Ecological Management Strategies for Impounded Harbours

by

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Abstract

Long-term physical, chemical and biological monitoring (May 1990 to November 1994) was conducted in order to quantify water and sediment quality changes arising from the impoundment of Sutton Harbour, Plymouth (UK). Spore tracer studies revealed that 95 % water renewal times increased from 45 h to 72 h after impoundment. Semi-diurnal (tidal) salinity variations of *circa* 5 $\times 10^{-3}$ were observed, revealing a mechanism which shunts contaminated estuarine water into the harbour during flood tides. Salinity typically varied from 17 $\times 10^{-3}$ to 34 $\times 10^{-3}$ seasonally, and exhibited strong inverse correlations with total oxidised nitrogen and orthophosphate, demonstrating the riverine source of dissolved nutrients. These varied seasonally in concentration by 2 to 3 orders of magnitude. Impoundment restricted the flux of riverborne nutrients but greater retention of brackish bottom waters produced a stronger concentration gradient, resulting in possible nutrient storage by diffusion into the porewaters. Sewage outfalls and sediments were the main sources of ammonium. Following impoundment, the evidence suggests that a balance between nutrients from reduced external (riverine) fluxes and increased internal (porewater) fluxes has developed. Phytoplankton blooms were regular but short-lived features in summer, and continued after impoundment. Sewage contamination, with faecal coliform bacteria occasionally exceeding 30,000 cfu 100 ml^{-1}, improved unequivocally after impoundment, but stricter controls on internal sources are required. The permanently anoxic harbour sediments, consisting mainly of silt, contained Cd (1.8 $\mu g\ g^{-1}$), Cu (160 $\mu g\ g^{-1}$), Hg (1.2 $\mu g\ g^{-1}$), Pb (200 $\mu g\ g^{-1}$) and Zn (290 $\mu g\ g^{-1}$) in the <63 μm fraction. Sedimentary Cu, Pb and Zn concentrations increased during the monitoring period. The benthic macrofauna consists mainly of polychaete worms, with species diversity decreasing during construction, and then attaining a new, impoverished equilibrium after impoundment. Multivariate analysis revealed changes in community structure involving loss of sensitive taxa and appearance of opportunists. The ecological impact of impoundment was minimal, in that the harbour ecosystem was able to withstand the imposed environmental stresses. The management strategy adopted will ensure that water and sediment quality are maintained in Sutton Harbour; recommendations equally applicable to future harbour impoundment projects.

Contents

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Declarations

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for any other University award.

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A programme of advanced study was undertaken which included guided reading in marine environmental issues, a development of proficiency in the wide-ranging water industry Blue Book analyses of dissolved and particulate trace metals, faecal indicator bacteria, chlorophyll *a*, BOD_5 and grain size analysis, and training in the use of flame and flameless atomic absorption spectrophotometry and inductively coupled plasma mass spectrometry.

Relevant scientific seminars and conferences were regularly attended at which work was often presented; external institutions were visited for consultation purposes, and several papers prepared for publication.

Data Availability

The original data collected during this study are available as spreadsheets created using Microsoft Excel 5.0 for Windows and are stored on 3½" diskettes compatible with IBM PC's. The diskettes are contained in an appendix to copies of this thesis held by the University of Plymouth Library.

Publications

Smith, J. A., Millward, G. E. and Babbedge, N. H., 1995. Monitoring and management of water and sediment quality changes caused by a harbour impoundment scheme. *Environment International*, **21**: 197-204.

Smith, J. A., Millward, G. E., Babbedge, N. H., Attrill, M. J. and Jones, M. B., 1996. An assessment of changes in benthic community structure caused by construction of a harbour impoundment scheme. *Netherlands Journal of Aquatic Ecology*, in press.

Conferences and Presentations

Oral presentation entitled "*Perturbation of seasonal water quality trends in a tidally-defended UK Harbour*" at an Institute of Environmental Science and Institute of Physics joint meeting, **Inland and Coastal Water Quality 1993 - Measurement and Modelling**, held at Warren Springs Laboratory, Stevenage, UK, 29th September 1993.

Oral presentation entitled "*Water quality management of a tidally-defended UK harbour*" at the 12th Biennial International Estuarine Research Federation Conference, **The Science and Management of Coastal Estuarine Systems**, held on Hilton Head Island, South Carolina, USA, 14th to 18th November 1993.

Oral presentation entitled "*Monitoring and management of water and sediment quality changes caused by a harbour impoundment scheme*" at ECSA-24, **Northern and Southern Estuaries and Coastal Areas**, held at the University of Aveiro, Portugal, 5th to 9th September 1994.

Three oral presentations of current work to CONAPEG (Chemistry of Natural and Polluted Environments Group) and Marine Biology Research Group, University of Plymouth, September 1992 to July 1994.

One-day seminar entitled "*Environmental aspects of harbour design*" at Hydraulic Research Ltd., Wallingford, UK, 6th October 1994.

Meetings and external visits

Numerous scientific and coordination meetings with NRA personnel and Sutton Harbour Commissioners to discuss progress and to obtain advice.

Meeting with Mr. Tim Mason, Manager of Port Solent Marina, Portsmouth, UK, to discuss his experience of environmental management of an impounded marina development, August 1994.

Meetings with scientific staff of Plymouth Marine Laboratory, Plymouth, UK, to discuss approaches to nutrient analysis and modelling of water quality in Sutton Harbour.

Signed James A Smith

Date 16/12/95

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Chapter One

Introduction

Chapter 1 - Introduction

Sutton Harbour in Plymouth (UK), impounded in 1994 for tidal defence and operational purposes, has provided the opportunity for a novel three year study into the environmental effects of marine civil construction and impoundment on an anthropogenically modified marine ecosystem. Initially, this chapter examines the vital importance of harbours to local and national economies, the environmental issues arising from their shared use and the increasing future need to protect harbours from rising sea levels. Current environmental legislation with a bearing on harbour operation and water quality is then discussed, and experiences from other impounded saline ecosystems are reviewed. Finally, the case of Sutton Harbour is examined in detail: The regional setting is described in economic, geographic and environmental terms and the issues that led to impoundment are discussed. The ambitious civil engineering works are described, together with the environmental implications of impoundment. The chapter concludes with the detailed aims and rationale of the current study.

1.1 Ports and Harbours as a Commercial Resource

Ports and harbours have a great commercial value both globally and in the UK. They are essential as safe anchorages for cargo transfer and are often popular centres for property development and itinerant businesses (Couper, 1989). The term port implies a large commercial installation serving bulk carriers, tankers, cargo and large passenger vessels, whilst the term harbour generally means a smaller haven with docks for the local fishing industry and services for recreational sea users. However, ports can often be considered as constituting a multitude of smaller harbours in proximity. Harbours have historically attracted tourists and are often centres for this industry in many UK coastal cities and towns. This shared use can result in conflicting interests. In badly managed harbours, the presence of industry and commerce may cause visually obtrusive pollution from uncontrolled discharges and discarded materials. Tourism and recreational use of the harbour and quayside can be hazardous and unwelcome in a busy industrial or commercial setting. However, with good management these potential conflicts are often resolved.

1.1.1 Environmental issues

Public awareness of environmental issues increased considerably during the 1980's, a trend which continues today. Attitudinal surveys reported by the DoE (1992) in England and Wales show that environmental issues have an increasingly high profile: During the 1970's, environmental and pollution issues were considered important by between 1 % and 7 % of the population. By 1989, this figure had increased to 35 % and Environment was one of the most important issues alongside the National Health Service and Poll Tax. In 1988, 67 % of the UK population felt that pollution and environmental damage was an urgent and immediate problem faced by the EC. In addition, there have been changes in concern about specific issues: Between 1971 and 1989, the proportion of respondents saying that pollution of seashore and beaches was "very serious" has increased from 46 % to 77 %, and pollution of water from 50 % to 76 %. These two issues were of greatest concern in England and Wales in 1989 and in Scotland in 1990, ranking above nuclear waste, the ozone layer and global warming. On a local level, 67 % of people in 1989 identified litter and rubbish as an important problem in their area, with 14 % saying it was the most important issue. It is clear from these figures that action to limit pollution and environmental damage is an increasing component of public opinion that must be addressed by all sectors of industry and commerce. This is further highlighted by increased membership of environmental organisations in recent years: Greenpeace UK membership rose from 30,000 in 1985 to 372,000 in 1990, whilst Friends of the Earth membership (in England and Wales) rose from 18,000 in 1981 to 110,000 in 1990 (DoE, 1992).

Many environmental issues associated with harbour use stem from contaminant sources on land and water. Sewage treatment in coastal areas has traditionally been achieved through discharge directly to the sea and it has long been acknowledged that these short sea outfalls return sewage-contaminated water to the shore on a rising tide or favourable wind (Morris, 1991). However, it has recently been established that bathing in sewage-contaminated water increases the risks of gastrointestinal disease, ear, throat and skin infections (Cabelli *et al.*, 1982; Cabelli, 1989; Balarajan *et al.*, 1991; Kay and Jones, 1992) particularly in the age group up to four years (Fattal *et al.*, 1987). Sewage outfalls are often located near harbours because of their mutual association with coastal centres of population. Combined with direct sewage inputs from vessels inside the harbour, this can lead to poor microbiological water quality and increased risk of illness to those in contact with the water. The problems are often exacerbated by seasonal population growth and increasing recreational use of the sea.

The sheltered nature of harbours often results in low water velocities and a net import of fine sediments, particularly when the harbour is situated in or near an estuary. The fine sediments possess a high specific surface area (De Groot *et al.*, 1976) and therefore carry relatively high concentrations of organic and inorganic micropollutants (Olsen *et al.*, 1992) including trace metals (Millward and Turner, 1995) and polychlorinated organic compounds (Tyler *et al.*, 1994). Other pollutants in the water column (for example, hydrocarbons from diesel and oil spillage, bilge pumping and road runoff, Cu from antifouling paints (Claisse and Alzieu, 1993) and Zn from sacrificial anodes and galvanised structures) form various pollutant-particle associations and are transported to the sediments. Thus, harbour sediments are often highly contaminated, posing problems for their disposal (Fraser, 1993) owing to the frequent requirement for operational dredging of net accretions.

Semi-enclosed bodies of water such as harbours and estuaries in limited interplay with the open sea are identified by GESAMP (1990) as prone to anthropogenically induced eutrophication or nutrient enrichment (specifically with inorganic and organic forms of nitrogen and phosphorus) chiefly from sewage and detergents, agricultural run-off and industrial effluents. The many stages of the eutrophication process are summarised in Fig. 1.1. The deleterious effects of eutrophication, apart from the eventual loss of all benthic macrofauna, are manifold. Fouling growth of macroalgae may increase on pontoons, walls and particularly on vessel hulls, thus encouraging the use of antifouling treatments that may pollute. *Phaeocystis* blooms may occur which produce visually obtrusive foam and unpleasant odours (Lancelot *et al.*, 1987). Potentially more serious to harbours is the proliferation of a number of toxin-producing algal species that can cause Neurotoxic (NSP), Paralytic (PSP) and Diarrhoeic (DSP) Shellfish Poisoning (GESAMP, 1990). Various shellfish, particularly bivalves, kept alive by storage in mesh sacks in the harbour, may concentrate the toxins and cause these poisonings in the consumer. Toxic blue-green algae (cyanobacteria) are a potential cause of illness associated with contact or ingestion of freshwater that contains cyanobacterial toxins (Fewtrell and Jones, 1992). They may also proliferate in warm, brackish waters such as those found in estuaries and

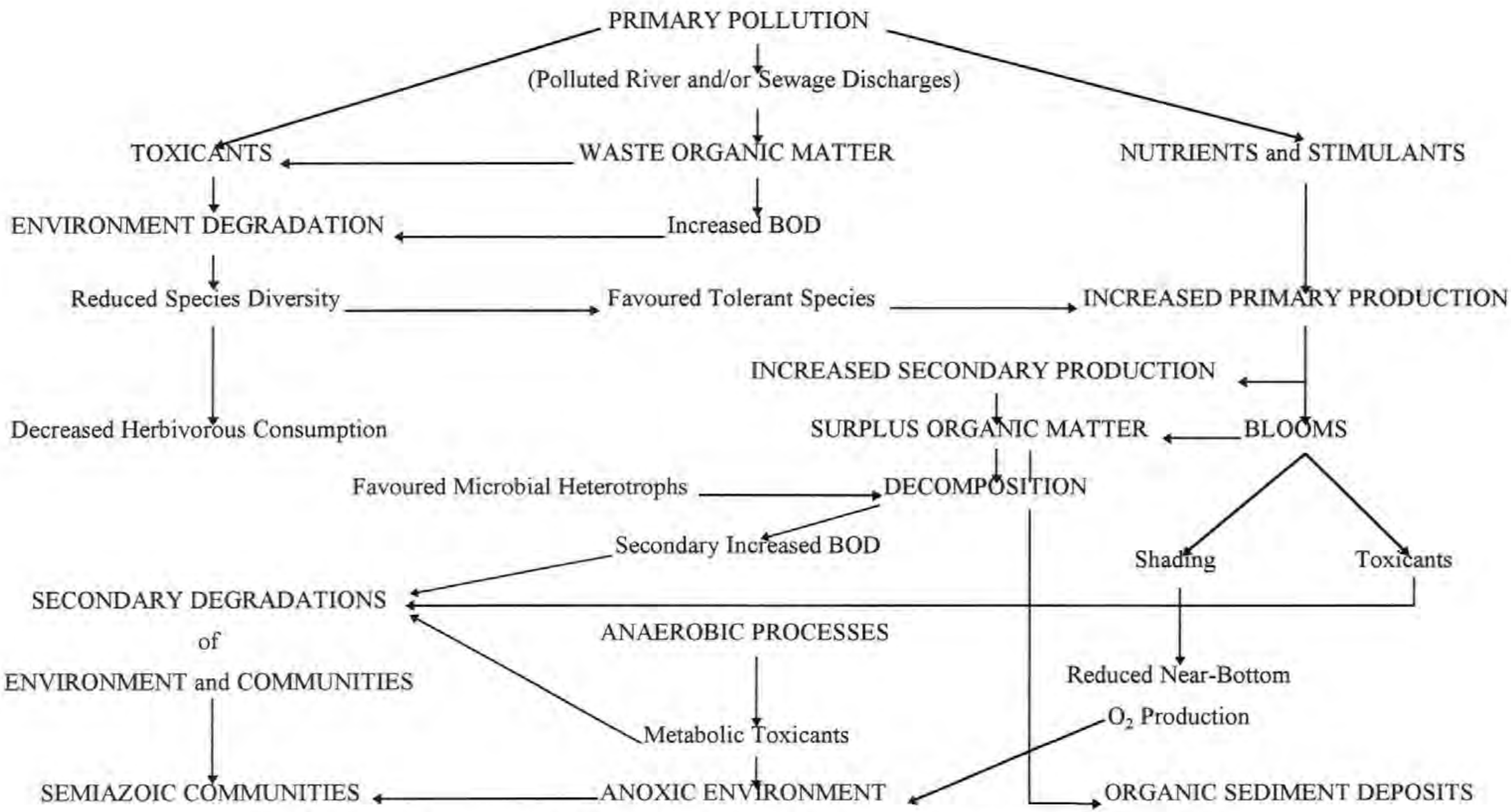


Fig. 1.1. Schematic representation of eutrophication processes in the marine environment (after GESAMP, 1990).

harbours during the summer. Clearly, preventative action to reduce the concentrations of nitrogen and phosphorus and thus the risk of eutrophication is desirable.

The issue of amenity is often a prime consideration in the management of harbours that attract tourism and recreation. Litter from the quayside, from vessels and from outside the harbour can accumulate in areas dependent on wind direction and become visually obtrusive. Aesthetic quality is generally poorest at low water (LW), when the intertidal zone is exposed. Anoxic mud may emit sulphide odours, whilst rotting macroalgae and litter can add to the problem. Litter accumulating above the HW mark is the responsibility of the land owner (such as the Harbour Authority) or the Local Authority. However, no clear responsibility exists for litter below the HW mark. In many instances, the Harbour Authority, marina operators or Local Authority extend their operation into this zone within the confines of the harbour, but they cannot legislate against waterborne litter entering the harbour by wind, wave or tidal action. The 'Polluter Pays' principle does extend to refuse from shipping through annexe V of the MARPOL Convention 1973/1978 (Howarth, 1988), but is difficult to enforce in areas of high vessel density.

1.1.2 The need for tidal defence

Modern trading ports have evolved in areas that afford deep water to prevent grounding when ships are at berth. Shallow waters are generally encountered in the approaches to these ports, so that regions with a semi-diurnal macrotidal regime, such as that around the UK, offer two periods per day when access to the ports is facilitated. However, this applies only for large vessels with a turnaround time of days, whereas it may be a distinct disadvantage to smaller fishing vessels and yachts. Harbours tend to be more shallow and many partially dry at LW, thus presenting a problem to the small boat operator. The solution to this problem for many harbours is impoundment, whereby a lock structure is constructed at the mouth, behind which an operational head of water can be maintained inside the harbour. This has the added advantage of increasing the amenity and hence the commercial value of the harbour and surrounding properties.

In future there will be a growing need for coastal defences to protect valuable land against flooding due to the increase in mean sea level (MSL) associated with climatic change. Tidal barrages will become essential both in harbours and on a larger scale across entire estuaries. One such large-scale barrage, the Thames Barrier, was completed near Woolwich in 1982 and now protects large areas of London from the flooding that would otherwise occur during exceptional surges and tides (Couper, 1989), particularly the 100-year wave. Barrage closure is triggered by the Storm Tide Warning System (STWS), a network of coastal tide gauges reporting to the Meteorological Office in Bracknell and established after catastrophic flooding from the Humber to the Thames in February 1953 (DoE, 1992). At present, world oceans are rising at a rate of 1 to 2 mm a^{-1} due to eustatic changes (continental drift and ocean basin evolution) and thermal water expansion due to global warming (Hansen, 1985). Further changes due to isostatic readjustment of the UK land mass after the last ice age (Pethick, 1993) and gradual subsidence of the North Sea basin margins (DoE, 1992) both increase the observed rise. These changes are manifest as a north-south tilt in the UK: analysis of tidal records shows that MSL at Aberdeen is falling at 0.1 mm a^{-1}, whilst MSL at Newlyn is rising at 2.2 mm a^{-1} (George, pers. comm.). Moreover, the effects of accelerated global warming from increases in atmospheric greenhouse gas concentrations have been estimated by the Intergovernmental Panel on Climate Change (IPCC) to result in an 18 cm global MSL rise by 2030 (Warrick and Oerlemans, 1990). The increases in MSL will alter the tidal range in areas of converging coastline such as the Bristol Channel, by tuning or detuning the local tide to the 12.5 h semi-diurnal regime, thereby increasing or decreasing the locally observed tidal range (Dyer, pers. comm.). The overall effects of sea level rise may be exacerbated by possible increases in storm frequency in the Atlantic and hence altered wave climate along the exposed western coasts: Carter and Draper (1988) suggest that the north east Atlantic has become considerably rougher during the last 30 years. All of these factors will combine to ensure that an increasing number of harbours will require tidal defence by impoundment in the future, thereby introducing associated problems of pollution dispersal.

1.2 Environmental Legislation Concerning Ports and Harbours

Ports and harbours in the UK are covered by a plethora of legislation at the European Union (EC) and national levels that have evolved to safeguard the environment. Much of the national legislation has been revised in the last decade in line with EC Directives, which are binding as to the desired results but leave the choice of form and methods to each member state (Haigh, 1990).

1.2.1 European Union legislation

The Discharge of Dangerous Substances Directive (CEC, 1976a) proposes a framework for the elimination or reduction of pollution in controlled (inland, coastal and territorial) waters. The Directive sets out Lists I and II of dangerous substances on the basis of toxicity, bioaccumulation and persistence, largely in line with the black and grey lists of the Paris Convention (PARCOM) adopted in 1974 (Howarth, 1988). List I includes Cd and Hg compounds that should be eliminated from the water environment. However, elimination of pollution does not mean zero concentrations, but rather zero effects (Haigh, 1990). List II contains less dangerous substances such as Zn, Cu and Pb compounds that should be reduced. There are a number of Daughter Directives adopted between 1982 and 1990 that set emission limits and water quality objectives for specific List I substances (Couper *et al.*, 1994).

The Bathing Water Directive (CEC, 1976b) sets out a framework intended to raise the quality of Designated Bathing Waters (fresh or sea water in which bathing is explicitly authorised or not prohibited and traditionally practised by large numbers of bathers) by imposing values for nineteen physical, chemical and microbiological parameters. In the UK, the most important of these are total coliform (TC) and thermotolerant faecal coliform (TFC) bacteria indicative of sewage contamination. Fortnightly samples during the bathing season must exhibit 95 % compliance with imperative (I) values (10,000 cfu 100 ml^{-1} TC; 2,000 cfu 100 ml^{-1} TFC) and 80 % compliance with more stringent guideline (G) values (500 cfu 100 ml^{-1} TC; 100 cfu 100 ml^{-1} TFC). The deadline for compliance with this legislation was 1985. By 1990 the European Commission had begun infringement proceedings against certain UK waters (Haigh, 1990).

No harbours are included in the 455 UK bathing waters identified for monitoring since 1985 (Kay and Jones, 1992) although many are in proximity. The Urban Waste Water Treatment Directive (CEC, 1991) is therefore more pertinent to harbour water quality. This defines minimum standards for effluent treatment based on organic load and the nature of the receiving waters, defined as sensitive and less sensitive by the Member States (Couper *et al.*, 1994). The first deadline for compliance with this Directive is 31/12/2000. Harbour waters will benefit from the implementation of this Directive owing to their general proximity to population centres.

The final main body of European environmental legislation concerning harbours is the Environmental Assessment Directive (CEC, 1985) that requires an environmental impact assessment to be carried out before consent is granted for development projects. Annex II of this Directive covers infrastructure, including flood relief works and yacht marinas.

1.2.2 UK national legislation

The Water Act 1989 (DoE, 1989b) allowed privatisation of the 10 Water Authorities and established the National Rivers Authority (NRA) as the competent authority for control of waterborne pollution in England and Wales. The NRA has jurisdiction over controlled waters from inland waterways to coastal waters extending three miles offshore and is responsible for ensuring that water quality criteria are met by consenting discharges, by planning schemes for effective effluent dispersion and by implementing large scale monitoring programmes. In particular, the NRA ensures that improvements made to coastal sewage treatment plants by the privatised Water Utility Companies meet with stringent requirements of the EC Bathing Water and Urban Waste Water Treatment Directives.

The Environmental Protection Act 1990 (DoE, 1991) empowered Her Majesty's Inspectorate of Pollution (HMIP) to apply the policy of Integrated Pollution Control (IPC) to prescribed processes having the potential to discharge significant quantities of harmful waste to any environmental medium. Authorisations are given subject to the operator employing Best Available Techniques (including technology) Not Entailing Excessive Cost (BATNEEC). Discharges to water must still satisfy stipulations required by the NRA.

There is no current UK legislation to cover dredging, although unlicensed dumping of dredged spoil or other waste on the UK Continental Shelf Limits is an offence under section 146 of the Environmental Protection Act 1990 that carries a maximum fine of £50,000. Licences to dump dredged spoil are issued by the Ministry of Agriculture, Fisheries and Food (MAFF) under part II of the Food and Environmental Protection Act 1985 (Howarth, 1988). This legislation also applies to building materials deposited below mean high water springs (MHWS), particularly where a construction project will disturb historically contaminated sediments. Licensing is generally conducted on the basis of criteria in the London and Oslo/Paris Conventions, but only after all alternatives to criteria in the London and Oslo/Paris Conventions, but only after all alternatives to disposal at sea have been considered by the operator (Vivian, 1994). Alternatives used in the US include Contained Aquatic Disposal (CAD) cells, where trenches are dug in the harbour bed to contain the contaminated sediments and are then capped off with clean sediments, or Confined Disposal Facilities (CDF), where diked areas along the shore are filled with contaminated sediments to reclaim land (Fraser, 1993).

The EC Environmental Assessment Directive is implemented for UK harbours by two statutory instruments (DoE Welsh Office, 1989): the Harbour Works (Assessment of Environmental Effects) Regulations 1988 (SI 1988 No. 1336) and the Harbour Works (Assessment of Environmental Effects No. 2) Regulations 1989 (SI 1989 No. 424).

The most current UK environmental legislation is the Environment Bill 1994 (DoE, 1994) that proposes to establish an Environment Agency for England and Wales (Scottish Environment Agency for Scotland). Powers of HMIP, NRA and the Waste Regulation Authorities will be transferred to the Agency, thereby creating the first fully integrated body for the control and regulation of all forms of pollution and further facilitating the use of established principles such as 'Polluter Pays', IPC and BATNEEC for the good of the environment at large.

1.3 Impounded Saline Ecosystems

Despite the extensive legislation, there is a dearth of published studies of the impact of impoundment on harbour waters, and relatively little data available for impounded marine ecosystems in general. The main body of evidence comes from the Netherlands, where for example The Eastern Scheldt, as an integral part of the SW Netherlands Delta Plan, is the only widely researched example of an impounded saline water body. However, Nienhuis *et al.* (1994) list the Thames Barrier in London, the Nieuwe Waterweg Barrier near Rotterdam, the 25 km-wide Neva River Barrier near St. Petersburg and 60 years of flood protection in the Venice Lagoon as European systems that can be permanently or temporarily impounded. Furthermore, proposals for Tidal Power Barrages in the Severn and Mersey Estuaries would have a similar effect for a different purpose (Couper, 1989). The impoundment of several UK Marinas will also be discussed, although there are no published studies of water quality in these locations.

1.3.1 The Eastern Scheldt (Oosterschelde) and the Netherlands Delta Project

Following disastrous flooding during a North Sea storm surge in February 1953, the decision was taken to dam 5 of the 7 estuaries in the SW Netherlands (van Westen and Leentvaar, 1988). However, following concerns over deleterious effects on the extensive aquaculture in the Oosterschelde (Elgershuizen, 1981; Smaal and van Stralen, 1990), plans for the impoundment were revised and a permeable storm surge barrier (SSB) was constructed at the mouth, with 2 compartment dams up-estuary that redirect seaward runoff elsewhere (Scholten *et al.*, 1990). The reduced freshwater inputs resulted in the creation of a permanently saline water body, with decreased tidal exchange with the North Sea, a situation similar to impounded harbour schemes although on a much larger scale. The Delta Region of the Netherlands is shown for reference in Fig. 1.2.

Fig. 1.2. Map of the delta area of the rivers Rhine, Meuse and Scheldt in the SW Netherlands (after Nienhuis and Smaal, 1994). Numbers refer to dam components of the Delta Plan (year of completion in brackets): 1 = Zandreekdam (1960); 2 = Veersegatdam (1961); 3 = Grevelingendam (1964); 4 = Volkerakdam (1969); 5 = Haringvlietdam (1970); 6 = Brouwersdam (1971); 7 = Markiezaatsdam (1983); 8 = Oosterschelde Storm Surge Barrier (1986); 9 = Philipsdam, (1987); and 10 = Oesterdam (1986).

In an early appraisal of the environmental effects of the scheme, Elgershuizen (1981) recognised that although the main purpose of the SSB was for protection of the low-lying land to the east, it would also be involved in active water quality management. The appraisal suggested that salinity in the Oosterschelde would decrease owing to greater residence time and despite a reduction in freshwater inputs. This would cause an associated rise in nutrient concentrations that would, in combination with enhanced settling and greater transparency, give rise to increases in primary production. Eutrophication might ensue in key areas, resulting in an eventual degradation of the valuable benthic macrofauna. Following commissioning of the scheme in 1987, ebb and flood tidal volumes were reduced by 30 % and the tidal range by 12 % (ten Brinke *et al.*, 1995). This caused an increase in residence time from 10 to 20 tides in the west (nearest the SSB and the North Sea) and from 100 to 175 tides in the east (nearest the Oesterdam) (Wetsteyn *et al.*, 1990). Mean freshwater flows decreased from 70 m^3s^{-1} to 25 m^3s^{-1}, representing less than 1 % of the tidal volume. Therefore the estuary was reclassified as a tidal bay (Smaal and Nienhuis, 1992) with high, stable salinities (ten Brinke *et al.*, 1995).

During the period before barrier closure, dissolved nutrient concentrations exhibited distinct seasonal cycles with minima in the late summer and maxima in the late winter. Chlorophyll *a* concentrations were typically 1 to 2 $\mu g\ l^{-1}$ in the winter and 20 to 40 $\mu g\ l^{-1}$ in the summer. Nitrogen was not considered to be a limiting nutrient (Wetsteyn *et al.*, 1990). Phytoplankton assemblages evolved with the spring, early summer and late summer turbidity/light gradient. During construction, the growth season began earlier and lasted longer, owing to lower current velocities and greater transparency coupled with unaltered freshwater/nutrient inputs (Bakker *et al.*, 1994). During this transitional period, Bakker *et al.* (1994) also noted the impact of short-term climatic conditions; several severe winters led to increases in summer primary production owing to the early establishment of *Biddulphia aurita*, a benthic/pelagic diatom with a high tolerance of low water temperatures. After construction, studies revealed that summer and winter nitrate and phosphate concentrations decreased and that nitrogen became the limiting nutrient (Wetsteyn and Kromkamp, 1994), balancing enhanced primary production from increased transparency (Scholten *et al.*, 1990). The altered light/nutrient/salinity regime led to a change in seasonal succession: the phytoplankton assemblage formerly only seen in the summer period formed earlier and persisted longer (Bakker *et al.*, 1994). Primary

(Wetsteyn and Kromkamp, 1994) and water quality more closely resembled the North Sea than before impoundment (Nienhuis *et al.*, 1994).

The construction of the SSB had a dramatic effect upon the sediment size distribution: the fine fraction increased from 60 % to 90 % in many areas, changing the system from a net exporter of sand to a net importer of fine material (ten Brinke *et al.*, 1995). This increased sedimentation and lower current velocities favoured suspension and deposit feeders, thus causing a change in community structure (Leewis and Waardenburg, 1990). However, it was suggested that *Mytilus edulis*, consuming 60 % of the particulate organic matter in the Oosterschelde (Scholten *et al.*, 1990) would be degraded in the western part due to a reduction in food import from the North Sea (Smaal and van Stralen, 1990) and preliminary data from 1987 confirmed this. However, continued study revealed that the biota showed a resilient response to the physico-chemical changes; productivity and hence the quality of the internationally renowned fishery was maintained (Smaal and Nienhuis, 1992) and the Oosterschelde continued to be a self-sustaining ecosystem despite the considerable anthropogenic alterations (Scholten *et al.*, 1990).

The outcome of major alterations to the large estuarine systems of the Delta Region is not always clear cut. One example is the Haringvliet to the north of the Oosterschelde (Fig. 1.2) which receives the combined discharges of the Rhine and Meuse rivers. The Haringvlietdam at its seaward end was completed in 1970, effectively sealing out saltwater intrusion from the North Sea whilst allowing discharge of excess freshwater *via* sluices (Knoester, 1983). Seventeen 56 m discharge sluices are opened to discharge river water at almost every period of low water, but can be restricted during periods of low Rhine/Meuse flows or whenever more fresh water is required elsewhere in the Netherlands (MTPW, 1988). For this reason, the Haringvlietdam is known colloquially as the stop-cock of the Netherlands, although the primary reason for its construction was the same as the Oosterschelde SSB: the protection of the valuable hinterland from flooding. Recently, the wisdom of creating a large freshwater lake supplied by the heavily industrialised and contaminated Rhine has been brought into question (G. T. M. van Eck, Rijkswaterstaat, pers. comm.). It appears that the scheme may need to be revised in the future, by allowing the incursion of North Sea water to exchange with the increasingly contaminated waters of the Haringvliet. This potential alteration to the operation of an integral part of the Delta Plan is an extremely useful illustration of the need for continued long-term monitoring after the fact; in the case of the Haringvliet the insidious environmental problems have taken over 20 years to become apparent.

1.3.2 Impounded recreational marinas in the UK

A number of marinas in the UK have been impounded, although this decision has generally occurred during the design phase rather than as a later solution to a developing problem. These marinas tend to be concentrated along the south coast of England and the location of several is shown in Fig. 1.3.

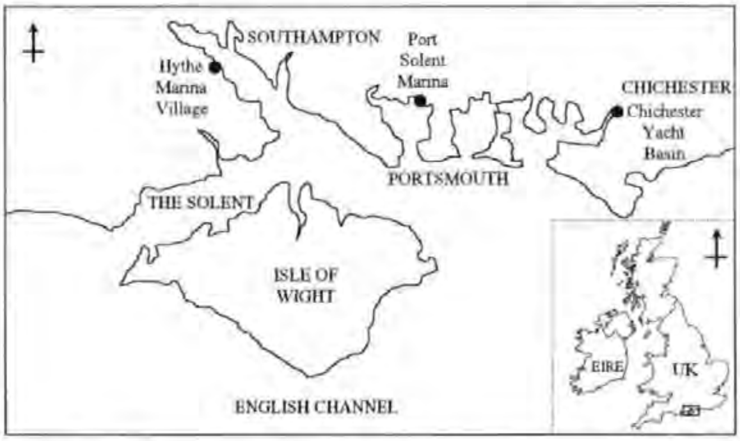


Fig. 1.3. Map of part of the south coast of England showing the location of several impounded marinas.

Hythe Marina is situated on Southampton Water (Fig. 1.3) and was purpose-built in the mid 1980's as the first Marina Village in Northern Europe (Adlam, pers. comm.). Properties and itinerant businesses are situated around the perimeter of the marina, but these do not discharge waste to the water. Litter blown in from Southampton water is regularly collected and microbiological water quality is regularly passed as excellent by Southampton City Council Inspectors. Port Solent Marina in the north of Portsmouth Harbour (Fig. 1.3) is another purpose-built marina complex with peripheral property development. Water quality is maintained through a combination of free flow periods either side of HW and a combined pumping and sluicing operation during the night (Mason, pers. comm.). The marina only houses recreational craft and litter patrols operate daily using a purpose-built dory. Portsmouth City Council conduct routine monthly checks on microbiological water quality and reported counts of total coliforms and thermotolerant faecal coliforms are regularly below 100 and 20 cfu $(100\ ml)^{-1}$ respectively. The main reported problem with water quality arises from road surface runoff; on one reported problem with water quality arises from road surface runoff; on one occasion inputs of oil were traced to a restaurateur disposing of used vegetable oil *via* the road drain outside and were swiftly curtailed by the marina management. Other impounded marinas are planned or are in development. The most recently completed was the Sovereign Marina in Eastbourne (Russell, 1992). Clearly the design of these developments is such that problems of water quality are not inherent and the high amenity value ensures that visually obtrusive pollution is dealt with swiftly and effectively by the marina operators. Chichester Yacht Basin (Fig. 1.3) is a notable exception to the normal design process. It was created from a disused gravel pit at the head of a tidal creek and has been impounded by lock gates for 30 years (Martin, pers. comm.). However, it is situated in a rural location with no riverine inputs, industrial or domestic discharges apparent nearby. The only reported problems with water quality are experienced after periods of summer rainfall when land drainage water pumped into the north of the basin tends to stimulate and sustain considerable algal blooms.

1.4 The Sutton Harbour Impoundment Scheme

1.4.1 Background

Sutton Harbour (Fig. 1.4) is an important commercial resource and attraction for the City of Plymouth, UK, not least because of its heritage: the Mayflower sailed to colonise North America from the steps on the west pier at the harbour mouth in 1620 (Gill, 1976). The harbour covers an area of 1×10^5 m^2 and contains a fish dock and marina that service a combined fleet of *circa* 500 fishing and recreational vessels (Marshall, pers. comm.). The Barbican area to the west combines with the harbour to form a centre for land- and water-based recreation and commerce vital to the local economy (Gunnell, 1977). The fishing industry plays a central role, employing 721 people aboard 193 vessels in 1992 (Seafish, 1995) and a further *circa* 3,500 to 5,000 people onshore in associated trades (Willerton, pers. comm.). Total landings of demersal, pelagic and shellfish in Plymouth in 1990 amounted to 1.2×10^4 tonnes with a value of over £5.2 million (MAFF, 1992). Apart from 5.4×10^3 tonnes (£600,000) of mackerel landed by Interfish of Cattedown Wharf, the vast majority of this catch passed through Sutton Harbour. The principal species were soles, lemon soles, monkfish/anglers, plaice and whiting (demersal), mackerel and pilchards (pelagic) and scallops, crabs and squid (shellfish). The strengths of Plymouth fishing port are excellent motorway links to the rest of the UK, proximity to very productive fishing

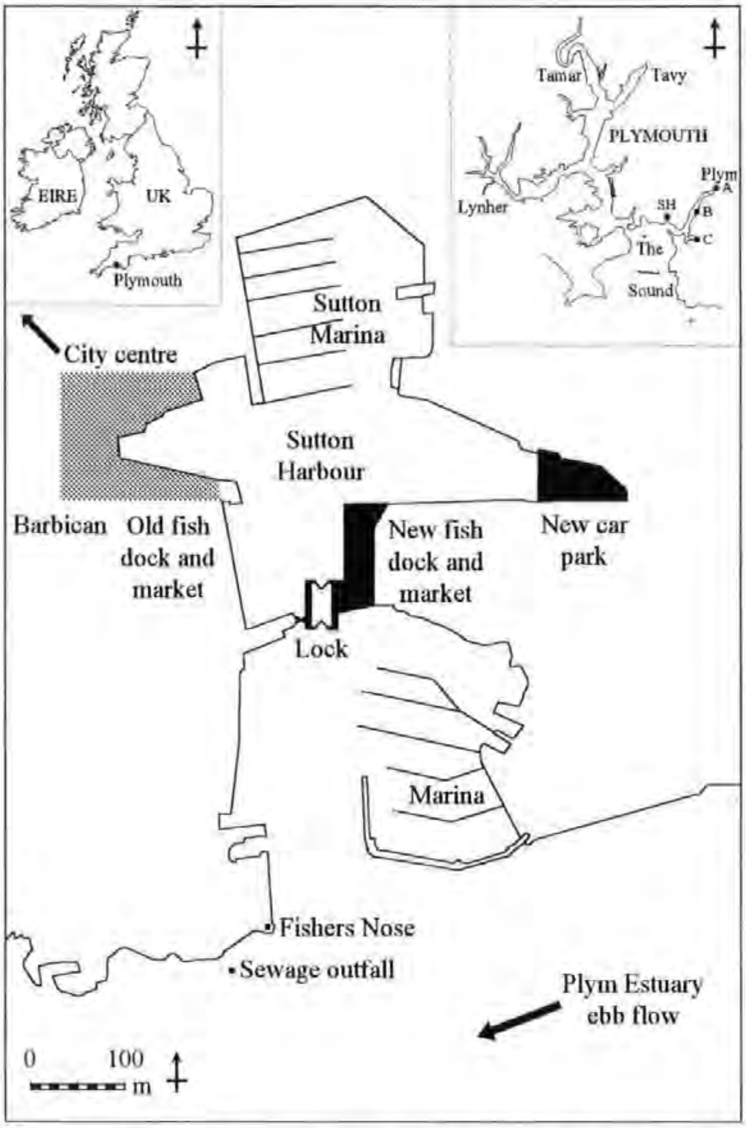


Fig. 1.4. Plan of Sutton Harbour and environs. The filled areas to the east and at the harbour mouth represent areas of reclaimed land (including the lock structure) and the shaded area to the west represents the commercial region periodically inundated before impoundment. Inset (top right) shows location of harbour (SH), Marsh Mills STW outfall (A), Chelson Meadow outfall (B) and Radford STW outfall (C).

grounds in the English Channel, the Western Approaches and the Bay of Biscay and the frequent ferry services to Brittany and Spain (Seafish, 1995). The disadvantages of Sutton Harbour as the main fishing port were that the harbour had insufficient water depth at LW to enable large beam trawlers to operate efficiently, that the existing fish market was too small and its location presented local traffic difficulties for traders.

The harbour is situated to the south east of the city centre at the mouth of the Plym Estuary. The Plym has a catchment area of 192 km^2 (Hiscock and Moore, 1986) and receives significant pollutant inputs from four sewage treatment works or STW's and leachate discharge from a refuse site (Table 1.1). The pollutants from these discharges (Table 1.2) disperse into an estuarine volume of 8.3 x10^6 m^3 at MHWS (Millward, 1993).

Table 1.1. Details of the four STW discharges (Wimpey Environmental, 1991) and the Chelson Meadow leachate discharge to the Plym Estuary.

Source	Outfall location	Effluent treatment	Population served
Marsh Mills STW	Near estuary head	Secondary	90,000
Radford STW	Hooe Lake[a]	Primary	20,000
Billacombe STW[b]	Mid-estuary	Primary	8,000
Cattedown	Near estuary mouth	Crude	4,000
Chelson Meadow	Mid-estuary	Settling lagoon	*circa* 350,000[c]

[a]Small, semi-enclosed embayment on south-east bank of lower estuary. [b]Billacombe was closed in 1994, burden transferred to Radford. [c]Entire population of city (Seafish, 1995).

Table 1.2. Mean flow rates (m^3 d^{-1}) of natural and industrial discharges to the Plym Estuary and mean daily fluxes of selected contaminants (ammonium, total zinc and thermotolerant faecal coliforms) in 1991 (1990[b]).

Source	Water flow[a], m^3 d^{-1}	NH_4^+ flux[a], g d^{-1}	Zn flux[a], g d^{-1}	TFC flux[b], cfu d^{-1}
Marsh Mills STW	3.0 x10^4	7.6 x10^5	1.3 x10^3	5.0 x10^{15}
Radford STW	5.9 x10^3	1.6 x10^5	124	1.7 x10^{15}
Billacombe STW	746[b]	-		3.6 x10^{14}
Cattedown STW	345[b]	-		-
Chelson Meadow	1.6 x10^3	3.2 x10^5	71	-
Total mean anthropogenic flux	3.9 x10^4	1.2 x10^6	1.5 x10^3	7.1 x10^{15}
River Plym flux	1.8 x10^5	7.7 x10^3	860	-
% anthropogenic flux	17.8 %	99.4 %	63.6 %	-

[a]NRA 1991 PARCOM data (Millward, 1993).

[b]SWWSL 1990 data (Wimpey Environmental, 1991).

It is evident from Table 1.2 that the anthropogenic discharges to the Plym are high relative to the low riverine discharge. This can result in generally poor water quality (Wimpey Environmental, 1991), consistent with classification as a heavily industrialised urban estuary. Furthermore, there are additional pollutant sources such as crude sewage outfalls at Fishers Nose and West Hoe, respectively, serving a population of 1,000 to 10,000 and 25,000 to 50,000 (METOCEAN, 1990) and inputs from marinas inside and outside Sutton Harbour, that complicate the role and significance of contaminants. The tidal streams in Plymouth Sound and the lower Plym and Tamar Estuaries are shown in Fig. 1.5, whilst Fig. 1.6 illustrates the tidal streams in the Thalweg just outside the approaches to Sutton Harbour in greater detail.

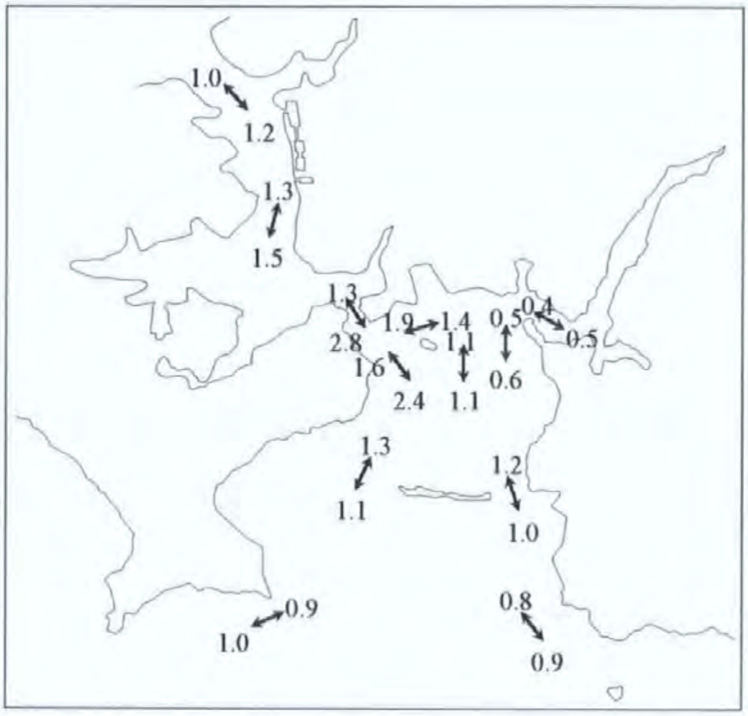


Fig. 1.5. Approximate rate (knots) and set (arrow direction) of maximum flood and ebb tidal streams at springs in Plymouth Sound from Admiralty tidal stream data (after METOCEAN, 1990).

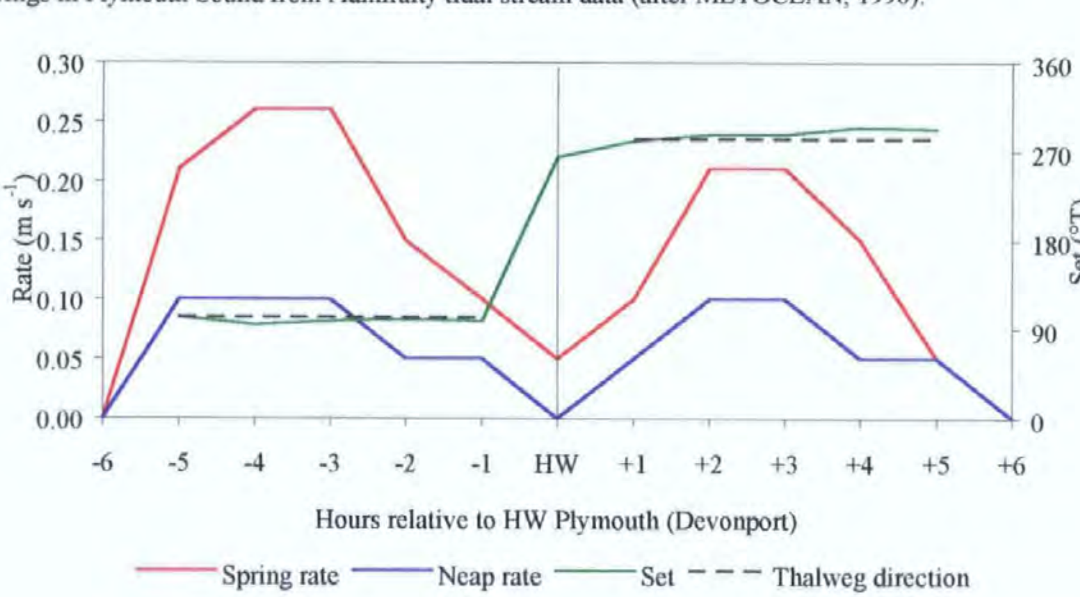


Figure 1.6. Tidal stream diagram for tidal diamond K (Mount Batten) near the mouth of the Plym Estuary (Hydrographer of the Navy, 1988).

It is evident from Fig. 1.5 that the most energetic tidal streams are those in the Narrows at the mouth of the Tamar, whilst the least energetic are those in the Cobbler Channel and Cattewater at the mouth of the Plym. Current measurements and dye and spore tracer studies conducted by Wimpey Environmental (1992) revealed several features of tidal circulation that are of direct consequence to the water quality in Sutton Harbour: First, strong convergence was observed between the Plym ebb and the main deep channel ebb (Smeaton Pass) *circa* 150 m west of the tip of Mount Batten Breakwater. The subsequent formation of a front of buoyant Plym water overlying energetic ebb flow through Smeaton Pass led to efficient mixing and dispersion of the West Hoe discharge but potentially restricted the mixing and dispersion of Plym water. Second, a weak counter current was observed during the ebb from east to west across the Hoe frontage, resulting in ponding of Plym water and Fishers Nose effluent in this area. Dissolved and particulate pollutants from the Plym and Fishers Nose are therefore only partially dispersed by the ebb tide into The Sound. The partially dispersed pollutants together with the ponded water along the Hoe frontage may be shunted into the harbour by the following flood tide, an effect first observed during an appraisal of siltation in nearby Queen Anne's Battery Marina (HR, 1984) and partially confirmed by an incursion of rhodamine dye and *Bacillus globigii* spores into the harbour approaches during tracer studies of the Chelson Meadow discharge in the Plym (Millward, 1993).

The Barbican was formerly prone to periodic flooding several times each year during equinoctial and other exceptional spring tides (Plate 1.1). Moreover, episodic inundation occurred as a result of storm surges in the English Channel (George and Thomas, 1976) which in combination with an onshore wind could increase the predicted HW by up to 1.3 m. These less predictable events were extremely costly to local businesses and residents in water damaged property and led to increasing calls for flood defences.

1.4.2 The impoundment solution

Flood prevention measures finally became economically viable when the NRA undertook to impound the harbour, partially funded by a grant in aid from MAFF, in conjunction with the Sutton Harbour Company's plans for a new fish market on the eastern side providing greater water frontage and easier road transport access. The scheme consists of a lock structure to seal the harbour entrance and provide a greater operational depth whilst allowing continual access to vessels, together with large areas of reclaimed land upon which the new fish market is situated. The lock operates in two modes: the routine operation is to close the gates at *circa* HW+3h in order to prevent further tidal fall inside the harbour. Entry or exit requests are made by VHF radio and effected by locking in or out of the harbour. The lock gates are opened to free flow again at *circa* HW-3h, when the rising tide reaches the impounded level inside the harbour. During freeflow, entry or exit through the restricted harbour mouth is controlled by signal lights. The second mode of operation is employed whenever an exceptionally high tide is predicted, or when a storm surge warning is in force. The lock is closed to act as a tidal defence approximately one hour either side of HW. However, during this period entry to and exit from the harbour is still possible *via* the lock. The benefits of flood alleviation to the local economy are clear and established. However, those accruing to the fishing industry have begun to emerge following the opening of the new fish market in March 1995 (Bowles, pers. comm.). The lock control tower now provides a point of radio contact for the fishing fleet, allowing the arrangement of landing crews and services before arrival. The harbour has begun to attract deep water fishing vessels that were formerly excluded, particularly French and Belgian beam trawlers. As a direct result of impoundment, Plymouth is set to become the main fishing port in the South West.

Work began on the £7 million impoundment phase of the joint project in April 1992 with initial site preparation (Plate 1.2). Key dates in the impoundment project are provided in Table 1.2. In July 1992 a clay-cored coffer dam was constructed across the eastern arm of the harbour and the water behind pumped out (Plate 1.3) in order to provide a dry dock for lock construction. The caisson-style lock chamber, 66 m long by 33 m wide and weighing 9,000 tonnes, was completed in April 1993. Each end of the lock was sealed with temporary gates to allow the structure to float when the dry dock was breached. The entire chamber was then towed into the prepared mouth of the harbour, where it was sunk into position by filling the caisson voids with sand (Plate 1.4). In the subsequent year the two pairs of sector gates were fitted at either end of the lock, during which time the temporary harbour entrance to the west of the structure was used. The lock was incorporated into the reclaimed land to the east and the control room, lock mechanism and pedestrian swing bridge were completed. When the final work and testing inside the lock chamber were complete in April 1994, the temporary harbour entrance was sealed with a removable stop-log gate, and the impoundment scheme was commissioned.

Plate 1.1. View of the Three Crowns in the western arm of the harbour to illustrate typical flooding of the Barbican at HW during exceptional spring tides.

Plate 1.2. View to the east of the harbour of site preparations for the construction of the fish market and new quay walls. Note the rammer-breaker (centre-right) used to remove bed-rock for lock foundations causing physical sediment disturbance and the input of non-marine sediments to the area.

Plate 1.3. View to the south in the eastern arm of the harbour showing the clay-cored coffer dam (left) and the extensive pumping of water from behind the coffer dam into the main harbour to maintain the dry dock.

Plate 1.4. View to the north east overlooking the harbour mouth with the lock chamber (centre) in its final position. The control room can be seen and the scheme is in the final stages of completion. Note the restricted temporary harbour entrance (behind vessel) before the removable stop-log gate was positioned.

Table 1.2: Key dates in the Sutton Harbour impoundment scheme and monitoring study.

DATE	EVENT
May 1990	Initial environmental assessment survey conducted by InstallOcean Ltd.
November 1991	Routine monitoring programme began with surveys by Wimpey Environmental Ltd.
April 1992	Site preparation began for construction work.
May 1992	Routine monitoring programme taken over by University of Plymouth
July 1992	Coffer dam constructed in the eastern arm of the harbour and water pumping commenced to provide dry dock for the lock construction.
April 1993	Coffer dam breached to flood dry dock and lock structure floated into position at the harbour mouth.
April 1994	Scheme commissioned and harbour impounded.
November 1994	Main body of monitoring programme ended.
January 1995	Additional monitoring of primary variables commenced.
July 1995	Additional monitoring of primary variables ended.

1.4.3 The environmental implications of impoundment

Millward (1990) summarised the conditions in the water and sediments of the harbour from the data provided by Houston (1990) and recognised certain implications of the impoundment scheme upon environmental quality:

- The water renewal time of the harbour would increase, allowing far longer for biogeochemical changes that would potentially involve pollutants to occur.

- The lock would act as a barrier to certain processes such as flux of suspended particulate matter (SPM). An accumulation of SPM might lower dissolved oxygen concentrations outside the harbour with implications for the timing of flushing events.

- Timing would also be critical when flows in the River Plym were high leading to considerable fluxes of dissolved and particulate contaminants from the estuary.

- Biodiversity of the harbour fauna would decrease, with loss of intertidal benthos and wading birds, the restricted passage and possible loss of migratory species, and alterations to the community structure of the subtidal benthic macrofauna.

- The internal sewage inputs identified might become more significant after impoundment and should therefore be addressed at the earliest opportunity.

Other environmental issues identified in Section *1.1.2* could be exacerbated by impoundment. In particular, the increased water renewal times might enhance the potential for eutrophication with the associated problems. The issue of the biogeochemical status of the historically contaminated sediments was a cause for concern, particularly if increased remobilisation or accretion led to a requirement for their dredging and disposal.

1.4.4 The aims and rationale of the monitoring programme

The Sutton Harbour Impoundment Scheme provided an ideal opportunity to study a saline system before and after impoundment leading to a better understanding of the environmental effects of both the large-scale civil construction and the subsequent operation of the system. The study was undertaken on behalf of the NRA South West Region Tidal Waters Investigation Unit (TWIU) under contract no. TWU 0002. The objective of the collaborative study was to acquire long-term data covering the period before, during and after construction, and to design a management plan to maintain and improve water and sediment quality according to established criteria. The monitoring strategy was devised within the framework of the NRA requirements for the contract (NRA, 1991). Key dates in the monitoring programme are included in Table 1.2. The primary aims of the study were as follows:

- To estimate the revised water renewal times and flushing efficiency of the harbour under the restricted tidal regime, and to make recommendations for the nature and extent of remedial flushing events as necessary. Within this context the need for an alternative mechanism for water renewal in stagnant areas was to be considered.

- To assess the potential for eutrophication in the impounded harbour with implications for the growth of nuisance or toxic algae, and for the development of benthic anoxia.

- To discern the various inputs of sewage contamination to the harbour and to establish the potential for changes in microbiological water quality after impoundment under varying meteorological and hydrodynamic conditions.

- To monitor the biogeochemical status of the sediments viz. changes in trace metal concentrations and redox potential that could lead to remobilisation of metals into the water column.

- To assess the changes in benthic macrofauna community structure occurring as a result of construction, and then as a result of the altered hydrodynamic regime.

- Overall, to provide a strategy for ecological management of the impounded harbour, and thus to aid the NRA in their remit to ensure continued water quality in controlled waters.

Chapter Two

Methods

Chapter 2 - Methods

2.1 The Monitoring Strategy

Monitoring of water and sediment quality in Sutton Harbour was conducted at different intervals, according to the schematic diagram in Fig. 2.1. These were selected in order to cover seasonal variability. Two subsets of primary variables were determined bimonthly, whilst secondary variables, sediment properties and benthic macrofauna assemblages were studied semi-annually. The resulting series of surveys was conducted aboard appropriate vessels on adjacent spring and neap tides (six to eight days apart) commencing with a study of primary and secondary variables in November 1991.

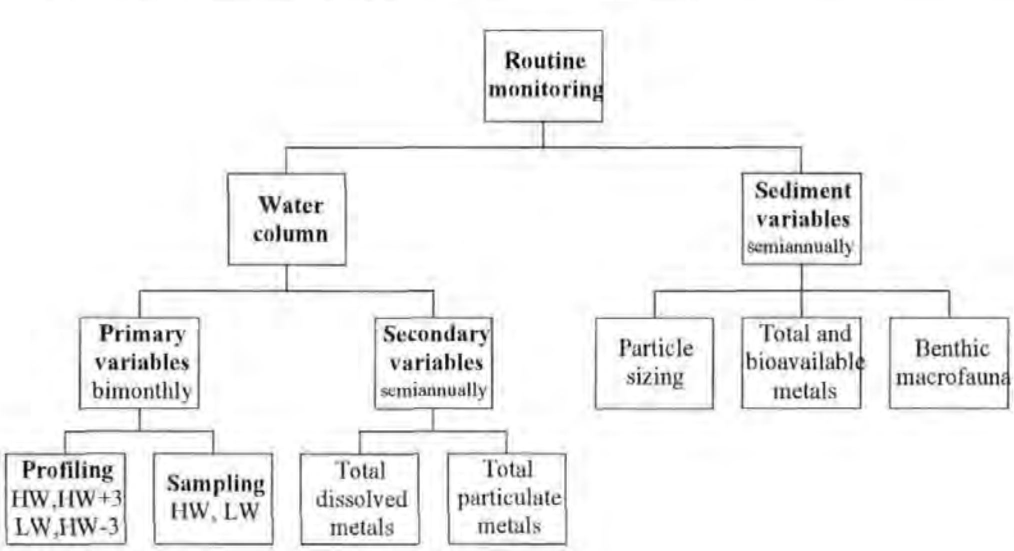


Fig. 2.1. Schematic representation of the routine monitoring programme employed in the study of Sutton Harbour.

Each survey utilised the fixed stations shown as numbered points in Fig. 2.2. These were visited in sequence at key stages throughout the spring and neap tidal cycle. The strategy was to begin at high water (HW), sample through maximum ebb (HW+3h) and low water (LW) and to finish with maximum flood (HW-3h). This sequence ensured that the character of the water body remained consistent between survey periods. However, for logistical reasons it was not always possible to begin at HW. In these cases, the logical sequence of four tidal states was still maintained. In order to obtain near-synoptic data for the entire harbour, each survey run began at station one half an hour before the relevant tidal state and finished at station four half an hour after.

