Phytoplankton as indicators for eutrophication in Europe's regional seas

by

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Abstract

Eutrophication of marine and coastal waters is a growing concern throughout Europe's regional seas and an historical problem in the Black Sea and regions of the North-East Atlantic, particularly the North Sea. As the base of the marine pelagic food web, phytoplankton are sensitive indicators of environmental change and therefore may be used as indicators of eutrophication for the monitoring, management and mitigation of the effects of nutrient loading on coastal and marine ecosystems. However, due to the interactive effects of climate and eutrophication, it can be difficult to separate the climate-driven response of phytoplankton from changes induced by excess nutrients. Therefore, the aim of this work is to separate these two signals in order to explore eutrophication effects.

Without historical knowledge of 'pristine' or unimpacted ecosystem states it is difficult to identify and assess the severity and magnitude of change. Even where spatially and temporally comprehensive ecological datasets are available, equivalent nutrient time-series are rare and a method of linking phytoplankton dynamics to eutrophication is required. Because open sea ecosystems are less impacted by anthropogenic nutrients than those near shore, offshore regions may be used as reference areas in comparison with coastal systems to investigate the effects of nutrient loading. Changes observed solely in coastal systems are most likely a result of local processes (such as eutrophication) while those observed in both open sea and coastal areas are probably a response to large-scale drivers (such as climate). Therefore the comparison of coastal and open sea data may reveal different (or similar) patterns of change in phytoplankton indicators.

Throughout most of the North-East Atlantic climate appears to override the effects of nutrients on phytoplankton dynamics, although the two drivers have been found to have synergistic effects resulting in increasing chlorophyll levels in the coastal North Sea. Additionally, the 1980s North-East Atlantic regime shift is clearly visible in coastal and open sea chlorophyll concentrations and diatom and dinoflagellate abundances, demonstrating the sensitivity of phytoplankton as indicators at both the biomass and functional group scales. In the Black Sea, an observed decrease in chlorophyll appears to be at least partially a result of changes in climate and is not solely attributable to the 'recovery' of the Black Sea ecosystem. Black Sea chlorophyll has also undergone a possible recent (2002) regime shift, although its significance is difficult to determine due to the short time-series of chlorophyll data available.

The successful use of phytoplankton as indicators of eutrophication in these two disparate sea regions at two different ecological scales suggests that the method of comparing coastal and open sea phytoplankton data could be applied to other European seas as a means of distinguishing between the effects of climate and eutrophication.

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Terms and abbreviations

BS-SAP – Black Sea Strategic Action Plan

CAP – Common Agricultural Policy

Chl *a* – Chlorophyll *a*

CIL – Cold Intermediate Layer

CPR – Continuous Plankton Recorder

EEA – European Environment Agency

EcoQOs – Ecosystem Quality Objectives

Eutrophication – an increase in the rate of supply of organic matter to an ecosystem, most commonly caused by anthropogenic nutrient enrichment

GEF – Global Environmental Facility

HAB – Harmful algal bloom. Includes blooms of nuisance as well as toxic species.

ICPDR – International Commission for the Protection of the Danube River

NAO – North Atlantic Oscillation

NCNEA – Northern and Central North-East Atlantic

PCI – Phytoplankton Colour Index

Regime shift – Stepwise alteration in composition and productivity of an ecosystem at a regional scale that reflects major hydrographic change

Resilience – The ability of a system to recover from disturbance

SAHFOS – Sir Alister Hardy Foundation for Ocean Science

SeaWiFS – Sea-viewing Wide Field-of-view Sensor

SNEA – Southern North-East Atlantic

SST – Sea surface temperature

SLP – Sea level pressure

TDA – Transboundary diagnostic analysis

TN – Total nitrogen

TP – Total phosphorus

WFD – Water Framework Directive

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Author's declaration

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for any other University award without prior agreement of the Graduate Committee.

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Chapter 1

Introduction

This chapter provides the basis and justification for examining the use of phytoplankton as indicators for eutrophication in Europe's seas. The need for eutrophication indicators in Europe is assessed, followed by an analysis of the suitability of plankton as effective indicators. Finally, the aims and objectives of this thesis are presented, along with a rationale for its structure.

Chapter 1: Introduction

1.1 A eutrophic Europe and the need for indicators

Each of Europe's regional seas, the Baltic, Black, Mediterranean, and North-East Atlantic, is experiencing increasing anthropogenic pressures including shipping activity, coastal development, habitat loss, and depletion of fish stocks (Fig. 1.1). Additionally, eutrophication of marine and coastal waters is an escalating concern in regions of each of these seas. In order to manage and mitigate environmental damage caused to marine ecosystems by excess nutrients, appropriate eutrophication indicators are needed for European seas.

Eutrophication is defined as an increase in the rate of supply of organic matter to an ecosystem, most commonly caused by anthropogenic nutrient enrichment (Nixon, 1995). In its 2001 topic report on eutrophication in Europe's coastal waters, the European Environment Agency (EEA) expanded Nixon's definition to state that eutrophication is 'enhanced primary production due to excess supply of nutrients from human activities, independent of the natural productivity level for the area in question' (Aertebjerg *et al.*, 2001). In the EU, eutrophication is assessed and monitored through changes in indicators, quantified information used to explain spatial or temporal changes in environmental quality (Baan and van Buuren, 2003). Indicators may be used to provide an early warning signal of environmental change, to assess the current state of the environment, to monitor environmental trends, or to diagnose the cause of an environmental problem (Cairns *et al.*, 1993). However, data availability and cost and ease of monitoring do not often allow for the selection of eutrophication indicators that fulfil all four of these uses (Dale and Beyeler, 2001).

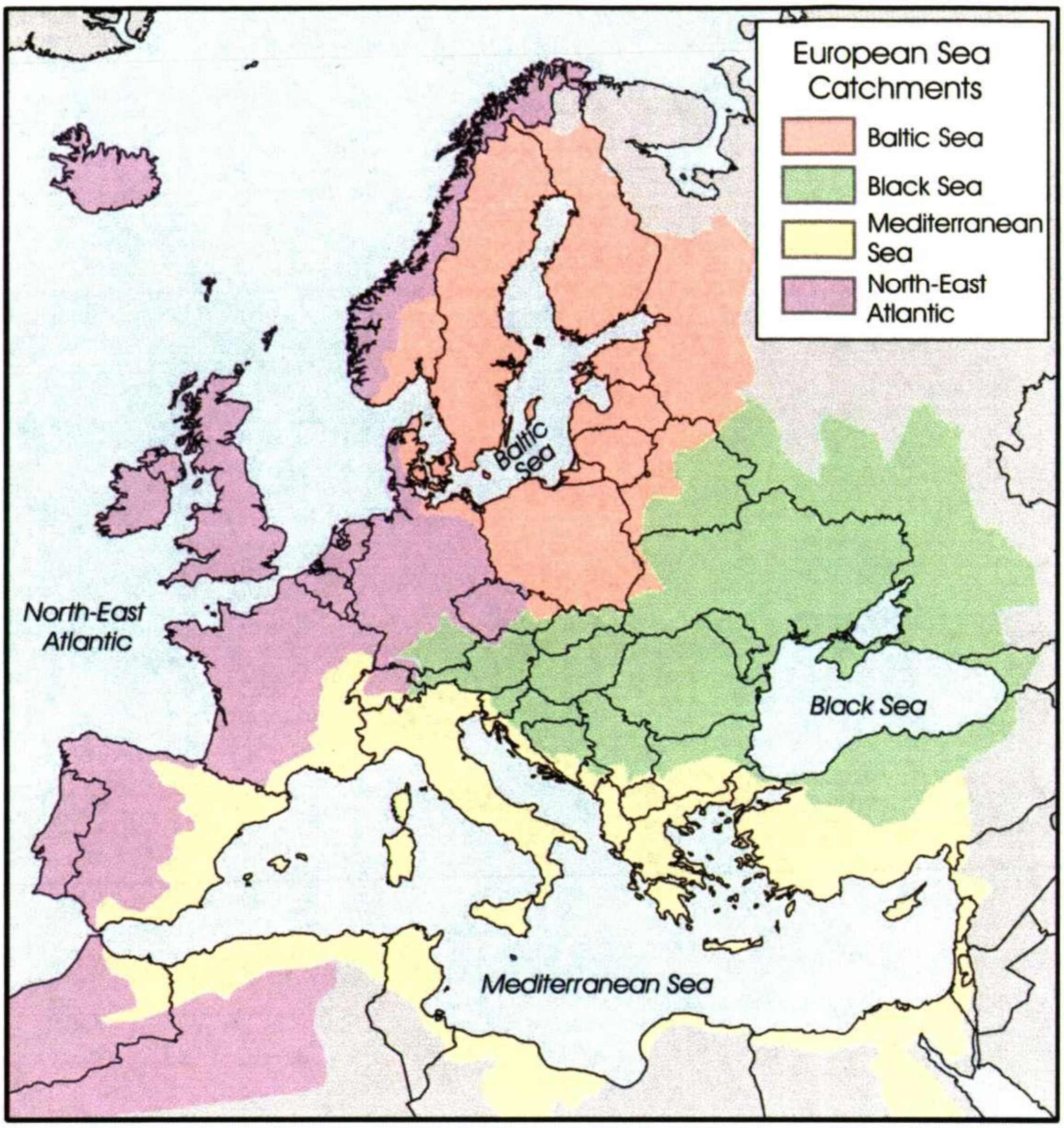


Figure 1.1. Map of catchment locations for each European regional sea.

Several frameworks or criteria for ecological indicator selection exist (Bossel, 2002; Cairns *et al.*, 1993; Dale and Beyeler, 2001; Rice, 2003; Xu *et al.*, 2001). However, indicator selection is a difficult task in Europe's seas due to the biological and physical variability of marine and coastal ecosystems and the temporal and spatial heterogeneity of available data. The framework proposed by Dale and Beyeler (2001) offers an example of suggested requirements needed for ecological indicator selection. Due to its simple but comprehensive approach, these criteria may be useful in choosing indicators for eutrophication in Europe's seas (Table 1.1).

Table 1.1. Criteria for ecological indicator selection (after Dale and Beyeler, 2001).

1. Are sensitive to ecosystem stress
2. Respond to stress in a predictable manner
3. Are easily measured
4. Provide an early warning of impending ecosystem change
5. Predict changes that can be averted by management action
6. Are integrative: can be used as part of a suite of indicators capable of assessing key components of ecosystem
7. Have a known response to natural disturbance and anthropogenic stresses over time
8. Have a low variability in response

Unfortunately, it is difficult to identify suitable eutrophication indicators, particularly those that will provide managers with an early warning sign. One problem lies in the separation of the impacts of eutrophication from the effects of natural stressors and other anthropogenic pressures such as overfishing, toxic contamination, exotic species introductions, aquaculture, hydrological alterations, habitat destruction, and climate change (Cloern, 2001; Niemi *et al.*, 2004). Multiple stressors may have synergistic, additive, or antagonistic effects on biological responses (Niemi *et al.*, 2004). Climate change in particular seems to have many of the same impacts as eutrophication including an increase in harmful algal blooms (HABs), altered food webs, and changes in species composition. In fact, climate change may even exacerbate eutrophication as increased precipitation and flooding flush nutrients into coastal waters (Hama and Handa, 1994). Also, global warming may enable changes in community composition (for example, higher temperatures lead to increased stratification and a consequent change from a diatom dominated phytoplankton community to one dominated by dinoflagellates) which in turn can alter food webs and affect ecosystem structure (Bopp *et al.*, 2005; Verity and Smetacek, 1996). With such heavy exploitation of Europe's coastal zone as well as simultaneous changes in climate, it is difficult to assess the effects of eutrophication without concurrently considering the impacts of the aforementioned stressors. Furthermore, disparate geographical regions are likely to require different indicators for eutrophication depending on the extent to which they have been affected by additional anthropogenic factors and variations in hydrological, meteorological, and biological conditions.

Since an increase in nutrients directly affects marine primary production, phytoplankton are an appropriate starting point when selecting eutrophication indicators for European seas (Tett *et al.*, 2007). The use of plankton as ecosystem indicators has been well-studied and the EEA uses summer chlorophyll *a* concentration (as a proxy for phytoplankton biomass), changes in plankton community structure and frequency of toxic HABs as eutrophication indicators (Aertebjerg *et al.*, 2001; Baan and van Buuren, 2003; Beaugrand, 2004b; Dickson *et al.*, 1992; Edwards *et al.*, 2001b; Paerl *et al.*, 2003a; Paerl *et al.*, 2003b; Tett *et al.*, 2007). In addition, plankton inhabit all European coastal and marine waters and so have broad applicability as indicators for eutrophication.

While no perfect eutrophication indicator exists, plankton do exhibit many of the characteristics of an 'ideal' indicator (Table 1.2). Phytoplankton are the primary producers of the pelagic system and are thus the first biological component to respond to increased nutrients. Additionally, phytoplankton play a crucial role in ecosystem structure and functioning and are an important food source for higher trophic levels (see also section 1.3). Also importantly, plankton are sensitive to their surroundings and respond quickly to physical and chemical changes in their environment. Recent research shows that plankton may be even more sensitive indicators of change than environmental variables themselves, since the non-linear responses of biological communities can amplify subtle environmental changes (Taylor *et al.*, 2002). Because plankton have short lifecycles, population size is less influenced by the persistence of individuals from previous years, thereby closely linking plankton dynamics to environmental perturbations (Hays *et al.*, 2005). Also, unlike fish and other marine species, most plankton are not commercially exploited and so long-term changes in the plankton can be attributed to changes in their environment (Hays *et al.*, 2005). Plankton can serve as indicators across multiple scales; phytoplankton biomass (or chlorophyll) is indicative of the amount of primary productivity in a system while changes in plankton community composition can foretell changes in ecosystem structure and functioning that may precede future changes in upper trophic levels. Plankton community composition is an integral part of the ecosystem and affects the structure and function of food webs, nutrient cycling, habitat condition, fishery resources, and overall ecosystem state (Paerl *et al.*, 2003b). Analysis of long-term trends in phytoplankton over a large spatial area including both coastal regions affected by anthropogenic eutrophication and open water regions with little or no human impact may therefore assist in separating the signals of eutrophication from those of natural ecosystem variability and global climate change (Edwards *et al.*, 2001b; Hickel *et al.*, 1993).

Table 1.2. An assessment of the suitability of plankton as indicators for eutrophication, based on criteria provided by Dale and Beyeler (2001).

Ideal Ecological Indicator Quality (from Dale and Beyeler, 2001)	Plankton Suitability as Eutrophication Indicators
Are sensitive to ecosystem stress	Respond quickly to physical and chemical changes in their environment, short life cycles, primary producers
Respond to stress in a predictable manner	The first sign of an increase in nutrients is increased phytoplankton biomass and primary productivity, although these changes may occur due to climatic variability as well
Are easily measured	Chlorophyll a measured by remote sensing
Provide an early warning of impending ecosystem change	As base of food web plankton experience change before higher trophic levels
Predict changes that can be averted by management action	Anthropogenic eutrophication can be avoided or mitigated through proper land and water management actions
Are integrative: can be used as part of a suite of indicators capable of assessing key components of ecosystem	Phytoplankton biomass is a good indicator of primary productivity in a system while changes in plankton community composition affect food web structure and function, nutrient cycling, habitat condition, fishery resources, and overall ecosystem condition
Have a known response to natural disturbance and anthropogenic stresses over time	As cultural eutrophication occurs, plankton biomass increases, changes in ecosystem structure and functioning result, hypoxic/anoxic events may increase due to decomposing plankton and benthic mortalities can occur, HABs may increase in frequency
Have a low variability in response	Unfortunately, plankton response is often non-linear. Additionally, it is difficult to separate planktonic responses to eutrophication from responses to other natural and anthropogenic pressures

Although plankton are sensitive and respond quickly to environmental change they do have several shortcomings as ecological indicators. Phytoplankton biomass can be measured via remote sensing, but identifying individuals to the taxonomic or species level takes experience and skill and so may not be a practical component of a monitoring programme. Additionally, while phytoplankton biomass tends to increase with an increase in nutrient concentrations, different species and functional groups may respond to increasing nutrient levels in varying ways. Also, the hydrographic heterogeneity of European marine and coastal ecosystems influences the effect that increased nutrients have on the plankton. Perhaps the most significant downfall of using phytoplankton as indicators for eutrophication is that many phytoplankton responses usually linked to increases in nutrients or changes in nutrient ratios may also occur due to natural variability caused by climate change or other anthropogenic pressures (Edwards *et al.*, 2005). Developing the ability to differentiate between these two signals is paramount to understanding change in our environment via plankton indicators.

1.2 Natural variability, climate change and the ecosystem filter

In order to examine a system's response to anthropogenic pressure, its natural characteristics and variability must also be understood (Cadee, 1992; Yunev *et al.*, 2005). For example, the level of natural productivity in a system is an important determinant of its response to anthropogenic nutrient enrichment, and is therefore essential in understanding the trophic structure of the system (Cloern, 2001; Jackson, 2001; Jackson *et al.*, 2001). This is not as straightforward as it seems, however, as mankind has impacted coastal ecosystems for so long that there are few, if any, 'pristine' ecosystems remaining from which to derive reference conditions (Jackson, 2001; Jackson *et al.*, 2001; Niemi *et al.*, 2004). Without baseline conditions with which to compare recent data and observations, ecosystem change remains difficult to assess and interpret and natural variability hard to determine (Niemi *et al.*, 2004).

A particularly challenging issue is the separation of eutrophication effects from those resulting due to climate variability. To determine if changes observed in a system are a result of anthropogenic pressures, long-term trends in climate must be examined in addition to alterations in nutrient inputs and other anthropogenic factors.

When discussing eutrophication, it is important to consider not only the anthropogenic pressures on a system but also its natural filter, or the physical and biological attributes that together determine the sensitivity of that particular ecosystem to nutrient enrichment (Cloern, 2001; Yunev *et al.*, 2005). An area's hydrological characteristics such as stratification, strength of tidal energy, length of residence time, salinity, and the amount of suspended sediment in the water column, and meteorological characteristics such as level of precipitation, light availability and wind patterns also influence its sensitivity to enhanced nutrient loading (Cacciamani *et al.*, 1992; Cloern, 2001; Vollenweider, 1992).

1.3 Top-down control

Although often overlooked or ignored, top-down controls can also influence an ecosystem's response to nutrient enrichment through the regulation of phytoplankton biomass (Daskalov, 2002; Perez-Ruzafa *et al.*, 2002). An adequate population of benthic suspension feeders may exert enough grazing pressure to balance an increase in primary production (Cloern, 2001; Gifford *et al.*, 2004; Peterson *et al.*, 2001). Overfishing also acts as a top-down control which can further compound the effects of eutrophication. A declining population of large commercially exploited fish can lead to an increase in planktivorous fish and consequently more grazing pressure may be exerted on herbivorous zooplankton thereby reducing its abundance and enabling the overgrowth of phytoplankton biomass (Daskalov, 2002; Jackson *et al.*, 2001; Verity and Smetacek, 1996; Verity *et al.*, 2002). The Black Sea experienced the collapse of its ecosystem due to a combination of eutrophication and overfishing (Daskalov, 2002). Unregulated fishing during the 1950s and 1960s resulted in the depletion of top predators such as dolphins, bonito, bluefish, and mackerel (Daskalov, 2002). At the same time, nutrient loading to the Black Sea from the Danube, Dniester, and Dnieper Rivers also intensified. From the 1950s to the early 1980s, annual discharge of phosphates increased from 14,000 to 55,000 tons, nitrates increased from 155,000 to 340,000 tons, and organic matter increased from 2,350,000 to 10,700,000 tons (Zaitsev and Mamaev, 1997). During the 1970s, small planktivorous fish stocks and jellyfish biomass increased leading to heavier grazing of fodder zooplankton, which experienced a 50% loss in biomass compared to the 1960s. During the 1980s, the subsequent lack of zooplankton consumption and continued nutrient enhancement caused phytoplankton biomass to double resulting in heavy sedimentation of phytoplankton detritus (Daskalov, 2002). Hypoxia followed, leading to mortalities of benthic invertebrates, such as the mussel *Mytilus galloprovincialis*, which in turn meant a further decrease of detritus consumption and therefore an enhancement of eutrophication in the Black Sea (Mee *et al.*, 2005; Zaitsev and Mamaev, 1997).

1.4 Nutrient alterations

As long as adequate light and micronutrients (including iron) are available, increased nitrogen and phosphorus results in an increase in phytoplankton biomass and primary production as phytoplankton cease to be nutrient limited (Cloern, 2001; Colijn, 1992;

Perez-Ruzafa *et al.*, 2002). While natural supplies of these nutrients are augmented anthropogenically through fertilizer runoff, aquaculture activities, wastewater disposal and atmospheric deposition, the amount of dissolved Si only increases through the natural processes of weathering and erosion (Officer and Ryther, 1980b; Riegman *et al.*, 1993). The artificial supplementation of N and P without that of Si causes a deviation from the natural nutrient composition of a system and not only increases the levels of N and P but also changes the stoichiometric ratios of Si:N:P from the ideal Redfield ratio of Si:N:P=16:16:1 (Justic *et al.*, 1995; Redfield *et al.*, 1963). Thus, because diatoms are Si-limited, eutrophic conditions can favour dinoflagellates and altered nutrient ratios may cause the phytoplankton community structure to shift from dominance by diatoms toward dominance by nitrogen- and phosphorus-limited dinoflagellates (Officer and Ryther, 1980b; OSPAR, 2003; Philippart *et al.*, 2000; Smayda, 1990). In the Dutch Wadden Sea, for example, as the summer average N:P ratio decreased to a ratio below that of Redfield, blooms of the foam producing flagellate *Phaeocystis* increased in both duration of bloom period and maximum biomass attained (Riegman *et al.*, 1993). Furthermore, two drastic changes in phytoplankton community composition in that area coincided with a switch from a phosphorus-controlled environment to an even more eutrophic but nitrogen-controlled environment and then back again (Philippart *et al.*, 2000).

1.5 Plankton community structure

Changes in phytoplankton community structure resonate upwards through the food web. Eutrophic waters, which are often Si-limited, can result in food webs structured around flagellates and jellyfish instead of the more trophically sound diatom-based fish food webs (Ryther and Officer, 1981; Sommer *et al.*, 2002). These flagellate-based food webs largely consist of heterotrophs (such as *Noctiluca*) and gelatinous zooplankton which, due to their low nutritional value, are less useful to upper trophic levels and so are essentially 'dead ends' (Verity and Smetacek, 1996). Furthermore, jellyfish both prey on fish larvae and also compete for food against planktivorous fish (Sommer *et al.*, 2002). In fact, a shift toward gelatinous species may be one way a marine ecosystem responds to stress, and could even be an alternative stable, and consequentially difficult to change, state of the system (Mee, 1992; Sommer *et al.*, 2002). This was a severe problem in the Black Sea when, in 1989 the gelatinous American ctenophore invader, *Mnemiopsis leidyi*, quickly became a prominent part of the zooplankton community. With the rise in *Mnemiopsis* biomass, the amount of prey zooplankton decreased drastically as did commercial fishery catches (Mee, 1992).

If zooplankton grazing pressure cannot control the enhanced phytoplankton growth resulting from eutrophication, water transparency may decrease as the sedimentation of organic matter increases. Initially this increase in food supply may lead to a rise in zoobenthic biomass (Grall and Chauvaud, 2002). However, if nutrient enrichment continues, this increased biomass cannot be sustained and the high rate of organic sedimentation can eventually lead to hypoxia and anoxia as bacteria consume oxygen while feeding on the organic matter. Most benthic organisms cannot tolerate hypoxic conditions and if they are unable to flee affected areas mass mortalities can occur. In addition to hypoxia and anoxia, decreased transparency results in diminished light penetration leading to a reduction in the growth depth and spatial coverage of seagrasses and perennial macroalgae (Duarte, 1993; Isaksson and Pihl, 1992).

1.6 Long-term data availability in European seas

The establishment of effective eutrophication indicators for Europe's seas is even more challenging because few continuous, long-term biological datasets are available (Reid, 1997). The lack of long time-series makes it difficult to measure the effects of eutrophication and to separate the eutrophication signal from ecological changes occurring due to climatic variability and/or other anthropogenic pressures. This paucity of data is a particular problem in the Black Sea (Langmead *et al.*, 2007).

Fortunately, one long-term dataset monitoring biomass and composition of marine plankton does exist for North Sea and North Atlantic waters. The Continuous Plankton Recorder (CPR) is operated by the Sir Alister Hardy Foundation for Ocean Science (SAHFOS) based in Plymouth, England. Ships of opportunity have towed the CPR across the North Sea and Atlantic Ocean since 1931, recording taxonomic and abundance data on approximately 170 phytoplankton taxa as well as the semi-quantitative measure of chlorophyll, the Phytoplankton Colour Index (PCI) (Batten *et al.*, 2003a). Recently, a relationship has been established quantitatively linking the PCI to chlorophyll concentration and thereby increasing its usefulness as a monitoring tool (Raitsos *et al.*, 2005).

Unfortunately the Black, Baltic and Mediterranean Seas do not have long-term plankton monitoring programmes with the wide spatial and temporal coverage of the CPR. However, several research institutes and projects have monitored plankton biomass and (less often) community composition in those regions. These efforts have focused mainly on coastal areas and, particularly in the Black Sea, sampling has occurred primarily during summer months.

Aside from *in situ* data, remotely-sensed ocean colour satellite data for all European seas are available for the periods 1978 – 1986 (Coastal Zone Colour Scanner) and 1997 – present (SeaWiFS). Ocean colour estimates chlorophyll *a* in surface waters, and while data are not available for a continuous period, the information provided is invaluable since satellite data are more regularly sampled in time and space than are most *in situ* data (Chu *et al.*, 2005). Although satellite data must be processed in order to eliminate atmospheric interference and difficulties in distinguishing between phytoplankton biomass and suspended solids in coastal waters exist, the data are free to use and downloadable from the internet. Unfortunately, due to differences in technology, data from the two monitoring programmes are not yet directly comparable, but changes within datasets can be observed.

Unlike biological datasets, data for many climatic variables are abundant and freely available. Long time-series of climate parameters such as sea surface temperature (SST), precipitation, photosynthetically active radiation (PAR), sea level pressure (SLP) and wind stress can be obtained from satellite sources. Nutrient concentration data for coastal and marine regions are more difficult to find. A quality-checked aggregation of nutrient data is available from the European Environmental Agency for areas of the Baltic, North and Mediterranean Seas, but Black Sea nutrient data must be obtained from the literature, research cruises or regional research institutes and agencies. In all seas nutrient data for coastal waters is more plentiful than that for open waters, but by its very nature is also inherently more variable.

1.7 Aims and hypotheses

The expansion of the EU to include 10 new countries in 2004 increased its population by nearly 75 million inhabitants and increased its size by 726 million square km (European Union, 2004). The accession countries, and indeed the original EU countries, have differing levels of waste and sewage treatment, types and intensities of land use, degrees of environmental degradation, and governmental policies regulating environmental issues. These issues differ within and between European sea catchments, but eutrophication is a concern in areas of each regional sea (Fig. 1.1). In order for the EU to monitor and remediate eutrophication-affected coastal ecosystems and prevent its coastal and marine areas from becoming further impacted by nutrient enrichment, appropriate eutrophication indicators are needed. Although each of Europe's seas suffers from some degree of eutrophication, two specific seas, the Black Sea and the North-East Atlantic/North Sea, are used as case studies in this thesis. However, methods applied to and lessons learned from these two marine ecosystems may be applicable to other sea areas in the future.

The Black Sea and some coastal regions of the North-East Atlantic (particularly the North Sea) have been impacted by eutrophication in recent decades (Hickel *et al.*, 1993; Mee, 1992). The North Sea has been monitored extensively during the last 50 years, while the Black Sea is somewhat less understood. Both seas have limited data available on unimpacted states making it difficult to analyse temporal changes and develop accurate indicators. Therefore, an alternate method of eutrophication assessment and indicator development must be established. Because areas further from the coast are less likely to be impacted by land-based nutrient inputs, data from open sea areas may be used as a working baseline with which to compare coastal data. Changes occurring due to climate variability are likely to be observed in both open sea and coastal waters while changes resulting from increased nutrients mostly affect coastal areas. Thus, the comparison of coastal indicators to those from open sea areas may provide some insight into changes resulting from eutrophication in the North-East Atlantic/North Sea and Black Sea.

The aim of this project is to assess the suitability of phytoplankton as indicators for eutrophication in the Black Sea and North-East Atlantic/North Sea system. Phytoplankton were selected as indicators because they exhibit the first biological response to increased nitrogen and phosphorus, they are the foundation of the marine food web and they are sensitive to environmental change. Two scales of phytoplankton assessment will be considered and compared. These include phytoplankton biomass (measured by remotely-sensed chlorophyll or the Continuous Plankton Recorder-derived Phytoplankton Colour Index) as an indicator of ecosystem productivity, and major phytoplankton group abundance (measured by diatom and dinoflagellate abundance) as an indicator of phytoplankton community composition. Changes in coastal and open sea phytoplankton will be analyzed in relation to nutrients (concentrations and ratios) and climatic factors (such as sea surface temperature, sea level pressure, wind stress, North Atlantic Oscillation, etc). Differences between open sea and coastal phytoplankton productivity and composition and their relationships with climate and nutrients will demonstrate the applicability of phytoplankton as indicators for eutrophication and may also provide insight into the status of coastal eutrophication in the North-East Atlantic/North Sea and Black Sea.

The main hypotheses tested in this research are:

- Phytoplankton community dynamics differ between coastal and open sea regions of the North-East Atlantic/North Sea and Black Sea
- Open sea ecological data may be used as a baseline where data from 'pristine' conditions are not available
- A comparison of open sea and coastal phytoplankton communities enables climate and eutrophication impacts to be distinguished
- Inter-annual changes observed in phytoplankton communities are likely to be non-linear and may include regime shifts and thresholds

1.8 Rationale

The rationale for the structure of this thesis is as follows:

Chapter 2 presents an overview of the general methodology used in this thesis.

Chapter 3 provides a brief background of eutrophication in the Black Sea, highlighting the importance of the Danube River to the Black Sea nutrient regime, followed by an analysis of the effectiveness of political tools established at various scales of governance in order to ameliorate and/or mitigate eutrophication in the sea. Data availability and regional research priorities and recommendations are discussed.

Chapter 4 investigates variability in inter- and intra-annual chlorophyll dynamics in the Black Sea through the use of SeaWiFS satellite data. Spatial patterns in phytoplankton biomass are explained and the role of climate in the recovery of the Black Sea is explored.

Chapter 5 explores the spatial patterns of diatom and dinoflagellate seasonal cycles in the North-East Atlantic. This chapter provides previously unreported 'baseline' data of the intra-annual dynamics of these two important phytoplankton functional groups.

Chapter 6 analyzes the differences in spatial and temporal dynamics of coastal and offshore diatom and dinoflagellate communities in the North-East Atlantic. The likelihood of influence of climate and eutrophication as possible drivers of phytoplankton dynamics in near-shore and open sea communities is assessed.

Chapter 7 investigates nutrient trends and climatic variability as possible drivers of the North Sea regime shift through a newly-created long-term chlorophyll dataset. The relationships between climate, nutrient concentrations and phytoplankton biomass are examined in the coastal and open North Sea. The synergistic effects of climate and eutrophication on phytoplankton production in the North Sea are discussed.

Chapter 8 discusses the main findings of this thesis and compares themes observed throughout the research presented here including the critical need for baseline data, the existence of non-linearities and regime shifts, and the use of open sea and coastal phytoplankton indicators for eutrophication assessment. Similarities between the North-East Atlantic/North Sea and Black Sea phytoplankton communities and their relationships with nutrient enrichment are also examined.

The appendix contains a series of monthly and annual mean chlorophyll concentration and monthly chlorophyll anomaly maps of the Black Sea for the 1997 - 2005 time period. These maps were created using SeaWiFS data and supplement Chapter 4.

Chapter 2

General methodology

This chapter outlines the general methodology underlying the analysis conducted in this research. Geostatistical techniques, data extraction and processing, mapping theory and methodology and potential spatial bias of datasets used are discussed here.

Chapter 2: General methodology

2.1 Introduction

Eutrophication is primarily a coastal problem and systems far removed from land are only minimally impacted by anthropogenic nutrient enrichment. The comparison of open sea and coastal phytoplankton populations and levels of productions was chosen as the cross-cutting method of evaluating and assessing eutrophication in the study areas under consideration. Spatial and temporal scales of datasets used in this work were carefully considered when selecting appropriate datasets and these scales influenced the type of analysis conducted in each regional sea.

2.2 Study areas

The North-East Atlantic/North Sea and the Black Sea were selected as case study areas for this research. The North-East Atlantic/North Sea has a long history of anthropogenic impacts and is data-rich. The Black Sea has experienced eutrophication-related problems since the 1970s, is new to the European Union and is, compared to the North-East Atlantic/North Sea, relatively data-poor. Thus, North-East Atlantic/North Sea and Black Sea provided an opportunity to test the 'open sea vs. coastal' methodology of eutrophication assessment in contrasting cast study areas.

2.3 Open sea vs. Coastal

A contrast between open sea and coastal waters (the region into which the majority of anthropogenic nutrients enter a sea) was used as a means of distinguishing between the effects of eutrophication and other ecosystem. This method was applied at various spatial scales depending on the sea under consideration. For example, the northwestern shelf of the Black Sea was considered as 'coastal' because, even though in some places the shelf extends up to 200 km from shore, it is shallow in depth, hydrographically distinct from the deep offshore or 'open sea' region of the Black Sea, and receives the majority of land-based anthropogenic nutrient input (Fig. 3.1).

The approach taken in the North-East Atlantic/North Sea was slightly different from that taken in the Black Sea as the North-East Atlantic/North Sea system has no consistent definition of coastal and open sea regions available in the literature. In the North-East Atlantic and North Sea, 'coastal' and 'open sea' regions were defined as a set distance from shore (Fig. 6.1; coastal: < 50 nm from shore, open sea: > 125 nm from shore). Between the open sea and coastal regions, a 'buffer' zone remained whose data were omitted from the research. Creation of a buffer emphasized differences between open sea and coastal dynamics.

2.4 Geographic Information Systems (GIS)

Geographic Information Systems (GIS) provides a method of displaying, selecting, aggregating, and exploring patterns, trends and relationships in data that have a spatial component. GIS stores data in 'point', 'polygon' or 'line' 'layers'. These layers can be overlain and their underlying information (contained in 'attribute tables') analyzed. The creation of maps is just one very small (but useful) part of GIS.

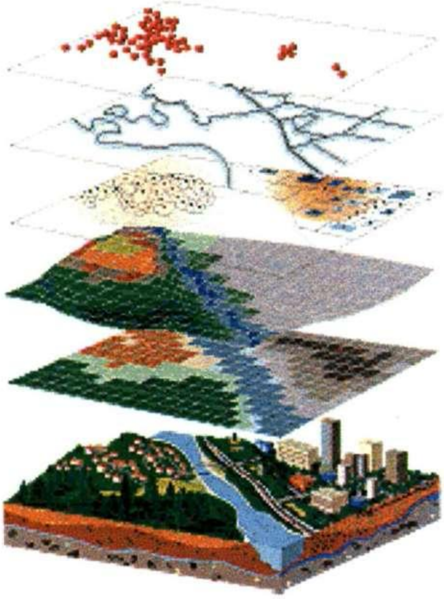


Figure 2.1. An example of point, line and polygon layers overlain in a GIS (figure from www.esri.com).

The GIS package ArcGIS 9.2, developed by ESRI, was used for all GIS analysis. Though ArcGIS is one of the most popular and comprehensive GIS packages available, it lacks some tools which are helpful when analyzing ecological and climatic datasets (such as the ability to 'grid' point data into polygon data – see section 2.5). Therefore, where needed, Hawth's Analysis Tools for ArcGIS (downloadable from: www.spatialecology.com), a free add-in for the ArcGIS software package, was used to conduct additional specialized spatial analysis.

In this research, polygons were created in order to distinguish between the open sea and coastal regions of the Black Sea and North-East Atlantic/North Sea systems. This was done slightly differently in each case study area. In the North-East Atlantic/North Sea, a buffer was created around the coastline denoting the 50 nm and 125 nm distance from the coast. Further subregions (see Fig. 6.1) were manually created as needed. The Black Sea was divided into shelf and open sea regions according to bathymetry. A bathymetric map was scanned and added to ArcGIS where it was then digitized and shelf and open sea polygons created. The shelf break is located approximately at the 200 m isobath and this feature was used to delineate the outer boundary of the northwestern shelf. The open Black Sea was further divided into western and eastern regions according to the approximate locations of the gyres.

In all aspects of this work, regional polygons were used to select 'open sea' and 'coastal' data points from all spatially referenced datasets used. This process will be explained in dataset-specific detail in section 2.5.

2.5 Data extraction and processing

Spatial coverage of data was a key priority and drove dataset selection. Where available, datasets with samples across a wide spatial area were chosen. The only parameters for which spatial comprehensiveness was not needed were the winter North Atlantic Oscillation index, Atlantic inflow to the North Sea, Rhine and Elbe river discharge, and nutrient concentrations from the mouths of the Elbe and Rhine rivers. Datasets describing the remaining hydroclimatic variables, all indicators of phytoplankton biomass and phytoplankton community composition and nutrient data were chosen for their long time-series and wide spatial spread of samples.

2.5.1 Remote sensing data: NCEP/NCAR climate data and SeaWiFS Chl a

All remote sensing datasets used in this thesis (NCEP/NCAR: sea level pressure (SLP), wind stress, precipitation, sea surface temperature (SST); SeaWiFS: Chl a) required the same method of data processing. The remotely sensed datasets were obtained in a gridded format as separate text files for each parameter for each month (for example, 100 months of SeaWiFS data resulted in 100 separate text files). A short program was used to 'stack' the dataset into one file (so 100 months of SeaWiFS data would then be in one text file with each month as a column). The 'stacked' text file was then imported into MS Access to ensure compatibility with ArcGIS. From MS Access, the data were added to ArcGIS as point data. Any datapoint with a 'no data' value was deleted (for example, SST measurements over land). The polygon grid onto which these points fall had to then be created so that each sample was located in the centre of the corresponding grid cell. The point layer was spatially joined to the polygon grid layer so that the attributes of each point (sample date, parameter measurement, etc) were copied to the attribute table of the corresponding grid cell. This resulted in a grid with each cell containing many parameter measurements (one for each month of data).

Hawth's Analysis Tools was used to calculate the Area Weighted Mean (AWM) of each parameter for each month for each region (open sea or coastal, or subregion). The AWM ensures that portions of a grid cell lying outside of the region under consideration are not included in the regional mean and that the calculated mean is correctly weighted by the area of each grid cell that lies in the region. In this way, for example, the portion of a grid cell that overlaps onto land is excluded from the regional mean. This process resulted in a time-series of each parameter for each month for each geographic region (for example: a mean measurement of SeaWiFS Chl *a* for the coastal Black Sea for each month from September 1997 to December 2005).

2.5.2 Continuous Plankton Recorder (CPR) data

The method of processing of Continuous Plankton Recorder (CPR) data was similar in some ways to that of remote sensing data. The primary difference is that CPR data are not gridded; this is because the CPR, unlike remote sensing satellites, does not sample regularly in space or time. Thus, in order to eventually arrive at an AWM for open sea and coastal regions, it was necessary to first grid the CPR data. A 1 by 1 degree grid polygon layer covering the North-East Atlantic was created using Hawth's Analysis Tools. CPR sample locations were then added to the GIS and one time period was selected (for example, all samples collected during the month of January 1958-1962). The mean of all samples was spatially joined to the grid cell in which those samples were located, giving each grid cell a mean number of diatoms, dinoflagellates and/or Phytoplankton Colour Index. This resulted in a grid with each cell containing many parameter measurements (one for each month-period of data). A point layer with a datapoint located in the centre of each grid cell was constructed and the parameter measurements from the grid layer were joined to the point layer. This point layer was not needed to determine the AWM, but was later used in creating seasonal (monthly) and decadal maps of diatom and dinoflagellate abundances (see section 2.6.1). Once all data from all time periods were gridded, the same method of calculating the AWM was used as for the remote sensing data (section 2.5.1).

2.5.3 Secchi disk and EEA nutrient data

Secchi disk and EEA nutrient data for the North Sea were treated differently than the climate and phytoplankton datasets. The EEA's Waterbase is designed using separate tables for each parameter measured as well as for sample locations so the first step in processing the EEA nutrient data was to join the sample parameters table to the station locations table in MS Access. This resulted in a spatially referenced dataset of nutrient concentrations for the North Sea. This was not necessary with Thorkild Aarup's Secchi Disk Data Collection (Aarup, 2002) which was already spatially referenced.

Because it was not necessary to create maps using the EEA nutrient dataset or the Secchi Disk Data Collection, these datasets were not gridded. Instead, both datasets were plotted as point data in a GIS and then selected using the open sea and coastal zone polygons resulting in an 'open sea' dataset and a 'coastal' dataset. The data were exported to a spreadsheet for use in Excel. Once in spreadsheet format, the data could then be aggregated into annual or seasonal means.

2.6 Data analysis

2.6.1 Maps

Due to the spatial nature of this work, mapping was a key tool used in data analysis. While it is possible to map data using ArcGIS, the majority of the maps created for this thesis (all CPR and SeaWiFS chlorophyll maps) were created using Golden Software's Surfer. Surfer is a mapping package rather than a GIS package and therefore requires fewer computer resources than ArcGIS, enabling quick and efficient map creation (over 150 maps were created). With Surfer it is also possible to create a file with all mapping parameters for a particular project (such as longitude and latitude of map extent, map units, final output size of a map, interpolation method and parameters, etc) which eliminates the need to input parameters manually each time a map is created and easily ensures a uniform look to a series of maps.

All maps were created using the Inverse Distance Weighting (IDW) method of interpolation. IDW creates a continuous distribution of a parameter (chlorophyll concentration, diatom or dinoflagellate abundance, etc) across the study area from a set of discrete points (here the gridded data, see section 2.5). IDW assumes that interpolated points are more influenced by nearby data than data that are further away (Caruso and Quarta, 1998), and like all geostatistical methods, assumes that spatial structures are stable in space and time for the duration of the sampling period (Simard *et al.*, 1992). However, this cannot be assumed of CPR data due to its 46 year sampling period. This problem has been resolved by portioning CPR data into appropriate temporal periods (such as decadal or half decadal) and treating each temporally-partitioned dataset individually (Edwards, 2000). Unlike CPR data, SeaWiFS data are sampled regularly in both space and time, making interpolation possible for short time periods (monthly SeaWiFS data were used here). More details referring to creation of specific maps can be found in the 'Methods' sections of Chapters 4, 5, and 6.

2.6.2 Anomalies

In order to remove the long-term mean, to emphasize subtle temporal patterns in Chl *a* or phytoplankton abundance, and to allow for the relative comparison of these parameters, standardized anomalies (z) were calculated as:

$$z = \frac{(x - \bar{x})}{\sigma}$$

where $\bar{x}$ = the long-term mean and σ = the standard deviation. This was repeated for each SeaWiFS measurement or CPR sample at three possible temporal scales: 1) the intra-annual (monthly composite) anomaly was calculated based on the long-term mean of each calendar month relative to the composite annual mean; 2) the inter-annual anomaly was calculated from the yearly mean of each complete calendar year (or in the case of the CPR decadal period) available relative to the composite annual (or in the case of the CPR time-series) mean; and 3) individual SeaWiFS monthly Chl *a* anomalies were calculated for each month (September 1997 – December 2005, n = 100) relative to the long-term monthly means. Anomalies were then mapped as needed, following the procedure outlined in section 2.6.1.

2.7 Potential spatial bias of datasets

As mentioned previously, spatial comprehensiveness was a key factor in dataset selection. An overview of the sources of potential spatial bias of the spatially referenced datasets follows.

2.7.1 SeaWiFS Chl a data

Although the spatial coverage of a remotely sensed dataset is remarkable, there are several limitations. The SeaWiFS satellite samples Chl *a* on a daily basis, however, many areas of the Earth's surface are obscured due to cloud cover at this temporal scale. Therefore 8-day (Chapter 7) and monthly (Chapter 4) data products readily available from NASA and that consist of aggregated averages of SeaWiFS Chl *a* estimates were used.

Another limitation of SeaWiFS data is the difficulty in distinguishing Chl *a* from particulate matter and/or yellow substances (dissolved organic matter) in Case II (optically complex coastal) waters (IOCCG, 2000). This can lead to an overestimation of Chl *a* concentration in coastal waters. One method of using SeaWiFS Chl *a* data in Case II waters is to pair the satellite data with *in situ* data (see Chapter 7). Additionally, anomalies may be used to measure relative change in SeaWiFS Chl *a* is Case II waters over a particular time period (see Chapter 4).

2.7.2 CPR data

The Continuous Plankton Recorder has sampled much of the North-East Atlantic since 1948. However, the CPR is normally towed further than 1 km offshore and therefore may not record changes in some near-shore plankton communities (Edwards *et al.*, 2006). In this research, CPR data were selected for regions < 50 nm from shore in order to overcome this spatial limitation. Additionally, CPR data have not always been sampled regularly in some regions of the North-East Atlantic (*i.e.* there are 'data voids' in some months in some regions). Where > 8 months of data per calendar year exist in a region, temporal interpolation was used to estimate abundance for missing months (see Chapter 6). If < 8 months of CPR data were available for a particular year in a region, that year was excluded from analysis of the regional time-series.

2.7.3 EEA nutrient data

In order to determine if the EEA nutrient data (Chapter 7) were spatially representative throughout the study period, all TN and TP samples were plotted in a GIS. Between 1980 and 2002, n = 1541 TN samples and n = 2060 TP samples were collected in the coastal North Sea while n = 46 TN samples and n = 179 TP samples were collected in the open North Sea (Fig. 2.2). Because of the low number of open North Sea samples, all conclusions drawn from those data were considered tentative.