The survey stations were chosen in order to represent varying conditions inside the harbour, as revealed by the detailed assessment of an initial environmental survey conducted in May 1990 (Millward, 1990; Houston, 1990). Station one was in the north of the harbour furthest from the entrance as this area was likely to suffer from long water renewal times. Station two was positioned centrally, in line with the axial flow through harbour entrance, and therefore received most tidal flushing. Station three was located in the eastern arm of the harbour, where water renewal is thought to be sluggish owing to low current velocities. This station was then moved slightly to the west in July 1992 after the construction of a coffer dam and later land reclamation. Station four was located outside the harbour entrance, initially serving as the only control. However, a fifth station was added in July 1992 (see Fig. 2.2) at which a limited suite of primary variables was determined in order to gauge the effect of the Plym Estuary and the untreated sewage discharges along the Plymouth waterfront, particularly that from Fishers Nose.

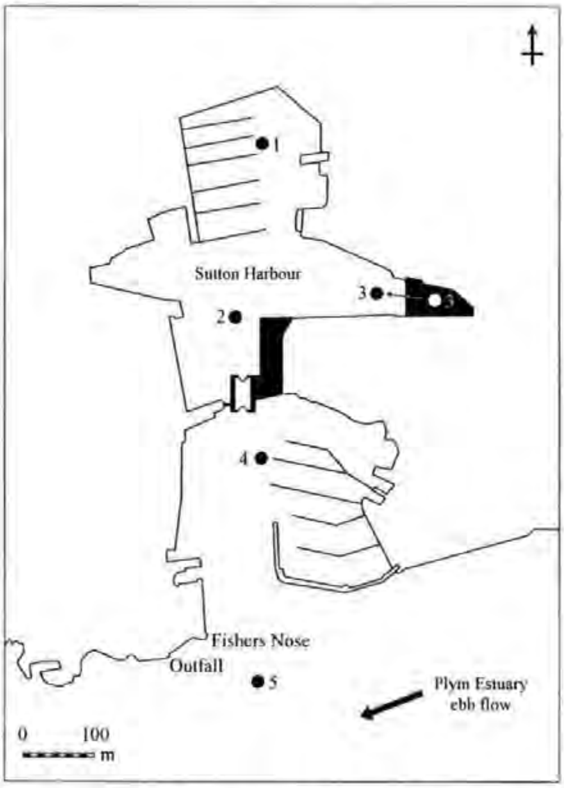


Fig. 2.2. Location of sampling sites in Sutton Harbour, Plymouth (UK). Station 3 shown in pre-July 1992 (open circle) and post-July 1992 (filled circle) positions.

2.1.1 Water column profiling

Profiles of the first subset of primary variables (viz. temperature, salinity, dissolved oxygen, pH, suspended particulate matter (SPM) and chlorophyll *a* concentration) were determined *in situ* at each station at every tidal state.

The apparatus employed (Table 2.1) consisted of a metal-framed rig with salinity, temperature and depth sensors (NBA Controls Ltd. model TDS-7), to which all other instruments were attached, with all cables leading to topboxes on board the survey vessel. At each station, total depth was measured with an on-board echo sounder. The TDS-7 rig was then carefully lowered by line to 0.5 m above the bed ('bottom') ensuring that the sediments were not disturbed. Readings of all variables were then recorded at this depth, synchronous with a 'bottom' water sample. The rig was raised in suitable depth intervals not exceeding 2 m and readings taken at each depth to 0.5 m below the surface ('surface'). The final readings were taken at this depth, synchronous with a 'surface' water sample.

Table 2.1. Instrumentation used for *in situ* vertical profiling.

Parameter	Instrument	Range	Resolution
Temperature,	NBA (Controls) Ltd.	T 0.0 - 30.0 °C	$\Delta T \pm 0.2$ °C
Salinity and	TDS-7	S 3.0×10^{-3} - 40.0×10^{-3}	$\Delta S \pm 0.5 \times 10^{-3}$
Depth		Z 0.0 - 100.0 m	$\Delta Z \pm 2$ % f.s.d.
Chlorophyll *a* concentration	Sea Tech Ltd. fluorometer	0.0 - 11.0 µg l^{-1} (0.0 - 5.7 V f.s.d.)	± 0.1 µg l^{-1}
SPM concentration	Partech suspended solids monitor (SSM)	0 - 75 mg l^{-1}	± 0.5 mg l^{-1}
Dissolved oxygen	Clandon/YSI Ltd.	0.0 - 200.0 %	± 0.3 %
concentration	YSI 58	0.00 - 20.00 mg l^{-1}	± 0.03 mg l^{-1}
pH	Hanna Instruments Ltd. HI 9024	0.00 - 14.00	± 0.01

2.1.2 Calibration

All field instruments were calibrated on a frequent but irregular basis, dependent upon the nature of the measurement. Calibrations were conducted in accordance either with manufacturers' instructions or with standard procedures.

Salinity on the TDS-7 was calibrated by dilution of indigenous Sutton Harbour water with distilled water. A Plessey laboratory salinometer ($\Delta S = 0.005 \times 10^{-3}$) calibrated against a vial of IAPSO standard seawater was used to determine the salinity of each dilution. These were plotted against the digital readout from the TDS-7 to produce a calibration graph, an example of which is shown in Fig. 2.3. This mixing of two waters with different temperatures also provided an opportunity to check the temperature calibration of the TDS-7 against a mercury-in-glass thermometer, which was found to be satisfactory on all occasions.

The suspended solids monitor (Partech Ltd.) was calibrated in a similar way, by dilution of an indigenous particle population in Sutton Harbour water. For each dilution, the SPM concentration was determined gravimetrically. Glass fibre filters (Whatman 47 mm GF/C) were weighed on a five-figure analytical balance and used to filter an aliquot of sample. The filters were washed with distilled water to remove salts and dried to constant weight in a drying cupboard. The volume of the filtrate was carefully measured. The resulting SPM concentrations in mg l^{-1} were plotted against the readings of the monitor to produce calibration graphs, as shown in Fig. 2.4.

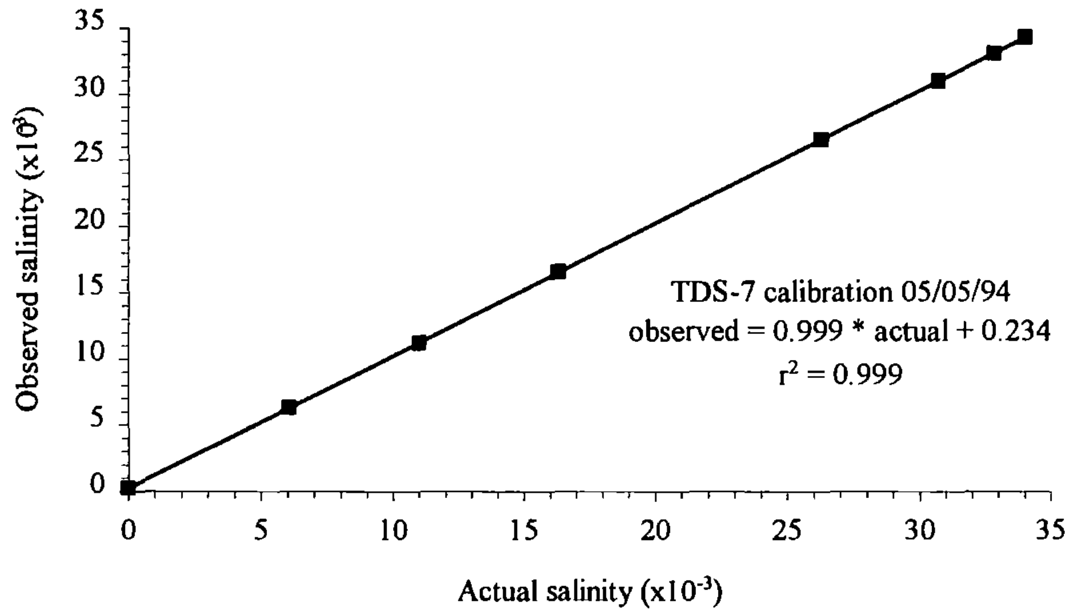


Fig. 2.3. Salinity calibration example for the TDS-7 against diluted standard seawater.

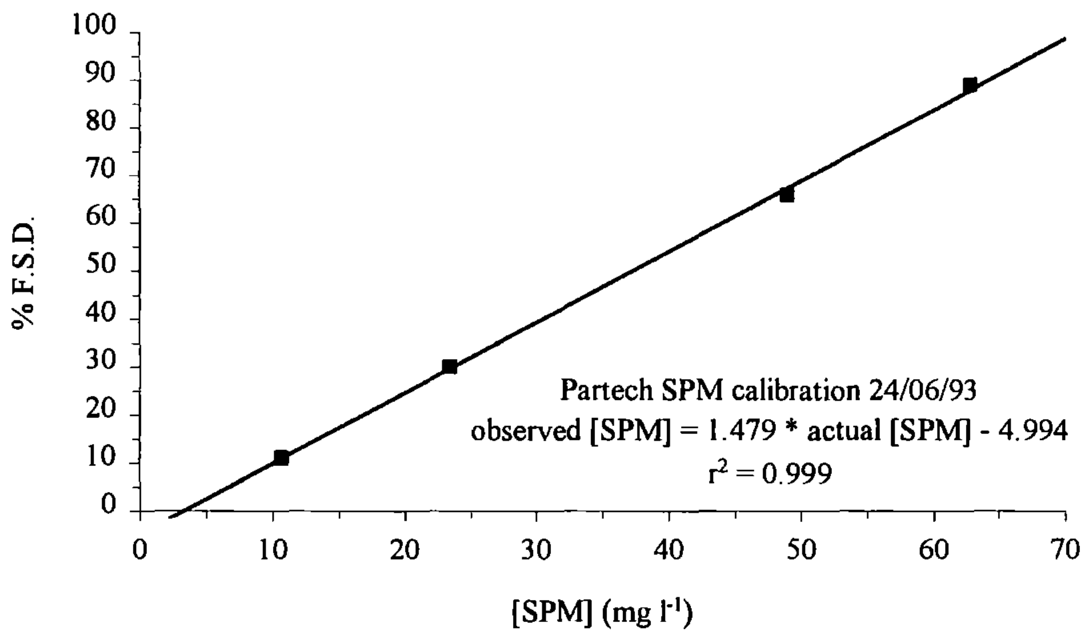


Fig. 2.4. Example of calibration graph for SPM concentration on the Partech.

Difficulties were encountered in calibrating the fluorometer. The instrument range was factory-set at 0.0 - 10.0 μg l^{-1} for an output of 0.0 - to 5.0 V and an in-house calibration using a *Skeletonema costatum* culture was provided (Lambert, pers. comm. 1992) that was linear to 11.0 μg l^{-1} (5.5 V). An additional laboratory calibration was carried out on 07/07/94 following the collection of a bulk water sample from Sutton Harbour during bloom conditions. The instrument was immersed in this sample in an opaque plastic container. The sample was diluted with distilled water and thoroughly mixed until a steady output voltage (*circa* 5.5 V) was achieved. At this point, the voltage was recorded and an aliquot of sample (*circa* 1.0 l) was abstracted and filtered through a glass fibre filter (Whatman 47 mm GF/C). The filter was stored in the dark at 4 °C to await spectrophotometric determination of chlorophyll *a* concentration (Section *2.2.6*) and the exact volume of filtrate was recorded for the concentration calculation. This process was repeated four more times with increasing dilutions of sample and again with distilled water only after all apparatus had been thoroughly rinsed. The resulting calibration line is shown in green in Fig. 2.5. However, when the calibration was compared with Lambert's (1992) calibration, a great disparity was observed. The calibration on 07/07/94 had been conducted with a local natural microphytoplankton population, since chlorophyll *a* can vary from 0.4 % to 4.0 % of the total dry weight of different microphytoplankton species (DoE, 1983). However, the fluorescence also depends on the physiological state of the phytoplankton (Parsons *et al.*, 1984) and the bulk sample for the calibration on 07/07/94 had unavoidably been stored for 48 h before use. The presence of chlorophyllides from the hydrolytic decomposition of chlorophyll that do not fluoresce but are spectrophotometrically identical to the parent chlorophyll would explain the apparently lower fluorescence from higher concentrations in the latter calibration. The remaining data in Fig. 2.5 are a suite of survey samples and extra samples where fluorescence was determined *in situ* and concurrent water samples were analysed for chlorophyll *a* spectrophotometrically. These results confirm that Lambert's 1992 calibration of the instrument remained valid.

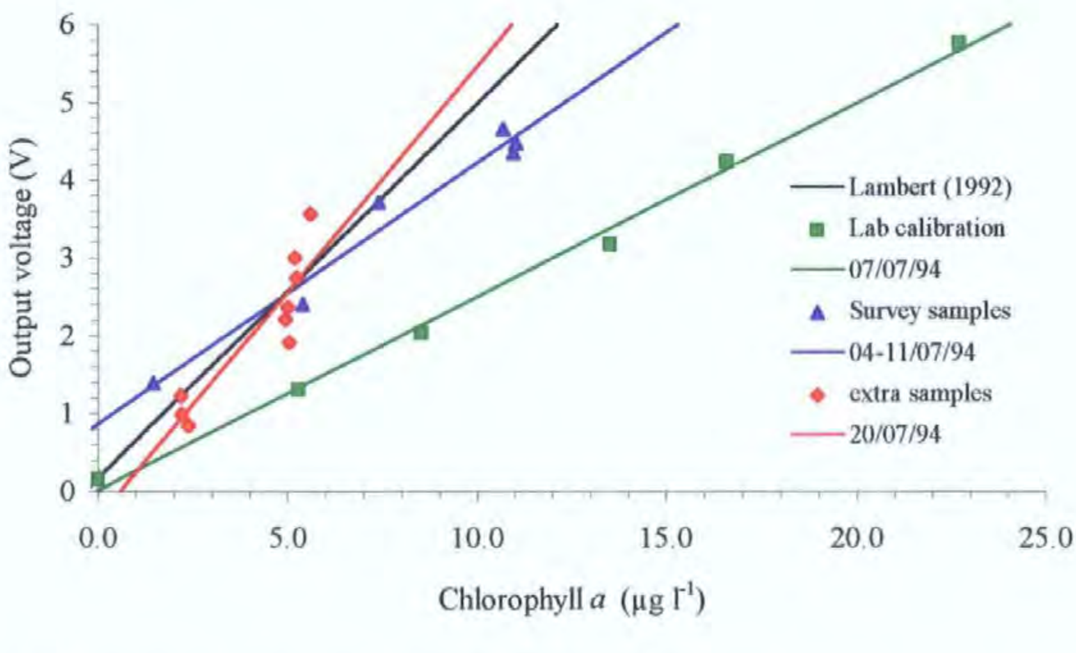


Fig 2.5. Comparative calibration and validation of the SeaTech Fluorometer.

The pH meter (Hanna Instruments Ltd. model HI 9024) was calibrated in the lab before each survey, according to manufacturer's instructions, using ready-made buffer solutions pH 7 and pH 9 at 20 °C. The solutions were also taken into the field and used as necessary. Since it did not produce a current at zero-oxygen tension (and therefore did not require a zero-oxygen solution), the dissolved oxygen meter needed only to be equilibrated at 100 % with a moistened cap over the electrode. The meter was therefore recalibrated before each tidal state in order to compensate for any alterations in atmospheric pressure. Salinity compensation was effected *via* the salinity dial on the meter to the value determined by the TDS-7 for each sample. Depth/pressure compensation was automatic *via* a diaphragm on the electrode.

2.1.3 Discrete water quality measurements

Two higher-frequency sampling exercises were mounted in order to increase spatial and temporal resolution. Firstly, two surveys of *in-situ* primary variables and of faecal indicator bacteria were conducted one week either side of the regular July 1993 spring and neap primary surveys as a component of a joint study (Parker, 1993). The number of stations for profiling and bacteria sampling was increased from five to ten on these occasions (Fig. 2.6).

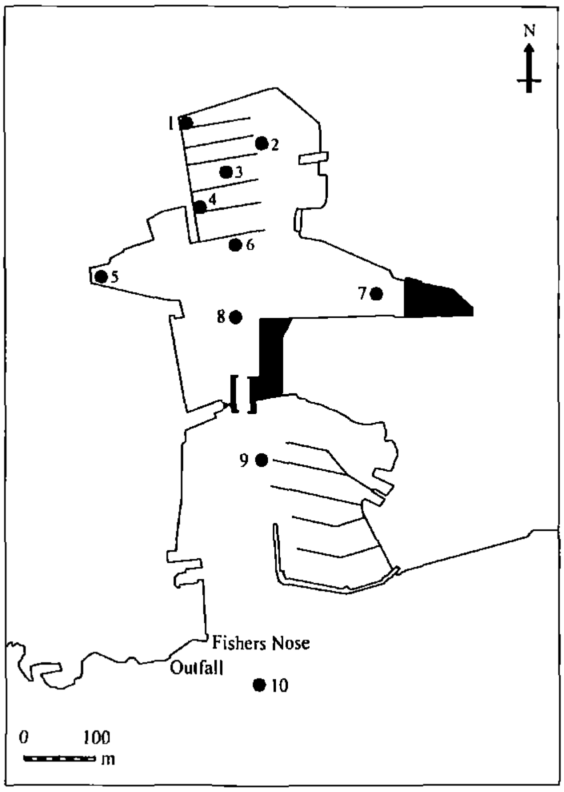


Fig. 2.6. Increased spatial resolution of sampling during July 1993 (Parker, 1993).

The rationale for site selection is summarised in Table 2.2. All sampling operations and analyses were conducted as in Sections *2.2.1* and *2.2.4* respectively. A fifth survey was conducted two weeks after this series, comprising an 'axial' survey of the harbour and approaches, from site 2, through site 8 and site 9, to site 10. This took place either side of HW and was designed to test whether the major input of sewage material was from the

Fishers Nose discharge by estimating the bacterial dispersion from south to north and vertically through the water column. To this end, four water samples were collected at each site (surface, two intermediate depths and bottom) together with a sediment sample obtained by stainless steel gravity corer, in order to examine the role of the sediments in the fate of faecal indicator bacteria. Surficial material from each sediment core (1 g) was added to a glass bottle (1000 ml) containing sterile saline water (100 ml) and shaken vigorously. The samples were treated as 100 ml dilutions and processed as in Section *2.2.4*.

Table 2.2. Site selection rationale for July 1993 intensive bacterial study.

Site number	Rationale for selection
1	Adjacent to Sutton Marina ablution block and furthest from harbour mouth.
2	Corresponding to Station 1 (Fig. 2.2).
3	Additional coverage of marina area to assess possible use of yacht heads.
4	Adjacent to known storm sewer outfall in harbour wall beneath Sutton Pier.
5	Western arm of harbour with limited water exchange, as at site 7 (Station 3).
6	Location of continuous water monitoring apparatus (used for comparison).
7	Corresponding to Station 3 (Fig. 2.2).
8	Corresponding to Station 2 (Fig. 2.2).
9	Corresponding to Station 4 (Fig. 2.2).
10	Corresponding to Station 5 (Fig. 2.2).

The second higher-frequency sampling exercise consisted of a series of spot measurements determined at various times of HW throughout June 1994 in order to follow the progression of an algal bloom observed inside the harbour during the May 1994 surveys (immediately after impoundment). Samples were collected by filling an acid-washed (10 % HCl, AnalaR) Nalgene HDPE bottle (2 l) from sites 2, 6 and 9 (Fig. 2.6) on 7th and 9th June 1994, then from site 6 (Fig. 2.6) only (in concert with cleaning, recalibration and downloading of data from the continuous water monitoring apparatus; refer to Section 2.7) on 16th, 24th and 29th June 1994. Temperature, salinity and dissolved oxygen concentration were determined *in situ* using a calibrated TS bridge (MC-5) and a YSI 58 oxygen meter in concert with each sample. On immediate return to the laboratory, the samples were filtered for chlorophyll *a* analysis, and concentration of TON and dissolved orthophosphate were determined in aliquots of filtrate (50 ml).

2.1.4 Visual aesthetics surveys

Throughout each primary and secondary survey, particularly during each HW and LW circuit, the aesthetic condition of the harbour was assessed by shore-walk inspection (subject to availability of personnel), or by observation from the vessel. The general cleanliness of the harbour was noted, along with specific incidents of diesel slicks, foam, plastic and paper litter, turbidity plumes and any miscellaneous flotsam and jetsam. Furthermore, any operational activities deemed likely to be deleterious to water and sediment quality were noted.

2.2 Sampling and Analysis of Primary Variables

The remaining primary variables in the water column (viz. dissolved ammonium, total oxidised nitrogen (TON), dissolved orthophosphate, five-day biochemical oxygen demand (BOD_5) and faecal indicator bacteria) were determined by water column sampling. This component of the study was carried out according to the stringent Blue Book requirements for quality control, maintaining compatibility with existing NRA data. All plastic and glassware was soaked in laboratory detergent (Decon 90) to remove any organic layers residual from manufacture. Polypropylene syringes (50 ml), Swinnex filter holders and brown glass bottles (250 ml) for dissolved nutrients were acid-washed (10 % HCl, AnalaR) and thoroughly rinsed with deionised water (MilliQ). Brown glass bottles (500 ml) and clear glass flasks (250 ml) for BOD_5 were washed with an acidified solution (I_2/KI in 1 % H_2SO_4, AnalaR) and then with MilliQ. Glass bottles (1000 ml) for microbiological determination were machine-washed, fitted with foil-covered caps, and sterilised in an autoclave for 15 minutes at 15 p.s.i. and 121 °C.

2.2.1 Water column sampling

Each spring and neap survey required that surface and bottom water samples be obtained at HW and LW from each station. These were collected using an IOS water bottle, consisting of a cylindrical polypropylene body with hinged rubber caps at either end that were triggered shut from the surface, thus enclosing a water sample from the required depth. Polypropylene taps were fitted at the top and bottom of the bottle for sample abstraction. The absence of internal metal components meant that the bottle was equally suitable for sampling for metal and nutrient determinations, since it could be acid-rinsed (10 % HCl, AnalaR) before each survey. At each station, the weighted bottle was primed by cocking the caps and closing the top and bottom taps. It was then lowered by a line marked at 1.0 m intervals to a depth 0.5 m above the bed as indicated by the on-board echo sounder. A messenger was clipped to the line and released at the time of determination of the bottom profile readings. The water sample was retrieved and the relevant aliquots abstracted, as described below. The process was repeated to collect a water sample from a depth of 0.5 m below the surface concurrent with the surface profile readings. However, before impoundment, on some occasions the recorded depth was approximately only 1 m at LW. Only one mid-depth sample and set of measurements were recorded on these occasions.

2.2.2 Dissolved ammonium and un-ionised ammonia (UIA)

The top cap of the IOS bottle was opened and an aliquot (10 ml) of sample drawn into the Swinnex syringe and expelled in order to rinse it. Another aliquot (10 ml) was extracted, the Swinnex filter holder attached and the aliquot passed through to rinse the pre-weighed filter (Sartorius 25 mm cellulose acetate, 0.45 μm pore size). The filter holder was then removed, an aliquot (50 ml) drawn from the IOS bottle and the holder replaced. The sample aliquot was filtered into a labelled brown glass bottle (250 ml) and placed in the dark in a coolbox at <4 °C for analysis within 6 hours.

Dissolved ammonium concentrations (limit of detection (LOD) 4 $\mu g\ l^{-1}$) were determined using a flow injection analysis (FIA) technique whereby ammonia gas liberated from the sample in a strong alkali (NaOH) donor stream diffuses across a membrane into an indicator (bromothymol blue) acceptor stream where it is detected spectrophotometrically (van Son *et al.*, 1983; Clinch *et al.*, 1988). Un-ionised ammonia concentrations, which cause stress in marine life but which cannot be determined directly, were calculated as a percentage of dissolved ammonium concentrations from tabulated values dependent upon temperature, pH and salinity (Bower and Bidwell, 1978).

2.2.3 Dissolved total oxidised nitrogen (TON) and dissolved orthophosphate

For these determinations, another aliquot (50 ml) of sample was drawn into the syringe and passed through the filter used in Section *2.2.2* into another labelled brown glass bottle (250 ml), and again placed in the dark in a coolbox at below 4 °C for analysis within 6 hours. The pre-weighed filter, having now processed 110 ml of sample, was retained for supplementary gravimetric [SPM] determination.

Total oxidised nitrogen (LOD 10 $\mu g\text{-N}\ l^{-1}$; relative standard deviation (RSD) 0.2 %) and orthophosphate concentrations (LOD 2 $\mu g\text{-P}\ l^{-1}$; RSD 2.0 %) were determined spectrophotometrically, using the Blue Book methods adapted for FIA. Concentrations of TON (essentially nitrate + nitrite) were determined by reduction to nitrite with copperized cadmium and derivatization by N-1-naphthylethylenediamine dihydrochloride (N1NED) and sulphanilamide to a pink azodye (DoE, 1982b), with LED photometric detection at 542 nm (Clinch *et al.*, 1987; McCormack *et al.*, 1994). Concentrations of orthophosphate were determined using the molybdenum blue reaction in a reagent-injection system adapted for seawater use, with LED photometric detection at 660 nm (Worsfold *et al.*, 1987).

2.2.4 *Faecal indicator bacteria*

An aliquot (500 ml) of sample was drawn from the bottom tap of the IOS bottle into a glass bottle (1000 ml), with due attention to maintaining sterility of the bottle neck and screw-cap. The sample was placed in the dark in a coolbox at <4 °C and delivered for filtration within one hour of completion at each tidal state.

Bacteriological determinations were conducted using Blue Book techniques for membrane filtration onto selective media (DoE, 1983). Determinations were made of total coliform (TC), thermotolerant faecal coliform (TFC) and faecal streptococci (FS) counts in each sample. The results were expressed as colony forming units per 100 ml (cfu $(100\ \text{ml})^{-1}$). Hands were thoroughly washed before and after labwork. All funnels and filter holders were sterilised with 70 % industrial methylated spirit (IMS) and rinsed with sterile water. Forceps were sterilised with 70 % IMS and flamed before each use, with the bunsen flame near the apparatus also creating an updraught. Sterile petri dishes were marked on the base with dilution, sample number, date and test (TC, TFC or FS). In an attempt to ensure the number of colonies on each membrane remained above 20 (for statistical significance) and below 200 (to prevent inhibited growth due to crowding), two different dilutions of each sample were made for each test, by reducing the filtered volume from 100 ml to 10 ml or 1 ml. Table 2.3 contains the dilutions and incubation conditions used for each determinand.

Table 2.3. Faecal indicator bacteria test dilutions and incubation conditions.

Test	Dilutions	Pre-incubation		Incubation	
	(ml)	Temperature (°C)	Duration (h)	Temperature (°C)	Duration (h)
TC	10, 1.	30	4	37	14
TFC	100, 10.	30	4	44	14
FS	100, 10.	37	4	45	44

For the TC and TFC tests, sterile pads were dispensed and saturated with membrane lauryl sulphate broth / phenol red by pipette. For the FS test, sterile sodium azide pads were saturated with sterile water by pipette. Filtration for all three tests was performed simultaneously with a stainless steel triple filter holder attached to one vacuum pump. Membrane filters (Sartorius 47 mm cellulose acetate, 0.45 μm pore size) were placed on the filter holder with sterile forceps and the funnel clamped in place. For 1 ml dilutions, sterile saline solution (*circa* 10 ml) was added to the funnel to ensure coverage of the filter. The appropriate volume of sample was pipetted into each funnel and filtered under vacuum. Each filter was removed with sterile forceps and carefully rolled onto the appropriate pad, without trapping air. The petri dishes were covered, loaded into the appropriate pre-incubators for 4 hours and then into the appropriate incubators for the duration (Table 2.3). Immediately after incubation, the petri dishes were removed and the colonies counted. For the TC and TFC tests, all yellow colonies were counted, ignoring pink or colourless colonies. For the FS test, all red, maroon or pink colonies were counted. The large number of samples and dilution duplicates required for each survey run precluded the routine assessment of replicate samples, although occasional replicates ($n = 3$) showed typical RSD's of between 3 % and 5 % for each test. Moreover, the inherent variability of these presumptive counts dictates that results are viewed with care (Fleisher, 1990) and are used as an indication of trends rather than as absolute values, particularly since variation at the source is generally greater than the errors introduced by proper laboratory procedures (Tillett, 1993).

2.2.5 Biochemical Oxygen Demand

Determinations of BOD_5 were conducted according to Blue Book recommendations (DoE, 1989). An aliquot (300 ml) of sample was drawn from the bottom tap of the IOS bottle into a brown glass bottle (500 ml) and tightly capped. The bottle was shaken vigorously to begin the oxygen saturation of the sample and placed in the dark in a coolbox at <4 °C. Following the survey, the samples were brought to 20 °C and full oxygen saturation in a constant temperature store (20 °C). The samples were decanted into clear glass BOD flasks (250 ml) and the initial dissolved oxygen concentrations, D_1 (mg l^{-1}) were determined using a YSI 5730 self-stirring BOD probe together with the YSI 58 DO meter. The flasks were topped up fully with remaining sample and tapered ground-glass stoppers were fitted tightly, displacing all air at the mouth of the flask. The flasks were left for five days in a light-tight box inside the constant temperature store at 20 °C. After this time, the flasks were removed and the final dissolved oxygen concentrations, D_2 (mg l^{-1}) were determined. No dilution of sample was necessary throughout the study due to the generally low organic load of the samples. Therefore, BOD_5 values were given by the equation:

$$BOD_5\ (\text{mg l}^{-1}) = D_1 - D_2 \qquad \text{Eq. 2.1.}$$

2.2.6 Chlorophyll a *concentration*

Determinations of chlorophyll *a* were generally made *in situ* using the fluorometer, but where this was not possible, during intense blooms out of the range of the instrument, during fluorometer calibration, or during discrete water quality measurements, they were conducted in accordance with Blue Book recommendations (DoE, 1983). Samples (*circa* 1.5 l) were collected in clean Nalgene HDPE bottles (2 l) and stored in the dark in a coolbox at <4 °C for transportation to the laboratory. Aliquots (*circa* 1 l) of sample were filtered (Whatman 47 mm GF/C) under vacuum using Millipore glass filtration apparatus. The volume of filtrate was recorded (± 10 ml) and the filtrate discarded. The filter was transferred to a centrifuge tube and sufficient methanol added (20 ml) to cover the filter.

allowing the methanol to boil for 10 seconds. The tube was removed, stoppered and kept in the dark for 5 minutes, before careful removal of the filter paper with squeezing to minimise solvent removal. The solvent was centrifuged (3500 rpm, 7 min.) to yield a clear solution, which was carefully decanted into a glass cuvette (path length 40 mm). The absorbance of the solution at 665 nm and 750 nm was determined in a UV/visible spectrophotometer zeroed on methanol in the reference beam. The chlorophyll *a* concentration in the sample was then calculated in the following way:

$$[\text{chlorophyll } a] = \frac{13.9 \times \mathrm{A} \times \mathrm{v}}{\mathrm{d} \times \mathrm{V}} \,\mu\mathrm{g}\,\mathrm{l}^{-1} \qquad \text{Eq. 2.2.}$$

Where A = absorbance (units) = $A_{665\ nm} - A_{750\ nm}$; v = volume of methanol (ml); V = volume of filtrate (l); and d = cell path length (cm).

2.3 Sampling and Analysis of Secondary Variables

Sampling for secondary variables in the water column (viz. total dissolved and total particulate Cd, Cu, Hg, Pb and Zn) was conducted semi-annually in concert with determination of primary variables, but the greater sample volumes necessitated a revised sampling strategy. This was carried out according to Blue Book requirements for quality control, thus maintaining compatibility with existing NRA data. Samples were collected using a larger volume IOS bottle, from which faecal indicator bacteria and BOD_5 samples were abstracted. The remainder were drawn into acid-washed (10 % HCl, AnalaR) HDPE carboys (10 l) for transport to the laboratory. All plastic and glassware was initially soaked in Decon in order to remove any organic layers residual from manufacture and then prepared according to Blue Book methods. Millipore glass filtration apparatus was acid-washed in 10 % HCl (AnalaR). Nalgene HDPE bottles (1000 ml) for dissolved metal and glass bottles (250 ml) for dissolved Hg determination were acid-washed in 10 % HNO_3 (AnalaR). All apparatus was rinsed thoroughly with MilliQ. On arrival at the laboratory, the bulk samples were filtered (pre-weighed Sartorius 47 mm cellulose acetate, 0.45 μm pore size) under vacuum in order to obtain the maximum of material for [SPM] and particulate heavy metal determinations. An aliquot (250 ml) of each filtrate for dissolved metal determination was placed in a plastic bottle, acidified (250 μl conc. HNO_3, AristaR) and frozen below -18 °C. Another aliquot (50 ml) for dissolved Hg determination was placed in a glass bottle, acidified (50 μl conc. HNO_3, AristaR) and frozen below -18 °C. Two aliquots were recovered for determination of ammonium (50 ml) and TON and dissolved orthophosphate (50 ml), placed in glass bottles and refrigerated at below 4 °C pending analysis within 6 h.

2.3.1 Dissolved metal analyses

Concentrations of dissolved metals Cd, Cu, Pb and Zn in samples collected during May 1992 and November 1992 were determined directly by inductively-coupled plasma mass spectrometry (ICP-MS, Varian) using indium as an internal standard. However, despite the benefits of multi-element analysis and rapid sample throughput, the technique proved unsatisfactory. The high salinity of the samples meant that a tenfold dilution was necessary in order to reduce salt interference, thereby raising detection limits, particularly with respect to Cd and Pb, where high blank values precluded meaningful analysis.

From May 1993 onward, dissolved metals in samples were preconcentrated by the Blue Book complexation-solvent extraction technique (Danielsson *et al.*, 1982; Campbell *et al.*, 1985; DoE, 1988). All plastic and glassware were soaked in Decon, acid-washed (10 % HNO_3, AnalaR) and thoroughly rinsed with MilliQ. All apparatus was reserved solely for dissolved metal analyses and all operations were carried out under ultra-clean conditions in a laminar flow hood. Aliquots (200 ml) of sample were decanted into glass separating funnels (300 ml) and mixed reagent (7.5 ml) was added. This contained an ammonium acetate buffer solution to raise the pH to between 4.4 and 5.3 and an ammonium pyrrolidine dithiocarbamate / diethylammonium diethyldithiocarbamate (APDC/DDDC) solution as a chelating agent. The funnels were shaken and the pH checked with narrow range indicator paper. Measured volumes (10 ml) of 1,1,2-trichlorotrifluoroethane (Freon) were added to the separating funnels and the samples were extracted for two minutes, with frequent venting. The phases were allowed to separate fully and the lower organic layer containing the metal chelates was drained into a small polyethylene bottle (50 ml), to which was added conc. HNO_3 (500 μl, AristaR) in order to "back-extract" the metals. The extraction was repeated with a further volume of Freon (10 ml), which was then drained into the small bottle. Finally, MilliQ (20 ml) was added to the small bottle and shaken, in order to complete the "back-extraction" and to provide sufficient volume of aqueous phase for analysis. The sample extracts were stored below 4 °C. Standards and blanks for the analysis were made from doubly-extracted water retained from the sample extractions. Four aliquots (200 ml) were spiked with different volumes (250 μl, 500 μl, 750 μl, 1000 μl) of a mixed standard cocktail (BDH Spectrosol, 1 mg Cd l^{-1}, 2 mg Cu l^{-1}, 1 mg Pb l^{-1}, 10 mg Zn l^{-1}) in order to provide a range of standards for each metal. Two aliquots (200 ml) were retained as blanks. All six aliquots were extracted and back-extracted in the same way as the samples. These were analysed by flame atomic absorption spectrophotometry (FAAS, Varian SpectrAA 300/400) with microcup and continuum background correction for Zn and by graphite furnace AAS (Perkin Elmer 4100ZL THGA) with Zeeman background correction for Cd, Cu and Pb. The detection limits achieved by this method were more than acceptable, as can be seen from Table 2.4.

Table 2.4. Comparison of actual detection limits from secondary surveys (3σ of blank values) with method detection limits (DoE, 1988), NRA requirements and EQS values.

Dissolved metal	ICP-MS May 1992-Nov 1992	Solvent extraction May 1993-Nov 1994	(DoE, 1988)	NRA[1]	EQS[1]
Cd ($\mu g\ l^{-1}$)	-	0.008 - 0.021	0.035	0.125	2.5
Cu ($\mu g\ l^{-1}$)	0.84	0.08 - 0.24	0.65	0.5	5.0
Pb ($\mu g\ l^{-1}$)	-	0.05 - 0.165	0.75	2.5	25.0
Zn ($\mu g\ l^{-1}$)	1.26	1.17 - 1.65	2.62	4.0	40.0

[1]NRA (1991).

2.3.2 Quality control

The quality of the dissolved metal determinations was assured by the periodic analysis of Certified Reference Materials CASS-2 (coastal seawater) and SLEW-1 (estuarine water) provided by the National Research Council of Canada (NRCC) through the Marine Analytical Chemistry Standards Program (MACSP). Aliquots (200 ml) were analysed under identical conditions to the samples. The results of these quality control procedures are summarised in Table 2.5.

Table 2.5. Certified and determined concentrations of dissolved Cd, Cu, Pb and Zn in Certified Reference Materials CASS-2 and SLEW-1.

Dissolved	CASS-2 (Sands, pers. comm.)				SLEW-1 (Zhou, pers. comm.)			
metal	Certified	SD	Determined	SD	Certified	SD	Determined	SD
Cd ($\mu g\ l^{-1}$)	0.019	0.004	0.024	0.004	0.018	0.003	0.019	0.002
Cu ($\mu g\ l^{-1}$)	0.675	0.039	1.067	0.006	1.760	0.090	1.570	0.030
Pb ($\mu g\ l^{-1}$)	0.019	0.006	ND[1]	-	0.028	0.007	ND[1]	-
Zn ($\mu g\ l^{-1}$)	1.970	0.120	0.825	0.001	0.860	0.150	0.760	0.040

[1]Below limit of detection.

It must be stressed that, except for Cu, the certified values were at or below the LOD obtained by the analyses and below both the literature LOD and the NRA requirements. The quality control procedures did not therefore give useful information on the accuracy of the analyses, but confirmed that contamination of the samples throughout the procedure was practically nil.

2.3.3 Particulate metal analyses

All plastic and glassware were soaked in Decon, acid-washed (10 % HNO_3, AnalaR) and thoroughly rinsed with MilliQ. Where possible, operations were carried out in a laminar flow hood, with due attention to the risks of contamination at all times. The filters from Section 2.3 were rinsed with MilliQ to remove salt, placed in covered plastic petri dishes and dried to constant weight. Very few of the filters held sufficient SPM for digestion, due to a combination of low concentrations and high percentage organic loadings that seemed to block the filters and prevent the filtration of more than *circa* 500 ml of sample. Those filters with most SPM (>5 mg) were placed in Teflon bombs with conc. HNO_3 (AristaR, 10 ml). These were then sealed with screw lids, placed in stainless steel heating collars to ensure even heating and heated on a hotplate in a fume cupboard (70 °C, 8 h). Once cooled, the bombs were opened and the digests partially diluted with MilliQ (10 ml). They were then filtered (Sartorius 25 mm celluluse acetate, 0.45 μm pore size) with careful rinsing of the bombs and filter funnel. The filtrates were transferred to volumetric flasks (25 ml) and made up to the mark with MilliQ. The digests were analysed by flame AAS (GBC Scientific 902) for Fe, Mn and Zn and by graphite furnace AAS (Perkin Elmer 4100ZL) for Cd, Cu and Pb. All instruments were calibrated with acidified metal standard solutions (BDH Spectrosol).

Consideration was given to the use of an in-house large-volume (8 l) pressure filtration apparatus (Glegg, 1987) which enables collection of sufficient SPM for meaningful analysis, but the idea was dismissed owing to the presence of a metal filter support causing unquantifiable contamination of dissolved and particulate metal samples.

2.3.4 Dissolved and particulate Hg analyses

Concentrations of Hg in the dissolved phase were determined directly by pre-treatment with bromide/bromate to reduce organo-mercury species and analysis by cold vapour generation with PSA Merlin atomic fluorescence detection (Stockwell and Corns, 1994). Particulate Hg concentrations in the SPM digests were determined in the same manner. The instrument was calibrated with acidified Hg standard solution (BDH Spectrosol). The samples were analysed against matrix-matched blanks with the results being blank-deducted. The limit of detection was 0.37 ng l^{-1} in water samples and 0.1 μg l^{-1} in digests (*circa* 0.02 μg g^{-1} of SPM).

2.4 Sampling and Analysis of Sediment variables

The sampling of sediment characteristics (viz. total and bioavailable fractions of Cd, Cu, Hg, Pb and Zn and particle size analysis) was conducted semi-annually in concert with a spring or neap survey of primary and secondary variables and took place between tidal states. Replicate samples (between three and six) were collected from each of sites one to four (Fig. 2.2) with a Shipek grab. One replicate from each site was reserved for the determination of sediment variables and the remainder were put aside for benthic macrofaunal analysis. All plastic and glassware were soaked in Decon, acid-washed (10 % HNO_3, AnalaR) and thoroughly rinsed with MilliQ. Surficial sediments to a depth of *circa*

1-2 cm were taken from the grab samples with a plastic scoop and placed in labelled screw-top glass jars (250 ml). The samples were frozen at <-18 °C pending analysis.

2.4.1 Particle size analysis

Particle size analysis was performed using a combination of wet sieving and Coulter counting (Buller and McManus, 1979) to yield the percentage weights of gravel (>2 mm), sand (<2 mm), silt (<63 μm) and clay (<4 μm). The sample was thoroughly mixed in the jar with a plastic spatula to evenly redistribute the original porewater and two sub-samples (*circa* 50 g) were carefully weighed in tared porcelain crucibles. One sub-sample was dried to constant weight (105 °C, *circa* 24 h) and the percentage water content was calculated by deduction. The second sub-sample was thoroughly dispersed in sodium hexametaphosphate solution (1 g l^{-1}) for 1 h with gentle agitation and then wet-sieved across a nest of brass sieves (2 mm, 63 μm, fines pan). Each fraction was thoroughly rinsed through with distilled water. The retained material from the 2 mm sieve and the 63 μm sieve was washed into tared porcelain crucibles and dried to constant weight, whilst the fine material that passed the 63 μm sieve was poured into a plastic bottle (2 l) and retained for further analysis. The dry weight of the second sub-sample was calculated from the percentage water content of the first, enabling the percentage weight of the gravel (>2 mm), sand (<2 mm) and fine fractions (<63 μm) to be calculated.

The fine material (<63 μm) was further fractionated by Coulter counting (Coulter Electronics Ltd. Industrial model D) into silt (<63 μm) and clay (<4 μm) fractions. The instrument was originally developed for counting blood cells and works on the following principle: particles in an electrolyte suspension under continual mechanical stirring are drawn by manometer pressure through a small aperture (140 μm) across which a constant electric field is maintained. Interruptions to this field as particles pass through it produce a series of signals proportional to the volume of electrolyte displaced by each particle. By manipulating the sensitivity of the instrument through different settings of aperture current, signal attenuation and signal threshold, counts are made of particles in discrete volume bands in a known electrolyte volume (0.5 ml).

The instrument was calibrated according to manufacturer's instructions with commercially available latex beads (12.5 μm diameter) and blank readings of clean electrolyte solution (Isoton II, Coulter Electronics Ltd.) were taken in duplicate at each combination of settings. The plastic bottle containing the fine fraction was shaken vigorously to resuspend the particles evenly and several drops of suspension were added to the clean electrolyte. Several counts were made on the lowest combination of settings, to ensure that the mean count fell in the range 10,310 (the minimum number for 5 % particle loss) to 14,580 (the maximum for 10 % coincidence counts). The settings were returned to the starting conditions and a systematic series of counts was made in triplicate from coarse to fine settings. The assumption that all particles were spherical allowed the volume bands to be converted to diameter bands. The assumption that the density of the particulate material was constant allowed the calculation of percentage weights of silt (<63 μm) and clay (<4 μm) fractions.

2.4.2 Analysis of trace metals in sediments

Owing to the greater sorptive capacity of the fine fraction for trace metals (De Groot *et al.*, 1976) and the high anthropogenic content (battery fragments, paint flakes etc.) of the coarse fraction of Sutton Harbour sediments, the samples were fractionated and treated separately as fine (<63 μm) and coarse (>63 μm) fractions. Each underwent two assessments: hot nitric acid digestion (Millward and Herbert, 1981) and bioavailable/non-detrital metals concentration (those which are weakly bound to the sediments and therefore available to biota during ingestion) using a 25 % acetic acid (HOAc) digest (Chester and Hughes, 1967; Loring and Rantala, 1988). Heavy metal analyses were performed using standard procedures. All plastic materials and glassware were soaked in Decon, acid-washed (10 % HNO_3, AnalaR) and thoroughly rinsed with MilliQ. Where possible, operations were carried out in a laminar flow hood, with due attention to the risks of contamination.

Each sediment sample (*circa* 5 g) was fractionated through a nylon sieve (63 μm mesh diameter) into a Millipore glass filter funnel. The material retained on the sieve (>63 μm fraction) was washed into a plastic petri dish and covered. That present in the funnel was filtered (Sartorius 47 mm cellulose acetate, 0.45 μm pore size) under vacuum, rinsed with MilliQ to remove salts, the filter transferred to a plastic petri dish and covered. Both size fractions were then dried to constant weight (30°C, *circa* 12 h) in covered plastic petri dishes.

For bioavailable metal analysis, weighed samples of each dried sediment fraction (*circa* 0.5 g) were transferred to polyethylene digestion vessels and 25 % v/v HOAc (AristaR, 20 ml) added. The vessels were covered with watch glasses and left to digest in the laminar flow hood (room temperature, 12 h). The digests were filtered (Sartorius 25 mm cellulose acetate, 0.45 μm pore size) with careful rinsing of the digestion vessels and filter funnel. The filtrates were transferred to volumetric flasks (50 ml) and made up to the mark with 25 % v/v HOAc (AristaR).

Further weighed samples of each dried sediment fraction (*circa* 0.5 g) were transferred to Teflon bombs (50 ml) and conc. HNO_3 (AristaR, 20 ml) added. The bombs were sealed with Teflon screw lids, placed in stainless steel heating collars to ensure even heating and heated under pressure on a hotplate in a fume cupboard (70 °C, 8 h). Once cooled, the Teflon bombs were opened and the digests partially diluted with MilliQ (20 ml). The

(20 ml). The digests were filtered (Sartorius 25 mm celluluse acetate, 0.45 μm pore size) with careful rinsing of the Teflon bombs and filter funnel. The filtrates were transferred to volumetric flasks (50 ml) and made up to the mark with MilliQ.

The HNO_3 and HOAc digests were analysed by flame AAS (GBC Scientific 902) with microcup for Fe, Mn and Zn, by graphite furnace AAS (Perkin Elmer 4100ZL) with standard addition for Cd, Cu and Pb and by cold vapour generation with PSA Merlin atomic fluorescence detection for Hg. All instruments were calibrated with matrix-matched standards for each digest using acidified metal standard solutions (BDH Spectrosol). The detection limits achieved by each method can be seen in Table 2.6.

Table 2.6. Comparison of detection limits (3σ of blank values) for each analysis technique and sediment digest with NRA requirements.

Metal	Technique	HNO_3 digest[1]		25 % HOAc digest[1]		NRA (25 % HOAc)
		($\mu g\ l^{-1}$)	($\mu g\ g^{-1}$)	($\mu g\ l^{-1}$)	($\mu g\ g^{-1}$)	($\mu g\ g^{-1}$)
Fe	FAAS	1.1	c. 0.11	0.5	c. 0.05	-
Mn	FAAS	0.1	c. 0.01	0.7	c. 0.07	-
Zn	FAAS	0.1	c. 0.01	0.1	c. 0.01	20.0
Cd	GFAAS	0.2	c. 0.02	0.2	c. 0.02	1.00
Cu	GFAAS	1.5	c. 0.15	1.8	c. 0.18	20.0
Pb	GFAAS	2.3	c. 0.23	6.6	c. 0.66	10.0
Hg[2]	AFS	0.1	c. 0.01	0.1	c. 0.01	0.20

[1]Approximately 0.5 g of material digested in each case. [2]Operator-imposed LOD; Actual LOD for Hg method 0.37 ng l^{-1} (Stockwell and Corns, 1994).

2.4.3 Quality control

The quality of the determinations made in Section *2.4.2* was assured by the periodic analysis of certified reference materials PACS-1 (coastal sediment) and BCSS-1 (shelf sediment), provided by the NRCC through MACSP. Samples (*circa* 0.5 g) of each were digested with hot nitric acid and analysed under identical conditions to the samples. The results of these quality control procedures are summarised in Table 2.7.

These standard sediments need not be of the same form as Sutton Harbour sediments and the metals need not be held in the same phases. Therefore, they may be inaccessible to a hot nitric acid digest. For example, PACS-1 and BCSS-1 have a high organic carbon content (3.7 % and 2.2 %, respectively) and these organic coatings are resistant to acid attack, resulting in low recovery of detrital Fe, Mn and Zn, but high recovery of non-detrital Cu and Pb in both cases. We must therefore take metal concentrations as operationally defined values, the reproducibility of which allow spatial and temporal comparisons to be made.

Table 2.7. Certified and determined concentrations of Fe, Mn, Cd, Cu, Pb and Zn in Certified Reference Materials. Certified concentrations are total (HF) digests; determined concentrations are HNO_3 digests, n = 3.

Metal	PACS-1 ($\mu g\ g^{-1}$)			BCSS-1 ($\mu g\ g^{-1}$)		
	Certified	Determined	Recovery	Certified	Determined	Recovery
Fe	48500 ± 835	15617 ± 90	32 %	32400 ± 966	12239 ±244	38 %
Mn	470 ± 12	140 ± 3	30 %	229 ± 15	98 ± 1	43 %
Cu	452 ± 16	381 ± 19	84 %	18.5 ± 2.7	16.5 ± 1.0	89 %
Pb	404 ± 20	342 ± 1	85 %	22.7 ± 3.4	32.5 ± 6.1	143 %
Zn	824 ± 22	278 ± 1	34 %	119 ± 12	49 ± 6	41 %

2.5 Sampling and Analysis of Benthic Macrofauna

The remaining replicates of sediment samples collected between May 1992 and November 1994 were reserved for benthic macrofaunal analysis. On-board, replicates were gently agitated by hand in individual buckets of water to break up the cohesive sediments and washed through a 500 μm sieve. The retained macrofauna and coarse material were washed into screw-top glass jars (250 ml) and preserved in 4 % buffered formalin (Warwick, 1983). Samples from May 1992 and November 1992 were treated with a protein stain (Rose Bengal), but this precluded identification of certain species from natural colour, so this technique was not used after November 1992.

2.5.1 Identification and enumeration

Benthic macrofaunal samples were separated from the retained (>500 μm) sediment fraction by freshwater elutriation: the glass sample jars were placed on a sieve (500 μm) in a sink under a running tap until the overspill ran clear. The material remaining in the jar was carefully checked for heavier fauna such as molluscs and large polychaetes. Identification was carried out to species level where possible, using compound and stereoscopic binocular microscopes with appropriate keys (Hayward and Ryland, 1990a and 1990b). Enumeration was effected using head counts to avoid duplication. To enable direct comparison with previous samples, collected by Day grab in May 1990 and by diver in November 1991, all data were normalised to 1 m^2 of surface sediment.

2.6 Flushing Experiments

Two investigations into the nature and extent of hydrodynamic flushing in Sutton Harbour have been conducted. The first was a component of the initial environmental assessment of the Harbour in May 1990 (Houston, 1990) and the second was carried out in July 1994 following impoundment as part of a joint investigation (Wilkinson, 1994). Both experimental designs employed the same principle: dosing of the harbour waters with a biological tracer organism and subsequent sampling at intervals, in order to assess the rate of removal and hence water renewal times. The tracer used for each experiment (manufactured by Biotrace and supplied by the NRA) contained *Bacillus globigii* spores, a microorganism shown to exhibit conservative behaviour in seawater over a period of days (Pike *et al.*, 1969) and used extensively by the Water Industry, particularly to assess the pattern of dispersion from projected sewage outfalls into marine and fresh waters (Houston *et al.*, 1989). The spores are not indigenous to the marine environment and do not germinate in seawater (no addition) but exhibit relatively high survival rates over several days (measurable mortality after 1 week from insolation; limited unquantifiable removal by zooplankton grazing). The experimental details of the May 1990 study are documented elsewhere (Houston, 1990), but were the basis of and therefore similar to the July 1994 joint investigation, described here and by Wilkinson (1994).

The tracer was provided in suspension (8 l) at a nominal titre of 4 x 10^9 cfu l^{-1}. The suspension was pumped on to the water surface *via* a 12V peristaltic pump, while the dosing vessel (dinghy with outboard engine) followed a prescribed track around the harbour (Fig. 2.7). Dosing was carried out around LW, and was completed one hour before the gates were opened to freeflow, so that maximum dispersion would be effected by the limited flood tide. At regular intervals during dosing, samples were taken from the pumped stream in order to check the injection titre and the carboy containing the suspension was vigorously shaken to resuspend the microorganisms evenly. Following dosing, the dinghy was dry-berthed outside the harbour and thoroughly disinfected before further use. Sampling for *B. globigii* was conducted at the intervals shown in Table 2.8.

Table 2.8. Timetable for the *B. globigii* tracer study, July 1994.

Date	Action	Time (BST)			Nominal
		Start	HW/LW	Finish	time (h)
11/7/94	Dosing harbour	1400	1424	1500	
	Baseline sampling at HW	2000	2035	2100	0
12/7/94	Sampling at LW	0230	0248	0400	6
	Sample delivery (Exeter)	0900			
	Sampling at HW	0830	0901	0930	12
	Sampling at HW	2045	2113	2145	24
13/7/94	Sample delivery (Exeter)	0900			
	Sampling at HW	2120	2150	2220	48
14/7/94	Sample delivery (Exeter)	0900			
15/7/94	Sampling at HW	1030	1059	1130	84
18/7/94	Sample delivery (Exeter)	0900			

Surface and bottom samples were obtained from each of the fifteen sites shown in Fig. 2.7 by IOS bottle, poured into bar-coded glass bottles and stored in a coolbox at <4 °C for immediate transport to laboratories at NRA Exeter, where counts of *B. globigii* (cfu(100 ml)$^{-1}$) were determined by membrane filtration onto selective media.

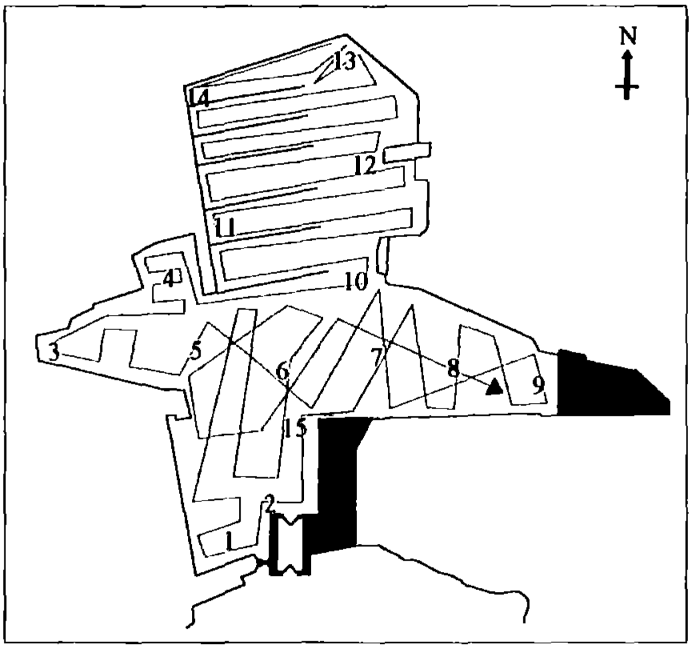


Fig. 2.7. Plan of dosing track and sampling locations for *B. globigii* tracer study, July 1994 (Wilkinson, 1994).

2.7 Continuous Monitoring

The bimonthly monitoring strategy was intended to detect seasonal and inter-annual variability in water quality. Within that strategy, separate spring and neap tidal watches were designed to provide information on processes occurring at semi-diurnal, diurnal and weekly frequencies. However, in order to ensure that these processes were interpreted correctly, it was necessary to monitor key water quality parameters continuously inside the harbour at certain locations, for extended periods of the study. The monitoring instruments used (DMP/pHOX Water Loggers) consisted of a sonde housing various probes (Table 2.9) connected by cable to a top-box, with sufficient power and memory capacity to record data for at least one month at half-hourly intervals.

Table 2.9. Specifications of the DMP/pHOx Water Logger instrument.

Parameter	Range	Resolution
Temperature	-5.00 - 60.00 °C	± 0.02 °C
Salinity	0.00 $\times 10^{-3}$ - 40.00 $\times 10^{-3}$	± 0.02 $\times 10^{-3}$
Dissolved oxygen	0.0 - 200.0 %	± 0.2 %
pH	0.00 - 14.00	± 0.02
SPM	0 - 500 FTU	± 1 FTU

The instruments were provided and maintained by the NRA. One instrument was installed on a marina pontoon at site 6 (Fig. 2.6) taking surface measurements in a central location from February 1993 to October 1994. Two further instruments were installed from February 1994 to October 1994 to give greater coverage throughout the period of commissioning: one was suspended on a buoyed system to obtain readings 0.5 m above the bed at site 6 (Fig. 2.6), the other was installed in the north of the harbour at site 2 (Fig. 2.6) taking surface readings in a more remote location. A routine of fortnightly cleaning and recalibration by NRA staff in concert with data download to laptop PC by the author was established from February 1993 to February 1994. The frequency of these visits was increased to weekly during periods of high fouling growth on the sonde and after February 1994.

A digital low-pass filter was applied to each time series to remove the considerable semi-diurnal and diurnal variability, leaving only the longer-term trends in evidence. The time series were broken in places as a result of equipment failure and routine maintenance. For this reason a Doodson X_0 filter (Pugh, 1987) was chosen because only 9½ h of data are lost from the beginning and end of each filtered series. The performance of the filter can be seen in Fig. 2.8 in comparison with a 12½ h moving average, which is simple but does not truly filter out all high frequency variations. Furthermore, the moving average rapidly goes out of phase with semi-diurnal cycles.

2.7.1 Validation and interpolation of primary variables

The continuous monitoring database provided an important means of validating the bimonthly measurements and *vice versa*. Continuous filtered values of temperature, salinity and dissolved oxygen are shown in Fig. 2.9 from May 1993 to October 1994, together with superimposed tidally-averaged values of the matching parameters from the bimonthly surveys of water temperature, salinity and dissolved oxygen. The two series are in good agreement in each case; the slight disparity between the dissolved oxygen series may be explained by the method of measurement. The oxygen electrode requires a steady flow of water across the membrane (*circa* 15 cm s^{-1}) for a true reading. These conditions were generally met during the on-board measurements due to the movement of the survey vessel as it manoeuvred on station, but may not always have been met during periods of slack water for the *in situ* continuous monitors.

Another value of the continuous monitoring was in interpolation between survey dates. It is clear from Fig. 2.9 that the snapshot nature of each survey led at times to biasing in the subsequent time series, with ephemeral features, particularly with respect to salinity and dissolved oxygen, lost in the intervening period. For example, a significant rain event in May 1993 caused a considerable pulse of low salinity water, in theory laden with nutrients, which was not detected by either of the May 1993 surveys. A considerable supersaturation

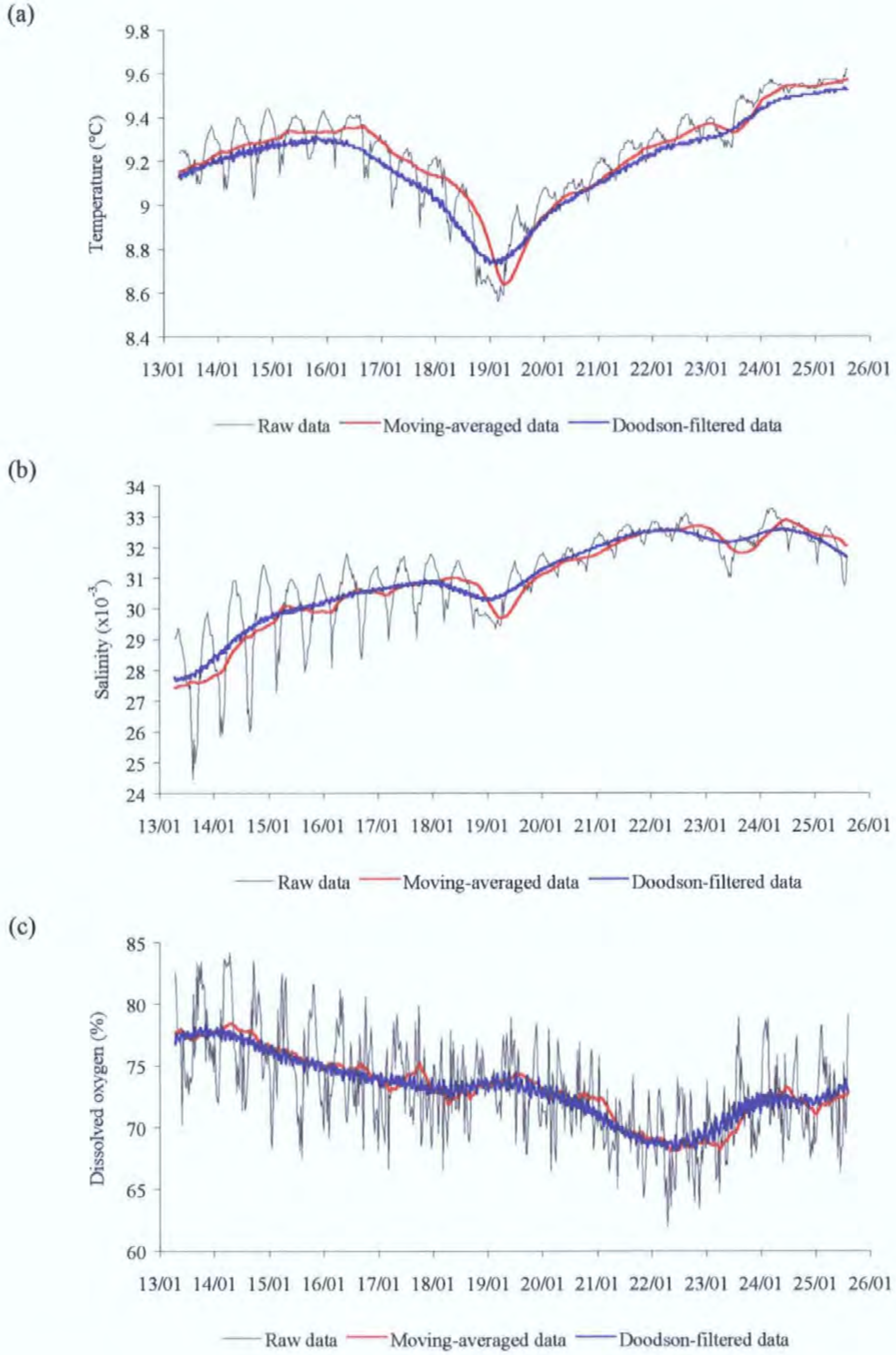


Fig 2.8. Comparison of raw, moving-averaged and Doodson-filtered data from continuous water monitoring in January 1994 for (a) temperature, (b) salinity and (c) dissolved oxygen.

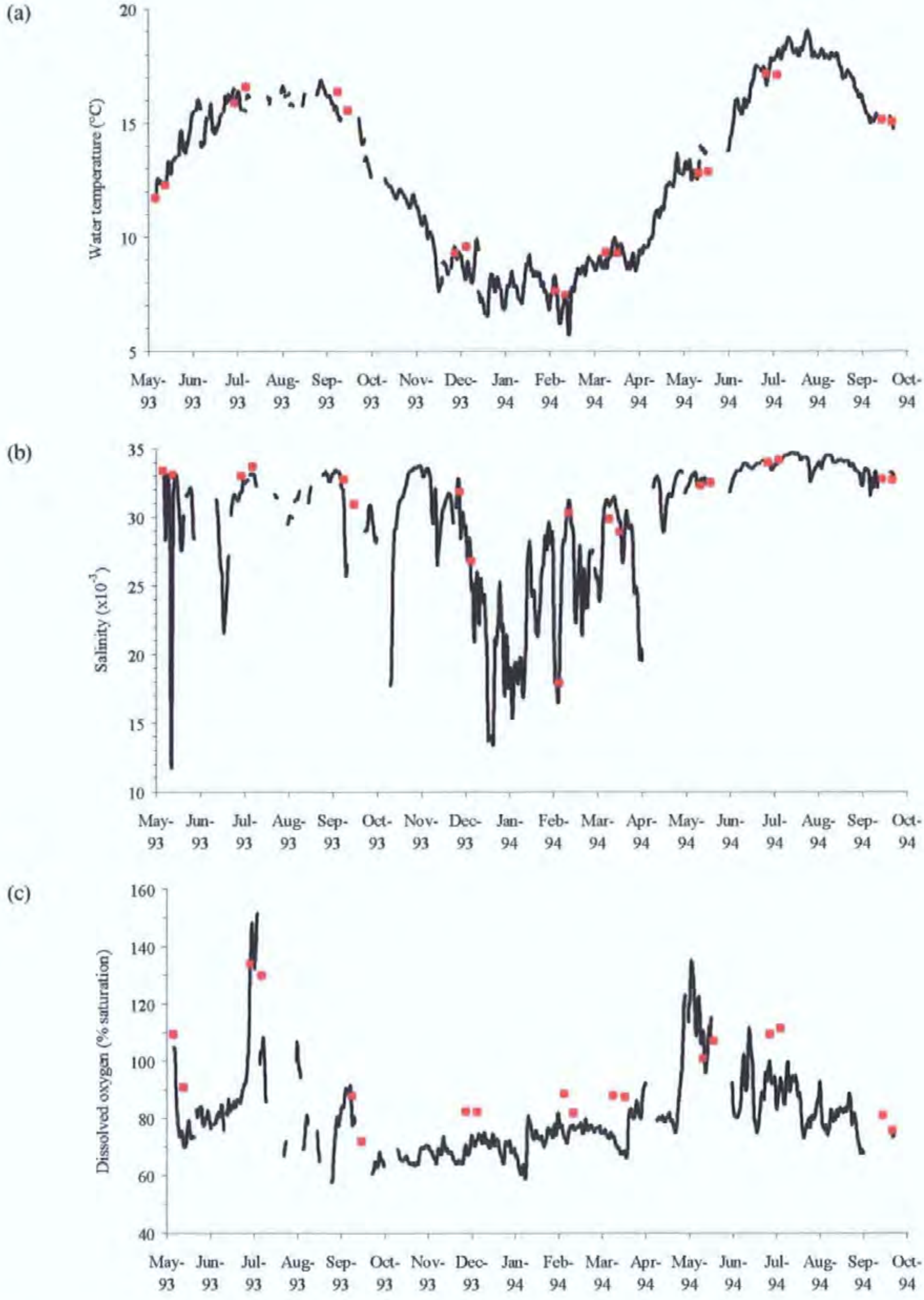


Fig. 2.9. Doodson-filtered time series of (a) temperature, (b) salinity and (c) dissolved oxygen inside Sutton Harbour (black lines) with superimposed routine monitoring tidal averages (red squares).

in oxygen was observed in May 1994 due to primary production that had declined considerably at the time of the May 1994 surveys. Careful consideration of these continuous data led to a better understanding of the timescales and extents of natural variability in the Sutton Harbour system.

Chapter Three

Hydrodynamics
and
Hydrography

Chapter 3 - Hydrodynamics and Hydrography

3.1 The Hydrodynamic Regime

Plymouth (Devonport) has a semi-diurnal tidal regime with mean spring and neap tidal ranges of 4.7 m and 2.2 m respectively (Table 3.1). Prior to impoundment the harbour, with an average depth of *circa* 0 m above chart datum (ACD), was subject to these full ranges (Fig. 3.1a) and up to 40 % of the harbour area dried at extreme springs. The main effect of impoundment was a major reduction in the volume of harbour water exchanged with the outside during each tidal cycle (Fig. 3.1b) and hence a reduction in the flushing efficiency of the system.

Table 3.1. Tidal levels for Plymouth (Devonport) (Hydrographer of the Navy, 1988).

Tidal level	Height (m ACD)
Mean high water springs (MHWS)	5.5
Mean high water neaps (MHWN)	4.4
Mean low water neaps (MLWN)	2.2
Mean low water springs(MLWS)	0.8

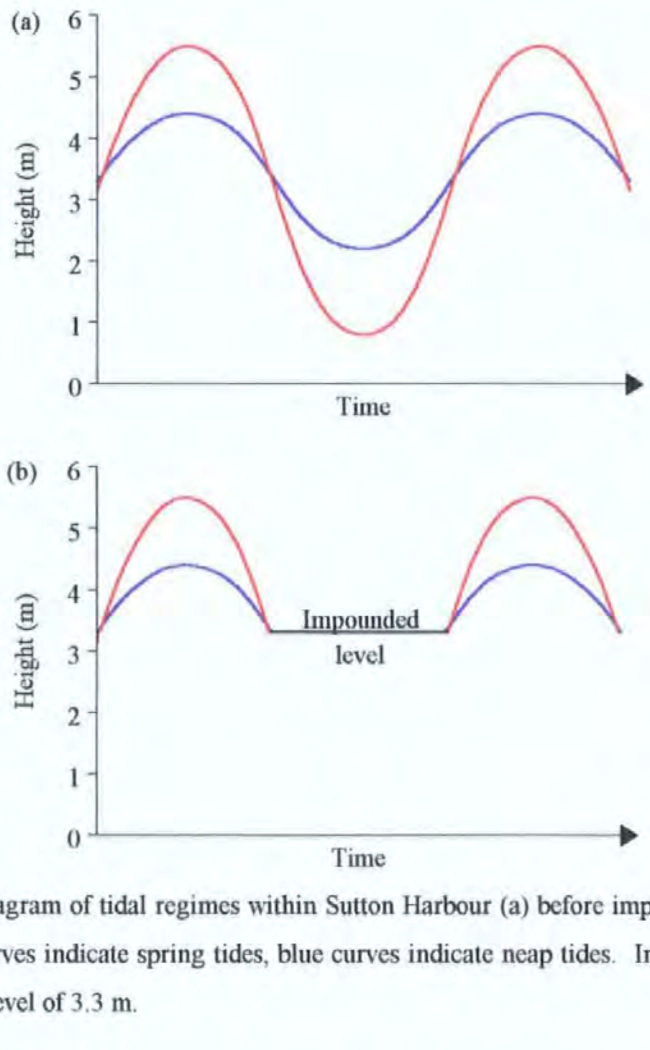


Fig. 3.1. Schematic diagram of tidal regimes within Sutton Harbour (a) before impoundment and (b) after impoundment. Red curves indicate spring tides, blue curves indicate neap tides. Impounded level denotes the target mean water level of 3.3 m.

The NRA has stipulated that the mean of the water levels inside the harbour at the end of impoundment in any 14 day period should not exceed 3.3 m ACD. In order to achieve this in practice, the operators should close the harbour to free flow when the level inside reaches 3.5 m ACD (*circa* HW+3h) and open it to free flow when the level outside the harbour corresponds with that inside (*circa* HW-3h). Log sheets are provided by the NRA for the lock operators to record precise times and water levels at the beginning and the end of each impoundment, in order to allow calculation of the mean impounded levels. Analysis of typical locking operations in August 1994 (Table 3.2) suggests that, allowing for a depth reduction of 1.4 cm/lock, ten locking operations (mean - one standard deviation) would result in a final impounded level of 3.35 m ACD and 28 (mean + one standard deviation) in 3.1 m ACD. It is evident from Table 3.2 that during August 1994, the prescribed flushing was not achieved: the mean water level at the beginning of impoundment was 0.3 m too high, resulting in a similar excess at the end. A simple rule of thumb that may help the lock operators is as follows: when deeper water is required in the harbour for operational purposes, for every 0.1 m of impounded level above 3.5 m ACD, it should be impounded at 0.1 m below 3.5 m ACD on a separate occasion during the same 14 day period. In this way the prescribed impounded levels could be routinely achieved within operational constraints.

Table 3.2. Statistical analysis of lock operations between 01/08/94 and 31/08/94.

Parameter	August 1994
Number of impoundments, n.	57
Mean number of lock operations per impoundment.	19 ± 9
Mean reduction in water level (cm) per lock operation.	1.4 ± 1.1
Mean water level (m) at start of impoundment.	3.81 ± 0.07
Mean water level (m) at end of impoundment.	3.59 ± 0.11

Concern about the impact of impoundment on water quality suggested that the flushing time of the harbour be investigated thoroughly. This was examined using the criterion of the time required to renew 95 % of harbour water. Two complementary approaches to this problem were made, one theoretical and one practical.

3.1.1 Theoretical flushing times

The theoretical flushing times for the harbour before and after impoundment were calculated on a tidal volume exchange basis, from total volumes and tidal volumes at MHWS and MHWN provided by the NRA (Millward, 1991), using the following equations:

$$x = \frac{\log(1-w)}{\log(1-\frac{v}{V})} \qquad \text{Eq. 3.1.}$$

$$T = 12.5x \qquad \text{Eq. 3.2.}$$

where x = renewal time (tidal cycles); w = renewal factor (i.e. 0.95); v = tidal volume (m^3); V = total volume (m^3); and T = renewal time (h).

Several large assumptions were necessary in these theoretical calculations: (i) no water leaving the harbour on the ebb tide returned on the flood tide; (ii) full spatial mixing occurred during each tidal cycle; and (iii) most importantly, the tidal range was constant between tidal cycles. The results are summarised in Table 3.3.

Table 3.3. Results of theoretical 95 % water renewal time calculations.

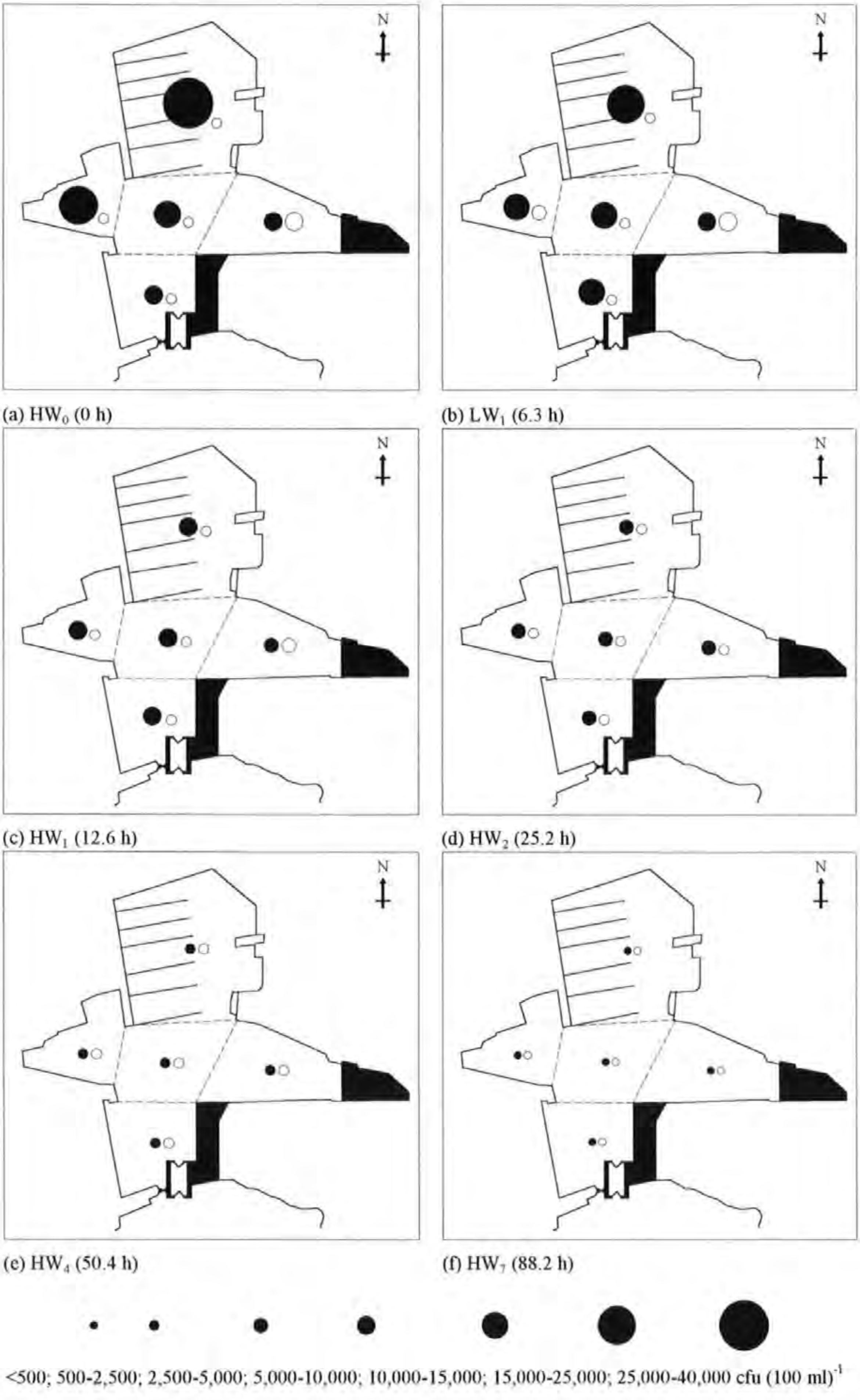
Parameter	MHWS		MHWN	
	Pre-Lock	Post-Lock	Pre-Lock	Post-Lock
Total volume (m^3)	610,740	555,773*	497,930	453,116*
Tidal volume (m^3)	450,580	139,867*	217,410	37,210*
Volume exchange (%)	73.8	25.2	43.7	8.2
95% renewal time (tidal cycles)	2.2	10.3	5.2	35.0
95% renewal time (h)	28	129	65	437

*Volumes corrected using reduction in area from 97500 m^2 to 88400 m^2 after infill and depths from bathymetric survey.

The results show that before impoundment, 95 % water renewal took approximately 1 day at springs and 3 days at neaps, whilst after impoundment it would take over 5 days at springs and over 18 days at neaps. Whilst these results showed that water renewal would be significantly slower after impoundment, the assumptions introduced considerable errors to the actual times, particularly where the effects of tidal range fluctuations were ignored. Thus, the theoretical values were tested using field studies with a conservative tracer.

3.1.2 Tracer studies of water renewal

The results of the tracer studies conducted in May 1990 and July 1994 were analysed to provide 95 % water renewal times, complementing those calculated from volume exchange. One of the problems with the method was that it was essential that the harbour waters were evenly dosed with tracer organisms, to ensure that mixing on the flood tide produced an homogeneous water body. A major difficulty arose because the *B. globigii* suspension settled during dosing. Regular shaking of the container was required to ensure uniform dosing, as the data in Table 3.4 show. The data for each survey during the July 1994 study are shown in Fig. 3.2 as averages for the northern, southern, eastern, western and central boxes within the harbour as delineated by the dashed lines.



(a) HW_0 (0 h)
(b) LW_1 (6.3 h)
(c) HW_1 (12.6 h)
(d) HW_2 (25.2 h)
(e) HW_4 (50.4 h)
(f) HW_7 (88.2 h)

<500; 500-2,500; 2,500-5,000; 5,000-10,000; 10,000-15,000; 15,000-25,000; 25,000-40,000 cfu (100 ml)$^{-1}$

Fig. 3.2. Results of the *B. globigii* tracer surveys in July 1994. Values shown are mean (n=2 to n=5) for each box (delineated by dashed lines). Filled circles are surface samples, open circles are bottom samples.

Table 3.4. *Bacillus globigii* injection titres for dosing events in May 1990 and July 1994.

	Bacillus globigii titre ($x10^9$ cfu l^{-1})	
	May 1990	July 1994
Number of samples, n	11	5
Mean	3.3	4.0
SD	2.4	0.7
RSD	71.8 %	17.2 %

The validity of the theoretical volume exchanges was tested by applying the observed volume exchange rates from each survey period to the depth-averaged initial mean spore concentrations. Tidal volumes were calculated using the HW and LW levels for 1990 and the HW and impounded levels (at the start of free flow) for 1994, together with the surface area of the harbour before and after construction estimated from an Admiralty chart (Hydrographer of the Navy, 1988) and a plan of works (NRA, 1991). The minimum tidal levels during the 1990 and 1994 periods were 1.8 m and 3.6 m respectively and the harbour area was assumed to be approximately uniform above these levels, facilitating the tidal volume calculation. Total volumes at each HW were calculated by subtraction from the total volume at MHWS. The volume of original water remaining after each tide ($1-{}^{v}\!/_{V}$ in Eq. 3.1) was then applied in succession to the preceding mean total spore concentration until the point of 95 % water renewal was reached (Fig. 3.3). It is apparent that these 95 % renewal times (41 h and 240h) were within 12 % and 16 % of the mean of the spring-neap range of renewal times in Table 3.3 (46.5 h and 283 h) for 1990 and 1994.

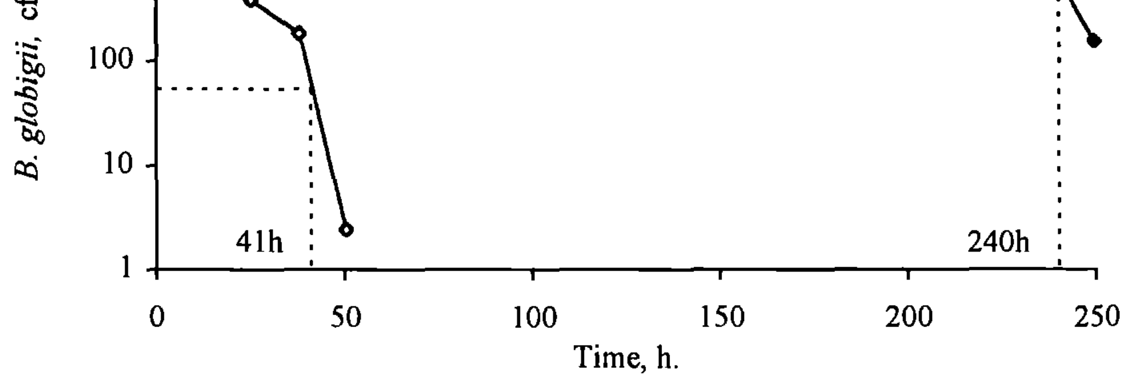


Fig. 3.3. Theoretical reduction in *B. globigii* concentrations from initial mean concentrations by volume exchange (calculated from each tidal cycle during the May 1990 and July 1994 study periods). Horizontal dashed lines represent 95 % reduction from initial concentrations.

The *B. globigii* concentrations at all stations were averaged to give mean surface and bottom concentrations and mean total concentrations for the harbour at each sample interval. These results are plotted on a logarithmic scale in Fig. 3.4. Removal of *B. globigii* from the harbour in each case was approximately exponential, so that an exponential regression was performed on each data set. The equation of each of these regression lines was of the form

$$y = c.e^{-mx} \qquad \text{Eq. 3.3.}$$

Substitution of y as 5 % of the initial mean concentration in each case gave x, the 95 % water renewal time. The values of x and y for 95 % water renewal are indicated by the dashed lines in Fig. 3.4. The depth-averaged 95 % water renewal times for 1990 and 1994 (Fig. 3.4a) were 45 h and 72 h respectively. For 1990, this fell precisely between the theoretical spring (28 h) and neap (65 h) renewal times (Table 3.3) and therefore gave good agreement with them. Separating these data into mean surface and bottom concentrations (Fig. 3.4b) revealed the overbearing effect of the surface data on the vertical mean at each station, as a direct result of low vertical mixing. The renewal time for the bottom water is just under twice that for the surface water, owing to a slightly lower rate of removal. This evidence suggests that exchange of the surface water with the outside is considerably more efficient than that of bottom water, but confirms that an efficient flushing of the entire water column took place before impoundment.

The situation was more complicated after impoundment in 1994. The depth-averaged 95 % water renewal time of 72 h (Fig. 3.4a) is much lower than the anticipated range of 129 h to 437 h. Separating the data into mean surface and bottom concentrations (Fig. 3.4c) shows that the rate of decay in the bottom water is much slower (75 %) than before impoundment and the renewal time falls within the anticipated range. However, the rate of decay in the surface waters is only 33 % slower than before impoundment. The inference from this observation is that a mechanism of removal other than tidal exchange takes place. Two possible explanations were postulated: First, there is now much less vertical mixing, and this, combined with continued efficient exchange at the surface, means that a discrete layer of water enters the harbour during the flood and then leaves almost unmodified by vertical mixing during the ebb. Second, the mortality of the *B. globigii* spores was considerably greater than anticipated, resulting in a departure from conservative behaviour. These possibilities are examined below.

The first hypothesis was examined using data obtained at the first low water during each tracer study in order to calculate a coarse estimate of the vertical mixing due to one ebb tide. For each site, the ratios of bottom to surface of concentration of *B. globigii* were calculated for HW_0 and LW_1. The value at HW_0 was then deducted from the figure for LW_1 (to correct for initial bottom concentrations) and the resultant ratio expressed

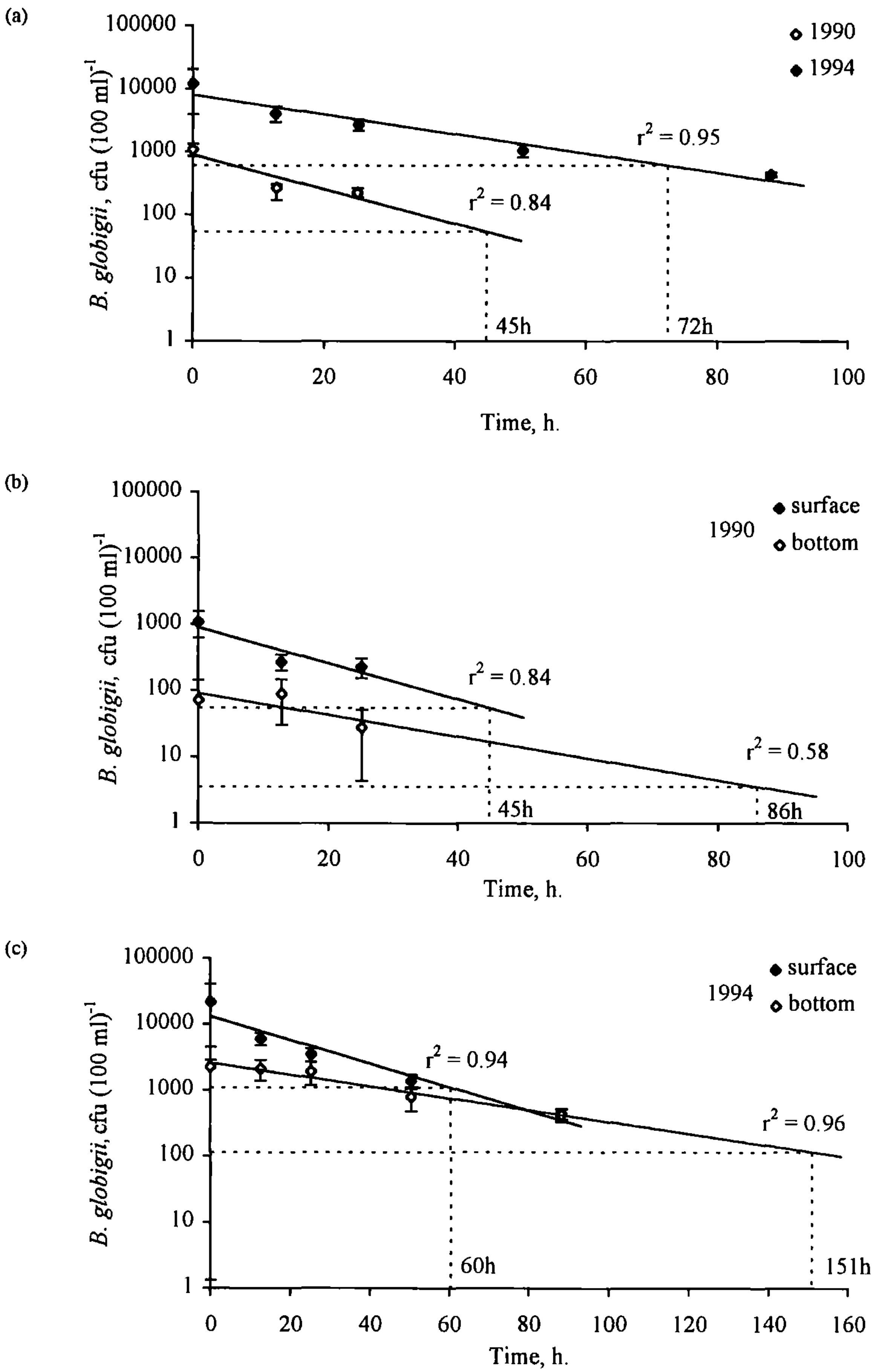


Fig. 3.4. *B. globigii* concentrations against time for (a) depth-averaged results, (b) surface and bottom 1990 and (c) surface and bottom 1994. Error bars show one standard deviation, r^2 quoted for each regression. Dashed lines represent 95 % renewal from initial concentrations.

as a percentage value. The percentage mixing values were divided into bands of low (<10 %), medium (10 % to 50 %) and high mixing (>50 %) and are shown in Fig. 3.5 with an arrow to indicate net downward flux (positive ratio) or upward flux (negative ratio). These must only be viewed as an indication of vertical fluxes, since during flood and ebb the surface and bottom waters move relative to each other and absolutely, resulting in spatial causes of changes in vertical distribution.

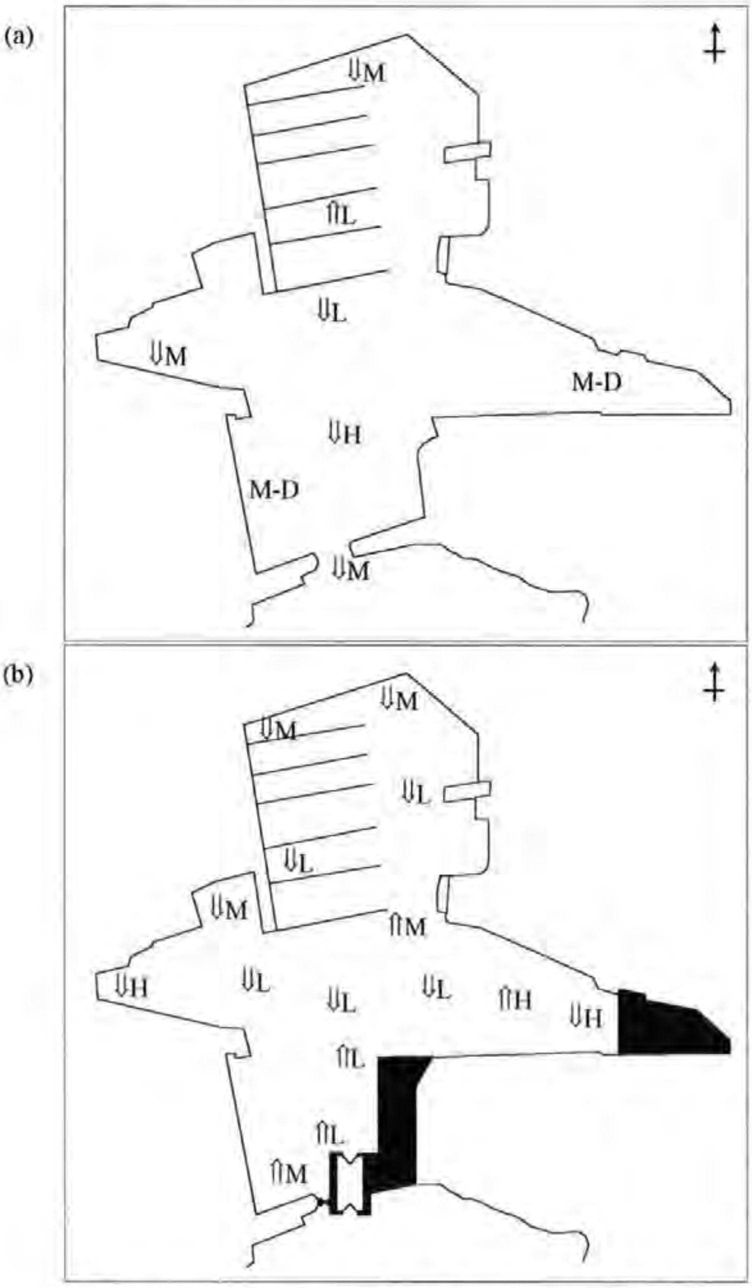


Fig. 3.5. Estimated vertical mixing bands for the first ebb tide during *B. globigii* tracer studies in (a) May 1990 and (b) July 1994. Letter denotes mixing band: Low mixing (L) <10 %, medium mixing (M) 10 % to 50 % and high mixing (H) >50 %. The direction of each arrow indicates net upward or downward mixing. M-D indicates mid-depth only samples at LW.

In May 1990 (Fig. 3.5a), the greatest downward fluxes were observed in the centre of the harbour and at the harbour mouth, consistent with the relatively high tidal energy at these locations. The lowest vertical fluxes were observed in the southern half of Sutton Marina, consistent with restricted water movement in shallower water around the pontoons. The most northerly site showed a medium downward flux. In July 1994 (Fig. 3.5b), a distinct contrast was seen with the mixing in May 1990. The greatest downward fluxes were observed in the extremes of the eastern and western arms. The adjacent site in the eastern arm had a high upward flux suggesting that an enclosed circulation cell was present. Low mixing was observed in the central and harbour mouth areas where high mixing previously occurred. The extreme north and the marina areas continued to exhibit moderate and low vertical mixing respectively. The area to the west of the lock chamber that is not aligned with water flow through the harbour mouth showed a low to moderate upward flux.

It is apparent from these differing fluxes that the impoundment affected the complex circulation patterns in the harbour. Of particular interest is the increased downward flux in the western arm, matched by a large downward flux in the eastern arm. These regions were considered to be at risk from reduced flushing, but it would appear that they remain at least as energetic as before. However, on average, the extent of vertical mixing in the harbour has decreased by a factor of 5 following impoundment (Table 3.5), whilst the RSD about this mean has increased by a factor of 6 despite a larger sample population. Therefore, rapid surface water renewal cannot be attributed to enhanced downward mixing following impoundment and must be attributed to efficient exchange with the water outside the harbour.

Table 3.5. Mean percentage mixing for the entire harbour after the first ebb tide during the May 1990 and July 1994 *B. globigii* tracer studies.

	May 1990	July 1994
Number of samples, n	6	15
Mean mixing value (%)	25	5
SD (%)	28	33
RSD (%)	110	651

The second hypothesis of unexpectedly rapid mortality of *B. globigii* was tested by the NRA in their laboratories (Babbedge, pers. comm.). Spores were added to a large drum of salt water (S *circa* 23 $\times 10^{-3}$) and thoroughly mixed. Near-surface and near-bottom samples and salinity measurements were taken to define the initial conditions and the drum was stored in the dark for 10 days. Samples and measurements were taken periodically and the results are summarised in Fig. 3.6. During the first seven days the water column became partially stratified at an almost constant rate, after which time the stratification rate reduced to zero and the water column became stable. During the first two day period there was evidence of spore settling, with increasing bottom concentrations and decreasing surface concentrations. The mean surface and bottom counts in the first two days were unchanged showing that early mortality of the spores was negligible. After a further five days, surface and bottom counts had fallen to 0.4 % and 1.4 % of their original values respectively. It must be assumed that mass mortality of the spores took place during this period, although no quantification could be made of absolute settling to the base of the drum or of adherence to the walls of the drum and the drum was not stirred at the end of the experiment to resuspend settled spores that remained viable. The viable spores remaining in suspension after this time showed lower mortality rates over the final three days of the experiment (10 % to 15 % d^{-1}) and were still detected at the end of the 10 day period. Moreover, this experiment was conducted in the dark. Experiments by Pike *et al.* (1969) in which *B. globigii* spores were suspended in seawater and left in the sun in glass bottles showed a detectable mortality after 4 days and a mortality of 16 % to 58 % after 6 days. During this period, the insolation was equivalent to two days of midsummer sunshine. However, concurrent experiments in which spores were kept in the dark revealed negligible mortality after 9 days. The inferences from these combined experiments are as follows: (i) spore settling occurs in still water over a period of a few days; (ii) solar radiation causes a measurable mortality rate after *circa* 1 d of strong insolation; and (iii) viable spores survive in seawater for up to 10 days. With the knowledge that the hydrodynamic marine environment should keep the spores in suspension and mix them between surface and bottom, together with the fact that UV radiation is rapidly attenuated in seawater (Jerlov, 1968), it is concluded that *B. globigii* spores behave as a near-conservative tracer when measured over a period of up to 10 days in sufficiently deep water (*circa* 2 m).

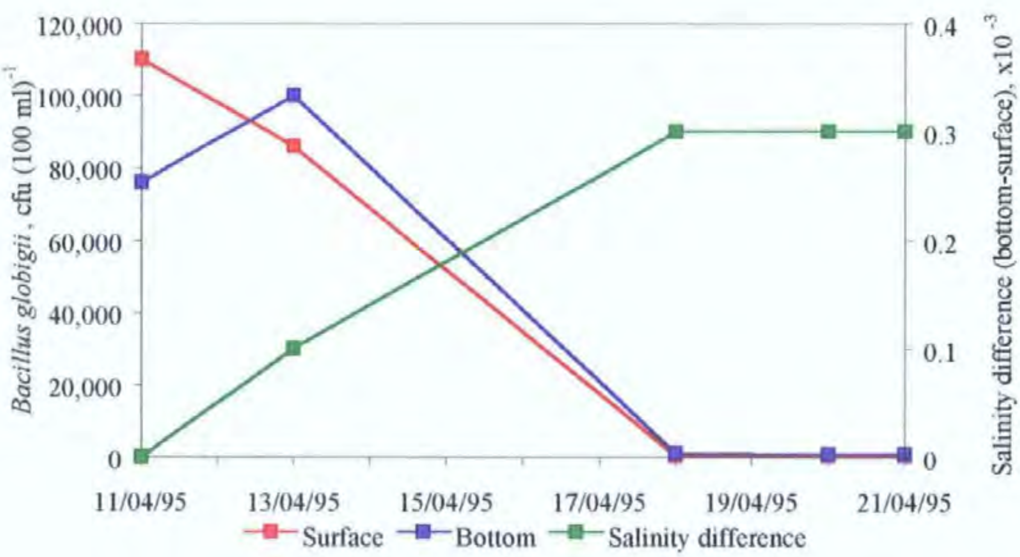


Fig. 3.6. Results of *B. globigii in vitro* mortality experiment conducted by the NRA (Babbedge, pers. comm.).

The greater number of sampling locations in July 1994 provided better spatial resolution of water renewal. The percentage removal of *B. globigii* from five sections of the harbour is shown in Fig. 3.7, with all boxes showing >90 % removal after the 7th HW. It would appear that sectional differences in water renewal after several tidal cycles are slight following drain water from the harbour arms to the outside, thus enhancing circulation, is not considered necessary at present.

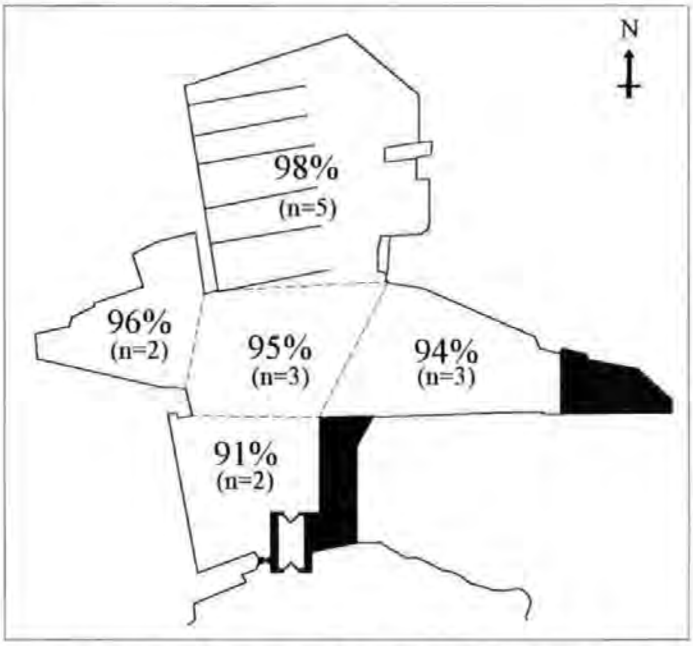


Fig. 3.7. Mean percentage removal (n=2 to n=5) of *B. globigii* from each of five boxes (delineated by dashed lines) after 88.2 h or 7th HW (after Wilkinson, 1994).

It would appear from this study that the exchange of Sutton Harbour surface water with outside was relatively increased after impoundment, given the reduction in volume exchange, due to considerably lower vertical mixing. As a result, bottom water exchange with outside slowed considerably with implications for biogeochemical cycling of pollutants. However, this negative effect of impoundment may be ameliorated to an extent if the nature of the pollutant inputs are considered. The majority of these will be to the surface waters of the harbour, whether associated with brackish water from the Plym, surface runoff and aeolian inputs, or localised anthropogenic inputs from commercial and leisure activities. Whilst some particle-associated pollutants may settle out, other pollutants should continue to be effectively flushed from the harbour. Conversely, those pollutants which do enter the bottom water from the overlying water and from the sediments may accumulate to a greater extent and require periodic full flushing on a spring tide in order to remove them.

The experimental design of the tracer studies enabled the required flushing estimates to be made, but certain flaws in the design limited the worth of the data. One problem in both studies was the lack of homogeneity in the initial concentrations at HW_0, leading to exaggerated estimates of the early flushing as high spore concentrations were removed in the surface water or mixed downwards during the following ebb tide. Future studies might attempt to reduce the problem by co-injecting the surface and bottom waters by means of a pole-mounted pipe maintained at 0.5 m above the bed along the dosing track, although a complete solution would require co-injection at many small depth intervals from surface to bottom, entailing excessive effort for minimal improvement in the data. Spatial homogeneity could also be improved by injection at HW_{-1}, thus allowing a full tidal cycle to mix and disperse the tracer before the baseline survey, or by co-injecting at the harbour mouth during the flood tide. A simple calculation of harbour volume during injection (at LW or during impoundment), and added volume during the ensuing flood, would enable the tracer volume to be proportionally divided between the two dosing periods. The turbulent flow through the harbour mouth might also mix the tracer vertically. A flaw in the experimental design of the May 1990 study that was corrected in July 1994 was the number of subsequent sampling exercises. In the case of May 1990, the interval allowed (from 3rd HW to 15th HW) was too great: *B. globigii* concentrations had fallen almost to zero at an indeterminable point during the intervening period. This resulted in a need for extrapolation rather than interpolation to the point of 95 % water renewal, thereby introducing greater error. It was therefore important in July 1994 (and in future studies) to remain flexible over the timing and number of sampling exercises, in order to react to the emerging flushing rates and thus to ensure data validity.

3.2 The Hydrographic Regime

Both temperature and salinity were employed as hydrographic markers during the study. Temperature is important for its effect on the rate of biogeochemical reactions, the solubility of oxygen and the rate of primary productivity. The main control on seawater temperature is direct insolation, whilst that on freshwater temperature is that of the catchment, i.e. indirect insolation. These factors combined to control water temperature in the harbour before impoundment. It was hypothesised that after impoundment, under reduced water exchange, seasonal variations in water temperature might be more extreme. Summer temperatures might increase through direct insolation, whilst winter values might decrease through convective loss to the atmosphere. More extreme variations would then have an impact on the water quality and ecology of the harbour. Salinity is an important indirect marker of pollutant input to the harbour from external sewage and riverine sources, and of the degree of stratification or mixing of harbour water. Salinity is seasonally controlled by freshwater flow, such that it decreases under spate and increases under dry conditions. However, salinity is a more ephemeral feature than temperature, with greater seasonal and inter-annual variability. It was also hypothesised that impoundment might affect the freshwater fraction of the harbour water through reduced water exchange.

This section of the study is structured in order to examine the variations in temperature and salinity along an expanding time scale, from semi-diurnal and diurnal, through spring-neap cycles, to seasonal and inter-annual variability. First, those environmental conditions that drive such variations are discussed.

3.2.1 Environmental conditions and riverflow

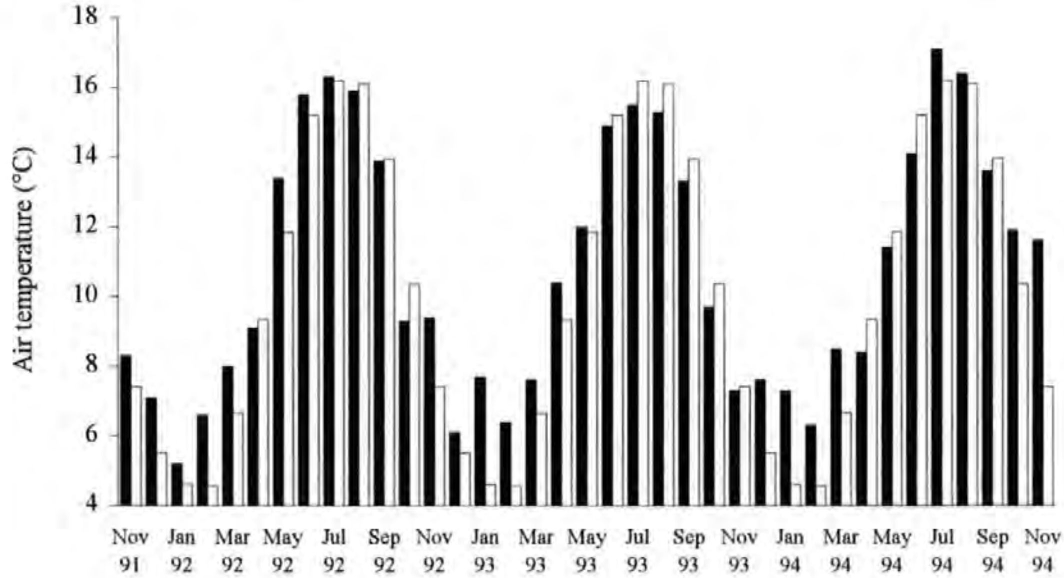
The meteorological conditions during the study were important in influencing temperature and salinity of the harbour waters. Inter-annual variations in direct insolation during the biologically active seasons contributed to observed variations in algal bloom intensity. Seasonal variations in rainfall influenced flow in the River Plym and hence the salinity of the estuarine water exchanging with the harbour water. The prevailing meteorological conditions for Plymouth during the study compared with 30-year climatic mean data for Mount Batten RAF Station (Meteorological Office, 1982) are shown in Fig. 3.8 and discussed in Section *3.2.2*. Flow data for the River Plym for the study period were unavailable because the then South West Water Authority closed its gauging station at Carnwood in 1978. However, the available mean monthly data from 1974 to 1978 were averaged to provide an idea of seasonal riverflow conditions and variability (Fig. 3.9). Mean winter riverflows were 4.4 ± 2.5 $m^3 s^{-1}$ during January and 5.4 ± 3.1 $m^3 s^{-1}$ for February, with a maximum of 8.8 $m^3 s^{-1}$ during February 1974. Such high variability may cause rapid changes in salinity in estuarine and harbour waters (as will be illustrated in Section *3.2.4*), with implications for short-term variations in water quality. Summer riverflows were ten times less than those in winter, with 0.4 ± 0.1 $m^3 s^{-1}$ in June and 0.5 ± 0.3 $m^3 s^{-1}$ in July. A minimum of 0.23 $m^3 s^{-1}$ for the drought of 1976 may have biased the mean values. However, a general low degree of flow variability in summer leads to stable salinity of estuarine and harbour waters, an important factor allowing inter-annual comparisons before and after impoundment to be drawn(Section *3.2.3*). March riverflows (2.9 ± 1.3 $m^3 s^{-1}$) were greater than September (1.6 ± 2.4 $m^3 s^{-1}$) despite lower mean rainfall in March, because of water table differences between winter and summer. However, greater variability in September is due to variability in rainfall (177 ± 77 mm (RSD = 44 %) in March; 189 ± 132 mm (RSD = 70 %) in September).

Murray (1985) found a strong positive correlation (r = 0.82) between mean monthly Plym riverflow and total monthly rainfall at Princetown Prison (obtained from the Meteorological Office, Bracknell). Linear regression of the data forced through the origin (Fig. 3.10) yielded the equation:

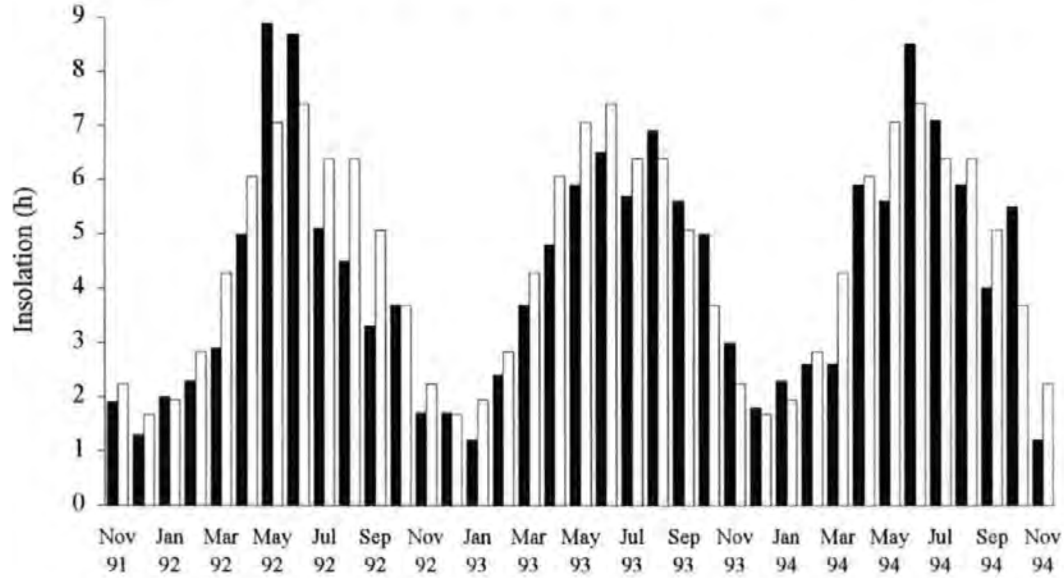
$$R = 0.0139 P \qquad \text{Eq. 3.4.}$$

where R = mean monthly riverflow (m^3s^{-1}) and P = monthly rainfall (mm). This algorithm was highly significant ($p < 0.1\%$) and was used as part of the validation of the ECoS model of the Plym Estuary and Sutton Harbour (Section *4.2.2*).

(a)



(b)



(c)

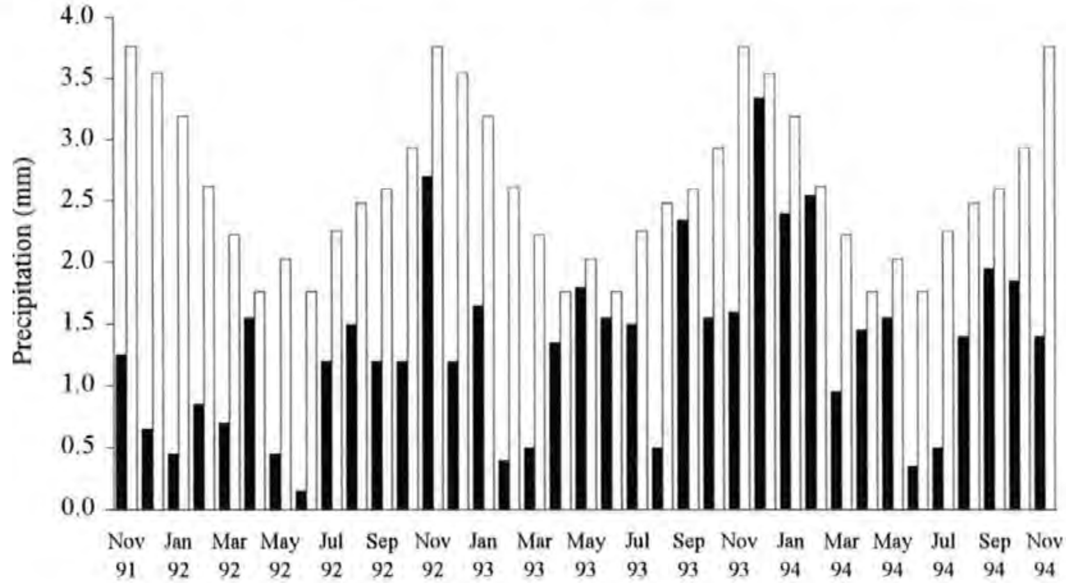


Fig. 3.8. Meteorological data for Plymouth from November 1991 to November 1994 comprising (a) mean monthly air temperatures (°C), (b) monthly mean insolation (h) and (c) monthly mean precipitation (mm). Filled bars represent monthly mean, open bars represent 30 year mean for Plymouth (Mount Batten).

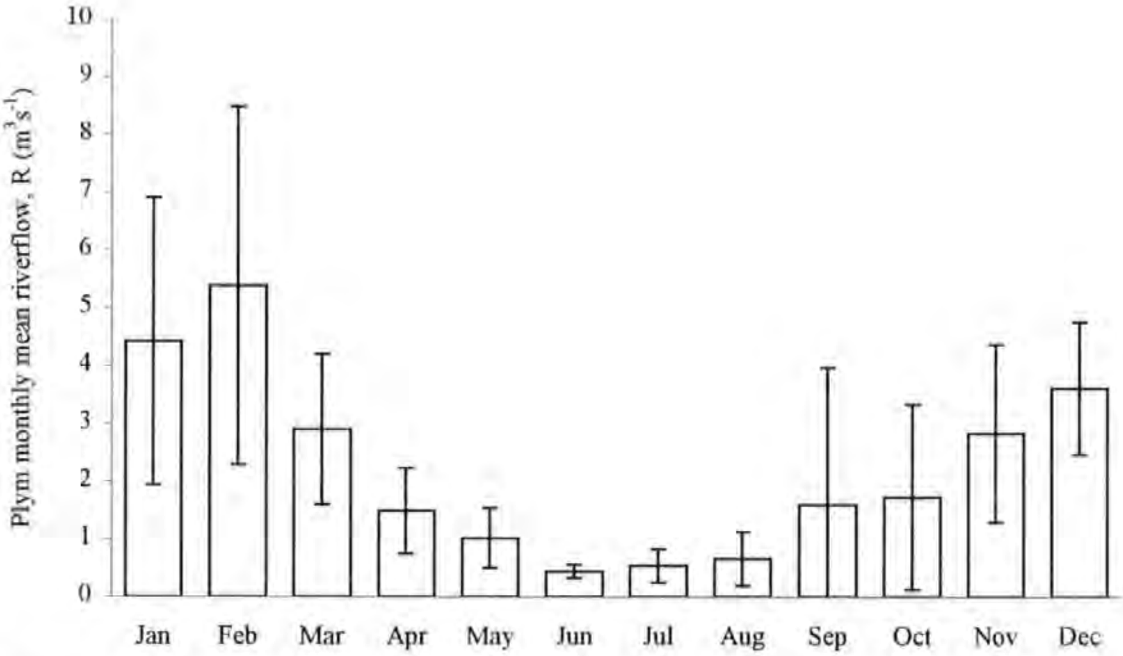


Fig. 3.9. Mean monthly Plym riverflow (m^3 s^{-1}), measured at Carnwood by South West Water Authority between 1974 and 1978. Error bars indicate one standard deviation, n = 5.