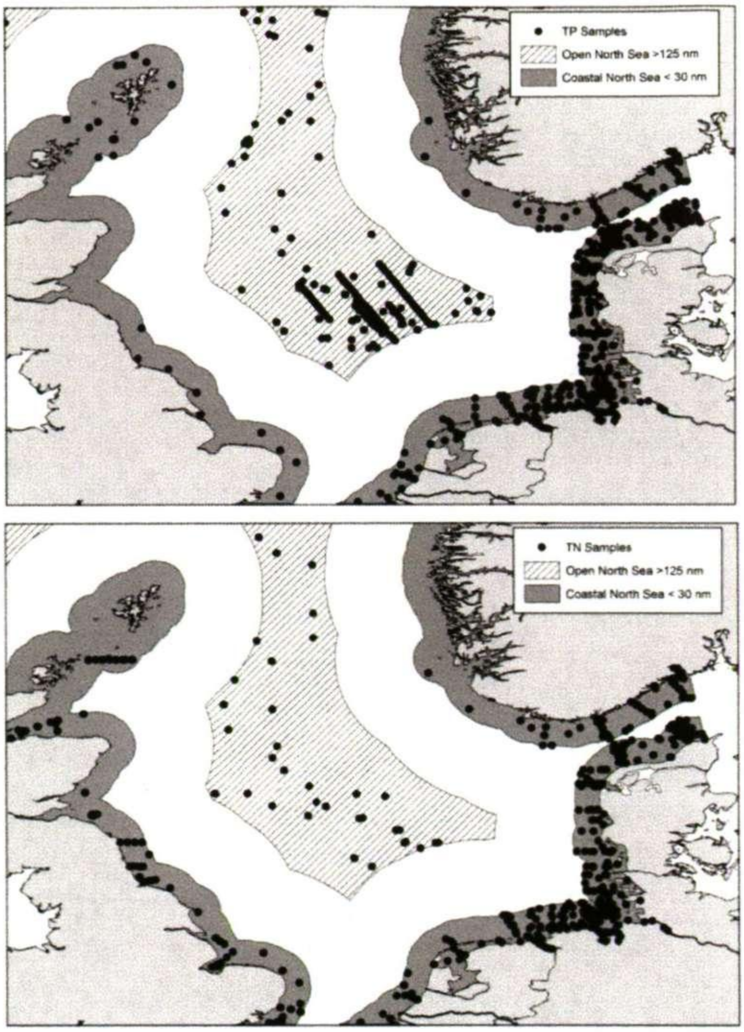


Figure 2.2. Locations of TP (top) and TN (bottom) EEA nutrient samples in the North Sea between 1980 and 2002.

The coastal North Sea was divided into finer geographic regions in order to more thoroughly explore spatial variability of nutrient data. The southern coastal North Sea and Danish coast were the most heavily sampled areas for both TN and TP (Fig. 2.2) while the UK and Norway coastal regions contained far fewer samples. The southern coastal North Sea was the region with the highest concentrations of both nutrients, with concentrations up to 10 times richer than those in other coastal regions (see Fig. 7.5). Thus, the general decreasing nutrient trends observed for the coastal North Sea as a whole are heavily weighted by nutrient dynamics occurring in the southern coastal North Sea (see Chapter 7 for further explanation).

2.7.4 Secchi disk data

As with the EEA nutrient data, it was necessary to confirm that data from the North Sea region of the Secchi Disk Data Collection (Aarup, 2002) were spatially representative throughout the period of study (Chapter 7). All Secchi disk samples were plotted as point data on a GIS and then selected by open sea or coastal North Sea region. Between 1950 and 2000, a total of $n = 5057$ samples were available for the coastal North Sea while only $n = 68$ were available for the open North Sea region. The limited number of open North Sea samples prevented any type of time-series analysis in that region, so those data were excluded from the database. Like the EEA nutrient data, the coastal North Sea region was further divided into subregions to explore the spatial spread of data. The southern coastal North Sea was the most heavily sampled subregion while the UK was the least sampled. The 1990s were the most heavily sampled decade in all three regions, followed by the 1980s and 1970s in the southern North Sea and Norway and the 1970s and then the 1980s in the UK. Thus, in the southern North Sea and Norway, the accuracy of trend in Secchi depth increases with time. The southern North Sea is the region most often considered to be impacted by eutrophication and was the region with the greatest number of Secchi depth samples. The data were divided into yearly summer (July - August) and winter (December - February) seasonal averages for the entire coastal North Sea region, leaving several years, particularly in the early part of the time-series, with no data (see Fig. 7.3).

Table 2.1. Secchi disk samples by decade and coastal North Sea subregion.

	Number of Secchi disk samples		
Decade	Southern North Sea	Norway	UK
1950s	0	3	3
1960s	40	2	5
1970s	692	41	25
1980s	1335	348	5
1990s	1542	947	69
Total	3609	1342	107

Chapter 3

A brief review of eutrophication in the Black Sea

Eutrophication is a historical concern in the Black Sea, and with the recent accession of Bulgaria and Romania to the EU, nutrient enrichment in the Black Sea is now a European problem. The causes and effects of eutrophication are surrounded by uncertainty, a situation exacerbated by rapidly changing political and economic regimes and a catchment shared between 6 countries, the majority of which are non-EU. Although regional, international and European initiatives have attempted to regulate nutrient input to the Black Sea, a lack of stakeholder involvement, poor enforcement of environmental regulations, and insufficient funding constrains the effectiveness of these measures. Nevertheless, the Black Sea has shown some recent signs of ecosystem improvement during the last 15 years. However, with the Black Sea's economies undergoing rapid transitions, the path to ecosystem recovery is uncertain. Increased environmental research and monitoring, extensive educational programmes, and adequate funding for the Black Sea Commission are the most likely routes through which further ecosystem recovery may be possible.

Aspects of this chapter are included in the following:

McQuatters-Gollop A. and L.D. Mee. (2007). Eutrophication in the Black Sea, p 20. World Resources Institute.

Chapter 3: A brief review of eutrophication in the Black Sea

3.1 Background

Eutrophication is a matter of growing concern in Europe's seas (Aertebjerg *et al.*, 2001), and is a long established problem in the Black Sea (Mee, 1992). The Black Sea is largely isolated from the World Ocean with only limited exchange with the Mediterranean through the Bosporus Strait. Due to its shallow depth and because it receives considerable riverine input, the northwestern shelf is particularly susceptible to the effects of nutrient loading (Fig. 3.1) and has suffered extensive ecosystem changes due to eutrophication. Except in the northwest, the continental shelf does not extend more than a few kilometres from the coast, and eutrophication events are few in those areas. The deep, or open, Black Sea comprises approximately 75% of its surface area, is permanently anoxic below 200m depth, and acts as a sink for nutrients. The impact of eutrophication on the open Black Sea has been considerably less severe than that occurring in the shelf area.

In total, the Black Sea drains a catchment consisting of 23 countries, covering a land area of 2,400,000 km^2, and receiving waste water from more than 190 million people (daNUbs, 2005). The Danube, Dniester, and Dnieper Rivers transport water from much of Europe and Russia to the northwestern shelf area, and together are responsible for 85% of the freshwater entering the Black Sea (Sorokin, 2002). The Danube is the most important river in the basin with an 800,000 km^2 catchment containing 43% of the Black Sea basin's population (daNUbs, 2005). The Danube contributes 55% of total river discharge into the Black Sea and, because of its extensive catchment area, a considerable proportion of its nutrient load.

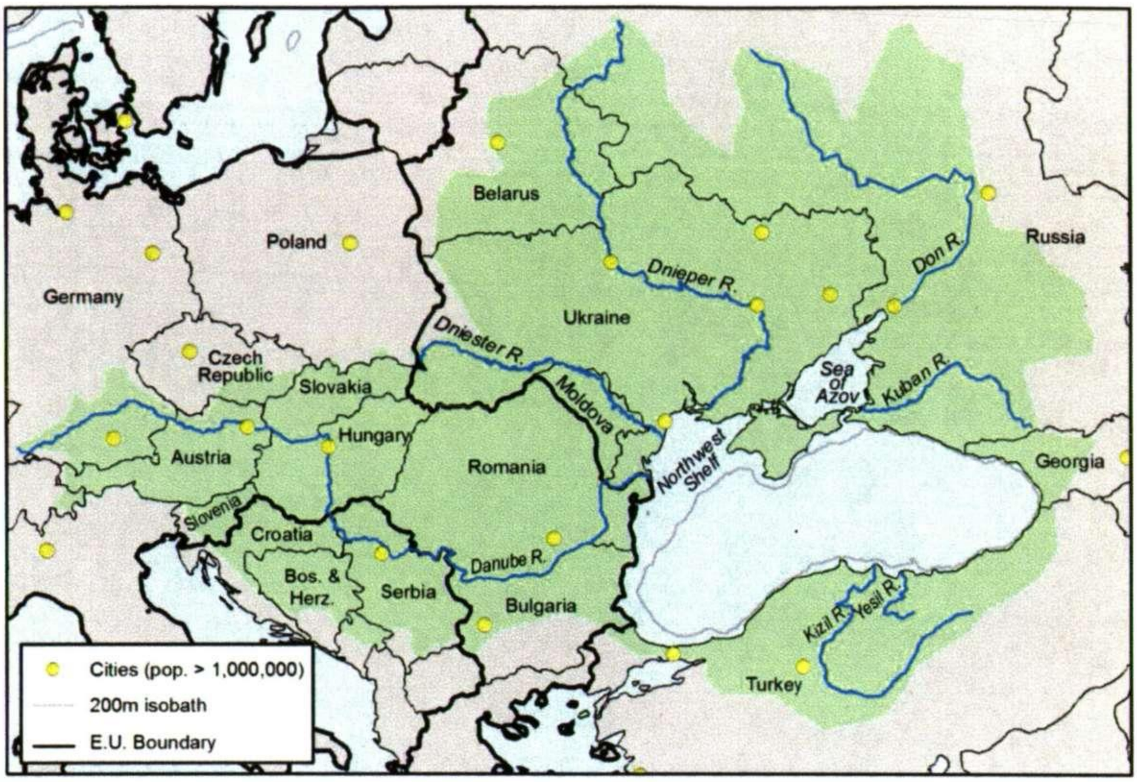


Figure 3.1. Catchment map of the Black Sea.

During the 1960s the Soviet agricultural revolution resulted in intensive fertilizer use and livestock production in the Black Sea catchment (Mee *et al.*, 2005). Winter phosphorus concentrations in shelf waters increased rapidly from the late 1960s (Cociasu *et al.*, 1998) while data suggest a six-fold increase in nitrogen between in 1960 and 1980, although limited pre-1980 nitrogen data available exist, and this conclusion must therefore be considered tentative (Cociasu *et al.*, 1996; Cociasu and Popa, 2004; Cociasu *et al.*, 1998) (Fig. 3.2). From the 1950s to the early 1980s, the annual discharge of phosphates increased from 14,000 to 55,000 tons, nitrates increased from 155,000 to 340,000 tons, and organic matter increased from 2,350,000 to 10,700,000 tons (Zaitsev and Mamaev, 1997). As nitrogen and phosphorus increased in shelf waters, silicon concentrations decreased due to the construction of the Iron Gate dam in 1974 (Humborg *et al.*, 1997) as well as intense consumption during the eutrophication-induced algal blooms of the 1970s (Cociasu *et al.*, 1998). Though concentrations have decreased, the Black Sea is not silicon-limited (Ragueneau *et al.*, 2002). Overall, shelf waters of the Black Sea are phosphorus-limited and open waters are nitrogen-limited (Cociasu *et al.*, 1998; daNUbs, 2005). More specifically, spring phytoplankton growth is limited by phosphorus while summer growth is regulated by both phosphorus and nitrogen. In winter, the open Black

Sea is nitrogen- and phosphorus-limited, and the Danube plume is phosphorus-limited (Mee *et al.*, 2005). The artificial supplementation of nitrogen and phosphorus without that of silicon causes a deviation from ideal relative nutrient ratios (Redfield *et al.*, 1963), which favors the growth of dinoflagellates rather than the more trophically useful diatoms (Officer and Ryther, 1980a; Smayda, 1990; Turner, 2002).

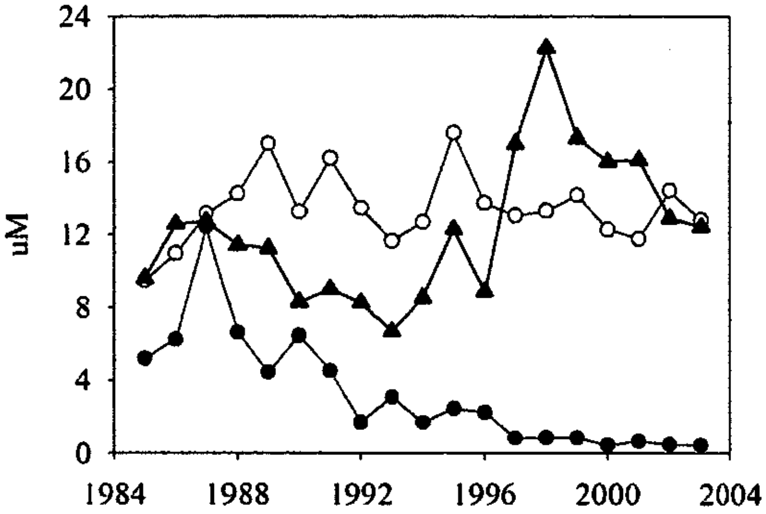


Figure 3.2. Trends in nutrient concentrations measured at Constanta, Romania: (total nitrogen: O; phosphate: ●; silicate: ▲) (from Cociasu and Popa, 2004).

As these eutrophic conditions accelerated, the northwestern shelf experienced an increase in the magnitude, extent, and frequency of algal blooms (Bodeanu, 1993) as well as the occurrence of a number of harmful algal bloom (or red tide) events (Moncheva *et al.*, 2001). The phytoplankton composition changed drastically between 1960 and 1990 – the number of non-diatom algal blooms increased and the proportion of diatoms in the phytoplankton community fell from 92% in the 1960s to 38% in the 1980s (Bodeanu, 2002). Diatoms were replaced with less trophically useful phytoplankton groups such as dinoflagellates, mixotrophs, and cyanophytes (Bodeanu *et al.*, 2004). Diatoms are the basis of the healthy copepod-fish food web and this change in community composition can affect higher trophic levels; dinoflagellate dominated communities may lead to food webs culminating in gelatinous 'trophic dead ends', while diatom dominated communities lead to 'muscular', trophically sound food webs culminating in fish (Turner, 2002; Verity and Smetacek, 1996).

Alterations to the phytoplankton community precipitated changes in the zooplankton. A decrease in non-gelatinous zooplankton species, including trophically important copepods, was recorded. Concurrently, edible zooplankton were replaced by gelatinous plankton, such as *Noctiluca scintillans*, *Aurelia aurita*, *Rhizostoma pulmo* and *Mnemiopsis leidyi* (Shiganova and Bulgakova, 2000). Species diversity and numbers of ichthyoplankton and eggs of several valuable fish species (bonito, *Sarda sarda*; bluefish, *Pomatomus saltatrix*; flounder, *Platichthys flesus*; turbot, *Psetta maxima*; sole, *Solea lascaris*) decreased (Shiganova and Bulgakova, 2000). These disruptions to the food web damaged area fisheries (Daskalov, 2002).

The increased phytoplankton biomass caused accelerated sedimentation of organic matter from the water column to the seafloor, resulting in hypoxic conditions as bacteria consumed the dead plankton. The first recorded hypoxic event occurred in 1973, and caused a mass mortality of benthic biota and fish in a 3500 km^2 area near the Danube delta (Zaitsev and Mamaev, 1997). Hypoxia increased in shelf waters throughout the 1970s and 1980s, covering areas of up to 40,000 km^2 (Zaitsev and Mamaev, 1997). Hypoxia, along with increased water column turbidity resulting from increased phytoplankton biomass, limited light and oxygen available to benthic plants, and resulted in the decline of macroalgal species such as the red alga, *Phyllophora*, and the brown alga, *Cystoseira*. By 1991, only 5% of *Phyllophora*'s original area remained (Mee, 1992) and *Cystoseira* had nearly disappeared from the shelf by 1981 (Zaitsev and Mamaev, 1997). Not only were *Phyllophora* and *Cystoseira* key benthic habitats for a large group of associated species, they are important oxygen generating species (Zaitsev and Mamaev, 1997). The increased hypoxic area also caused the loss of mussel beds. Because mussels are filter feeders, they provide important water filtering capacity in the Black Sea, and their decline, as with the loss of oxygen-producing macroalgae, further exacerbated the effects of eutrophication.

Increased phytoplankton biomass, changes in plankton community composition, hypoxic events, and the loss of benthic assemblages did not occur in isolation, but concurrently with the continued overexploitation of commercially important fish stocks (Daskalov, 2002). During the 1950s and 1960s stocks of top predators, such as tuna (*Thunnus thynnus*) and swordfish (*Xiphias gladius*), were dramatically reduced through overfishing. As the number of top predators declined a trophic cascade occurred, resulting in an increase in small planktivorous fish, causing a consequent decrease in zooplankton grazers. The reduced abundance of zooplankton removed grazing pressure from the growing phytoplankton biomass (Daskalov, 2002).

Weakened ecosystem resilience due to eutrophication and the effects of overfishing left the Black Sea ecosystem vulnerable to invasive species. In the mid 1980s, the Black Sea was invaded by the gelatinous ctenophore *Mnemiopsis leidyi*, probably transported in ballast water from eastern North America. Lacking predators and finding favourable prey and environmental conditions, its biomass grew rapidly, effectively dominating the food chain and precipitating a collapse in small pelagic fish stocks within 5 years of its arrival (Kideys, 2002b). In 1989 the live biomass of *Mnemiopsis* reached an astonishing 800 million tons in the Black Sea (Vinogradov, 1990). Because *Mnemiopsis* preys on anchovy (*Engraulis encrasicolus*) larvae and eggs and competes with anchovy for food (Kideys, 2002a), the amount of non-gelatinous zooplankton available as fish fodder decreased drastically after its establishment (Kideys, 2002b; Shiganova and Bulgakova, 2000). As a result, catches of anchovy decreased sharply, economically damaging this important fishery (Kideys, 2002b). In 1997, the Black Sea was invaded by another ctenophore, *Beroe ovata*, which, as a natural predator of *Mnemiopsis*, significantly reduced its population (Shiganova *et al.*, 2001), resulting in the increase of non-gelatinous zooplankton, anchovy larvae, and eggs, and increased biomass of native gelatinous plankton (*Aurelia aurita*, *Rhizostoma pulmo*) (Kideys, 2002b).

Due to economic decline in the 1990s of all Black Sea coastal countries except Turkey, and the consequent lack of support for centrally administered agriculture, the marine ecosystem has recently shown signs of recovery (Mee *et al.*, 2005). The late 1990s saw fewer phytoplankton blooms than previous decades and the community structure has returned to a diatom-dominated state (Bodeanu, 2002). No hypoxic events were recorded on the shelf between 1993 and 2001 (Mee, 2006) and *Phyllophora* has once again been found on the northwestern shelf, although species associated with the habitat are different than those common prior to eutrophication (Mee *et al.*, 2005).

Although possible, recovery of the Black Sea is far from certain. During the exceptionally warm years of 2001 and 2002, 15 monospecific blooms occurred in Romanian waters and the phytoplankton community was again dominated by non-diatoms (Bodeanu *et al.*, 2004). Additionally, the climatic conditions of 2001 triggered a large scale hypoxic event (Mee *et al.*, 2005) which resulted in fish mortalities in shelf waters (Boicenco, personal communication). These signs indicate that nutrient concentrations in shelf waters remain sufficiently elevated to support eutrophic events, particularly in combination with anomalous climatic conditions. Additionally, overfishing continues, with further depletion of remaining stocks likely (Mee *et al.*, in prep.).

However, the Black Sea's future remains uncertain. Because of expected rapid recovery of farming in Eastern Europe and additional pressures from shipping, poorly regulated fisheries, and sewage treatment without nutrient removal, the Black Sea may potentially revert back to its eutrophic state (Mee *et al.*, in prep.). Alternatively, this period of newfound regional prosperity, particularly with the expansion of the EU to include Bulgaria, Romania, and possibly Turkey, may be used as an opportunity for remediation of the Black Sea ecosystem. Even if eutrophication and overfishing are mitigated, there is no guarantee that the ecosystem will return to its pre-eutrophic state (Mee *et al.*, 2005).

3.2 Sources of nutrient enrichment

Riverine input is the primary source of land-based nutrients to the Black Sea (Black Sea Commission, 2002), contributing approximately 63% of nitrogen and 76% of phosphorus (Fig. 3.3a,b). The Danube is responsible for 75% of the fresh water reaching the Black Sea (Cociasu and Popa, 2004) and is thus the most important source of nutrients, contributing 52% of nitrogen and 50% of phosphorus (Black Sea Environmental Programme, 1996). Although the amount of fertilizer consumed in the Danube catchment has decreased significantly (Mee, 2006) (Fig. 3.4b), the amount of nitrogen in Danube waters has decreased only slightly due to retention of the nutrient in catchment soils and groundwater (daNUbs, 2005). The decreased load since 1990 (Fig. 3.4a) may be attributed to implementation of the Nitrates Directive in EU countries (Black Sea Commission, 2002; European Union, 1991a), reduced fertilizer usage after the collapse of the communist bloc (Black Sea Commission, 2002), and greater waste water treatment (Black Sea Commission, 2002). Conversely, the Danube's phosphorus load has shrunk considerably since 1990 due to policy initiatives such as the Urban Waste Water Treatment Directive (European Union, 1991b). As a result of decreasing nutrient loads in the Danube, the concentrations of phosphorus and nitrogen on the Black Sea shelf have also begun to decrease (Cociasu and Popa, 2004), with phosphorus responding more quickly than nitrogen.

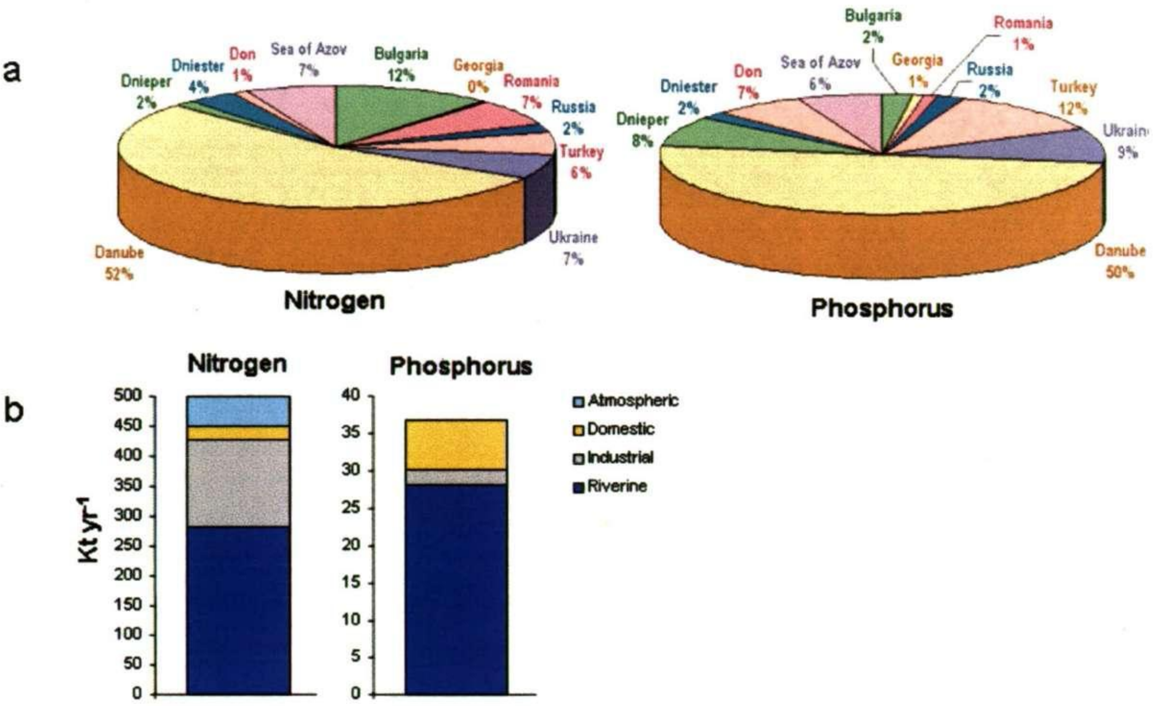


Figure 3.3. Nutrient inputs to the Black Sea (a) by source (after Black Sea Environmental Programme, 1996) and (b) by anthropogenic sector. Atmospheric contribution applies to northwestern shelf exclusively (data from Black Sea Commission, 2002; Langmead *et al.*, 2007).

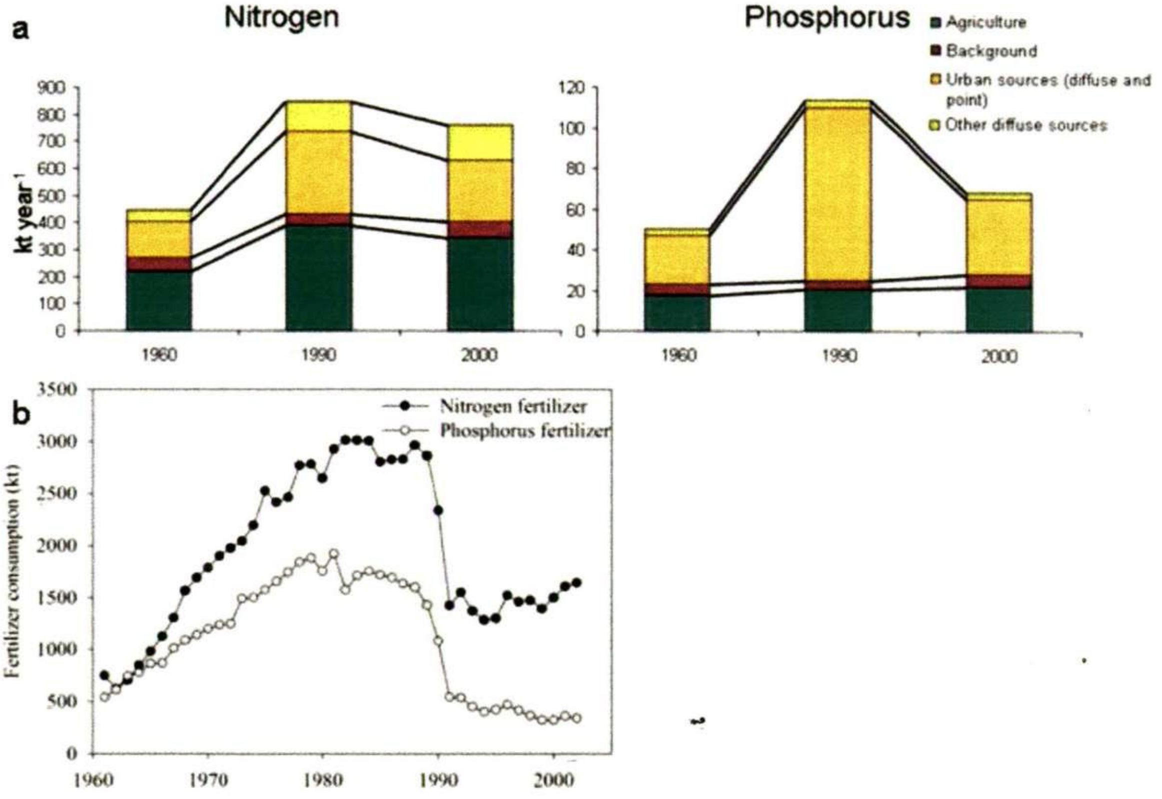


Figure 3.4. (a) Sources of nutrients to the Danube (daNUbs, 2005; Mee *et al.*, 2005). (b) Consumption of fertilizer in the Danube catchment (after Mee, 2006).

Industry is the second greatest contributor of land-based nitrogen (30%) directly to the Black Sea, although the amount of phosphorus input is relatively low (<5%) (Black Sea Commission, 2002). The Black Sea Transboundary Diagnostic Analysis (TDA) completed in 1996 found that most industrial nitrogen entering the Black Sea comes from Ukraine (49%), Russia (30%) and Bulgaria (18%), while Ukraine contributes 84% of industrial phosphorus (Fig. 3.5a) (Black Sea Environmental Programme, 1996). Domestic waste water is a minor contributor of nitrogen (<5%) and a modest contributor of phosphorus (18%) to the Black Sea (Fig. 3.5b). Bulgaria is the leading contributor of domestic nitrogen (47%), followed by Georgia (27%) and Ukraine (12%) while Ukraine (38%) and Turkey (33%) are the most important contributors of domestic phosphorus to the Black Sea (Fig. 3.5b) (Black Sea Environmental Programme, 1996). The majority of industries in all Black Sea coastal states are connected to municipal wastewater treatment systems. Therefore, with implementation of the Urban Waste Water Treatment Directive in the accession states and recovering economies in other Black Sea nations enabling more efficient sewage treatment, the amount of untreated domestic and industrial waste entering the Black Sea directly is expected to decrease (Black Sea Commission, 2002). Atmospheric deposition contributes approximate 10% of the nitrogen load to the Black Sea (Fig. 3.3b), a minor amount compared to the riverine and industrial contributions.

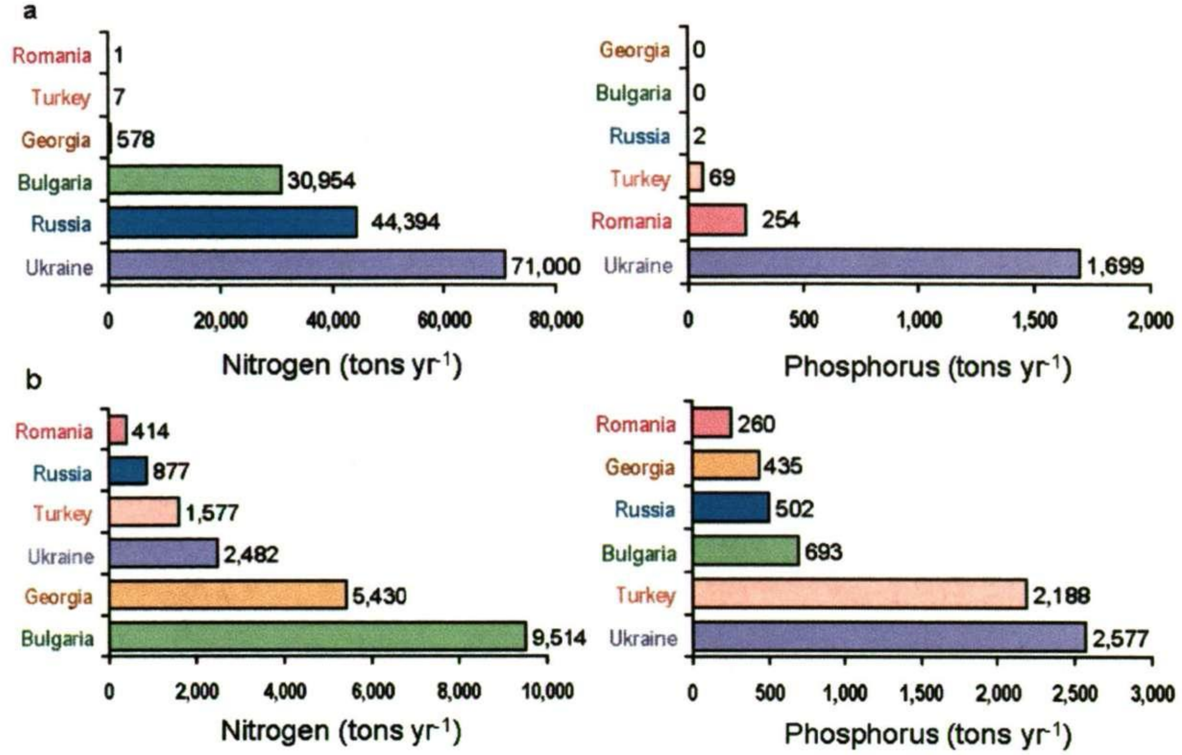


Figure 3.5. (a) Industrial and (b) domestic sources of nutrients to the Black Sea (from Black Sea Environmental Programme, 1996).

3.3 Policies and institutions

3.3.1 Regional initiatives

The Bucharest Convention for the Protection of the Black Sea, signed in 1992 and ratified in 1994, was the first step toward protection of the Black Sea ecosystem (Mee, 2005b). The Bucharest Convention (Black Sea Commission, 1992) was signed by the six Black Sea coastal countries and called for the establishment of the Black Sea Commission, to be based in Istanbul (see Table 3.1). However, the Commission did not establish its Secretariat until 2000. Due to budget constraints, the Commission Secretariat has undergone financial difficulties, but has managed to implement its workplan and develop new legal instruments such as the Protocol on Biodiversity and Landscape Conservation, signed in 2003 (Mee, 2005b). The Convention Secretariat relies on information on compliance volunteered from the coastal countries and has no independent means of verifying whether or not the Bucharest Convention is being successfully implemented. Fortunately, the Global Environmental Facility (GEF) has continued to support the Secretariat in specific areas, particularly the control of eutrophication which was signalled as a clear priority in the BS-SAP (see below). Finance for this work is now guaranteed until 2008.

The Odessa Ministerial Declaration (Black Sea Ministers of Environment, 1993), signed in 1993, was a response to the UN Conference on Environment and Development (the Rio Summit) held in 1992. The Rio Summit's Agenda 21 encouraged a new comprehensive approach to policy. The Odessa Declaration was signed by environmental ministers in the Black Sea for the purpose of implementing joint policy regarding the protection of the Black Sea. The Declaration features nineteen specific actions designed to facilitate the rapid development of practical measures for controlling pollution from land-based and marine sources (including the harmonization of environmental standards); restore, conserve and manage natural resources; respond to environmental emergencies; improve the assessment of contaminants and their sources; introduce integrated coastal zone management policies and compulsory environmental impact assessments; and to create a transparent and balanced mechanism for reviewing and updating the Declaration on a triennial basis (Mee, 2005b).

Because the Odessa Declaration came at a time of political tension in the Black Sea basin, its enactment was interpreted as a gesture of willingness to cooperate between the Black Sea countries. The GEF and European Commission provided US$11 million to establish the Black Sea Environmental Programme (BSEP) in order to facilitate capacity building for assessing and controlling pollution in the Black Sea, to create a long-term 'Black Sea Strategic Action Plan (BS-SAP)' and to facilitate further investment in environmental protection (Black Sea Commission, 1996; Mee, 2005b). In 1996 the BS-SAP was adopted with the purpose of defining a long-term policy agenda and establishing an institutional framework for achieving it (Mee, 2005b, see also Adaptive management section). Although the BS-SAP's Project Implementation Unit is formally part of the Istanbul Commission Secretariat, it is supported financially by the EU and GEF.

Because eutrophication in the Black Sea is fundamentally linked with nutrient loads in the Danube, the river has also been subjected to a number of conventions and plans. The Danube River Protection Convention was ratified in 1998 with the purpose of preventing, controlling and reducing significant adverse transboundary impacts from the release of hazardous substances and of nutrients into the aquatic environment within the Danube Basin (International Commission for the Protection of the Danube River, 1994). The Convention established a Secretariat, known as the International Commission for the Protection of the Danube River (ICPDR). The Danube River Basin also has its own Strategic Action Plan, also under the auspices of the ICPDR, which was adopted in 1995 with the support of the European Commission, UNDP, and GEF (International Commission for the Protection of the Danube River, 1999). Both the Convention and the SAP name the protection of the Black Sea as a priority (Mee, 2005b).

3.3.2 European Union initiatives

In addition to political and legal instruments affecting the Black Sea and Danube Basins, several EU policies now affect the Black Sea. In 1991 the Urban Waste Water Treatment Directive (also referred to as the 'Phosphates Directive') was adopted by the EU (European Union, 1991b). The Directive mandates the secondary treatment of waste water (a key source of phosphate) in towns with a population >2000 and its establishment has effectively reduced the amount of phosphorus in Danube and Black Sea waters (Cociasu and Popa, 2004). The Nitrates Directive was also established in 1991, and promotes the use of sound agricultural practices, the most important source of nitrate pollution, in EU countries (European Union, 1991a). The Urban Waste Water Treatment and Nitrates Directives were superseded in 2000 by the Water Framework Directive (WFD), a key piece of legislation that requires all European waters to achieve 'good ecological status' for waters within 1 nm of the coast by 2015 (European Union, 2000). The WFD promotes the sustainable use of water and a transboundary system of water management which will be achieved through the establishment of river basin-scale monitoring networks; in most cases, regulations imposed by the WFD are more rigorous than those imposed by the Nitrates and Urban Waste Water Treatment Directives. The ICPDR is the coordinating body responsible for the development of WFD compliance plan in the Danube catchment. As an EU state, Bulgaria has also established a River Basin Directorate to ensure national compliance with the WFD in the Bulgarian Black Sea catchment (Pratt *et al.*, 2005). The implementation approach differs in Romania, where two river basins, the Somes and the Arges, were used as pilot sites, with the results and methodologies to be disseminated for application elsewhere in Romania (Romanian Ministry of Environment and Sustainable Development, 2007). In Turkey, a potential EU Member State, a similar process is underway, with the designation of the Buyuk Menderes river basin as a pilot site from which to establish a river basin management plan that will then be replicated in other Turkish basins (de Bruin *et al.*, 2005). The process is proving difficult in Turkey, however, due to a lack of (but improving) integrated management and the need for improvements in inter-institutional coordination and cooperation (de Bruin *et al.*, 2005).

In addition to the Directives mentioned above, the EU's Common Agricultural Policy (CAP) influences nutrient loads to the Black Sea and Danube. The CAP uses monetary incentives (such as direct income support) to encourage 'good farming practices' (European Union, reformed 1999, 2003). Measures beyond 'good farming practices' (such as organic farming, extensification, landscape preservation, etc) are further rewarded. New Member States may receive subsidies to aid their populations in the transition from semi-subsistence farms to commercially viable farms which will likely result in the intensification of farming practices and creation of larger farms in new Member States (European Union, reformed 1999, 2003).

3.3.3 Adaptive management in the Black Sea

In 2001, a Memorandum of Understanding (MOU) was issued between the Black Sea Commission and the ICPDR, establishing the Danube-Black Sea Task Force (DABLAS) whose mission is to secure financing for the implementation of investment projects for pollution reduction and the rehabilitation of ecosystems in the Black Sea region (International Commission for the Protection of the Danube River, ; Mee, 2005b). These objectives are based on the 'adaptive management' technique (Fig. 3.6), which recognizes that long-term management decisions based upon conceptual modelling or knowledge of only a limited part of the system are not practical due to high levels of scientific uncertainty in natural systems (Holling, 1978; Mee *et al.*, 2005). Adaptive management offers the opportunity for taking a flexible and pragmatic approach to restoring and protecting the Black Sea - it actually treats a system as a management 'experiment', adapting management policies and goals based upon knowledge gained. It is strongly rooted in the 'Ecosystem Approach' a concept that regards humans as an integral part of the ecosystem and environmental management as more a matter of managing humans than managing the environment (Mee, 2005b). Adaptive management can accommodate unexpected events by encouraging approaches that build system resilience (the capacity of a system to absorb change without losing its basic functionality) (Mee *et al.*, 2005). To do this, it integrates social, economic and ecological knowledge, all essential for development of realistic targets for ecosystem improvement. Thus, good scientific knowledge of the system is crucial in order to reduce management uncertainties and cooperation of all key actors in the region is necessary in order to achieve the success of an adaptive management plan (Mee, 2005b).

Adaptive management sets both a long-term vision (supported by measurable Ecosystem Quality Objectives – EcoQOs) as well as short-term goals (operational targets) for ecosystem improvement. The formulation of the long-term vision and EcoQOs is often contentious as the desired outcomes are not always possible (Mee *et al.*, 2005). Unlike top-down management schemes, EcoQOs must be understandable by both the general public and stakeholders and stakeholders are integral in the development of EcoQOs and the operational targets needed to meet them (Mee *et al.*, 2005). Progress towards operational targets and the responses of the socio-ecological system are continually monitored; this information reduces uncertainty about management outcomes and helps policymakers to define the next step (Mee *et al.*, 2005).

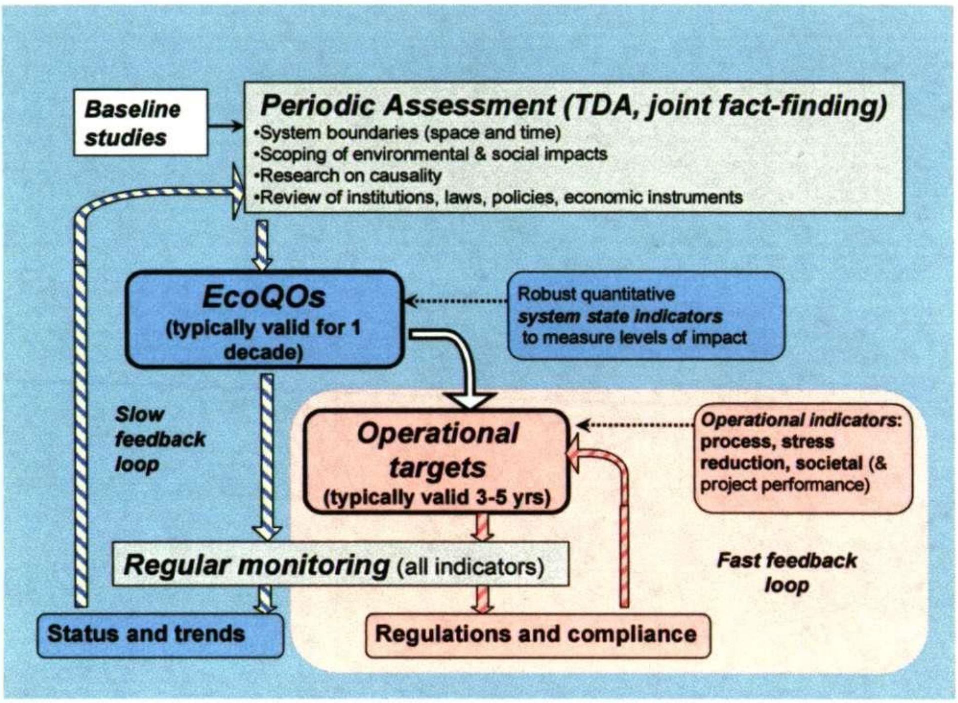


Figure 3.6. General scheme for implementing adaptive management currently applied in the Black Sea (from Mee, 2005b).

Before EcoQOs and operation targets can be set, the state of the marine system must be assessed. This is achieved through a Transboundary Diagnostic Analysis (an overall assessment of the state of the marine environment and the social and economic reasons for its decline which is now applied in most GEF international waters projects) (Mee, 2005b). The Black Sea TDA (Black Sea Environmental Programme, 1996) was completed in 1996, and integrated information from existing research and monitoring. The TDA highlighted knowledge gaps in the understanding of the Black Sea socio-ecological system, but provided a sufficient factual base to allow the agreement of an overall vision of the Black Sea's future and the preliminary steps (EcoQOs and operational targets) needed to accomplish it (Mee *et al.*, 2005).

The Black Sea vision statement expresses the desire for:

> *a biologically diverse Black Sea ecosystem with viable natural populations of higher organisms, including marine mammals and sturgeons, and which will support livelihoods based on sustainable activities such as fishing, aquaculture and tourism in all Black Sea countries* (Black Sea Commission, 1996; Mee *et al.*, 2005).

Achievement of the vision statement is the goal of the Black Sea Strategic Action Plan (BS-SAP). Eutrophication is named as a priority issue in the BS-SAP and the first milestones to its abatement were clarified in the 2001 Memorandum of Understanding issued by the Black Sea and Danube Commissions:

> **Long-term objective (EcoQO):** *"to take measures to reduce the loads of nutrients and hazardous substances discharged to such levels necessary to permit Black Sea ecosystems to recover to conditions similar to those observed in the 1960s"*
>
> **First operational target:** *"urgent measures should be taken in the wider Black Sea Basin in order to avoid that the loads of nutrients and hazardous substances discharged into the Seas exceed those that existed in the mid 1990s (these discharges are only incompletely known)"* (from Mee *et al.*, 2005)

The GEF and the European Union, working closely with the Secretariat to the Black Sea Commission, have funded a number of activities that will help to achieve and monitor the attainment of the first operational target (from Mee, 2005b):

- > US$100 Million of large demonstration projects in nutrient reduction. The projects are being conducted in the framework of a Black Sea Strategic Partnership involving the GEF, the World Bank and governments. They target key activities in agriculture, urban waste-water treatment and wetland restoration. Though these will not by themselves make a significant difference to nutrient discharges, they are highly replicable and generate a number of other benefits that should make replication highly attractive. The EU's DABLAS programme has similar aims and objectives.
- An International Study Group on the Black Sea with a programme of active research to reduce some of the scientific uncertainties impeding the refinement of the operational targets for adaptive management.
- Technical support for specific aspects of the work of the Black Sea Commission to help meet national and international obligations for monitoring and assessment. This has lead to a new 'State of the Black Sea' TDA report to be published at the end of 2007.

Table 3.1. International legal and policy instruments addressing eutrophication and nutrient control in the Black Sea and Danube basins (adapted from Mee, 2005b).

Instrument	Secretariat	Observations/Objectives	Further Information
The Bucharest Convention for the Protection of the Black Sea Against Pollution. Signed 1992, Ratified, 1994	The Black Sea Commission with a Secretariat in Istanbul (in place since 2000)	Protocols for Land base source pollution, dumping, accidental releases and biodiversity and landscape conservation are in force.	http://www.blacksea-commission.org/OfficialDocuments/Convention_iframe.htm
The Odessa Declaration for the Protection of the Black Sea. Signed, April 1993	Black Sea Environmental Programme, Programme Co-ordination Unit (became PIU in 1998, see below)	A pragmatic 3-year policy agreement largely implemented with financial and technical support from the GEF and EC.	http://www.blacksea-commission.org/OfficialDocuments/OdessaDeclaration_iframe.htm
The Danube River Protection Convention. Signed 1994, Ratified, 1998	The International Commission for the Protection of the Danube River (ICPDR), Vienna, operational since 1999)	The ICPDR's role is rapidly developing following entry into force of the Convention.	http://www.icpdr.org/
Strategic Action Plan for the Danube River Basin. Adopted in 1995	Originally managed by the Environmental Programme for the Danube River Basin with a co-ordinating unit in Vienna, Austria. Now a responsibility of the ICPDR.	A wide-ranging basin-wide plan developed with the support of the European Commission.	http://www.icpdr.org/
The Black Sea Strategic Action Plan. Signed and adopted in October 1996.	Currently managed by a Project Implementation Unit (PIU) in Istanbul, now formally part of the Istanbul Black Sea Commission Secretariat but supported financially by the GEF and the EC.	A wide ranging decadal plan covering many aspects of environmental protection in the Black Sea. Sets goals and milestones. Revision anticipated in 2002.	http://www.blacksea-commission.org/OfficialDocuments/BSSAP_iframe.htm

Danube Pollution Prevention Programme. Agreed in 1999	Its implementation is the responsibility of the ICPDR. Practical management is through the Programme Management Task Force. The GEF has provided new (2002) funding for a project to support its implementation.	Developed through a project funded by the GEF, it includes detailed studies of actions to reduce transboundary pollution in the Danube River.	http://www.icpdr.org/
Memorandum of Understanding (MOU) between the Black Sea Commission and the ICPDR on common strategic goals (Nov. 2001)	DABLAS Task Force for co-operation on water protection in the wider Black Sea Region (formed Nov. 2001), Secretariat: EC DG-Environment to arrange.	DABLAS has 'effective implementation of the MOU' as its first objective. DABLAS itself was established on the principles of the EC Water Framework Directive and the EC Communication 615 (Oct. 2001). Sets targets for protecting Black Sea from eutrophication.	http://ec.europa.eu/environment/enlarg/pdf/danube_memorandum.pdf http://www.icpdr.org/icpdr-pages/dablas.htm
EU – Urban Waste Water Treatment Directive (also referred to as the 'Phosphates Directive'), adopted 1991	Applicable to EU countries (Bulgaria, Romania, most countries in the Danube catchment) and potential EU Member States (Turkey).	Objective: to protect the environment from the adverse effects of urban waste water discharges and discharges from certain industrial sectors Applicable to EU countries (Bulgaria, Romania, most countries in the Danube catchment) and potential EU Member States (Turkey).	http://ec.europa.eu/environment/water/water-urbanwaste/index_en.html
EU – Nitrates Directive, adopted 1991	Applicable to EU countries (Bulgaria, Romania, most countries in the Danube catchment) and potential EU Member States (Turkey).	Objective: designation of nitrate vulnerable zones (NVZs), establishment of code of good agricultural practice, but limited monitoring requirements.	http://ec.europa.eu/environment/water/water-nitrates/index_en.html

EU – Common Agricultural Policy (CAP), adopted 1962. 1999 and 2003 reforms integrated environmental requirements into CAP	Applicable to EU countries (Bulgaria, Romania, most countries in the Danube catchment) and potential EU Member States (Turkey).	Objective: self sufficiency for food in Europe, efficient and sustainable farming practices, aid for rural development, establishment of food quality standards.	http://ec.europa.eu/agriculture/index_en.htm
EU – Water Framework Directive, adopted 2000	Applicable to EU countries (Bulgaria, Romania, most countries in the Danube catchment) and potential EU Member States (Turkey).	Objective: requires achievement of 'good ecological status' for all European waters by 2015. More stringent than Urban Waste Water and Nitrates Directive. Only applies to 1nm from coast.	http://ec.europa.eu/environment/water/water-framework/index_en.html For the Danube: http://www.icpdr.org/icpdr-pages/river_basin_management.htm

3.4 Successes and failures

The political and institutional instruments in Table 3.1 demonstrate that a clear willingness to protect and rehabilitate the Black Sea exists, although adequate funding often does not. When the Soviet Union collapsed, many ex-communist countries restructured their social welfare institutions, resulting in considerable financial and influential down-scaling of environmental ministries (Mee, 2005b). Fortunately, funding bodies such as the GEF and the EC have supported the Black Sea Commission when local financial support was unavailable. The Black Sea Commission plays a key role in our understanding of eutrophication and the identification of nutrient sources in the Black Sea. Outside support of organizations such as the Black Sea Commission and the Danube Pollution Prevention Programme is crucial as although the region is recovering economically, environmental protection is not a priority in many non-EU Black Sea states.