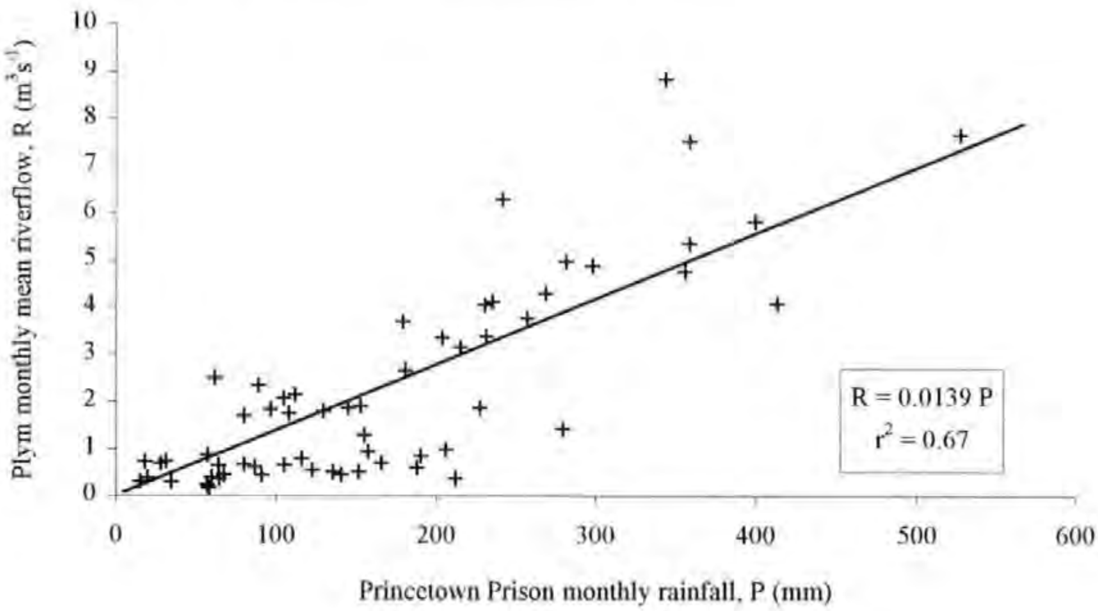


Fig. 3.10. Model used to predict mean monthly Plym riverflow, R, from Princetown Prison monthly rainfall, P (After Murray, 1985).

3.2.2 Comparisons of environmental conditions and inter-annual variability

The winter of 1991-1992 was considerably drier than the climatic mean, thereby limiting the river flow and maintaining higher surface salinities inside and outside the harbour than in subsequent winters. This dry period lasted until July 1992, punctuated only by near-average precipitation in April 1992. May 1992 and June 1992 received 25 % and 20 % higher insolation than average, respectively, with an associated increase in mean water temperatures inside and outside the harbour over those observed in subsequent summer periods, particularly at neaps. An increase in rainfall in July 1992 had little effect on surface or bottom salinities inside or outside the harbour, although localised runoff may have been significant. The conditions during the biologically active season from April 1992 to September 1992 were therefore generally favourable for primary production.

The period from November 1992 to January 1993 was considerably wetter than in the previous year, resulting in enhanced river flows that lowered surface and bottom salinities inside and outside the harbour, particularly in January. Conversely, February 1993 and March 1993 were drier than in the previous year, resulting in reduced river flows and an increase in salinity to maximum values throughout the system in March. However, the period from April 1993 to July 1993 received more precipitation and less insolation than in the previous year, resulting in lower mean water temperatures throughout the system. Despite slightly lower salinities, conditions were less favourable for primary production than in the previous biologically active season until July 1993. After this period, the opposite was true, with considerably higher insolation than in the previous year resulting in higher mean water temperatures for September 1993 than September 1992 both inside and outside the harbour. An increase in precipitation lowered salinities, promoting favourable conditions for primary productivity through September 1993. The biologically active season was effectively delayed in 1993 by up to six weeks over that observed in 1992 and differences in prevailing weather conditions were a causal factor.

The winter period from December 1993 to February 1994 received the highest rainfall observed throughout the study, almost reaching the climatic mean. Surface salinities were particularly low throughout the system at neaps in January 1994. The period March 1994 to May 1994, spanning the commissioning of the scheme, received rainfall comparable with previous years. Insolation in April 1994 and May 1994 was comparable with that in 1993, but mean water temperatures were higher inside the harbour and therefore more favourable for primary production than outside in May 1994. Insolation in June 1994 and July 1994 was greater than in previous years and further increased the difference in temperature between inside and outside. Despite similarly high salinities, conditions for primary production were again more favourable inside the harbour.

The inter-annual variability in water temperature and salinity as a result of these varying environmental conditions is shown in Fig. 3.11 and Fig. 3.12, respectively. Water temperatures in November were greater and more variable than in January both inside and outside the harbour at springs and neaps. The lowest and most variable salinities were observed between November and March, particularly in January at neaps inside the harbour (Fig. 3.12c). Water temperatures in March showed little inter-annual variation, but salinities were particularly variable at neaps outside the harbour. The greatest inter-annual variability in water temperatures was seen inside and outside the harbour at neaps

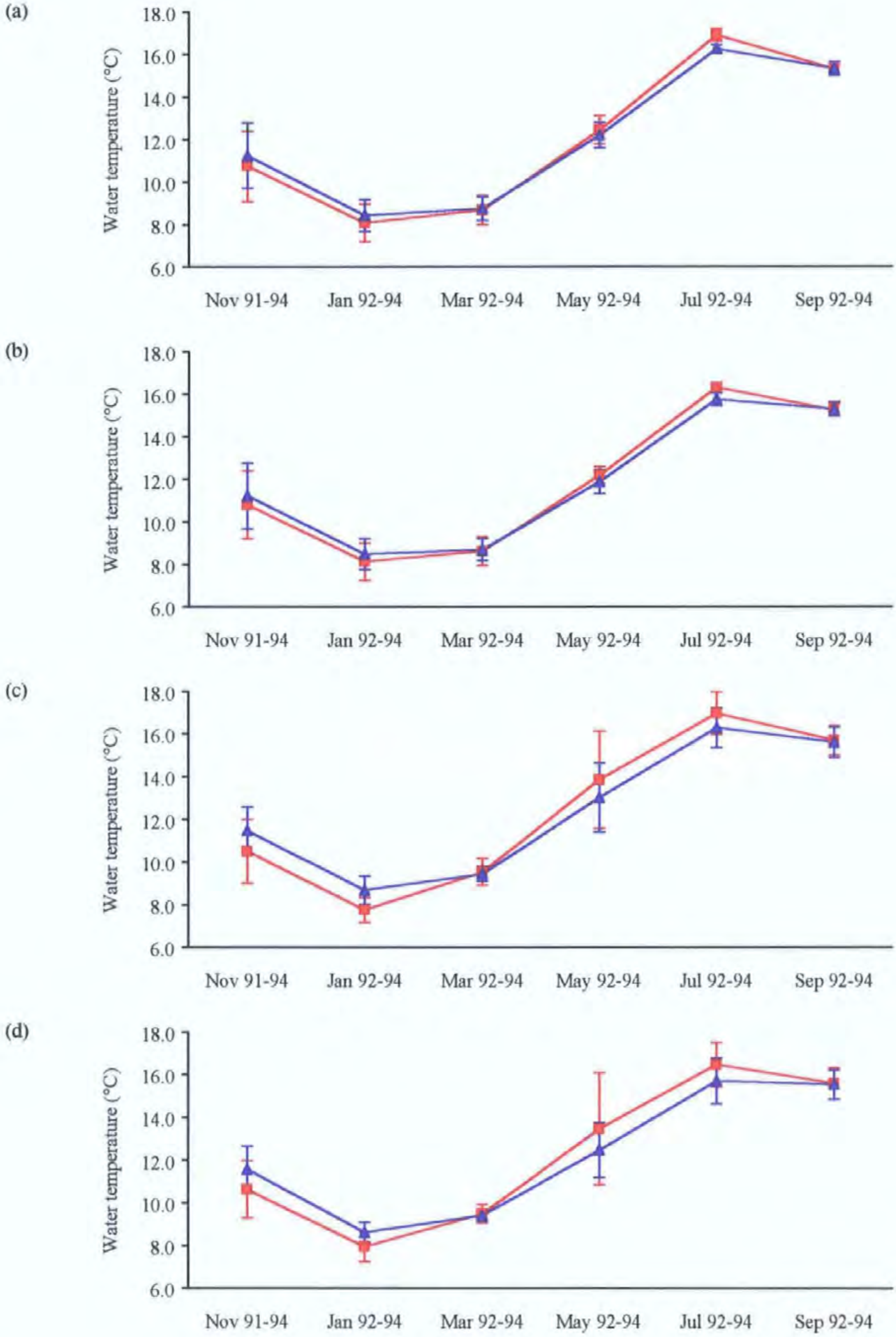


Fig. 3.11. Mean of tidally-averaged water temperatures (°C) for (a) inside harbour at springs, (b) outside harbour at springs, (c) inside harbour at neaps and (d) outside harbour at neaps. Red lines represent surface water, blue lines represent bottom water. Error bars indicate one standard deviation.

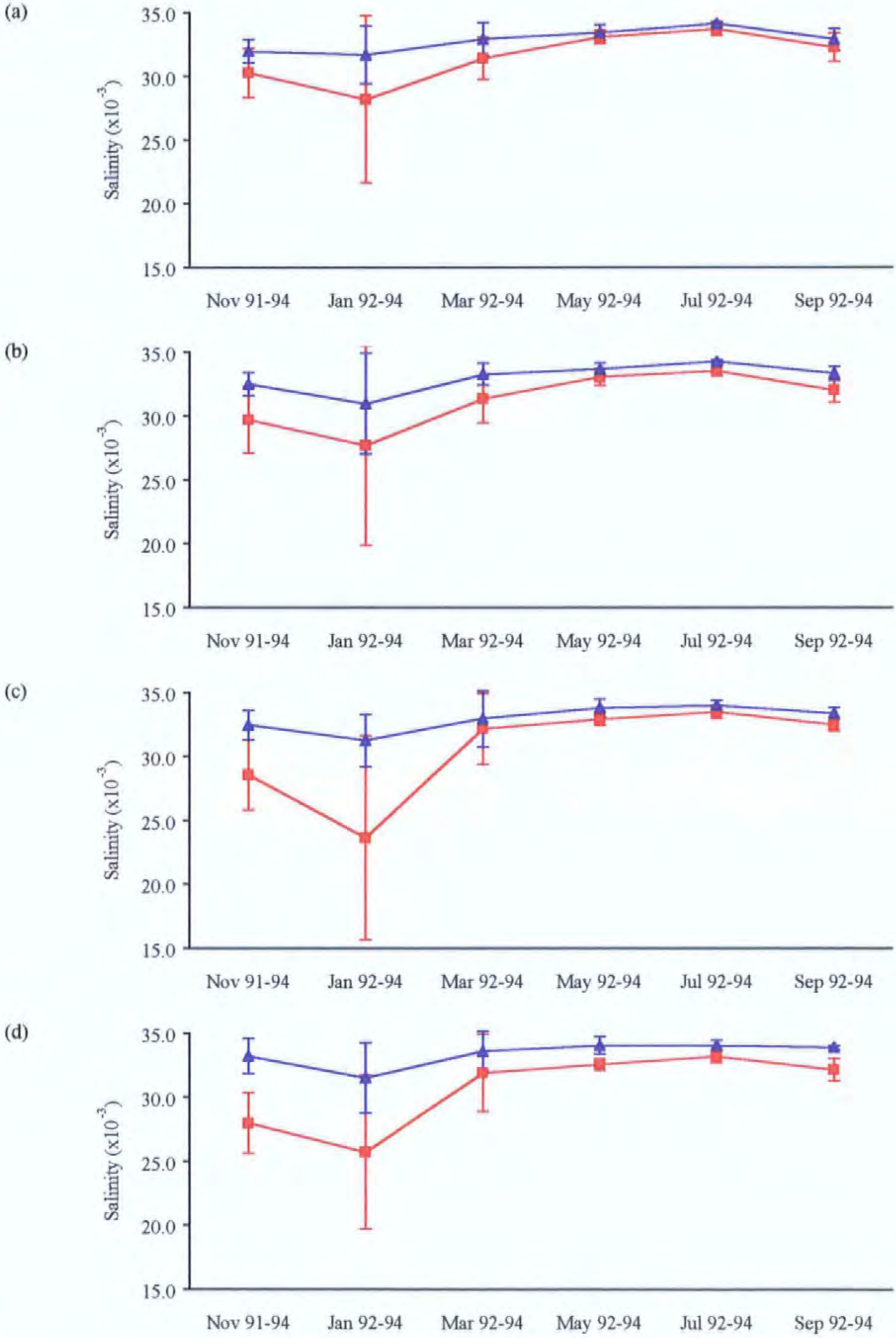


Fig. 3.12. Mean of tidally-averaged salinity (x10^{-3}) for (a) inside harbour at springs, (b) outside harbour at springs, (c) inside harbour at neaps and (d) outside harbour at neaps. Red lines represent surface water, blue lines represent bottom water. Error bars indicate one standard deviation.

in May (Figs. 3.11c and d) as a result of the high insolation in May 1992 compared with subsequent years. Similar circumstances were observed in July at neaps, as a result of higher insolation in July 1994 than in previous years. However, at springs in May and July, water temperatures exhibited inter-annual stability (Figs. 3.11a and b) matched by stable salinities at springs and neaps. September water temperatures were less variable than those of summer, although some departure at neaps reflects the high insolation of September 1993. Salinities in September were less variable than in March, although this conflicts with the trends in riverflow between 1974 and 1978 and highlights the requirement for contemporary riverflow data for studies of this nature.

3.2.3 Semi-diurnal and diurnal variability

During summer periods, strong insolation caused considerable warming of the shallow waters inside and outside the harbour. The possible effect of impoundment upon this warming was investigated by comparison of the results of two July surveys, one made before and one after impoundment. The low inter-annual variability in environmental conditions for July has been shown in Section *3.2.2,* thus facilitating this comparison. Water temperature profiles for representative sites on 07/07/93 and 11/07/94 are plotted against non-dimensional depth in Fig. 3.13. Differing water depths between stations and between tidal states were normalised to facilitate comparison using the following equation:

$$\text{non-dimensional depth, } \eta = \frac{z_t - z}{z_t} \qquad \text{Eq. 3.5.}$$

where z = probe depth (m) and z_t = total depth (m).

In July 1993, temperatures inside the harbour (Fig. 3.13a) increased steadily from HW to LW, particularly near-bottom, as the water became shallower. A similar result was observed outside the harbour (Fig. 3.13b), although the deeper water exhibited lower mean temperatures and stronger thermoclines. Between LW and HW-3h, the temperature inside the harbour decreased as cooler water flooded in and the depth increased. Outside the harbour, the observed decrease was greater as cooler, deeper water flooded in.

In July 1994, the temperature inside the harbour (Fig. 3.13c) again increased steadily from HW to LW, although this was more limited to the surface. A stronger thermocline developed in the deeper impounded water. Outside the harbour (Fig. 3.13d) temperatures increased to a higher mean than inside, owing to the shallow depth. The greatest difference was again observed between LW and HW-3h as cooler seawater entered the area outside the harbour. However, on this occasion the temperature inside the harbour steadily increased above that observed at LW until the start of free flow. The reduced influx of cooler water between HW-3h and HW would therefore also have contributed to higher mean temperatures inside than outside at the following HW.

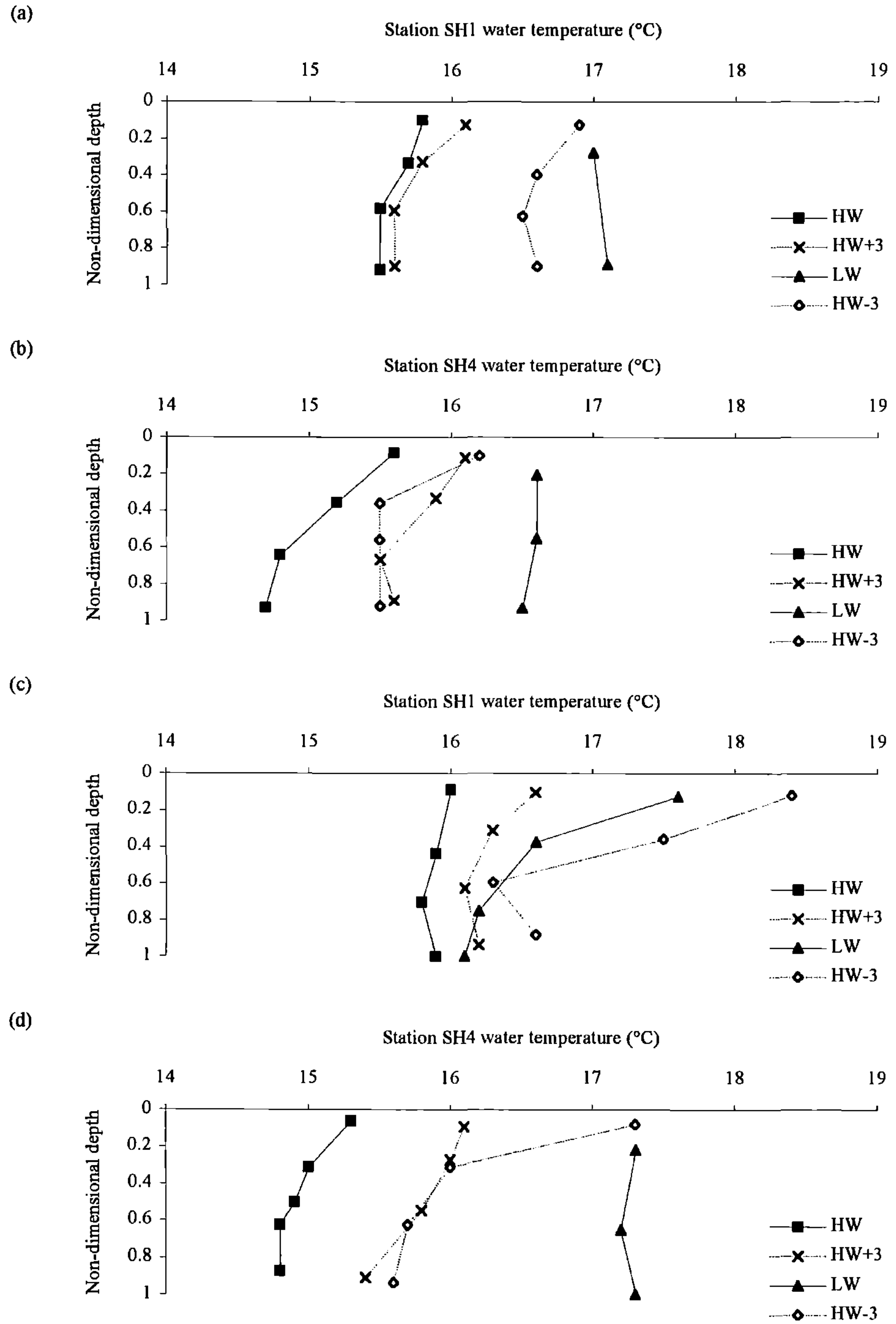


Fig. 3.13. Water temperature profiles for representative stations (a) inside harbour and (b) outside harbour at springs in July 1993 (before impoundment) and (c) inside harbour and (d) outside harbour at springs in July 1994 (after impoundment).

The salinity of the harbour water underwent semi-diurnal variations before impoundment, driven by tidal oscillations at the mouth of the Plym Estuary. In the estuary at HW, salinities were at a maximum as seawater flooded in. From HW to LW, salinity decreased to a minimum as brackish water flowed seaward from the estuary on the ebb tide. This cycle drove the salinity variations observed in the harbour, examples of which are shown in Fig. 3.14. The upward trend in salinity between 09/04/94 and 11/04/94 was attributed to recovery after spate conditions in which 67 mm of rain fell in 10 days. However, the semi-diurnal variations remained clearly visible. Salinity was at a maximum inside the harbour at each LW due to the relative retention of bottom, more saline water. During the first 1-2 h of the flood tide, the minimum salinity water in the mouth of the estuary flowed into the harbour, causing a rapid reduction in surface salinity. Later in the flood tide through HW-3h to HW the salinity of the estuarine water flooding into the harbour had increased by mixing with seawater. Then, from HW to LW, the salinity of the harbour surface water continued to increase steadily as it mixed with and was replaced by the more saline underlying water during the ebb tide. This evidence was proof of a hypothetical tidal mechanism (HR, 1984) that shunted contaminated estuarine water into the harbour during the flood tide and further showed that the process was limited to the early flood.

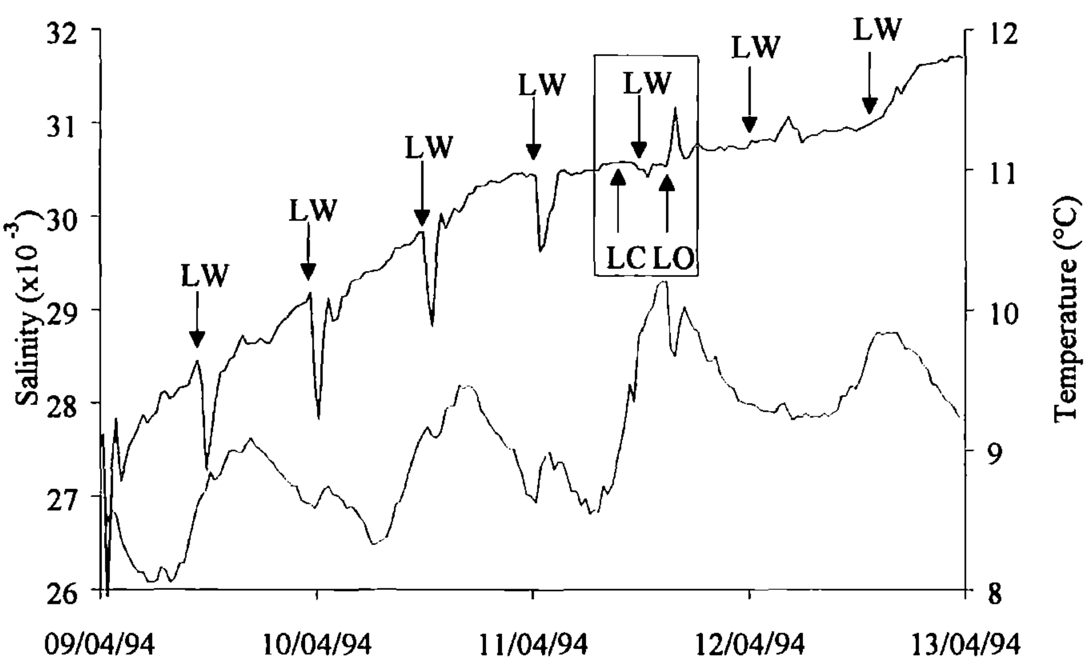


Fig. 3.14. Surface salinity and water temperature traces from the continuous monitoring instrument at site 6 (Fig. 2.6) during the commissioning period. Arrows indicate the precise times of LW, lock closure (LC) and lock opening (LO). Boxed region represents inaugural impoundment. Blue line represents salinity, red line represents water temperature.

The effect of impoundment upon semi-diurnal salinity variations is shown in Fig. 3.14, the time scale of which spans the commissioning date of the scheme. Immediately after impoundment, the familiar semi-diurnal cycle was disrupted. The low salinity water of the early flood was prevented from entering the harbour. At the start of free flow (HW-3h), a spike of higher salinity water entered the harbour. It would therefore appear that the lock gates provide an effective barrier against the influx of pollutant laden estuarine water for the first 2-3 h of the flood tide, an unanticipated benefit of impoundment. A slight perturbation to the diurnal variation of water temperature after impoundment is also evident in Fig. 3.14, where temperature apparently rises more rapidly than on the previous two days. However, temperature is predominantly diurnal in variation, not tidal, and is driven by environmental conditions. The mean air temperature on 11/04/94 was 9.4 °C, 3 °C warmer than the previous two days, although there were only 7 h of direct insolation compared with 12 h on 10/04/94.

The effect of impoundment upon short-term variations in salinity was further investigated by comparison of the results of surveys before and after impoundment (Fig. 3.15). The surveys were conducted at spring tides on 11/01/93 and 22/11/94. The rainfall during the preceding 4 day period was 30 mm and 14 mm, respectively. Inside the harbour in January 1993 (Fig. 3.15a), the water column remained stratified ($\Delta S = 8 \times 10^{-3}$ to 12×10^{-3}) throughout the tidal cycle. Surface salinity increased steadily from HW to LW, decreasing again at HW-3h as brackish estuarine water flooded in. Bottom salinity was almost unchanged between HW and LW, but decreased sharply during the flood to HW-3h. These variations were consistent with the shunting mechanism, driven by changes outside the harbour (Fig. 3.15b) where the stratified water column ($\Delta S = 7 \times 10^{-3}$ to 11×10^{-3}) varied from maximum salinity at HW to minimum at LW at both surface and bottom. The salinity of the harbour water therefore varied typically whilst in free connection with the estuarine water and the harbour received a significant proportion of low salinity water during each tidal cycle. This is in contrast with observations in November 1994, when the harbour water column (Fig. 3.15c) was stratified ($\Delta S = 4 \times 10^{-3}$ to 5×10^{-3}) but salinity did not change significantly throughout the tidal cycle. The water outside the harbour was more stratified ($\Delta S = 5 \times 10^{-3}$ to 8×10^{-3}) and exhibited typical decreases in salinity between HW and LW. Short-term variations in salinity were therefore curtailed, with significantly less low salinity water entering the harbour during each tidal cycle. The shunting mechanism is shown to be consistent before impoundment in Section *3.2.5*.

3.2.4 Spring-neap tide variability

The mean spring and neap tidal ranges for Sutton Harbour (Plymouth Devonport) are 4.7 m and 2.2 m, respectively (Table 3.1), but can be as extreme as 5.6 m (29/03/94) and 1.9 m (21/03/94). These differing tidal ranges lead to variations in mixing and advection, in that the more energetic spring tides cause greater mixing of the water column and bring a greater volume of water across the interface between harbour and estuary. Therefore, the spring-neap tidal cycle, with a period of *circa* 14.5 days, causes sympathetic variations in water quality.

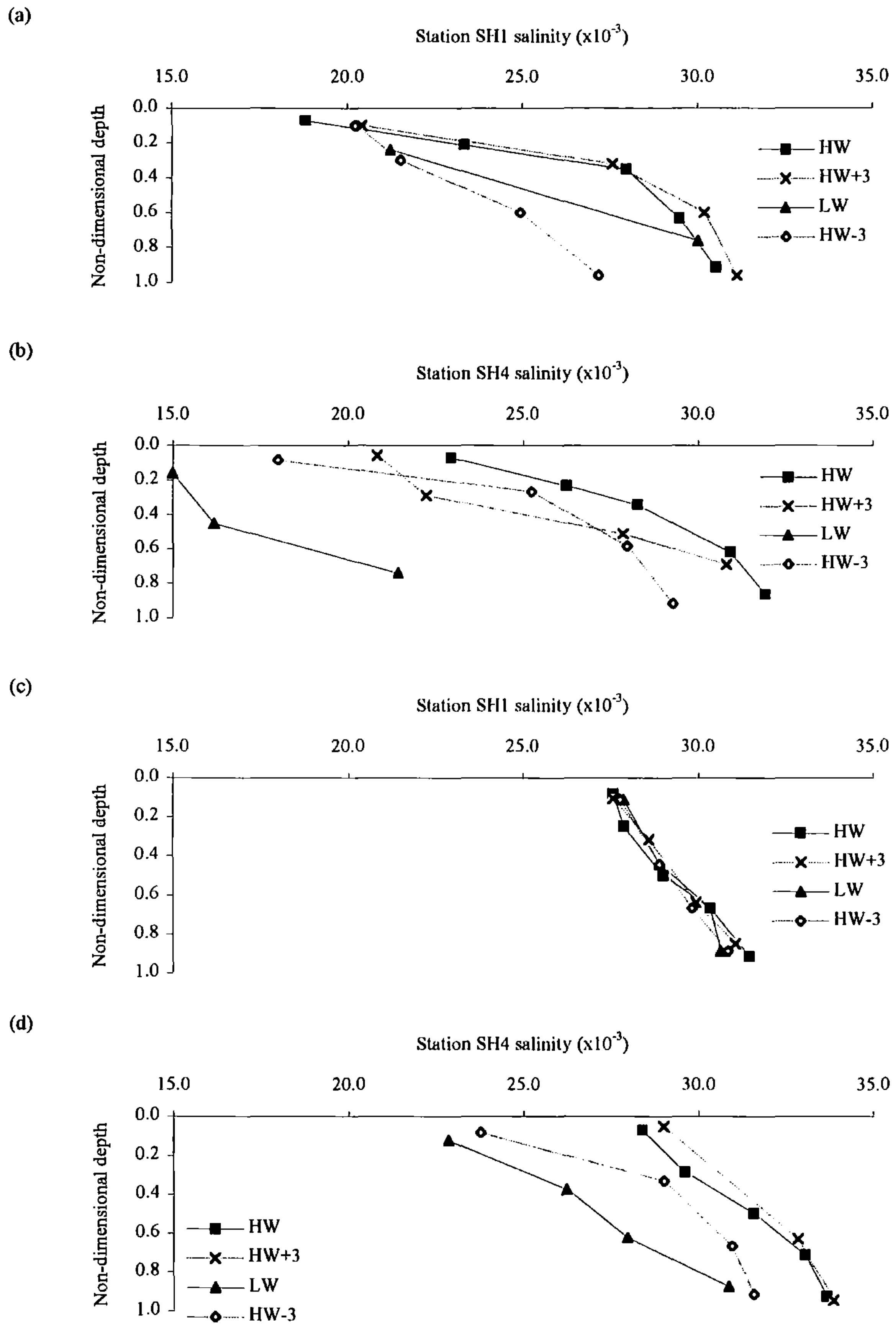


Fig. 3.15. Salinity profiles for representative stations (a) inside harbour and (b) outside harbour at springs in January 1993 (before impoundment) and (c) inside harbour and (d) outside harbour at springs in November 1994 (after impoundment).

Water temperatures inside the harbour were examined on consecutive spring and neap tides before and after impoundment (Fig. 3.16), during the summer period when the greatest spring-neap variability was anticipated. At neaps in July 1993, (Fig. 3.16a), the surface water was heated by 10 h of direct insolation (from LW to HW+3h), following 20 h of direct insolation in the previous three days. However, the less energetic neap tide did not efficiently mix this warm water downwards and bottom temperatures remained at 15 °C, 2 °C cooler than at the surface by HW+3h. One week later during greater mixing at springs (Fig. 3.16b), the water column was heated throughout its depth (from HW to HW-3h) by 13 h of direct insolation, after 36 h of direct insolation in the previous three days. During one week, mean surface and bottom temperatures had risen by 0.5 °C and 1 °C, respectively. After impoundment, at neaps in July 1994 (Fig. 3.16c), the water column received only 4 h of direct insolation between LW and HW-3h following 17 h of direct insolation in the previous three days, but a small increase in surface temperature from 17 °C to 17.5 °C was still detected during the day. One week later at springs (Fig. 3.16d), the water column received 12 h of direct insolation, following 16 h of direct insolation in the previous three days. The water column exhibited more stratification than at neaps due to this increased insolation and more than at springs before impoundment under similar insolation, due to the decreased tidal range after impoundment. Mean surface and bottom temperatures had fallen by 0.5 °C during the week. Spring-neap variations in temperature were therefore slightly less marked after impoundment, particularly in the bottom waters, because the increased mean harbour volume and the decreased difference in volume exchange between springs and neaps acted to buffer any changes.

Salinity inside the harbour was also examined on consecutive spring and neap tides before and after impoundment (Fig. 3.17) during the winter period when the greatest spring-neap variability in salinity was anticipated. In the absence of January 1995 data, November 1994 data were taken as representative of winter after impoundment. In January 1994 (Fig. 3.17a), the water column was highly stratified throughout the neap tidal cycle due to the heavy winter precipitation discussed in Section *3.2.2*, with 30 mm of rainfall in the previous four days. Surface salinities were in the range 17×10^{-3} to 20×10^{-3}, whilst bottom salinities were in the range 29×10^{-3} to 31×10^{-3}. The potential for pollutant input from riverine and sewage sources was considerable under these conditions. One week later at springs, more efficient tidal mixing of the water column was evident (Fig. 3.17b), with little stratification despite 12 mm of rainfall in the previous four days. Surface and bottom salinities varied little throughout the tidal cycle, at *circa* 30×10^{-3} and 31×10^{-3}, respectively. The freshwater fraction of the entire water column decreased considerably in one week, highlighting the ephemeral nature of salinity in the harbour system due to the short flushing time of the Plym Estuary. Following impoundment, at springs in November 1994 (Fig. 3.17c), the water column was partially stratified after 14 mm of rainfall in the previous four days, with surface salinities in the range 26×10^{-3} to 28×10^{-3} and salinities at

(a)

(b)

(c)

(d)

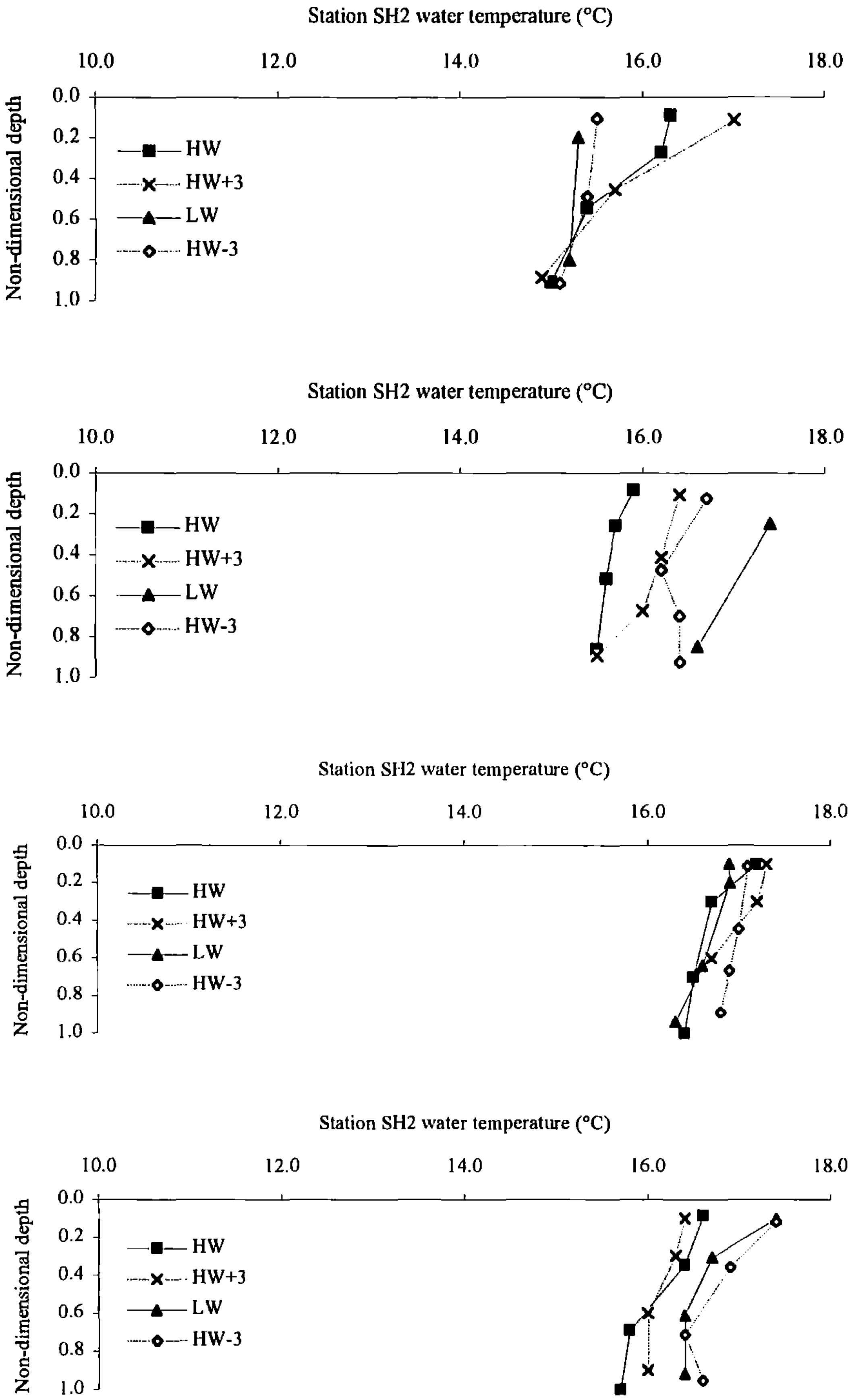


Fig. 3.16. Water temperature profiles for representative station SH2 before impoundment in July 1993 during (a) neap tide (LW 0801, HW 1422) and (b) spring tide (HW 0844, LW 1503) and in July 1994 after impoundment during (c) neap tide (LW 0849, HW 1518) and (d) spring tide (HW 0821, LW 1424).

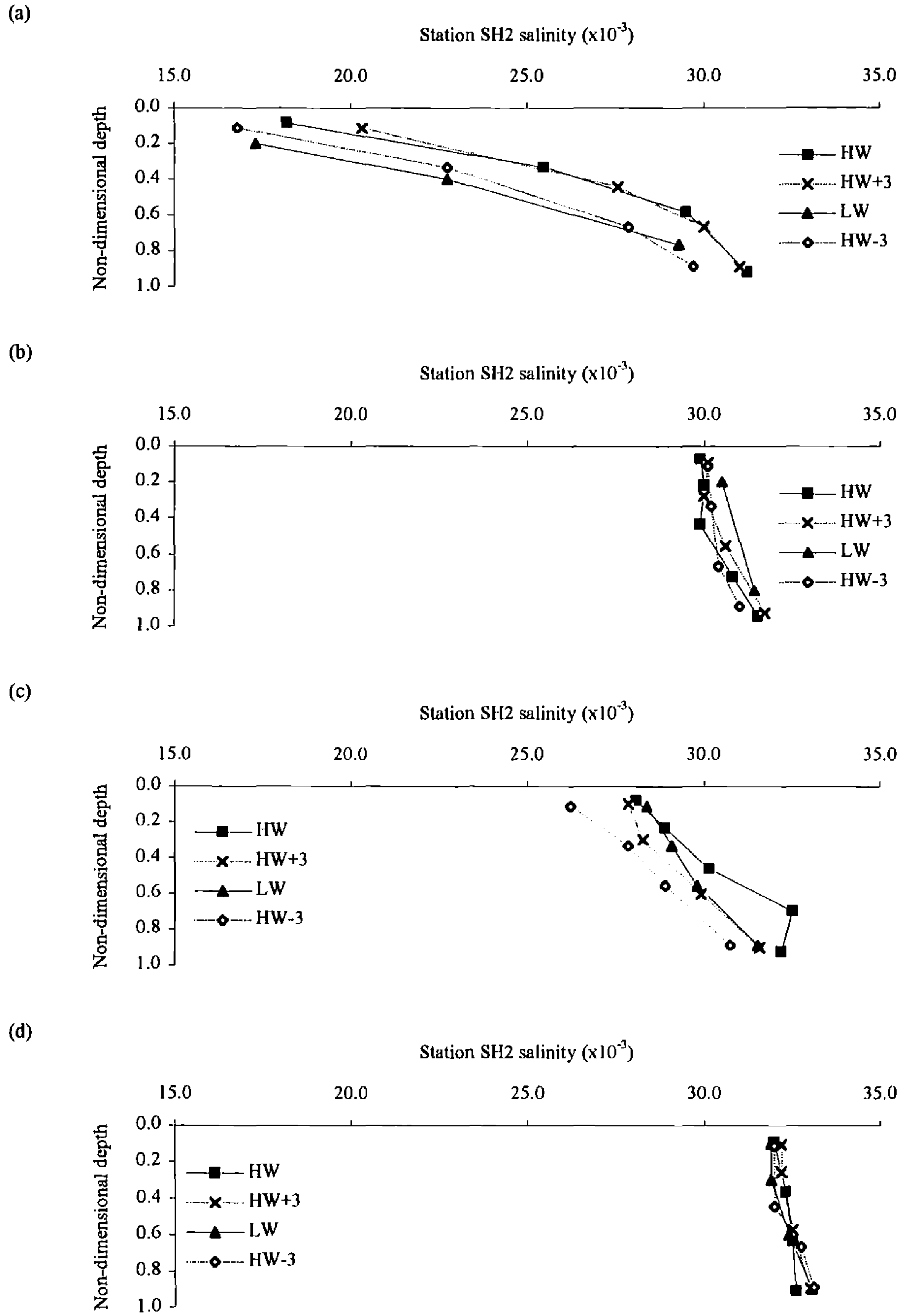


Fig. 3.17. Salinity profiles for representative station SH2 before impoundment in January 1994 during (a) neap tide (LW 0923, HW 1528) and (b) spring tide (HW 0749, LW 1414) and after impoundment in November 1994 during (c) spring tide (HW 0740, LW 1349) and (d) neap tide (LW 0801, HW 1400). November 1994 taken as representative of winter 1994-1995 in absence of January 1995 data.

the bottom values in the range 31 $\times 10^{-3}$ to 32 $\times 10^{-3}$, as a result of less effective tidal mixing. However, as shown in Figs. 3.15c and d, the reduced exchange of volume led to a lower potential for pollutant influx than before impoundment. One week later at neaps (Fig. 3.17d), the water column was almost fully mixed despite the occurrence of 11 mm of rainfall during the previous four days, with surface and bottom salinities of *circa* 32 $\times 10^{-3}$ and 33 $\times 10^{-3}$ observed throughout the tidal cycle. The freshwater fraction of the entire water column had again decreased considerably in one week. However, in this case, the potential for external pollutant inputs was lower at springs than at neaps, owing to the reduction in volume exchange after impoundment. These observations suggest that salinity of the harbour water continued to vary with the spring-neap tidal cycle after impoundment, but that the magnitude of variations was reduced, as with temperature, by a decrease in volume exchange differences.

3.2.5 Seasonal variability

For the purposes of describing seasonal variability in Sutton Harbour, the seasons have been operationally defined as winter (January), spring (March and May), summer (July) and autumn (September and November). It is acknowledged that there may be some overlap between these seasons, particularly with May and November which tend towards summer and winter, respectively.

Seasonal variations in temperature were approximately sinusoidal (Fig. 3.18), varying in response to the cycle of solar input. The amplitude of these variations was greater in the surface than in the bottom water. In winter, surface temperatures were up to 2 °C lower than bottom temperatures inside and outside the harbour owing to a combination of cooler, fresher water from the catchment, and convective loss to the atmosphere. Surface temperatures were generally at a minimum of 8 °C in January. In summer, they exceeded bottom temperatures by up to 2 °C inside and 3 °C outside the harbour and were generally at a maximum of 17 °C in July. Spring and autumn saw the transition between these opposites, when surface and bottom temperatures were generally the same, at a mean of 9 °C in spring and 15 °C and autumn. Temperature variations were greatest during neap tides under reduced vertical mixing (Figs. 3.18c and 3.18d).

The time series shown in Fig. 3.19 are composed of differences in temperature between inside and outside the harbour (such that a positive value represents a greater temperature inside). They are useful in determining the effect of impoundment. In winter, there was little difference at surface or bottom between inside and outside the harbour, with a maximum of 0.6 °C greater outside at neaps. Impoundment reduced these differences to zero in November 1994. Spring and autumn saw similarly small differences between inside and outside the harbour, with a maximum of 0.2 °C greater inside at springs and

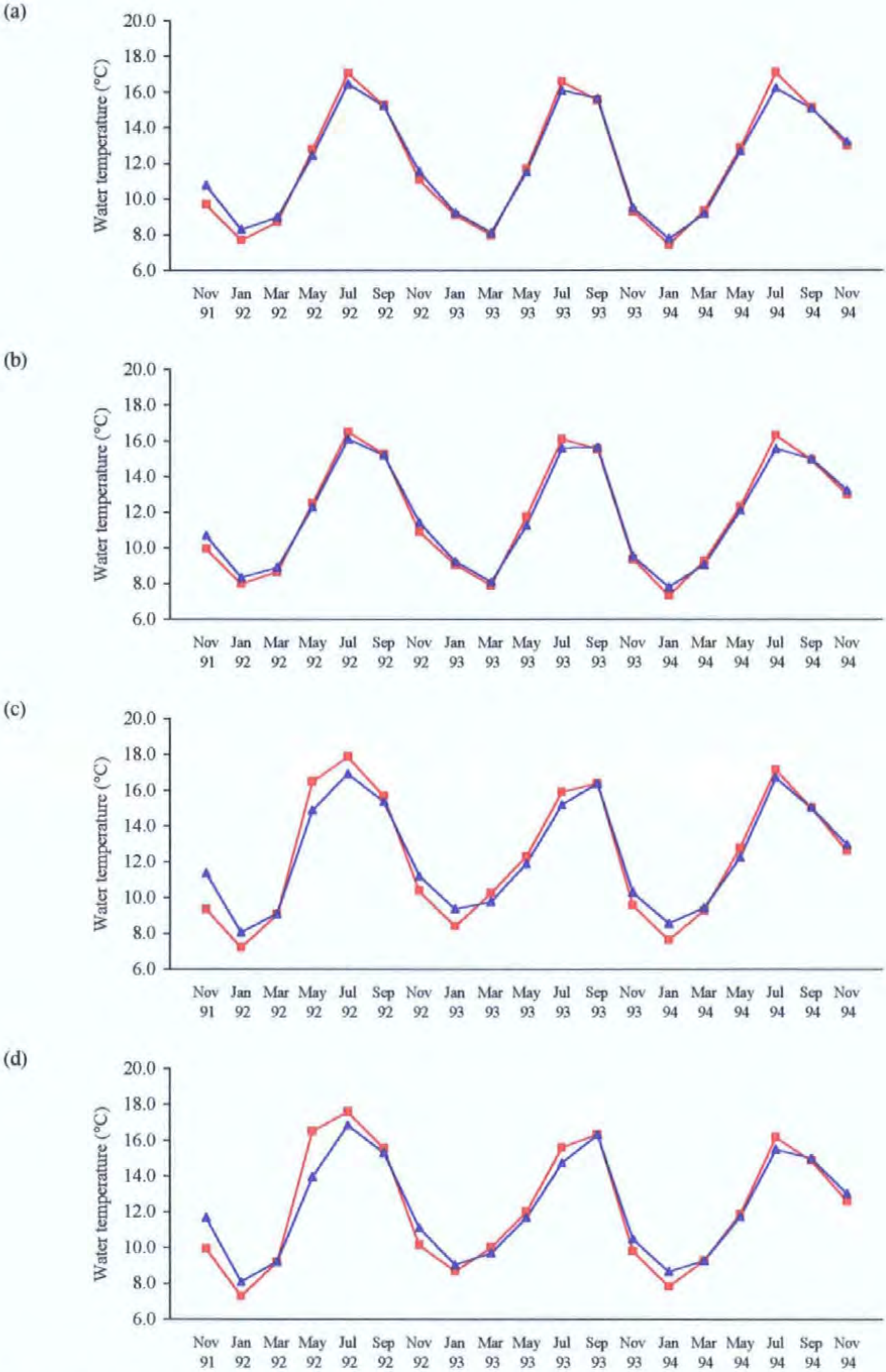


Fig. 3.18. Time series of tidally-averaged water temperature (°C) (a) inside harbour at springs, (b) outside harbour at springs, (c) inside harbour at neaps and (d) outside harbour at neaps. Red lines represent surface, blue lines represent bottom. Mean values of SH1 to SH3 (inside) and SH4 and SH5 (outside).

(a)

(b)

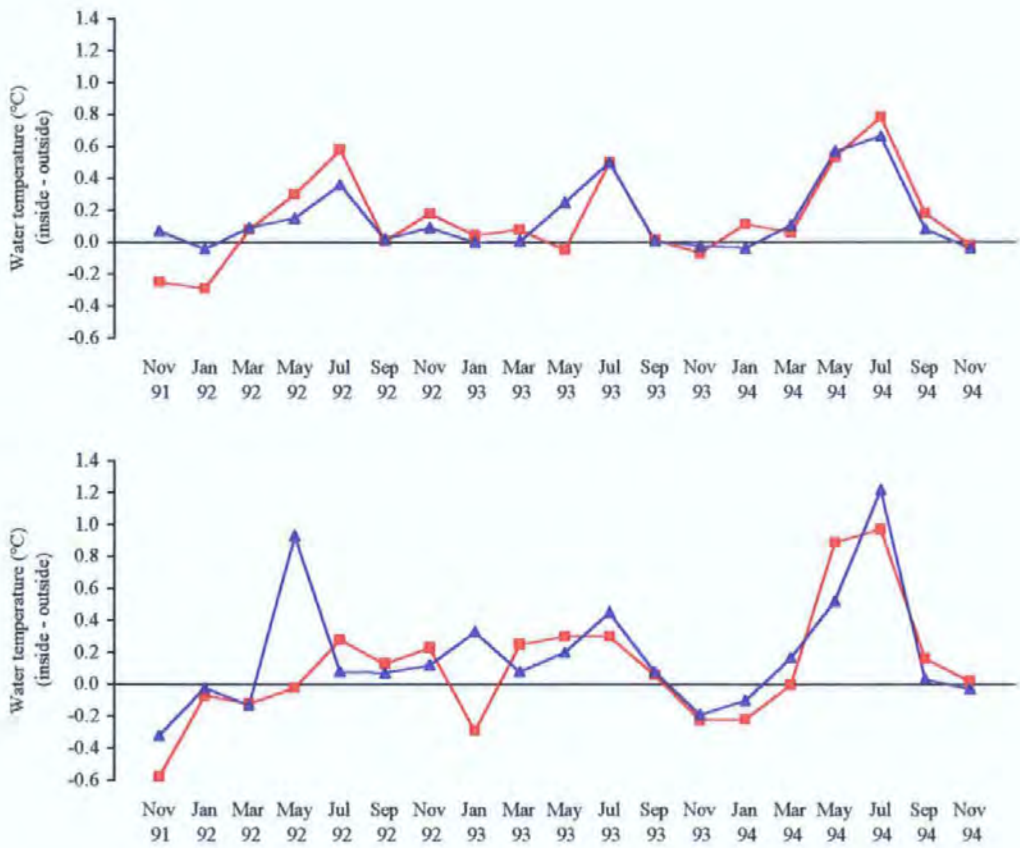


Fig. 3.19. Time series of differences in water temperature (°C) between inside and outside Sutton Harbour during (a) spring tides and (b) neap tides. Red lines represent surface, blue lines represent bottom.

neaps after impoundment in September 1994. The greatest differences were observed in summer, especially at neaps, when surface and bottom temperatures were consistently greater inside the harbour, except in May 1992 (Fig. 3.19b) and May 1993 (Fig. 3.19a), when strong insolation produced high diurnal surface temperatures throughout the system. Following impoundment, the differences were enhanced, with surface temperatures 1 °C greater and bottom temperatures 1.3 °C greater inside than outside at neaps in July 1994. It would therefore appear that impoundment did cause the anticipated enhancement of temperatures inside the harbour. It should also be noted that the summer of 1994 was not particularly warm and the differences may be further increased under prolonged periods of intense insolation. However, those differences observed during the present study are unlikely to have had a significant effect upon water quality.

The seasonal variations in salinity were dominated by pulses of brackish water during the winter periods when the River Plym was in spate (Fig 3.20), particularly during January surveys. At neaps, stratification of the water column was greater inside the harbour ($\Delta S = 13 \times10^{-3}$ in January 1994) than outside ($\Delta S = 11 \times10^{-3}$ in January 1994) before impoundment, because the tidally-averaged surface salinities inside the harbour were 3×10^{-3} to 4×10^{-3} lower than those outside (Fig. 3.21b). At springs, stratification was also greater inside ($\Delta S = 9 \times10^{-3}$) than outside the harbour ($\Delta S = 8 \times10^{-3}$) in January 1993

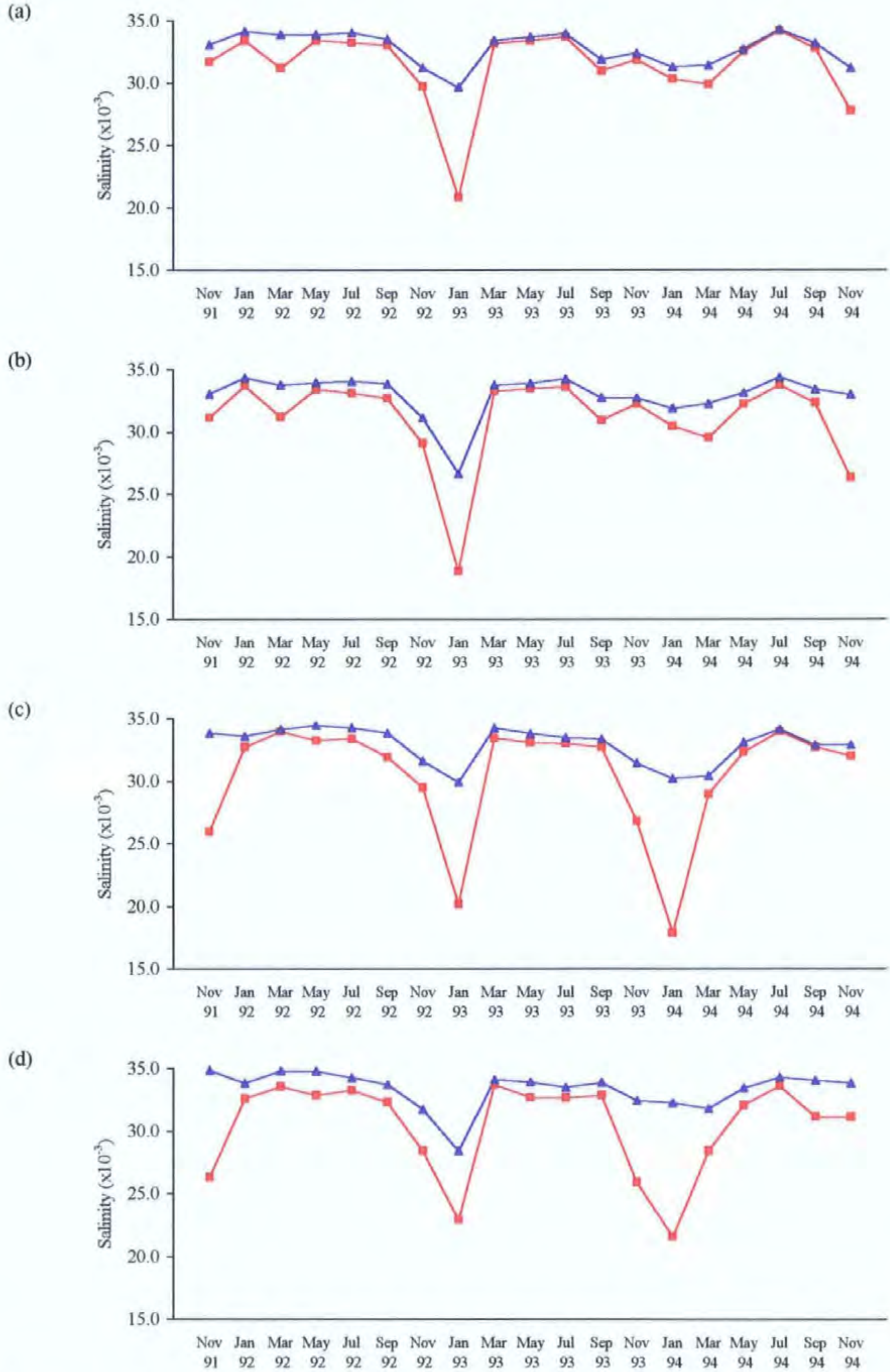
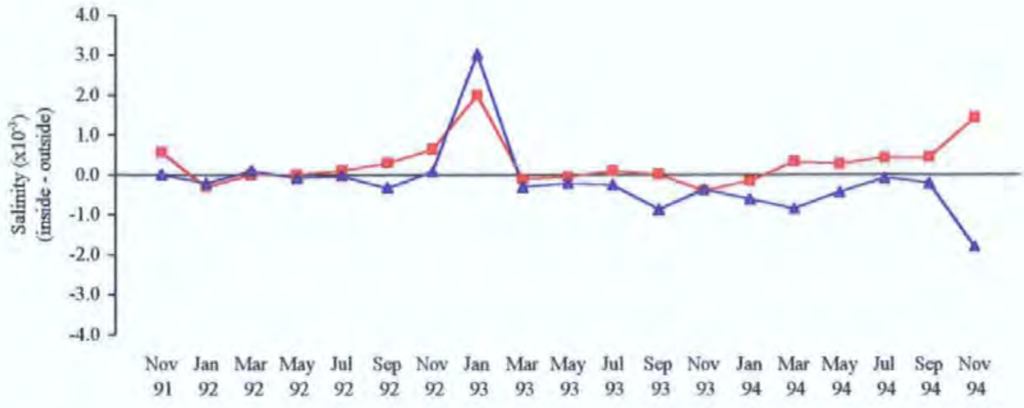


Fig. 3.20. Time series of tidally-averaged salinity ($x10^{-3}$) (a) inside harbour at springs, (b) outside harbour at springs, (c) inside harbour at neaps and (d) outside harbour at neaps. Red lines represent surface, blue lines represent bottom. Mean values of SH1 to SH3 (inside) and SH4 and SH5 (outside).