Compliance with the regulations, plans and directives in Table 3.1 is key to determining their effectiveness. National environment sectors, particularly in the non-EU states, may not have the means to enforce compliance or to monitor implementation (Mee, 2005b). Stakeholder involvement in the development of many regional policies, plans, and institutions was limited, which may result in low compliance. For example, top-down management strategies are still dominant in the Black Sea region and environmental education is limited (Mee, 2005b).

In a region undergoing economic, political and social transition, adaptive management provides a flexible approach to ecosystem restoration in the Black Sea. However, although the BS-SAP's EcoQOs and operational targets were developed with stakeholder involvement, the BS-SAP was not distributed in local languages to regional stakeholders (Mee, 2005b). The support of local stakeholders is critical for the success of an adaptive management scheme.

European accession can be a 'double edged sword' as it will improve environmental regulations but has the potential to damage the recovering Black Sea ecosystem. Eastern European countries are working to strengthen their economies (including the agricultural sector) and environmental protection may not be a priority. The Common Agricultural Policy may actually result in increased fertilizer application in Eastern Europe as it is likely to accelerate changing farming practices in the new Member States, thereby increasing nutrient loads to the Black Sea and heightening the risk of eutrophication (Mee *et al.*, 2005). The Urban Waste Water Treatment Directive will improve sanitation in the region, but if waste is not sufficiently treated (*i.e.* tertiary treatment which removes nutrients as opposed to primary or secondary treatment which does not), nutrient loads may increase as more people are connected to sewage treatment (Mee, 2005b).

It is too early to determine if the Water Framework Directive will be a success or a failure for the Black Sea. The WFD is only applicable to 1 nm from the coast, an arbitrary distance that results in limited authority over the coastal zone (de Bruin *et al.*, 2005). Furthermore, the 'good ecological status' required by the directive is never explicitly defined in the WFD and refers more to rivers, lakes and estuaries than marine receiving waters, a clear mismatch in scale. In this way although eutrophication may be a problem in coastal waters into which a river drains, the river basin management body may not consider the reduction of nutrient loading a priority (de Bruin *et al.*, 2005). However, the establishment of a river basin-scale level of management is generally positive as are the requirements for monitoring, particularly in countries that aspire to join the EU, such as Turkey, and, eventually, Georgia and Ukraine. The European Marine Strategy (EMS), currently under negotiation in the EU, will reinforce the work of the Black Sea Commission and aid in the implementation of the WFD and future European Directives. The EMS will require countries to agree on common environmental targets in order to achieve 'good environmental status' for each regional sea. For the Black Sea, setting these targets will be a complex issue as the sea is shared between EU and non-EU countries. A strong Black Sea Commission will be essential for this.

An investigation into the future of the Black Sea ecosystem has recently been completed as part of the EU-funded Framework Programme 6 European Lifestyles and Marine Ecosystems (ELME) project (Langmead *et al.*, 2007; Mee *et al.*, in prep.). ELME integrated relevant information concerning current major ecosystem state changes, the pressures on the marine environment causing these changes, and the socio-economic drivers leading to these pressures. These data were used to model plausible scenarios of socio-economic change in Europe during the next 25 years and the resultant effects on the marine environment (Langmead *et al.*, 2007). Results indicate that under current political and socio-economic conditions in Europe the exploitation of environmental resources is likely to increase due to the continued economic development of post-Soviet countries.

However, increased infrastructure development is likely to include increased urban waste water treatment, although the expansion of infrastructure may result in a loss of wetland habitat (essential for filtering nutrients). Due to the CAP livestock production may intensify, resulting in increase nitrogen input to the Black Sea and a subsequent increase in phytoplankton biomass, decrease in water clarity, decline in seagrass and macroalgal health, and increased hypoxic events. ELME models also indicate that if nutrient levels can be controlled though regulation of farming practices, wetland destruction is reduced, and fishing pressure eased eutrophication may still be ameliorated (Langmead *et al.*, 2007; Mee *et al.*, in prep.).

3.5 Data availability and knowledge gaps

ELME identified existing datasets and useful proxies and compiled new datasets through extensive metadata analyses, resulting in a comprehensive database containing information about nutrient concentrations and loads; agricultural activities; waste disposal; invasive species; fishing effort, catch and stocks; turbidity; extent and frequency of hypoxia; phytoplankton biomass and composition; seagrass and macroalgal habitat status; and changes in areal coverage of wetland habitats (Langmead *et al.*, 2007; Mee *et al.*, in prep.). The ELME database focused on the western Black Sea as it is the region most extensively impacted by eutrophication and also contains the most comprehensive data, although these data vary in quality, ease of access and frequency of sampling. Limited data exist for the open and eastern Black Sea, but data are often difficult to access (unpublished or published in non-English languages) and regular monitoring programmes are few. Data for non-coastal Black Sea regions comes from research cruises, primarily carried out during the summer months, and so are limited in temporal and spatial coverage. A useful method of assessing long-term change due to eutrophication is through the establishment of nutrient budgets for different time periods using historical and modelled data (Artioli *et al.*, in prep.; Langmead *et al.*, 2007). However, although a 'contemporary' nutrient budget exists for the Black Sea (Artioli *et al.*, in prep.), there is a fundamental gap in our knowledge concerning likely 'baseline' and 'peak eutrophication' nutrient budgets. The lack of historic data, particularly with respect to nutrient budgets, makes it difficult to separate natural variability from anthropogenically-induced changes (Jackson, 2001). However, this may be achieved through interpretation of sediment core records, a technique that has already been applied in the Baltic Sea.

Like other socio-ecological systems, the Black Sea region suffers from mismatches in scale. For example, socio-economic data is often collected at a country-scale as opposed to a catchment-scale. This creates difficulties in determining likely drivers of ecosystem change. Additionally, some sectoral drivers of ecosystem change relating to coastal land use and resource exploitation, such as coastal urbanization, coastal development, coastal defence, and aggregate dredging, are difficult to quantify. These activities are expected to increase in the future but without a good understanding of past trends and their effects on the Black Sea's ecosystem, our ability to forecast future change is limited (Langmead *et al.*, 2007).

Modelling exercises may fill some of these gaps mentioned above, but no amount of model refinement can counter poor or absent data (Langmead *et al.*, 2007). These gaps in the basic knowledge of the Black Sea's ecosystem states and processes resulting in ecosystem change lead to difficulties in management caused by high levels of uncertainty. A coherent monitoring programme is crucial to understanding changes occurring in the Black Sea. For example, the Black Sea lacks a long-term spatially comprehensive plankton monitoring system. The Continuous Plankton Recorder (CPR), a long-term monitoring programme run by the Sir Alister Hardy Foundation for Ocean Science, has been in operation for more than 60 years in the North-East Atlantic and has been used extensively to assess climatic and anthropogenic ecosystem changes (Batten *et al.*, 2003a; McQuatters-Gollop *et al.*, 2007; Reid *et al.*, 1998). The extension of the CPR or a comparable monitoring system into the Black Sea could provide invaluable data on the pelagic system, aid our interpretation of observed ecosystem changes, and could even provide an early warning signal for future change.

In addition to establishing new monitoring programmes, scope also exists for the reanalysis of existing datasets to make them more applicable to the Black Sea. For example, NASA's SeaWiFS satellite system has been recording world-wide estimates of chlorophyll (as a measure of phytoplankton biomass) since 1997. This remote sensing dataset offers excellent spatial and temporal coverage of the Black Sea, however, the algorithm used to interpret the satellite measurements is inaccurate in the turbid, shallow Black Sea waters (Nezlin, 2001). The development of a regionally-specific algorithm would aid the re-interpretation of existing and future SeaWiFS data, creating a decade-long dataset of spatially comprehensive information about changes in Black Sea phytoplankton biomass.

3.6 Recommendations

The Black Sea is a complex socio-economic system and further research into nutrient dynamics is needed in order to fully understand the impact of nutrient loads on the Black Sea ecosystem. Research priorities include the creation of historical nutrient budgets which would allow valuable insight into the relationship between anthropogenic pressures and ecosystem effects. Additionally, further data on ecosystem responses are needed, particularly in non-coastal areas. Most knowledge gained recently has been through modelling (Mee *et al.*, 2005); while this information is useful, it does not replace field measurements. Field surveys, including video surveys which uniquely illustrate the state of the Black Sea, would be especially useful. For example, a long-term comprehensive plankton monitoring programme, such as an extension of the Continuous Plankton Recorder survey to include routes in the Black Sea, would be particularly valuable in the region. Time-series data concerning phytoplankton biomass and community dynamics could provide insight into eutrophication-induced changes in the Black Sea food web.

The eutrophication problem in the Black Sea cannot be effectively addressed without stakeholder involvement and education. While adaptive management offers the flexibility appropriate to a region undergoing rapid socio-economic change, no form of management is truly effective without good compliance. Because environmental education is not a priority in regional schools, the importance of the environment and the services it provides is severely underestimated. A school curriculum stressing the value of the Black Sea's resources and encouraging children to value their local environment from a young age could encourage environmental protection. At the same time, the involvement of stakeholders in the management decision making process and the distribution of educational materials and information regarding best use practices to stakeholders and members of the public would encourage compliance.

The Black Sea Commission has the potential to provide an effective institution through which to implement eutrophication-related policies and management strategies but is currently poorly funded and understaffed. Without adequate funding the Commission is unable to operate at the level required to mitigate nutrient-related problems in the Black Sea. The Commission should be reinforced but this must be matched against a commitment from governments to support it and carry forward its recommendations into national policies and legislation.

Chapter 4

Non-linearities, regime shifts and recovery: The recent influence of climate on Black Sea chlorophyll

The Black Sea ecosystem experienced severe eutrophication-related degradation during the 1970s and 1980s. However, in recent years the Black Sea has shown some signs of recovery which are often attributed to a reduction in nutrient loading. Here, SeaWiFS chlorophyll, a proxy for phytoplankton biomass, is used to investigate spatio-temporal patterns in Black Sea phytoplankton dynamics and to explore the potential role of climate in the Black Sea's recovery. Maps of chlorophyll anomalies, calculated relative to the 9 year mean, emphasise temporal variability of phytoplankton biomass in the Black Sea, particularly between the riverine-influenced Northwest shelf and the open Black Sea. Evolution of phytoplankton biomass has shown significant spatial variability of the 'system memory' between three major regions of the Black Sea. With the exception of 2001, chlorophyll has generally decreased during this 9 year time-series. However, the winter of 2000-2001 was anomalously warm with low wind stress, resulting in reduced vertical mixing of the water column and retention of nutrients in the photic zone. These conditions were associated with anomalously high levels of chlorophyll throughout much of the open Black Sea during the following spring and summer. The unusual climatic conditions occurring in 2001 may have caused a shift in the Black Sea's chlorophyll regime; however, the significance of this shift is still uncertain due to its recent occurrence. Nevertheless, climate appears to have played an important role in the recovery of the pelagic Black Sea ecosystem, although further ecosystem change may be difficult to predict due to the complex relationship between climate and chlorophyll.

Aspects of this chapter are included in the following:

McQuatters-Gollop, A., L.D. Mee, D.E. Raitsos, G. I. Shapiro (submitted). Non-linearities, regime shifts and recovery: The recent influence of climate on Black Sea chlorophyll. *Journal of Marine Systems.*

Chapter 4: Non-linearities, regime shifts and recovery: The recent influence of climate on Black Sea chlorophyll

4.1 Introduction

The pelagic Black Sea ecosystem has undergone significant changes during the last 30 years including habitat loss (Zaitsev and Mamaev, 1997), collapse of predatory fish stocks (Daskalov, 2002), wide-spread establishment of the invasive ctenophore *Mnemiopsis leidyi* (Kideys, 2002b), and massive phytoplankton blooms resulting in hypoxia and loss of benthic communities (Cociasu *et al.*, 1996). It is widely accepted that these changes are at least partially attributable to intense eutrophication, particularly in coastal waters (Mee, 1992; Mee, 2006; Niermann, 1999; Yunev *et al.*, 2002; Zaitsev and Mamaev, 1997). In recent years, the Black Sea has shown some signs of improvement such as an increase in the proportion of diatoms in the phytoplankton community (Bodeanu *et al.*, 2004), a decrease in the number of monospecific algal blooms (Bodeanu *et al.*, 2004), a decrease in phytoplankton biomass (Yunev *et al.*, 2002), and decreased area of hypoxia (Mee, 2006). This system recovery appears to be linked to a reduction in intensive farming practices after the collapse of the Soviet Union (Mee *et al.*, 2005). However, the role of climate in the Black Sea's recovery is unclear.

The aforementioned alterations to the Black Sea's ecosystem are all directly or indirectly connected to changes that occurred in the phytoplankton community. As phytoplankton comprise the base of the marine food web, alterations in phytoplankton production and community composition may have profound consequences for higher trophic levels (Edwards and Richardson, 2004). Due to their short life cycles and quick response to changes in their environment phytoplankton are sensitive to ecosystem change (Hays *et al.*, 2005). However, most of the historical ecological data available regarding plankton in the Black Sea are the result of near-shore monitoring programmes or occasional research cruises and are therefore limited in temporal and spatial extent and, consequently, the amount of information the data can provide. The SeaWiFS satellite, however, provides daily remotely-sensed spatially-comprehensive estimates of chlorophyll concentration, a proxy for phytoplankton biomass. Although SeaWiFS is not without limitations (see Oguz and Ediger, 2006 for more information on calibration issues), the comprehensive spatio-temporal nature of this dataset enables the exploration of changes in and the possible recovery of the pelagic Black Sea system.

The aims of the work are 1) to investigate variability in recent inter- and intra-annual chlorophyll dynamics in the Black Sea through the use of SeaWiFS satellite data, 2) to explain spatial patterns in phytoplankton biomass, and 3) to explore the role of climate in the recovery of the Black Sea.

4.2 Materials and methods

4.2.1 Study area

The Black Sea drains a catchment area containing large parts of 12 countries, covering a land area of 2 million km^2, and receiving waste water from more than 100 million people (Mee, 1992). The Black Sea is a nearly enclosed body of water, with only a narrow inlet to the Mediterranean through the Bosporus Strait. In its northwestern region, the Black Sea has a wide and biologically active continental shelf while the open sea is permanently anoxic below 100-150 m (Sorokin, 2002). Hydrographically, the Black Sea is divided into two distinct regions: the shallow (< 200 m) Northwest Shelf and the deep (> 1000 m) central sea (Fig. 4.1). The Northwest Shelf receives most of the nutrient load to the Black Sea through riverine inputs from the Dniester, Dnieper and Danube rivers and is therefore the region most severely impacted by eutrophication (Cociasu and Popa, 2004). A large part of the terrestrial nutrients entering the Black Sea originate in central and western Europe, particularly those transported by the Danube, which alone is responsible for 75% of total nutrient input to the Black Sea (Mee, 1992; Zaitsev and Mamaev, 1997). A rim current creates a fluid boundary at the edge of the shelf, separating shallow shelf and deep open waters (Simonov and Altman, 1991). This liquid boundary is intermittently crossed by mesoscale eddies and filaments providing an efficient exchange mechanism between nutrient-rich shelf waters and the less productive waters of the central basin (Enriquez *et al.*, 2005; Zatsepin *et al.*, 2003).

For the purpose of this study, the Black Sea is separated into three regions: (1) the shallow Northwest shelf, (2) the deeper western gyre and (3) the eastern open Black Sea (Fig. 4.1). The far eastern open sea, although deep, was excluded from this study due to the complex nature of the relatively poorly studied Batumi gyre. Polygons representing the open and coastal Black Sea study areas were used to regionally partition each geographically referenced data set used in the analysis (SeaWiFS chlorophyll, wind stress, sea surface temperature (SST)).

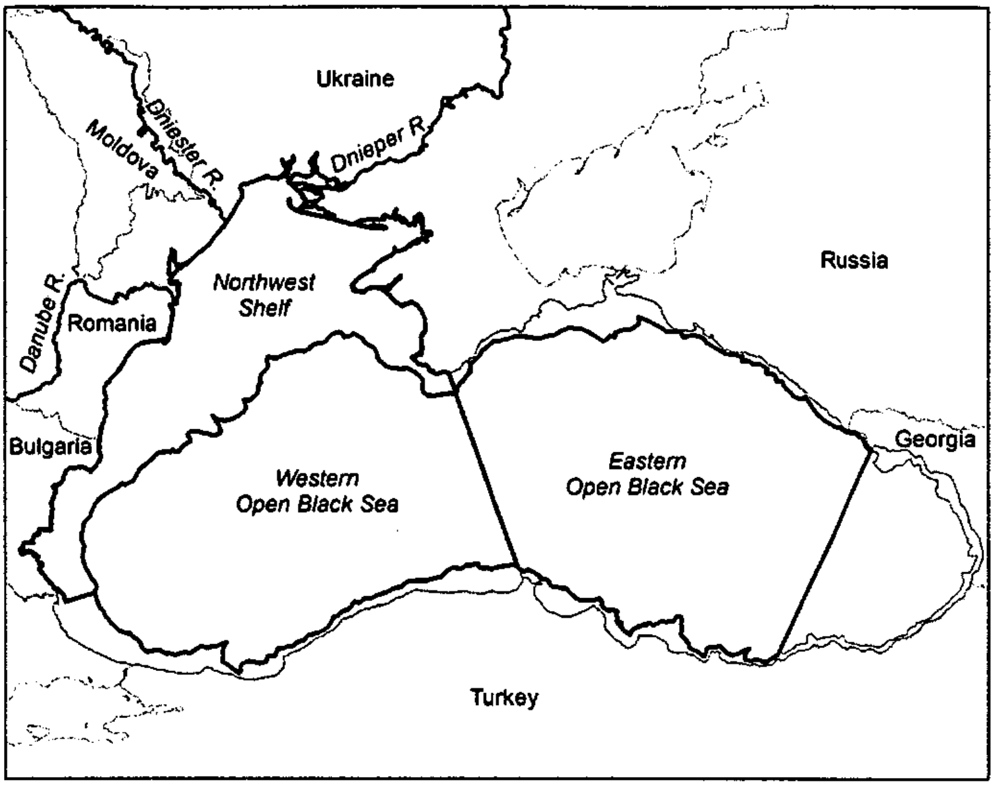


Figure 4.1. Black Sea regions used in this study. The line surrounding the open Black Sea is also the 200m isobath.

4.2.2 Data

Wind speed and sea surface temperature (SST) data were obtained from the NCEP/NCAR Reanalysis Project at the NOAA-CIRES Climate Diagnostics Center (NOAA-CIRES Climate Diagnostics Center, 2006). Wind speed was converted into wind stress, which is a function of wind speed, non-dimensional drag coefficient and boundary layer air density (Pond and Pickard, 1978). Wind stress regulates the dynamics of the boundary layer and is connected to the production of wind-driven surface currents, the generation of surface waves and upper-ocean mixing (Pond and Pickard, 1978). Therefore, low wind stress contributes to formation of highly stratified waters. Monthly wind stress and SST anomalies were calculated by subtracting from each monthly value the corresponding long-term monthly mean of the 100 month time-series. Danube river discharge data were obtained from the Global Runoff Data Centre, a digital worldwide repository of discharge data and associated metadata (Global Runoff Data Centre, 2005).

SeaWiFS current reprocessed version (v5.1) data produced by Ocean Biology Processing Group were acquired from the NASA Ocean Color website (NASA Ocean Biology Processing Group, 2006). The data were Level 3, 8-day products (9 km x 9 km square resolution) of the near-surface Chl *a* concentration (mg m^{-3}), estimated using the ocean Chlorophyll 4 - version 4 (OC4-v4) algorithm (O'Reilly *et al.*, 1998):

$$\mathrm{Chl}\ a = 10^{(0.366 - 3.067x + 1.930x^2 + 0.649x^3 - 1.532x^4)},$$

where $x = \log_{10}((R_{rs}443 > R_{rs}490 > R_{rs}510)/R_{rs}555)$ and R_{rs} is the satellite calculated remote sensing reflectance. The first (x) equation takes the highest reflectance value from a waveband at 443, 490, or 510 nm and divides it by the reflectance at 555 nm; reflectance maximum moves towards higher wavebands as the Chl *a* concentration increases. This x value then goes into an equation that results from a statistical fit of this empirical algorithm to a large *in situ* database. NASA processed these data using a series of radiometric corrections (*e.g.*, atmospheric) to eliminate the presence of clouds, haze, and water vapour (Mueller and Austin, 1995). The 8-day products were aggregated into monthly means from September 1997 through December 2005.

Standardized chlorophyll anomalies (z) were calculated as:

$$z = \frac{(x - \bar{x})}{\sigma}$$

in order to remove the long-term mean, to emphasize subtle temporal patterns in productivity, and to allow for the relative comparison of chlorophyll dynamics. This was done for each SeaWiFS measurement (n=6683) at three temporal scales: 1) the intra-annual (monthly composite) anomaly was calculated based on the long-term mean of each calendar month relative to the composite annual mean; 2) the inter-annual chlorophyll anomaly was calculated from the yearly mean of each complete calendar year available (1998-2005) relative to the composite annual mean; and 3) the individual monthly chlorophyll anomalies were calculated for each month (September 1997 – December 2005, n=100) relative to the long-term monthly means.

Interpolated maps illustrating standardized chlorophyll anomalies in the Black Sea were created for each month using the inverse distance weighting (IDW) method of interpolation on a 9 x 9 km data grid (Caruso and Quarta, 1998) thereby creating a continuous distribution of chlorophyll concentration across the Black Sea. IDW assumes that interpolated points are more influenced by nearby data than data that is further away

(Caruso and Quarta, 1998), and like all geostatistical methods, assumes that spatial structures are stable in time for the duration of the sampling period.

The Black Sea includes both Case I (open ocean) and Case II (optically complex coastal) waters (IOCCG, 2000). In Case II waters, chlorophyll is difficult to distinguish from particulate matter and/or yellow substances (dissolved organic matter) and so global chlorophyll algorithms (such as OC4-v4) are less reliable (IOCCG, 2000). SeaWiFS has also been found to overestimate chlorophyll concentrations by a factor of 4 in the Black Sea (Oguz and Ediger, 2006). Nevertheless, the observation of near-coastal chlorophyll with remote sensing satellites has been found to provide important information on potential relationships with climate and nutrient enrichment in the Black Sea (Barale *et al.*, 2002; Nezlin, 2001).

Pearson correlation analysis was used to calculate 'system memory' effects in each region of the Black Sea (see 'Results' section for more information). Each monthly chlorophyll anomaly was correlated with the anomalies of each of the previous 4 months. Two methods were used to assess changes in monthly anomalies of each variable. First, the relatively simple and well known cumulative sums method was applied in order to summarize major changes by smoothing high frequency variability and highlighting changes in local mean values along the time-series. Successive positive anomalies produce an increasing slope, while successive negative anomalies produce a decreasing slope. The occurrence of shifts was then confirmed and their significance evaluated using an automatic sequential algorithm, which detects regime shifts by accounting for statistically significant differences between the means in consecutive segments of a time-series (Rodionov, 2004). This method is based on a regime shift index (RSI) combined with a sequential t test (Rodionov, 2004). Absolute value of RSI indicates magnitude of shift while its sign indicates change in direction of mean between regimes. More information on the RSI may be found in Rodionov (2004; 2007). An 18 month minimum regime length was chosen in order to increase the likelihood of selecting real shifts as opposed to small variations in the mean; however, any shift detected must be considered in context of the relatively short (100 months) length of the time-series evaluated here.

4.3 Results and discussion

Changes in climate affect productivity in the Black Sea through changes in temperature, wind patterns, and riverine inputs, but the spatial variation of impact, extent of change to the pelagic system and the exact mechanisms through which change will occur are not fully understood. The open and shelf regions of the Black Sea are distinct systems, whose productivity is regulated by different drivers. Production in the shelf system is linked to both freshwater inflow and climatic processes (Bodeanu, 2002; Bodeanu *et al.*, 2004) while the open Black Sea is predominantly influenced by climatic forces, primarily wind and temperature, aspects of which regulate stratification, upwelling, and pattern and magnitude of circulation (Mikaelyan, 1995; Sorokin, 2002). Inter- and intra-annual chlorophyll concentrations throughout the Black Sea are patchy, with an obvious difference between the chlorophyll-rich shelf and comparatively less productive open regions (Fig. 4.2, 4.3, A1-A19 in Appendix) (Sorokin, 2002).

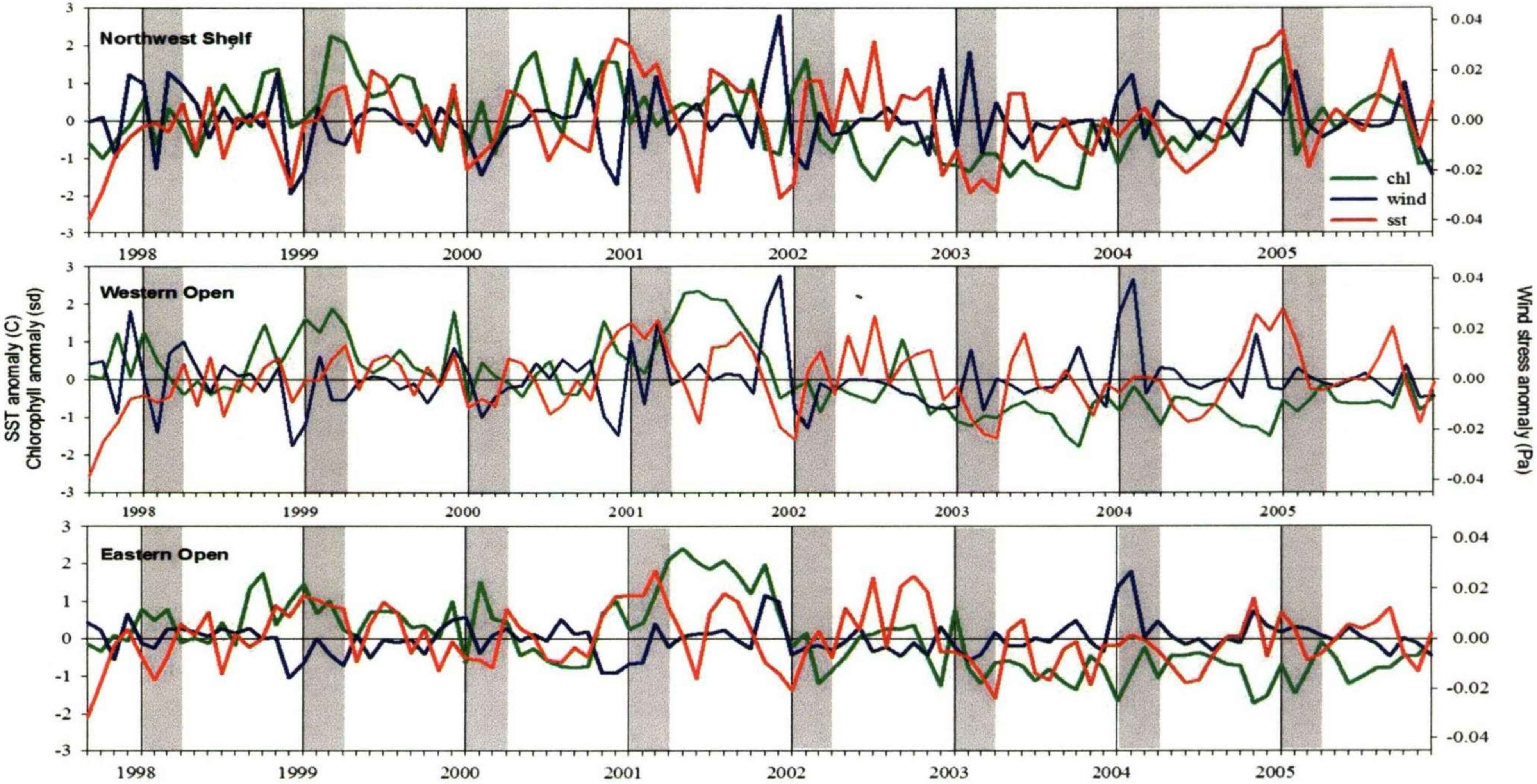
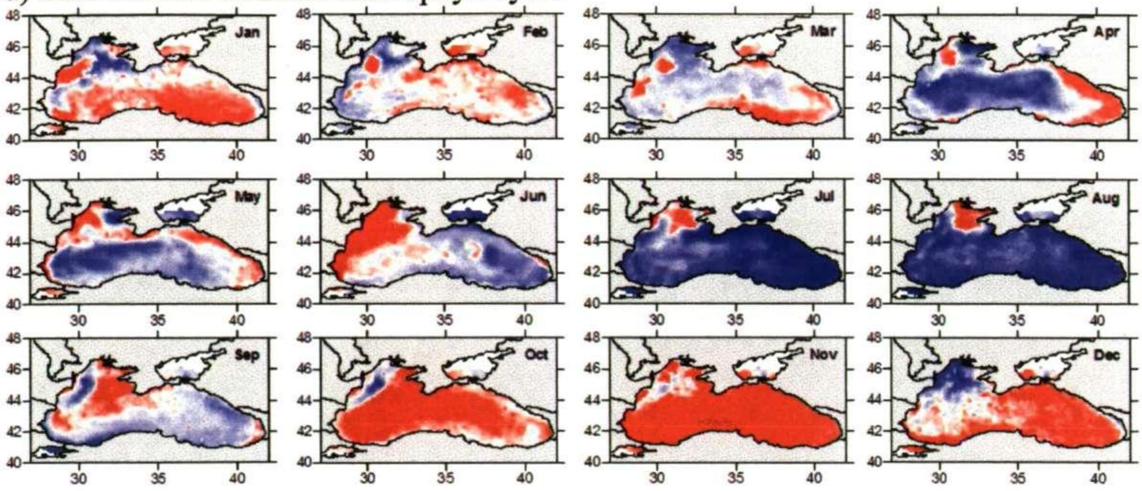


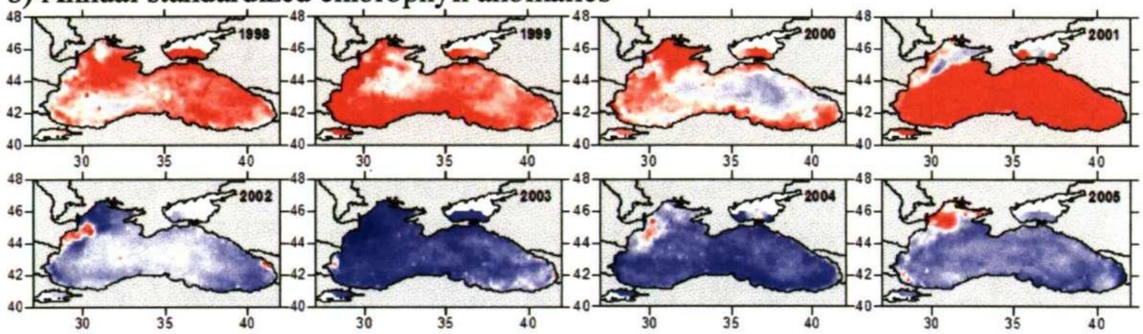
Figure 4.2. Time-series of chlorophyll (standardized) and climatic anomalies in the open and Northwest Shelf of the Black Sea. Shaded areas highlight season of CIL formation and nutrient subduction.

The chlorophyll seasonal cycle can be used to provide a baseline of 'typical' relative chlorophyll conditions in the Black Sea throughout the year (Fig. 4.3a). It is immediately clear that the seasonal cycle of chlorophyll is not spatially uniform across the Black Sea. The open Black Sea experiences its chlorophyll maximum during autumn and winter with minimum levels found during the summer months (as first described by Vinogradov *et al.*, 1999). The bloom begins near the shelf slope in the northwestern region during September and progresses eastward across the open Black Sea, covering the entire basin during October and November. The blooming cycle in the open Black Sea is primarily a response to the erosion of the seasonal pyconocline in autumn which replenishes the photic zone with nutrients from the mixed layer (Vinogradov *et al.*, 1999). The bloom ends as stratification occurs, nutrients are depleted and grazer biomass increases (Vinogradov *et al.*, 1999). The eastern Black Sea is the last region in which the phytoplankton bloom disperses in early spring. Unlike the open Black Sea, the seasonal cycle of the Northwest Shelf undergoes two blooms, an annual pattern typical of temperate waters (Sorokin, 2002). The spring bloom is dependent upon Danube River flow, and commences during April and May, the months of maximum Danube discharge, when nutrient-rich shelf waters are sufficiently warm for phytoplankton growth (Fig. 4.3a) (Cociasu and Popa, 2004; Yunev *et al.*, 2007). The decline of the spring bloom on the shelf is most likely a combination of nutrient depletion and increased zooplankton grazing pressure (Chu *et al.*, 2005). A late summer/autumn bloom also occurs in shelf waters as zooplankton grazing pressure is reduced, but is not as intense as the spring bloom. The chlorophyll minimum occurs during autumn and winter in the shelf region, when Danube discharge is lowest and shelf waters well-mixed and cool.

a) Standardized seasonal chlorophyll cycle



b) Annual standardized chlorophyll anomalies



c) Standardized chlorophyll anomalies for 2001

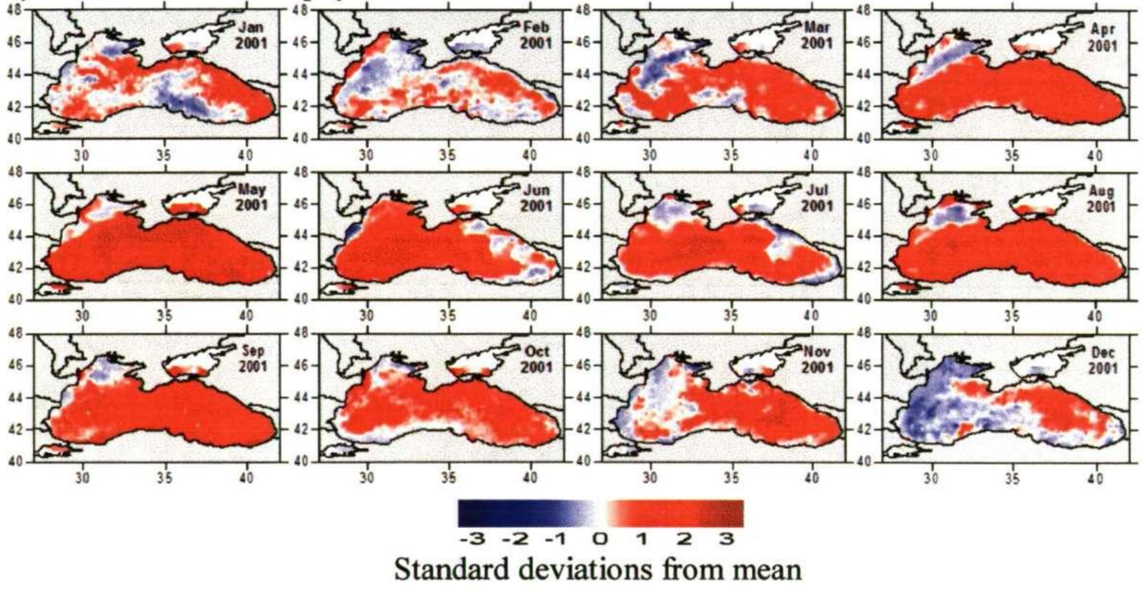


Figure 4.3. Standardized anomaly maps illustrating (a) the mean intra-annual seasonal chlorophyll cycle, standardized across the year; (b) interannual chlorophyll, standardized across all years; and (c) a detail of 2001, an exceptionally productive year in the Black Sea.

Differences between the three regions may be partially described by variability in regional 'system memory', or the persistence of a bloom-sustaining state from one month to the next (Table 4.1). Such conditions are reflected in chlorophyll anomalies; *i.e.*, a positive chlorophyll anomaly indicates that the conditions conducive to phytoplankton growth are also present. The relationship (quantified as a correlation) between chlorophyll anomalies in consecutive months provides an indication of the time it takes for environmental conditions (*e.g.*, nutrients, stratification, horizontal exchanges, etc) to change in a particular region. The system memory is defined as a length of time over which the correlation remains above a certain level (in this case above r^2=0.50). Thus, strength of system memory is a function of the rate of change of flux of chlorophyll into and out of a region (through large scale circulation or mesoscale eddies) as well as local conditions (*e.g.*, weather conditions, nutrients, local mixing) which regulate phytoplankton growth and death. Strength of system memory is variable between regions; in other words, the temporal pattern at which bloom conditions develop and the length of time for which they are sustained is not consistent across the whole Black Sea. Memory is longest in the western open Black Sea, with a correlation of r^2>0.50 at a 4 month lag period. System memory is considerably shorter in the eastern open Black Sea and Northwest Shelf with memory in both regions decreasing rapidly after only one month. The shallow shelf is a physically dynamic system that is strongly influenced by sharp variations in riverine input as well as short term weather changes and active hydrodynamic processes (Zatsepin *et al.*, 2003), resulting in limited system memory in that region. The memory of the eastern open Black Sea is shorter than that of the open western Black Sea. The reasons for the difference between system memory in the two open regions are not entirely clear but may be attributed to regional differences in gyres, eddies, and variability in the rim current (Enriquez *et al.*, 2005; Zatsepin *et al.*, 2003). This variability in system memory indicates that the western open Black Sea appears to be the most stable region, with bloom events, and the underlying hydrodynamic conditions that enable them, likely to persist for longer in that region than in either the shelf or eastern open regions. Further investigation is needed to determine the relative importance of chlorophyll flux and local processes to the system memory of each region.

Table 4.1: Correlation matrix illustrating variability in chlorophyll system memory length between Black Sea regions. Italicized values denote significant relationships ($p<0.05$).

	Northwest Shelf							Western Open Black Sea							Eastern Open Black Sea					
	month 0	month 1	month 2	month 3	month 4	month 5		month 0	month 1	month 2	month 3	month 4	month 5		month 0	month 1	month 2	month 3	month 4	month 5
month 0		*0.56*	*0.35*	*0.38*	*0.38*	*0.37*			*0.75*	*0.67*	*0.60*	*0.52*	*0.44*			*0.56*	*0.29*	*0.27*	*0.21*	0.15
month 1	*0.56*		*0.55*	*0.35*	*0.38*	*0.39*		*0.75*		*0.75*	*0.67*	*0.60*	*0.52*		*0.56*		*0.55*	*0.29*	*0.27*	*0.21*
month 2	*0.35*	*0.55*		*0.55*	*0.35*	*0.39*		*0.67*	*0.75*		*0.76*	*0.67*	*0.60*		*0.29*	*0.55*		*0.56*	*0.28*	*0.26*
month 3	*0.38*	*0.35*	*0.55*		*0.55*	*0.35*		*0.60*	*0.67*	*0.76*		*0.76*	*0.67*		*0.27*	*0.29*	*0.56*		*0.56*	*0.28*
month 4	*0.38*	*0.38*	*0.35*	*0.55*		*0.55*		*0.52*	*0.60*	*0.67*	*0.76*		*0.77*		*0.21*	*0.27*	*0.28*	*0.56*		*0.56*
month 5	*0.37*	*0.39*	*0.39*	*0.35*	*0.55*			*0.44*	*0.52*	*0.60*	*0.67*	*0.77*			0.15	*0.21*	*0.26*	*0.28*	*0.56*	

Due to the differences in the shelf and open water systems, the response of the Black Sea to changes in climate is not spatially homogeneous. A clear example of the decoupled nature of the two systems occurred in 2001 when an extensive bloom encompassed most of the open Black Sea from March through November, with chlorophyll levels reaching > 2 sd above the long-term mean (Fig. 4.2, 4.3b, 4.3c, A1, A6). The cause of the 2001 bloom was almost certainly climate-related. The winter of 2000-2001 was exceptionally warm with very low wind stress in the central Black Sea. The warm, stable winter resulted in stratification throughout the season and constrained the formation of the Cold Intermediate Layer (CIL), an intrusion of cold water between the pycnocline and thermocline at approximate 50-150 m depth (Oguz and Ediger, 2006). The CIL normally traps nutrients below surface waters, locking them out of the photic zone, until mixing energy caused by winter storms or upwelling returns them to the photic zone (Yunev *et al.*, 2005). However, in the absence of subduction, nutrients may have remained in the surface waters, available for uptake by plankton.

The bloom of 2001 was decoupled from conditions in shelf waters. The shelf, though quantitatively richer in chlorophyll than the open Black Sea, contained anomalously low levels of chlorophyll during most of the year. Phytoplankton biomass on the Northwest Shelf is intimately related to riverine outflow, and Danube discharge is correlated with chlorophyll (r^2=0.30, p=0.015, n=64). Danube outflow was uncommonly low during the winter of 2000-01, causing the front between high chlorophyll Danube-influenced and low chlorophyll open sea waters to be close to the coastline (the boundary is clearly observable in Figure 3.3c, particularly during February and April). During winter, nutrient-rich waters are usually subducted and during cold winters, the contribution of nutrient-rich water from the northwestern continental slope and Northwest Shelf may constitute 60% of the CIL water mass (Stanev *et al.*, 2003). Since this process did not occur in 2001, the nutrient rich waters stayed on the surface of the shelf where they were later mixed into the open Black Sea. The intrusion of low chlorophyll waters from the open Black Sea and the out-mixing of the nutrient rich shelf waters resulted in anomalously low chlorophyll levels in the shelf region during most of 2001. Concurrently, close inshore, where blooms were still fed by river discharge, high temperatures and low wind stress encouraged stratification and resulted in severe hypoxia in bottom waters (Kondratiev and Lemeshko, 2003).

The warm, stratified conditions occurring throughout the winter of 2000-2001 lengthened the phytoplankton growing season and caused alterations to phytoplankton community composition across the Black Sea. Dinoflagellates, which are well-suited to stratified conditions (Margalef, 1978), occurred in very high numbers (91% of biomass) in both open (Soydemir *et al.*, 2003) and shelf (Bodeanu *et al.*, 2004) waters. Although the chlorophyll concentration in the shelf region was relatively low during most of 2001, nine algal blooms occurred during summer, with 13 species reaching bloom concentrations, conditions similar to those seen in shelf waters during the eutrophic 1980s (Bodeanu *et al.*, 2004). Additionally, there was no coccolithophore bloom in the open Black Sea during the summer of 2001 (Soydemir *et al.*, 2003). This shift in community composition was indirectly visible through remotely-sensed chlorophyll, which shows a distinct spring bloom in the open sea in addition to the usual high chlorophyll level that occurs during late autumn (Fig. 4.4). The double bloom structure was previously common in the Black Sea during periods of non-diatom dominance in both open and shelf waters and is visible in Coastal Zone Color Scanner data from the late 1970s and early 1980s (Bodeanu *et al.*, 2004; Oguz *et al.*, 2003).

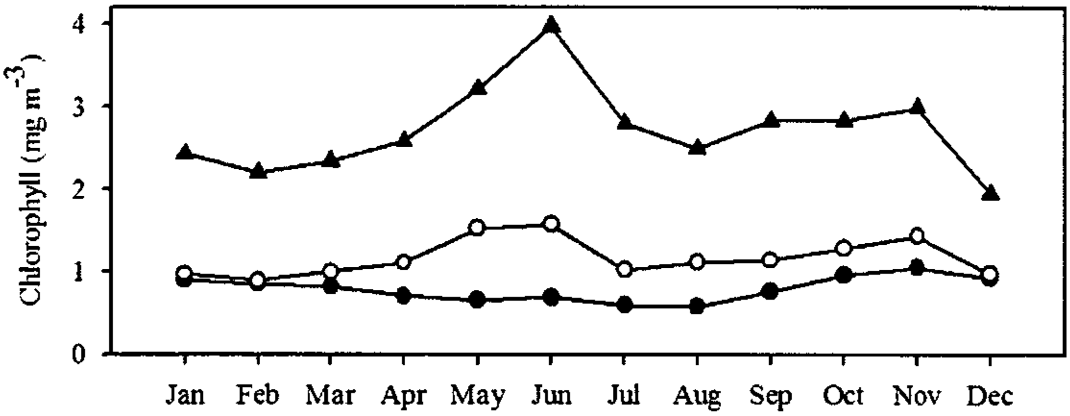


Figure 4.4. Seasonal chlorophyll cycles for the Northwest Shelf 1997-2005 (▲), the open Black Sea during 2001 (○), and the open Black Sea excluding 2001 (●). The 2001 seasonal cycle for the open Black Sea resembles that of the Northwest Shelf with a spring and an autumn bloom.

Changes in the phytoplankton community such as those observed during 2001 may have profound consequences for higher trophic levels. It is already known that warming seas are detrimental to diatoms due to increased stratification and consequent nutrient depletion of surface waters (Bopp *et al.*, 2005). A shift to a non-diatom dominated phytoplankton community may result in an increased number of 'trophic dead-ends'

(Verity and Smetacek, 1996). This was the case during the hot summer of 2001 when blooms of jellyfish and *Noctiluca scintillans* occurred in shelf waters (Velikova and Mihneva, 2005). Abundance of *Noctiluca*, a heterotrophic dinoflagellate used as an indicator of water quality in the Black Sea, had previously been found to be decreasing (daNUbs, 2005) and its resurgence in 2001 has been linked to the unusual climatic conditions (Velikova and Mihneva, 2005). Additionally, a hypoxic event on the shelf resulting in mass fish mortalities took place in 2001 (Boicenco, personal communication; Kondratiev and Lemeshko, 2003). High numbers of algal blooms, hypoxic events, faunal mortalities, and a non-diatom dominated phytoplankton community are all conditions reminiscent of those regularly found in the Black Sea during the period of peak eutrophication in the 1970s and 1980s. Such trophic changes and their related consequences may become increasingly common if they are also a symptom of a warming climate, and could have serious impacts for higher trophic levels, including commercially important fish species.

The response of phytoplankton production to warming SST is geographically variable on a global scale: increases in chlorophyll have been observed in temperate seas as rising SST extends the growing season and prolonged periods of stratification reduce light limitation (*e.g.* North Sea (McQuatters-Gollop *et al.*, 2007)), but, conversely, warming SST has resulted in decreased productivity in much of the tropical to mid-latitude World Ocean as stratification prevents nutrient upwelling (Behrenfeld *et al.*, 2006). In general, the second scenario is true of the open Black Sea; apart from 2001, the open Black Sea experienced a statistically significant decrease in annual mean (western open: r^2=0.75, p=0.012; eastern open: r^2=0.89, p=0.001; 2001 excluded) and winter (November-March, r^2=0.49-0.73, p<0.05) chlorophyll concentrations between 1998 and 2005. During all years except 2001, the open Black Sea also underwent a bloom cycle similar to that of the nearby oligotrophic Mediterranean (Bricaud *et al.*, 2002) with an autumn chlorophyll peak and summer minimum, but no spring bloom. However, the winter of 2000-01 was exceptionally warm and still and the water remained stratified throughout autumn and winter thereby extending the growing season. That year the Black Sea responded as a temperate system with a double bloom pattern characteristic of mid-latitude oceans. The chlorophyll anomalies observed after 2001 were nearly the opposite of those observed before, with most of the Black Sea comparatively low in chlorophyll (Fig. 4.3b, see also Fig. A1).

It is now clear that the role played by winter weather in the production of phytoplankton biomass is non-linear in the Black Sea. It has previously been believed that windy, cold winters lead to bigger spring/summer blooms due to enhanced vertical mixing and stronger upwelling and that the effects of eutrophication are magnified during years with severe winters (such as during the late 1980s and early 1990s) and depressed during warm winters (Oguz, 2005; Oguz and Gilbert, 2007). The trend toward milder winters in recent years has also been cited as the reason for the disappearance of the spring bloom in the open Black Sea (Oguz, 2005). Yet the warmest winter of the last 50 years resulted in an extensive bloom encompassing nearly the entire Black Sea as well as the return to the previously-observed double bloom structure. Furthermore, the winter of 2003 was comparatively windy and cool, which, according to the previously postulated relationship between winter weather and phytoplankton production, should have resulted in higher chlorophyll, yet 2003 had the lowest chlorophyll anomaly of the time-series (Fig. 4.2, 4.3b, see also Fig. A1).

The non-linear relationship between winter weather and phytoplankton biomass may be explained by the degree of subduction of water below the photic zone, most noticeably in the CIL. During years when subduction occurs, nutrient rich water is stored below the photic zone where nutrients remain unavailable throughout spring and summer; however, the disappearance or severe erosion of the CIL in 2001 kept nutrients available in surface waters, resulting in high levels of chlorophyll. The degree of subduction of nutrient rich water depends on the regeneration of the CIL, which in turn depends on the severity of winter weather, particularly from January through March when CIL replenishment is at its strongest (Sorokin, 2002). For formation of the CIL to be prevented it may be necessary for SST to be warmer than some 'threshold' temperature during those key months; in other words, the warming of the open Black Sea may indeed result in decreased chlorophyll, but only if winter temperature remains below a certain threshold level and nutrients are subducted and stored below the photic zone. The gradual erosion of the CIL due to warmer winters has already been documented (Oguz *et al.*, 2003) and the lack of its formation as observed during 2001 could become a regular feature as warm winters become more common.

The anomalous climatic conditions that occurred during 2001 may have triggered a shift in the Black Sea chlorophyll regime. A distinct switch to a predominantly negative chlorophyll anomaly post-2001 is clearly observable in the open Black Sea and, to a lesser extent, on the shelf (Fig. 4.2, 4.3b). A clear downward trend in the chlorophyll anomaly beginning in January 2002 was observed for the open Black Sea and in March 2002 for the Northwest Shelf (Fig. 4.5). A second, positive trend also began in shelf waters in October 2004. These three shifts are all statistically significant (open region: January 2002 (RSI: -0.67; $p<0.01$); shelf region: March 2002 (RSI: -0.88; $p<0.01$), October 2004 (RSI: 0.66; $p<0.01$)) (Fig. 4.5). None of these shifts corresponded with a shift in wind stress but the October 2004 shift in the Northwest Shelf region coincided with a shift in SST (RSI: 0.10, $p<0.05$). The lack of direct relationship between climate and chlorophyll is a factor of the synergistic and interactive impacts of climate on phytoplankton biomass. Thresholds and non-linearities make it difficult to model the climate-chlorophyll relationship and accurately predict the consequences of changes in climate.

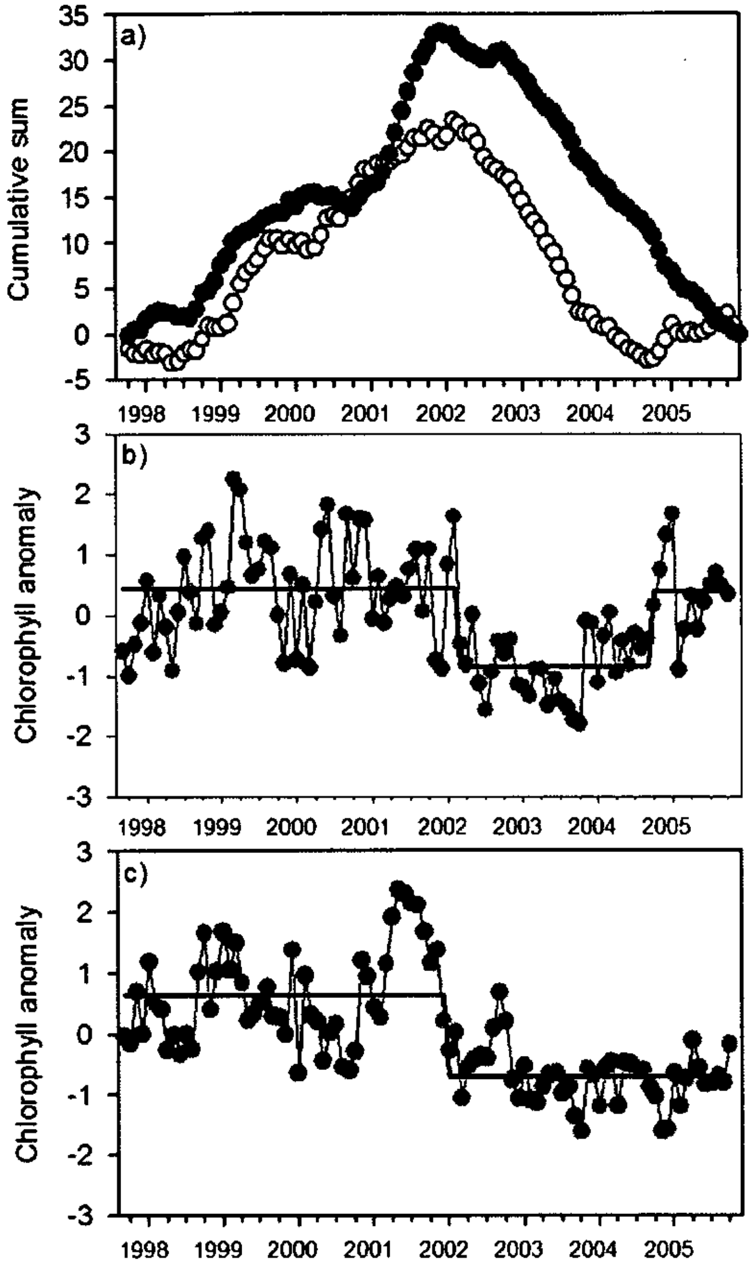


Figure 4.5. Shifts in chlorophyll anomaly revealed by (a) cumulative sums method and (b, c) automatic sequential algorithm. A major shift in the Black Sea's chlorophyll regime occurred after 2001 (a; Northwest Shelf (o), Open Black Sea (•)). Statistically significant shifts were identified in the shelf region (b) in March 2002 and October 2004 and in the open region (c; west and east combined for brevity) in January 2002.

Due to the limited length of the SeaWiFS dataset, it is difficult to assess the full significance of these shifts on the Black Sea ecosystem. As more data, particularly those from other ecological time-series, become available, an increasingly thorough examination of the importance of the detected shifts can be made. If the shifts in chlorophyll coincide with changes in other trophic levels, they may be part of an ecological regime shift, a stepwise alteration in the composition and productivity of the whole ecosystem at a regional scale that reflects major hydrographic change (Beaugrand, 2004c). Alternately, the shifts described here could be caused by natural variability or might be part of an oscillatory cycle. Regardless of their cause, these changes in the chlorophyll regime cannot be underestimated. Non-linear responses in biological communities have been found to amplify subtle environmental changes; in other words, environmental shifts may be detectable in the phytoplankton before they are detectable in the environmental variables themselves (Taylor *et al.*, 2002).

The relationship between climate and chlorophyll in the Black Sea is complex; however it is highly likely that climate has played a significant role in the recovery of the Black Sea. A decreasing trend in chlorophyll and primary production has been observed world wide as SST increases (Behrenfeld *et al.*, 2006), suggesting that reduced phytoplankton biomass in the Black Sea ecosystem is not solely a factor of diminished nutrient loading. The Black Sea nutrient regime is also dependent on degree of nutrient subduction, a function of CIL formation and winter temperature; however, the relationship between chlorophyll and these aspects of climate appears to be non-linear, with a possible temperature threshold constraining subduction. A further non-linearity was evidenced in a shift in the chlorophyll regime which coincided with the anomalous climatic conditions occurring in 2001. Although the significance of the chlorophyll shift has yet to be determined, it may be a precursor of further ecosystem change. These unpredictable responses to climatic variability emphasize the uncertain future the Black Sea faces as our climate changes.

Chapter 5

Spatial patterns of diatom and dinoflagellate seasonal cycles in the North – East Atlantic Ocean

Within the phytoplankton community, diatoms and dinoflagellates have diverse roles, different spatial patterns and contrasting trophic value: diatoms are the foundation of the copepod-fish food web while dinoflagellates appear less valuable. Changes in relative abundance of these two phytoplankton groups have been linked to pressures such as climate change and eutrophication. Spatially comprehensive data on the seasonal distribution of diatoms and dinoflagellates in non-coastal waters is limited; thus, information concerning their distribution in the open ocean is particularly useful. Here spatial and temporal patterns of diatom and dinoflagellate seasonal cycles in the coastal and open North-East Atlantic Ocean based on >100,000 Continuous Plankton Recorder (CPR) samples are presented. This analysis is presented in the form of monthly composite maps of the spatial distribution of diatoms and dinoflagellates and their relative abundances from 1958 – 2003, whose monthly time scale allows the biogeographical. exploration of seasonal cycle patterns for each group. The diatom bloom peaks first during May, with a smaller peak in late summer, while dinoflagellate abundance reaches its peak in August. Spatially, the blooms of both groups begin in the North Sea and spread outward across the North-East Atlantic region. Throughout the year, dinoflagellates and diatoms are generally most abundant in the central and southern North Sea, while the minimum abundances of both groups occur to the south of Iceland. These spatially detailed seasonal data are not yet available from remote sensing sources and may be used for the validation of current models and research as well as coastal and resource management.

Aspects of this chapter are included in the following:

McQuatters-Gollop, A., D.E. Raitsos, M. Edwards, and M.J. Attrill. (2007). Spatial patterns of diatom and dinoflagellate seasonal cycles in the North – East Atlantic Ocean. *Marine Ecology Progress Series* **339:** 301-306.

Chapter 5: Spatial patterns of diatom and dinoflagellate seasonal cycles in the North – East Atlantic Ocean

5.1 Introduction

Variations in phytoplankton community composition impact the pelagic ecosystem through changes in trophic transfer efficiency (Nagata *et al.*, 1996), food web structure and nutritional content provided to higher trophic levels (Pedersen *et al.*, 1999). Diatoms and dinoflagellates play unique roles in ecosystem processes. Within the phytoplankton, diatoms form the base of the copepod-fish food web while the smaller, unpalatable, and less nutritious dinoflagellates are often considered trophic dead ends and can result in food webs culminating in non-fodder gelatinous organisms to the detriment of fish (Verity and Smetacek, 1996). Thus the spatial distributions of diatoms and dinoflagellates throughout the year may influence the seasonal distributions of higher trophic organisms. Additionally, such shifts in phytoplankton community composition have been attributed to eutrophication (Micheli, 1999) and climate change (Edwards *et al.*, 2006; Richardson and Schoeman, 2004). In order for these links to be confirmed and for these environmental changes to be assessed and monitored, some indication of 'typical' community composition throughout the year is first needed; however, for most of the North-East Atlantic, particularly open waters, a spatially detailed description of the seasonal patterns of diatom and dinoflagellate abundances is unavailable.

Despite the significant roles of phytoplankton functional groups in our world oceans, information about their distribution and comparative seasonal cycles in the open oceanic areas is limited due to the lack of comprehensive datasets (Edwards *et al.*, 2001b). The Continuous Plankton Recorder (CPR) survey, an upper layer plankton monitoring programme, is unique in providing comprehensive data on the spatial and temporal distributions of diatoms and dinoflagellates in the North-East Atlantic Ocean and North Sea. Because it is not yet possible to differentiate diatoms and dinoflagellates based on current remote sensing algorithms, the CPR survey is able to provide large scale information not available through remote sensing (Sathyendranath *et al.*, 2004).

Recently, Beaugrand et al (2004a) published a comprehensive CPR plankton atlas detailing the abundance and presence of various phytoplankton species in the North-East Atlantic. Whilst the atlas is an invaluable reference illustrating the spatial distribution of individual phytoplankton taxa, it does not explore the distribution of diatoms and dinoflagellates at the functional group scale. However, temporal variation of major phytoplankton groups across large areas of the North-East Atlantic has recently been investigated (Leterme *et al.*, 2005; Leterme *et al.*, 2006), but this was done at a large scale and seasonal cycles were not examined.

Here spatial and temporal patterns of diatom and dinoflagellate seasonal cycles in the coastal and open North-East Atlantic Ocean based on >100,000 CPR samples are presented. This analysis is presented in the form of monthly composite maps of the spatial distribution of both major taxonomic groups from 1958 – 2003, as well as maps of their relative abundances throughout the calendar year. These are the first maps of this kind to be developed in the North-East Atlantic and the monthly time scale allows the biogeographical exploration of seasonal cycle patterns for each group.

5.2 Methods and materials

The CPR has been operating in the North-East Atlantic and North Sea since 1931 (figure 1 in Edwards *et al.*, 2001b) and measures the abundance of approximately 200 phytoplankton taxa (Warner and Hays, 1994). Full details of the operation of the CPR have been published extensively elsewhere (e.g. Batten *et al.*, 2003a; Warner and Hays, 1994) but are summarized here. The CPR collects samples using a high-speed plankton recorder that is towed behind 'ships of opportunity' in the surface layer of the ocean (~10 m depth). Water passes through the recorder, and plankton are filtered by a slow moving silk (mesh size of 270 µm). A second layer of silk covers the first and both are reeled into a tank containing 4% formaldehyde. Because phytoplankton cells are recorded as either present or absent across 20 microscopic fields spanning each section of silk, CPR phytoplankton abundance is a semiquantitative estimate (*i.e.* the species is recorded once per field independent of the number of cells in a field). However, the proportion of cells captured by the silk reflects the major changes in abundance, distribution, and community composition of the phytoplankton (Robinson, 1970), and is consistent and comparable over time. For more information on CPR sampling methods see Batten *et al.* (2003a). The collection and analysis of CPR samples have been carried out using a consistent methodological approach since 1958, making the CPR survey the longest continuous dataset of its kind in the world (Edwards and Richardson, 2004).

The total diatom and dinoflagellate abundances were calculated for each CPR sample; because unarmoured dinoflagellates are often destroyed by the CPR collection process, only armoured flagellates were considered in the group's total abundance. Abundance data were $\log_{10}(x+1)$ transformed to ensure the data approximate a normal distribution. Monthly composite maps of diatoms and dinoflagellates were produced using the kriging method of interpolation on a 1°×1° data grid (Caruso and Quarta, 1998) thereby creating a continuous distribution of phytoplankton group abundances. Kriging, like all geostatistical methods, assumes that spatial structures are stable in time, at least for the duration of the sampling period (Simard *et al.*, 1992). However, this cannot be assumed of CPR data due to its 45 year sampling period. This problem has been resolved by portioning the data into shorter temporal periods and treating each temporally-partitioned dataset individually (Edwards, 2000). Thus, the period of study was refined into nine shorter time periods of generally five years each (1958-1962, 1963-1967, 1968-1972, 1973-1977, 1978-1982, 1983-1987, 1988-1992, 1993-1997, and 1998-2003). Next, twelve monthly-averaged maps were created for each of the five-year intervals using the kriging method described above. For each calendar month, all nine of the five-year maps of that month were then averaged at each grid node and kriging was used to create monthly composite maps showing the mean spatial distribution for each month over the entire study period. This process was performed separately for diatoms and dinoflagellates. Each monthly composite map is made up of approximately 8,000 *in situ* CPR samples taken between 1958 and 2003. In order to explore phytoplankton community composition, the monthly composite maps of diatoms and dinoflagellates were then used to calculate the relative abundance (measured as percent diatoms) for each calendar month.

5.3 Results

To explore the seasonal patterns of diatoms and dinoflagellates, the monthly means of both datasets were plotted (Fig. 5.1). The seasonal cycles show dissimilar bloom patterns. The diatom spring bloom peaks during May, abundance then gradually declines through summer before a weaker peak occurs in late summer. Dinoflagellates bloom during late summer, peaking in August, then progressively decline throughout autumn. Both groups maintain only a minimum abundance during winter months. The 95% confidence interval calculated for each month is very small, indicating that the measured abundances are consistent (Zar, 1984). The spatial patterns of abundance during the seasonal cycle and the relative percent composition of the functional groups were further examined using the monthly abundance and monthly relative community composition (Fig. 5.2) composite maps.

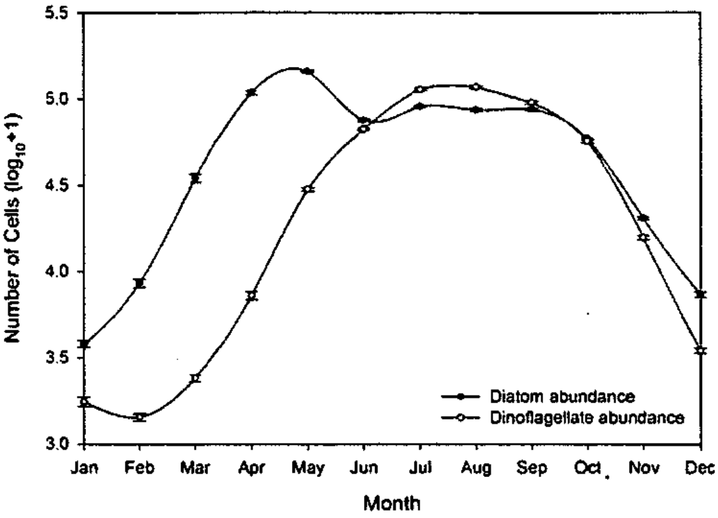
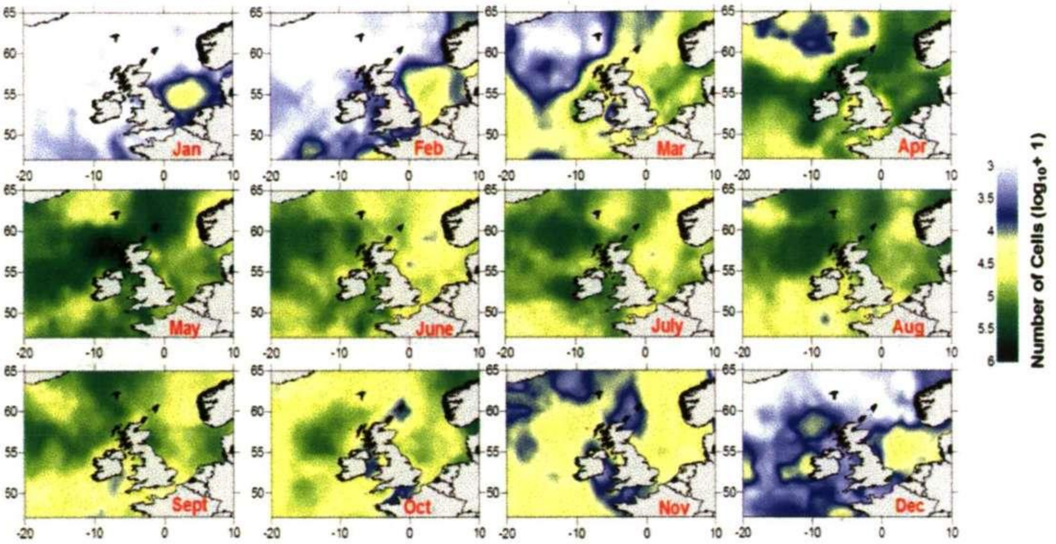


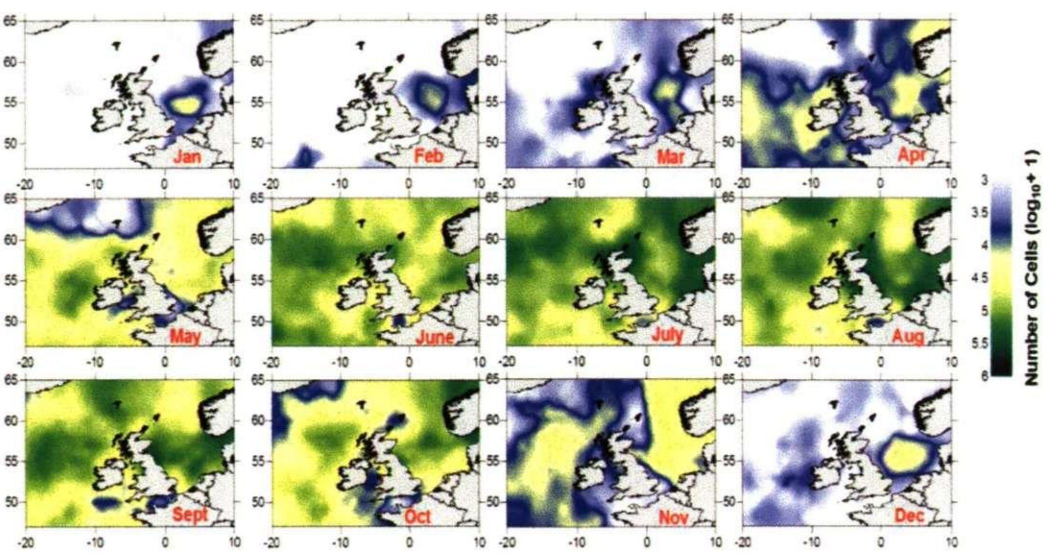
Figure 5.1. Averaged seasonal cycles of diatoms and dinoflagellates between 1958 and 2003 in the North-East Atlantic and North Sea (ninety-five percent confidence intervals are indicated but are very small). Diatoms bloom strongly in spring and more weakly in late summer while dinoflagellates reach maximum abundance in late summer.

Across the survey time period, the spring diatom bloom, defined here as the annual rapid increase in phytoplankton growth and abundance, commences in the shallow areas of the North Sea in March (Fig. 5.2a, Fig. 5.3). Its timing is predominantly controlled by light penetration, the amount of light available in the euphotic zone being determined by day length, strength of solar radiation, cloudiness (most notably in winter), degree of mixing and amount of suspended matter in the water column (Edwards, 2000). Therefore, in March the shallow areas of the North Sea are the first to show an increase in diatom abundance as light penetration increases. The diatom bloom also began in March in the Skagerrak, but here as a result of haline stratification due to Baltic inflow rather than because of shallow depth and low suspended matter (Richardson and Christoffersen, 1991).

a) Spatial distribution of diatoms



b) Spatial distribution of dinoflagellates



c) Relative abundance of diatoms and dinoflagellates

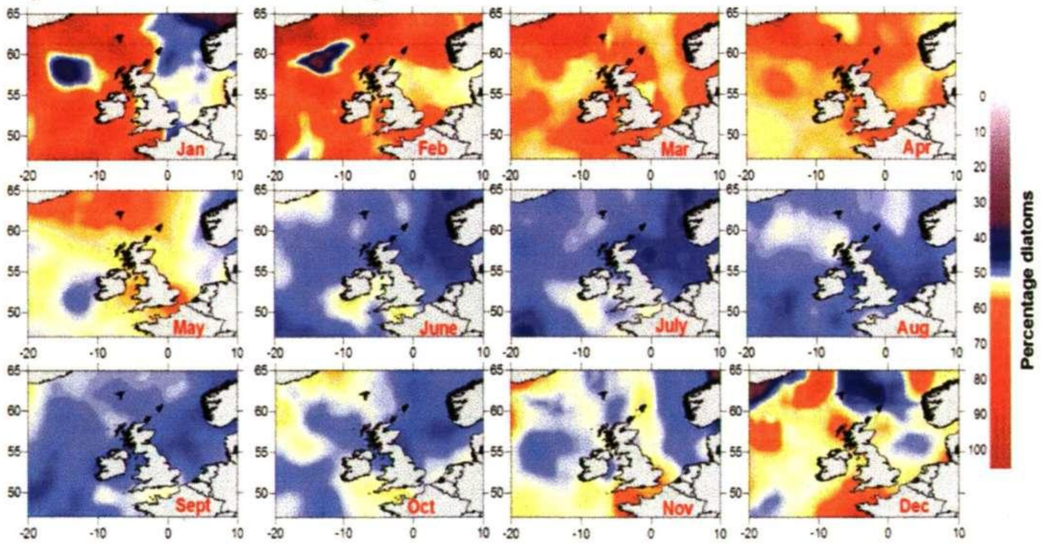


Figure 5.2. The mean monthly spatial patterns of (a) diatoms, (b) dinoflagellates, and (c) their relative community abundances (measured in percentage diatoms) in the North-East Atlantic during the period 1958 – 2003.

During April, diatom abundance increases outwards from shallow to deeper waters, encompassing nearly the entire main shelf area; the diatom bloom was most intense in the North Sea and the shelf area west of Scotland. At this time, diatom abundance to the south of Iceland remained somewhat lower, probably due to the colder water and deep vertical mixing in that region which delays the onset of stratification, and consequently the spring bloom, at higher latitudes (Edwards, 2000).

By May, the diatom bloom was at its peak, covering nearly the entire North-East Atlantic, although a reduction in abundance in the Skagerrak and Norwegian Trench indicated the end of the diatom spring bloom in those areas. During May, the dinoflagellate bloom began in the Skagerrak, the Rockall Trough and the Porcupine Bank (Fig. 5.2b). In late spring and summer, the phytoplankton community composition across most of the North-East Atlantic consists of a greater number of dinoflagellates than diatoms (Fig. 5.2c). Because the water column is stratified, phytoplankton deplete upper layer nutrients and diatoms (opportunists) give way to flagellates (competitors) and eventually dinoflagellates (stress-tolerators), both of which are adapted to survival in low nutrient, stable waters (Holligan, 1987; Margalef, 1978).

During June, the spring diatom bloom gradually continued to recede across the region, while the dinoflagellate bloom spread throughout most of the North-East Atlantic, except for the coastal area around the southern United Kingdom and Ireland, which remains well mixed and unstratified due to tidal turbulence (Pingree and Griffiths, 1978). Diatom abundance again increased across the entire region in July, and by August was especially high in the Skagerrak, the German Bight, and the waters north of Norway as well as those to the northwest of Ireland. The dinoflagellate bloom peaked during July and August, with particularly high abundances persisting in the productive areas of the Skagerrak, German Bight and the Shetland and Orkney Island areas of Scotland. After August, the blooms of both taxonomic groups began to decrease in abundance, starting in the Celtic Sea and continuing northwards throughout the autumn months.

During autumn, cooler water temperatures and increased storms cause the thermocline to start to erode, allowing the mixing of cold, nutrient rich bottom waters with warm, nutrient depleted surface waters (Holligan, 1987). Thus, as stratification decreased, autumn diatom and dinoflagellate blooms occurred in parts of the North Sea, Norwegian

Trench and deeper shelf waters. During September, the diatom bloom continued to gradually recede in the Celtic Sea, Irish Sea, the Norwegian Trench and south of Iceland, although it remained strongest around the Faroe Islands and Rockall Trough. The dinoflagellate bloom also still covered much of the North-East Atlantic, though abundance continued to decline when compared to previous months, especially around the British Isles and to the south of Iceland.

By October, diatom and dinoflagellates only remained abundant in the Rockall Trough, shallow areas of the North Sea, the Porcupine Bank and north of Norway. Strengthening wind and wave action result in increased turbidity, which along with decreasing solar radiation cause the abundance of both phytoplankton groups to decline during November. Because North-East Atlantic waters are light limited in winter, and remain cool, turbulent, and well-mixed throughout the winter months (OSPAR, 2000) the abundance of both phytoplankton groups decreases throughout winter, with diatoms reaching a minimum in January and dinoflagellates reaching a minimum in February. However, even during the winter, the abundance of both groups was highest in the shallow areas of the Dogger Bank and southern areas of the North Sea. These areas are the first to bloom in March when the seasonal cycle begins again.

Like the monthly abundances of diatoms and dinoflagellates, the relative community composition of each functional group (measured as percent diatoms) is patchy across both space and time (Fig. 5.2c). However, some definite seasonal patterns can be observed. During winter and spring diatoms are generally more abundant than dinoflagellates across the North-East Atlantic region, while the phytoplankton community is richer in dinoflagellates than diatoms in summer and autumn. The North Sea, Norwegian Trench, Skagerrak and Rockall regions show the lowest percentage of diatoms during summer months (< 45%), while relative diatom abundance remains higher (~ 55%) throughout the summer in the English Channel, the Celtic Sea and to the south and east of Iceland. During winter months, the phytoplankton in most of the North-East Atlantic may consist of up to 90% diatoms; however, this is not the case with the North Sea where phytoplankton is rarely composed of more than 70% diatoms. Hydrographic explanations for the variations in diatom and dinoflagellate abundance have been provided above, but additional research is required to rule out influence by anthropogenic factors, particularly in coastal areas.

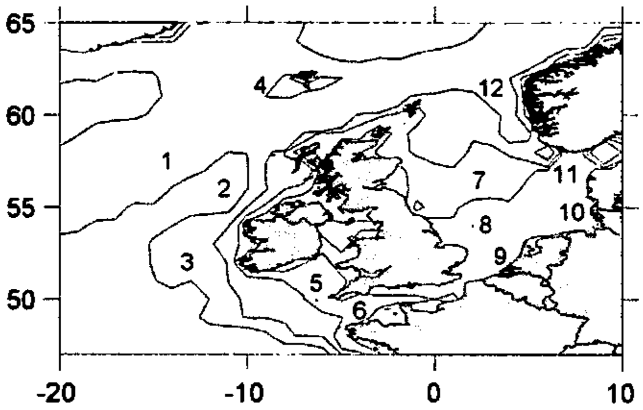


Figure 5.3. Bathymetric map with detailed location names of the North-East Atlantic and North Sea regions mentioned in the text: 1) Rockall Plateau, 2) Rockall Trough, 3) Porcupine Bank, 4) Faroe Islands, 5) Irish Sea, 6) English Channel, 7) North Sea, 8) Dogger Bank, 9) Southern Bight, 10) German Bight, 11) Skagerrak, 12) Norwegian Trench. Contour lines denote 50 m, 100 m, 200 m, and 2000 m of depth.

5.4 Discussion

The analysis presented here provides a description of phytoplankton seasonal cycles across the North-East Atlantic and North Sea and their relationships to the hydrographical characteristics of the region based on > 50 years of data. These maps contribute information intended to help us better understand how the spatial distribution of phytoplankton groups can influence ecosystem structure and functioning as well as to illustrate the patchy nature of phytoplankton distribution in the North-East Atlantic.

In summary, the two phytoplankton groups display dissimilar bloom patterns with the diatom spring bloom peaking during May before gradually declining through mid-summer and then weakly peaking again in late summer. Dinoflagellates bloom most intensely during late summer, peaking in August, before progressively declining throughout autumn. Both groups maintain only a minimum abundance during winter months. Spatially, dinoflagellates and diatoms both exhibit highest abundances in the central and southern North Sea, while dinoflagellates also display particularly high abundance in the Skagerrak and diatoms are highly abundant in waters to the west of Scotland. The minimum abundances of both groups occur to the south of Iceland. These general patterns provide the framework for more detailed exploration of spatial and temporal trends in smaller geographic areas and across more refined time periods in order to explore finer scale changes. Additional knowledge regarding trends in long-term spatial and temporal variation in phytoplankton functional group distribution can provide insights into the effects of climate change and anthropogenic pressures on primary productivity in the North-East Atlantic.

Chapter 6

Separating the long-term influences of climate variability and eutrophication on North-East Atlantic phytoplankton communities

Alterations in phytoplankton community composition have been associated with both hydroclimatic change and anthropogenic eutrophication. However, because of the similar (or synergistic) effects of these drivers, it is difficult to ascertain if changes observed in North-East Atlantic phytoplankton are responses to increased nutrients, natural variability, or a combination of both factors. Because comprehensive long-term nutrient data are lacking, phytoplankton dynamics cannot be related to nutrient concentrations on a large spatio-temporal scale and an alternative method is needed to separate the eutrophication and climate signals. Changes due to climatic variability are likely to be observed in both open and coastal phytoplankton communities while changes resulting from increased nutrients will primarily affect coastal areas. A comparison between open sea and coastal phytoplankton dynamics reveals similar patterns of change in diatom and dinoflagellate abundance in open and coastal regions of the North-East Atlantic. These results indicate that the same factors drive abundance of phytoplankton populations in both regions, suggesting that changes in diatom and dinoflagellate abundances in the North-East Atlantic are primarily a result of climatic variability and that eutrophication remains a localized problem in the North-East Atlantic, rather than a large scale driver of phytoplankton community dynamics.

Aspects of this chapter are included in the following:

McQuatters-Gollop, A., D.E Raitsos, M.J. Attrill, and M. Edwards (submitted). Separating the long-term influences of climate and eutrophication on North-East Atlantic phytoplankton communities. *Marine Ecology Progress Series.*

Chapter 6: Separating the long-term influences of climate variability and eutrophication on North-East Atlantic phytoplankton communities

6.1 Introduction

Phytoplankton community composition is a sensitive indicator widely used to monitor environmental change in European coastal regions (Bodeanu, 2002; Hickel, 1998; Marasovic *et al.*, 2005; Philippart *et al.*, 2000). Because of the artificial supplementation of N and P without the addition of Si, eutrophic conditions tend to favour dinoflagellates rather than the silicon-limited diatoms, causing a shift in community composition (Officer and Ryther, 1980a; Smayda, 1990; Turner, 2002). In addition to functional group abundances, the relative community composition of diatoms and dinoflagellates (measured as the ratio of diatoms to dinoflagellates) may be used as an indicator of eutrophication; the abundance of diatoms is likely to decrease in eutrophic waters while the relative abundance of dinoflagellates increases (Smayda, 1990; Sommer *et al.*, 2002). Such a situation has been observed, for example, in the eutrophic Black Sea (Zaitsev, 1992) and regions of the Northern Adriatic (Marasovic *et al.*, 2005). These changes in phytoplankton community composition have, however, also been linked to hydro-climatic change making it difficult to clearly identify the eutrophication signal (Bopp *et al.*, 2005; Edwards and Richardson, 2004; Leterme *et al.*, 2005; Leterme *et al.*, 2006; Richardson and Schoeman, 2004).

Some areas of the North-East Atlantic are commonly considered to be impacted by eutrophication (Allen *et al.*, 1998; Dederen, 1992; Fock, 2003; Hickel *et al.*, 1993), although, because of the similar (or even synergistic) effects of climate change and eutrophication, it is difficult to ascertain if changes observed in North-East Atlantic phytoplankton are responses to increased nutrients, natural variability, or a combination of both factors (Cloern, 2001; Niemi *et al.*, 2004). The Continuous Plankton Recorder (CPR), although limited in collecting small and unarmoured flagellates, provides a spatially extensive dataset with which to observe long-term changes in diatom and dinoflagellate dynamics. However, because comprehensive long-term nutrient data are not available for much of the North-East Atlantic, relating biological and nutrient time-series is not feasible on a broad spatial scale. Therefore, an alternative method of eutrophication assessment is needed. Because regions further from the coast are less likely to be impacted by land-based nutrient inputs, phytoplankton composition data from open sea areas may be compared to that of near-shore areas. Changes in diatom and dinoflagellate abundances occurring due to climatic variability are likely to be observed in both open and coastal waters while changes resulting from increased nutrients will mostly affect coastal areas. Comparison of long-term trends in phytoplankton composition over a large spatial area, including both coastal regions affected by anthropogenic eutrophication and open water areas with little human impact, may therefore assist in separating the signals of eutrophication from those of natural ecosystem variability and global climate change (Edwards *et al.*, 2001b; Hickel *et al.*, 1993). Similar methods have been applied to the North Sea (McQuatters-Gollop *et al.*, 2007), the Adriatic (Marasovic *et al.*, 2005) and the Aegean (Tsirtsis and Karydis, 1998).

The objectives of this research are: (1) to explore and compare changes in diatom and dinoflagellate abundance in coastal and open sea regions of the North-East Atlantic; (2) to assess the relationship between diatom and dinoflagellate abundance; and (3) to determine if phytoplankton dynamics in coastal areas are the result of climatic variability or anthropogenically-induced change.

6.2 Methods and materials

6.2.1 Study area

The North-East Atlantic Ocean is increasingly impacted by anthropogenic activities including fishing, eutrophication, shipping activity and habitat destruction. These impacts are much more severe in coastal areas than in the open sea. For the purpose of this study, the North-East Atlantic was originally separated into 'coastal' (<50 nm (93 km) from shore) and 'open' (>125 nm (231 km) from shore) regions, which were further divided into eight coastal and four open subregions. GIS was used to select the corresponding data for each subregion. Analysis of variance (ANOVA) was performed on the subregional diatom and dinoflagellate time-series and significantly different subregions were identified. Adjoining subregions whose diatom and dinoflagellate time-series were not significantly different were combined, resulting in a final study area consisting of three open areas (Southern North-East Atlantic, Northern & Central North-East Atlantic, Open North Sea) and five coastal areas (Coastal North Sea, English Channel, Shelf Sea, Irish Sea, Coastal Iceland). However, the Irish Sea and coast of Iceland were later excluded due to insufficient phytoplankton abundance data (Fig. 6.1), leaving three sub areas in each region.

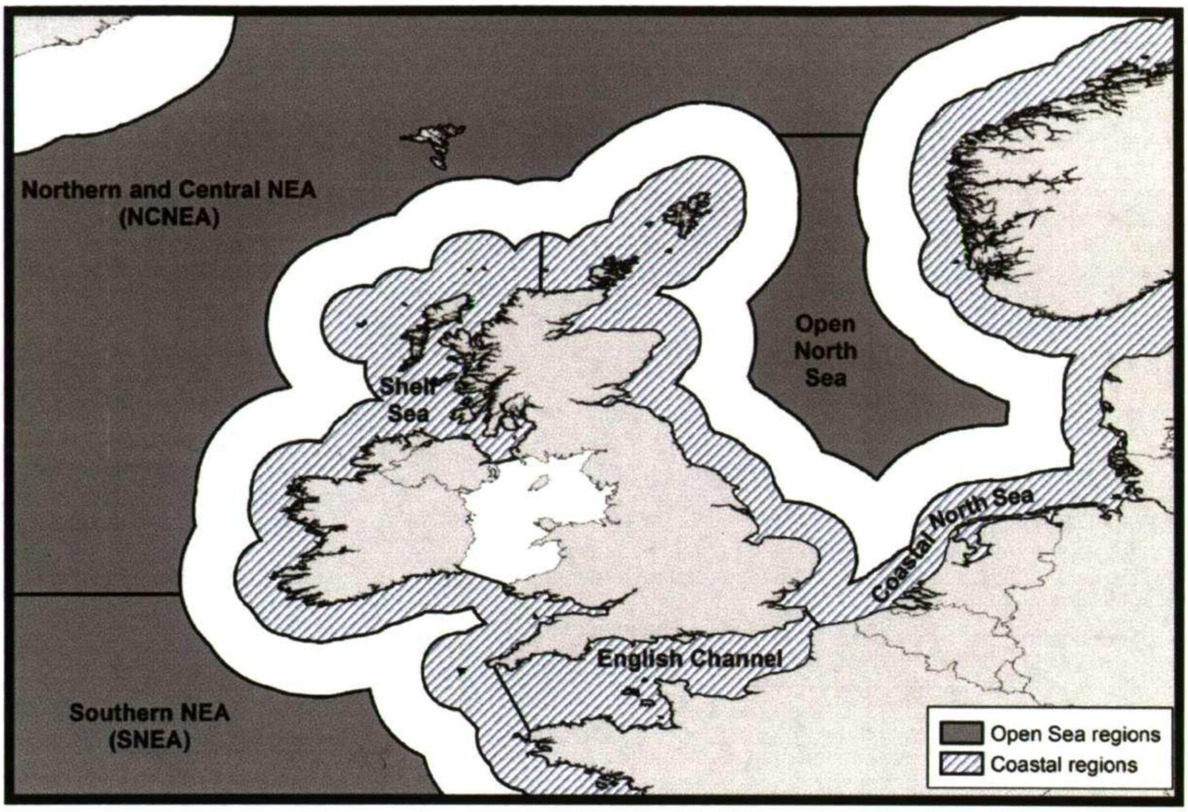


Figure 6.1. Map of the study area and subregions used in the analysis. Gray shaded areas indicate open sea regions (>125 nm from land) and hatched areas indicate coastal regions (<50 nm from land).

6.2.2 Data

Phytoplankton abundance data were obtained from the Continuous Plankton Recorder (CPR) database. The CPR is towed behind ships of opportunity in the surface layer of the ocean (~10 m depth). Plankton are filtered onto a 270 μm silk. The collection and analysis of CPR samples have been carried out using a consistent methodological approach since 1958, making the CPR survey the longest continuous dataset of its kind in the world (Edwards and Richardson, 2004). The CPR survey measures the abundance of approximately 200 phytoplankton taxa (and species) and has been used to describe the seasonal and long-term variations of phytoplankton abundance in the North Atlantic since 1958 (Warner and Hays, 1994). Because phytoplankton cells are recorded as either present or absent across 20 microscopic fields spanning each section of silk, CPR phytoplankton abundance is a semiquantitative estimate (*i.e.* the species is recorded once per field independent of the number of cells in a field). The total diatom and dinoflagellate abundances were calculated for each CPR sample in the study area; because unarmoured dinoflagellates are often destroyed by the CPR collection process, only armoured flagellates were considered in the group's total abundance. Additionally, due to its large mesh size the CPR underestimates small flagellate taxa and primarily counts large dinoflagellates such as *Ceratium* spp. (Dickson *et al.*, 1992). However, the proportion of cells captured by the silk reflects the major changes in abundance, distribution, and community composition of the phytoplankton (Robinson, 1970), and, most importantly, is consistent and comparable over time. For more information on CPR sampling methods see Batten *et al.* (2003a).

A total of 69,324 samples fell within the areas of study (32,135 of these fell within open sea areas, the remainder within designated coastal areas) during the 1958 – 2003 period. Only years with 8 or more months of data available (per subregion) were included in the analysis. For mapping and time-series analysis, abundance data were $\log_{10}(x+1)$ transformed to ensure the data approximated a normal distribution. The Southern North-East Atlantic area has data for 39 years, the English Channel for 42 years and all other subregions have 46 years of data. The white areas visible in Figure 5.2 signify areas with too few samples for effective interpolation to occur; the waters in the centre of the study area are the most highly sampled with sampling frequency declining toward the far outer limits of the study area, particularly the northern region.

6.2.3 Analysis

Decadal composite maps of diatoms and dinoflagellates were produced using the inverse distance weighting (IDW) method of interpolation on a 1°×1° data grid (Caruso and Quarta, 1998) thereby creating a continuous distribution of phytoplankton group abundances; IDW assumes that interpolated points are more influenced by nearby data than data that are further away (Caruso and Quarta, 1998). IDW, like all geostatistical methods, assumes that spatial structures are stable in time for the duration of the sampling period (Simard *et al.*, 1992). However, this cannot be assumed of CPR data due to its 46 year sampling period. This problem has been resolved by portioning the data into shorter temporal periods and treating each temporally-partitioned dataset individually (Edwards, 2000). Thus, the period of study was refined into 4 decadal and one half-decadal time periods (1958-1962, 1963-1972, 1973-1982, 1983-1992, and 1993-2003). Next, twelve monthly-averaged maps were created for each of the decadal intervals using the interpolation method described above; these twelve maps were then averaged into one composite map per time period. This process was performed separately for diatoms and dinoflagellates. The subregional time-series of abundance indicates comparable spatial and temporal patterns to those illustrated in the decadal maps. This method of cross validation confirms the accuracy of the interpolation methods employed to create the decadal maps.

Simple correlation analysis (Pearson) and linear regression were used to determine the existence and strength of possible relationships between annual abundances of diatoms and dinoflagellates within and between regions. Abundances were formally compared using a 3-factor ANOVA design (1: open/coastal region [fixed], 2: subregion [nested, random], 3: decade [orthogonal, fixed]) and analysed with the programme 'GMAV 5' (Underwood and Chapman, 1998). Cochran's C-test was used to test for homogeneity of variances, and in cases of significance ($p < 0.05$), data were $\log_e(x+1)$ transformed. Where significant ($p < 0.05$) differences between open and coastal regions, subregions, or time were identified, the Student-Newman-Keuls (SNK) post hoc test was used to determine which pairs of levels differed significantly.