(a)



(b)

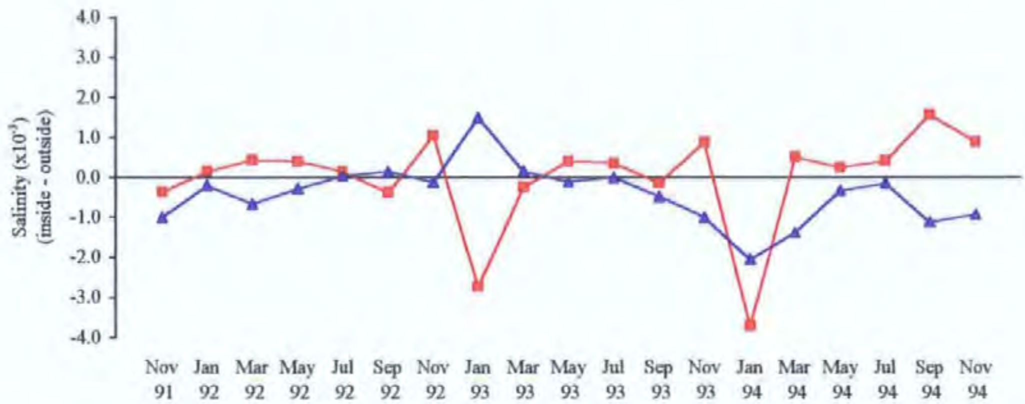
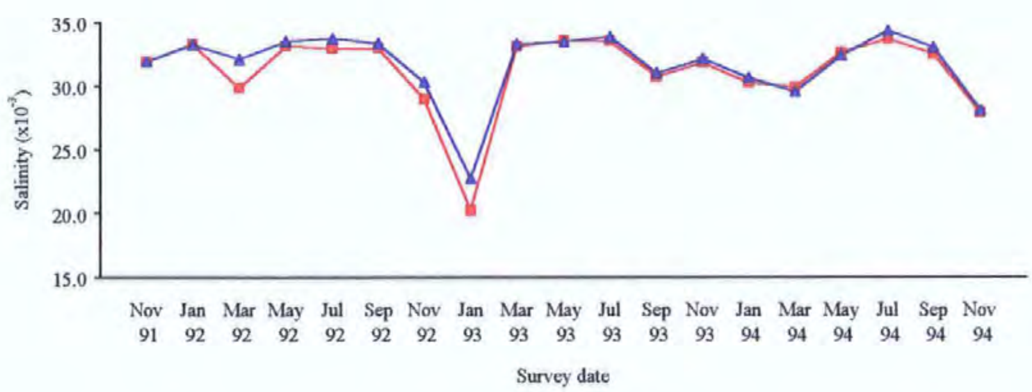


Fig. 3.21. Time series of differences in salinity (x10^{-3}) between inside and outside Sutton Harbour during (a) spring tides and (b) neap tides. Red lines represent surface, blue lines represent bottom.

before impoundment, despite the lowest surface salinity being observed outside the harbour (S = 18 x10^{-3}), because the bottom salinity was 3 x10^{-3} greater inside the harbour (Fig. 3.21a). Following impoundment, the limited winter data (November 1994) suggest that the water column was more stratified outside the harbour (ΔS = 7 x10^{-3} at springs, ΔS = 3 x10^{-3} at neaps) than inside the harbour (ΔS = 3 x10^{-3} at springs, ΔS = 1 x10^{-3} at neaps) as a result of reduced influx of lower salinity water to the harbour (Fig. 3.20) and possibly due to increased tidal mixing in the constricted harbour entrance. Salinity variations were less marked during the summer periods of low river flow. The water column inside the harbour was generally well-mixed, particularly in July (ΔS <1 x10^{-3}), with surface salinities reaching a seasonal maximum of 34 x10^{-3} (Fig. 3.20). Outside the harbour, the water column was more stratified (ΔS <2 x10^{-3} at neaps) with surface salinities ranging from 32 x10^{-3} to 34 x10^{-3}. Surface salinities were higher inside the harbour and bottom salinities were higher outside the harbour in summer (Fig. 3.21), but the differences were slight. After impoundment, these differences were not significantly greater, but remained constant. The potential for pollutant influx to the harbour was clearly lower during the summer months. Salinity variations in the spring and autumn again marked the transition between the opposing conditions of winter and summer. Inside the harbour, the water column was partially stratified (ΔS <3 x10^{-3} in March) with surface salinities to a minimum of 28 x10^{-3} (Fig. 3.20), although these conditions were variable depending on the meteorological conditions. Outside the harbour, the water column was again more stratified ($\Delta S < 4 \times 10^{-3}$) as a result of higher bottom salinities, but surface salinities reached the same minimum as inside the harbour ($S = 28 \times 10^{-3}$).

The time series of salinity at HW and LW in Fig. 3.22 further illustrate the consistent nature of the shunting mechanism discussed in Section *3.2.3* prior to impoundment, particularly under higher river flow conditions such as in March 1992, November 1992 and January 1993 where surface salinities were lower at HW than LW inside the harbour (Fig. 3.22a) and the opposite was true outside the harbour (Fig. 3.22b). The time series also show the difference after impoundment during high river flow conditions in November 1994.

(a)



(b)

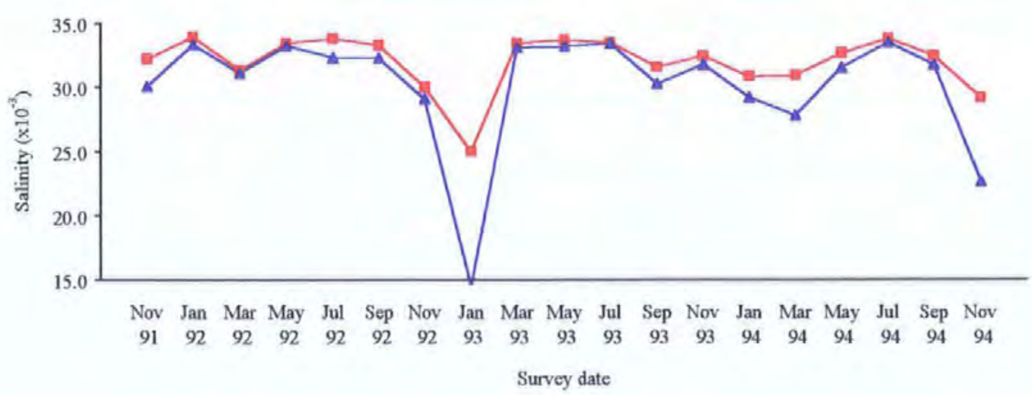


Fig 3.22. Time series of salinity ($\times 10^{-3}$) at the surface during spring tides (a) inside Sutton harbour (mean of SH1 to SH3) and (b) outside Sutton harbour (mean of SH4 and SH5). Red lines represent HW conditions, blue lines represent LW conditions.

3.3 Summary

The flushing of the harbour was less efficient after impoundment, as revealed by the increase in tracer-determined 95 % water renewal times from 45 h in 1990 to 72 h in 1994. However, this observed increase was considerably less than the theoretical five-fold increase predicted from volume exchange-based renewal times, as a result of considerable non-mixing between the dosed surface waters and the non-dosed bottom waters. It was found that vertical mixing in the harbour had effectively decreased by a factor of five and the surface waters were still effectively renewed over several tidal cycles, whilst the bottom waters were retained for almost a week. Although the flux of pollutants to the bottom waters may have decreased after impoundment, a slow accumulation of contaminated water could necessitate one monthly flushing on a spring tide, within the operational constraints of Sutton Harbour Company. The data suggest that the need to enhance water renewal in the harbour arms artificially by the installation of gravity recirculation pipes is unnecessary under present operational conditions.

Temperature and salinity, the hydrographic markers used in the study, were found to vary significantly on a number of time scales, depending on driving variables. Semi-diurnal variations were driven by the tidal cycle, diurnal and weekly variations by short-term variations in environmental conditions and the spring-neap tidal cycle. Seasonal and inter-annual variations are related to changes in environmental conditions. High variability in environmental conditions made it extremely difficult to predict water quality in the short- or long-term. In winter, riverflow varied significantly over a period of days, causing large variations in salinity and hence in pollutant influx. During summer, water temperatures varied significantly on a similar time scale owing to changes in insolation, with implications for short-term variation in primary production. Analyses of semi-diurnal and diurnal variations demonstrated an increase in mean temperatures inside the harbour during the summer after impoundment, confirmed the influx of low salinity water into the harbour during the early flood tide before impoundment, and illustrated that operation of the lock effectively curtailed this. Analyses of spring-neap variations showed that after impoundment the increased mean volume and decreased tidal range in the harbour buffered changes in water temperature and salinity on this time scale, thus reducing the potential for weekly variations in water quality. Analyses of seasonal variations demonstrates that regular cycles of temperature and salinity variation continued after impoundment, with some perturbations inside the harbour. Mean summer temperatures increased by *circa* 1 °C over those outside, a perturbation that may be enhanced during a warmer summer than 1994. The limited winter data for the period after impoundment suggest a slight increase in surface, and a slight decrease in bottom salinity. This effect may be linked to decreased flushing which causes a greater accumulation of brackish water in the bottom waters, which affects deposition of nutrients in winter. It may also be linked to the reduction of the shunting mechanism which limits the influx of low salinity estuarine water to the harbour surface waters. The combination of higher summer temperatures and increased winter nutrient deposition may promote eutrophicated conditions in the harbour, if these perturbations are enhanced in future.

Chapter Four

Water Column Ecology and Contaminants

Chapter 4 - Water Column Ecology and Contaminants

The water quality parameters examined in this chapter varied over similar timescales to temperature and salinity, whether through causal relationships to these hydrodynamic markers, or through response to the same forcing mechanisms as discussed in the previous chapter. For this reason and to aid comparison of all variables, the following sections have been arranged into semi-diurnal and diurnal, spring-neap and seasonal variabilities.

4.1 Dissolved Nutrients

Dissolved nutrient fluxes into Sutton Harbour appear to be important in determining the intensity and duration of algal bloom events in the spring and summer and macrophyte coverage of the harbour walls and pontoons. There are four main sources of nutrients to the harbour system associated with (i) brackish water from the Plym Estuary, (ii) untreated sewage outfalls along the Plymouth waterfront and diffuse inputs from leisure craft, (iii) anoxic sediment porewaters (Jacobson *et al.*, 1987; Seiki *et al.*, 1989) and (iv) mineralisation of organic matter in the water column. If the main source of nutrients is the catchment area of the Plym, then plots of dissolved nutrient concentration *versus* salinity should be linear in the absence of alternative sources and sinks (Knox *et al.*, 1981). These nutrient/salinity relationships are shown in Figs. 4.1a, b and c. It is evident from Fig. 4.1b that TON concentrations showed a strong inverse correlation with salinity in the surface and bottom waters ($r = -0.73$ and $r = -0.62$ respectively). This conservative behaviour was consistent with the observations of Morris *et al.* (1981) and resulted from low particle reactivity. The main source of TON to the lower estuary was therefore from terrestrial sources associated with runoff. However, both ammonium (Fig. 4.1a) and orthophosphate (Fig. 4.1c) showed only slight inverse correlations with salinity in the surface waters ($r = -0.48$ and $r = -0.45$, respectively) and neither showed any correlation with salinity in the bottom waters. The departure from conservative behaviour at the surface can be attributed to anthropogenic inputs: Morris *et al.* (1981) demonstrated domestic and industrial inputs of orthophosphate in the lower 10 km of the nearby Tamar Estuary and Robards *et al.* (1994) attributed dissolved ammonium in surface waters to domestic pollution. An explanation of the non-conservative behaviour of ammonium and orthophosphate concentrations in the bottom waters of Sutton Harbour would have benefited from analyses of particulate and interstitial dissolved nitrogen and phosphorus. Consequently, the enrichment of the sediments with these nutrient species is not known. Orthophosphate has a tendency to adsorb more effectively onto particles with iron oxide surface coatings (Crosby, 1984), a mechanism whereby dissolved orthophosphate is deposited to the sediments, from where it can diffuse back into the water column following diagenetic release through the formation of FeS (Enoksson, 1993). In contrast, nitrate diffuses into anoxic sediment porewaters along the concentration gradient set up by bacterial reduction of nitrate to nitrite and a significant proportion is mineralised and then

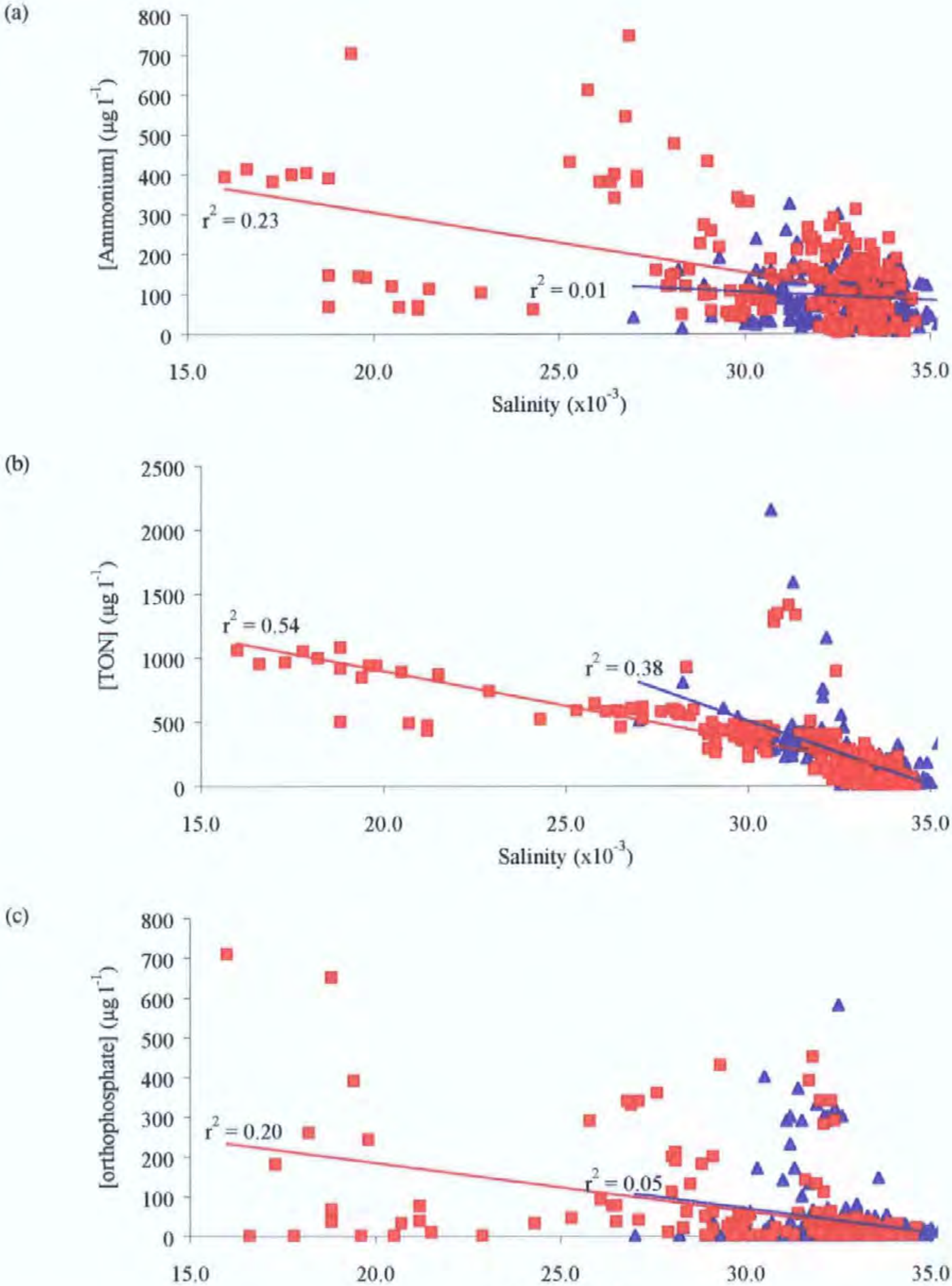


Fig. 4.1. Dissolved nutrient concentration versus salinity inside harbour (SH1-SH3) from November 1991 to November 1994 (n = 228) for (a) ammonium, (b) TON and (c) orthophosphate. Red squares represent surface samples and blue triangles represent bottom samples.

returns to the water column as ammonium (Nedwell, 1982; Caffrey, 1995), thereby representing a cyclic process of drawdown and release of nitrogen across the sediment-water interface. Knox *et al.* (1981) showed that the diffusion of ammonium into the water column from sediment porewaters could account for observed mid-estuary maxima of ammonium. Therefore, strong circumstantial evidence exists for sediment porewaters as an important source for both ammonium and orthophosphate. Watson *et al.* (1985a) estimated the fluxes of dissolved ammonium and orthophosphate from the sediment porewaters of the nearby lower Tamar Estuary. Their values were used to estimate the daily inputs from the Sutton Harbour sediments to the water column (Table 4.1). Although these values do not appear sufficient to explain the variability in Figs. 4.1a and c it must be noted that they are calculated for the harbour volume at MHWS and not for the bottom water layer extending perhaps 1 m from the bed in which the most non-conservative behaviour was observed. In combination with the body of evidence to suggest that several tidal cycles were required to renew the bottom water pre- and post-impoundment (Section 3.1) the values in Table 4.1 could increase by a factor of 10, thereby increasing the feasibility of the porewaters as a major source of ammonium and orthophosphate. The concentrations of TON compared favourably with those measured during a variety of studies involving axial transects of the nearby Tamar Estuary (Table 4.2), whilst the concentrations of orthophosphate and ammonium exhibited a wider range.

Table 4.1. Estimated fluxes of dissolved ammonium and orthophosphate from the porewaters of the lower Tamar Estuary (Watson *et al.*, 1985a) and estimated inputs to the harbour waters pre- and post-impoundment at MHWS using data from Table 3.3.

Nutrient	Estimated Tamar flux (mg m^{-2} d^{-1})	Input to harbour water (µg l^{-1} d^{-1}) Pre-impoundment	Post-impoundment
NH_4^+	2.5 - 16.7	0.4 - 2.7	0.4 - 2.7
PO_4^{3-}	5.7 - 37.1	0.9 - 5.9	0.9 - 5.9

Table 4.2. Comparative values of dissolved nutrient concentration measured in the nearby Tamar Estuary during axial transects and in Sutton Harbour during the present study.

Nutrient	Location	Range (µg l^{-1})	Reference
NH_3-N	Lower Tamar (S >20 x10^{-3})	10 - 200	Morris *et al.*, 1985
		20 - 300	Knox *et al.*, 1981
	Sutton Harbour	7 - 580	
NO_3-N	Lower Tamar (S >20 x10^{-3})	100 - 2000	Morris *et al.*, 1981
	Sutton Harbour	12 - 1470	
PO_4-P	Lower Tamar (S >20 x10^{-3})	5 - 65	Morris *et al.*, 1981
	Sutton Harbour	2 - 350	

4.1.1 Semi-diurnal and diurnal variability

The processes affecting short term variability of dissolved nutrient concentrations in the harbour system can be thought of as sources and sinks. Semi-diurnal differences arise due to nutrient association with the freshwater flux (for example TON) and should vary inversely with salinity (Morris *et al.*, 1981) exhibiting minima at LW and maxima at HW inside the harbour. Conversely, nutrients from sources within the harbour such as those associated with anoxic porewaters (for example ammonium) should vary with harbour volume (by dilution) and therefore exhibit higher concentrations at LW than at HW. These simplistic relationships are complicated by diffuse inputs such as the sewage outfalls outside the harbour and the vessels inside the harbour. Diurnal changes arise due to the differential uptake of nutrients for primary production, such that during the day when phytoplankton photosynthesis exceeds respiration the uptake of nutrients could be greater than during the night when respiration exceeds photosynthesis. Within the framework of the present study, these conditions should manifest as high nutrient concentrations in early morning samples and low nutrient concentrations in late afternoon samples and may act with or against the semi-diurnal variations. In this respect, the occasional inability to begin surveys at HW (Section 2.1) was viewed as an advantage, enabling otherwise complex semi-diurnal and diurnal processes to be resolved. Clearly the diurnal processes were prevalent during high primary productivity, whilst the semi-diurnal processes acted continually.

Ammonium concentrations generally exhibited greater semi-diurnal differences outside the harbour than inside. Fig. 4.2 illustrates this for the bottom waters during spring tides. Outside the harbour, ammonium concentrations were generally greater at LW than at HW, apart from in May 1992 and July 1993 when on both occasions intense algal blooms were observed during the mid-afternoon LW periods (Section 4.3), thus depleting the ammonium concentrations. A similar distribution was observed inside the harbour before impoundment, although the difference between HW and LW concentrations was less marked because HW concentrations were considerably greater than those outside. This may have been the result of outside ammonium-rich bottom water from LW being shunted into the harbour on the flood. The diurnal decreases in ammonium concentration observed inside the harbour in May 1994 and July 1994 were probably the result of uptake by algal blooms. In November 1994 the elevated HW concentrations could no longer be fully attributed to the tidal shunting mechanism and may therefore have been augmented by a significant flux of ammonium from the sediments. This seems likely since the Fickian diffusion of solutes from the sediment is known to be temperature dependent (Watson *et al.*, 1985a) and the bottom waters in November 1994 were considerably warmer (13.1 ± 0.1 °C) than in previous November surveys (10.8 ± 0.8 °C). Other studies have shown that epibenthic fluxes of solutes are also correlated with average primary production in the preceding month as a result of benthic anoxia (Hunt, 1983) and that this effect can persist in shallow sea regions (Southern Bight) from the phytoplankton bloom in April/May until February of the following year (Dehairs *et al.*, 1989). The concentrations of UIA varied with that of ammonium from which they were calculated.

(a)

(b)

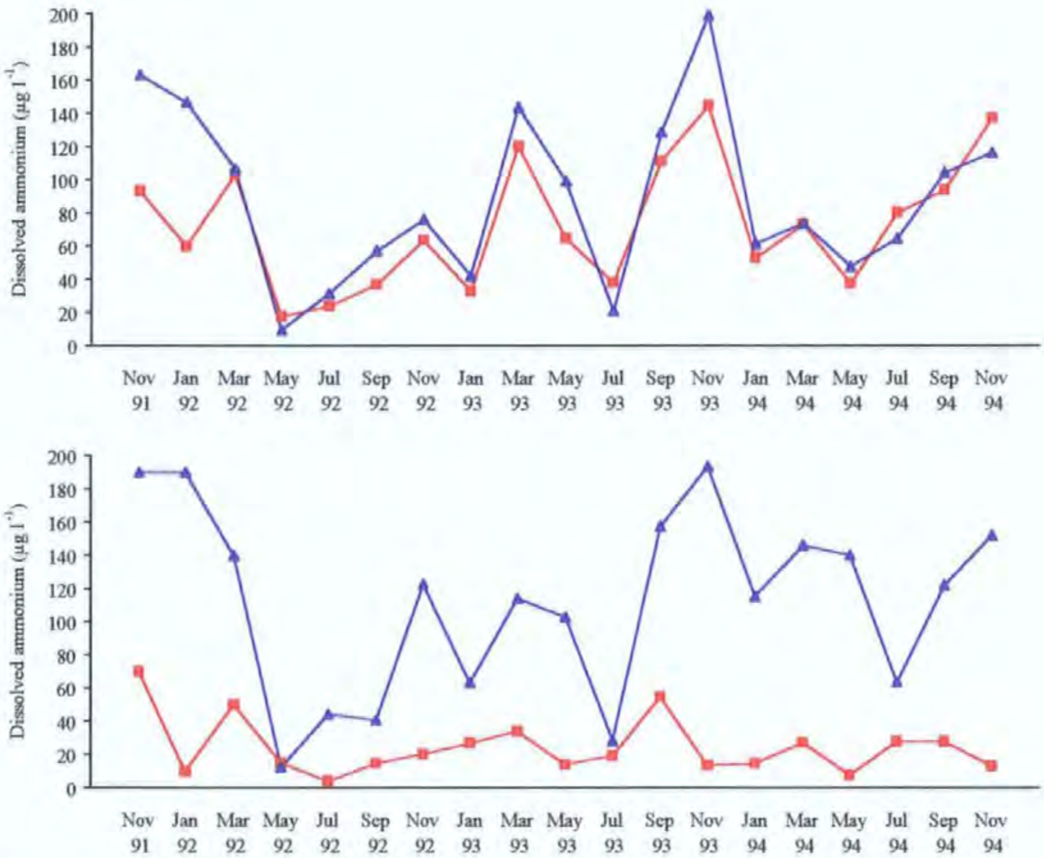


Fig. 4.2. Semi-diurnal variability of dissolved ammonium (μg l^{-1}) in the bottom waters during spring tides (a) inside harbour (mean of SH1 to SH3) and (b) outside harbour (mean of SH4 and SH5). Red squares represent HW and blue triangles represent LW.

Semi-diurnal and diurnal changes in dissolved nutrient concentrations were examined during periods of low inter-annual variability in environmental conditions (as defined in Section *3.2.2*) to provide evidence for inter-annual nutrient differences. These are discussed with reference to Fig. 4.3, in which histograms are presented comparing HW and LW concentrations during neap tides in July 1992, July 1993 and July 1994. With low riverine influx observed during these periods, the main sources of nutrients were sewage inputs and diffusion from sediment porewaters. Chlorophyll *a* concentrations indicated that primary productivity was high but variable during each occasion.

July 1992 (Figs. 4.3a and b). All nutrient concentrations exhibited a small decrease in concentration inside the harbour from the late morning HW to the late afternoon LW (ammonium decreased from 72 μg l^{-1} to 55 μg l^{-1} subsequently expressed as 72-55 μg l^{-1}; TON 61-58 μg l^{-1}; orthophosphate 21-17 μg l^{-1}) consistent with uptake by a modest bloom that was in decline following an intense bloom observed in the previous week rather than with any increase in salinity at LW. Outside the harbour, ammonium concentrations also decreased from HW to LW (79-51 μg l^{-1}), whilst TON and orthophosphate concentrations increased slightly from HW to LW (TON 63-78 μg l^{-1}; orthophosphate 18-20 μg l^{-1}), consistent with an incursion of brackish water from the Plym.

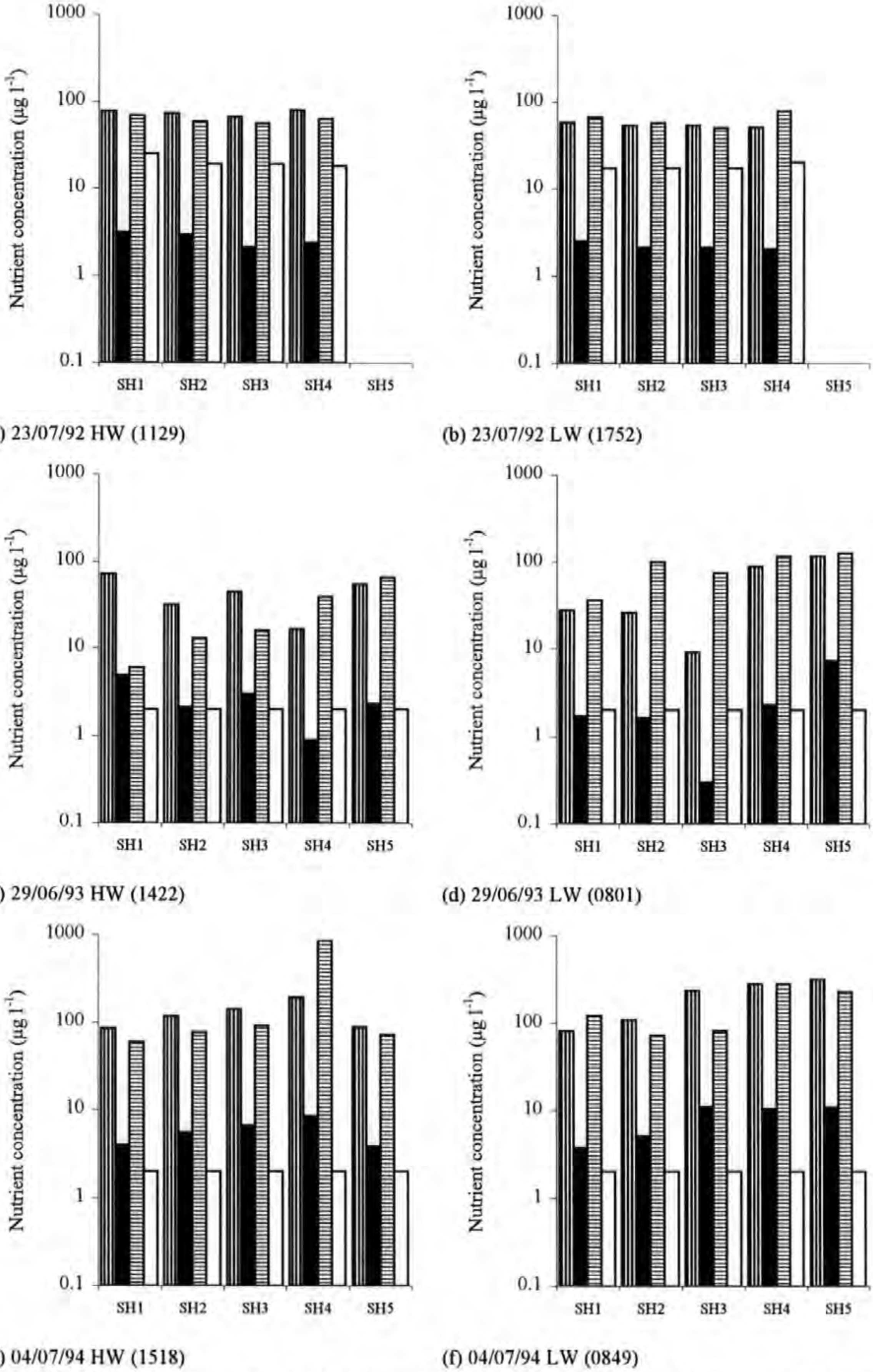


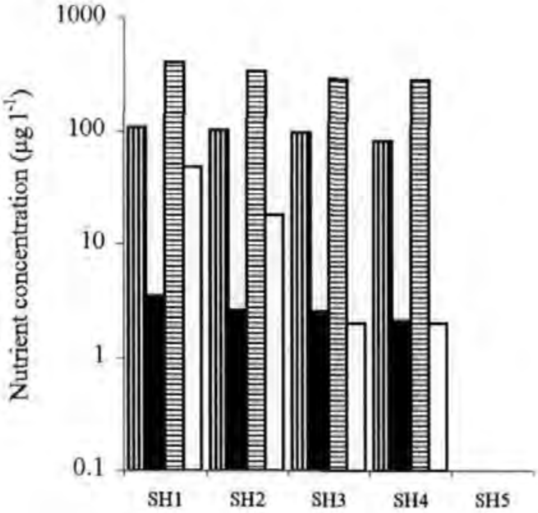
Fig. 4.3. Comparison of dissolved nutrient concentrations in surface water samples at neaps during July 1992, July 1993 and July 1994 surveys. Vertically striped bars represent ammonium, filled bars represent UIA, horizontally striped bars represent TON and open bars represent orthophosphate. Captions (a) to (f) denote survey date, tidal state and time of HW or LW (BST).

July 1993 (Figs. 4.3c and d). Ammonium concentrations increased from early morning LW to mid-afternoon HW inside the harbour (21-50 $\mu g\ l^{-1}$) as high concentrations flooded in from outside, whilst TON concentrations decreased from LW to HW (70-12 $\mu g\ l^{-1}$) despite high concentrations outside and must therefore have been depleted by the intense bloom observed throughout the system. Outside the harbour both ammonium and TON concentrations decreased from LW to HW (TON 120-52 $\mu g\ l^{-1}$; ammonium 100-36 $\mu g\ l^{-1}$) as a result of dilution by the flood tide and depletion by the intense bloom. Orthophosphate concentrations were below the limit of detection (2 $\mu g\ l^{-1}$) throughout.

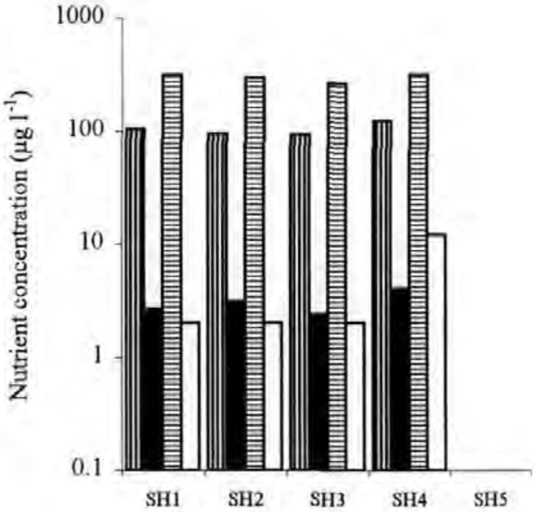
July 1994 (Figs. 4.3e and f). Greater inter-site variability was observed inside the harbour following impoundment. Differences between tidal states were no longer consistent, possibly resulting from reduced mixing and flushing. Ammonium concentrations increased slightly from early morning LW to mid-afternoon HW (95-100 $\mu g\ l^{-1}$) at SH1 and SH2 despite an intense bloom but decreased at SH3 (240-140 $\mu g\ l^{-1}$), whilst TON concentrations decreased at SH1 (120-59 $\mu g\ l^{-1}$) but increased slightly at SH2 and SH3 (77-85 $\mu g\ l^{-1}$). Bloom intensity was much reduced outside the harbour over previous July conditions and nutrient concentrations were therefore unusually high. Ammonium concentrations decreased by dilution with the flood tide from LW to HW, whilst TON concentrations decreased at SH5 (230-73 $\mu g\ l^{-1}$) yet increased to an unseasonal high at SH4 (280-850 $\mu g\ l^{-1}$). This large increase in TON concentration cannot readily be explained in the absence of high riverine influx, but does help to explain the slight HW increase observed at SH2 and SH3. Orthophosphate concentrations were again below the limit of detection (2 $\mu g\ l^{-1}$) throughout, but did not appear to biolimit the intense bloom inside the harbour.

Semi-diurnal and diurnal variability of dissolved nutrients during periods of low primary productivity was also examined and is discussed with reference to Fig. 4.4, in which histograms are presented comparing HW and LW concentrations during spring tides in November 1992, November 1993 and November 1994. River flow was increasing towards winter spate conditions during these periods and was therefore the major source of nutrients. Ammonium and TON concentrations compared favourably with the Tamar data (Table 4.2) but orthophosphate concentrations were considerably greater. Morris *et al.* (1981) observed consistent peaks at 30 $\times 10^{-3}$ salinity during 10 axial transects in one year and attributed these to anthropogenic inputs. It would seem that a significant additional input was required to cause the greater orthophosphate concentrations observed during the present study, namely fluxes from sediment porewaters. Chlorophyll *a* concentrations confirmed a period of low primary productivity when diurnal variability was not expected to be significant in comparison with semi-diurnal variability.

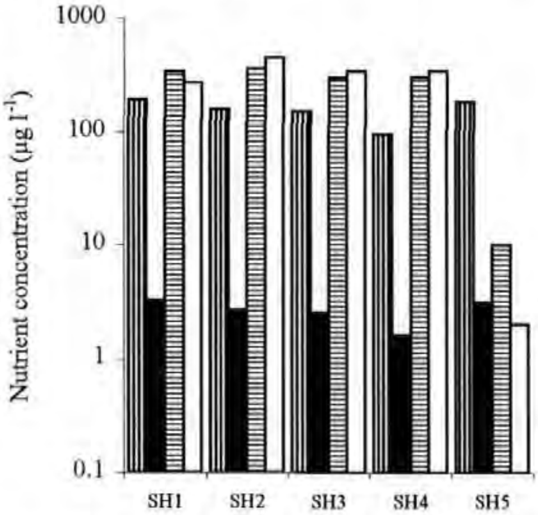
November 1992 (Figs. 4.4a and b). The concentration of nutrients inside the harbour decreased to varying extents from early morning HW to mid-afternoon LW (ammonium



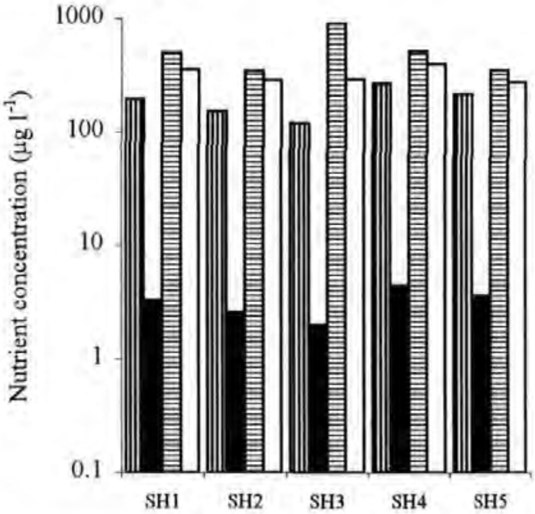
(a) 13/11/92 HW (0716)



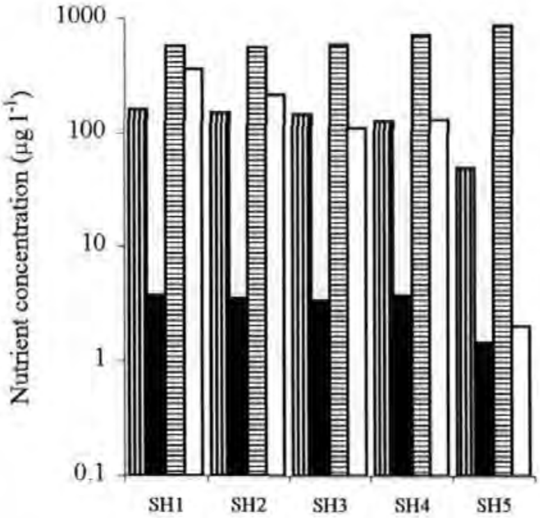
(b) 13/11/92 LW (1329)



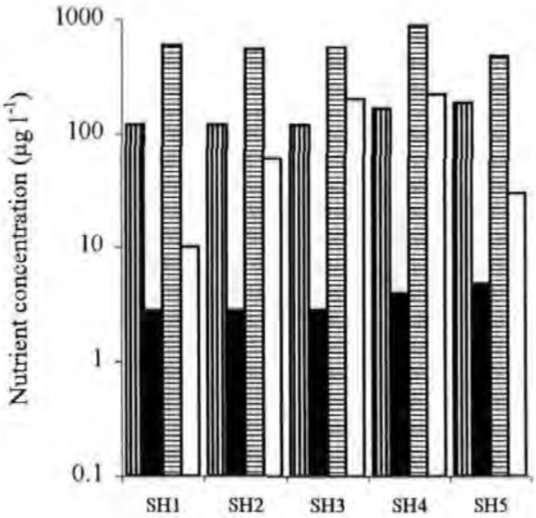
(c) 29/11/93 HW (1752)



(d) 29/11/93 LW (1144)



(e) 22/11/94 HW (0740)



(f) 22/11/94 LW (1349)

Fig. 4.4. Comparison of dissolved nutrient concentrations in surface water samples at springs during November 1992, November 1993 and November 1994 surveys. Vertically striped bars represent ammonium, filled bars represent UIA, horizontally striped bars represent TON and open bars represent orthophosphate. Captions (a) to (f) denote survey date, tidal state and time of HW or LW (GMT).

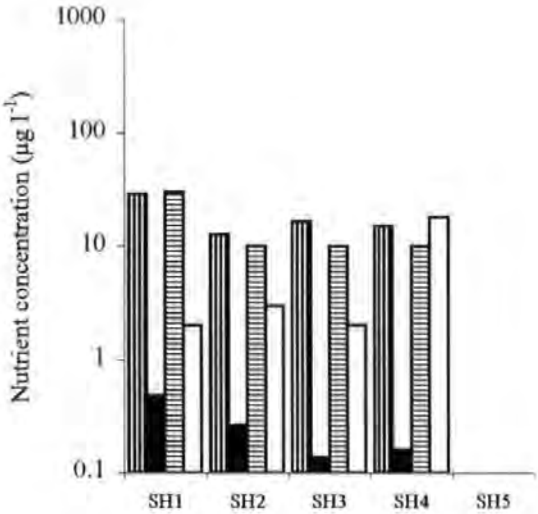
100-97 μg l^{-1}; TON 340-290 μg l^{-1}; orthophosphate 23-2 μg l^{-1}), whilst outside the harbour values rose (ammonium 81-120 μg l^{-1}; TON 280-310 μg l^{-1}; orthophosphate 2-12 μg l^{-1}) from HW to LW. This distribution is entirely consistent with the tidal shunting mechanism already discussed (Section *3.2.3*).

November 1993 (Figs. 4.4c and d). Concentrations of ammonium inside the harbour increased from late morning LW to late afternoon HW (150-170 μg l^{-1}), as did those of TON at SH2 and orthophosphate at SH2 and SH3 (TON 340-360 μg l^{-1}; orthophosphate 290-400 μg l^{-1}). However, TON concentrations at SH1 and SH3 and orthophosphate concentrations at SH1 decreased from LW to HW (TON 694-321 μg l^{-1}; orthophosphate 350-270 μg l^{-1}). This observation may have been the result of contribution by localised nutrient sources, particularly for orthophosphate, where LW bottom concentrations at SH1 reached 580 μg l^{-1}. Outside the harbour, a more typical distribution of decreasing nutrient concentrations (ammonium 240-140 μg l^{-1}; TON 430-160 μg l^{-1}; orthophosphate 330-170 μg l^{-1}) was observed from late morning LW to late afternoon HW as the flood tide diluted the nutrient-rich estuarine waters.

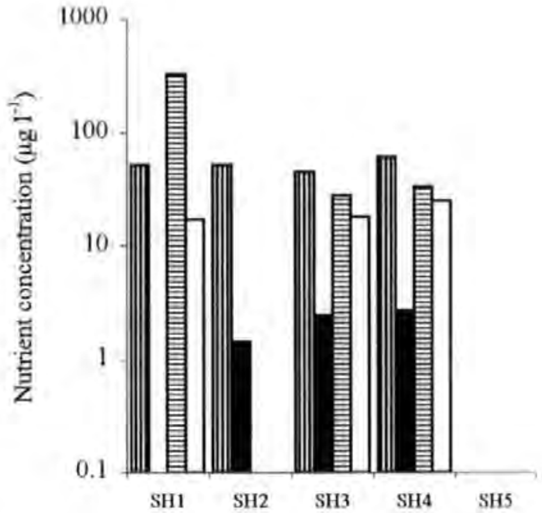
November 1994 (Figs. 4.4e and f). Ammonium concentrations inside the harbour decreased from early morning HW to mid-afternoon LW (150-120 μg l^{-1}), apparently in contradiction of the earlier observation that ammonium was diffusing significantly from the sediments. However, another possibility (assuming low biological activity) is that ammonium continued to be shunted into the harbour in the latter stages of the flood tide, having been released from sources other than the estuary, from mid-flood onwards. Concentrations of TON at SH2 and SH3 and of orthophosphate at SH1 and SH2 also decreased from HW to LW (TON 570-560 μg l^{-1}; orthophosphate 290-35 μg l^{-1}) but increased concentrations of TON at SH1 and orthophosphate at SH3 were observed over the same period (TON 580-600 μg l^{-1}; orthophosphate 110-200 μg l^{-1}). These are attributed, as in November 1993, to localised inputs. Outside the harbour, nutrient concentrations increased from HW to LW as brackish estuarine water pervaded (ammonium 88-180 μg l^{-1}; TON 730-890 μg l^{-1}; orthophosphate 66-130 μg l^{-1}). The exception was TON at SH5, where the concentration decreased considerably from HW to LW (880-480 μg l^{-1}). All the evidence points to the sediments as an additional near-field source that is sometimes nutrient-specific. For example, Watson *et al.* (1985b) observed that interstitial orthophosphate concentrations varied seasonally in the lower Tamar Estuary and in inner Carmarthen Bay, according to the rates of microbial degradation and formation of ferro-phosphate minerals.

4.1.2 Spring-neap tide variability

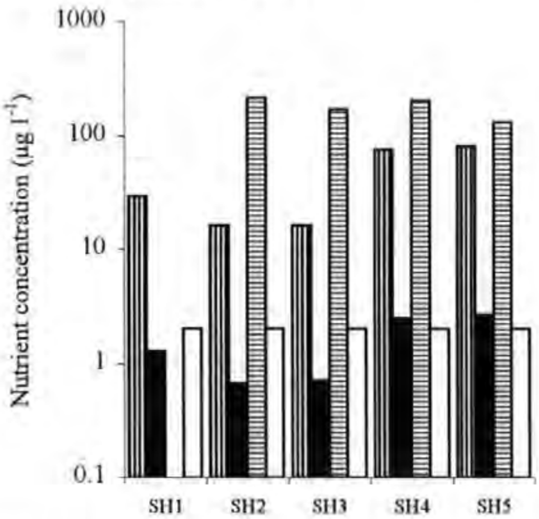
The contrast between of dissolved nutrient concentrations at spring and at neap tides was examined during periods of low variability in environmental conditions (Section *3.2.2*) and high variability in primary production as indicated by chlorophyll *a* concentrations. In May 1992, an intense phytoplankton bloom was observed throughout the harbour system during the spring survey that was completely absent one week later at neaps. In May 1994, after impoundment, a moderate phytoplankton bloom was observed inside the harbour at neaps and became an intense bloom one week later at springs, whilst outside the harbour primary productivity was low throughout the survey period. It was postulated that these variations should also occur in dissolved nutrient concentrations, which are shown for HW surface samples in Fig. 4.5.



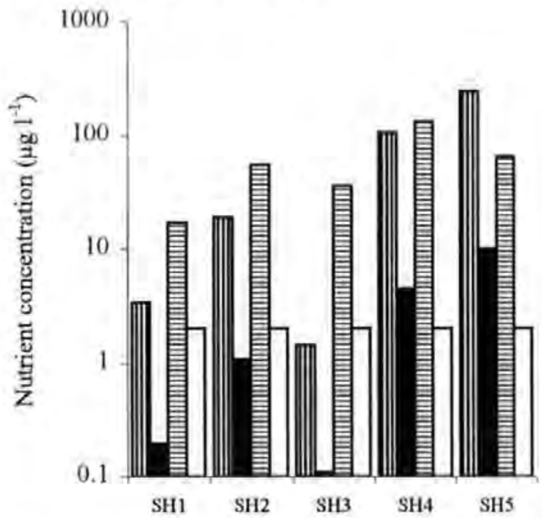
(a) 18/05/92 Springs (HW 0740)



(b) 26/05/92 Neaps (HW 1325)



(c) 17/05/94 Neaps (HW 1040)



(d) 24/05/94 Springs (HW 1819)

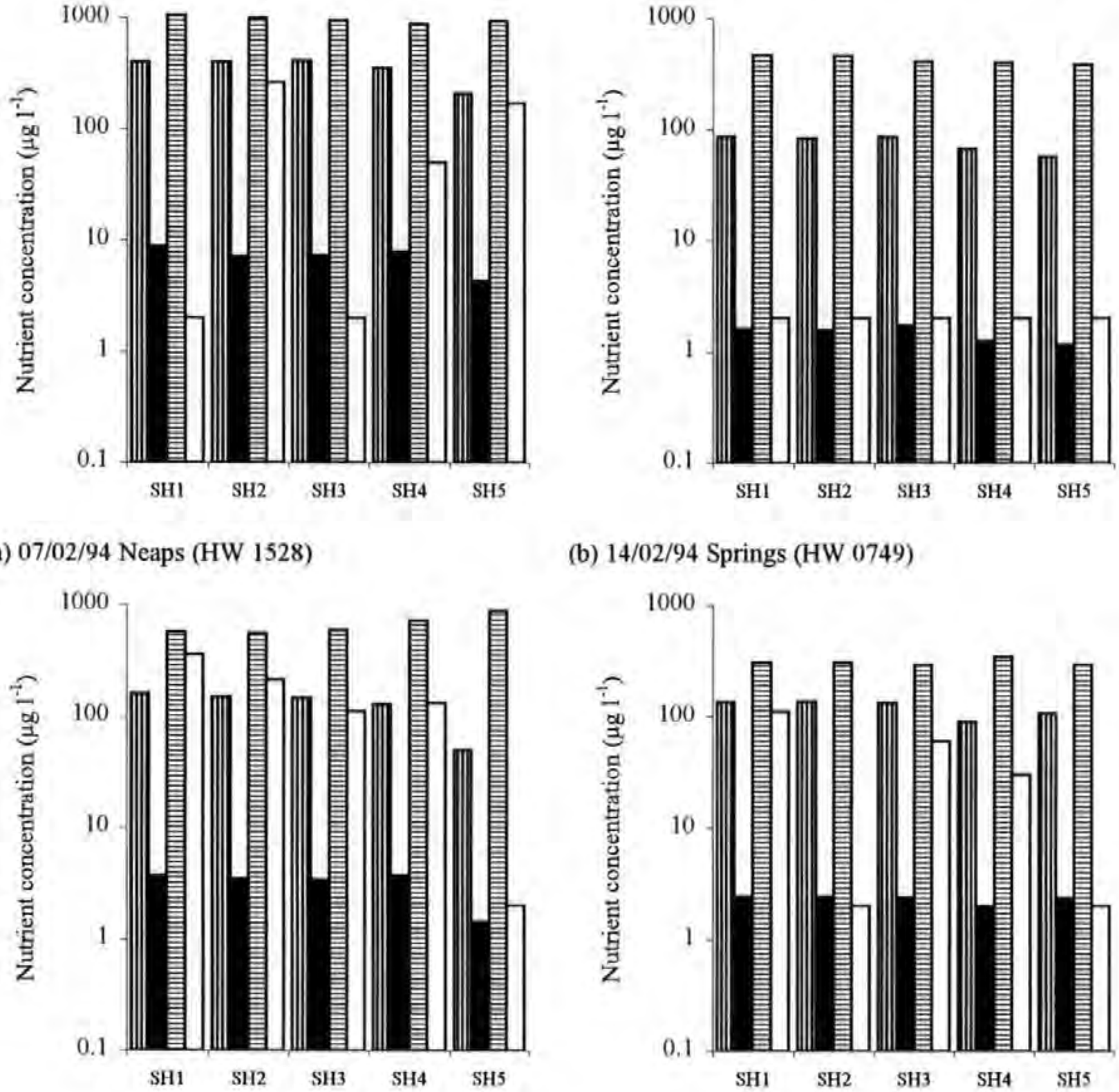
Fig. 4.5. Comparison of dissolved nutrient concentrations in surface water samples at HW during May 1992 and May 1994 spring and neap surveys. Vertically striped bars represent ammonium, filled bars represent UIA, horizontally striped bars represent TON and open bars represent orthophosphate. Captions (a) to (d) denote survey date, tidal range and time of HW (BST).

May 1992 (Figs. 4.5a and b). Early morning HW concentrations of all nutrients inside and outside the harbour (ammonium 19 and 15 $\mu g\ l^{-1}$; TON 17 and 10 $\mu g\ l^{-1}$; orthophosphate 2 and 18 $\mu g\ l^{-1}$, respectively) were atypically low at springs. These low values may have limited further phytoplankton growth, thereby altering the trophic balance. Early afternoon HW concentrations of all nutrients inside and outside the harbour (ammonium 50 and 61 $\mu g\ l^{-1}$; TON 180 and 33 $\mu g\ l^{-1}$; orthophosphate 18 and 25 $\mu g\ l^{-1}$, respectively) had increased considerably at neaps. According to the salinity distribution, influx of nutrients from the Plym was negligible. Increased nutrient concentrations were probably therefore chiefly attributable to the degradation of phytoplanktonic material in the water column, and to porewater inputs.

May 1994 (Figs. 4.5c and d). At neaps, mid-morning HW concentrations of ammonium were lower inside the harbour than outside (20 and 79 $\mu g\ l^{-1}$, respectively) as the moderate bloom utilised ammonium preferentially to TON. Concentrations of the latter were higher inside the harbour than outside at this time (190 and 170 $\mu g\ l^{-1}$, respectively). One week later at springs, an intense bloom had developed inside the harbour, thus depleting concentrations of ammonium (8 $\mu g\ l^{-1}$) and TON (36 $\mu g\ l^{-1}$), whilst low primary productivity was evident outside despite elevated concentrations of ammonium (170 $\mu g\ l^{-1}$) and TON (99 $\mu g\ l^{-1}$). Orthophosphate concentrations were at or below the limit of detection of 2 $\mu g\ l^{-1}$ throughout the May 1994 survey period. This may therefore have limited primary productivity outside the harbour, whilst a supply from the porewaters inside the harbour supported the bloom there. The implications of this observation are discussed in Section *4.3.2*.

The contrast between dissolved nutrient concentrations at spring and neap tides was also examined during periods of high variability in environmental conditions (Section *3.2.2*) and low primary productivity as indicated by chlorophyll *a* concentrations. High precipitation during the winter of 1993/1994 led to extreme variability of surface salinities. The January 1994 survey period spanned a transition from conditions of low surface salinity at neaps to high surface salinity at springs throughout the harbour system. In November 1994, during the first significant (but short-lived) period of precipitation since impoundment, surface salinities were lowered at springs, but recovered at neaps. It was postulated that the variations in salinity should reflect changes in dissolved nutrient concentrations to a greater extent in winter than during periods of variable primary productivity. HW surface dissolved nutrient concentrations during these periods are shown in Fig. 4.6.

January 1994 (Figs. 4.6a and b). Concentrations of all nutrients were high throughout the harbour system at neaps, as indicated by low salinity (18 to 21 $\times 10^{-3}$). Tidal shunting was evident in the shape of higher concentrations of ammonium and TON inside the harbour than outside (ammonium 410 and 280 $\mu g\ l^{-1}$; TON 1000 and 910 $\mu g\ l^{-1}$, respectively).



(a) 07/02/94 Neaps (HW 1528)

(b) 14/02/94 Springs (HW 0749)

(c) 22/11/94 Springs (HW 0740)

(d) 29/11/94 Neaps (HW 1400)

Fig. 4.6. Comparison of dissolved nutrient concentrations in surface water samples at HW during January 1994 and November 1994 spring and neap surveys. Vertically striped bars represent ammonium, filled bars represent UIA, horizontally striped bars represent TON and open bars represent orthophosphate. Captions (a) to (d) denote survey date, tidal range and time of HW (GMT).

Orthophosphate exhibited the opposite trend with concentrations of 88 $\mu g\ l^{-1}$ inside and 110 $\mu g\ l^{-1}$ outside. One week later at springs, the combination of increased tidal mixing and reduced riverflow that increased surface salinities to 30 $\times 10^{-3}$ also resulted in a considerable reduction in dissolved nutrient concentrations throughout. Tidal shunting was still in evidence through higher concentrations of ammonium and TON inside than outside (ammonium 84 and 61 $\mu g\ l^{-1}$; TON 440 and 390 $\mu g\ l^{-1}$) but orthophosphate concentrations were <2 $\mu g\ l^{-1}$ throughout the harbour system.

November 1994 (Figs. 4.6c and d). Low surface salinities prevailed again at springs (26 to 27 $\times 10^{-3}$). Concentrations of TON that showed the strongest inverse correlation with salinity in Section *4.1.3* were lower inside the harbour than outside (580 and 800 μg l^{-1}) illustrating the reduction in tidal shunting. However, concentrations of ammonium and orthophosphate were higher inside the harbour than outside (ammonium 150 and 88 μg l^{-1}; orthophosphate 230 and 66 μg l^{-1}) and may therefore have received a significant contribution from the anoxic porewaters. One week later at neaps under increased surface salinities (32 $\times 10^{-3}$) concentrations of all nutrients had decreased accordingly, but the relative concentrations inside and outside the harbour were unchanged (ammonium 130 and 98 μg l^{-1}; TON 300 and 320 μg l^{-1}; orthophosphate 57 and 16 μg l^{-1}). This evidence supports the hypothesis that near-field sources of nutrients (specifically ammonium and orthophosphate) were more significant after impoundment.

The spring-neap tide variability exhibited in January 1994 was surprising given the high precipitation during the winter period and could be used to advantage in the harbour management plan following impoundment, if the intention to periodically flush the harbour is upheld. Clearly opening the lock gates to free flow over a spring tidal cycle (preferably at night to minimise inconvenience for harbour users) would significantly reduce the nutrient concentrations in the harbour waters, provided that the surface water salinity was sufficiently high outside the harbour on the preceding LW. This could be effected with a permanently mounted solid-state device with no need for more expensive nutrient determinations. If this process was repeated several times each winter after successive periods of high river flow the potential for nutrient retention in the harbour and hence the intensity of spring phytoplankton blooms could be reduced considerably.

4.1.3 Seasonal and inter-annual variability

Seasonal variability in dissolved nutrient concentrations was driven to varying extents by seasonal cycles in river flow (a primary nutrient source) manifested as salinity and then modified by seasonal variability in primary production (a primary nutrient sink). The understanding of seasonal nutrient variations in the harbour system is central to the management plan if the potential for development of eutrophic conditions is to be minimised. To examine the salinity-nutrient relationships, correlation coefficients between salinity and dissolved nutrient concentrations were calculated over the entire study period (Table 4.3). No significant correlation was observed between salinity and ammonium, UIA or orthophosphate at springs, suggesting either that the high degree of tidal mixing effectively homogenised the nutrient distribution or that the secondary nutrient sources were more significant. The latter explanation is more plausible since TON concentrations, shown to be strongly associated with river flow in Section 4.1, exhibited slight significant inverse correlations with salinity at surface and bottom.

Table 4.3. Correlation coefficients (r) between salinity and dissolved nutrients inside harbour from November 1991 to November 1994 (n = 19). All data were tidally-averaged between HW and LW and spatially averaged between SH1, SH2 and SH3.

Nutrient	Springs		Neaps	
	Surface	Bottom	Surface	Bottom
Ammonium	NS	NS	-0.54[a]	NS
UIA	NS	NS	-0.48[a]	NS
TON	-0.41[a]	-0.55[a]	-0.97[b]	-0.90[b]
Orthophosphate	NS	NS	-0.74[b]	-0.68[b]

NS - not significant at $p < 0.05$; [a]$p < 0.05$; [b]$p < 0.005$.

Ammonium and UIA concentrations continued to exhibit no correlation with salinity at the bottom at neaps, suggesting that the porewater source was dominant, whilst slight significant inverse correlations with salinity were observed at the surface. However, TON and orthophosphate concentrations showed strong significant inverse correlations with salinity at surface and bottom, with an almost linear TON-salinity relationship ($r^2 = 0.94$) at the surface. The relationships of TON and orthophosphate with salinity at springs and neaps are illustrated by the time series in Figs. 4.7a, b, c and d. It is immediately apparent that both TON and orthophosphate undergo dynamic seasonal variations of 2 to 3 orders of magnitude.

Neaps (Figs. 4.7c and d). The response of nutrient concentrations to the winter pulses of brackish water was particularly evident at neaps when increased stratification enhanced the surface-bottom concentration gradients. TON and orthophosphate concentrations exhibited distinctive seasonality, with surface and bottom maxima between November and March and surface and bottom minima between May and September each year. The impoundment in 1994 had no apparent effect on established seasonal trends. Surface concentrations of both nutrients were generally greater than bottom concentrations inside and outside the harbour, with maximum vertical differences between November and March and minimum vertical differences between May and September, concordant with observed variations in salinity (Section *3.2.5*) and primary productivity (Section *4.3.3*).