6.3 Results

ANOVA revealed no statistically significant differences in diatom or dinoflagellate abundance or relative community composition between open and coastal regions (Table 6.1). Furthermore, areas of high phytoplankton abundance are not confined to near shore waters but exist in both open and coastal regions (Fig. 6.2). Although some significant differences between subregions were identified, no single subregion displayed changes that were clearly different from those in other subregions and which would indicate that eutrophication was driving phytoplankton dynamics. Decadal differences in dinoflagellate abundance and the diatom:dinoflagellate ratio were also identified; however, due to the non-normal distribution of the diatom:dinoflagellate ratio, these differences must be considered tentative.

Table 6.1. 3-factor ANOVA showing differences between diatom and dinoflagellate abundances and relative abundances between open and coastal regions, subregions and over time. Diatom data are natural log (x+1) transformed. Although relative abundance data are not normally distributed, transformation of a ratio is not appropriate; therefore differences in the diatom:dinoflagellate ratio over time must be considered tentative. Values denoted by ** are significant at the $p < 0.01$ level.

	Source	DF	MS	F	*p*
Diatoms	Open/Coastal Region (Re)	1	0.01	0.01	0.94
	Subregion (Su)	4	1.99	10.01	<0.01**
	Decade (De)	3	1.33	7.95	<0.01**
	Re x De	3	0.12	0.69	0.57
	De x Su	12	0.17	0.84	0.61
Dinoflagellates	Open/Coastal Region (Re)	1	5055459.66	0.15	0.72
	Subregion (Su)	4	134201.57	15.18	<0.01**
	Decade (De)	3	2050759.67	2.57	0.10
	Re x De	3	4442944.05	0.35	0.79
	De x Su	12	6522039.46	1.19	0.30
Diatom: Dinoflagellate ratio	Open/Coastal Region (Re)	1	0.05	0.96	0.38
	Subregion (Su)	4	0.05	10.80	<0.01**
	Decade (De)	3	0.03	5.71	0.01**
	Re x De	3	0.01	1.96	0.17
	De x Su	12	0.01	1.08	0.38

ANOVA indicated some significant differences in phytoplankton dynamics between coastal subregions. The English Channel contained significantly fewer diatoms than the other coastal subregions (Table 6.1) but, like the coastal North Sea, showed no temporal trend in abundance (Fig. 6.3). In the Shelf Sea subregion, however, diatom abundance decreased significantly ($r^2 = 0.23$, $p < 0.001$, $n = 39$) (Fig. 6.2, 6.3). Despite these differences, diatom abundances in all coastal regions were significantly correlated, suggesting that large scale processes are driving coastal diatom dynamics (Table 6.2). Dinoflagellate abundance in the coastal North Sea was significantly different from that in the English Channel and Shelf Sea subregions (Table 6.1), with a slight but significant decrease occurring during the study period ($r^2 = 0.15$, $p < 0.01$, $n = 46$) (Fig. 6.3). The English Channel was the only coastal region demonstrating an increase in dinoflagellate abundance ($r^2 = 0.29$, $p < 0.001$, $n = 42$) (Fig. 6.4) and, unlike coastal diatoms, abundances of coastal dinoflagellate populations were not significantly correlated (Table 6.2). The relative community composition (measured by the ratio of diatoms:dinoflagellates) was significantly different between all coastal subregions (Table 6.1). A decline in the proportion of the phytoplankton community occupied by diatoms occurred in both the English Channel ($r^2 = 0.31$, $p < 0.001$, n = 42) and Shelf Sea ($r^2 = 0.09$, $p = 0.043$, n = 46) while analysis of the coastal North Sea revealed no significant trend (Fig. 6.4).

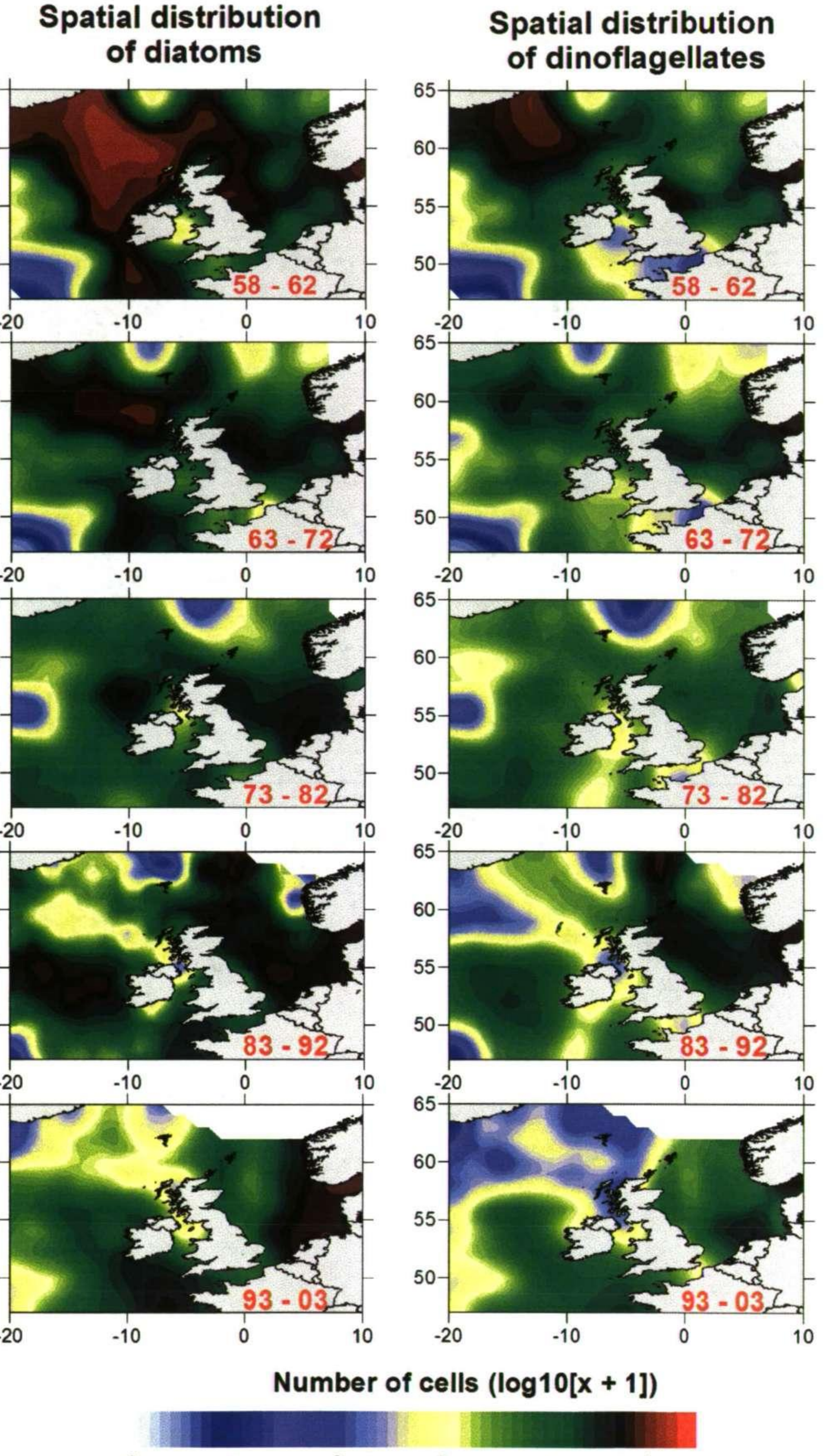


Figure 6.2. Decadal maps illustrating the spatial distribution of diatoms and dinoflagellates in the North-East Atlantic for five time periods since 1958.

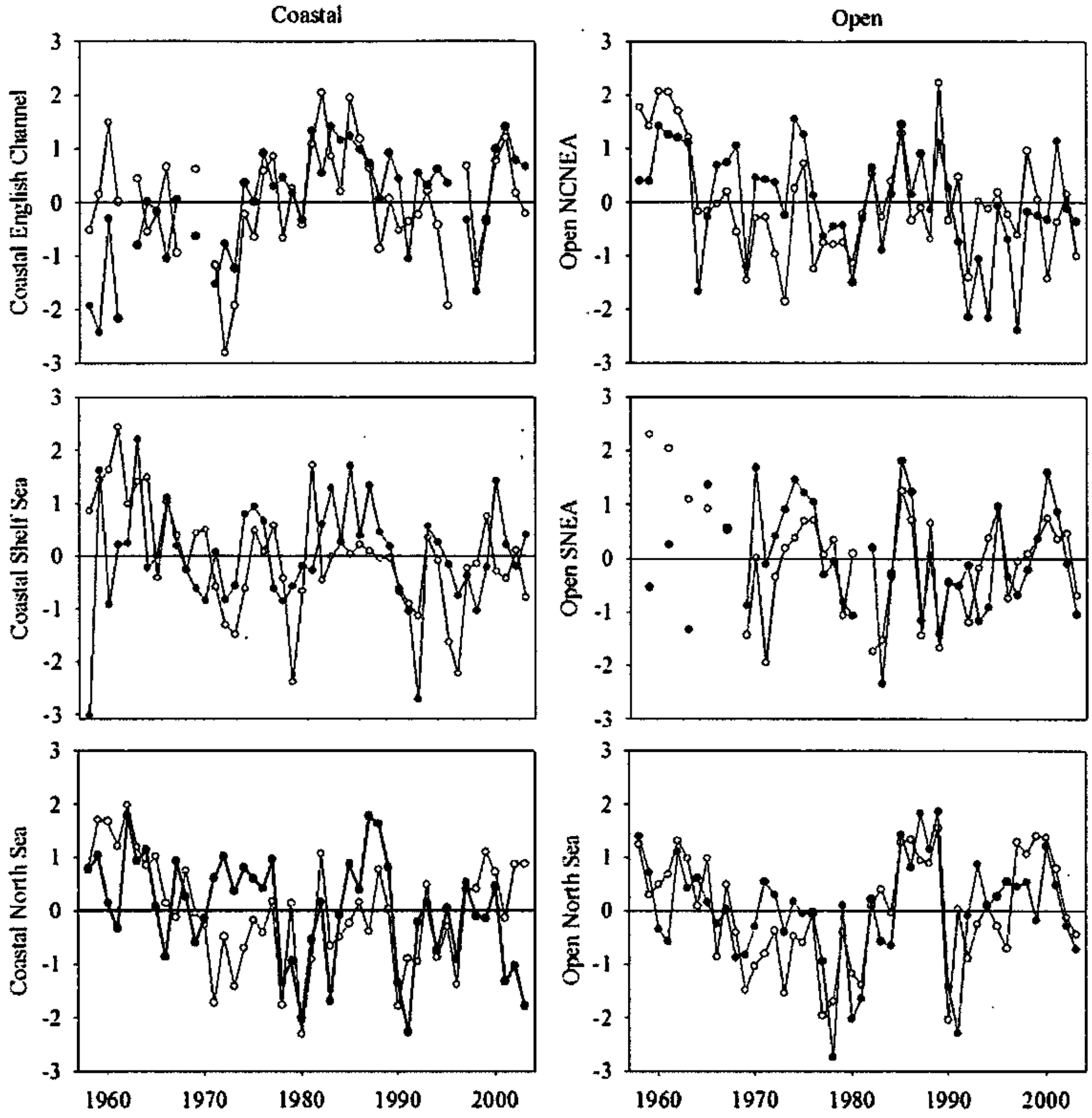


Figure 6.3. Standardized time-series of diatom and dinoflagellate abundances in the six North-East Atlantic subregions (diatoms: ○; dinoflagellates: ●)

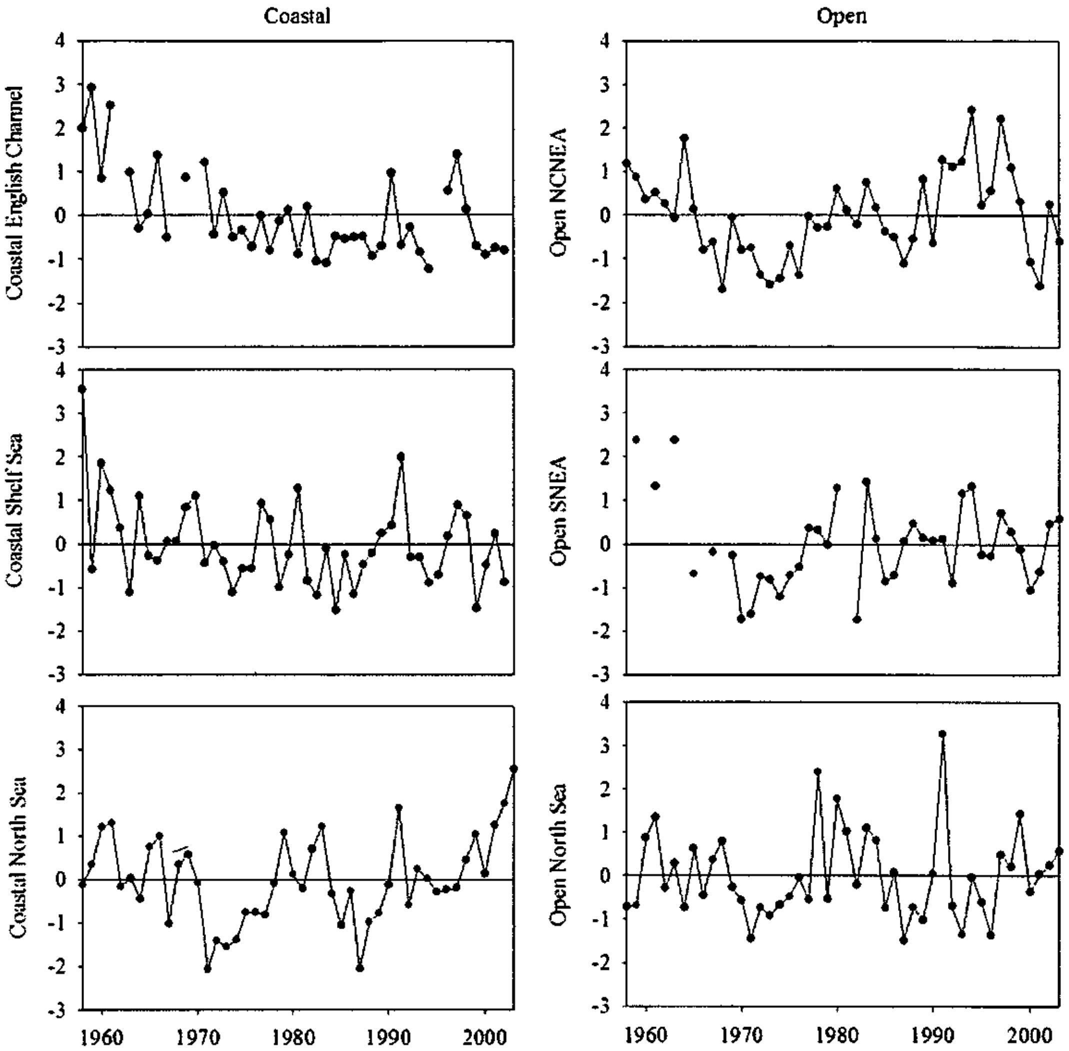


Figure 6.4. Time-series of standardized diatom:dinoflagellate ratio for each North-East Atlantic subregion.

Table 6.2. Correlations between annual diatom and dinoflagellate abundances in coastal and open North-East Atlantic regions. Values denoted by ** are significant at the $p < 0.01$ level, * at the $p < 0.05$ level.

		Diatoms						**Dinoflagellates**				
		English Channel (Coastal)	Shelf Sea (Coastal)	North Sea (Coastal)	SNEA (Open)	NCNEA (Open)	North Sea (Open)	English Channel (Coastal)	Shelf Sea (Coastal)	North Sea (Coastal)	SNEA (Open)	NCNEA (Open)
Diatoms	Shelf Sea (Coastal)	0.37*										
	North Sea (Coastal)	0.31*	0.53**									
	SNEA (Open)	-0.06	0.47**	0.38*								
	NCNEA (Open)	0.20	0.53**	0.49**	0.27							
	North Sea (Open)	0.23	0.25	0.60**	0.25	0.52**						
Dinoflagellates	English Channel (Coastal)	0.42**	-0.23	-0.21	-0.25	-0.29	0.00					
	Shelf Sea (Coastal)	0.36*	0.25	0.21	0.33*	0.13	0.29**	0.24				
	North Sea (Coastal)	-0.08	0.30*	0.42**	0.21	0.28	0.39**	-0.12	0.22			
	SNEA (Open)	-0.04	0.01	0.07	0.49**	-0.07	0.07	0.01	0.05	0.27		
	NCNEA (Open)	0.15	0.27	0.29	0.25	0.56**	0.24	-0.07	0.35*	0.36*	0.35*	
	North Sea (Open)	0.09	0.05	0.43**	0.03	0.33*	0.68**	0.01	0.28	0.70**	0.17	0.26

The only significant difference between open subregions occurred in the Southern North-East Atlantic (SNEA) which contained fewer diatoms than the remaining open subregions. The Northern and Central North-East Atlantic (NCNEA) was the only region in which either functional group showed a (slight) trend in abundance; both diatoms ($r^2 = 0.10$, $p < 0.05$, n = 46) and dinoflagellates ($r^2 = 0.14$, $p = 0.01$, n = 46) decreased, a pattern that can clearly be seen in Figure 5.2. However, all three open subregions had comparable levels of dinoflagellates and the relative community compositions were also similar, with no significant trends in relative community composition (Fig. 6.4). Annual diatom abundances in the open North Sea and NCNEA subregions were significantly correlated, as were dinoflagellate abundances in the open NCNEA and SNEA (Table 6.2).

6.4 Discussion

Eutrophication and climate change are two issues of concern in the North-East Atlantic Ocean; both are thought to impact phytoplankton composition, yet it has been difficult to determine if changes in diatom and dinoflagellate abundances are driven by increased nutrients, climatic variability, or the synergistic effects of both. Separating the effects of climate change and eutrophication is especially challenging because although long-term spatially and temporally comprehensive data exist regarding phytoplankton abundance and climate in the region, no equivalent nutrient data set is available. Thus, traditional deductive methods of directly relating changes in plankton abundance to nutrient enrichment (*i.e.* statistical analysis) are not possible and an alternative inductive method of assessment must be employed. Because anthropogenic eutrophication primarily affects coastal waters, changes observed in open sea phytoplankton abundance are most likely in response to changes in climate rather than to nutrient enhancement, while changes observed in coastal phytoplankton may be a response to either (or both) driver. Therefore, comparing long-term phytoplankton dynamics between coastal and open waters may serve to separate the signals of eutrophication and climate change.

No statistically significant differences in the abundances of diatoms or dinoflagellates or in the relative community composition of the two groups were found between coastal and open regions. Instead coastal and open regions shared many similarities in phytoplankton dynamics despite inter-subregional differences in anthropogenic pressure and therefore are likely a result of larger scale, such as hydroclimatic, processes, as opposed to anthropogenic influences. Diatom and dinoflagellate abundances in open and coastal subregions were often correlated, indicating that common factors are driving phytoplankton dynamics in both near-shore and offshore areas (Table 6.2).

Additionally, throughout the study period, open and coastal phytoplankton communities underwent near-concurrent periods and patterns of change. In the late 1970s all open and coastal subregions (apart from the English Channel and open SNEA) experienced a period of low diatom and dinoflagellate abundance that coincided with a period of exceptionally low temperature and salinity across the North-East Atlantic, as well as reduced inflow from the Atlantic Ocean to the North Sea, resulting in a region-wide low temperature/low salinity hydroclimatic anomaly (Edwards *et al.*, 2001b). This anomaly affected much of the pelagic community, including decreased zooplankton abundance and biomass, decreased phytoplankton biomass and diversity and anomalous seasonal bloom patterns, decreased meroplankton abundance, and a crash in the population of *Ceratium macroceros*, one of the most dominant dinoflagellates found in the North-East Atlantic (Edwards *et al.*, 2002; Edwards *et al.*, 2001a).

During the 1980s, diatom and dinoflagellate abundances increased throughout the study area (again, with the exception of the English Channel) and culminated in peak abundances in the late 1980s (Fig. 6.3). These changes reflect the North-East Atlantic regime shift, a stepwise alteration in the composition and productivity of an ecosystem at a regional scale that reflects major hydrographic change, which occurred during the 1980s (Beaugrand, 2004c; Reid *et al.*, 2001a). Like the hydroclimatic anomaly of the late 1970s, the regime shift manifested as a change in many biological and ecosystem processes and individual species throughout the North Atlantic including changes in phenology, resulting in trophic mismatch (Edwards and Richardson, 2004), an influx of oceanic species to the North Sea (Lindley *et al.*, 1990), increased phytoplankton biomass (Reid *et al.*, 1998) and changes in zooplankton community structure and salmon abundance (Beaugrand and Reid, 2003). Additionally, the relative abundance of diatoms decreased in the open and coastal North Sea, from 2 s.d. above the long-term mean in the early 1980s to 2 s.d. below in the late 1980s (Fig. 6.4). While these inter-subregional changes occurred at slightly different magnitudes and during slightly different times, they suggest that similar large scale drivers influence coastal and open sea diatom and dinoflagellate abundances throughout the North-East Atlantic.

Although open and coastal phytoplankton dynamics were not significantly different, some differences were observed between subregions within the coastal and open regions. The coastal English Channel in particular exhibited different abundance patterns from other coastal subregions. For example, although phytoplankton abundances were high in most of the North-East Atlantic (up to 2 s.d. above the long-term mean in some subregions) during the 1980s, abundance of both groups clearly decreased in the English Channel at that time. Additionally, the English Channel was the only region in which dinoflagellate abundance significantly rose, leading to an increase in their relative community abundance. Spatially, the English Channel is the smallest coastal subregion of this study, and therefore may be particularly susceptible to the affects of anthropogenic and climatic pressures (Hoch and Garreau, 1998).

Dinoflagellate abundance in the coastal North Sea was significantly different from that in other coastal regions, and although areas of the coastal North Sea are commonly considered to be eutrophic, abundance decreased during the period of study. However, an increase in nanoflagellate (< 20 μm) biomass has been recorded at Helgoland research station in the southern North Sea since the end of the 1970s (Hickel, 1998). Because nanoflagellates are too small to be counted by the CPR, this increase is not reflected in the abundance data presented here, but is consistent with a recorded increase in phytoplankton biomass in North Sea waters (McQuatters-Gollop *et al.*, 2007). Since eutrophication is thought to select for flagellates and other small phytoplankton groups, it is possible that the increase in nanoplankton observed at Helgoland may be a result of elevated nutrient levels (Margalef, 1978; Sellner *et al.*, 2001; Smayda and Reynolds, 2001; Zaitsev, 1992); however modelling results also suggest that the warming climate may favour smaller plankton such as dinoflagellates at the expense of diatoms (Bopp *et al.*, 2005; Edwards *et al.*, 2006).

Because of its remoteness from land, the NCNEA is the subregion least likely to be affected by anthropogenic nutrients and the decreases in diatom and dinoflagellate abundances observed there are almost certainly climatically driven. The NCNEA is the only subregion in which both diatoms and dinoflagellates decreased significantly (although slightly) during the study period (Fig. 6.2, 6.3); this decrease has previously been observed for phytoplankton colour in part of the region and may be related to a negative relationship between the North Atlantic Oscillation (a natural mode of climatic variability affecting SST and wind magnitude and direction) and phytoplankton dynamics in that area (Edwards *et al.*, 2001b). Perhaps most surprising is the fact that the open NCNEA supported the highest abundances of any subregion (Fig. 6.2). It is interesting to note that the coastal North Sea (probably the most heavily anthropogenically impacted subregion) and the open NCNEA (the least impacted subregion) both underwent decreases in dinoflagellate abundance during the period of study; this reinforces the suggestion that climate is the overriding regulator of phytoplankton dynamics throughout the North-East Atlantic.

The results of this study show no clear differences between diatom and dinoflagellate dynamics in open and coastal North-East Atlantic waters. Two typical symptoms of eutrophication, a decrease in diatom abundance and an increase in dinoflagellate abundance, were observable only in the coastal Shelf Sea and coastal English Channel subregions, respectively. Neither of these symptoms presented in the coastal North Sea, an area commonly considered eutrophic; instead the coastal North Sea experienced a decline in dinoflagellate abundance during the study period.

Overall, variations in diatom and dinoflagellate abundance in open and coastal areas of the North-East Atlantic are at times significantly correlated and display many simultaneous inter-subregional trends, indicating that the same agents generally regulate abundance of both coastal and open sea phytoplankton populations. It is unlikely that coastal and open phytoplankton would follow such similar trends in abundance and be so closely correlated if coastal diatom and dinoflagellate populations were responding to nutrient enrichment. Therefore, at this scale, the changes observed in North-East Atlantic diatom and dinoflagellate dynamics are most likely a result of climatic variability and are not directly induced through anthropogenic eutrophication. This work suggests that eutrophication remains a localized problem in the North-East Atlantic and is not a large scale driver of phytoplankton dynamics.

Chapter 7

A long-term chlorophyll dataset reveals regime shift in North Sea phytoplankton biomass unconnected to nutrient trends

During the 1980s, a rapid increase in the Phytoplankton Colour Index (PCI), a semiquantitative visual estimate of algal biomass, was observed in the North Sea as part of a region-wide regime shift. Two new datasets created from the relationship between the PCI and SeaWiFS Chlorophyll *a* (Chl *a*) quantify differences in the previous and current regimes for both the anthropogenically affected coastal North Sea and the comparatively unaffected open North Sea. The new regime maintains a 13% higher Chl *a* concentration in the open North Sea and a 21% higher concentration in coastal North Sea waters. However, the current regime has lower total nitrogen and total phosphorus concentrations than the previous regime, although the molar N:P ratio in coastal waters is now well above the Redfield ratio and continually increasing. Besides becoming warmer, North Sea waters are also becoming clearer (*i.e.*, less turbid), thereby allowing the normally light-limited coastal phytoplankton to more effectively utilize lower concentrations of nutrients. Linear regression analyses indicate that winter Secchi depth and sea surface temperature (SST) are the most important predictors of coastal Chl *a* while Atlantic inflow is the best predictor of open Chl *a*; nutrient concentrations are not a significant predictor in either model. Thus, despite decreasing nutrient concentrations, Chl *a* continues to increase, suggesting that climatic variability and water transparency may be more important than nutrient concentrations to phytoplankton production at the scale of this study.

Aspects of this chapter are included in the following:

McQuatters-Gollop, A., D. E. Raitsos, M. Edwards, Y. Pradhan, L. D. Mee, S. J. Lavender, and M. J. Attrill. 2007. A long-term chlorophyll dataset reveals regime shift in North Sea phytoplankton biomass unconnected to nutrient levels. *Limnology and Oceanography* **52:** 635-648.

McQuatters-Gollop, A., D.E. Raitsos, M. Edwards, Y. Pradhan, L.D. Mee, S.J. Lavender, and M.J. Attrill (2007). Climate exacerbates eutrophication in the North Sea. *IMBER Update* **7:** 3-4.

McQuatters-Gollop, A. (2007). The synergistic effects of climate and eutrophication in the North Sea. *GLOBEC International Newsletter* **13:** 10-11.

Langmead, O., **A. McQuatters-Gollop** and L.D. Mee (Eds.). (2007). European Lifestyles and Marine Ecosystems: Exploring challenges for managing Europe's seas. 43pp. University of Plymouth Marine Institute, Plymouth, UK.

Chapter 7: A long-term chlorophyll dataset reveals regime shift in North Sea phytoplankton biomass unconnected to nutrient trends

7.1 Introduction

Phytoplankton are the primary producers of pelagic marine waters, the base of the marine food web and thus an integral part of the ecosystem, affecting trophic dynamics, nutrient cycling, habitat condition, and fishery resources (Paerl *et al.*, 2003b). Additionally, plankton are closely coupled to environmental change (Hays *et al.*, 2005), making them sensitive indicators of environmental disturbance.

Between 1983 and 1988 a rapid change occurred in many biological and ecosystem processes and individual species in the North Atlantic region (Beaugrand, 2004c). A shift in the proportion of cold and warm water species of *Calanus* (Reid *et al.*, 2003a), phenological changes in production resulting in trophic mismatch (Edwards and Richardson, 2004), an influx of oceanic species (Lindley *et al.*, 1990) and changes in zooplankton community structure and salmon abundance (Beaugrand and Reid, 2003) occurred in the North Sea during the mid to late 1980s. Additionally, phytoplankton production demonstrated a marked increase across the North Atlantic and North Sea regions during that period (Edwards *et al.*, 2001b; Reid *et al.*, 1998). These changes, observable across multiple trophic levels, are linked to what has been described as a regime shift, a stepwise alteration in the composition and productivity of the whole ecosystem at a regional scale that reflects major hydrographic change (Beaugrand, 2004c; Reid *et al.*, 2001a).

In part, the cause of the late 1980s North Sea regime shift may have been a response to a switch in the behaviour of the winter North Atlantic Oscillation (NAO) from a negative phase to its longest ever positive phase (Alheit *et al.*, 2005; Beaugrand, 2004c; Weijerman *et al.*, 2005). A positive phase NAO influences the North Sea ecosystem through regional climate effects including increased sea surface temperature, strong westerly winds (Alheit *et al.*, 2005; Beaugrand, 2004c; Weijerman *et al.*, 2005), and increased inflow of warm, salty water from the Atlantic Ocean (Beaugrand, 2004c; Edwards *et al.*, 2001b; Reid *et al.*, 2003a).

Although oceanic algal production is an important component of the marine system, few long-term biological datasets exist for European waters. The lack of time-series data has made it difficult to identify trends in phytoplankton production dynamics and establish linkages to natural variability or anthropogenic change (Edwards *et al.*, 2001b). The Continuous Plankton Recorder (CPR) survey, an upper-layer plankton monitoring programme operated since 1931, provides the only long-term biological plankton dataset with spatial coverage across coastal and open North Sea waters (Edwards *et al.*, 2001b). The first level of analysis the CPR offers is the Phytoplankton Colour Index (PCI), a semiquantitative *in situ* measurement of phytoplankton biomass (see Methods section). The PCI was recently used in conjunction with phytoplankton Chlorophyll-a data estimated by the Sea-viewing Wide Field-of-view Sensor (SeaWiFS) satellite to extrapolate a new 50 year Chl *a* dataset for the Northeast Atlantic and North Sea (Raitsos *et al.*, 2005).

As an area rich in natural resources, much of the North Sea has been affected by anthropogenic impacts including fishing, nutrient runoff and oil, gas and aggregate extraction (Clark and Frid, 2001). Because of its proximity to land, the degree of impact on the coastal area is likely to be greater than that affecting the open North Sea. The aim of this paper is to separately extrapolate the SeaWiFS dataset back 50 years for the open and coastal North Sea in order to assess the long-term variability of Chl *a* within and between these two regions and to relate the Chl *a* time-series to environmental and climatic variation. In this paper an attempt is made to explain trends observed in phytoplankton biomass based on environmental, climatic and nutrient related factors and processes. The biological, environmental, and climatic differences between the pre- and post- regime shift North Sea system were also examined and factors generating change investigated, including nutrient concentrations, water transparency, sea surface temperature (SST), sea level pressure (SLP), precipitation, wind stress, and climatic variability.

7.2 Methods and materials

7.2.1 Area of study

The North Sea is bordered by some of the most densely populated and highly industrialized Western European countries; its catchment covers an area of 850,000 km^2 and contains 184 million people (OSPAR, 2000). The North Sea has been a productive fishing ground and is heavily exploited for oil, gas, and aggregates. Additionally, the

North Sea is extensively used for transport as well as for the dumping of dredged material. From an environmental perspective, the North Sea is an ecologically rich and diverse environment; a number of endangered species and important habitats, some of which are now protected, are found within the North Sea ecosystem (OSPAR, 2000).

For the purpose of this study, the limit of the coastal North Sea was defined as a standard 30 nm (56 km) distance from land (Fig. 7.1). In order to minimize the impact of land-based activities, the open North Sea was designated as the area greater than 125 nm (231 km) from the shore. Polygons representing the open and coastal North Sea study areas were constructed with ESRI ArcMap 9.0. These polygons were used to select the corresponding data points for each geographically referenced dataset used in this analysis (PCI, SeaWiFS, PCI – SeaWiFS match-ups, SST, sea level pressure, wind stress, precipitation, nutrient concentrations, and transparency measurements).

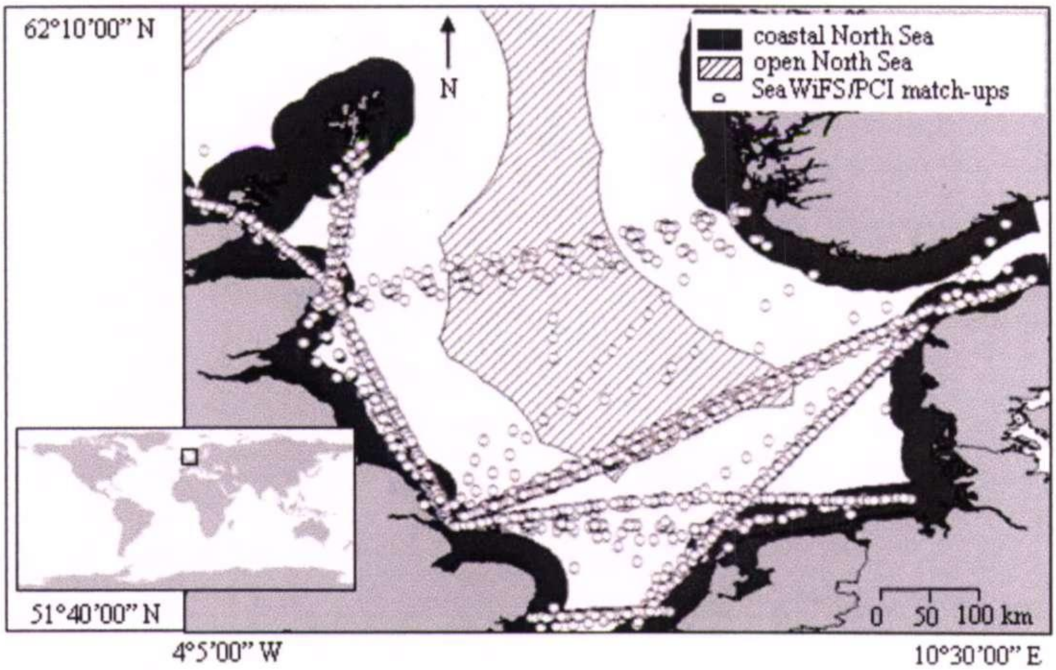


Figure 7.1. Location of coastal and open North Sea areas used in study overlain by CPR samples temporally corresponding with SeaWiFS Chlorophyll measurements (n=3,695).

7.2.2 Data extraction

7.2.2.1 Environmental data

Annual surface (top 10 m of water column) nutrient concentration data for the North Sea and Elbe and Rhine rivers were obtained from the European Environment Agency's

Waterbase (EEA) via their website (http://dataservice.eea.eu.int/dataservice/). Waterbase contains reliable and validated data collected from the national monitoring programmes of the member countries of the EEA and therefore comprehensively covers a large geographical area, a requirement which is paramount to this study. To assess variability of nutrient input, Elbe and Rhine River discharge data were obtained from the Global Runoff Data Centre, a digital world-wide repository of discharge data and associated metadata (Global Runoff Data Centre, 2005).

Coastal water transparency data were extracted from the Secchi Disk Data Collection for the North Sea and Baltic Sea (available at http://www.ices.dk/ocean/project/secchi/) compiled by Thorkild Aarup (Aarup, 2002). While the transparency dataset contains a considerable number of samples (n = 5,057) with a wide spatial and temporal resolution, they are not uniformly distributed in time or space. Temporally, the nineties were the most heavily sampled decade (n = 3,056) followed by the eighties (n = 1,188) and the seventies (n = 758); thus, the accuracy of trend in Secchi depth increases with time. The majority of samples were taken in the Dutch, German, Danish and Norwegian waters of the southern North Sea (n = 2,730) and Skagerrak (n = 2,116). The remaining samples are from the coastal areas of the UK and western Norway. Water transparency is dependent on phytoplankton biomass, dissolved particulate organic matter (POM), suspended sediments, and yellow substances in the water column (Sanden and Hakansson, 1996) as well as water column stability.

The North Atlantic Oscillation (NAO) is an important index of climatic variability affecting the North Sea and influences its ecology through SST, wind direction and magnitude, and precipitation (Ottersen *et al.*, 2001) and is strongly linked to oceanic inflow into the North Sea (Reid *et al.*, 2003a). The winter (December through March) NAO index (NAOI) was acquired from the website of Jim Hurrell at the National Center for Atmospheric Research. The NAOI is based on the difference in normalized sea level pressure between Lisbon, Portugal, and Stykkisholmur, Iceland (Hurrell, 1995). Mean annual SST data were obtained from the Hadley Centre, UK Met Office (HadISST v1.1). Monthly mean precipitable water content (as a measure of precipitation) and sea level pressure data were obtained from the NCEP/NCAR Reanalysis Project at the NOAA-CIRES Climate Diagnostics Center (http://www.cdc.noaa.gov/cdc/reanalysis/). Wind speed data, also obtained from the NCEP/NCAR Reanalysis Project, were converted into wind stress data. Wind stress is a function of wind speed, non-dimensional drag coefficient and boundary layer air density (Pond and Pickard, 1978). Wind stress regulates the dynamics of the boundary layer and is connected to the production of wind-driven surface currents, the generation of surface waves and upper-ocean mixing (Pond and Pickard, 1978). Therefore, low wind stress is associated with highly stratified waters.

Data on the influx of Atlantic waters entering the North Sea between the Orkney Islands and Utsira were obtained from the NORWECOM 3-D hydrodynamic model (Iversen *et al.*, 2002). Model generated data were used as no long-term measured time-series exist.

7.2.2.2 Primary production data

PCI data were extracted from the CPR database for the North Sea. During the period 1948-2003, the CPR survey collected approximately 52,000 samples. Although the CPR has been sampling in the North Sea since 1931, data from 1948 onwards were used as the methodology of sampling and measurement of PCI has remained consistent since 1948 (Batten *et al.*, 2003a). Samples are collected by a high-speed plankton recorder (~15-20 knots (28 – 37 km h^{-1})) that is towed behind 'ships of opportunity' in the surface layer of the ocean (~10 m depth); one sample represents 18 km of tow. Accumulation of phytoplankton cells on the silk gives it a greenish colour (Batten *et al.*, 2003b); the Phytoplankton Colour Index (PCI) is based on a relative scale of greenness and determined on the silk by reference to a standard colour chart. There are four different 'greenness' values: 0 (no greenness), 1 (very pale green), 2 (pale green) or 6.5 (green). Categories of PCI are assigned numerical values based on acetone extracts (Colebrook and Robinson, 1965). PCI is a unique measurement of phytoplankton biomass, as small phytoplankton cells that cannot be counted under the microscope contribute to the colouration of the filtering silk (Batten *et al.*, 2003b).

SeaWiFS current reprocessed version (v5.1) data produced by Ocean Biology Processing Group (OBPG) were acquired from the NASA Ocean Color website (http://oceancolor.gsfc.nasa.gov/). The data were Level 3, 8-day products (9 km x 9 km square resolution) of the near-surface Chl *a* concentration (mg m^{-3}), estimated using the ocean Chlorophyll 4 - version 4 (OC4-v4) algorithm (O'Reilly *et al.*, 1998):

$$\text{Chl } a = 10^{(0.366 - 3.067x + 1.930x^2 + 0.649x^3 - 1.532x^4)},$$

where $x=\log_{10}((R_{rs}443>R_{rs}490>R_{rs}510)/R_{rs}555)$ and R_{rs} is the satellite calculated remote sensing reflectance. The first (x) equation takes the highest reflectance value from a waveband at 443, 490, or 510 nm and divides it by the reflectance at 555 nm; reflectance maximum moves towards higher wavebands as the Chl *a* concentration increases. This x value then goes into an equation that results from a statistical fit of this empirical algorithm to a large *in situ* database.

NASA processed these data using a series of radiometric corrections (*e.g.*, atmospheric) to eliminate the presence of clouds, haze, and water vapour (Mueller and Austin, 1995). The 8-day data products were used in order to increase the number of CPR – SeaWiFS match-ups, as the daily data were highly obscured by cloud cover.

7.2.3 Data analysis

7.2.3.1 Matching SeaWiFS and PCI data

Seventy-six months (September 1997 - December 2003) of *in situ* measurements of PCI and satellite Chl *a* values were compared for the North Sea. Concurrent SeaWiFS and CPR measurements were collated for the same spatial and temporal coverage. Then, refining the technique developed and used on a larger scale by Raitsos *et al.* (2005), coastal and open North Sea samples were selected based on the areas shown in Fig 6.1. The finer geographical scale helped us to establish more accurate relationships between the PCI and SeaWiFS Chl *a* for the open and coastal North Sea study areas. In the North Sea, the CPR survey collected 6,294 different samples for the 6-year period; 2,311 of which fell within 30 nm (56 km) of the coast and 723 of which were located in the defined open North Sea area. After screening the SeaWiFS dataset for CPR match-ups, 1,272 samples could be used for comparison in the coastal North Sea (44.96 % of coastal data did not have a SeaWiFS match-up, primarily due to cloud coverage) and only 412 could be used in the open North Sea. As 412 is too small a sample size to establish a reliable relationship, all available non-coastal North Sea match-ups (3,695) were used to construct the relationship to be applied to the open North Sea (Fig. 7.1). PCI data are on a ratio scale (*i.e.*, not only can PCI categories be ranked but differences are quantified). Thus, Pearson correlation (or linear regression) is appropriate to assess the strength of the relationship between SeaWiFS and PCI data (Raitsos *et al.*, 2005; Zar, 1984). SeaWiFS data were log-transformed to improve homogeneity of variance and normality (Zar, 1984).

The PCI and SeaWiFS datasets demonstrate an overall increase in Chlorophyll during the study period. Highly significant positive relationships exist between PCI and SeaWiFS data for the entire North Sea ($r = 0.26$, $p < 0.001$) and for the coastal area ($r = 0.30$, $p < 0.001$). As these relationships are non-linear, the mean SeaWiFS Chl *a* value was calculated for each PCI category in each North Sea region (Fig. 7.2). Using the significant relationships between PCI and SeaWiFS Chl *a* for each region (Fig. 7.2) and the total number of CPR samples analyzed (approximately 52,000) for the period 1948 – 2003, retrospective calculations of Chl *a* for the coastal and open North Sea could be produced.

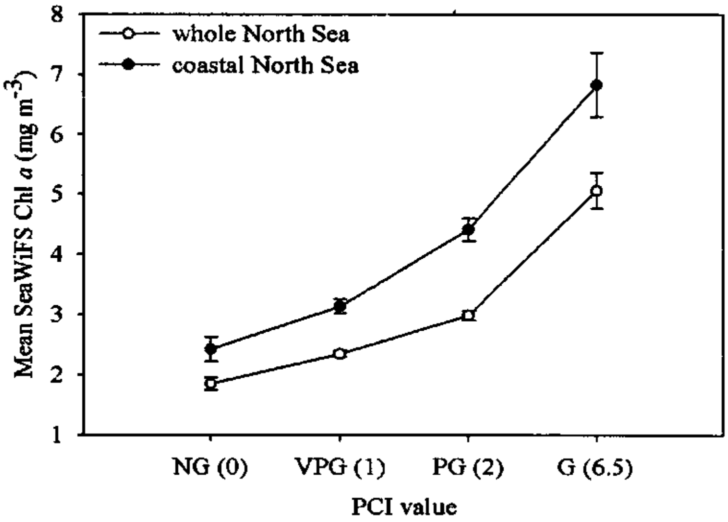


Figure 7.2. Equivalent SeaWiFS – PCI values for whole and coastal North Sea areas with 95% confident limits. This relationship was applied to the PCI dataset to extrapolate the new Chl a time-series. The PCI is a ratio scale of Phytoplankton Colour with four 'greenness' values: 0 (NG - no greenness), 1 (VPG - very pale green), 2 (PG - pale green), and 6.5 (G - green). Note there is no overlap between confidence intervals for each PCI category.

Simple correlation analysis (Pearson) and multiple linear regression were used to determine the existence and strength of possible relationships between environmental and biological variables. For multiple regression modelling, data were first assessed for normality (Kolmogorov-Smirnov test) and transformed if necessary. Models were estimated using forward selection stepwise regression procedures with Chl *a* (coastal or open) as the dependent variable and a suite of environmental parameters as candidate independent variables. Robustness of resulting models (α set at 0.05) was assessed by testing the residuals for normality (K-S test) and homoscedasticity (scrutinizing plots of standardized residuals). All significant analyses conformed to these regression assumptions. Standardized regression coefficients were used to infer the relative importance of model variables for explaining variations in Chl *a* data.

7.2.3.2 Potential biases

Consistency and comparability of the methodology used in the CPR survey has been studied in depth (Batten *et al.*, 2003a). Although standard methods have been used for more than 50 years in the survey, the PCI has been measured by a number of different analysts during this time. However, evaluating greenness is a simple task that is typically undertaken by 2 to 3 people in a year, many of whom have done this work for more than a decade. As well as referring to a standard colour chart, apprentices are trained in assessing PCI for a year before performing the task on their own (Raitsos *et al.*, 2005).

The study area includes both Case I (open ocean) and Case II (optically complex coastal) waters (IOCCG, 2000). In Case II waters, Chl *a* is difficult to distinguish from particulate matter and/or yellow substances (dissolved organic matter) and so global Chlorophyll algorithms (such as OC4-v4) are less reliable (IOCCG, 2000). Previous work applying the PCI/SeaWiFS Chl *a* relationship has occurred primarily in Case I waters (Raitsos *et al.*, 2005). This is the first time the relationship has been used to create a Chl *a* time-series for a substantial geographic area featuring optically complex Case II waters; this analysis is possible due to the coupling between remotely-sensed (SeaWiFS) and *in situ* (PCI) measurements.

7.3 Results

7.3.1 Environmental measures

Since 1958, annual mean SST in the North Sea has demonstrated an increasing trend which was most pronounced during the late 1980s, and continues through to the present (Fig. 7.3) (Edwards *et al.*, 2002). Across the North Sea, mean SST increased by 0.48 °C between the 1960s and 1990s; locally, the increase was greatest in the southern North Sea waters (0.75°C) and least in the northern North Sea (0.18°C). The increasing trend in SST is related to changes in the NAO (Ottersen *et al.*, 2001) and is also highly correlated with Northern Hemisphere temperature (Beaugrand and Reid, 2003). During the 1950s and 1960s the NAO was in a negative phase, but from 1972 to 2002 was in its longest ever positive phase, reaching its highest recorded value in 1989. The phytoplankton growing season has been lengthened by the resulting warmer SST in the North Sea (Reid *et al.*, 1998).