Springs (Figs. 4.7a and b). The response of TON concentrations to the brackish water pulses was less clear at springs and orthophosphate concentrations showed little dependence on salinity although seasonal cycles were still evident. Seasonal maxima and minima were again observed between November and March and May and September, respectively, although with certain perturbations. For example, unseasonally high precipitation between neaps and springs in September 1993 resulted in the highest concentrations of TON observed during the study. Also, a reduction in river flow between neaps and springs in January 1994 (Section *4.1.2*) lowered the orthophosphate concentrations below their detection limit. Surface TON concentrations were generally

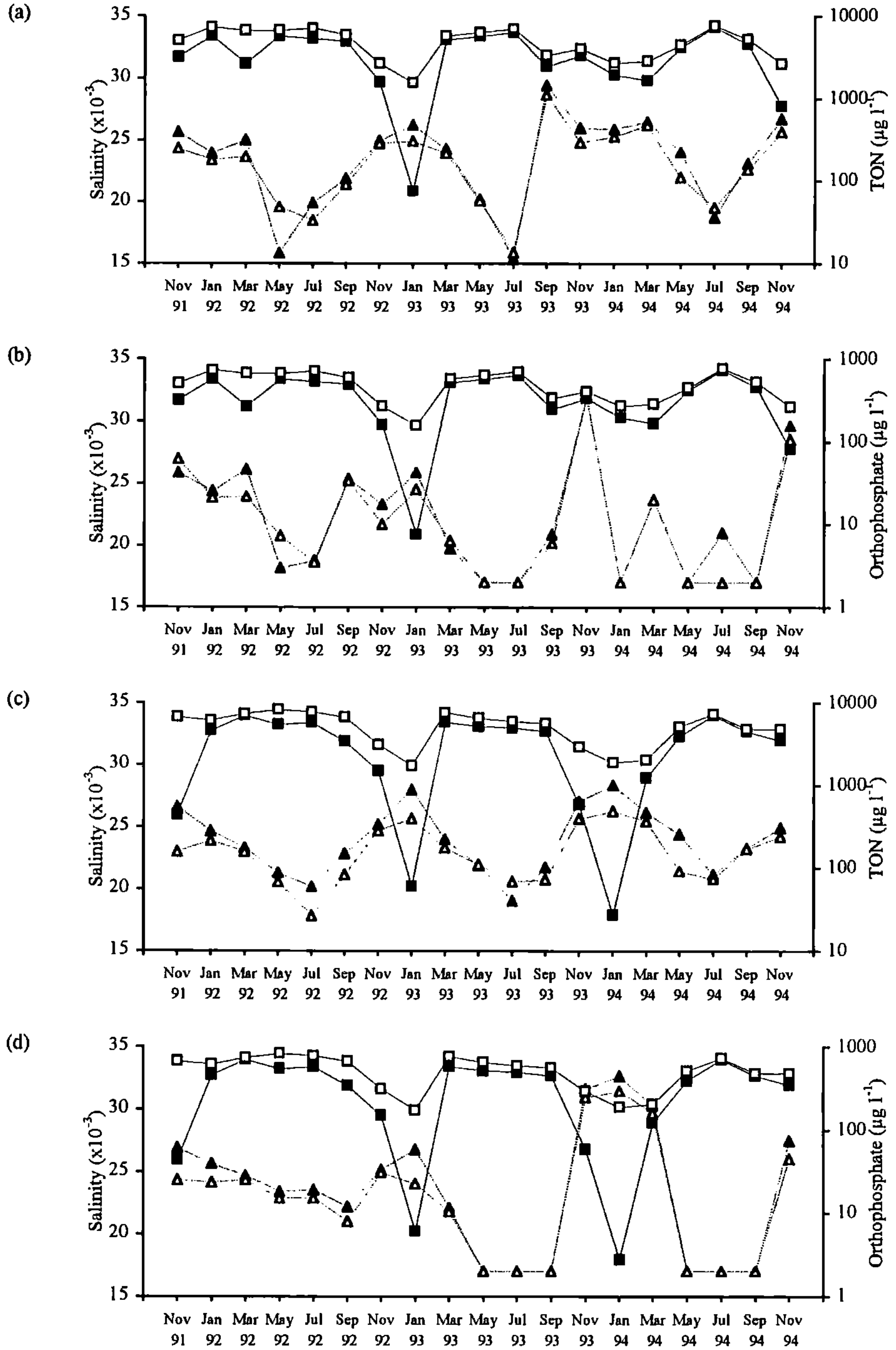


Fig. 4.7. Tidally-averaged time series of salinity and concentrations of (a) TON at springs, (b) orthophosphate at springs, (c) TON at neaps and (d) orthophosphate at neaps inside Sutton Harbour from November 1991 to November 1994. Squares represent salinity, triangles represent dissolved nutrient; filled symbols are near-surface, open symbols are near-bottom. All values are mean of SH1 to SH3.

greater than those at bottom both inside and outside the harbour, apart from during periods of high primary productivity (May 1992, July 1993 and July 1994) when the reverse was observed. Surface orthophosphate concentrations were also generally greater than or equal to bottom concentrations inside and outside the harbour, except during November 1991 and March 1993 (possibly from enhanced porewater inputs) and May 1992, when primary productivity was greater at the surface than at the bottom.

Seasonal variability of dissolved ammonium concentrations is shown in Figs. 4.8a and b. The seasonality of reduced forms of nitrogen was not as distinct as that exhibited by by TON or orthophosphate. As a consequence, inter-annual variability was greater. At springs (Fig. 4.8a), dissolved ammonium exhibited surface and bottom maxima during March 1992 and March 1993 (the end of the winter period) and in November 1993 (the beginning), and surface and bottom minima during periods of high primary productivity (May 1992, July 1993 and May and July 1994). At neaps (Fig. 4.8b), surface and bottom maxima in ammonium were observed during November 1991 and November 1993 (before the winter period) and in March 1993 (after). Surface and bottom minima were again recorded during periods of high primary productivity (May 1992, July 1993 and May 1994). Surface concentrations of ammonium were generally greater than bottom values inside the harbour at springs except in November 1993 (low riverine input) and during phytoplankton blooms, and at neaps, except in September 1993 (low riverine input) and between May 1994 and September 1994. The latter observation may have been due to existence of high productivity in the surface waters after impoundment combined with a possible increase in porewater release as mean temperatures increased (Section *3.2.5*). Surface concentrations were always greater than bottom outside the harbour at springs and neaps owing to the influence of the sewage outfalls.

The seasonal variability of concentrations of UIA is shown in Figs. 4.8c and d. UIA is not strictly a nutrient; the lack of charge making it lipid-soluble and therefore more toxic to marine teleosts than hydrated ammonium ions (Bower and Bidwell, 1978), but it is considered here by association. Calculated values of UIA were proportional to temperature and pH and inversely proportional to salinity, such that the effects of the seasonal variability of the first and the last acted in opposition upon the equilibrium equation (Whitfield, 1974; DOE, 1982a):

$$NH_4^+ + H_2O \rightleftarrows NH_3 + H_3O^+$$

thus ameliorating the net effect. Surface and bottom maxima were observed at springs during winter between November and March (Fig. 4.8c) and minima were recorded in May 1992 and July 1993 during high productivity. A surface minimum was also observed in May 1994, although increased bottom temperatures and reduced salinities at

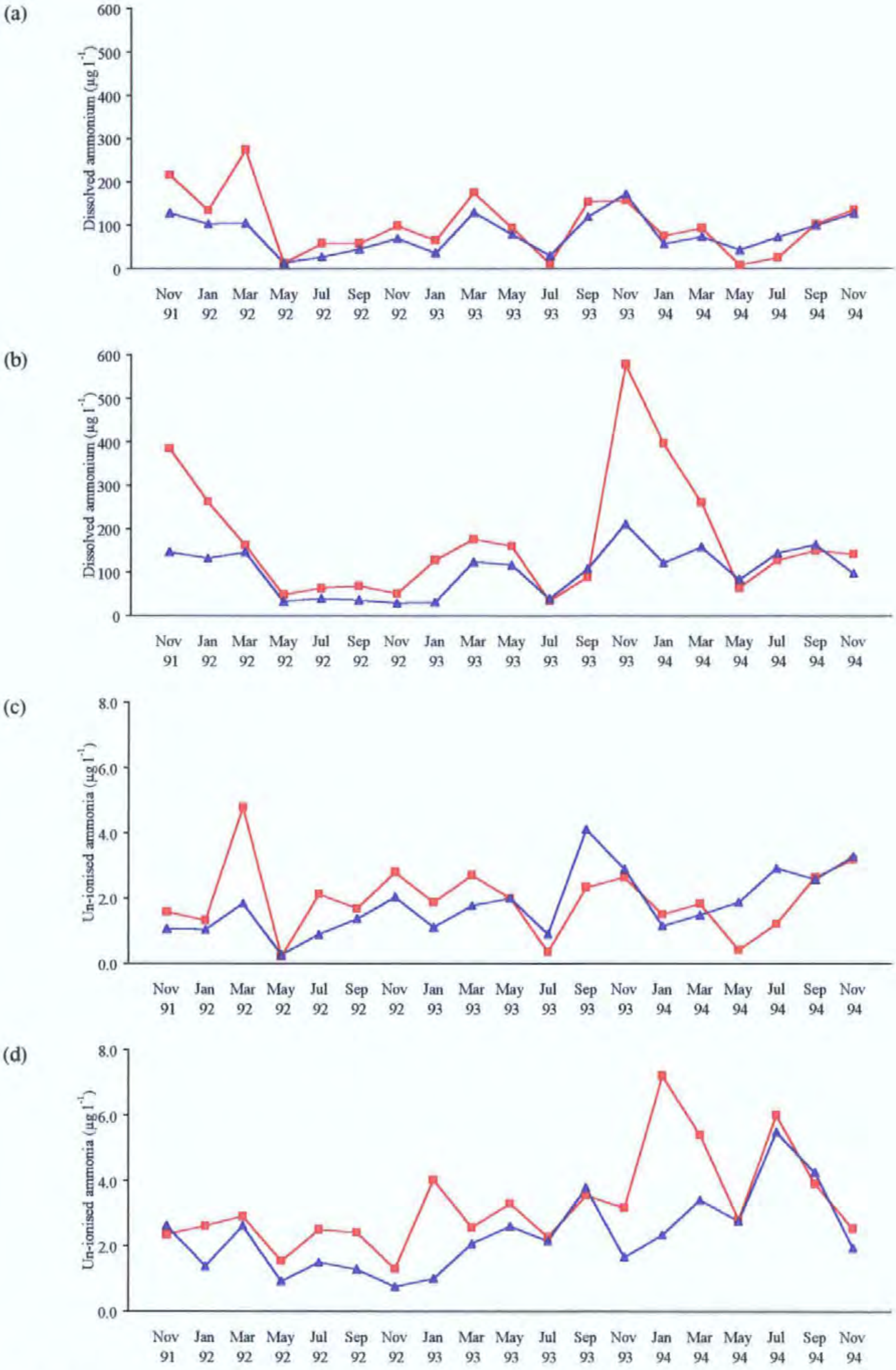


Fig. 4.8. Tidally-averaged time series of dissolved ammonium concentrations at (a) springs and (b) neaps and UIA concentrations at (c) springs and (d) neaps inside Sutton Harbour from November 1991 to November 1994. Red lines represent surface samples and blue lines represent bottom samples. All values are mean of SH1 to SH3.

the bottom combined to maintain bottom concentrations. Surface maxima were also observed in winter between November and March at neaps (Fig. 4.8d) with surface minima in May 1992 and July 1993. However, the bottom trend in the more stratified water column was reversed with bottom minima between November and January and bottom maxima between May and September. This reversed bottom trend continued after impoundment and was also observed in the surface waters with the highest summer concentrations of the entire study (*circa* 6 $\mu g\ l^{-1}$) in July 1994. This effect was caused by a combination of elevated July ammonium concentrations and higher temperatures and may be exacerbated during future summers that are warmer than in 1994. However, the concentrations were considerably lower than the draft EQS for UIA of 21 $\mu g\ l^{-1}$ (ENDS, 1992). Surface UIA concentrations were generally greater than or equal to bottom concentrations both inside and outside the harbour. However, bottom concentrations were greater than surface concentrations inside the harbour at springs in July and September 1993 and in May and July 1994 as a result of high chlorophyll *a*.

The time series in Fig. 4.9 show the difference in dissolved nutrient concentrations between inside and outside the harbour for the surface waters and the bottom waters at springs:

Ammonium (Fig. 4.9a). Surface concentrations of ammonium were generally greater outside the harbour than inside except during the winter periods of January 1992 to March 1992 and March to May 1993. The bottom concentrations inside the harbour were generally greater than or equal to those outside except during the winter periods of January 1993 and January 1994 to March 1994. These distributions may have been a consequence of the relative sources of ammonium inside and outside the harbour, with sewage outfalls dominating the surface inputs outside the harbour and porewater inputs dominating the bottom inputs inside the harbour. Surface and bottom concentrations were enhanced outside the harbour in May 1994 by the greatest margin of the study (+160 $\mu g\ l^{-1}$ and +40 $\mu g\ l^{-1}$, respectively) because of the rapid utilisation of ammonium throughout the water column by the phytoplankton bloom inside the harbour.

UIA (Fig. 4.9b). The concentrations of UIA followed an almost identical pattern to those of ammonium and again showed unprecedented surface and bottom elevation outside the harbour in May 1994 (+6.5 $\mu g\ l^{-1}$ and +2 $\mu g\ l^{-1}$, respectively).

TON (Fig. 4.9c). Surface and bottom TON concentrations reflected the differences in salinity between inside and outside (Section *3.2.5*) such that before impoundment concentrations inside the harbour were greater than (notably in September 1993 and November 1993) or equal to those outside, except in November 1991 and January 1993 when salinities were lower outside the harbour. Bottom concentrations outside were greater than or equal to those inside between May 1994 and November 1994, whilst the

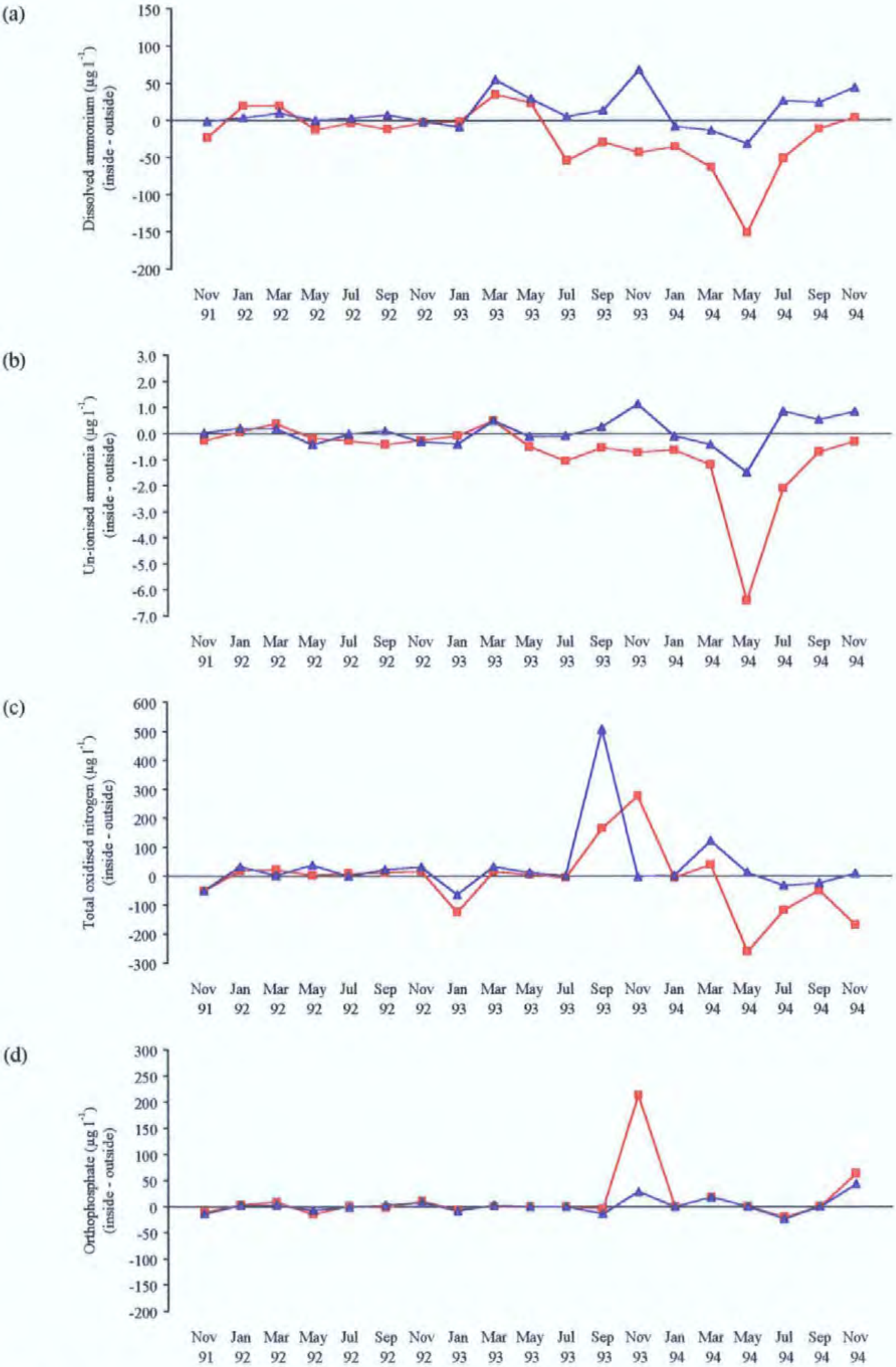


Fig. 4.9. Time series of spring tide differences in dissolved nutrient concentrations between inside harbour (mean of SH1 to SH3) and outside harbour (mean of SH4 to SH5) for (a) dissolved ammonium, (b) UIA, (c) TON and (d) orthophosphate. Red lines represent surface and blue lines represent bottom.

surface concentrations were greater outside than inside for this entire period. This was particularly evident in May 1994 when the greatest outside enhancement of the entire study was observed (+250 $\mu g\ l^{-1}$), owing to the relative scarcity of phytoplankton there as indicated by chlorophyll *a* concentration.

Orthophosphate (Fig. 4.9d). Concentrations of orthophosphate exhibited very little difference inside and outside the harbour except during November 1993, when surface concentrations were particularly great inside the harbour (210 $\mu g\ l^{-1}$) owing to lower surface salinity (ΔS 0.5 $x10^{-3}$). Similar circumstances were observed during November 1994, when elevated concentrations throughout the system were particularly high at the surface (50 $\mu g\ l^{-1}$) and the bottom (30 $\mu g\ l^{-1}$) inside the harbour coincident with lower bottom salinities (ΔS 1.8 $x10^{-3}$) but despite higher surface salinities (ΔS 1.3 $x10^{-3}$). These observations suggest that following impoundment, orthophosphate concentrations may be enhanced inside the harbour by internal sources during winter.

The time series in Fig. 4.10 demonstrate the difference between dissolved nutrient concentrations inside and outside the harbour for surface and bottom waters at neaps.

Ammonium (Fig. 4.10a). Similar patterns of ammonium distribution inside and outside the harbour were observed as at springs, although differences between surface and bottom were generally more marked as a result of increased stratification. Surface concentrations outside were again generally greater than or equal to those inside, except during the winter periods of January 1993 to March 1993 and January 1994 during which surface salinities inside were considerably lower than those outside (Fig. 3.21b). The greatest enhancement outside (+200 $\mu g\ l^{-1}$) was observed during November 1993 when a period of low salinity throughout occurred. Salinity was slightly lower (ΔS 1 $x10^{-3}$) outside. Bottom concentrations were generally greater inside than outside, except during the winter periods of January 1993 (when low concentrations throughout were slightly higher outside the harbour) and November 1993 (when higher concentrations throughout were slightly enhanced inside the harbour). Surface concentrations in May 1994 were greater outside (+180 $\mu g\ l^{-1}$) by almost as much as in November 1993. These conditions also prevailed during July 1994 (+90 $\mu g\ l^{-1}$) and September 1994 (+80 $\mu g\ l^{-1}$), when each occasion constituted the greatest outside enhancement observed during the study for that month. Bottom concentrations remained elevated inside the harbour between May 1994 and November 1994 (particularly in September 1994) illustrating the potential for increased ammonium release from the sediments after impoundment.

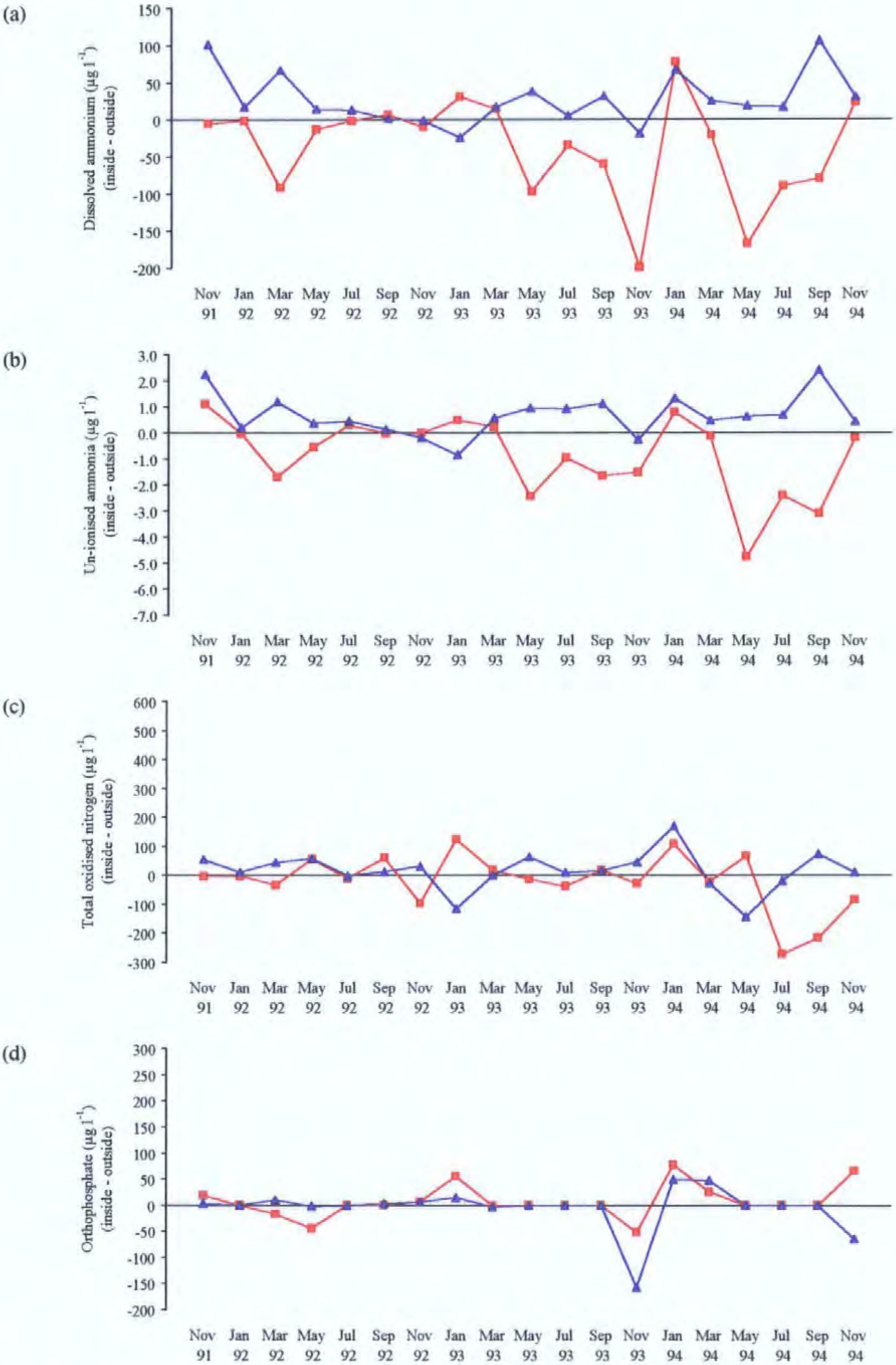


Fig. 4.10. Time series of neap tide differences in dissolved nutrient concentrations between inside harbour (mean of SH1 to SH3) and outside harbour (mean of SH4 to SH5) for (a) dissolved ammonium, (b) UIA, (c) TON and (d) orthophosphate. Red lines represent surface and blue lines represent bottom.

UIA (Fig. 4.10b). Concentrations of UIA again followed a similar pattern to that of ammonium, exhibiting unprecedented surface elevation (+5 μg l^{-1}) outside the harbour in May 1994.

TON (Fig. 4.10c). Surface and bottom TON concentrations were again found to vary inversely with the differences in salinity between inside and outside the harbour (Section *3.2.5*). Surface concentrations exhibited no clear trends until July 1994, when the greatest outside enhancement (+280 μg l^{-1}) was observed and continued thereafter. Bottom concentrations inside the harbour were greater than or equal to outside, except in January 1993 when the bottom salinity was particularly enhanced inside the harbour due to high riverflow outside and in May 1994 during intense primary productivity inside the harbour.

Orthophosphate (Fig. 4.10d). Concentrations of orthophosphate showed greater differences between inside and outside the harbour than at springs, particularly at the surface where concentrations were elevated outside (+50 μg l^{-1}) in May 1992 whilst the low concentrations from the intense bloom at springs prevailed inside the harbour and where concentrations were enhanced inside (+50 μg l^{-1}) in January 1993 in response to lower salinity. In November 1993 surface and bottom concentrations were augmented outside the harbour, whilst in January 1994 surface and bottom concentrations were greater inside the harbour. These differences were entirely consistent with the salinity enhancements exhibited during these periods. No spatial differences were observed after impoundment until November 1994 when concentrations inside the harbour were greater (+70 μg l^{-1}) at the surface and less (-70 μg l^{-1}) at the bottom than outside, consistent with the salinity variations and highlighting the stronger correlation exhibited between orthophosphate and salinity at neaps than at springs.

4.2 Modelling the Plym Inputs

The study has highlighted the influence of the Plym Estuary on the water quality in Sutton Harbour. A numerical model of the major contaminants transported by the Plym could be useful in a predictive capacity to optimise the efficiency of harbour flushing events. A model of salinity and ammonium concentration in the Plym has been devised using the NERC Estuarine Contaminant Simulator (ECoS) developed by workers at Plymouth Marine Laboratories.

4.2.1 Overview of ECoS

ECoS is a modelling shell that provides a user-defined representation of an idealised estuary (Harris *et al.*, 1993). The simulations are one-dimensional (axial), such that the resulting variables are averaged over a cross-section. A maximum of five contaminants can be modelled simultaneously through advection and dispersion by tidal movements and riverine flows, partitioning between dissolved or permanently suspended phases and bed exchangeable or sediment phases, and atmospheric exchange and degradation processes. Simulations may be tidally-averaged or tidal in nature. The example presented here is a tidally-averaged simulation of salinity and ammonium concentration under differing fresh water runoffs and ammonium inputs measured in 1992. Ammonium is assumed to behave conservatively in the Plym Estuary, as observed by Millward (1993).

4.2.2 Model inputs and refinements

The ECoS shell requires data on the physical characteristics of the estuary to be simulated and data on the contaminant inputs to be modelled. The main physical inputs to the Plym model are shown in Fig. 4.11. They comprise estuary length, cross-sectional area in each of 27 segments from the head to the mouth of the estuary, the aspect ratio and total freshwater input. The aspect ratio is the ratio of the width at MSL to the depth relative to MSL that enables conversion of water-column variables to bed variables and calculation of the air-water transfer interface area. Plym flow data were unavailable for 1992, so total freshwater inputs were calculated from 1992 Tavy flows, using the algorithm provided by correlation of Plym and Tavy flows between 1976 and 1978.

Other physical inputs to the model were mean spring and neap tidal ranges and the axial water dispersion coefficient, K_w, defined as:

$$K_w = U * S/(dS/dX) \qquad \text{Eq. 4.1.}$$

where U (m s^{-1}) is the net seaward velocity of fresh water, S (x 10^{-3}) is salinity and X (m) is axial distance along the estuary (Harris *et al.*, 1993). A value can be obtained for U by dividing the total freshwater input above a point in the estuary by the cross-sectional area at that point for each segment of the estuary, thus enabling the calculation of K_w given sufficient axial salinity observations. However, it is often preferable to obtain an estimate of K_w and then to vary this until a best fit between computed and observed salinities is achieved.

The contaminant input data used in the Plym model are given in Table 4.4. It is evident that the three fluxes of ammonium were relatively constant between March and July and it is important to note that these were consented discharges that were being met by the operators. Therefore the mean fluxes were valid for use in the Plym model.

Once the various physical and chemical parameters had been established, sensitivity tests were conducted on the model (Fig. 4.12) to determine the value of K_w that best fitted the estuary and the effect of varying river flows. It is evident from Fig. 4.12a that K_w has a great effect upon contaminant concentration associated with the fresh water inputs but a

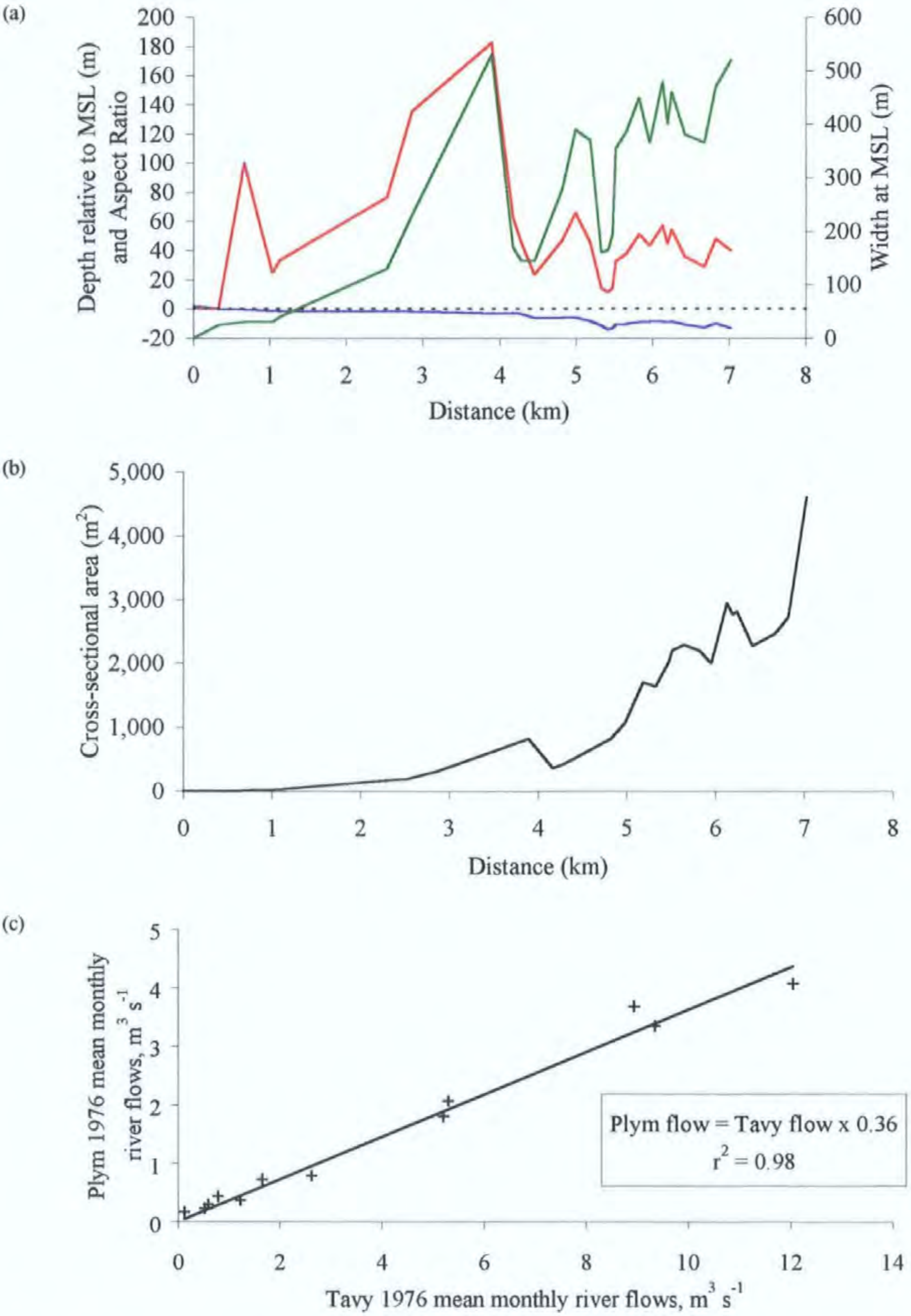


Fig. 4.11. Physical characteristic inputs to the ECoS Plym Estuary model: (a) depth relative to mean sea level (blue line), Aspect Ratio (red line) and width (green line) at mean sea level (black dashed line); (b) cross-sectional area; and (c) correlation of mean monthly Plym and Tavy river flows between 1976 and 1978 (1976 shown) to facilitate use of 1992 Tavy flow data for model.

Table 4.4. Fluxes of ammonium into the Plym used in ECoS simulations - 1991 NRA data from Millward (1993). Discharge distances from the weir.

Month/range	Plym River flow ($m^3 s^{-1}$)	Ammonium fluxes ($kg d^{-1}$) Marsh Mills STW (0.22 km)	Radford STW (4.33 km)	Chelson Meadow (2.8 km)
July/minimum	2.2 (0.40)	500		
July/mean	5.0 (1.50)	550	137	192
July/maximum	8.6 (2.48)	600		
March/minimum	1.5 (0.85)	400		
March/mean	1.5 (4.50)	550	106	198
March/maximum	2.0 (5.85)	700		

slight effect upon salinity distribution. This is as expected in an estuary with low river flows. The value of K_w chosen for the current simulation was 50 $m^2 s^{-1}$ on the criteria of best fit with observed salinity. The effect of varying river flow between 1.5 $m^3 s^{-1}$ and 4.5 $m^3 s^{-1}$ can be seen in Fig. 4.12b. In this case the greater effect is upon salinity, with lower salt water intrusion in the upper estuary under higher river flows. However, it is interesting to note the slight effect upon ammonium concentrations. The consented daily flux of ammonium from Marsh Mills (near the head of the estuary; Fig. 1.4) is almost constant. Under higher river flow, the ammonium concentrations are initially diluted by the fresh water, but further down estuary the concentrations are sustained by the Chelson Meadow and Radford inputs (Fig. 1.4).

The effects of minima and maxima in river flow upon salinity and ammonium concentration are better illustrated in Fig 4.13a and b, where all ammonium fluxes are constant and maximum and minimum river flow data are applied for March and July respectively. In these tests it is evident that ammonium concentrations are almost unchanged and are therefore shown to be dissociated from riverine/estuarine interactions, controlled instead by simple axial dispersion.

4.2.3 The ECoS Plym model as a management tool

The model was set up using mean river flows and mean ammonium fluxes for March and July so that it could be calibrated against observations made in the Plym Estuary on 09/03/92 and 27/07/92 (Millward, 1993) as shown in Tables 4.5 and 4.6, respectively. The observations were made throughout the available water column over a period of 7 to 8 h, such that the resulting mean values are essentially depth- and tidally-averaged.

Comparison between the model output and the observations is shown in Fig. 4.14. It is evident that the model gives good agreement with ammonium concentrations, confirming that ammonium does indeed appear to behave conservatively in the Plym. The discrete

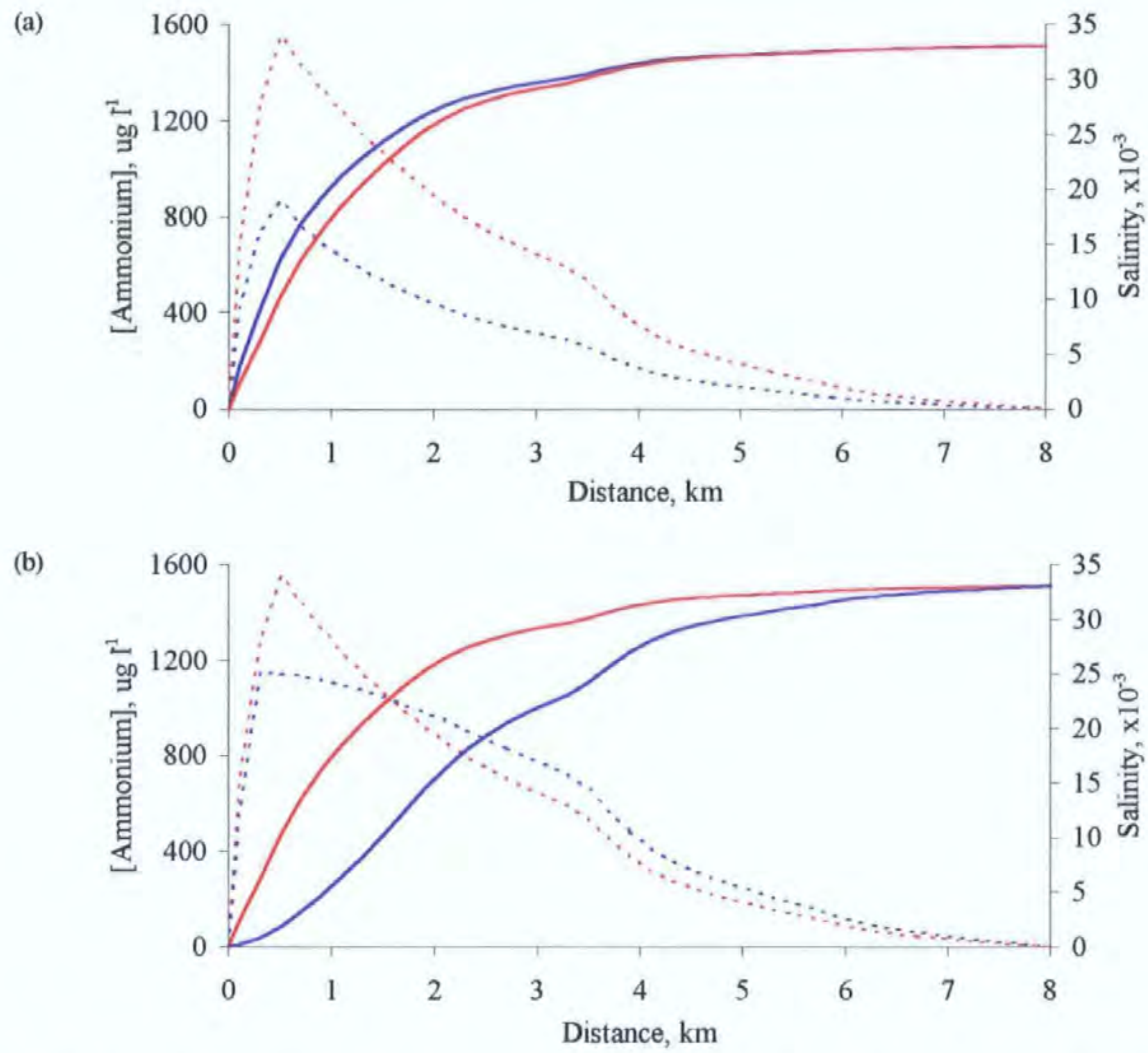
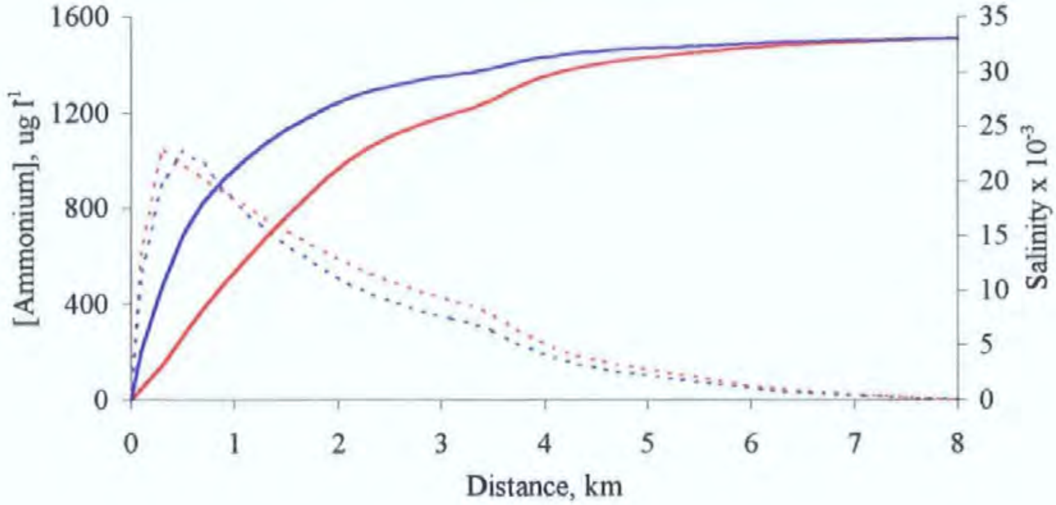


Fig. 4.12. Sensitivity tests of computed ammonium concentrations (dashed lines) and salinity (solid lines) in the Plym Estuary *versus* distance from the weir at Marsh Mills under (a) varying water dispersion coefficients between $K_w = 50$ m^2 s^{-1} (blue lines) and $K_w = 25$ m^2 s^{-1} (red lines) with constant river flow ($R = 1.5$ m^3 s^{-1}), and (b) varying river flow between $R = 4.5$ m^3 s^{-1} (blue lines) and $R = 1.5$ m^3 s^{-1} (red lines) under constant water dispersion coefficients ($K_w = 25$ m^2 s^{-1}).

kink in the ammonium curve between 3 and 4 km in March 1992 and July 1992 is caused by the additional input at Chelson Meadow and the observations confirm this in each case.

Departure of the curves from the elevated concentrations observed in the Cattewater is a consequence of the additional ammonium sources in the Cattewater and at Fishers Nose and West Hoe that were not included in the model. Moreover, ECoS assumes that the estuary is well mixed throughout its length, whereas the salinity observations suggest otherwise and therefore the ammonium fluxes introduced as buoyant plumes may be enhanced in the surface waters. The agreement of ammonium concentrations between model and observations was obtained despite lesser agreement between simulated and observed salinity values, particularly in the upper estuary, highlighting the separation of ammonium concentrations from riverine inputs.

(a)



(b)

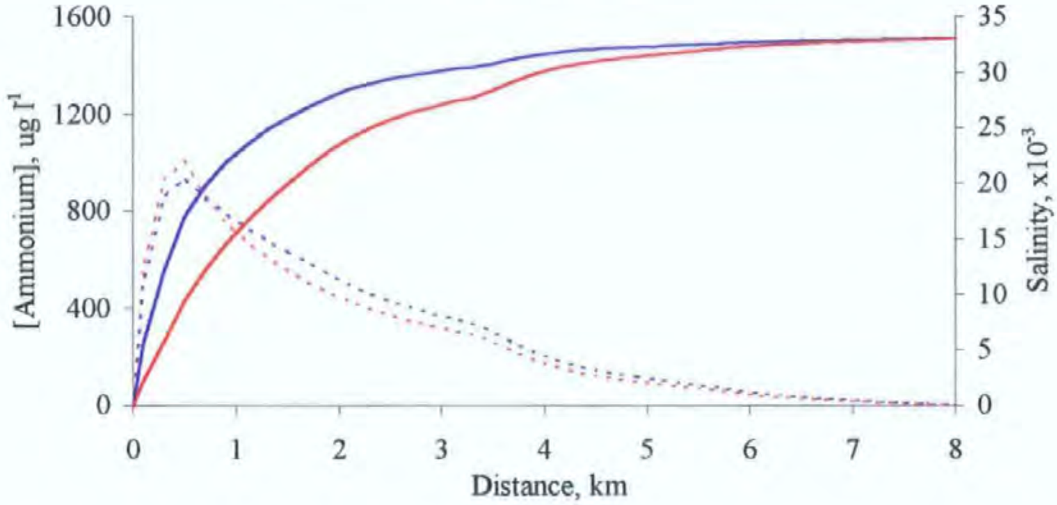


Fig. 4.13. Computed ammonium concentrations (dashed lines) and salinity (solid lines) in the Plym Estuary *versus* distance from the weir at Marsh Mills in (a) March 1992 and (b) July 1992 under minimum (blue lines) and maximum (red lines) river flow conditions. Minimum river flows 0.85 $m^3 s^{-1}$ (March 1992) and 0.4 $m^3 s^{-1}$ (July 1992); maximum river flows 4.5 $m^3 s^{-1}$ (March 1992) and 2.5 $m^3 s^{-1}$ (July 1992). $K_w = 50 m^2 s^{-1}$.

The current modelling effort has shown the great potential for this type of simulation to provide tidally-averaged predictions of contaminant concentration in the near-field waters exchanging with Sutton Harbour under differing river flows. Further work would involve the refinement of K_w for the estuary using measured Plym river flows and axial salinity observations and the inclusion of the additional ammonium sources in the lower estuary. The model would be made tidal by the inclusion of the subroutine TIDECOS (Harris *et al.*, 1993) based on the simplified harmonic method of tidal prediction. Moreover, the main thrust of further work would involve the modelling of Sutton Harbour as an embayment on the side of the estuary with water and contaminant exchange driven by the variations outside the harbour. This facility is provided by the new ECoS version 3β (Harris, pers. comm.), but was beyond the scope and time constraints of the present study.

Table 4.5. Data from Millward (1993) used in comparison with ECoS output for 09/03/92.

Survey times	Depth (m)	Saltram House (1.14 km from weir)		Laira Bridge (3.22 km from weir)		Cattewater (5.2 km from weir)	
		Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)	Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)	Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)
06:08	12.0					33.2	90
06:10	0.1					31.4	259
06:15	6.0					33.4	93
06:35	0.0			32.4	159		
06:48	0.0	29.0	700				
08:45	3.0	31.4	216				
08:54	0.0	28.6	660				
09:08	4.5			32.8	125		
09:15	2.5			32.6	133		
09:20	0.0			32.4	674		
09:40	12.0					34.1	84
09:45	6.0					33.9	77
09:50	0.1					32.8	140
11:35	0.0	15.2	1210				
11:50	0.0			27.5	497		
11:55	2.5			30.6	282		
12:05	11.0					33.1	140
12:10	0.1					31.4	213
12:20	5.5					32.6	200
14:25	0.0			11.9	1240		
14:35	9.0					33.1	174
14:40	4.5					32.0	265
14:55	0.1					27.3	484
Mean		26.1	697	28.6	444	32.4	185
SD		7.3	407	7.6	407	1.8	115

Table 4.6. Data from Millward (1993) used in comparison with ECoS output for 27/07/92.

Survey times	Depth (m)	Saltram House (1.14 km from weir)		Laira Bridge (3.22 km from weir)		Cattewater (5.2 km from weir)	
		Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)	Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)	Salinity ($x10^{-3}$)	NH_4^+ ($\mu g\ l^{-1}$)
07:15	0.0	13.8	2018				
07:30	0.0			31.8	504		
07:32	2.0			33.0	315		
07:46	10.0					33.3	189
07:48	0.1					29.7	1019
07:50	5.0					33.1	331
13:15	0.0	31.9	413				
13:24	2.8			33.6	201		
13:26	0.0			31.4	>500		
13:35	11.5					33.9	207
13:37	0.1					31.4	617
13:40	6.0					33.6	229
16:33	0.0	32.4	415				
16:35	2.8	33.0	283				
16:45	3.9			33.9	119		
16:47	0.0			31.4	418		
16:50	2.0			33.6	186		
17:00	12.0					34.2	114
17:02	0.1					33.1	258
17:05	6.0					34.2	124
Mean		27.8	782	32.8	320	32.9	343
SD		9.3	826	1.0	157	1.5	294

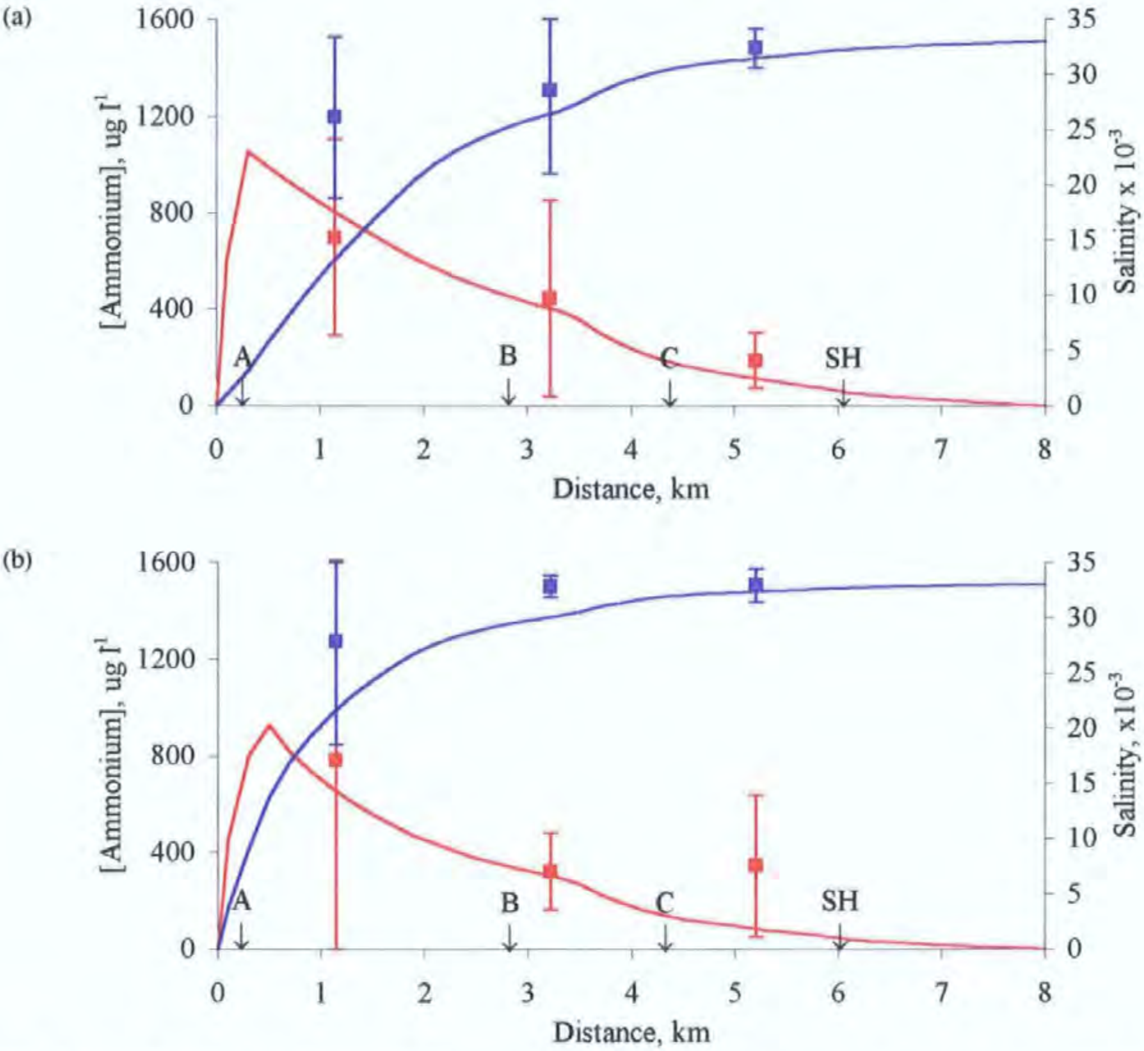


Fig. 4.14. Salinity (blue lines) and ammonium concentrations (red lines) in the Plym Estuary *versus* distance from the weir at Marsh Mills in (a) March 1992 ($R = 4.5\ m^3\ s^{-1}$) and (b) July 1992 ($R = 1.5\ m^3\ s^{-1}$). Solid lines represent values computed with ECoS, square symbols represent mean of in situ measurements with error bars indicating one standard deviation. Points highlighted on x axis represent inputs from Marsh Mills STW (A), Chelson Meadow (B) and Radford STW (C) (refer to Fig. 1.4) and the position of Sutton Harbour (SH). $K_w = 50\ m^2\ s^{-1}$.

4.3 Phytoplankton and Dissolved Oxygen

Studies have shown that the growth of phytoplankton is primarily controlled by seasonal variations of insolation, water temperature and the availability of dissolved nutrients (Tanaka *et al.*, 1981). In turn, phytoplanktonic activity controls the concentrations of nutrients and the dissolved oxygen saturation of the water column, whilst phytoplankton decay exerts an oxygen demand and returns inorganic forms of N and P to the water. Periods of high biological activity were therefore expected to show high variability in dissolved oxygen and chlorophyll *a* concentrations, whilst periods of diminished biological activity were expected to show low dissolved oxygen and chlorophyll *a* concentrations. It was hypothesised that after impoundment the combination of warmer water, increased transparency and increased nutrient availability would stimulate productivity in the phytoplankton and extend the duration of the spring and summer blooms. It was further postulated that the increased retention of decaying phytoplanktonic material would cause increasing anoxia in the water column and sediments, conditions commonly observed elsewhere in the marine environment (Desprez *et al.*, 1992; Rosenberg *et al.*, 1992).

4.3.1 Semi-diurnal and diurnal variability

The greatest short-term variations in dissolved oxygen saturation and chlorophyll *a* concentration were observed during periods of high primary productivity. Dissolved oxygen was mainly subject to diurnal variations in response to the balance of photosynthesis and respiration of the phytoplankton, such that minima were observed in the early morning and maxima were observed in the mid- to late afternoon. In contrast, chlorophyll *a* concentrations were mainly subject to semi-diurnal variations associated with tidal mixing and dilution, since the growth rate was not sufficiently rapid to show diurnal differences between HW and LW. These observations can be qualified with reference to Figs. 4.15a to d, which show profiles of dissolved oxygen and chlorophyll *a* inside and outside the harbour during neap tides before and after impoundment.

July 1993 (Figs. 4.15a and b). The water column was supersaturated with dissolved oxygen throughout the day inside the harbour as a result of the intense phytoplankton bloom and high insolation. The saturation of the surface waters increased from over 110 % at LW (0801) to over 140 % at HW (1422), whilst the saturation of the bottom waters remained at *circa* 110 %. Similar conditions were observed outside the harbour although surface oxygen saturations were lower (125 %) at HW. Any variation in the high chlorophyll *a* concentrations between LW and HW was undetectable inside or outside the harbour because all concentrations exceeded the fluorometer operational limit of 11 $\mu g\ l^{-1}$.

July 1994 (Figs. 4.15c and d). The dissolved oxygen saturations increased throughout the water column inside the harbour from *circa* 100 % at LW (0849) to *circa* 110 % at HW (1518) despite a decrease in chlorophyll *a* concentrations over the same period of up to 1 $\mu g\ l^{-1}$ throughout the water column. Outside the harbour, dissolved oxygen concentrations also increased from *circa* 100 % at LW to *circa* 110 % at HW throughout the water column, whilst chlorophyll *a* concentrations, considerably lower than those inside the harbour throughout the day, increased by 1 $\mu g\ l^{-1}$ at the surface and decreased by 0.5 $\mu g\ l^{-1}$ in mid-water. It would appear that water lower in chlorophyll *a* content flooded into the harbour from HW-3h to HW, producing the observed decrease in concentration.

Lesser short-term variations in dissolved oxygen saturation and chlorophyll *a* concentration were observed during winter periods of low primary productivity. With primary productivity and chlorophyll *a* concentrations reduced to a seasonal minimum, variations in dissolved oxygen were more sensitive to the semi-diurnal processes of tidal mixing and dilution and reflected more the oxygen demand of decaying organic matter.

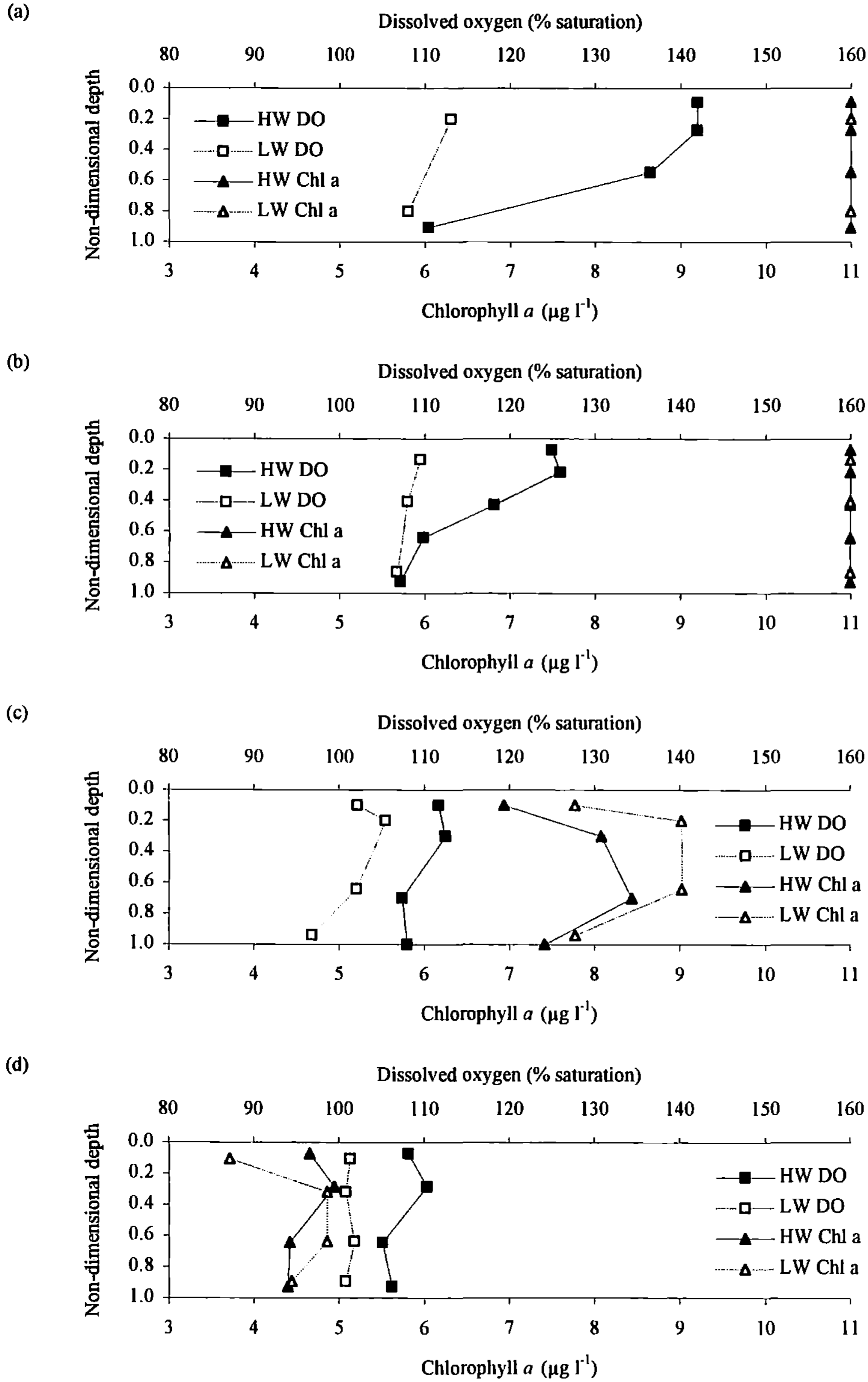


Fig. 4.15. Profiles of dissolved oxygen and chlorophyll *a* at neaps (a) inside harbour (SH2) and (b) outside harbour (SH4) in July 1993 and (c) inside harbour (SH2) and (d) outside harbour (SH4) in July 1994.