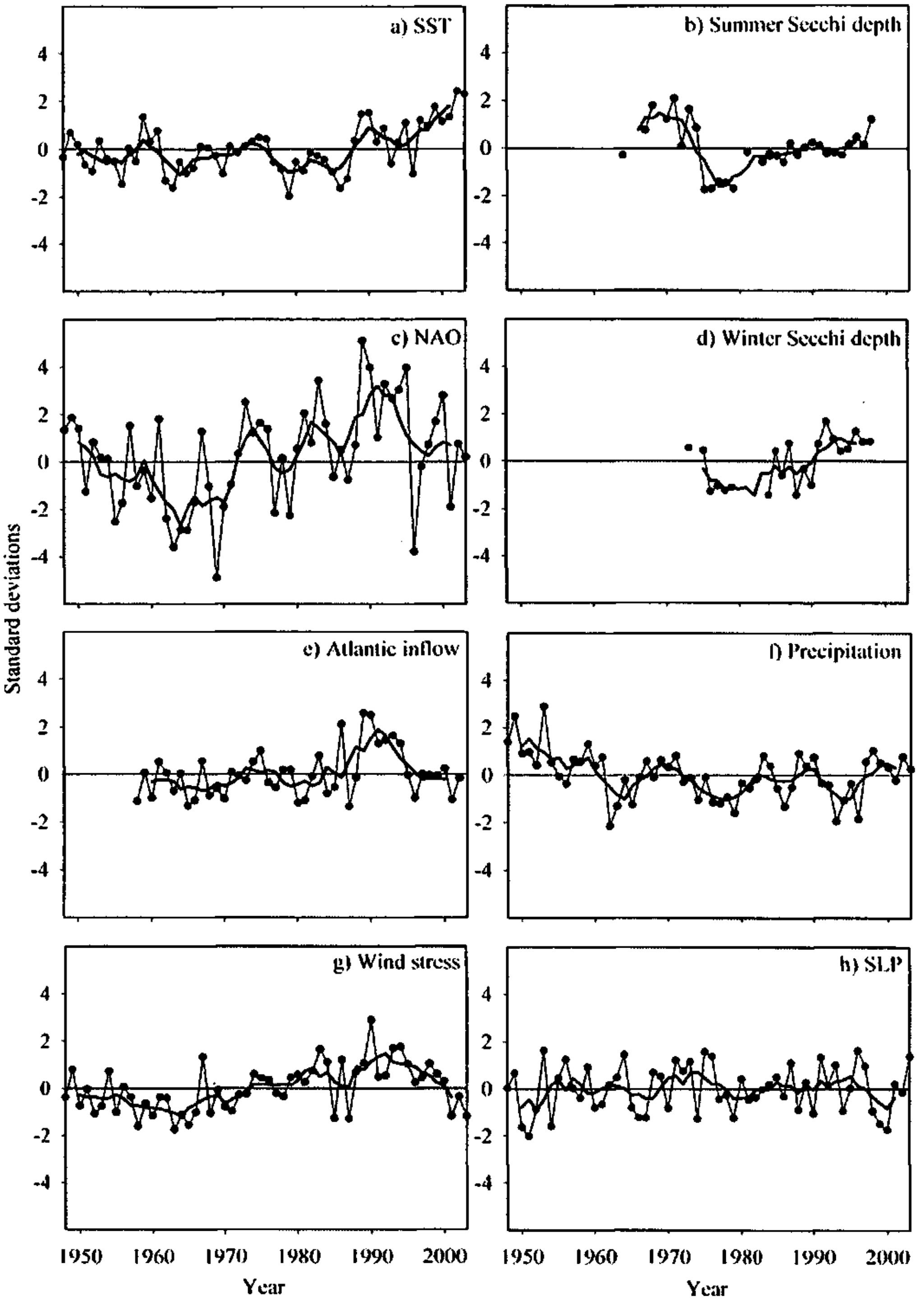


Figure 7.3. Time-series of (a) SST, (b) summer Secchi depth, (c) NAO, (d) winter Secchi depth, (e) Atlantic inflow, (f) precipitation, (g) wind stress, and (h) SLP with 5-year running means. With the exception of precipitation (panel f) and sea level pressure (panel h), all of the environmental variables considered in this study demonstrated a rapidly increasing trend during the 1980s. SST (panel a) and summer (panel b) and winter Secchi depth (panel d) continue to show increasing trends.

Secchi depth data ($n = 5057$) showed that, after a major decline during the early 1970s, water transparency in coastal North Sea waters has been increasing (Fig. 7.3). Improvement during both summer and winter has been greatest since the mid seventies, with a mean summer Secchi depth of 2.6 m and a mean winter depth of 1.8 m during the 1975-1979 period in contrast to a summer mean of 6.1 m and a winter mean of 5.7 m between 1991 and 1995.

Inflow of oceanic waters to the North Sea from the Atlantic Ocean displayed an increase throughout the 1960s and early 1970s before decreasing from 1976 through 1980 (Fig. 7.3). During the early and mid 1980s inflow continued to increase, reaching a maximum volume of more than 2.5 standard deviations above the long-term mean in 1989. From 1988 to 1995, influx of Atlantic waters remained consistently above the long-term mean, although volume of inflow began to decline in 1990. Inflow is significantly positively correlated with the NAOI ($r = 0.63$, $p < 0.001$) and SST ($r = 0.32$, $p = 0.035$) as well as significantly negatively correlated with Rhine ($r = -0.32$, $p = 0.03$) and Elbe ($r = -0.42$, $p < 0.001$) river discharge (Table 7.1 displays full details of correlation results).

Table 7.1. Results of correlation analysis between the coastal and open Chl *a* time-series and climatic and environmental variables. Significant ($p < 0.05$) results are in bold.

	Chl *a* open	Chl *a* coast	TN coast	TP coast	N:P coast	TN open	TP open	N:P open	Elbe flow	Rhine flow	Elbe TN	Elbe TP	Elbe N:P	Rhine TN	Rhine TP	Rhine N:P	Winter NAO	SST	Inflow	Summer Secchi	Winter Secchi	Precipitation	Wind stress
Chl *a* coast	**0.74**																						
TN coast	-0.25	**-0.65**																					
TP coast	-0.08	**-0.45**	**0.80**																				
N:P coast	0.07	0.27	**-0.63**	**-0.65**																			
TN open	-0.16	-0.18	0.51	-0.01	-0.42																		
TP open	-0.09	0.38	-0.22	-0.34	0.03	0.10																	
N:P open	0.00	-0.03	-0.21	0.06	0.34	**-0.55**	0.05																
Elbe flow	-0.06	-0.12	**0.43**	**0.59**	-0.33	0.12	0.24	**0.48**															
Rhine flow	0.15	0.12	0.34	0.23	0.06	0.33	0.05	0.35	**0.71**														
Elbe TN	-0.22	**-0.69**	**0.76**	**0.73**	**-0.58**	0.22	-0.26	-0.34	0.10	-0.02													
Elbe TP	**-0.75**	**-0.84**	**0.63**	**0.56**	-0.44	0.12	-0.37	-0.32	-0.21	-0.15	**0.93**												
Elbe N:P	**0.70**	**0.77**	**-0.50**	-0.41	0.37	-0.18	0.46	0.35	0.43	0.27	**-0.79**	**-0.93**											
Rhine TN	0.36	-0.36	**0.67**	**0.68**	-0.39	0.19	-0.16	-0.37	-0.14	-0.47	**0.90**	**0.71**	-0.46										
Rhine TP	-0.30	**-0.77**	**0.76**	**0.64**	**-0.52**	0.28	-0.31	-0.28	0.08	0.01	**0.97**	**0.94**	**-0.82**	**0.88**									
Rhine N:P	-0.29	0.07	**-0.55**	**-0.61**	0.27	-0.34	-0.01	0.38	0.26	0.47	**-0.72**	**-0.62**	0.48	**-0.56**	**-0.87**								
Winter NAO	**0.28**	0.15	0.13	0.01	0.04	0.00	-0.17	-0.19	-0.16	-0.03	0.11	0.07	-0.11	**0.54**	0.11	**-0.52**							
SST	**0.32**	**0.42**	**-0.59**	**-0.53**	**0.61**	-0.25	-0.10	-0.08	**-0.32**	-0.18	**-0.62**	**-0.62**	0.54	-0.21	**-0.64**	0.16	**0.50**						
Inflow	**0.36**	0.22	-0.01	-0.12	0.14	0.02	-0.19	-0.34	**-0.42**	**-0.32**	0.15	0.13	-0.28	**0.80**	0.13	**-0.73**	**0.63**	**0.31**					
Summer Secchi	0.19	0.35	**-0.60**	-0.44	-0.09	-0.10	0.11	-0.11	-0.04	-0.06	**-0.63**	**-0.63**	**0.59**	-0.42	**-0.66**	0.47	-0.03	0.24	-0.16				
Winter Secchi	0.27	**0.63**	**-0.55**	-0.48	-0.11	-0.10	0.36	-0.15	-0.17	-0.15	**-0.64**	**-0.58**	0.45	**-0.72**	**-0.54**	0.19	0.06	0.17	0.02	**0.53**			
Precipitation	-0.05	-0.09	-0.01	-0.04	0.17	-0.17	-0.49	-0.26	-0.19	-0.21	-0.05	-0.05	0.08	-0.15	-0.10	0.24	0.24	**0.48**	0.01	**0.43**	-0.08		
Wind stress	**0.30**	**0.26**	0.20	0.05	-0.12	0.28	0.03	-0.38	-0.11	-0.01	0.14	0.09	-0.13	**0.67**	0.09	**-0.57**	**0.64**	0.23	**0.70**	-0.18	-0.14	-0.02	
SLP	-0.11	0.12	0.07	0.18	-0.29	-0.06	0.32	-0.19	-0.21	**-0.40**	0.18	0.17	-0.19	0.16	0.14	-0.18	-0.15	0.01	-0.04	0.02	**0.47**	-0.08	-0.19

Until 1974, wind stress in the North Sea was predominantly below average for the study period (Fig. 7.3). After 1974, wind stress began to show an increasing trend, and reached its highest value in 1990. Although, like Secchi depth, wind stress displays an overall increasing trend throughout the late 1970s and 1980s, only a weak negative relationship exists between the two variables (see Table 7.1). Wind stress is however strongly correlated with Atlantic inflow ($r = 0.70, p < 0.001$) and NAOI ($r = 0.64, p < 0.001$) but not with SST or precipitation.

Unlike SST, NAO, wind stress, Atlantic inflow, and Secchi depth, precipitation did not show an increasing trend during the 1980s but instead showed considerable variability during that period (Fig. 7.3). Overall, precipitation was variable during the period of study with the 1970s being the driest decade and the 1950s the wettest, and showed positive relationships with SST ($r = 0.48$, $p < 0.001$) and summer Secchi depth ($r = 0.43$, $p = 0.02$), but no other environmental variables considered in this study.

Sea level pressure (SLP) is variable throughout the entire time-series (Fig. 7.3) and is correlated only with winter Secchi depth ($r = 0.47, p = 0.03$). There is a more pronounced low pressure signal at the beginning and end of the series.

7.3.2 Nutrient measures

When examining nutrient data from the North Sea regions, it is clear that the coastal zone was much richer than the open North Sea in total nitrogen (TN) and total phosphorus (TP) (Fig. 7.4a). Significantly decreasing trends in annual mean TN ($r^2 = 0.65$, $p < 0.001$, $n = 23$) and TP ($r^2 = 0.57, p < 0.001, n = 23$) were observed in coastal North Sea waters, TN and TP following very similar patterns (but note the difference in scales), with visible peaks in the late-eighties followed by dramatic decreases in TN and TP from 1988 - 2002. These are general trends, observable for nutrients in the coastal North Sea as a whole, and may not reflect local variability. For example, when examined at a finer scale, nutrient concentrations showed a significantly decreasing trend in the Southern Bight area since 1980 (TN: $r^2 = 0.57, p < 0.001, n = 23$; TP: $r^2 = 0.74, p < 0.001, n = 23$), but although TP has been decreasing in Norwegian coastal waters ($r^2 = 0.33$, $p < 0.01$, $n = 20$), TN displays an increasing trend in that area ($r^2 = 0.38, p = 0.01, n = 17$). However, because the concentrations of both TN and TP in the Southern Bight are up to 10 times richer than those in Norwegian coastal waters (Fig. 7.5), the general decreasing nutrient trends observable for the coastal North Sea as a whole are heavily weighted by nutrient dynamics occurring in the Southern Bight.

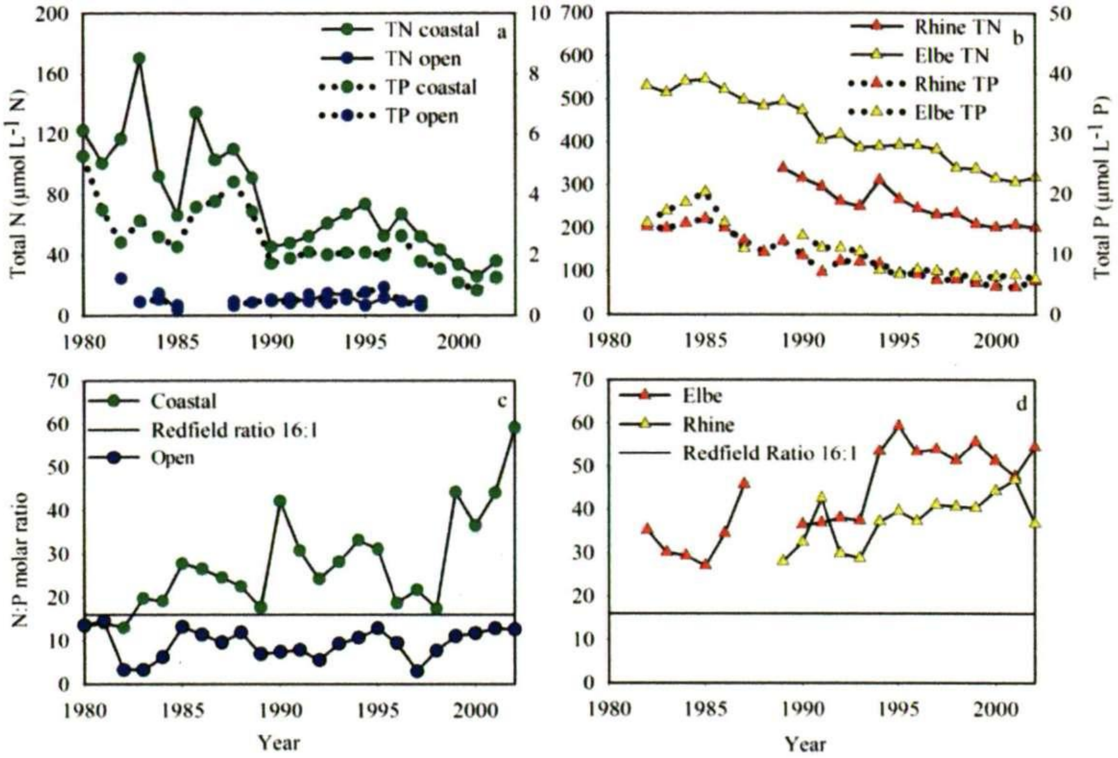


Figure 7.4. (a) Annual total nitrogen (TN) and total phosphorus (TP) concentrations in coastal and open North Sea waters and (b) Elbe and Rhine rivers during the period 1980 – 2002. Both datasets show similar decreasing trends in TN and TP in coastal North Sea and riverine waters, despite the differences in scale. As TN and TP concentrations decrease, the molar ratio of N to P is increasing in both (c) coastal North Sea and (d) Elbe and Rhine waters.

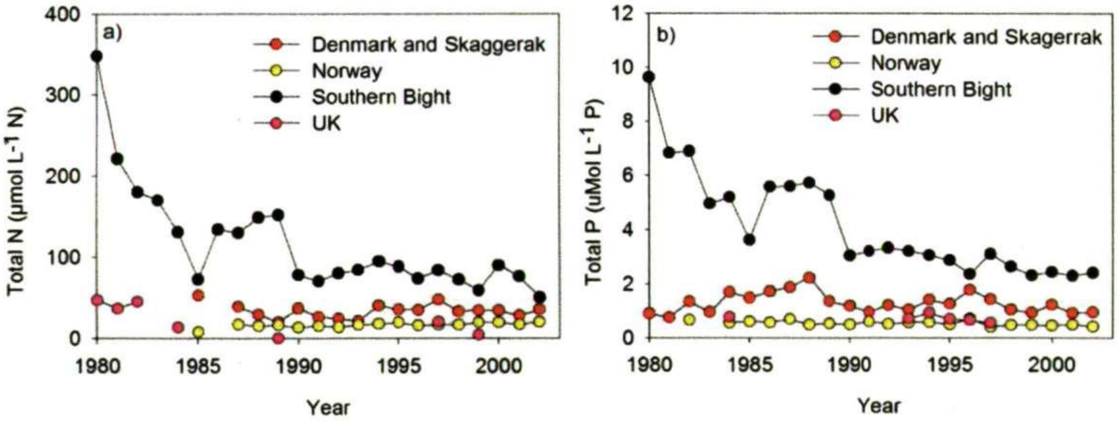


Figure 7.5. Annual total nitrogen (a) and total phosphorus (b) concentrations in regional areas of the coastal North Sea. Nutrient trends in coastal North Sea waters are dominated by the nutrient-rich waters of the Southern Bight, where both TN and TP have significantly decreased.

The open North Sea time-series of both nutrients have a much narrower range than their coastal counterparts; although the open North Sea lacks adequate data to distinguish any significant trend, there is no indication at all of a temporal change in nutrient levels. As the concentrations of TN and TP decreased in coastal North Sea waters, the molar ratio of N:P rose far above the Redfield ratio of 16:1, the molecular ratio at which diatoms require the two elements (Redfield *et al.*, 1963). The N:P ratio continued to increase in coastal waters (Fig. 7.4c) with a substantial jump in ~1988, while in the open North Sea N:P remained below the Redfield ratio. Nutrient concentrations in coastal waters are negatively correlated to SST (TN coast: $r = -0.59$, $p < 0.001$; TP coast: $r = -0.53$, $p = 0.01$) while N:P in coastal waters is positively correlated to SST ($r = 0.61$, $p < 0.001$). Possibly due to a lack of sufficient nutrient data, open North Sea nutrients and N:P show no significant relationship with SST or any other environmental variable.

Since the start of the time-series in 1982 annual mean nutrient concentrations have decreased in both the Elbe (TN: $r^2 = 0.94$, $p < 0.001$, $n = 21$; TP: $r^2 = 0.71$, $p < 0.001$, $n = 21$) and Rhine rivers (TN: $r^2 = 0.84$, $p < 0.001$, $n = 14$; TP: $r^2 = 0.91$, $p < 0.001$, $n = 21$) (Fig. 7.4b). Data from the International Commission on the Protection of the Rhine indicate that Rhine TP has displayed a decreasing trend since 1973 and Rhine NO_3 (pre-1982 data on Rhine TN was not available) increased steadily from 1954 before peaking in 1989 and declining thereafter (ICPR - International Commission for the Protection of the Rhine, 2006). Similarly to coastal North Sea waters, the difference in scale between the concentration of TN and that of TP in the Elbe and Rhine rivers has caused an increase in the riverine N:P (Fig. 7.4d). Concentrations of TN and TP in Elbe and Rhine waters are strongly correlated with coastal North Sea TN (Elbe TN: $r = 0.76$, $p < 0.001$; Rhine TN: $r = 0.67$, $p = 0.009$) and coastal North Sea TP (Elbe TP: $r = 0.56$, $p = 0.012$; Rhine TP: $r = 0.64$, $p = 0.002$) concentrations. Nutrient concentrations in both rivers demonstrate negative relationships with winter Secchi depth (Elbe TN: $r = -0.64$, $p = 0.01$; Elbe TP: $r = -0.58$, $p = 0.04$; Rhine TN: $r = -0.72$, $p = 0.02$; Rhine TP: $r = -0.54$, $p = 0.04$) and summer Secchi depth (Elbe TN: $r = -0.63$, $p = 0.01$; Elbe TP: $r = -0.63$, $p = 0.02$; Rhine TN: $r = -0.42$, $p = 0.23$; Rhine TP: $r = -0.66$, $p = 0.01$). SST is also negatively related to Elbe TN ($r = -0.62$, $p < 0.001$), Elbe TP ($r = -0.62$, $p < 0.001$), Rhine TN ($r = -0.21$, $p = 0.47$), and Rhine TP ($r = -0.64$, $p < 0.001$). Note that the correlations between Rhine TN and summer Secchi depth and SST are not statistically significant (Table 7.1); this may be due to the short time-series of Rhine TN (Fig. 7.4b). Furthermore, data for the Elbe river indicated no significant change in annual quantity of water discharged into the North Sea between 1948 and 2001, while output from the Rhine increased slightly ($r^2 = 0.09$, $p < 0.05$, $n = 54$) (Fig. 7.6). Additional analysis also revealed that there has been no seasonal change in discharge pattern for either river. Thus, the absence of change in quantity of river discharge in conjunction with the decline in riverine nutrients indicate that the Elbe and Rhine rivers have reduced their input of these nutrients to North Sea waters.

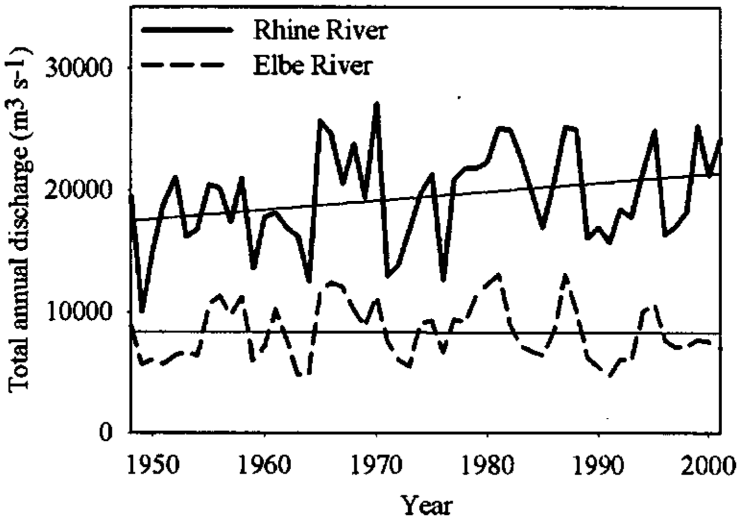


Figure 7.6. Annual mean discharge (1948 – 2001) of Elbe and Rhine rivers. There is no trend in Elbe discharge and a slight increase in Rhine discharge, indicating that the decrease in coastal nutrients can most likely be attributed to the decline in nutrient concentrations in both rivers rather than decreased discharge.

In general, coastal North Sea and riverine nutrient concentrations, have been decreasing since the early 1980s, while open North Sea nutrient concentrations show no clear trend. However, since the early 1980s, SST, Secchi depth, wind stress, and Atlantic inflow have displayed increasing trends, while precipitation and SLP remain variable.

7.3.3 Phytoplankton production measures

The newly created Chl *a* time-series shows considerable variability in both the open and coastal North Sea until the mid-1980s when a rapid increase in Chl *a* began to occur (Fig. 7.7). This increase resulted in a Chlorophyll peak in 1989, observable in both the coastal (3.92 mg m^{-3}) and open (3.15 mg m^{-3}) North Sea. After 1990, the coastal and open North Sea have both sustained Chl *a* concentrations well above those maintained before the 1980s rather than returning to pre-phase shift levels (Fig. 7.7). As mentioned earlier, this pattern of change for the North Sea as a whole has been observed and described as a regime shift (Reid *et al.*, 1998).

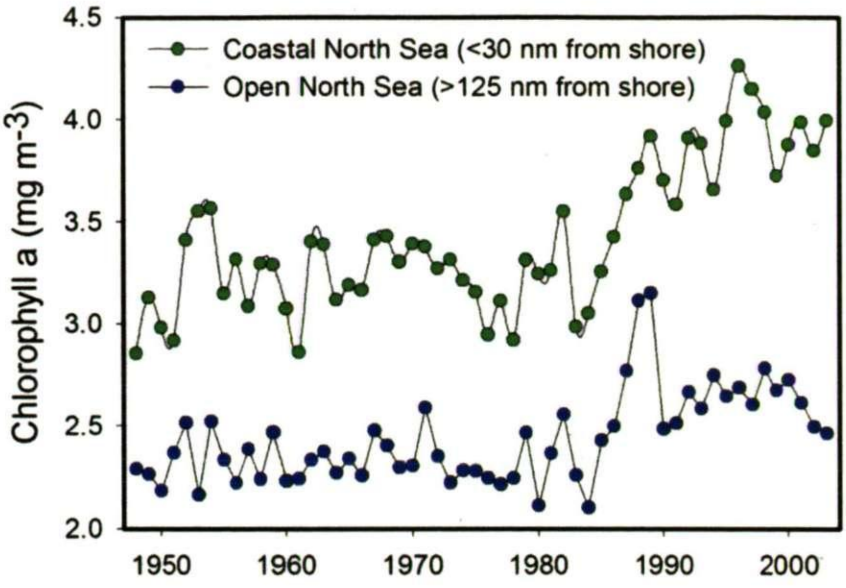


Figure 7.7. Time-series of the new Chl *a* data set (annual means) for the period 1948 to 2003 in the coastal and open North Sea. The regime shift is evident as a rapid increase in Chl *a* between 1982 and 1989 followed by mean annual Chl *a* concentration that is consistently higher than that observed prior to the regime shift. Both open and coastal North Sea time-series show a higher level of Chl *a* in the period after the regime shift than in the period before.

The changes in Chl *a* observed in coastal waters have occurred throughout the coastal North Sea. The regime shift and subsequent increased level of Chl *a* are visible in the northern (> 58° latitude), central (55° – 58° latitude), and southern (< 55° latitude) coastal regions of the North Sea (Figs. 6.8a-c). SST is significantly correlated with both open Chl *a* ($r = 0.32$, $p < 0.05$) and coastal Chl *a* ($r = 0.42$, $p < 0.01$). Also, winter NAO is significantly correlated with Chl *a* in the open North Sea ($r = 0.28$, $p = 0.036$) but not with coastal Chl *a*. Wind stress, however, is significantly positively correlated with both open Chl *a* ($r = 0.30$, $p = 0.03$) and coastal Chl *a* ($r = 0.26$, $p = 0.05$). Coastal Chl *a* also has a strong significant relationship with winter Secchi depth ($r = 0.63$, $p = 0.002$) while open Chl *a* is positively correlated with Atlantic inflow ($r = 0.36$, $p = 0.01$). Most interestingly, coastal Chl *a* was found to have a significant negative relationship with both TN ($r = -0.65$, $p = 0.001$) and TP ($r = -0.45$, $p = 0.029$) in coastal waters but an insignificant relationship with molar N:P ($r = 0.27$, $p = 0.216$). In addition to nutrients in coastal waters, coastal Chl *a* is negatively correlated with riverine nutrients as well (Elbe TN: $r = -0.69$, $p < 0.001$; Elbe TP: $r = -0.84$, $p < 0.001$; Rhine TN: $r = -0.36$, $p < 0.21$; and Rhine TP: $r = -077$, $p < 0.001$). Notice that the correlation between coastal Chl *a* and Rhine TN is not statistically significant; again this may be due to the short time-series of Rhine TN.

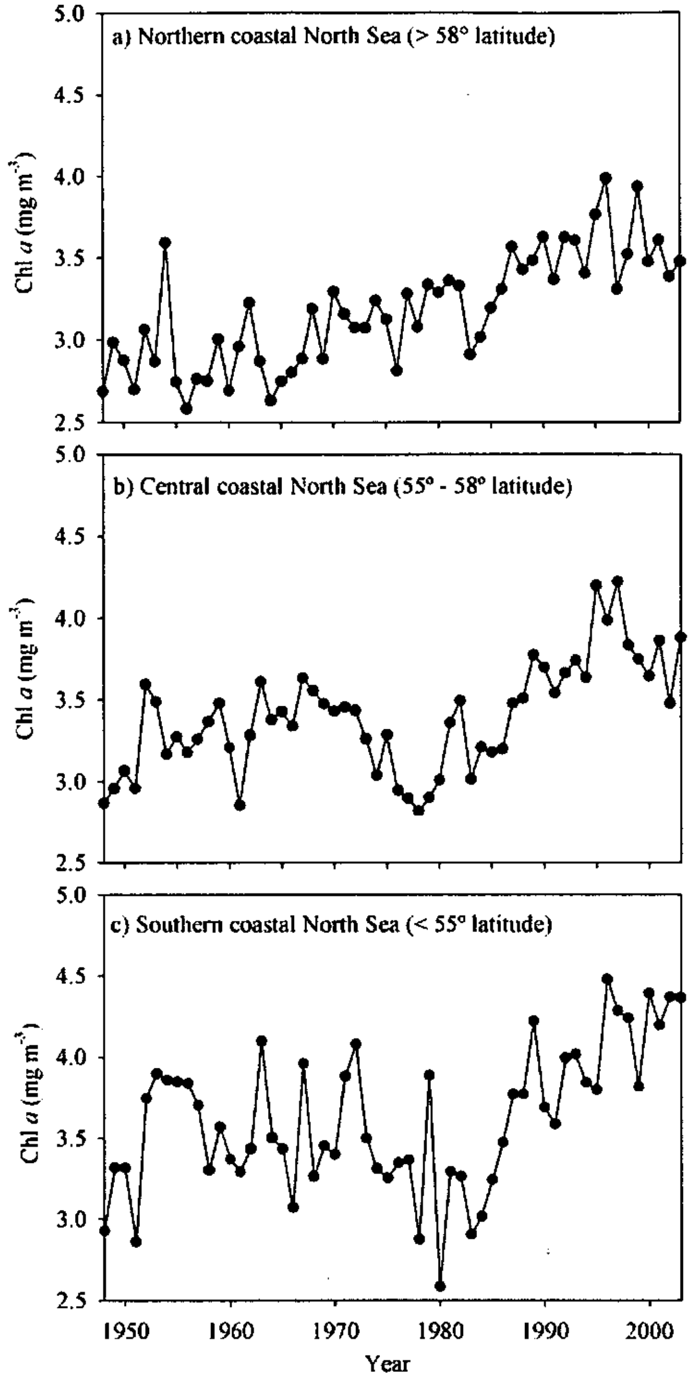


Figure 7.8. Trends in Chl *a* in the (a) northern, (b) central, and (c) southern coastal regions of the North Sea.

Separate multiple linear regression analyses were performed on the coastal and open Chl *a* datasets (Table 7.2). In the coastal North Sea model, winter Secchi depth (standardized regression coefficient, β, = 0.564, p = 0.017) and SST (β = 0.455, p = 0.045) were the most important predictors of Chl *a* (r^2 = 0.51, p = 0.015, n = 15) while in the open North Sea model no significant predictor was identified. Although no variable was significant when performing the multiple linear regression on the open North Sea, the exclusion of nutrients (due to their short time-series, $n = 12$) showed that Atlantic inflow ($\beta = 0.364$, $p = 0.14$) is the best predictor of Chl *a* in the open North Sea, although the relationship is relatively weak ($r^2 = 0.112$, $p = 0.014$, $n = 45$). Neither model, therefore, featured nutrients (TN, TP, or N:P) as significant predictors of Chl *a*.

Table 7.2. Results of multiple linear regression analyses, including coefficient (b) and standardized regression coefficient (β) for each significant predictor. In the coastal model, winter Secchi depth and SST were the most important predictors of Chl *a* while in the open North Sea model no significant predictor of Chl *a* was identifiable unless nutrients were excluded from the model (due to their short time-series). A diagnostic K-S test was performed to ensure normality of the residuals: both models conformed to the assumptions of linear regression.

	Significant Predictor	*b*	*p*	β	Model *r2*	Model *p*	Model *n*	Model diagnostic
Coastal Chl *a*	Winter Secchi	0.085	0.017	0.564	0.510	0.015	15	0.682
	SST	0.318	0.045	0.455				
	constant	0.173						
Open Chl *a* (nutrients excluded)	Atlantic inflow	0.883	0.014	0.364	0.112	0.014	45	0.392
	constant	1.382						

In summary, Chl *a* across the entire North Sea has displayed a rapid increase from the mid 1980s, culminating in the regime shift peak in 1989. After 1989, Chl *a* concentrations remained higher than in the pre-regime shift period in both the open and coastal North Sea. These results indicate that coastal Chl *a* is negatively correlated with coastal nutrient concentrations, but positively correlated with winter Secchi depth and SST. Open North Sea Chl *a* shows no significant relationship with nutrients and is most closely correlated with Atlantic inflow, wind stress, and SST.

7.4 Discussion

It is well documented that in 1989 the North Sea experienced a strong peak in phytoplankton abundance (Reid *et al.*, 1998) that has been correlated with warmer than average SST, a positive phase in the NAO and increased oceanic inflow from the North Atlantic (Beaugrand, 2004c). This same peak, preceded by a rapid increase in Chl *a*, can be clearly observed in the new coastal and open North Sea Chl *a* time-series (Fig. 7.7). The rapid increase in Chl *a* beginning during the mid-1980s and peaking in 1989 are part of what is now thought to be a regime shift, a stepwise modification in the composition and productivity of an entire ecosystem at a regional scale, reflecting substantial hydrographic change (Beaugrand, 2004c; Reid *et al.*, 2001a; Reid and Edwards, 2001). At that point, the system converted to an alternate resilient state. As this sequence of events is non-linear, a reduction of pressures on the system does not necessarily mean that the system will recover to a previous alternate state (Scheffer *et al.*, 2001), a clear example of hysteresis.

Because riverine input is the primary vector through which anthropogenic nutrients enter the North Sea, nutrient concentrations in coastal North Sea waters are fundamentally linked to riverine nutrient concentrations and discharge (de Vries *et al.*, 1998; van Bennekom and Wetsteijn, 1990). Nutrient concentration data from the Elbe and Rhine rivers, important sources of nutrients to coastal waters, demonstrated a significant decrease in TN and TP since the early 1980s, while quantity of water discharged changed little (Figs. 6.4b, d, 6.6). This suggests that the decreased anthropogenic nutrient loads to the North Sea have not occurred due to decreased river discharge but more likely due to declining riverine nutrient levels. This is further supported by the positive correlations between riverine nutrients and coastal nutrient concentrations and negative relationship between riverine nutrients and coastal Chl *a* as well as the lack of relationships between coastal Chl *a* and Elbe and Rhine discharge (Table 7.1). Surprisingly, Elbe and Rhine discharge do not appear to be related to precipitation. This could be because the precipitation data used in this study are for the North Sea itself and not the river catchments. However, Elbe discharge is significantly negatively related to SST (r = -0.32, p = 0.02), although Rhine discharge is not (r = -0.18, p = 0.19).

The declining riverine nutrient load is reflected in the decreasing nutrient concentrations observed in coastal waters and has caused the current regime in the North Sea to be lower in coastal nutrient concentrations than the previous regime. Decadal means of TP and TN in coastal waters were nearly 50% higher in the 1980s (TN = 110.48 μmol L^{-1}, TP = 3.43 μmol L^{-1}) compared to the post-regime shift 1990s (TN = 56.03 μmol L^{-1}, TP = 1.97 μmol L^{-1}). Other studies have also observed declining concentrations of phosphorus (de Vries *et al.*, 1998; Nixon *et al.*, 2003; OSPAR, 2000) and, to a lesser extent, nitrogen (de Vries *et al.*, 1998; Nixon *et al.*, 2003) in areas of the coastal North Sea in recent years. Agriculture is currently the main anthropogenic contributor of both N (63%) and P (45%) to North Sea waters (Anonymous, 2005; Nixon *et al.*, 2003). Since 1985, pollution from agricultural nitrates has been reduced by 21% (Nixon *et al.*, 2003); part of this reduction can be attributed to the Nitrates Directive which was adopted by the EU in 1991

(European Union, 1991a). Since 1985, agricultural P has been reduced by just 5% (Nixon *et al.*, 2003). In 1991 the EU also implemented the Urban Waster Water Treatment Directive (European Union, 1991b). This directive has been partially responsible for the 42% decrease of N and the 78% decrease of P from urban waste water treatment works since 1980 (Nixon *et al.*, 2003). Since 1985 there have also been significant reductions in N and P loads from industry (81% P reduction, 79% N reduction) and other sectors (62% P reduction, 43% N reduction) (Nixon *et al.*, 2003).

As concentrations of TN and TP have decreased, the molar ratio between N and P has steadily increased to > 40:1 in coastal (Fig. 7.4c) and to > 30:1 in riverine (Fig. 7.4d) waters; the change in N:P is potentially an artifact of the similar rate of decline in the two contrasting scales of concentrations. Since 1983, N:P has remained above the Redfield ratio of 16:1 in the coastal North Sea while in the open North Sea, the ratio has stayed consistently below Redfield. Throughout our time-series, both Elbe and Rhine waters have had an N:P greater than Redfield. Unlike the coastal North Sea, an increase in the N:P ratio is probably not observable for the open North Sea because a trend in nutrient concentrations cannot be found due to insufficient data (Fig. 7.4a). However, as nutrient concentrations in the open North Sea are predominantly influenced by natural variability and not anthropogenic (riverine) sources (Lenhart *et al.*, 1997), a rapid change in nutrient concentrations for open waters, as seen in coastal and riverine waters, would not be expected. Therefore as open North Sea nutrient concentrations change little, so does the N:P ratio. This suggests that P is presently the limiting nutrient in coastal North Sea waters while the open North Sea is N limited (Pätsch and Radach, 1997). Studies in the Marsdiep (Philippart *et al.*, 2000) and Helgoland (Hickel *et al.*, 1992; Radach *et al.*, 1990) have shown that a strong relationship exists between N:P and phytoplankton community composition, while laboratory experiments have demonstrated that a high N:P creates favourable conditions for *Phaeocystis*, a harmful algal bloom (HAB) causing species (Riegman, 1991; Riegman *et al.*, 1992). Thus, further investigation is needed to determine the effects of change in the N:P on the wider coastal phytoplankton community composition.

As a consequence of the regime shift, biological changes to new alternate states are revealed in the new Chl *a* datasets. The datasets are in good agreement with changes observed previously for the PCI (Reid *et al.*, 2001a) and then later confirmed and quantified based on the PCI/SeaWiFS relationship for the North Atlantic and North Sea as a whole (Raitsos *et al.*, 2005). As mentioned previously, the regime shift is evident as a rapid increase followed by a consistently high level of Chl *a* in both coastal and open North Sea waters, but these changes cannot be explained by nutrients alone as nutrient concentrations in the open North Sea have not changed significantly (Fig. 7.4a). The multiple linear regression models confirm this as they too indicate that nutrients are not the best predictors of Chl *a* in either region of the North Sea (Table 7.2). After the regime shift, neither coastal nor open North Sea Chl *a* returned to its pre-regime shift level, but instead the time-series indicates that the current North Sea regime maintains an average Chl *a* level 21% higher in the coastal zone and 13% higher in the open zone than that of the pre-1980 regime. Additionally, Chl *a* in coastal waters appears to be increasing still further, particularly in the southern North Sea (Fig. 7.7). Post regime shift alterations to the North Sea ecology have also been observed, such as a change in the proportion of warm and cold water species of *Calanus* (Reid *et al.*, 2003a), trophic mismatch due to changes in phenology (Edwards and Richardson, 2004), changes in distribution of horse mackerel (Reid *et al.*, 2001a) and a collapse of cod stocks (Beaugrand *et al.*, 2003). The current North Sea regime has fewer piscivorous top predators (Heath, 2005; Reid and Edwards, 2001), and a lower mean trophic level of fisheries landings (Pauly *et al.*, 1998).

Although nutrient concentrations in the coastal North Sea have decreased significantly since the regime shift, algal biomass has continued to increase. The coincident decrease in nutrients and increase in Chl *a* is surprising as, traditionally, increases in plankton production are triggered by increasing nutrients and are a symptom of eutrophication (Cloern, 2001; Nixon, 1995). In fact, areas of the coastal North Sea are commonly considered to be eutrophic (Hickel *et al.*, 1993; Philippart *et al.*, 2000; van Beusekom and de Jonge, 2002) and eutrophication was identified as a key issue affecting the North Sea in the 1987 North Sea Quality Status Report (Reid and Edwards, 2001).

While there is a strong relationship between algae and nutrients, it is considered to be non-linear in coastal North Sea waters (*i.e.*, a reduction in nutrient load does not lead to an equivalent reduction of phytoplankton biomass) (Lenhart, 2001). Furthermore, as North Sea nutrient concentrations have decreased, water transparency has improved (Fig. 7.3), allowing the light-limited coastal phytoplankton (Pätsch and Radach, 1997) to make better use of available nutrients. A similar increase in phytoplankton biomass was also documented in the Scheldt estuary where the construction of a storm surge barrier caused reduced river flow with a corresponding increase in water transparency and decrease in nutrient concentrations (Westetyn and Kromkamp, 1994). Improvement in water transparency has also been observed in the Western Wadden Sea (de Jonge *et al.*, 1996) and Marsdiep (Bot and Colijn, 1996). Although the explanation behind the increase in North Sea water transparency is still unclear, it may be linked to the increased inflow of warm, clear, oceanic water entering the North Sea (Reid *et al.*, 2003a). Additionally, multiple linear regression showed that Atlantic inflow is the best predictor of Chl *a* in the open North Sea when nutrient data are excluded from the analysis. The transparency minimum (Fig. 7.3) occurring during the mid to late 1970s is coincident with a period of reduced Atlantic inflow (Corten, 1999) and low Chl *a* values in both the open and coastal North Sea datasets (Fig. 7.7). This is in agreement with work by Edwards *et al.* (2001b) who also attributed low PCI values in the late 1970s and early 1980s to a period of reduced Atlantic inflow. Additionally, the increasing water transparency that occurred throughout the 1980s and 1990s coincides with a proposed increase in Atlantic inflow during that period (Corten, 1999; Edwards *et al.*, 2002; Edwards *et al.*, 2001b). The increase of indicator species with a southerly origin in the mid 1980s and sustained abundance thereafter in the North Sea (Corten, 1999; Edwards *et al.*, 2001b), may indicate a change in current patterns, thereby increasing the volume of warm, clear, southerly water entering the North Sea from the North Atlantic (Edwards *et al.*, 2001b; Reid *et al.*, 2001b).

However, the reduction in turbidity alone does not seem to be enough to explain the increase in Chl *a*, coincident with the decrease in nutrients. One possible explanation may be that after the regime shift, coastal waters have become more vulnerable to fluxes in nutrient concentrations. Because coastal phytoplankton are light limited, the cooler, more turbid state of the previous regime may have acted as a buffer, preventing algae from reaching the high biomass that is now possible. Multiple linear regression analysis supports this, indicating that while winter Secchi depth is the best single predictor of coastal Chl *a*, it only explains 30% (p = 0.035, n = 15) of the variability. Together SST and winter Secchi depth form the best model, explaining 51% of the variation in coastal Chl *a* (Table 7.2). As mentioned previously, significant correlations exist between both open and coastal Chl *a* and mean annual SST. This result is in agreement with the previously established positive relationship between PCI and SST (Beaugrand and Reid, 2003). The low Chl *a* values occurring in the late 1970s (Fig. 7.7) and the exceptionally high Chl *a* values observed in the late 1980s correspond to two hydroclimatic anomalies: the previously mentioned cold-boreal event in the late 1970s, associated with lower than average SST, and a warm-temperate event in the late 1990s associated with warmer than average SST (Edwards *et al.*, 2002). Phytoplankton are closely linked to the temperature of their environment and respond to increased SST directly (physiologically), indirectly (enhanced or earlier stratification of the water column results in changes in phytoplankton succession) and phenologically (certain species are now experiencing earlier spring blooms) (Edwards and Richardson, 2004; Richardson and Schoeman, 2004). Additionally, warmer SST in the North Sea has created a longer growing season, thereby resulting in increased phytoplankton biomass, particularly during summer (Reid *et al.*, 1998; Reid *et al.*, 2001b) and winter (Raitsos *et al.*, 2005; Reid *et al.*, 2001b). The enhanced response by the algal community to changes in nutrients is a characteristic of the new alternate state of the North Sea and may have triggered the other documented changes to the pelagic system. If, due to increased SST and clearer water, coastal waters are indeed more sensitive to changes in nutrients, it is more important than ever that the quantity of anthropogenic nutrients entering the North Sea is reduced.

Alternatively, or additionally, to bottom-up (resource limited) control, it is also important to consider the top-down (consumer regulated) control of phytoplankton biomass. Fishing is an important industry in the North Sea and the last few decades have seen changes in the catch of the top predators such as cod, haddock, and mackerel (Heath, 2005; Reid and Edwards, 2001). The decrease in piscivorous predators, and the subsequent dominance of planktivorous fishes (sprat, sandeel, herring, and Norway pout), in the North Sea indicate that we are fishing down the food web (Heath, 2005; Pauly *et al.*, 1998). In theory, the resulting trophic cascade may have increased the consumption of zooplankton (Heath, 2005) and therefore may also effect a subsequent lessening of grazing pressure on phytoplankton leading to an increase in algal biomass (Reid *et al.*, 2000). However, data suggest that the top-down control of North Sea phytoplankton is only important during times of ecosystem stress (Reid *et al.*, 2000; Riegman, 1995). Additionally, Richardson and Schoeman (2004) suggest that bottom up control dominates the North Sea plankton community. More research is needed to further quantify the importance of top down control on North Sea phytoplankton biomass.

Therefore, it is suggested that changes in large-scale climatological forcing (possibly exacerbated by anthropogenic pressures such as overfishing) have gradually eroded the resilience of the North Sea system until a critical threshold was reached in the mid 1980s. Since then, the North Sea has been in an alternate state with its own characteristic ecological and environmental features. The present regime, though lower in nutrient concentrations, maintains a higher (and possibly still increasing) Chl *a* level in both coastal and open North Sea waters; this level of phytoplankton biomass in North Sea waters is more closely related to climatic variability via sea surface temperature and water transparency than nutrient concentrations.

Chapter 8

General discussion

Eutrophication is a growing concern in Europe's regional seas and appropriate indicators are required for its monitoring and assessment. Phytoplankton biomass and community composition comprise an important component of any possible suite of eutrophication indicators. Spatial sampling extent and scale, time-series length, non-linear responses of phytoplankton and indicator covariance all increase the difficulty of using phytoplankton as eutrophication indicators. However, these limitations may be overcome through comparison of coastal and open sea phytoplankton dynamics. This method is a particularly useful tool for isolating the impacts of eutrophication from those caused by climatic variability. The comparison of open and coastal phytoplankton datasets is explored in two geographic regions (the Black Sea and North-East Atlantic/North Sea) using phytoplankton as indicators at two scales (chlorophyll as an indicator of biomass and functional group abundance as an indicator of community composition). The successful implementation of phytoplankton as indicators of eutrophication in these two disparate sea regions at two different ecological scales suggests that this method could be applied to other European seas as a means of assessing nutrient enrichment and distinguishing between the impacts of climate change and eutrophication.

Chapter 8: General discussion

8.1 Introduction

The recent expansion of the European Union has increased the EU's size and geographical coverage while extending its coastal area eastward to the Black Sea coast. Each of Europe's seas currently experiences eutrophication to some extent. The two seas used as case studies in this thesis, the Black Sea and the North-East Atlantic/North Sea, have disparate ecological and anthropogenic histories. The majority of the countries in the North Sea catchment are EU member states and the North-East Atlantic/North Sea system is well studied and monitored. Nevertheless, eutrophication is an historical problem, particularly in certain areas of the North Sea. The Black Sea, though new to the EU, has a history of extensive eutrophication-induced ecosystem degradation. Environmental management of the Black Sea is further complicated by the fact that 9 of the 16 countries comprising the majority of its catchment are non-EU states. Despite these differences between the North-East Atlantic/North Sea and Black Sea, the selection and monitoring of appropriate eutrophication indicators is required in both regions (and throughout Europe's seas) to prevent further damage to marine ecosystems caused by excess nutrients.

8.2 Eutrophication indicators

Chapter 1 of this thesis justifies the selection of phytoplankton as suitable indicators for eutrophication in Europe. At this point it seems appropriate to explore where phytoplankton fit into a larger suite of eutrophication indictors, particularly in reference to those representing eutrophication-related pressures (such as nutrient loads or concentrations) or other ecosystem responses to eutrophication (Table 8.1).

Table 8.1. Some eutrophication indicators and a selection of references to their application.

Eutrophication indicator	**References**
Nutrient discharge loads	(Baan and van Buuren, 2003; Cociasu and Popa, 2004; Dederen, 1992; Mee *et al.*, 2005; OSPAR, 2005; Yunev *et al.*, 2007)
Nutrient concentrations or ratios	(Aertebjerg *et al.*, 2001; Baan and van Buuren, 2003; Cloern, 2001; Cociasu and Popa, 2004; OSPAR, 2005; Philippart *et al.*, 2007; Philippart *et al.*, 2000; Yunev *et al.*, 2007)
Phytoplankton biomass	(Aertebjerg *et al.*, 2001; Baan and van Buuren, 2003; Bodeanu *et al.*, 2004; Cloern, 2001; Marasovic *et al.*, 2005; OSPAR, 2005; Philippart *et al.*, 2007)
Phytoplankton species composition	(Bodeanu *et al.*, 2004; Hickel, 1998; Marasovic *et al.*, 2005; Officer and Ryther, 1980b; OSPAR, 2005; Philippart *et al.*, 2007; Philippart *et al.*, 2000; Sommer *et al.*, 2002; Yunev *et al.*, 2007; Zaitsev, 1992)
Harmful algal bloom frequency	(Bodeanu, 2002; OSPAR, 2005; Richardson and Jorgensen, 1996; Smayda, 1990; Verity *et al.*, 2002)
Macrophyte biomass	(Mee, 1992; OSPAR, 2005; Tett *et al.*, 2007; Zaitsev, 1992)
Macrophyte species composition	(Cloern, 2001; Grall and Chauvaud, 2002; Isaksson and Pihl, 1992; OSPAR, 2005; Tett *et al.*, 2007)
Macrophyte depth distribution	(Domin *et al.*, 2004; Tett *et al.*, 2007; Zaitsev, 1992)
Macrozoobenthos biomass	(de Jonge *et al.*, 1996; OSPAR, 2005; Philippart *et al.*, 2007)
Macrozoobenthos species composition	(Grall and Chauvaud, 2002; OSPAR, 2005; Philippart *et al.*, 2007; Zaitsev, 1992)
Benthic hypoxia (area or frequency)	(Baan and van Buuren, 2003; Mee, 2006; OSPAR, 2005; Richardson and Jorgensen, 1996; Zaitsev, 1992)
Benthic mass mortality	(OSPAR, 2005; Richardson and Jorgensen, 1996; Tett *et al.*, 2007; Zaitsev, 1992)

Because eutrophication is a response to nutrient enrichment, nutrient concentrations or loads can potentially provide an early warning signal of possible eutrophication ecosystem effects. However, nutrient enrichment is not a condition which is actually harmful unless ecosystem change is produced as a result (Tett *et al.*, 2007). Additionally, the relationship between nutrients and eutrophication is complex as alterations in nutrients may not always correspond to ecosystem changes associated with eutrophication. For example, in the coastal North Sea nutrient concentrations and loads have decreased although chlorophyll has not (Chapter 7). The concentrations or loads at which nutrients may produce an effect in the ecosystem also vary between regions and are dependent on the hydrographical properties of a system. A well-flushed water body like the North Sea is likely to respond more slowly to increased anthropogenic nutrients than the Baltic or Black Seas, which have restricted exchanges and long residency times.

Unlike nutrient concentrations and loads which indicate *pressure* on an ecosystem, the remainder of the eutrophication indicators listed in Table 8.1 are indicators providing information on ecosystem *disturbance*. Pressure indicators may have the ability to provide an early warning of ecosystem change, but as mentioned previously, are difficult to directly relate to eutrophication consequences. Disturbance indicators, though often influenced by other pressures such as climate, may also provide an early warning of eutrophication, particularly those indicators measuring biological responses to nutrient enrichment.

The biological effects of increased nutrients initially manifest through changes in phytoplankton. In other words, eutrophication-induced changes in the plankton occur prior to changes in other ecosystem components, thereby providing an early warning signal for eutrophication. The aspects of phytoplankton listed as indicators in Table 1.7 can provide information on the trophic effects of eutrophication, allowing a comprehensive picture of ecosystem change. However, the relationship between phytoplankton dynamics and nutrient-related pressure varies between regional seas and plankton response to nutrient-climate interactions is not fully understood, making it difficult to separate the effects of eutrophication from changes caused by climate variability.

Indicators regarding the health of the zoobenthic and phytobenthic communities can also provide useful information on degree of eutrophication. However, nutrient enrichment usually affects the benthic system after the pelagic, through, for example, increased turbidity as a result of phytoplankton blooms or hypoxia due to the bacterial decomposition of elevated levels of phytoplankton detritus. Therefore, response of benthic communities to nutrient enrichment does not occur as early as the phytoplankton response. Like phytoplankton indicators, phytobenthos and zoobenthos may also be subject to multiple pressures such as harvesting, the effects of dredging and trawling, and climate variability, all of which can confound the eutrophication signal.

Although associated with eutrophication, hypoxia and benthic mortalities do not provide early warnings of ecosystem disturbance, as these are often catastrophic state changes that indicate a breakdown in ecosystem resilience (Mee, 2005a; Scheffer *et al.*, 2001). This scenario is exemplified in the Black Sea which, prior to the agricultural intensification of the 1960s and 1970s, was home to an extensive macroalgal habitat associated with the red alga *Phyllophora*. As eutrophication intensified and the area of hypoxia (and eventually anoxia) expanded, *Phyllophora* habitat was lost. Although nutrient loads to the Black Sea have decreased since the disintegration of the communist bloc, and *Phyllophora* habitat may be recovering, the species associated with this macroalgal community are different than those common prior to eutrophication (Mee *et al.*, 2005). Thus, by the time hypoxia and benthic mortalities occur, restoration of an ecosystem to its previous state may be impossible even with a reduction in nutrient enrichment.

8.3 Phytoplankton as indicators

Although nutrient enrichment first manifests biologically through changes in the phytoplankton, phytoplankton have some limitations as indicators. A scarcity of historical baseline data, the confounding influence of climatic variability, a lack of spatio-temporally comprehensive datasets, and non-linear responses to environmental change complicate eutrophication assessment through phytoplankton monitoring. However, it is possible to overcome these challenges.

8.3.1 Baselines

Without historical knowledge of unimpacted ecosystem states it is difficult to identify and assess the severity and magnitude of change. Furthermore, due to the prolonged impact of human existence on marine ecosystems, particularly those closest to shore, very few ecological datasets extend back the hundreds of years necessary to provide an indication of the state of marine ecosystems before human impact (Jackson, 2001). Even the Continuous Plankton Recorder dataset, the longest marine ecological time-series in the world, began only in 1931 (Batten *et al.*, 2003a). Because unimpacted ecosystem state data is rare, other types of data may be used as substitute baselines.

Before the effects of nutrient enrichment are investigated, it is helpful to understand 'typical' phytoplankton dynamics in a region. Although seasonal spatial change in phytoplankton biomass in the North-East Atlantic has previously been examined (Drinkwater *et al.*, 2003; Edwards, 2000; Edwards *et al.*, 2001b), the spatial intra-annual dynamics of the major phytoplankton functional groups, diatoms and dinoflagellates, are presented for the first time in map form in Chapter 5. This type of basic data is a vital tool to understanding ecosystem changes caused by climatic variability or anthropogenic pressures and provides a baseline of group abundance throughout the year. Diatoms and dinoflagellates are two of the most important groups of primary producers in the marine ecosystem and their spatial and seasonal variability have direct consequences for upper trophic levels. For example, climatically-induced seasonal temporal mismatch in phytoplankton blooming and zooplankton spawning times has had direct consequences on North Sea cod stocks (Beaugrand *et al.*, 2003; Edwards and Richardson, 2004). Unfortunately, the Black Sea does not possess the large amount of temporally and spatially extensive seasonal baseline data for which the CPR provides the North-East Atlantic; however, information gleaned from observations in the North-East Atlantic can be tentatively applied to the Black Sea and tested as ecological datasets in that region increase in scope and length. For example, we have seen changes in the Black Sea chlorophyll seasonal cycle as a result of a hydroclimatic anomaly (Chapter 4). From this we could tentatively hypothesize that due to a change in seasonal blooming pattern, climate-driven phenologically-induced trophic mismatches similar to those occurring in the North Sea could possibly manifest in the Black Sea, particularly if the hydroclimatic anomaly of 2001 was a result of global warming. However, more data are needed to test this or any other such hypotheses.

Although few ecological time-series monitoring unimpacted ecosystem states exist (Jackson, 2001), the long time-series that are available provide valuable ecological information about temporal variability in the phytoplankton community. Chapter 6 illustrates decadal-scale spatio-temporal changes in North-East Atlantic diatom and dinoflagellate abundances. However, this information alone is not sufficient to investigate impacts caused by eutrophication and the influence of climate must also be considered when interpreting observed changes in the abundance of functional groups. Hydroclimatically-induced fluctuations in abundance of these two groups are immediately obvious from Figures 6.2 and 6.3: low abundances of both groups during the late 1970s were a response to a region-wide low temperature/low salinity anomaly in the North-East Atlantic while the high abundances observed in the 1980s were part of a region-wide regime shift. Taken out of context without climatic (or, if possible, nutrient) data, the increase in coastal diatom and dinoflagellate abundances (and chlorophyll concentration – see Chapter 7) resulting from the regime shift may be misinterpreted as eutrophication-related. Misinterpretation of ecological time-series due to a lack of reference data is particularly likely where only short time-series exist or data are not spatially comprehensive. The Black Sea, for example, is thought to be recovering from eutrophication. Whether this 'recovery' is an actual response to decreased nutrient loading or a result of climate changes (Chapter 4) is difficult to determine because of the limited availability of spatio-temporally comprehensive ecological datasets.

8.3.2 Limitations in plankton datasets

It is important to consider temporal scales when drawing conclusions based on ecological indicators. Long time-series of ecological data are most desirable for eutrophication assessment (*e.g.*, CPR data for the North-East Atlantic, Chapters 6, 7); however it is possible to use shorter datasets, as in the Black Sea (Chapter 4). An advantage of remotely-sensed datasets, though shorter in length than data from the CPR, is their regular temporal sampling. However, conclusions drawn from shorter datasets, even if they are regularly sampled, must be considered more tentative due to natural ecosystem variability which may be misinterpreted as anthropogenically-induced change at short temporal scales.

Spatial scale is also an issue when evaluating change in ecological indicators. In addition, hydrographic study regions must be selected carefully and, although aggregating data geographically can increase its reliability (more samples, better temporal coverage), small scale local variation may be underestimated. This was demonstrated in Chapter 7, where although phytoplankton biomass is not increasing as a result of enhanced nutrient loading at the scale of the whole coastal North Sea, eutrophication may remain a problem in some local regions.

Although SeaWiFS and the CPR provide important contributions to phytoplankton monitoring, both datasets have some limitations. For example, SeaWiFS is less accurate in Case II (coastal, turbid) waters where phytoplankton biomass is often overestimated by the satellite (IOCCG, 2000). The development of region-specific processing algorithms for SeaWiFS data may help to solve this problem (Nezlin, 2001). Due to the mesh size of its silk, the CPR primarily counts large, armoured flagellates and may underestimate unarmoured flagellate species, as well as other small plankton (Dickson *et al.*, 1992). However, the proportion of cells retained by the CPR silk has been found to reflect major changes in abundance, distribution and composition of the phytoplankton community (Robinson, 1970). Additionally, the CPR is normally towed further than 1 km offshore and therefore may miss the effects of eutrophication in localized areas very close to the shore (Edwards *et al.*, 2006). These limitations in SeaWiFS and CPR data must be accounted for when interpreting ecological information from those two sources.

8.3.3 Non-linearities and regime shifts

The relationship between phytoplankton, nutrients and climate is complex. In the North Sea, hydroclimatic change appears to exacerbate eutrophication through extension of the growing season and greater water transparency from increased Atlantic inflow (Chapter 7). At the same time, nutrient loads to the North Sea have decreased while chlorophyll has increased; this is a departure from the common view that increasing nutrients lead to increasing chlorophyll. The excess nutrients added to the system, though declining, may now be more readily available for biological use due to changes in hydroclimatic factors (such as rising SST and increasing water transparency as a result of increased Atlantic inflow). This is a case of covariance masking causality and highlights the need for even stricter nutrient controls in the catchment. The cumulative influence of climate and nutrients on Black Sea phytoplankton also appears to be non-linear, with a warming climate leading to greater stratification and decreased chlorophyll. But this relationship may hold only until a threshold temperature is reached, preventing formation of the Cold Intermediate Layer and nutrient subduction, leading to favourable growing conditions for phytoplankton (Chapter 4). Additionally, in both seas historical nutrients are stored in catchment soils as well as bottom sediment (Mee *et al.*, in prep.). These nutrients are a 'historical legacy' and supplement nutrients presently supplied by rivers and point sources, thereby contributing to eutrophication many years after their introduction to the system. Nutrient input in the two catchments has decreased in recent years (Chapters 3, 7) but remains above natural levels and certain climatic conditions allow phytoplankton to take advantage of these excess nutrients. A similar scenario has also been observed in the Baltic Sea where the quantity of historical phosphorus retained in and resuspended from the sediment dwarfs the amount of new phosphorus arriving from the North Sea and rivers (Mee *et al.*, in prep.).

In these ways, past anthropogenic pressure has resulted in 'locked-in changes' to the pelagic ecosystem. In other words, regardless of current nutrient reduction measures, historical nutrient loads may remain biologically available for many years, locking us into ecosystem changes as a result of past loads. The locked-in changes observed through nutrient legacies are similar to the locked-in changes resulting from historical carbon emissions: because of our past use of fossil fuels, we will be unlikely to see an improvement in climate change for many years, even if carbon emissions are immediately significantly reduced (Mee *et al.*, in prep.). Clearly, historical legacy and locked-in changes are a problem for policy makers who want to see immediate results (Chapter 3).

The North-East Atlantic regime shift discussed in Chapter 7 occurred due to region-wide hydroclimatic change and, though the result was increased phytoplankton biomass, was not connected with excess nutrient concentrations. The possible shift in the Black Sea chlorophyll regime which occurred in 2002 and is identified in Chapter 4 also appears to be climate related. However, unlike the North-East Atlantic regime shift which is comparatively well studied, the significance of the shift in the Black Sea chlorophyll regime must be considered with great caution. The Black Sea shift is much more recent and therefore its temporal extent cannot yet be known. In fact, it is not certain that the change in chlorophyll observed after 2002 is a regime shift; it is possible that changes observed in chlorophyll are the result of natural system variability and phytoplankton biomass may increase in coming years. Additionally, because of its recent occurrence, it is not yet known if higher trophic levels underwent corresponding significant change, a characteristic trait of regime shifts (Beaugrand, 2004c). Conversely, it is possible that this is the first identification of a major regime shift in the Black Sea ecosystem, but clearly more information is needed before the significance of the shift in chlorophyll can be determined and such a statement must remain speculation for the time being. Without the benefit of long-term ecological data it is extremely challenging to identify and determine the importance of such changes.

8.3.4 Ambiguous indicators

Two HAB-forming species, *Noctiluca scintillans* and *Phaeocystis* spp., are at times used as indicators of eutrophication and of climate change. The representativeness of either of these species as a eutrophication indicator is somewhat controversial. Although it has been used as a eutrophication indicator in the North Sea (Zevenboom *et al.*, 1991) and the Black Sea (Shiganova and Bulgakova, 2000), recent work by Edwards *et al.* (2006) found that the observed increase in abundance of *Noctiluca* in areas of the North Sea is the result of increasing winter SST rather than elevated nutrient concentrations. The increase in *Noctiluca* abundance also coincided with a general increase in dinoflagellate abundance in the North Sea attributed to post-regime shift hydroclimatic changes (Edwards *et al.*, 2006; Edwards and Richardson, 2004). However, as *Noctiluca* is a heterotrophic dinoflagellate, rather than a strict autotroph, it is also possible that an increase in organic matter, such as that associated with eutrophication, may trigger increased abundance of the species (Vasas *et al.*, 2007). The applicability of *Noctiluca* as a eutrophication indicator must therefore be determined on a case-by-case basis.

Similarly to *Noctiluca*, *Phaeocystis* has also been used as an indicator for eutrophication in the North Sea (Zevenboom *et al.*, 1991). However, recent work using CPR data has shown that *Phaeocystis* blooms were especially frequent in the North Sea during the 1960s and after the 1980s, two periods with relatively low nutrient inputs, and that occurrence of *Phaeocystis* blooms is closely related to hydroclimatic factors such as Atlantic inflow through the Dover Strait (Gieskes *et al.*, 2007). Additionally, recent modelling work has also shown that *Phaeocystis* biomass in Belgian coastal waters is attributable to both Atlantic inflow and riverine nutrient input and that a reduction in anthropogenic N could significantly reduce *Phaeocystis* biomass in that region (Lacroix *et al.*, 2007; Lancelot *et al.*, 2007).

8.4 Open sea vs. coastal

Even when spatially and temporally comprehensive ecological datasets are available, equivalent nutrient time-series are rare and a means of linking phytoplankton dynamics to eutrophication is still needed. Because open sea ecosystems are less impacted by eutrophication than those near shore, they may be used as reference areas in comparison with coastal systems to investigate the effects of nutrient loading. Changes observed in coastal systems alone are most likely a result of local processes (such as eutrophication) while those observed in both open sea and coastal areas are probably a response to climatic drivers. By segregating datasets into open sea and coastal regions, the influence of anthropogenic nutrients can also be separated from that of natural background nutrient concentrations. For example, 90% of nutrients entering the North Sea do so through inflow from the Atlantic Ocean (OSPAR, 2000) indicating that nutrient concentrations in the open North Sea are predominantly influenced by climatic variability rather than anthropogenic sources. Therefore the comparison of synchronous coastal and open sea data may serve to reveal disparate (or similar) patterns of change in ecological indicators in the two regions.

A comparison of the composition of open sea and coastal phytoplankton communities indicates that, at the North-East Atlantic scale, climate appears to override the influence of eutrophication, with similar patterns of change in phytoplankton community composition observable in both open and coastal waters (Chapter 6). Open sea and coastal phytoplankton communities were not found to be significantly different, although significant differences in both diatom and dinoflagellate communities were observable between coastal and open sea subregions. These results were confirmed when changes in coastal and open North Sea chlorophyll were investigated; although the coastal area was richer in chlorophyll than the open North Sea possibly due to elevated coastal nutrient levels, chlorophyll in the two regions was highly correlated and showed nearly identical patterns of change during the last 55 years (Chapter 7). Furthermore, coastal chlorophyll was positively correlated with climatic factors (SST, wind stress, winter Secchi depth) but negatively correlated with coastal and riverine nutrient concentrations, emphasizing the importance of climatic drivers in coastal waters. Although at the North Sea scale climate may dominate change in chlorophyll, the increasing trend in coastal North Sea chlorophyll observable since the regime shift may be a result of the synergistic influences of climate and nutrient enrichment. Additionally, these results don't rule out eutrophication as a real problem affecting localized areas of the North-East Atlantic/North Sea region.

Results from the Black Sea are less clear as the SeaWiFS time-series is relatively short and no comprehensive nutrient data are available. However, as in the coastal North Sea, Black Sea shelf waters are richer in chlorophyll than open sea waters (Chapter 4). In general, chlorophyll levels in the Black Sea appear to be decreasing (except for the anomalous 2001). However, this trend is also found throughout much of the World Ocean (Behrenfeld *et al.*, 2006), which indicates that the decreasing chlorophyll is at least partly attributable to climatic variability and is not solely a result of nutrient reduction.

8.5 Recommendations

The comparison of open sea and coastal phytoplankton dynamics is a useful tool for monitoring eutrophication. However, this technique must be used in conjunction with additional data regarding climate and, if possible, nutrients. Determining phytoplankton responses to changes in climate is especially problematic because of the various ways through which climate changes manifest in the marine ecosystem (such as alterations in temperature, stratification, precipitation, circulation). Further research into the impacts of these (and other) aspects of climate variability on phytoplankton biomass and community composition is required in order to understand phytoplankton responses to changes in climate.

Comprehensive monitoring programmes are crucial in order to assess the biological effects of eutrophication through comparison of open and coastal phytoplankton dynamics. The North-East Atlantic/North Sea is well monitored by the CPR, yet additional routes would provide even more information on plankton dynamics. An extension of CPR routes to the Black Sea would provide that region with an important source of consistently analysed *in situ* ecological data, which is currently missing. In addition to *in situ* data, the refinement or development of regionally-specific algorithms for interpreting remotely-sensed chlorophyll would increase the usefulness of SeaWiFS satellite data in the coastal North Sea and Black Sea. While *in situ* monitoring programmes reveal valuable information about phytoplankton community composition, satellite data can provide an important source of information on changes in phytoplankton biomass, particularly if accuracy could be improved in Case II (coastal, turbid) waters.

Although relating changes in nutrients to the biological effects of eutrophication is not straightforward, data on nutrient concentrations are highly desirable when interpreting the causes of eutrophication-induced biological change. More research is required to determine the exact responses of phytoplankton biomass and community composition to increased nutrient concentrations and changes in nutrient ratios. Furthermore, the EU's Water Framework Directive aims to achieve 'background' values of nutrient concentrations in the marine environment (Baan and van Buuren, 2003). However, nutrients are often irregularly sampled in time and space, particularly in open sea waters, and most nutrient time-series are short. While it may be impossible to recreate interannual nutrient time-series where data are limited, modelling exercises can help to fill some of these gaps. Modelling is a particularly useful tool in the Black Sea where historical nutrient data are scare and historical nutrient budgets are lacking (Artioli *et al.*, in prep.). Information about past nutrient dynamics could enable valuable insight into the relationship between anthropogenic nutrient pressures and ecosystem effects, particularly in the Black Sea, which is possibly undergoing a process of ecosystem recovery.

8.6 General conclusions

This thesis explored the complex responses of phytoplankton to changes in nutrient concentrations and the difficulty in separating the influence of climate from that of eutrophication on phytoplankton dynamics. Comparing coastal phytoplankton dynamics to those in open sea waters is a useful tool in eutrophication assessment and allows the isolation of the eutrophication from the climate signal. This method of separating the influence of these two drivers has been demonstrated here in two geographic regions (the Black Sea (Chapter 4) and North-East Atlantic/North Sea (Chapters 6, 7)) using phytoplankton as indicators at two scales (chlorophyll as an indicator of biomass (Chapters 4, 7) and community composition (Chapter 6)). Thanks to the CPR, the North-East Atlantic/North Sea has a long, spatially comprehensive time-series of plankton data (Chapters 5, 6, 7) allowing the establishment of baseline, though not 'pristine', reference conditions. At the North-East Atlantic/North Sea scale climate appears to be more important than nutrients as a regulator of phytoplankton dynamics (Chapter 6), although the two drivers have been found to have synergistic effects resulting in increasing chlorophyll levels in the coastal North Sea (Chapter 7). In the Black Sea, it seems that the observed decrease in chlorophyll is at least partially a result of changes in climate and is not solely attributable to the 'recovery' of the Black Sea ecosystem (Chapter 4). Additionally, the North Sea regime shift is clearly visible in coastal and open sea chlorophyll concentrations (Chapter 7) while Black Sea chlorophyll has also undergone a possible recent regime shift, although the significance of that shift is difficult to determine at this early stage (Chapter 4). However, spatial sampling extent and scale (Chapters 5, 6, 7), time-series length (Chapter 4), non-linear responses of phytoplankton (Chapters 4, 7) and indicator covariance (Chapter 7) all increase the difficulty of separating the influence of eutrophication and climate. Limitations aside, the successful use of phytoplankton as indicators of eutrophication in these two disparate sea regions at two different ecological scales suggests that this method could be applied to other European seas as a means of distinguishing between the effects of climate and eutrophication.

Appendix

This appendix contains a series of maps depicting monthly and annual mean SeaWiFS chlorophyll concentrations and standardized monthly chlorophyll anomalies in the Black Sea for the time period spanning September 1997 to December 2005. The methodology for creating these maps can be found in Chapter 4.

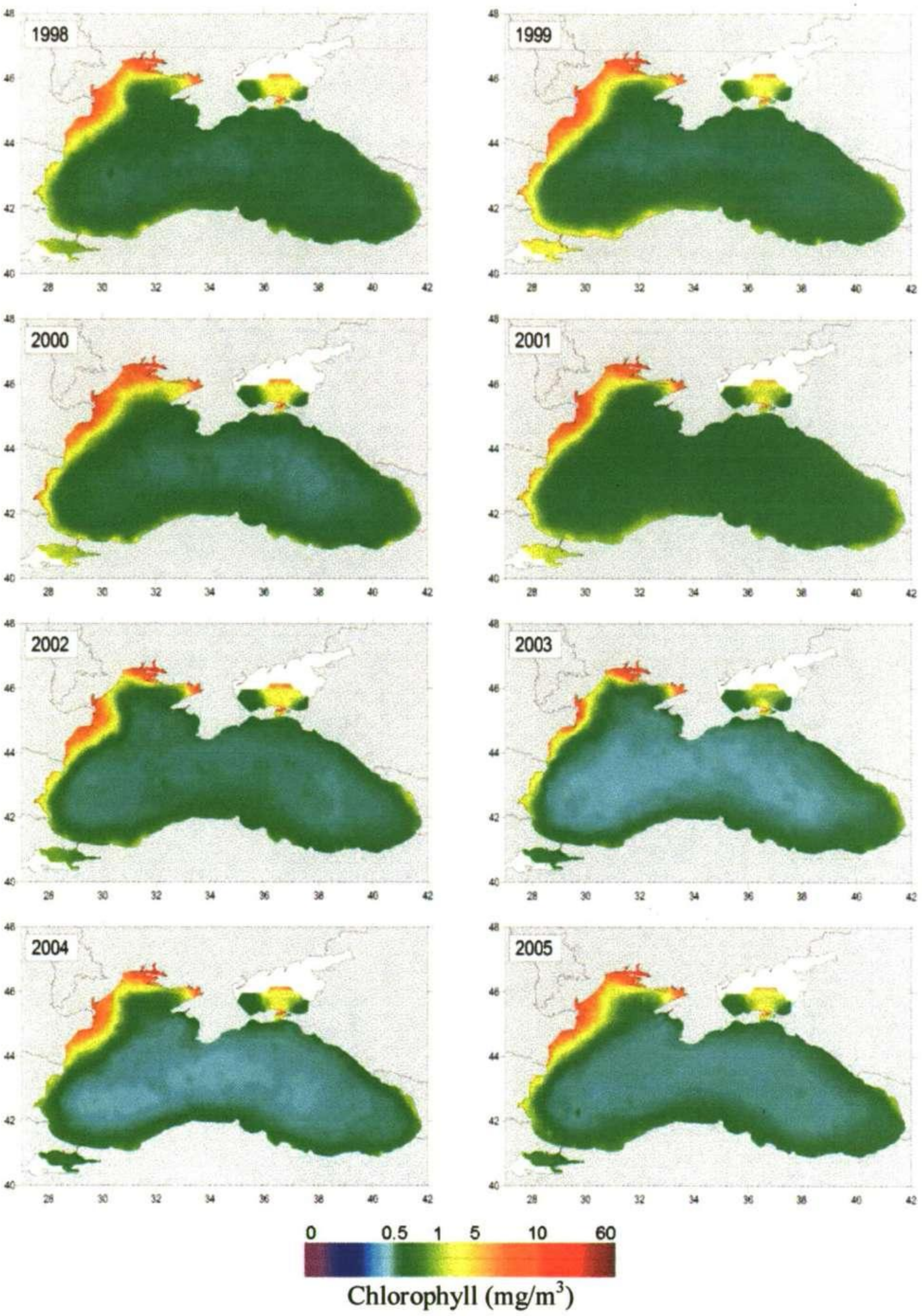


Figure A1. Mean annual SeaWiFS-derived chlorophyll in the Black Sea.

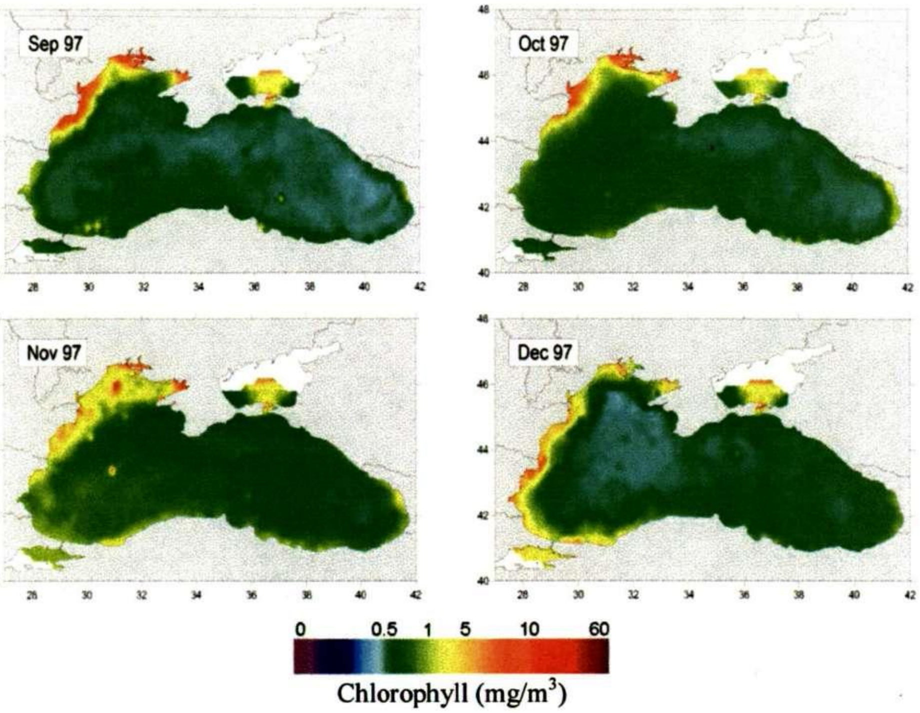


Figure A2. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 1997.

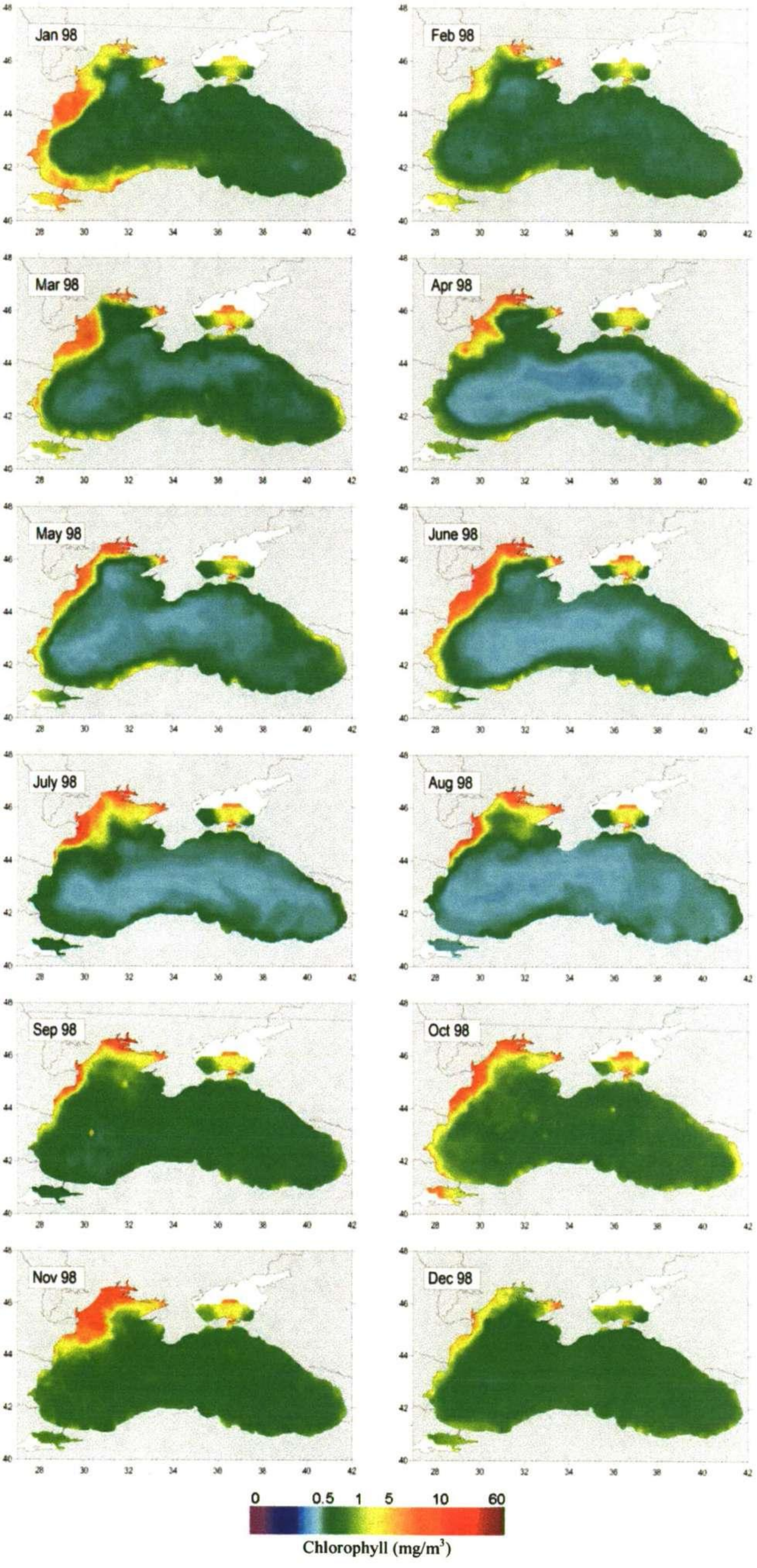


Figure A3. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 1998.

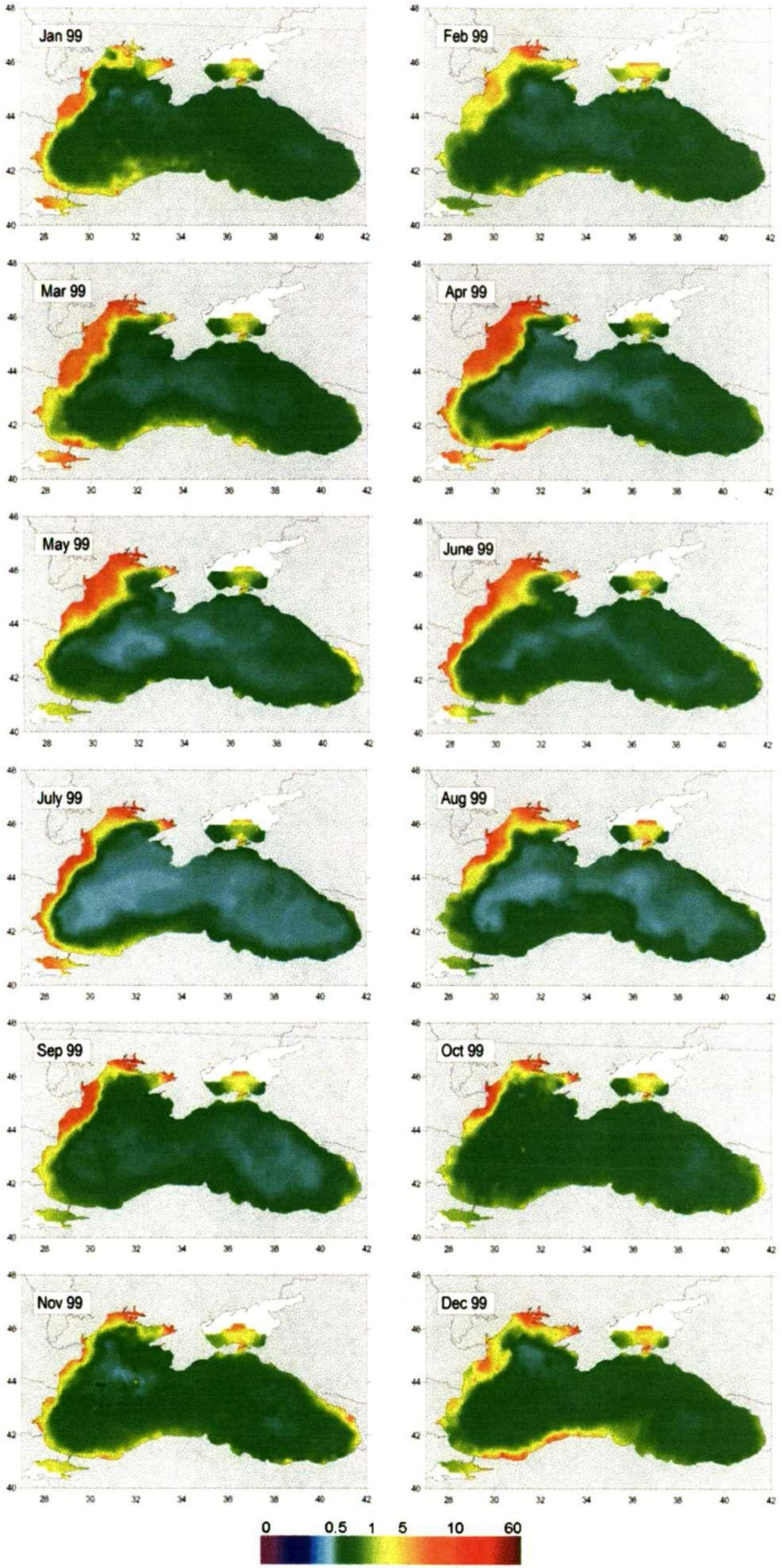


Figure A4. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 1999.

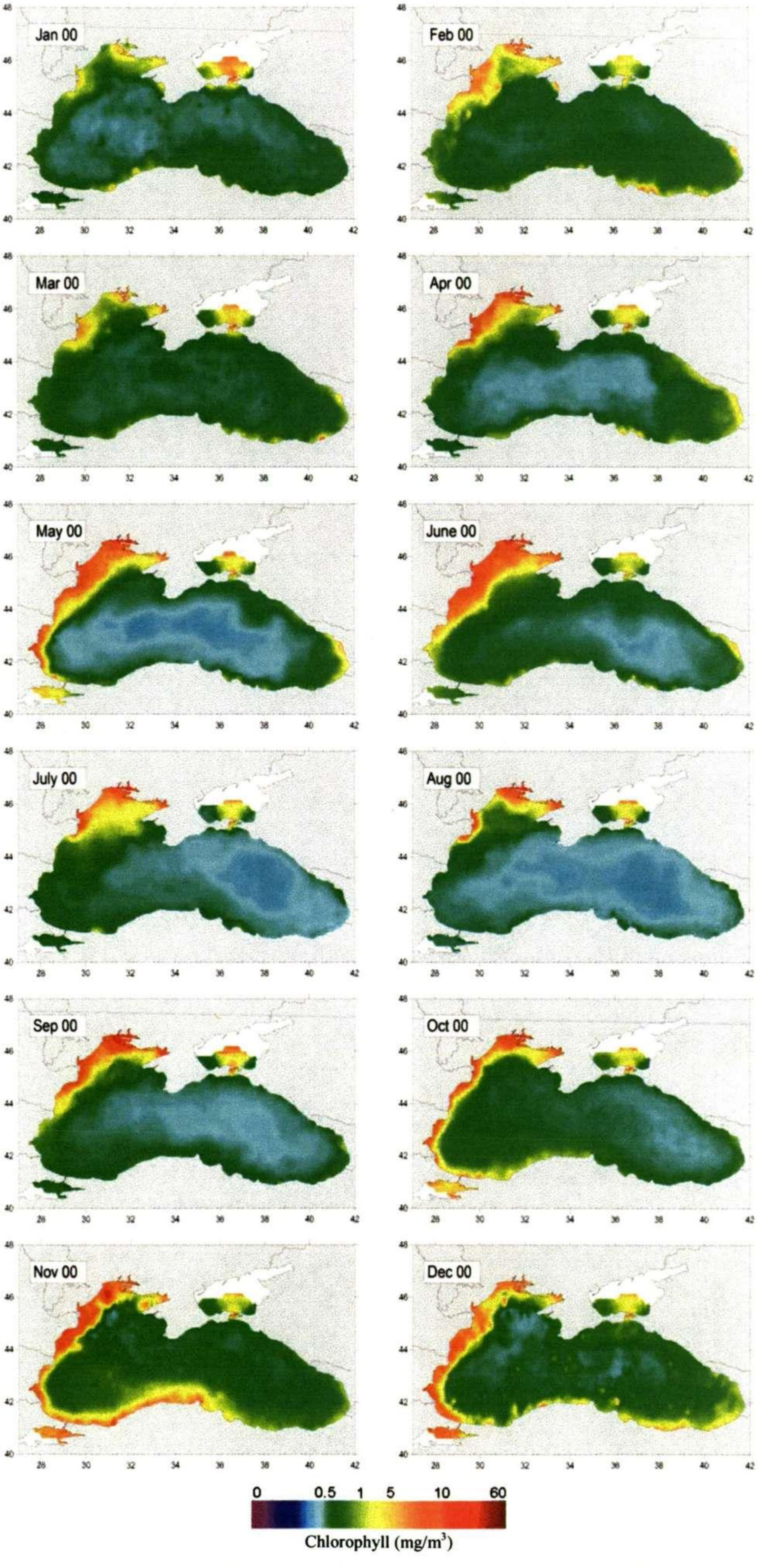


Figure A5. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2000.

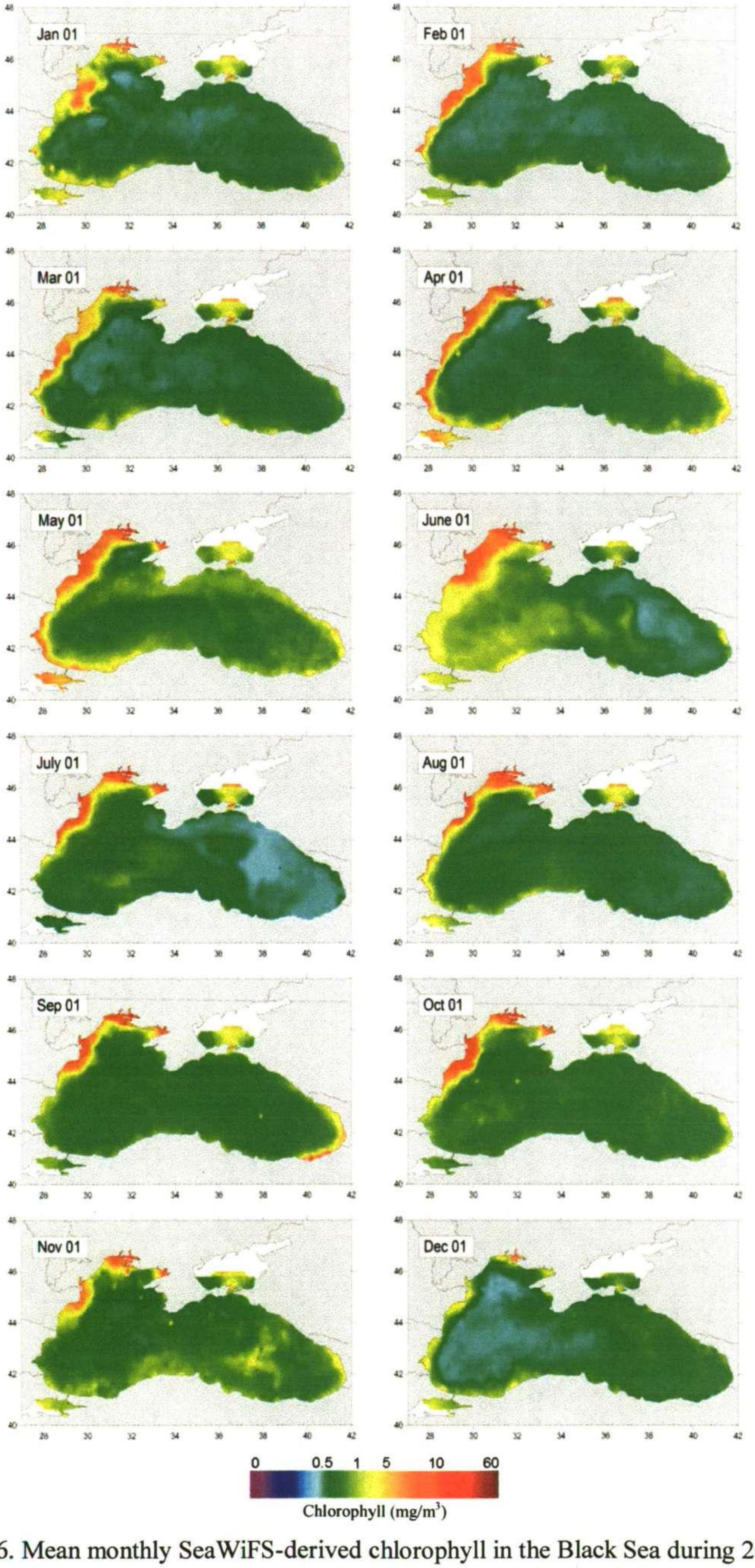


Figure A6. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2001.