Profiles of dissolved oxygen and chlorophyll *a* in Fig. 4.16 represent conditions inside and outside the harbour at neap tides in November 1993 and November 1994:

November 1993 (Figs. 4.16a and b). Dissolved oxygen saturations inside the harbour were almost identical to those outside the harbour with constant surface values between HW and LW of *circa* 83 %, LW bottom readings of *circa* 78 % and HW bottom saturations of *circa* 85 %. Chlorophyll *a* concentrations lay at a seasonal minimum of 0.5 to 1.0 $\mu g\ l^{-1}$ throughout except for a slight rise outside the harbour to 2 $\mu g\ l^{-1}$ at LW.

November 1994 (Figs. 4.16c and d). Chlorophyll *a* concentrations again lay at a seasonal minimum of 0.5 $\mu g\ l^{-1}$ throughout the tidal cycle both inside and outside the harbour. Surface dissolved oxygen saturations were constant between HW and LW outside, at *circa* 77 %, whilst bottom readings varied from 79 % at HW to 72 % at LW. Dissolved oxygen saturations were considerably lower inside the harbour. Surface saturations varied little from 69 % at HW to 68 % at LW, whilst bottom values varied slightly from 72 % at HW to 67 % at LW. These observations indicate limited dilution from HW-3h to HW of the oxygen-deficient waters inside the harbour with the slightly more oxygenated waters outside. The deep oxygen sag observed particularly at LW bottom (i.e. mid-impoundment) suggests that a significant amount of decaying epibenthic organic matter was present. Low BOD_5 values of 0.4 $mg\ l^{-1}$ at LW bottom indicate that the oxygen demand was not being generated in the water column.

4.3.2 Spring-neap tide variability

One of the most surprising observations was the extent to which primary productivity changed on weekly or spring-neap tidal timescales. The driving forces for these variations were numerous and included changes in insolation and nutrient availability. Such variations are examined with reference to the time series in Fig. 4.17 in which spring and neap tidal surface observations are plotted in sequence for each of the biologically active seasons (defined as May to September) of 1992, 1993 and 1994. Bottom observations were not significantly different from those at the surface and are therefore omitted. The greatest variations of dissolved oxygen and chlorophyll *a* (and hence of BOD_5) both inside and outside the harbour were observed between spring and neap tides in May 1992. An intense phytoplankton bloom was manifest at springs, with tidally-averaged chlorophyll *a* concentrations of >10 $\mu g\ l^{-1}$ inside, and >9 $\mu g\ l^{-1}$ outside the harbour. BOD_5 values of >3 $mg\ l^{-1}$ were also indicative of a considerable phytoplankton biomass. Intense insolation during the survey produced high photosynthetic rates and oxygen supersaturations of *circa* 130 %. One week later at neaps under similarly intense insolation, the bloom had decayed to winter background concentrations of 1 $\mu g\ l^{-1}$ throughout the harbour system. The water column was undersaturated (*circa* 80%) owing to organic matter decomposition and BOD_5

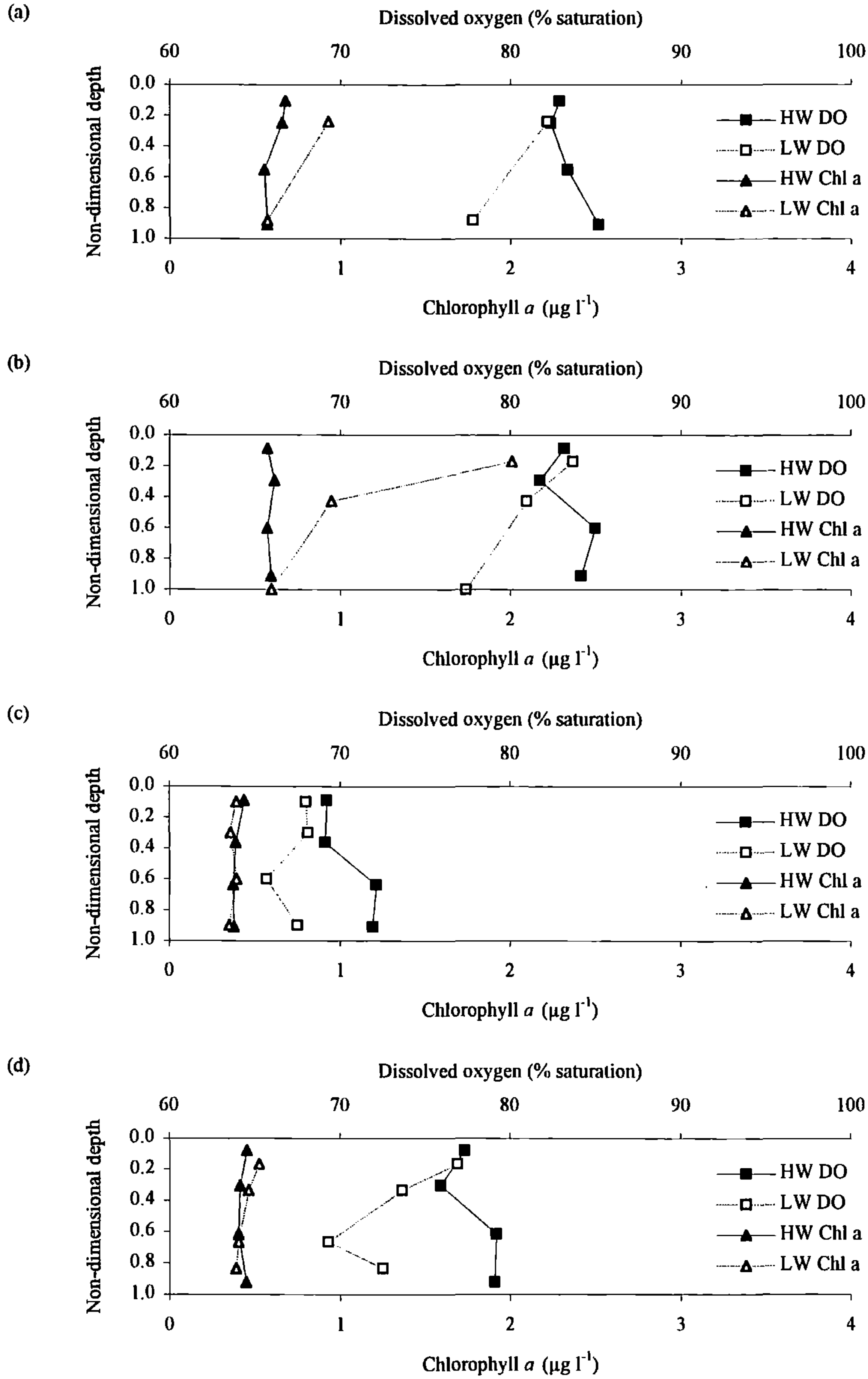


Fig. 4.16. Profiles of dissolved oxygen and chlorophyll *a* at neaps (a) inside harbour (SH2) and (b) outside harbour (SH4) in Nov. 1993 and (c) inside harbour (SH2) and (d) outside harbour (SH4) in Nov. 1994.

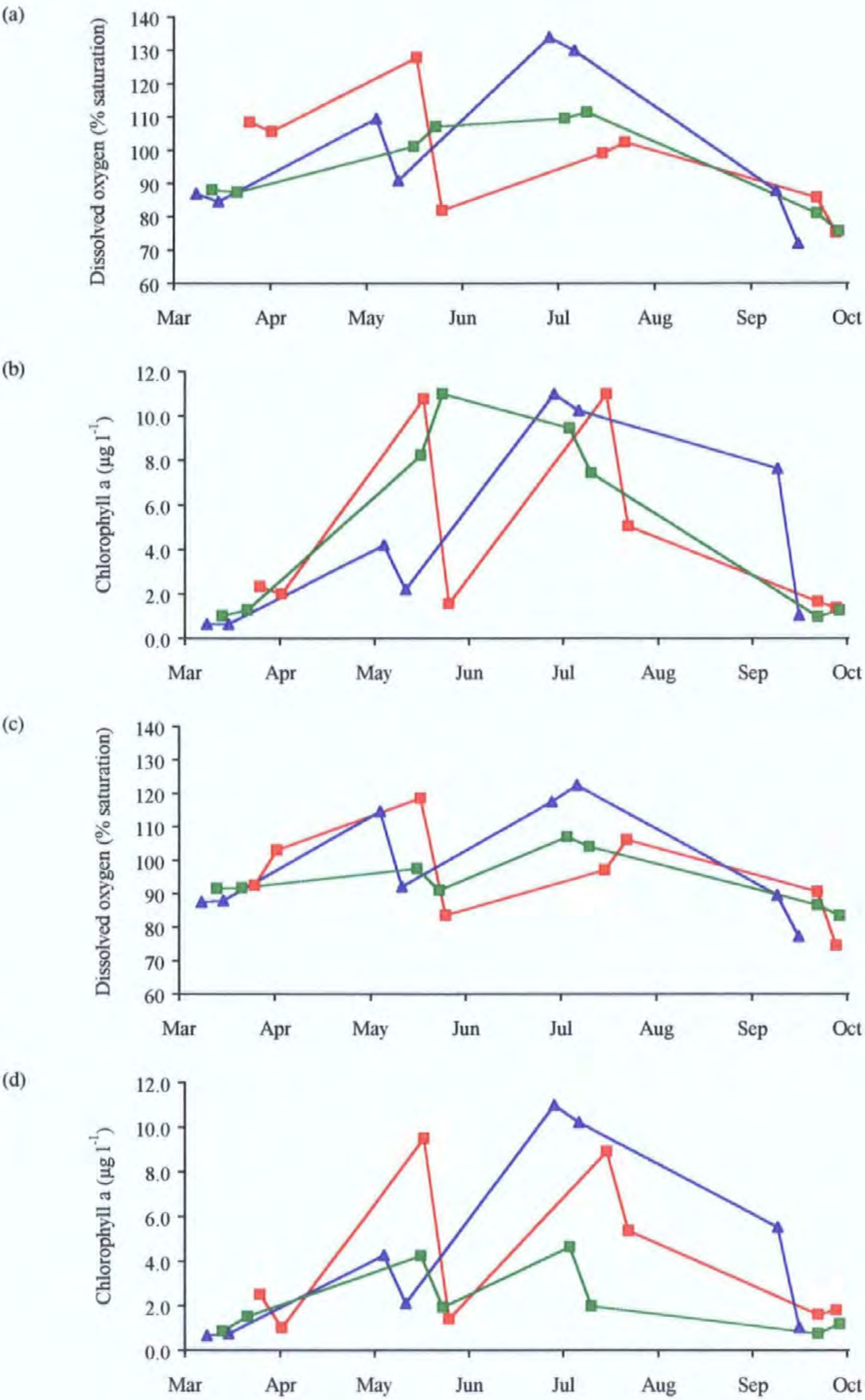


Fig. 4.17. Time series of (a) dissolved oxygen and (b) chlorophyll *a* inside harbour and (c) dissolved oxygen and (d) chlorophyll *a* outside harbour in 1992 (red lines), 1993 (blue lines) and 1994 (green lines).

values of *circa* 0.5 mg l^{-1} indicated that this process was almost complete. It would appear that low nutrient concentrations at springs were biolimiting as a consequence of negligible nutrient supply from outside the harbour, so that the rate of growth of the bloom was unsustainable.

Slightly different conditions were observed later during the same biologically active season, illustrating the variability resulting from changing insolation. At springs in July 1992, chlorophyll *a* concentrations were high inside (11 μg l^{-1}) and outside the harbour (9 μg l^{-1}), although BOD_5 values of >2 mg l^{-1} suggested that the phytoplankton biomass was not as great as in May 1992. The water column was marginally undersaturated with oxygen throughout due to only 0.5 h of direct insolation during the day. One week later at neaps, chlorophyll *a* concentrations had decreased to 5 μg l^{-1} and BOD_5 values had decreased to *circa* 1.5 mg l^{-1} inside and out, despite similar precipitation to springs (15 mm in each preceding 4 d period) and higher nutrient concentrations than in May 1992. However, 4 h of direct insolation was received during the day resulting in a marginal supersaturation of oxygen (*circa* 105 %).

The next biologically active season (May 1993 to September 1993) showed that the processes causing the previous observations of bloom at springs and decay at neaps were detached from the tidal cycle. The bloom in May 1993 was minimal, although the pattern of higher chlorophyll *a* concentrations (4 μg l^{-1}) and oxygen supersaturation (110 %) at springs followed by lower chlorophyll *a* concentrations (2 μg l^{-1}) and oxygen undersaturation (110 %) at neaps was repeated. A more intense bloom was observed in July 1993 and on this occasion the peak in chlorophyll *a* concentration was seen at neaps inside and out (>11 μg l^{-1}). The biomass as suggested by the BOD_5 values was greater inside (2.3 mg l^{-1}) than outside (1.7 mg l^{-1}) and oxygen supersaturations from 10 h of direct insolation were 140 % inside and 120 % outside. One week later at springs, chlorophyll *a* concentrations had declined moderately throughout to 10 μg l^{-1}. The phytoplankton biomass inside the harbour appeared to have declined slightly (BOD_5 2.0 mg l^{-1}) whilst remaining the same outside. Oxygen supersaturations under enhanced direct insolation (13 h) were 130 % inside the harbour and 120 % outside the harbour, apparently balanced by the slight reduction in chlorophyll *a* concentration.

The strongest evidence for the non-involvement of the spring-neap tidal cycle came in September 1993. A moderate bloom was seen at neaps inside (chlorophyll *a* 7 μg l^{-1}; BOD_5 1.7 mg l^{-1}) and outside the harbour (chlorophyll *a* 5 μg l^{-1}; BOD_5 1.2 mg l^{-1}), although the water column was undersaturated with oxygen throughout (*circa* 88 %) due to low insolation. One week later at springs, the bloom had almost completely disappeared inside and outside the harbour (chlorophyll *a* 0.8 μg l^{-1}; BOD_5 0.1 mg l^{-1}) and oxygen sags had developed inside (70 %) and outside the harbour (75 %) due to combustion of the phytoplankton biomass. The decay of the bloom occurred despite a marked reduction in salinity indicating an increased freshwater influx, which allowed the introduction of a fresh supply of dissolved nutrients to the system. It was therefore attributed to decreasing solar intensity at the end of summer.

Overall short-term variability within the harbour appeared to be reduced after impoundment, whilst differences between conditions inside and outside increased. In May 1994, at neaps, a moderate bloom was observed inside the harbour (chlorophyll *a* 8 $\mu g\ l^{-1}$; BOD_5 1.7 $mg\ l^{-1}$) and outside (chlorophyll *a* 4 $\mu g\ l^{-1}$; BOD_5 1.7 $mg\ l^{-1}$). Despite an absence of any direct insolation, respective oxygen saturations reached 100 % and 98 %. One week later at springs, the bloom inside the harbour had intensified (chlorophyll *a* >11 $\mu g\ l^{-1}$; BOD_5 1.6 $mg\ l^{-1}$) and dissolved oxygen saturations had increased to 110 % despite the continued absence of any direct insolation. However, the moderate bloom outside the harbour had declined considerably(chlorophyll *a* 2 $\mu g\ l^{-1}$; BOD_5 1.2 $mg\ l^{-1}$) and dissolved oxygen saturation had fallen to 90 %. This constituted a marked departure from the established patterns of similarity of conditions inside and outside which had been observed during the previous two biologically active seasons. Clearly the impoundment had had an effect upon water quality inside the harbour, supporting a phytoplankton bloom that was not sustainable in the waters outside the harbour at the same time. Conditions for bloom were no more favourable outside the harbour at neaps in July 1994 (chlorophyll *a* 4 $\mu g\ l^{-1}$; BOD_5 1.5 $mg\ l^{-1}$). However, more intense insolation produced dissolved oxygen supersaturation of 110 % throughout the system, despite existence of a more intense bloom (chlorophyll *a* 9.5 $\mu g\ l^{-1}$; BOD_5 1.6 $mg\ l^{-1}$) inside the harbour. One week later at springs, chlorophyll *a* concentration inside the harbour had partially declined (7.5 $\mu g\ l^{-1}$), although the BOD_5 value had increased (1.8 $mg\ l^{-1}$) and the dissolved oxygen supersaturation was unchanged (110 %). The bloom inside the harbour therefore showed little evidence of the rapid decay observed in May 1992, July 1992 or September 1993, whilst outside the moderate bloom had almost disappeared (chlorophyll *a* <2 $\mu g\ l^{-1}$; BOD_5 1.4 $mg\ l^{-1}$; dissolved oxygen 100 %). It would therefore appear that the harbour waters were now able to sustain a phytoplankton bloom for a longer period after impoundment, at a time when conditions did not favour growth outside. The implications of this higher productivity are that a greater mass of organic matter will be retained for a longer period in the water column and on the sediments, deepening the winter oxygen sag and benthic anoxia. A greater store of dissolved nutrients would also be provided for the subsequent growth season.

Owing to the nature of the variables discussed in this section, and the strong link with biological activity, the period between November and March each year exhibited little significant spring-neap variability. Chlorophyll *a* concentrations fell to background (below 2 $\mu g\ l^{-1}$) and dissolved oxygen saturations were reduced to the seasonal minimum, as discussed in the following section.

4.3.3 Seasonal and inter-annual variability

The seasonal variability of phytoplankton primary production in the harbour was coupled to the regular cycle of seasonal variations in solar input. However, this seasonality was modified by the availability of dissolved nutrients and by fluctuations in the amount of direct insolation received by the harbour waters. These modifications gave rise to the inter-annual variability observed during the study. Furthermore, the seasonality was perturbed by the impoundment of the harbour and these perturbations will be further examined here. The relationship of chlorophyll *a* concentrations with dissolved oxygen saturation and biochemical oxygen demand (BOD_5) was investigated by correlation during the biologically active seasons (i.e. summer) and the quiescent seasons (i.e. winter). The results in Table 4.7 show that in summer, dissolved oxygen saturation exhibited strong significant correlations with chlorophyll *a* concentration throughout the water column at springs and neaps. In winter, strong significant correlations were exhibited at the surface but only a slight significant correlation was exhibited at the bottom at springs, with no significant correlation at neaps. The data therefore suggest that dissolved oxygen concentrations were controlled throughout the water column in summer by phytoplankton photosynthesis and the combustion of decaying organic matter. In winter dissolved oxygen concentrations were possibly controlled at the surface by the combustion of organic matter carried from the catchment area, whilst the combustion of organic matter at the sediment surface deposited during the summer period may have controlled the dissolved oxygen saturation of the bottom waters. Strong significant correlations were exhibited between BOD_5 value and chlorophyll *a* concentration throughout the study during springs and neaps at surface and bottom, suggesting that the decay of phytoplankton was the main component of the biochemical oxygen demand in the harbour waters and legitimising the coarse assumption that BOD_5 values can be used to represent the phytoplankton biomass.

Table 4.7. Correlation coefficients (r) of dissolved oxygen saturation and BOD_5 values with chlorophyll *a* concentrations inside harbour from November 1991 to November 1994. All data have been tidally-averaged between HW and LW and spatially averaged between SH1, SH2 and SH3. Summer defined as May to September, winter as November to March.

Correlation	Season	Springs		Neaps	
		Surface	Bottom	Surface	Bottom
Dissolved oxygen *versus* chlorophyll *a*	Summer (n = 9)	0.81^c	0.88^c	0.84^c	0.87^c
	Winter (n = 10)	0.73^b	0.61^a	0.73^b	NS
BOD_5 *versus* chlorophyll *a*	All data (n = 19)	0.67^c	0.69^c	0.79^c	0.68^c

NS - not significant at $p < 0.05$; $^a p < 0.05$; $^b p < 0.01$; $^c p < 0.005$

The seasonal variabilities of dissolved oxygen saturation and chlorophyll *a* concentration are shown in the time series in Figs. 4.18a to d. A distinct seasonality is immediately evident in chlorophyll *a* concentrations throughout the water column at springs (Fig. 4.18c) and neaps (Fig. 4.18d), with minima of <2 $\mu g\ l^{-1}$ exhibited from November to March and maxima in the range 4 to >11 $\mu g\ l^{-1}$ exhibited in May and July. September marked the transition between biologically active and quiescent periods and this transition was complete at the time of the September 1992 and September 1994 surveys. However, as discussed in Section *4.3.2*, a bloom was still in evidence at neaps in September 1993 that had decayed one week later at springs. Dissolved oxygen saturations also exhibited clear seasonality at springs (Fig. 4.18a). Before impoundment, the water column was generally undersaturated with oxygen between November and March and supersaturated in May and July. September again marked a transition period, with seasonal minima of 75 % at surface and bottom possibly corresponding to the peak in combustion of the organic matter from the preceding summer. The bottom water was undersaturated with oxygen (95 %) in May 1994 and July 1994. The characteristic oxygen sag of 80 % in September 1994 was deepened still further in November 1994 to the lowest oxygen saturations observed during the study (70 %). The seasonality in dissolved oxygen saturations was less distinct at neaps (Fig. 4.18b). Low oxygen saturations were again observed between November and March (except in March 1992) and high saturations were observed each July. The low saturation in May 1992 resulted from the aforementioned rapid decay of the bloom at springs. The bottom water was undersaturated with oxygen in May 1994 and July 1994 and the surface and bottom oxygen sags in September 1994 (70 %) and November 1994 (65 %) were again the deepest observed during the study. The deeper oxygen sags in November 1994 at springs and neaps support the hypothesis in Section *4.3.2* that a greater biomass of phytoplankton had been retained inside the harbour following impoundment.

The time series in Figs. 4.19a to d show the difference in chlorophyll *a* concentration and dissolved oxygen saturation between inside and outside the harbour for the surface waters and the bottom waters:

Chlorophyll a (Figs. 4.19b and d). Concentrations of chlorophyll *a* showed little difference at springs or neaps during the quiescent period between November and March. The main differences were slight increases inside the harbour during periods of high primary productivity. At springs these periods were May 1992 and July 1992 with an elevation of 1 to 2 $\mu g\ l^{-1}$ in both surface and bottom waters and May 1993 and July 1993 with an enhancement of 1.5 to 3.5 $\mu g\ l^{-1}$ in the bottom waters only. High productivity was observed at neaps in July 1993 and September 1993. There was no difference in chlorophyll *a* concentration between inside and outside the harbour in July 1993, whilst an enhancement of 1 $\mu g\ l^{-1}$ at the bottom and 2 $\mu g\ l^{-1}$ at the surface was observed in September 1993. Phytoplankton growth therefore had a tendency to be slightly greater

(a)

(b)

(c)

(d)

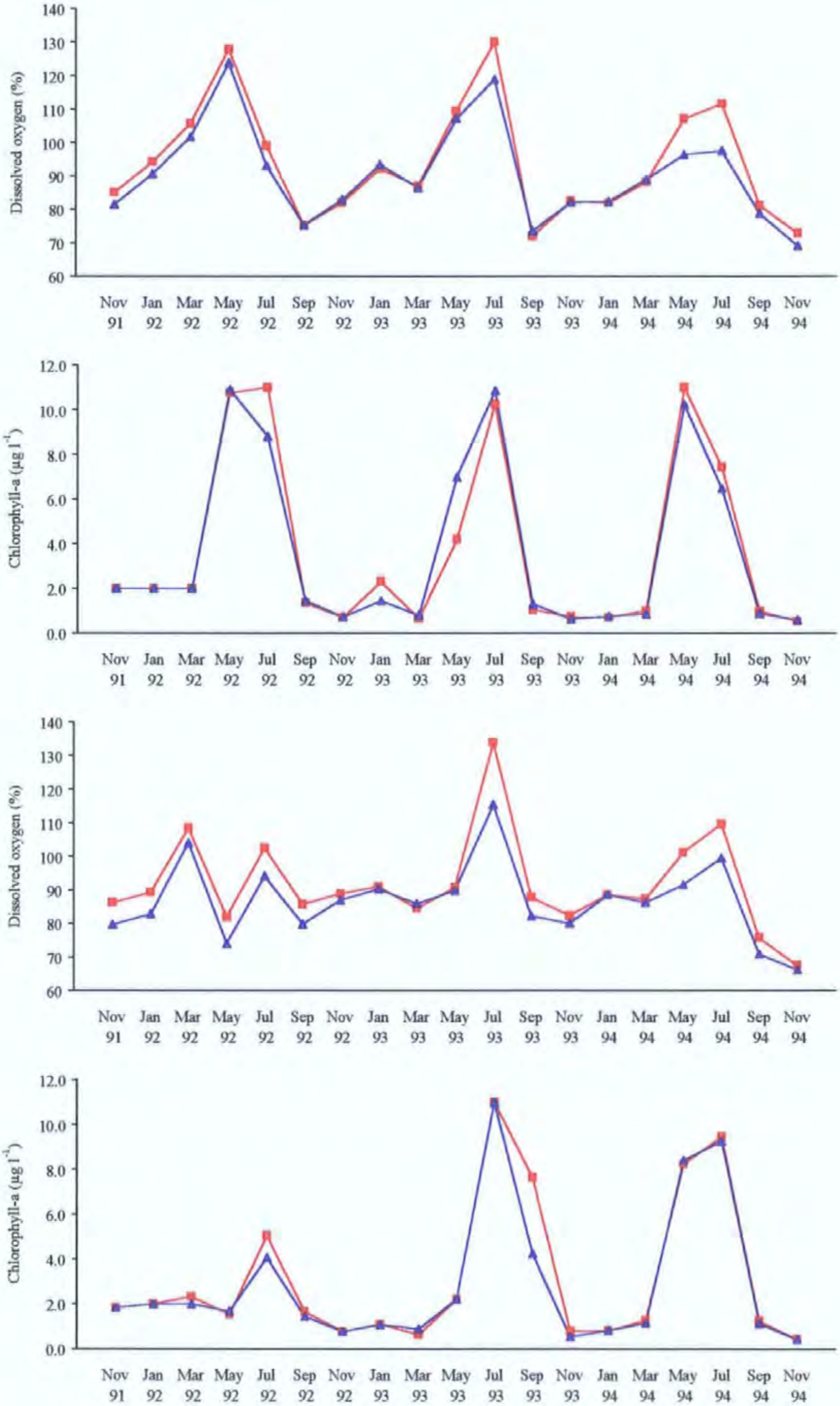


Fig. 4.18. Tidally-averaged time series of (a) dissolved oxygen saturation and (b) chlorophyll *a* concentration at springs and (c) dissolved oxygen saturation and (d) chlorophyll *a* concentration at neaps inside Sutton Harbour, from November 1991 to November 1994. Red lines represent surface samples and blue lines represent bottom samples. All values are mean of SH1 to SH3.

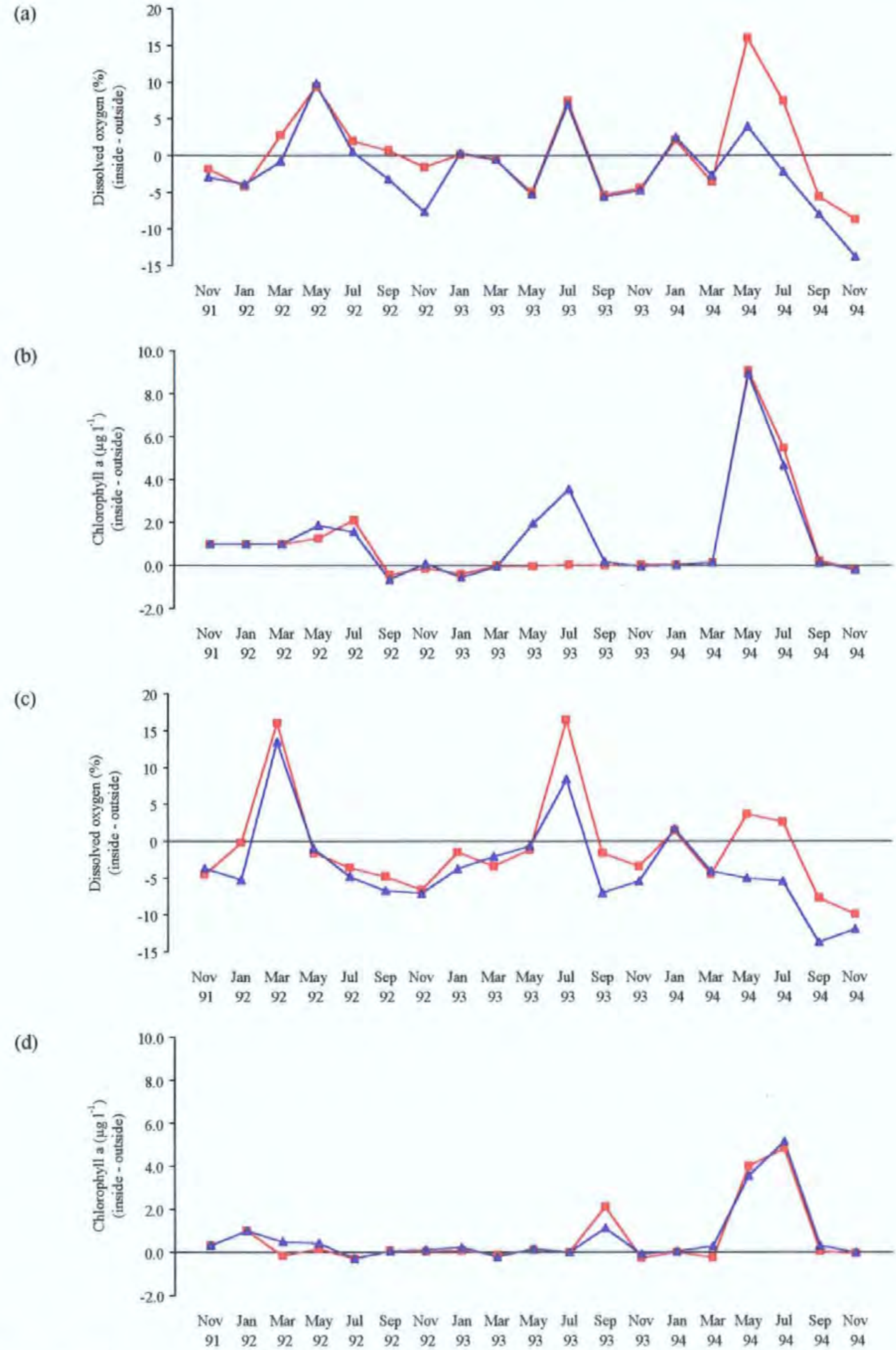


Fig. 4.19. Time series of differences between inside harbour (mean of SH1 to SH3) and outside harbour (mean of SH4 to SH5) for (a) dissolved oxygen saturation and (b) chlorophyll *a* concentration at springs and (c) dissolved oxygen saturation and (d) chlorophyll *a* concentration at neaps. Red lines represent surface water, blue lines represent bottom water.

inside the harbour than outside before impoundment, probably because of the shallower depth. Surface and bottom increases of 9 $\mu g\ l^{-1}$ at springs and 4 $\mu g\ l^{-1}$ at neaps were observed inside the harbour in May 1994, with surface and bottom elevations of 5 $\mu g\ l^{-1}$ at springs and neaps in July 1994. These enhancements were unprecedented during the study and gave clear evidence of an important effect of impoundment. It must be noted that the summer of 1994 was not unusually warm or sunny (Section *3.2.2*) and that increases in winter precipitation could potentially combine with greater summer insolation to produce eutrophic conditions within the harbour. Under these conditions, enhanced nutrient availability would promote unlimited phytoplankton growth and the subsequent oxygen sag when insolation decreased from September onwards could prove deleterious to the resident fauna. Furthermore, enhanced benthic anoxis from the deposition of organic matter on the sediment surface would lead to the remobilisation of toxic heavy metals and enhanced mineralisation reactions would increase the availability of dissolved nutrients for the next biologically active season. The possible development of these conditions is currently being monitored until July 1995 (Chapter 6) and if evident may require extensive flushing of the harbour during the summer in addition to that prescribed in Section *4.1.2* during winter.

Dissolved oxygen (Figs. 4.19a and c). Dissolved oxygen saturations followed a similar pattern to that of chlorophyll *a* concentrations, as would be anticipated from Table 4.7. The water column was more saturated with oxygen inside the harbour at springs only during the periods of high primary productivity in May 1992, July 1993 and May 1994 to July 1994. The greatest of these spatial differences was seen in May 1994, in concordance with chlorophyll *a* concentrations. At other times, the opposite condition generally prevailed except in January 1993 when no differences were observed and in January 1994. The greater oxygen saturation outside the harbour possibly resulted from more efficient renewal of the water in free connection with the Sound than inside the semi-enclosed harbour. At neaps (Fig. 4.19c) the water column was again only more saturated with oxygen inside the harbour during periods of high primary productivity in July 1993 (surface and bottom) and May 1994 to July 1994 (surface only). At other times, the oxygen saturation was generally higher outside the harbour except in March 1992 and January 1994. The greatest elevation in dissolved oxygen saturations outside over inside was observed at springs and neaps in November 1994, showing that the deep oxygen sag observed inside the harbour at this time was not a general feature.

The continuous monitoring trace of surface dissolved oxygen saturation in Fig. 4.20 further illustrates the seasonal variations in biological activity inside the harbour. The peaks in July 1993 and May 1994 corresponding to the intense blooms discussed in Section *4.3.2* can be clearly seen, together with an extended period of sub-oxic conditions from August 1993 until April 1994. The limited trace from the bottom water instrument suggests that oxygen saturations were generally even lower than at the surface.

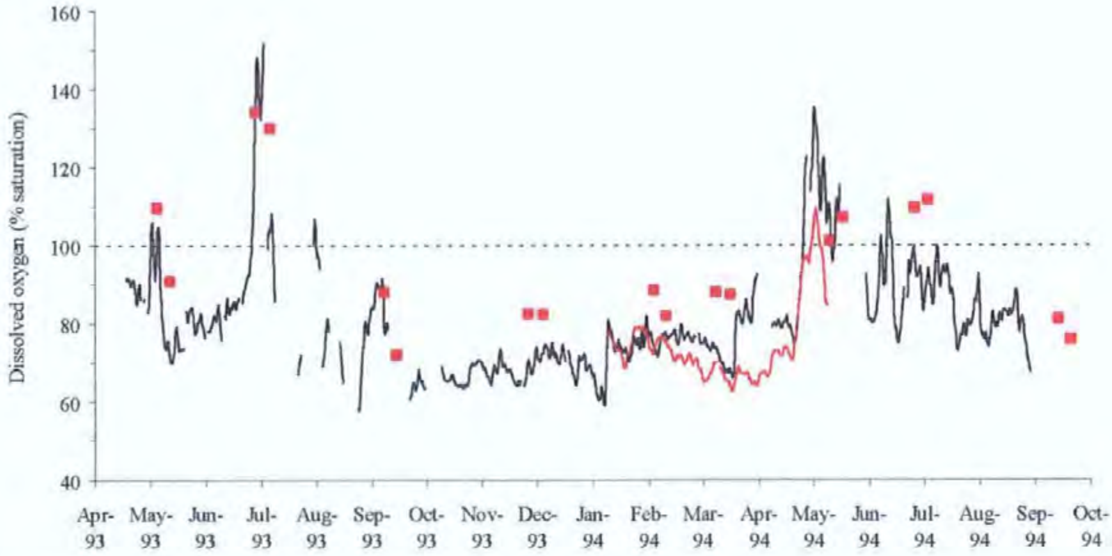


Fig. 4.20. Doodson-filtered dissolved oxygen data from continuous monitoring inside harbour at surface (black line) and bottom (red line) spanning the biologically active seasons of 1993 and 1994. Dashed line denotes 100 % saturation level. Red squares represent routine monitoring results (SH2 surface).

By implication, benthic anoxis during this period as a result of epibenthic deposits of decaying organic matter would have caused the release of reduced Fe, Mn and associated trace metals from the sediments and the subsequent enrichment in these elements of the overlying suspended matter (Dehairs *et al.*, 1989). The Fe and Mn oxy-hydroxide coatings thus formed on the SPM (Morris *et al.*, 1982) would scavenge trace metals from the dissolved phase, settle out and therefore accumulate in the sediments. The implications for impoundment are clearly that enhanced benthic anoxis and increased settling of SPM would lead to greater accumulation of trace metals in the sediments.

Other notable features were a small peak in dissolved oxygen saturation corresponding to a slight bloom in early May 1993, an oxygen sag of *circa* 80 % prevailing from mid May 1993 to late June 1993, the high variability with periodic undersaturation and supersaturation exhibited between July 1993 and August 1993 and between June 1994 and August 1994. These features were probably caused as lesser phytoplankton blooms cycled through periods of growth and decay during the mid-to-late summer periods.

4.3.4 Discrete monitoring of post-impoundment phytoplankton bloom

The phytoplankton bloom observed immediately after impoundment during the routine surveys in May 1994 was closely monitored throughout June 1994, so that any immediate and exceptional water quality perturbations could be detected and the extent and duration of the bloom characterised. Unfortunately, the pontoon at site 6 (Fig. 2.6) was damaged by a malfunction in lock operation on 24/05/94 and had to be removed along with the continuous water monitor. By the time this problem had been discovered and a replacement instrument and housing installed, two weeks had been lost from the data set.

However, the monitoring exercise was still valuable; the results are presented in Fig. 4.21 as Doodson-filtered time series of dissolved oxygen and salinity and discrete measurements of TON, orthophosphate and chlorophyll *a* concentration. The results are summarised in Table 4.8.

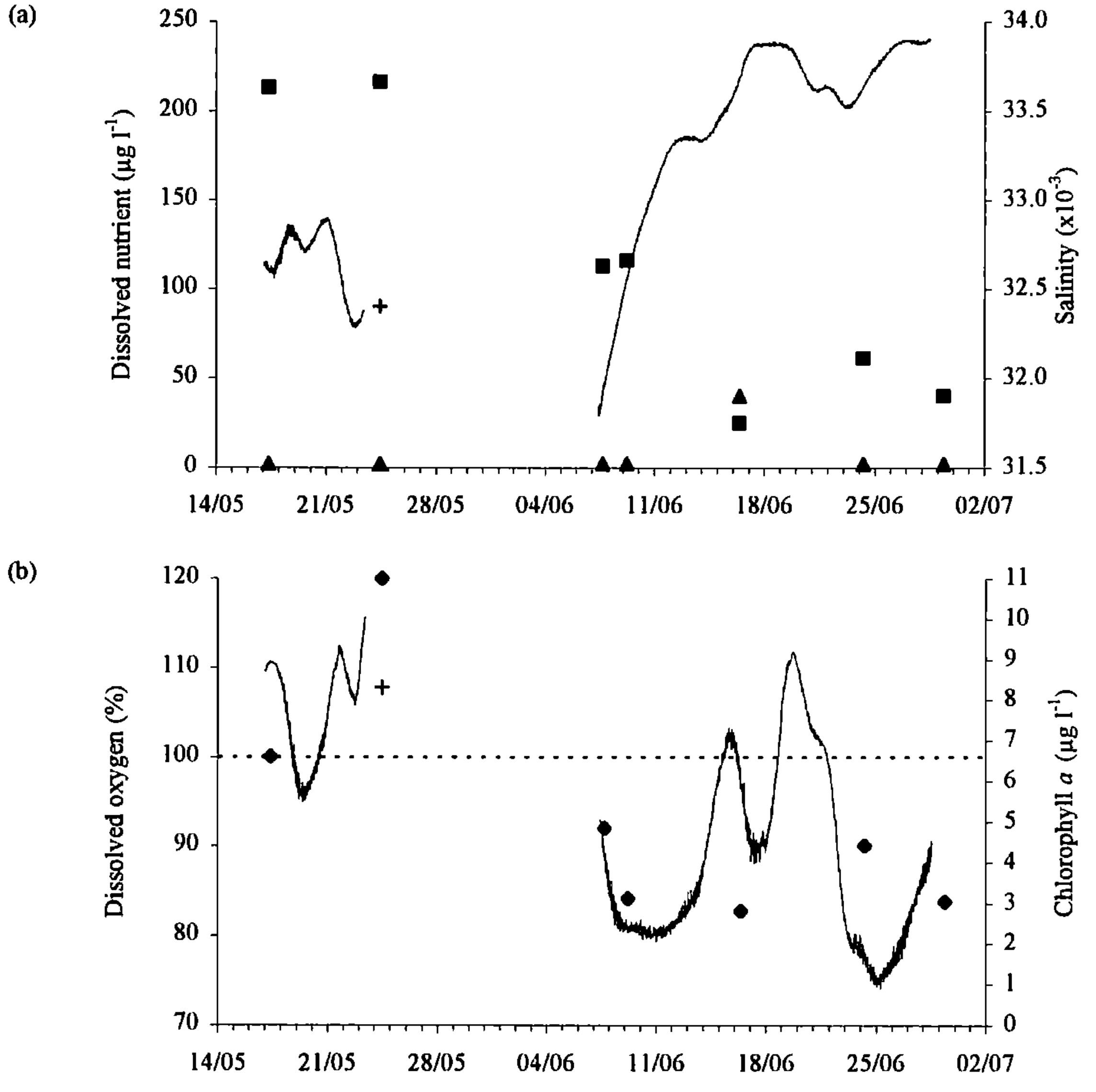


Fig. 4.21. Doodson-filtered time series of (a) salinity and (b) dissolved oxygen, with superimposed discrete measurements of (a) TON concentration (squares) and orthophosphate concentration (triangles) and (b) chlorophyll *a* concentration (diamonds), recorded at site 6 (Fig. 2.6) between 17/05/94 and 29/06/94. Crosses represent discrete measurements of (a) salinity and (b) dissolved oxygen in lieu of monitor values.

The harbour received 309 h of direct insolation during the period 17/05/94 and 29/06/94, an average of 7 h per day, although 211h (68 %) was received from 07/06/94 onwards. Only 76 mm of rain was recorded during the period, 55 mm (72%) of which fell between 17/05/94 and 26/05/94. The meteorological observations confirm the decreasing trend in salinity (Fig. 4.21a) observed midway between the neap survey (17/05/94) and the spring survey (24/05/94) and the increasing trend in salinity from 07/06/94 onwards. The high

Table 4.8. Change in each variable between sets of discrete observations.

Period	Salinity	TON	Orthophosphate	Chlorophyll *a*	Dissolved oxygen
17/05-24/05	decreased	high	low	increased	increased
24/05-07/06	decreased	decreased	low	decreased	decreased
07/06-09/06	increased	same	low	decreased	decreased
09/06-16/06	increased	decreased	increased	low	increased
16/06-24/06	decreased	increased	decreased	increased	decreased
24/06-29/06	increased	decreased	low	decreased	increased

chlorophyll *a* concentrations (Fig. 4.21b) at the end of the May 1994 survey period had declined by 07/06/94 and 09/06/94, whilst TON values had halved. Orthophosphate concentrations remained low. Oxygen saturation sagged to 80 % at this time, coincident with a period of low insolation. Seawater intrusion between 09/06/94 and 16/06/94 led to reduced concentrations of TON and chlorophyll *a*, although enhanced insolation raised the photosynthetic rate leading to increased dissolved oxygen saturation. Coincident with low values of chlorophyll *a*, orthophosphate concentrations increased slightly and may have promoted the phytoplankton growth observed between 16/06/94 and 24/06/94 that caused oxygen supersaturation, followed by undersaturation as the bloom decayed. Orthophosphate concentrations had decreased by 24/06/94, whilst TON values had risen in line with a slight decrease in salinity. Chlorophyll *a* concentrations had increased slightly from 16/06/94, but fallen again by 29/06/94 together with TON values, influenced by an increase in salinity.

Although an inverse correlation ($r = -0.66$) was exhibited between TON concentration and salinity during June 1994, a stronger positive correlation was found between concentrations of TON and chlorophyll *a* ($r = 0.82$). This may indicate a transition in nutrient supply and demand, from the pre-impoundment to the post-impoundment regime. Before impoundment, TON concentrations inside the harbour were generally in equilibrium with those outside, even during periods of enhanced primary productivity, and did not appear to biolimit growth. After impoundment, when the influx of estuarine brackish water was reduced, so too was input of TON. Concentrations *inside* the harbour may then have exerted more control upon phytoplankton growth, particularly during periods of low precipitation such as June 1994. The short-term cycling of phytoplankton growth and decay exhibited during this period of higher frequency sampling is consistent with observations described in Section *4.3.3* from Fig. 4.20. This illustrates the sensitivity of the water column to slight variations in salinity and hence the importance of nutrient concentrations when insolation is at a seasonal maximum. Any increases in nutrient concentration will clearly have a profound effect upon the phytoplankton growth within the impounded harbour.

4.4 Faecal Indicator Bacteria

The microbiological monitoring of Sutton Harbour was restricted to bacteriological determinations of total coliforms (TC), thermotolerant faecal coliforms (TFC, formerly referred to as *Escherichia coli*) and faecal streptococci (FS). The monitoring therefore represented only a partial characterisation in terms of the EC Bathing Water Directive (CEC, 1976b), because the area is not a Designated Bathing Water. No assays were made of *Salmonella sp.* or of enteroviruses and little data exist on the minimum infective dose of each pathogenic organism (UNEP/WHO, 1991). Therefore no direct assessments can be made of the potential risk to human health. However, recent epidemiological studies have confirmed that bathers in UK sea waters report higher incidences of minor illnesses than non-bathing controls (Balarajan *et al.*, 1991; Kay and Jones, 1992). These studies further improved upon the methods employed by the USEPA (Cabelli *et al.*, 1982) in which dose-response relationships were established between gastroenteritis and TFC densities far lower than the EC imperative limit shown in Table 4.9.

Table 4.9. Selected EC requirements for bathing water quality (CEC, 1976b).

Microbiological Parameter	Guideline (G) value	Mandatory (I) value
Total coliforms (TC), cfu $(100\ ml)^{-1}$	500[b]	10,000[a]
Faecal coliforms (TFC), cfu $(100\ ml)^{-1}$	100[b]	2,000[a]
Faecal streptococci (FS), cfu $(100\ ml)^{-1}$	100[b]	-
Salmonella, l^{-1}		0[a]
Enteroviruses, pfu $(10\ l)^{-1}$		0

[a]95 percentile; [b]80 percentile.

Although the harbour is not a Designated Bathing Water, it is a recognised Use Area (Fiddes and Lack, 1989) for marine recreation and is proximate to two Designated Bathing Waters (Plymouth East Hoe and West Hoe) and two non-designated bathing areas (Firestone Bay and Jennycliff Bay). The harbour should therefore be included in an integrated strategy for water quality in the surrounding area. However, it must be stressed that the EC guideline and imperative values do not apply to the harbour, and provide only a useful frame of reference against which trends are discussed in the following sections.

The relationship between salinity and faecal indicator bacteria was investigated by correlation (Table 4.10) in order to establish the relative importance of estuarine and local inputs to the harbour system. It is evident that the correlations were consistently negative, and it will be shown that counts were generally greater during or following periods of heavy rainfall. However, the correlations are not particularly strong which suggests that the major sources of faecal indicator bacteria to the harbour waters were localised rather than estuarine in origin. This hypothesis is supported by the knowledge that the localised raw sewage outfalls discharge *circa* 3×10^7 TFC l^{-1} compared with *circa* 3×10^5 TFC l^{-1} in

Table 4.10. Correlation coefficients (r) of TC, TFC and FS counts with salinity inside harbour from November 1991 to November 1994. All data were tidally-averaged between HW and LW and spatially averaged between SH1, SH2 and SH3. Summer is defined as May to September and winter as November to March.

Correlation	Season	Springs		Neaps	
		Surface	Bottom	Surface	Bottom
TC *versus* salinity	All data (n = 19)	-0.78[c]	-0.61[c]	-0.54[c]	-0.56[c]
	Summer (n = 9)	-0.76[b]	NS	NS	NS
	Winter (n = 10)	-0.78[c]	-0.64[a]	NS	-0.69[a]
TFC *versus* salinity	All data (n = 19)	-0.60[c]	-0.59[c]	-0.47[a]	-0.50[a]
	Summer (n = 9)	NS	NS	NS	NS
	Winter (n = 10)	NS	NS	NS	NS
FS *versus* salinity	All data (n = 19)	-0.47[a]	-0.54[b]	-0.55[b]	NS
	Summer (n = 9)	NS	NS	NS	NS
	Winter (n = 10)	NS	NS	NS	NS

NS - not significant at $p < 0.05$; [a]$p < 0.05$; [b]$p < 0.01$; [c]$p < 0.005$;

secondary-treated discharges from the four Sewage Treatment Works at Marsh Mills, Billacombe, Radford and Cattedown in the Plym Estuary (Babbedge, pers. comm.). Furthermore, UK storm sewers and foul sewers are generally combined, such that after a period of heavy rainfall stormwater overflows contain foul water and can be highly polluting (Grantham, 1993); one such stormwater overflow is thought to be situated in the harbour wall beneath Sutton Pier.

The relationship between counts of TC, TFC and FS was examined by correlation (Table 4.11) to see if the sources of each were the same. There were significant positive correlations ranging from strong to slight throughout the dataset suggesting that the major sources of each indicator were probably the same, namely the local sewage outfalls rather than the STW's. The strongest correlations were generally observed between TC and TFC. Meynard *et al.* (1989) demonstrated similarly strong positive correlations between TC and TFC during a three year inter-laboratory comparison and suggested that this illustrated a considerable redundancy of data. Recent NRA monitoring policy reflects this, and future studies of this nature where monitoring of TC is not obligatory should consider its omission in favour of doubling the TFC monitoring effort, either through increased replication or enhanced spatial resolution.

Table 4.11. Correlation coefficients (r) between TC, TFC and FS counts inside harbour from November 1991 to November 1994. All data were tidally-averaged between HW and LW and spatially averaged between SH1, SH2 and SH3. Summer is defined as May to September and winter as November to March.

Correlation	Season	Springs		Neaps	
		Surface	Bottom	Surface	Bottom
TC *v.* TFC	All data (n = 19)	0.89[c]	0.98[c]	0.77[c]	0.75[c]
	Summer (n = 9)	0.90[c]	0.81[c]	0.80[c]	0.74[b]
	Winter (n = 10)	0.88[c]	0.99[c]	0.62[a]	0.60[a]
TC *v.* FS	All data (n = 19)	0.68[c]	0.96[c]	0.71[c]	0.61[c]
	Summer (n = 9)	0.90[c]	0.82[c]	0.61[a]	0.74[b]
	Winter (n = 10)	0.56[a]	0.97[c]	0.69[b]	0.57[a]
TFC *v.* FS	All data (n = 19)	0.70[c]	0.95[c]	NS	NS
	Summer (n = 9)	0.69[b]	NS	NS	NS
	Winter (n = 10)	0.70[b]	0.98[c]	NS	NS

NS - not significant at $p < 0.05$; [a]$p < 0.05$; [b]$p < 0.025$; [c]$p < 0.005$.

4.4.1 Semi-diurnal and diurnal variability

The major sources of faecal indicator bacteria to the harbour system operate on a semi-diurnal cycle: the sewage discharges from Fishers Nose and other holding tanks along the waterfront are triggered hydrostatically between maximum ebb and LW. The discharge around LW was confirmed by the presence of a malodorous surface boil on many occasions during the study. The effluent was therefore available for advection into the harbour on the flood tide *via* the tidal shunting mechanism. This process is illustrated by the time series in Fig. 4.22, which show that counts of TFC in the surface waters were generally greater at LW outside the harbour but greater at HW inside. The removal processes operate on a variety of short time scales that are of varying importance dependent upon season. Increases in water temperature, salinity and solar intensity are known to increase the mortality rate of enteric bacteria (Mitchell and Chamberlin, 1978; Solic and Krstulovic, 1992) therefore the rate will be highest in summer. However, the effects will be greatest in the surface waters because the bactericidal UV radiation is rapidly attenuated in the water column (Jerlov, 1968) and semi-diurnal salinity variations and diurnal temperature variations are greater here than at the bottom.

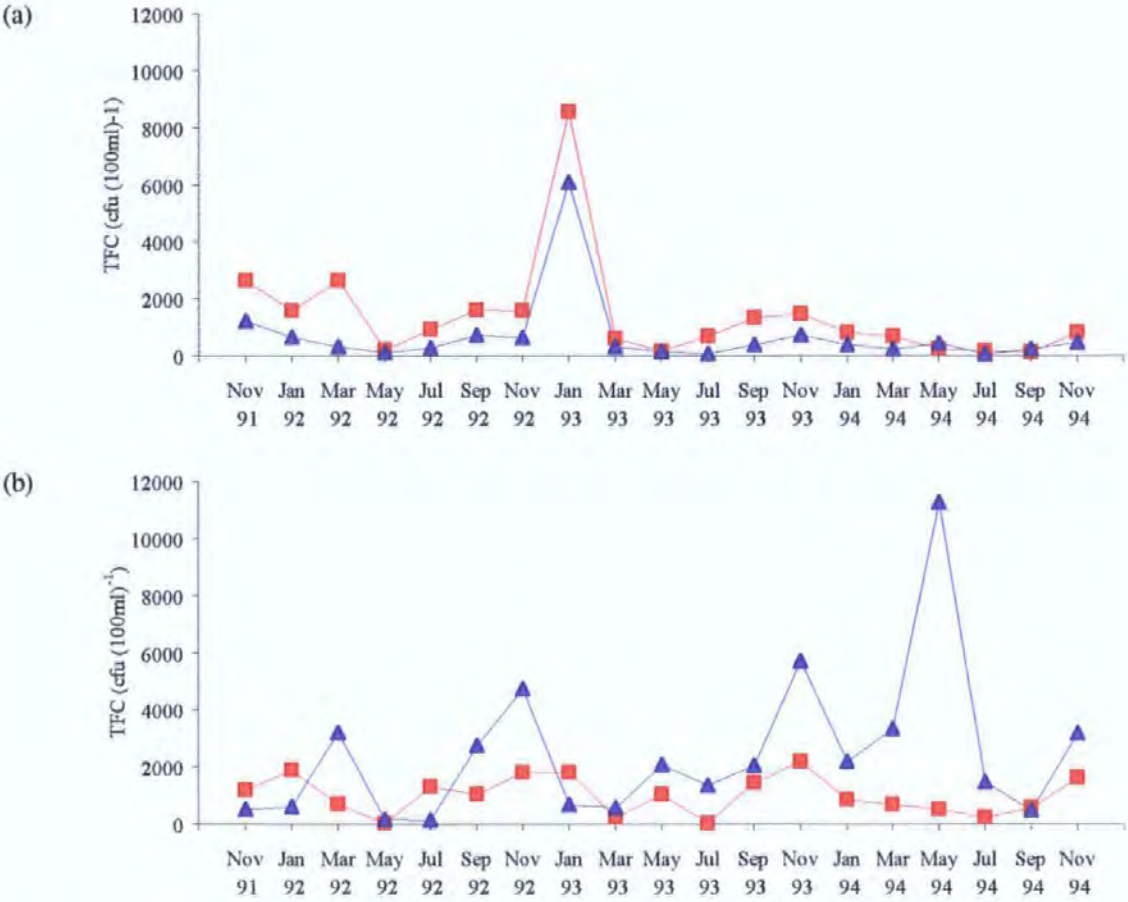
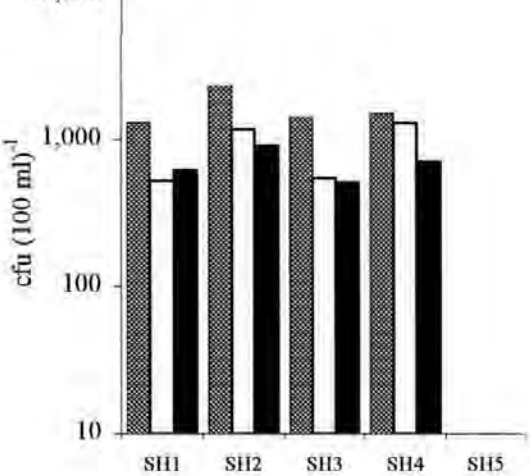


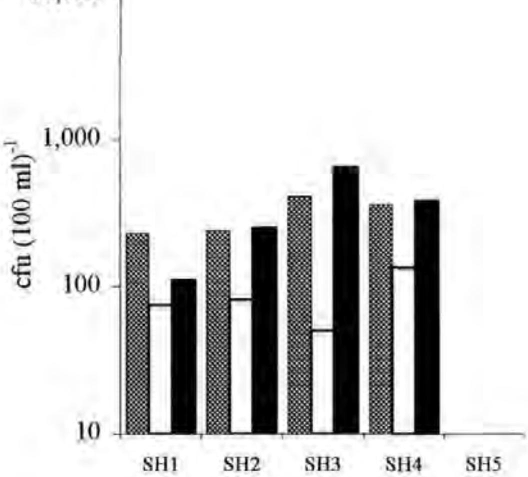
Fig. 4.22. Semi-diurnal variability of TFC (cfu $(100\ \text{ml})^{-1}$) in the surface waters during (a) spring tides inside harbour (mean of SH1 to SH3) and (b) neap tides outside harbour (mean of SH4 and SH5). Red squares represent HW and blue triangles represent LW.

Semi-diurnal and diurnal variability of faecal indicator bacteria counts was examined during periods of high environmental stress (high water temperatures, salinity and solar intensity) at neaps in July 1992, July 1993 and July 1994. The surface water counts of TC, TFC and FS at HW and LW on each occasion are shown in Figs. 4.23a to f on a logarithmic scale. These were also periods of low inter-annual environmental variability (Section *3.2.2*) with surface temperatures in the range 15 to 18 °C inside and 14.5 to 17.5 °C outside and salinities in the range 32.5 to 34.5 $\times 10^{-3}$ inside and 32.5 to 34.0 $\times 10^{-3}$ outside. The values of precipitation in the previous 4 d period and direct insolation on the day were 14 mm and 4 h in July 1992, 0 mm and 10 h in July 1993 and 0 mm and 4 h in July 1994.