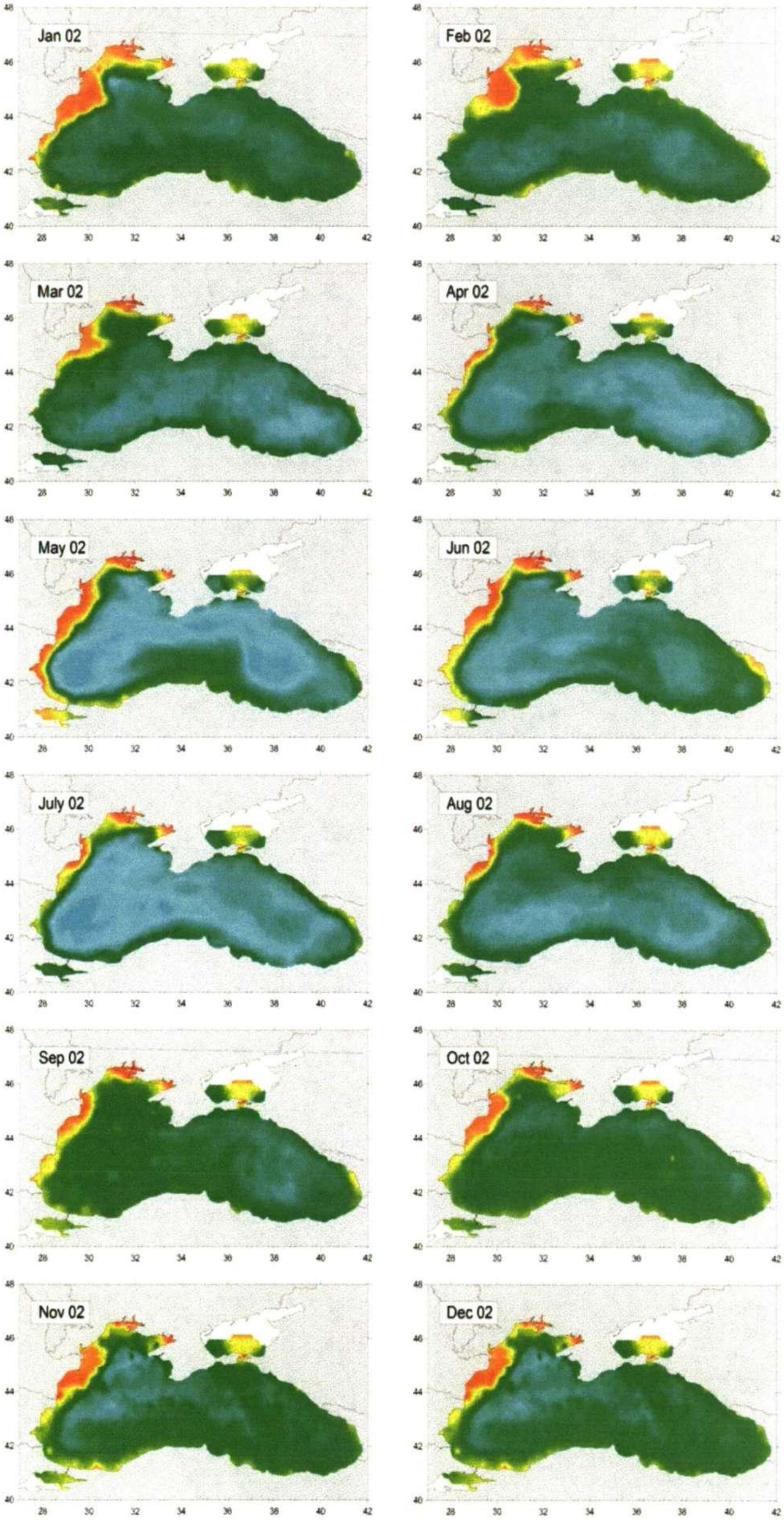


Figure A7. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2002.

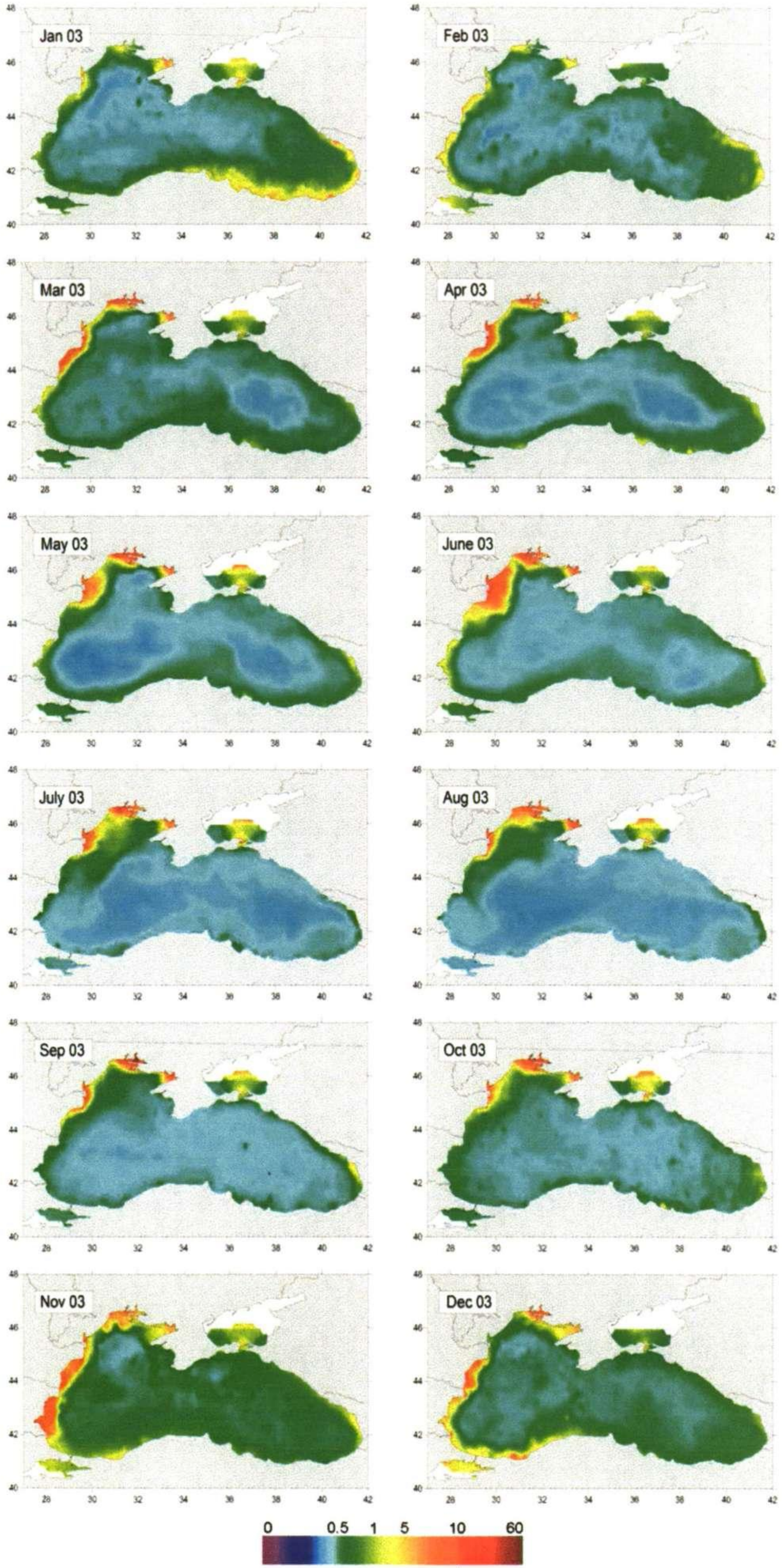


Figure A8. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2003.

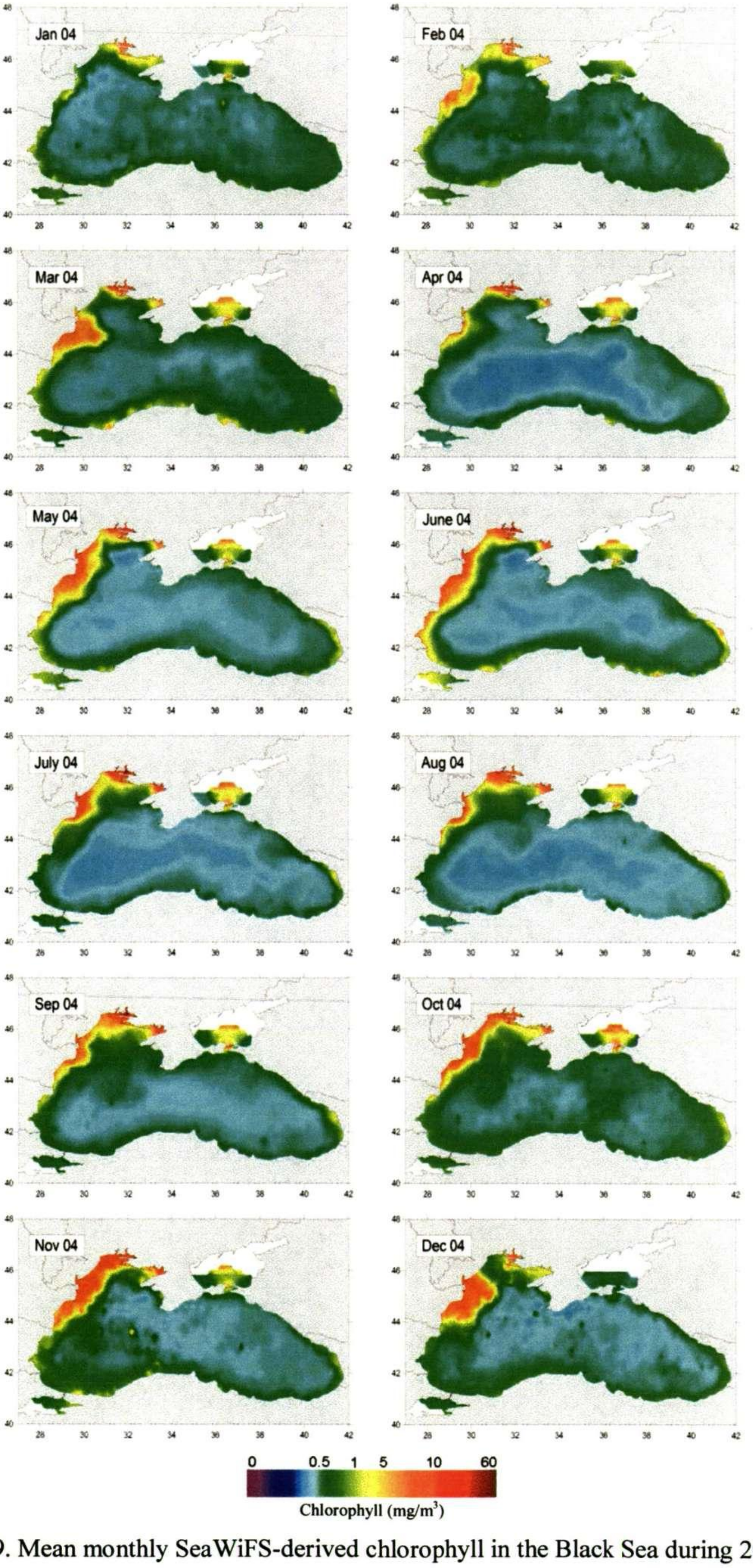


Figure A9. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2004.

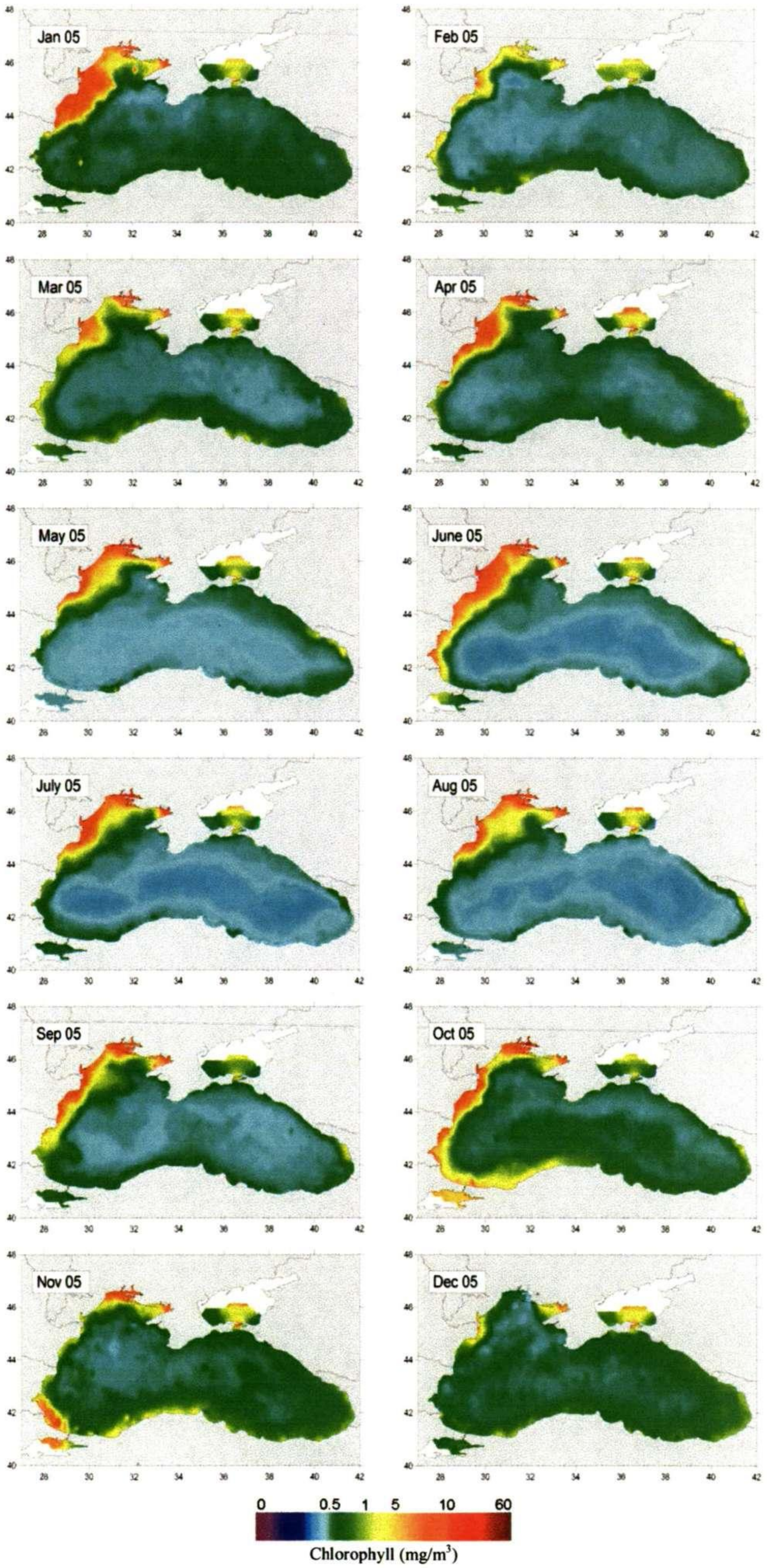


Figure A10. Mean monthly SeaWiFS-derived chlorophyll in the Black Sea during 2005.

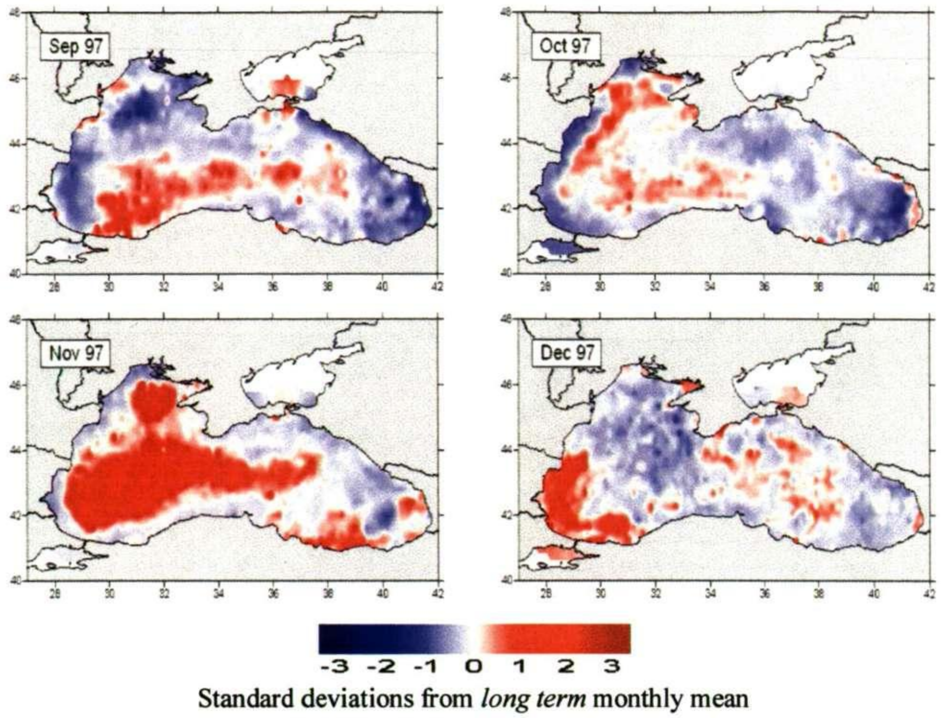


Figure A11. Black Sea standardized chlorophyll anomalies during 1997.

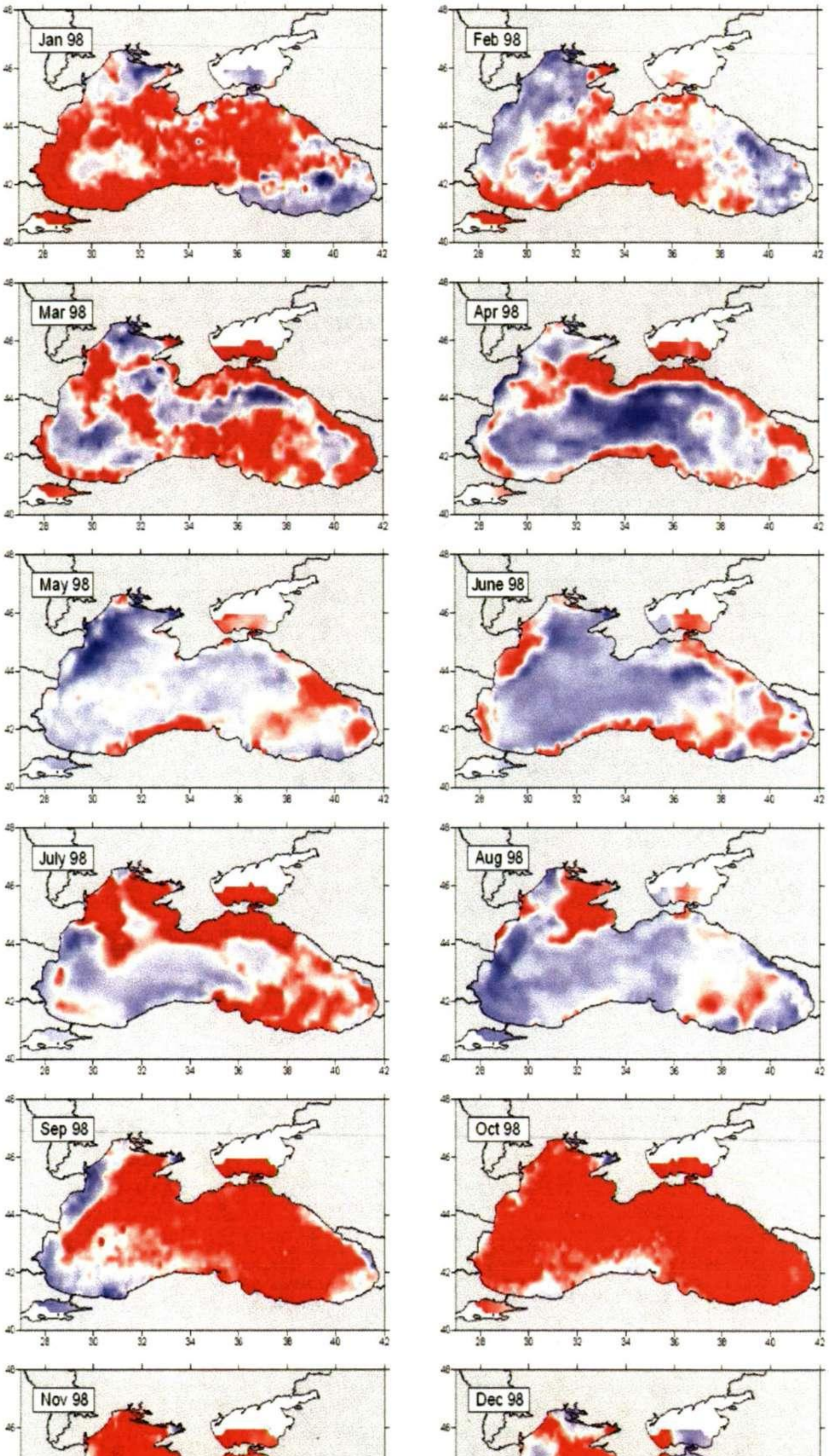


Figure A12. Black Sea standardized chlorophyll anomalies during 1998.

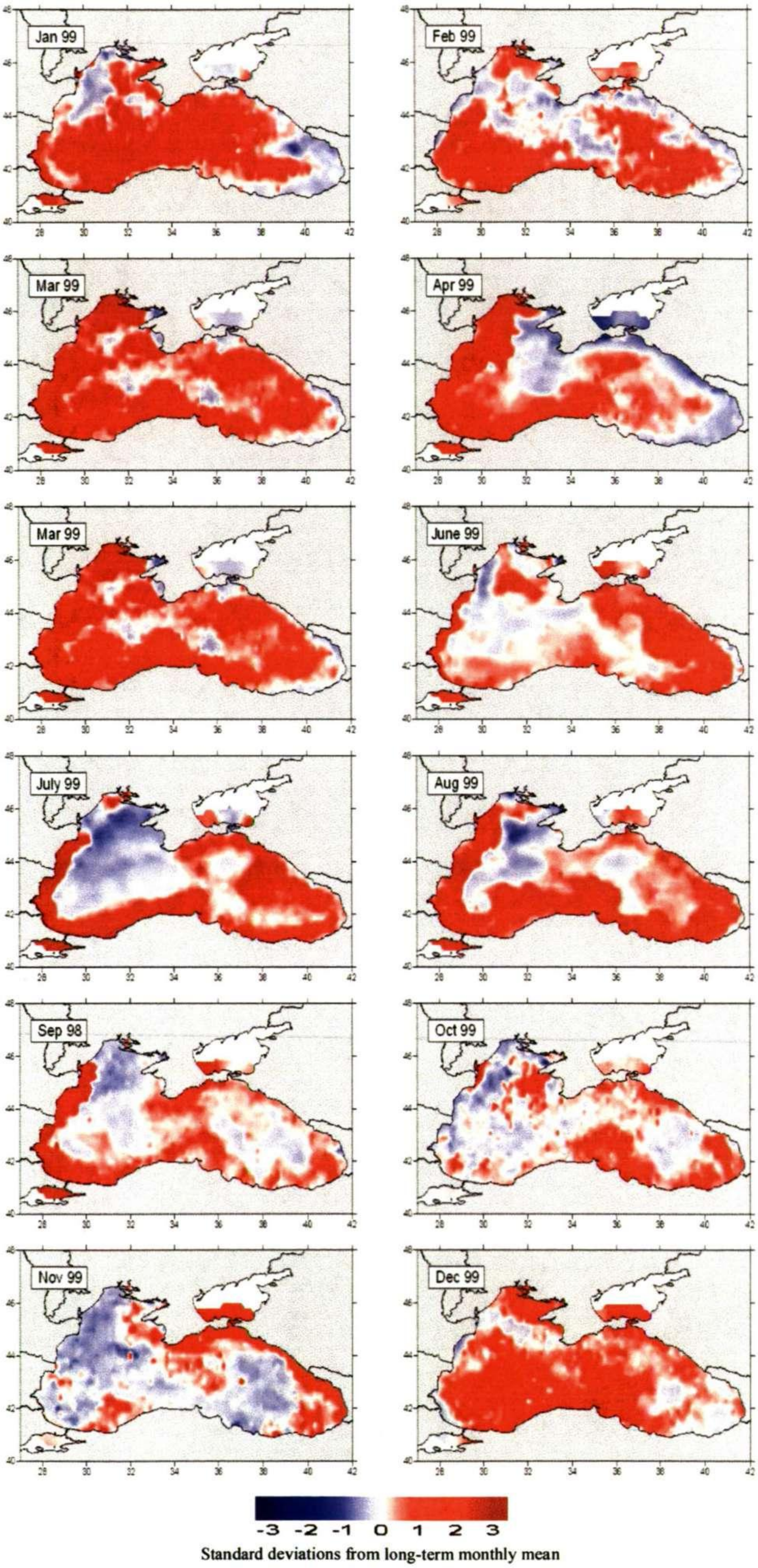


Figure A13. Black Sea standardized chlorophyll anomalies during 1999.

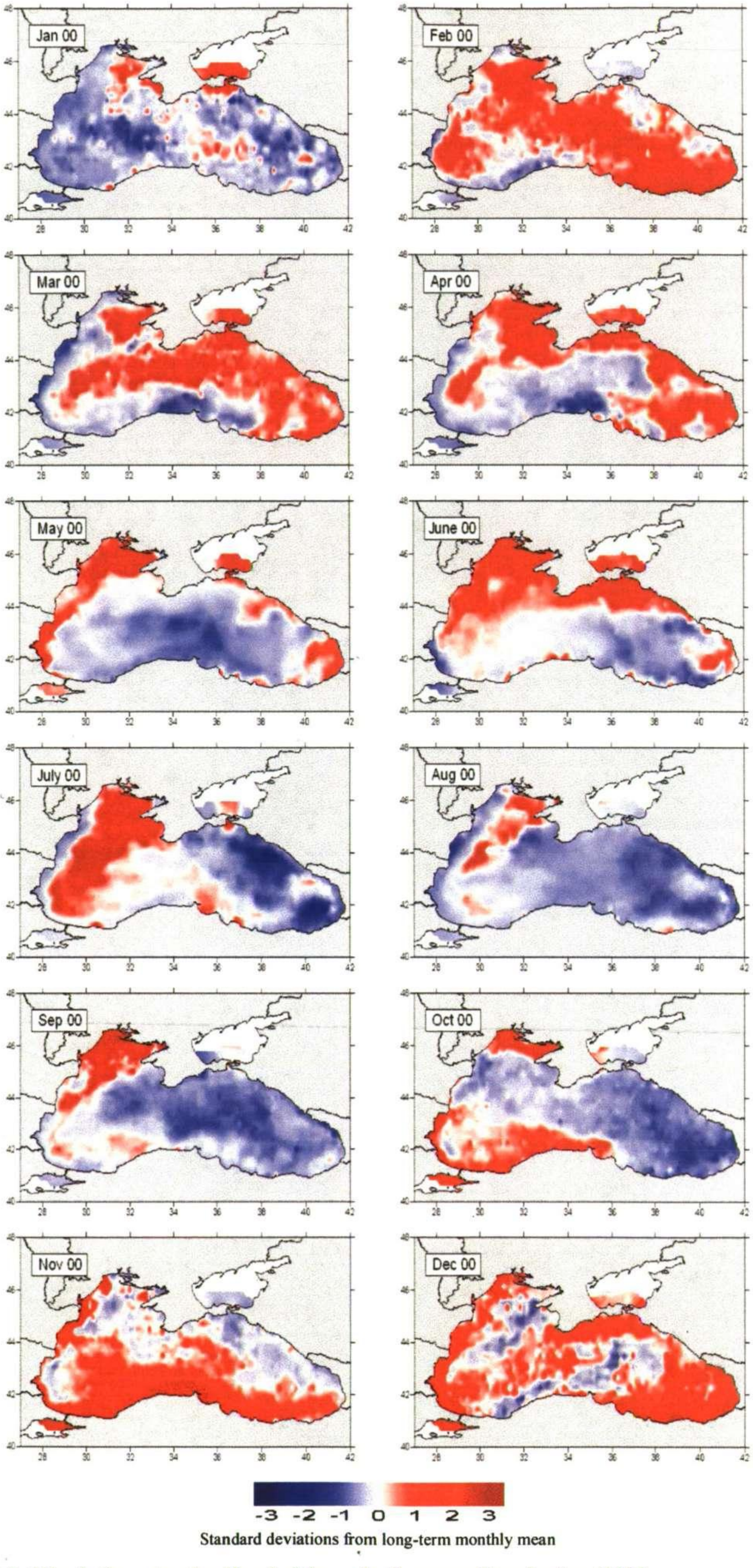


Figure A14. Black Sea standardized chlorophyll anomalies during 2000.

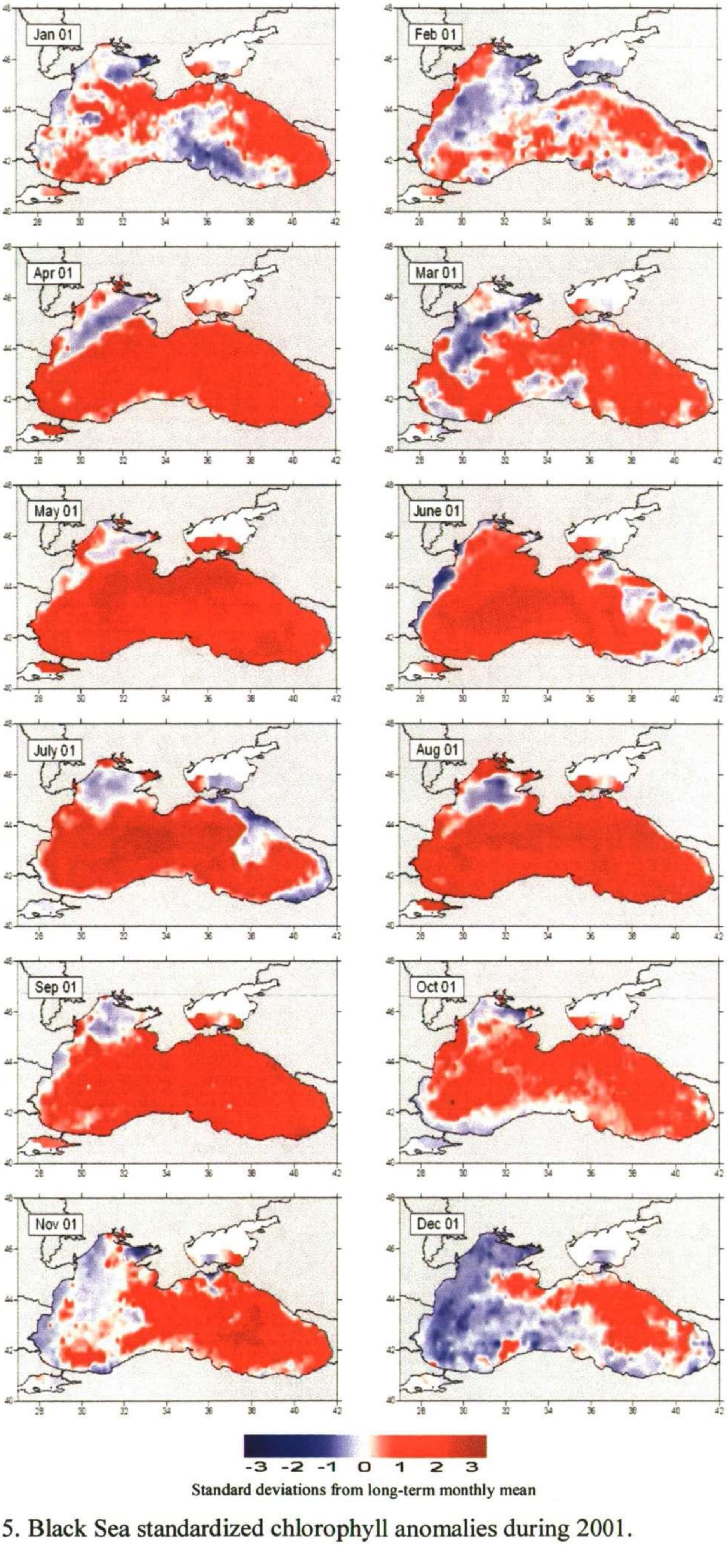


Figure A15. Black Sea standardized chlorophyll anomalies during 2001.

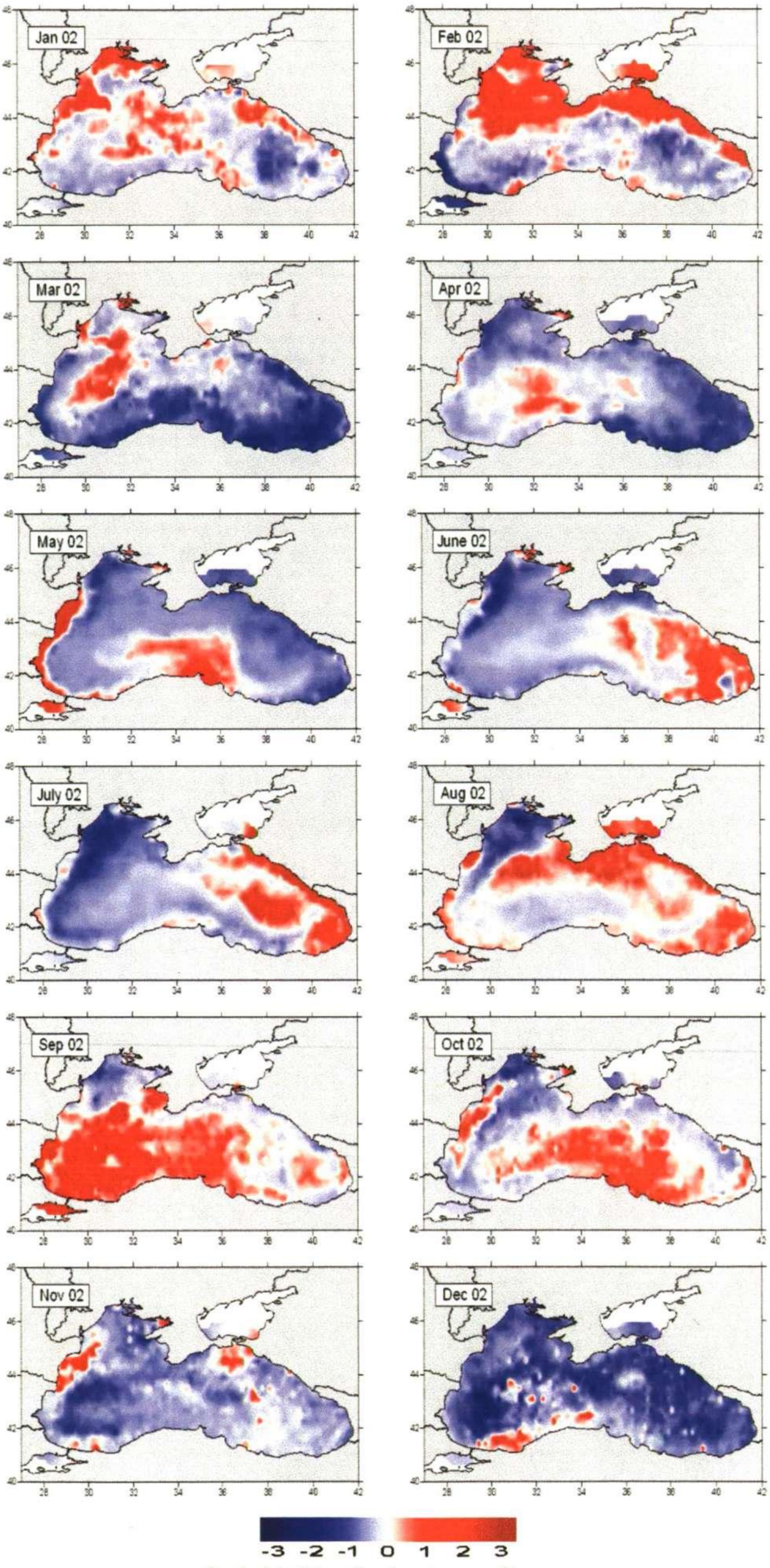


Figure A16. Black Sea standardized chlorophyll anomalies during 2002.

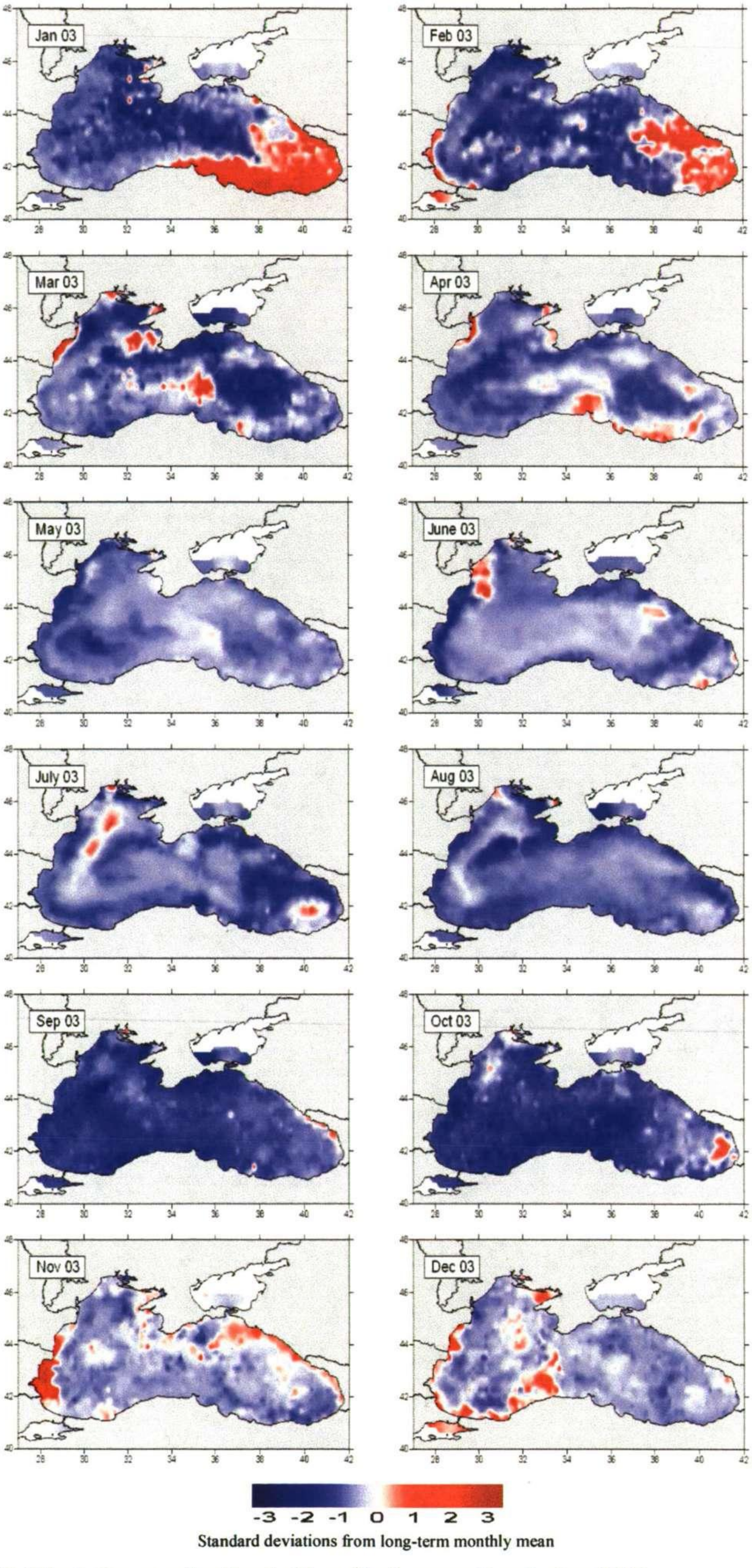


Figure A17. Black Sea standardized chlorophyll anomalies during 2003.

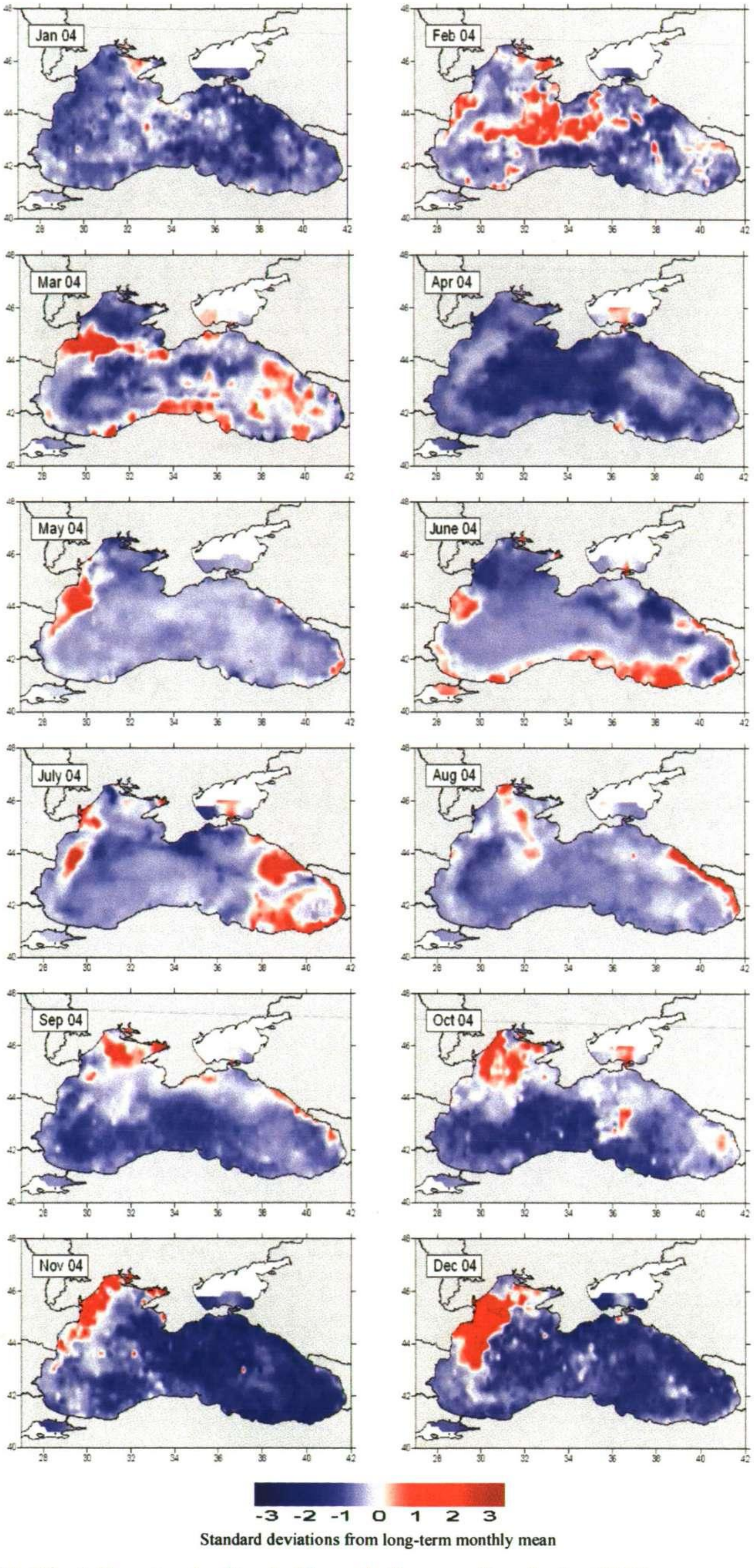


Figure A18. Black Sea standardized chlorophyll anomalies during 2004.

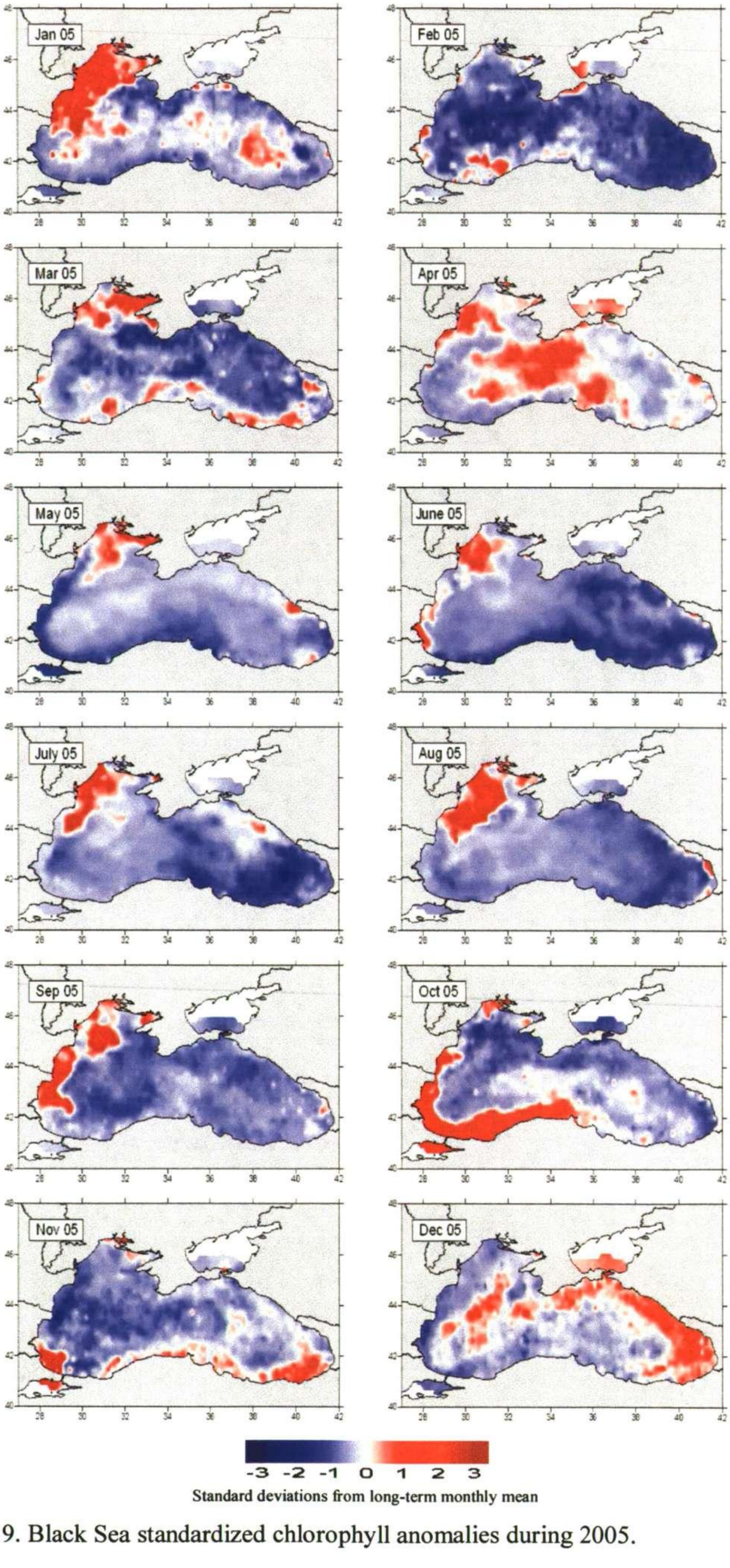


Figure A19. Black Sea standardized chlorophyll anomalies during 2005.

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