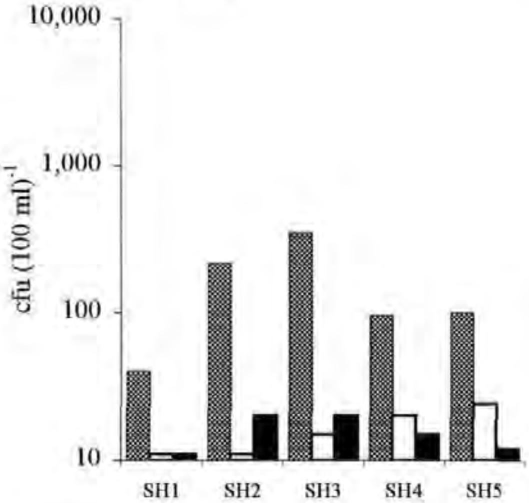
July 1992 (Figs 4.23a and b). Counts of all bacteria inside the harbour were generally greater at HW in the late morning than at LW in the late afternoon. This was also the case outside the harbour at SH4. Monitoring at the additional station SH5 was not undertaken in 1992, but it is safe to assume that the discharge around LW from Fishers Nose occurred as usual and therefore elevated counts would have been expected here. These distributions



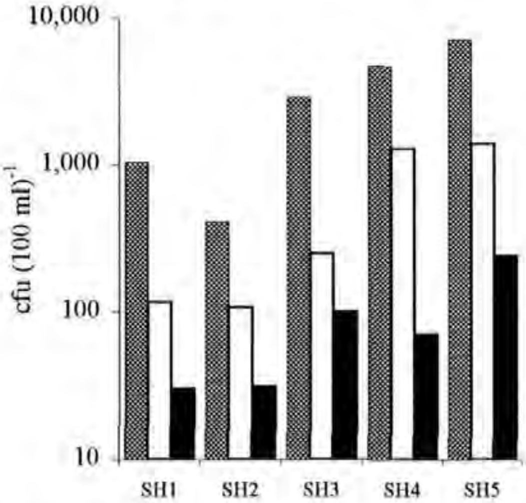
(a) 23/07/92 HW (1129)



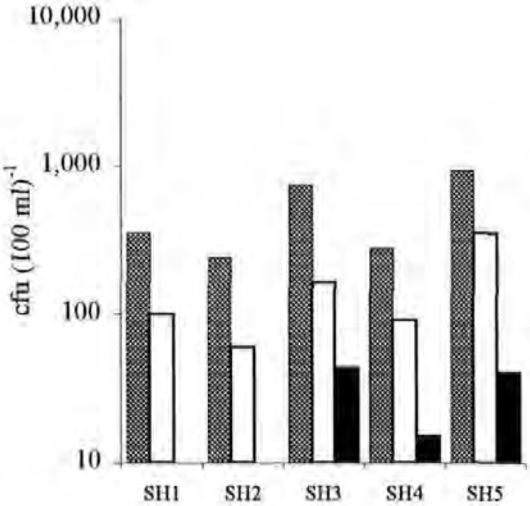
(b) 23/07/92 LW (1752)



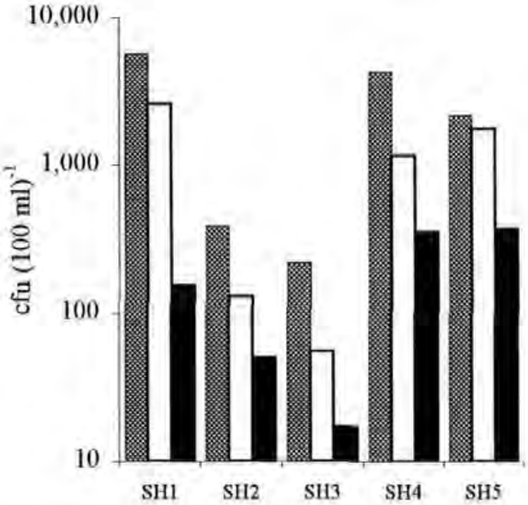
(c) 29/06/93 HW (1422)



(d) 29/06/93 LW (0801)



(e) 04/07/94 HW (1518)



(f) 04/07/94 LW (0849)

Fig. 4.23. Comparison of faecal indicator bacteria counts in surface water samples at neaps during July 1992, July 1993 and July 1994 surveys. Hatched bars represent TC, open bars represent TFC and filled bars represent FS. Captions (a) to (f) denote survey date, tidal state and time of HW or LW (BST).

were typical of those observed under similar conditions during the study, with high bacterial removal rates due to temperature, salinity, moderate insolation and physical removal by the ebb tide. The processes were not complicated by stormwater flows or high riverine influx. The data do reveal a persistence of FS relative to the other bacteria, particularly at SH3, with implications for the implied human health risk if this equates with the persistence of enteroviruses (Fattal *et al.*, 1983), despite counts of TC and TFC below the EC imperative limits.

July 1993 (Figs. 4.23c to d). Counts of all bacteria were higher outside the harbour than inside the harbour at LW in the early morning, particularly at SH5 where the influence of the sewage discharge is clearly evident. All counts had reduced considerably under intense insolation at HW in the mid-afternoon. However, the counts of TC and FS were marginally greater inside the harbour than outside as a result of the advection of contaminated water into the harbour and dilution of the water outside with uncontaminated seawater during the flood tide. All counts were below the EC imperative limits at LW and below the guideline limits at HW.

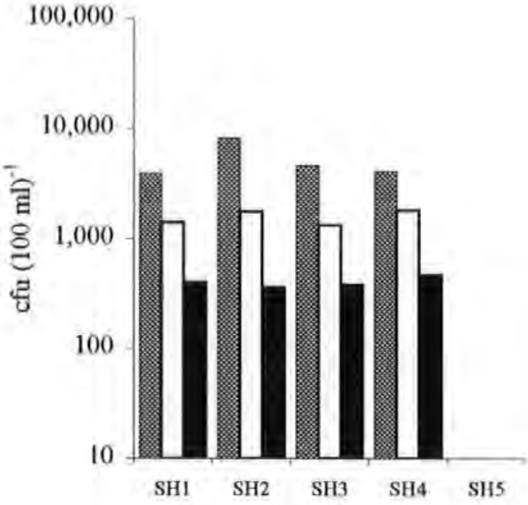
July 1994 (Figs. 4.23e to f). Counts of all bacteria outside the harbour were more than those inside the harbour at SH2 and SH3 at LW in the early morning, due to the consistent effect of the sewage discharge. However, the highest counts of TC and TFC were observed at SH1 in Sutton Marina. The dry weather rules out a stormwater overflow event and therefore the most likely explanation was the use of yacht heads in the busy marina with the localising effect of an 8 to 10 knot SW'ly wind. Whilst diligent yacht owners may use the marina facilities during daylight hours, the popular view is that many do not during the night and this hypothesis is supported by the timing of the observations. The only solution to a problem that could become acute in an impounded harbour is greater public education and awareness on the part of Sutton Harbour Company. Counts of all bacteria were lower both inside and outside the harbour at HW in the mid-afternoon, with the highest counts now at SH5. In the absence of strong insolation, it would seem that the physical barrier of the lock gates prevented the advection of contaminated water on the early flood, allowing only the advection of less contaminated seawater during the mid to late flood. This is seen as a very positive effect of the impoundment of Sutton Harbour, and has led to improved microbiological water quality. All counts were below the EC imperative limits at LW, and below the guideline limits at SH1 and SH2 at HW despite only moderate insolation.

Semi-diurnal and diurnal variability of faecal indicator bacteria counts was also examined during periods of lower environmental stress (low water temperatures, salinity and solar intensity) at neaps in November 1992, November 1993 and November 1994. The surface water counts of TC, TFC and FS at HW and LW on each occasion are shown in Fig. 4.24 on a logarithmic scale. These were also periods of higher inter-annual environmental variability (Section *3.2.2*) with surface temperatures in the range 9.5 to 12.5 °C inside and 9.0 to 12.5 °C outside and salinities in the range 24.0 to 32.0 $\times 10^{-3}$ inside and 20.0 to 32.0 $\times 10^{-3}$ outside. The values of precipitation in the previous 4 d period and direct insolation on the day were 24 mm and 4 h in November 1992, 17 mm and 3 h in November 1993 and 0 mm and 0 h in November 1994.

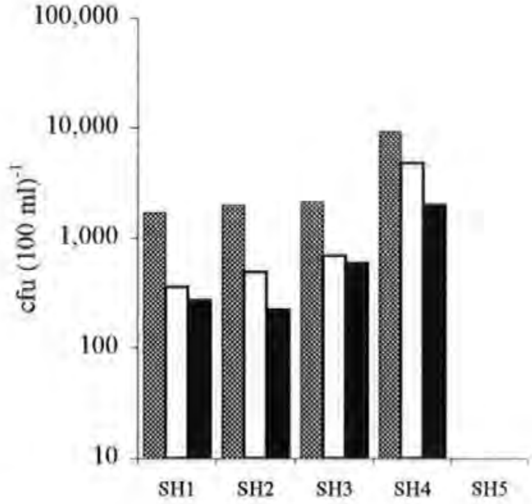
November 1992 (Figs. 4.24a and b). Counts of all bacteria were similar inside and outside the harbour at HW in the mid-morning, although TC counts were marginally higher at SH2. However, there was little evidence of spatial differences inside the harbour resulting from stormwater overflow activity or other localised inputs, despite the antecedent rainfall. All counts were reduced inside the harbour at LW in the late afternoon as a result of bacterial mortality processes including moderate insolation. Outside the harbour all counts had increased due to the LW discharge, with TC counts at almost the EC imperative limit and TFC counts over twice the limit. This illustrates the effect of greater precipitation and lower mortality rates as the winter period began.

November 1993 (Figs. 4.24c and d). Counts of all bacteria were slightly greater inside the harbour than outside at HW in the late morning. In particular, TFC counts were between one and three times the EC imperative limit. FS counts were thirty times the guideline limit at SH1, although this spatial enhancement was not exhibited in TC or TFC, so it is unlikely that this was a result of local inputs. Counts of all bacteria were only slightly reduced inside the harbour at LW in the late afternoon due to lower mortality rates. All counts outside the harbour were again greater due to the LW discharge, with TC counts over 1.5 times and TFC counts over 3 times the EC imperative limits, illustrating the consistently poor microbiological water quality.

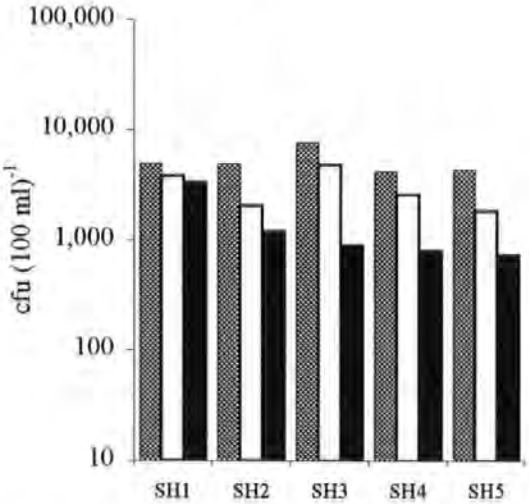
November 1994 (Figs. 4.24e and f). All counts were particularly low inside the harbour at LW in the early morning, whilst all counts were 5 to 10 times greater outside the harbour due to the sewage discharge. The counts outside were only partially reduced at HW in the mid afternoon, as a possible consequence of high discharges from sewage outfalls seaward of Fishers Nose combined with enhanced survival of bacteria from the previous tide under lower environmental stresses. However, the counts inside the harbour remained almost unchanged from those at LW and constituted the lowest observations of the present study for the winter period. This confirms that the lock acts in the same way under winter conditions as under summer conditions, preventing the ingress of contaminated water during the first half of the flood tide.



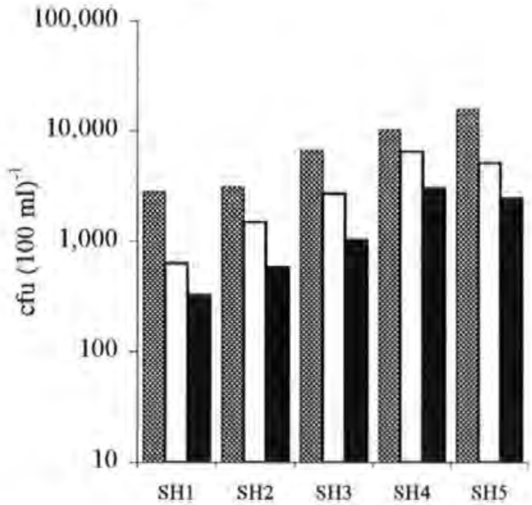
(a) 17/11/92 HW (1017)



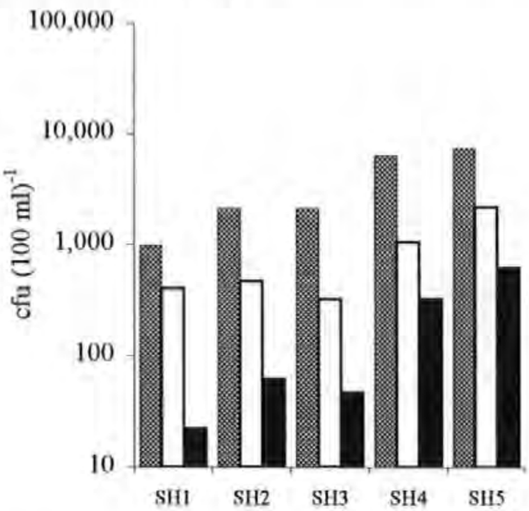
(b) 17/11/92 LW (1645)



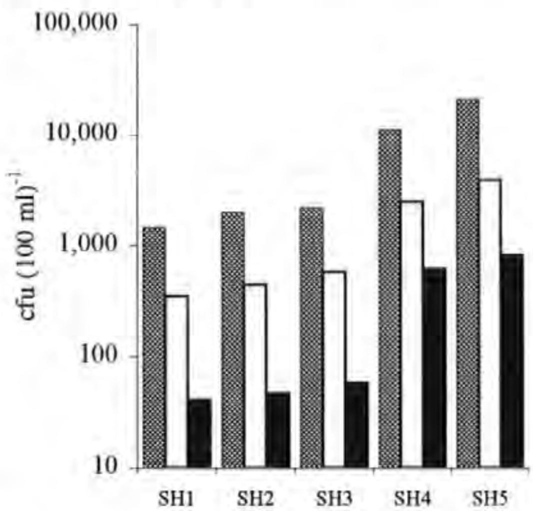
(c) 07/12/93 HW (1110)



(d) 07/12/93 LW (1734)



(e) 29/11/94 HW (1400)



(f) 29/11/94 LW (0801)

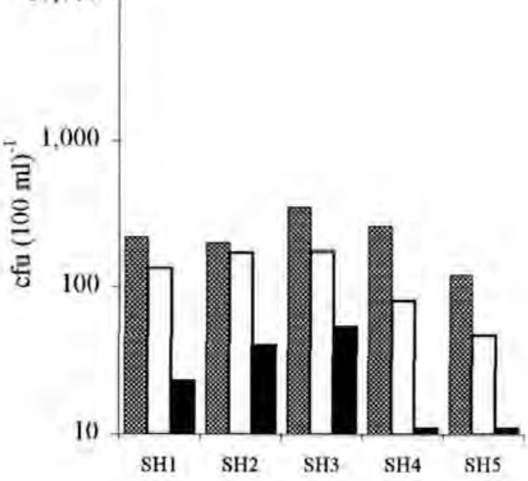
Fig. 4.24. Comparison of faecal indicator bacteria counts in surface water samples at neaps during November 1992, November 1993 and November 1994 surveys. Hatched bars represent TC, open bars represent TFC and filled bars represent FS. Captions (a) to (f) denote survey date, tidal state and time of HW or LW (GMT).

4.4.2 Spring-neap tide variability

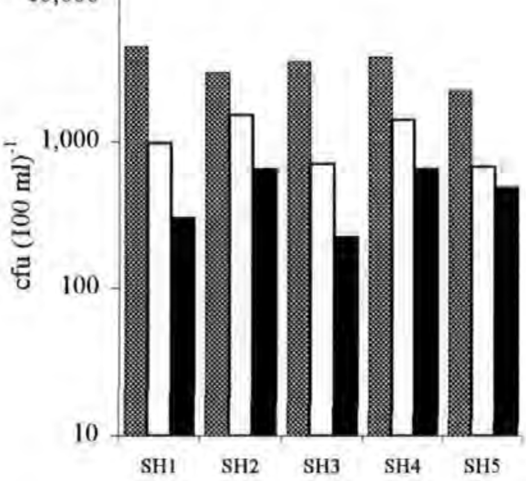
Variability of faecal indicator bacteria counts on the spring-neap tide timescale should occur primarily as a result of variations in magnitude of the source discharges promoted by precipitation. However, they may be modified by differences in insolation and to a lesser extent by the effects of salinity and temperature upon mortality rates. Differences in stratification resulting from spring-neap tidal differences may also be important. Greater stratification at neaps may enhance UV sterilisation at the surface, and survival at depth, whilst stronger mixing at springs may ensure even exposure throughout the water column to the processes of bacterial mortality. In order to investigate these ideas, spring-neap tide variability was first examined during periods of high environmental stress during May 1993 and May 1994. Surface counts of TC, TFC and FS on each occasion at HW springs and HW neaps are shown in Fig. 4.25. These were also periods of low inter-annual environmental variability (Section *3.2.2*) with surface temperatures in the range 11.5 to 12.5 °C, and salinities in the range 32 to 33 $\times 10^{-3}$, inside and outside the harbour.

May 1993 (Figs. 4.25a and b). As a consequence of no rainfall in the previous 4 d period and intense insolation (13 h) during the day, counts of TFC were low and values of TC and FS lay below the EC guideline limits at HW springs inside the harbour. Values were equally low at LW throughout (not shown). They were considerably greater throughout the system one week later at neaps after 12 mm of rain fell in the previous 4 d period, 10 mm on the morning of the survey. The typical LW pattern, of lower counts inside and higher values outside was observed, but all remained below the EC imperative limits despite an absence of direct insolation. These observations illustrate the profound effect of differences in insolation, and of short-term rain events, upon microbiological water quality.

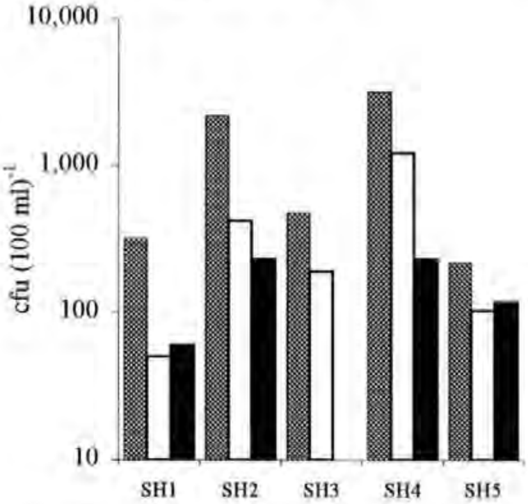
May 1994 (Figs. 4.25c and d). All counts were lower inside the harbour than outside at HW neaps despite LW values at Fishers Nose of 4 and 10 times the EC imperative limits for TC and TFC. The high counts at the discharge were related to the 35 mm of rain which fell during the previous 4 d period, 14 mm on the morning of the survey, with no direct insolation. This finding further illustrates the beneficial effects of lock closure during the early flood. Slight enhancement at SH2 was thought to be a close coupling with water from SH4 during the late flood period. One week later at HW springs an almost identical distribution was observed when all counts were lower inside than at SH4 despite LW values of 3 and 9 times the EC imperative limits for TC and TFC at Fishers Nose. Again, the high counts at the discharge were promoted by 17 mm of rain in the previous 4 d period and 4 mm on the morning of the survey, with no direct insolation. On this occasion, LW counts inside the harbour were low at SH2 and SH3, but higher at SH1. This may indicate the existence of localised stormwater inputs in the north of the harbour, curtailed and diluted at HW. Close coupling of the water at SH2 and SH4 was again in



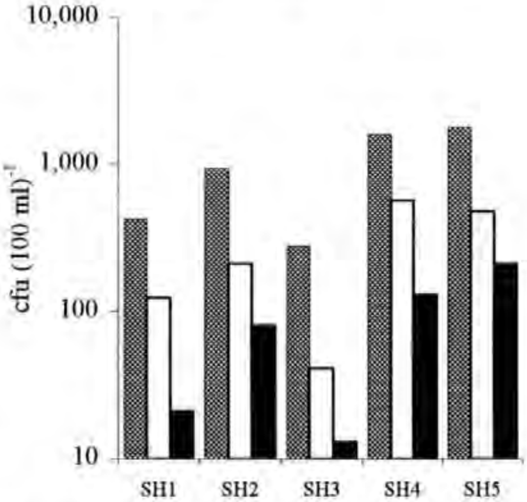
(a) 05/05/93 Springs (HW 1812)



(b) 12/05/93 Neaps (HW 1048)



(c) 24/05/94 Springs (HW 1819)

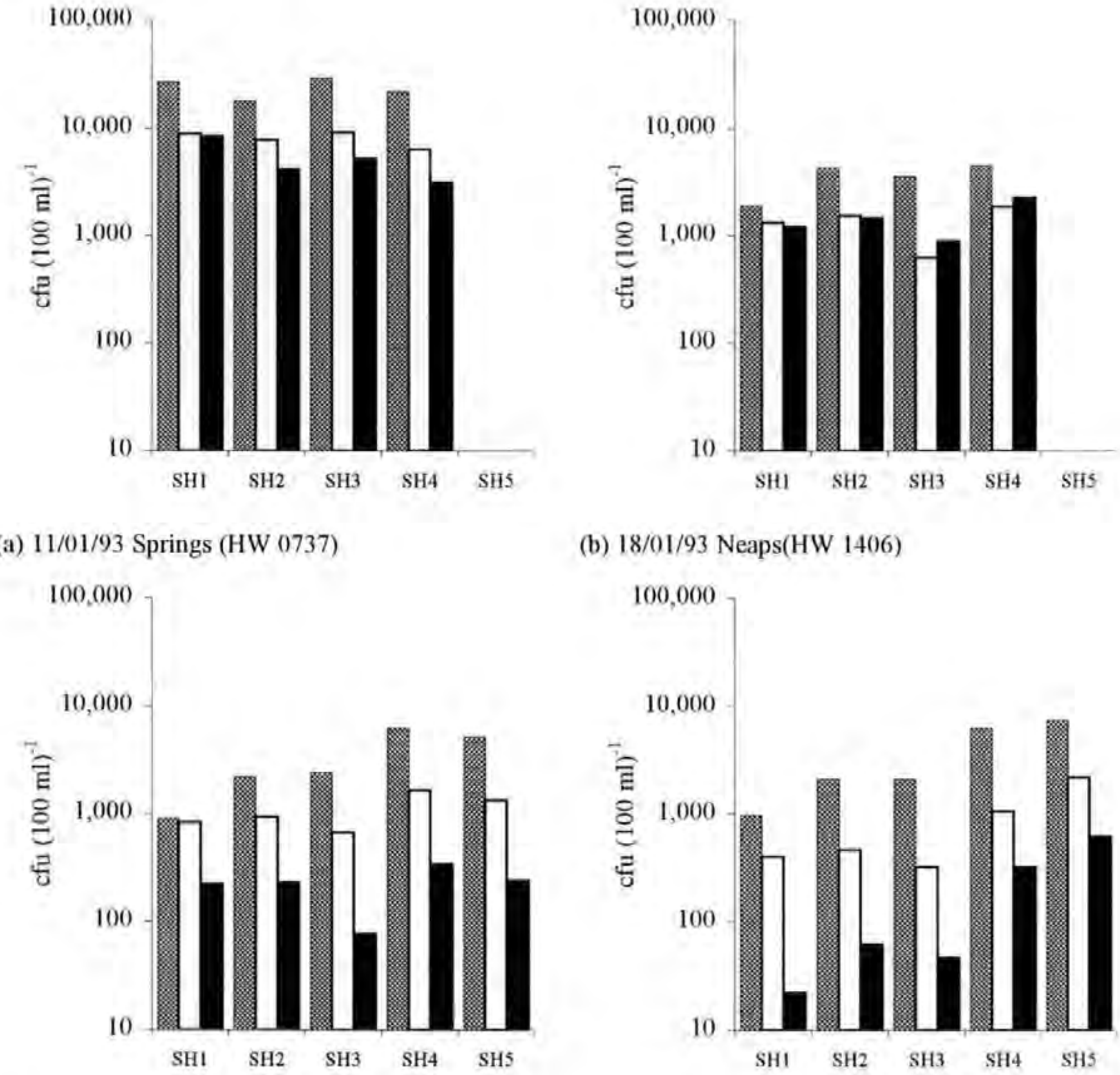


(d) 17/05/94 Neaps (HW 1040)

Fig. 4.25. Comparison of faecal indicator bacteria counts in surface water samples at HW during May 1993 and May 1994 spring and neap surveys. Hatched bars represent TC, open bars represent TFC and filled bars represent FS. Captions (a) to (d) denote survey date, tidal range and time of HW (BST).

evidence after the late flood period. Similar environmental and meteorological conditions produced similar distributions of faecal indicator bacteria after impoundment, and therefore spring-neap tide differences appeared to be minimal under these conditions.

Spring-neap tide variability was also examined during periods of lower environmental stress in January 1993 and November 1994. Surface counts of TC, TFC and FS on each occasion at HW springs and HW neaps are shown in Fig. 4.26. These were also periods of higher inter-annual environmental variability (Section *3.2.2*) with surface temperatures of 9 to 13 °C and salinities of 19 to 28 x10^{-3} at springs and surface temperatures of 8.5 to 12.5 °C and salinities of 20 to 32 x10^{-3} at neaps inside and outside the harbour.



(a) 11/01/93 Springs (HW 0737)

(b) 18/01/93 Neaps(HW 1406)

(c) 22/11/94 Springs (HW 0740)

(d) 29/11/94 Neaps (HW 1400)

Fig. 4.26. Comparison of faecal indicator bacteria counts in surface water samples at HW during January 1993 and November 1994 spring and neap surveys. Hatched bars represent TC, open bars represent TFC and filled bars represent FS. Captions (a) to (d) denote survey date, tidal range and time of HW (GMT).

January 1993 (Figs. 4.26a and b). All counts observed at springs both inside and outside the harbour were the highest recorded in routine monitoring during the present study. This finding was almost certainly the result of heavy rainfall (36 mm) over the previous 4 d period, with 6 mm falling on the morning of the survey, causing heavy discharges from sewage outfalls and probable functioning of stormwater overflows. TC counts of 1.5 to 3.5 times and TFC values of 2.5 to 4.5 times the EC imperative limits were observed, with only a slight reduction inside and a slight enhancement outside seen at LW. The microbiological water quality of the harbour was clearly very poor under these conditions of sewer spate and no direct insolation. One week later at neaps all counts at HW were considerably reduced below the EC imperative limits throughout the harbour system because of lower rainfall (4 mm) over the previous 4 d period. All counts were slightly enhanced throughout over the preceding LW probably as a result of a modest discharge from Fishers Nose (unmeasured). A 4 h period of broken winter sunshine during the day would have been unlikely to have a significant effect upon bacterial mortality, so that these observations are thought to illustrate the importance of stormwater flows in controlling levels of faecal bacteria independent of insolation.

November 1994 (Figs. 4.26c and d). Surface counts of all bacteria at HW springs were lower inside the harbour than outside despite 14 mm of rainfall over the previous 4 d period. The difference was still greater at LW owing to the effects of the discharge at Fishers Nose. However, all counts remained below the EC imperative limits. One week later at HW neaps all counts were further reduced inside the harbour with no rainfall over the previous 4 d period and no insolation. This finding was despite the recording of high counts at LW at the Fishers Nose discharge of twice the EC imperative limits. Microbiological water quality in the impounded harbour appears less prone to variation on the spring-neap tide timescale mainly because of the physical process of impoundment. However, it must be noted that conditions of high stormwater flow were not observed after impoundment, so that the effect of this process cannot be assessed from the current dataset.

4.4.3 Seasonal and inter-annual variability

It has been shown in previous sections that the important environmental factors in control of microbiological water quality are precipitation and insolation. The amount of precipitation controls the volume of sewage discharges, whilst the duration and intensity of insolation as a major influence on bacterial mortality rates in the marine environment, the effect of water temperature and salinity being negligible in comparison. These factors have been shown to vary on seasonal and inter-annual timescales (Section *3.2.1*), and to lead to the associated variability discussed here. Seasonal variabilities of TC, TFC and FS at springs and at neaps are shown in the time series in Figs. 4.27 and 4.28 respectively. The most striking features are the atypical peaks exhibited at springs in January 1993 particularly in the surface waters, which coincided with a storm event as discussed in Section *4.4.2*. It is evident that mean surface and bottom counts lay at or below EC imperative limits during the remainder of the present study. However, as will be illustrated in Section *4.4.4*, the frequency of such storm events was far greater than would be estimated from these time series alone. There are obvious consequences of the "snapshot" nature of discrete monitoring wherever it is employed, particularly in Designated Bathing Waters under current EC legislation. Designs of monitoring programmes may need to be more flexible and responsive to such ephemeral features.

Springs (Figs. 4.27a to c). Several trends were evident in the time series of seasonal bacterial counts at springs. Troughs were observed in May and July under decreased precipitation and high insolation. Conversely, peaks were recorded during November and January under increased precipitation and low insolation. The spring tidal mixing

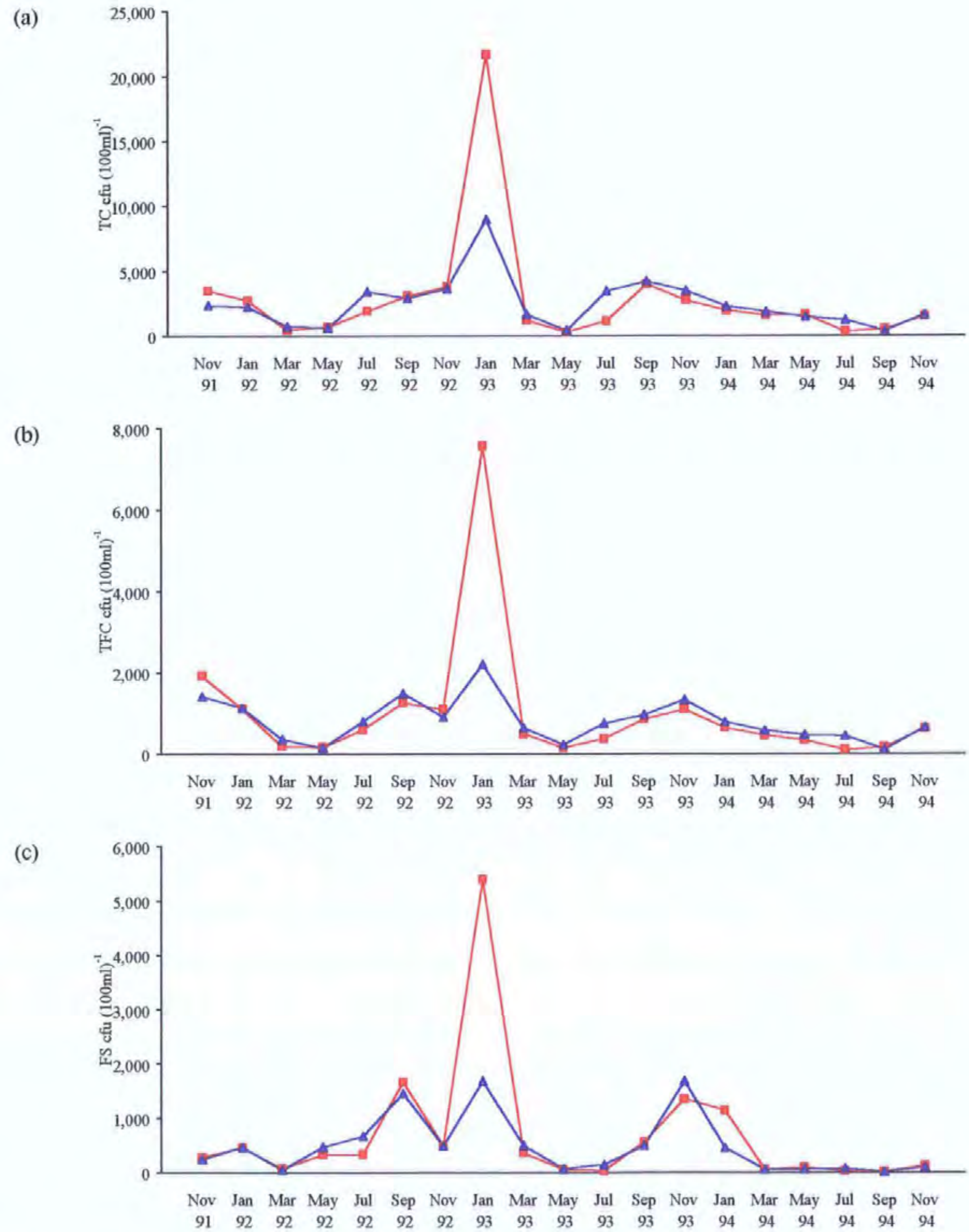


Fig. 4.27. Tidally-averaged time series of faecal indicator bacteria counts at springs inside Sutton Harbour from November 1991 to November 1994 for (a) TC, (b) TFC and (c) FS. Red lines represent surface samples and blue lines represent bottom samples. All values are mean of SH1 to SH3.

generally resulted in small vertical differences in counts. Vertical differences were greatest when the controlling factors were most extreme: high insolation in July 1992 and July 1993 resulting in relatively low surface counts and high precipitation in November 1991 and January 1993 resulting in relatively high surface counts. From May 1994 onwards counts were particularly low, with September 1994 and November 1994 exhibiting the lowest counts of TC, TFC and FS in each respective period during the present study.

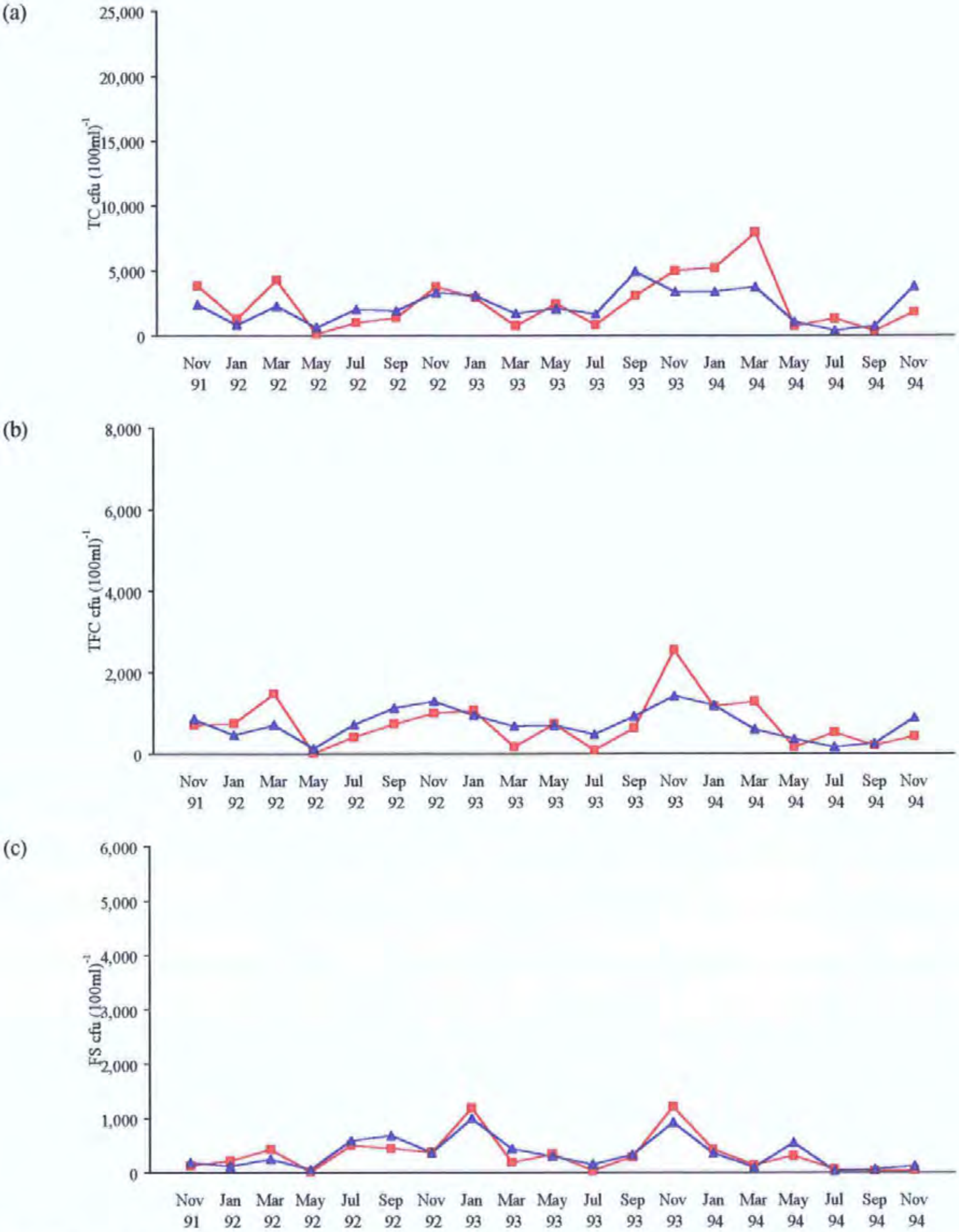


Fig. 4.28. Tidally-averaged time series of faecal indicator bacteria counts at neaps inside Sutton Harbour from November 1991 to November 1994 for (a) TC, (b) TFC and (c) FS. Red lines represent surface samples and blue lines represent bottom samples. All values are mean of SH1 to SH3.

Neaps (Figs. 4.28a to c). Similar trends were evident in the time series of counts at neaps, apart from the notable lack of high counts in January 1993 as previously discussed. Seasonal troughs were evident in May and July with seasonal peaks in November and January. However, these variations were slightly less pronounced than at springs, which might reflect the decrease in tidal incursion of outside water at neaps. The vertical differences in counts were slightly more marked than at springs under reduced tidal mixing, manifesting as enhanced surface TC and TFC counts during greater stratification from November 1991 to March 1992 and November 1993 to March 1994 and reduced surface counts during high insolation in July and September 1992 and July and September 1993. Counts were again particularly low from May 1994 onwards, with surface and bottom counts in September 1994 and surface counts in November 1994 as the lowest observed during the present study. One exception was the bottom water in November 1994, in which all counts were as high or higher than before impoundment, as a possible result of enhanced settling.

The time series in Figs. 4.29 and 4.30 show the difference in faecal bacteria counts between inside and outside the harbour at springs and neaps respectively.

Springs (Figs. 4.29a to c). Surface counts of all bacteria at springs were generally greater outside the harbour due to the location of the sewage discharges. The differences in bottom counts of all bacteria were smaller, but were generally greater inside the harbour. There were many possible explanations for this phenomenon: settling of bacteria in association with SPM may have been enhanced inside the harbour; the remoteness from the sewage discharges may have allowed time for more effective downward mixing of the contaminated surface water; the shallower water in the harbour may have allowed a greater relative penetration of UV radiation. The main exceptions to this bottom trend were observed in January 1993 under high stormwater flows outside the harbour and in May 1994 when the lock gates restricted the incursion of contaminated bottom and surface water. From May 1994 onwards, with the exception of July 1994 bottom counts, all counts were greater outside the harbour, further supporting the hypothesis that the influx of contaminated water was curtailed by the closure of the lock gates.

Neaps (Figs. 4.30a to c). Surface counts of all bacteria at neaps were again generally greater outside the harbour due to the location of the sewage discharges. Differences in bottom counts were again less extreme and no clear trend between inside and outside enhancement was seen in any of the indicator bacteria. As observed at springs, from May 1994 onwards all surface and bottom counts were greater outside the harbour than inside.

4.4.4 Higher resolution measurements

The extended and enhanced sampling period undertaken from 22/06/93 to 27/07/93 revealed evidence of a significant process that would otherwise have been undetected by the routine monitoring: the effect of stormwater overflows (Grantham, 1993) on the microbiological water quality of the harbour during the summer period. There was no rainfall in the 4 d period before the spring tide survey (07/07/93) whilst 27 mm of rain fell in the 4 d period before the additional neap survey (15/07/93) with 17 mm during the morning of the survey. Surface counts of TC, TFC and FS for each date are shown in Fig. 4.31. Site numbers and the rationale for site selection are given in Section *2.1.3*.

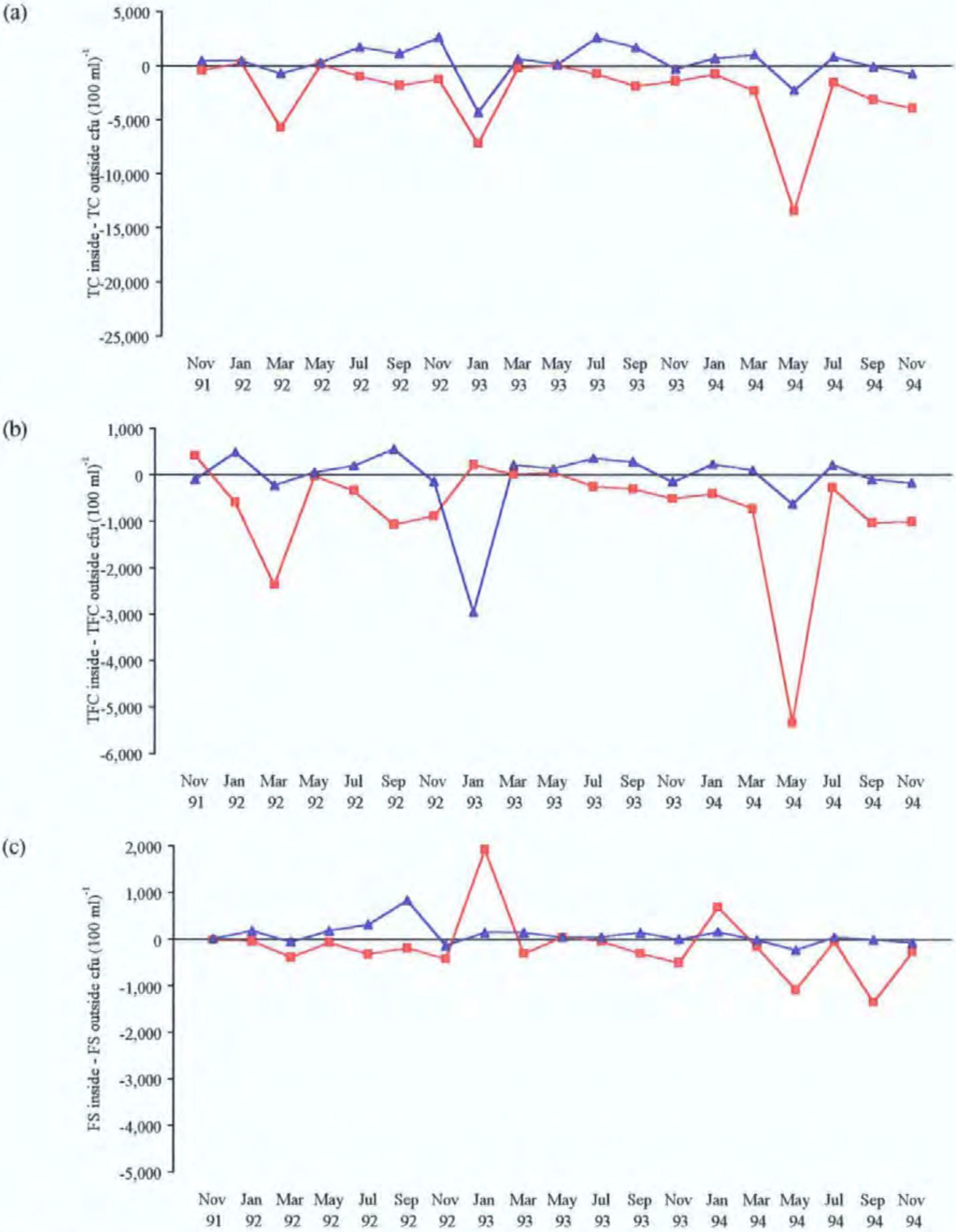


Fig. 4.29. Time series of spring tide differences in faecal indicator bacteria counts between inside harbour (mean of SH1 to SH3) and outside harbour (mean of SH4 and SH5) for (a) TC, (b) TFC and (c) FS. Red lines represent surface samples and blue lines represent bottom samples.

Springs 07/07/93 (Figs. 4.31a, c and e). Counts of all bacteria at LW were low (TC <500, TFC <200 and FS <100 cfu (100 ml)$^{-1}$) at six sites inside the harbour and immediately outside the harbour mouth. At Fishers Nose (site 10) counts were slightly greater due to the proximity of the outfall (TC 990, TFC 430 and FS 20 cfu (100 ml)$^{-1}$). At site 1 (NW corner of the Marina) and site 4 (near Sutton Pier) all counts were significantly greater (TC >2,000, TFC >1,000 and FS >500 cfu (100 ml)$^{-1}$) which suggests the presence of localised sources. However, it should be noted that all counts were below the EC imperative limits and most were below the EC guideline limits set for bathing waters.

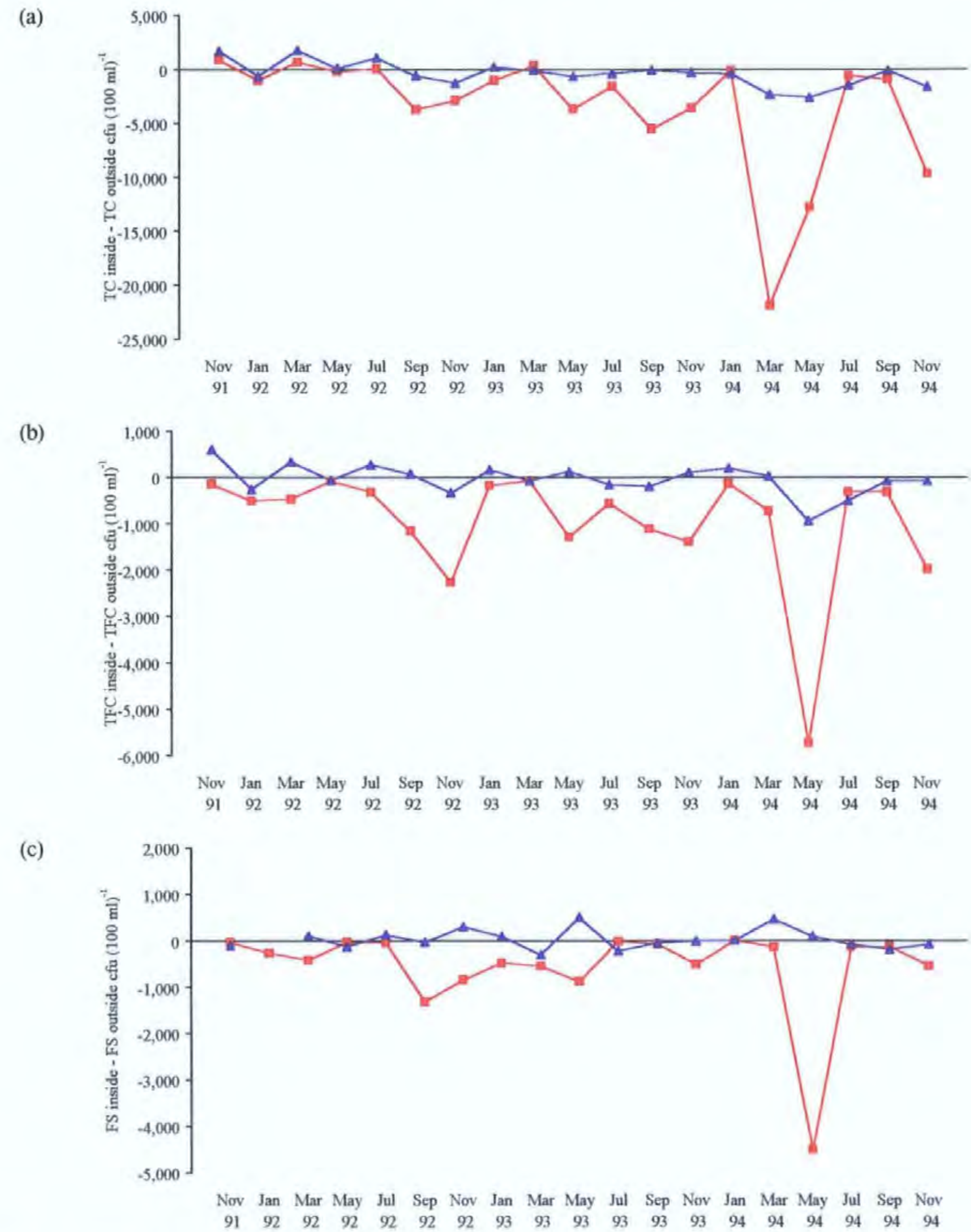
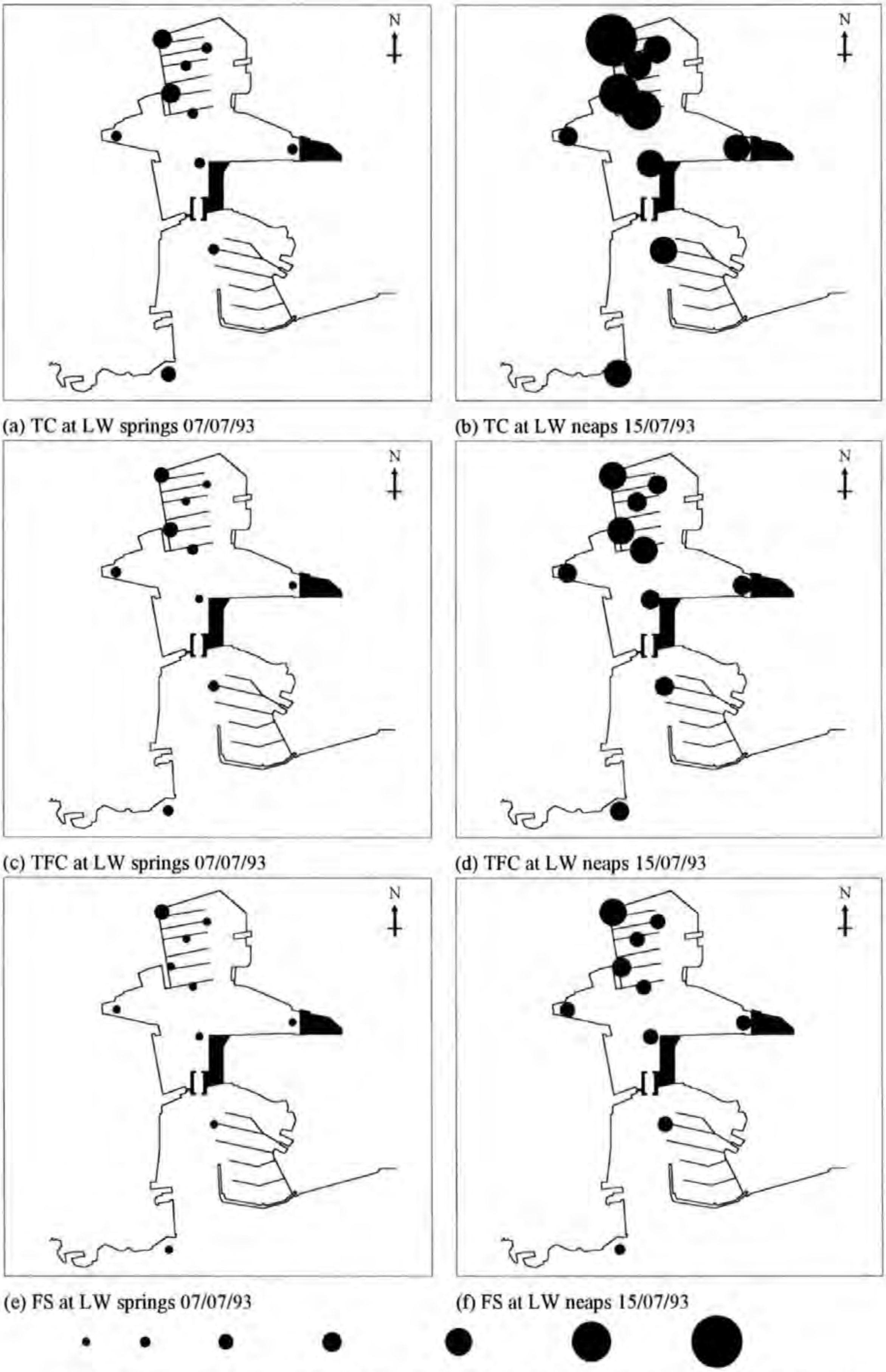


Fig. 4.30. Time series of neap tide differences in faecal indicator bacteria counts between inside harbour (mean of SH1 to SH3) and outside harbour (mean of SH4 and SH5) for (a) TC, (b) TFC and (c) FS. Red lines represent surface samples and blue lines represent bottom samples.

Neaps 15/07/93 (Figs. 4.31b, d and f). Counts of all bacteria were much higher at LW one week later because of greater stormwater inputs to the system. Counts at Fishers Nose outfall were up to twice the EC imperative limits (TC 20,000, TFC 3,000 and FS 440 cfu (100 ml)$^{-1}$) illustrating the effect of greater surface runoff upon foul water flows.



(a) TC at LW springs 07/07/93

(b) TC at LW neaps 15/07/93

(c) TFC at LW springs 07/07/93

(d) TFC at LW neaps 15/07/93

(e) FS at LW springs 07/07/93

(f) FS at LW neaps 15/07/93

<100; 100-500; 500-2,000; 2,000-10,000; 10,000-50,000; 50,000-100,000; >100,000 cfu (100 ml)$^{-1}$

Fig. 4.31. Results of the latter half of higher resolution bacteria surveys conducted in July 1993.

However, counts were even greater inside the harbour. The TC counts were at least three times the EC imperative limit at all but one site (site 5) whilst all TFC counts were above the EC imperative limit and all FS counts were above the EC guideline limit (there is no imperative limit for FS). At sites 4 and 6 near Sutton Pier and the storm sewer, counts of TC and TFC were all over five times the EC imperative limit, whilst FS counts were between 10 and 100 times the EC guideline limit. However, the highest counts inside the harbour and of the present study were observed at site 1 (TC >100,000, TFC 30,000 and FS 12,000 cfu $(100\ ml)^{-1}$) representing ten and fifteen times the EC imperative limits for TC and TFC. The fact that all concurrent bottom counts of TC and TFC inside the harbour were below the EC imperative limits precludes the sediments as the source of the high surface counts. It is also unlikely that such elevated populations could have originated from outside the harbour on the previous flood tide (between 6 h and 12 h earlier) given the bacterial mortality rate in the marine environment of up to 0.79 d^{-1} (Servais *et al.*, 1985). Therefore it must be concluded that the sewage contamination came either from foul water mixed with surface runoff or directly from yacht heads in the marina. The latter explanation is less likely since the marina is populated to varying extents throughout the year and this phenomenon would have been equally evident (if not more so) during periods of fine weather.

The problem of sewage inputs to Sutton Harbour from Fishers Nose and other outfalls along the waterfront should be curtailed by South West Water's Clean Sweep programme (SWW, 1990) implemented under the EC Bathing Water Directive. The Plymouth Area Sewage Scheme, to include tertiary (UV) treatment of the West Hoe and Fishers Nose outfalls, forms part of a £1 billion SWW investment and should be commissioned in 1998. However, it has been shown that these are not the only sources of sewage contamination to the harbour. The pipe below Sutton Pier was often seen to discharge considerable quantities of waste water of unknown origin when revealed at LW before impoundment. On several occasions during periods of heavy rainfall, water was seen to run down the harbour walls (NW corner near site 1) from spaces between the stone blocks. It is conceivable that this water emanated from a damaged combined sewer and would explain the observations in this region of the harbour. The microbiological water quality of the impounded harbour may therefore deteriorate during periods of persistent rainfall throughout the year and the conditions will be exacerbated by low water temperature, low salinity and low insolation (Mitchell and Chamberlin, 1978). It should also be noted that the survival of pathogenic enteroviruses is known to be greater than enteric bacteria in the marine environment (Fattal *et al.*, 1983; Borrego *et al.*, 1987) and therefore the potential risks to human health may become more acute under these conditions.

The results from an 'axial' profiling survey of the harbour and approaches on 27/07/93, conducted jointly with Parker (1993), are given in Fig. 4.32. Only 4 mm of rain fell in the

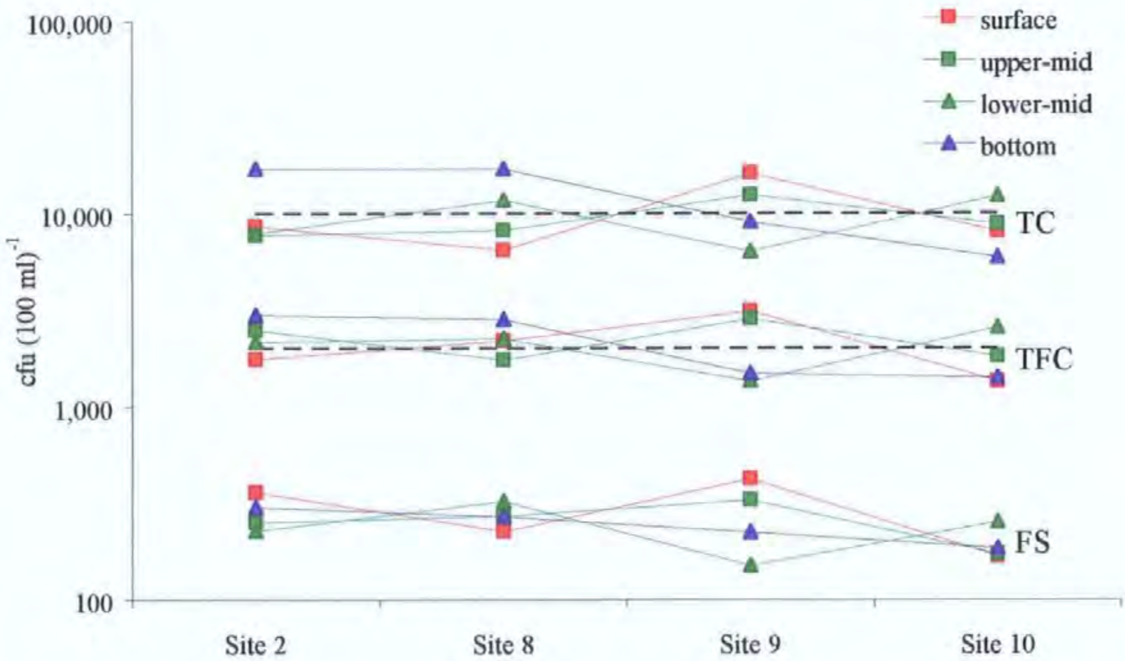


Fig. 4.32. Faecal indicator bacteria counts (TC, TFC and FS) from the axial survey on 27/07/93 at HW. Site numbers as shown in Fig. 2.6. Dashed lines represent EC imperative values for TC and TFC.

preceding 4 d period, precluding the activity of stormwater overflows. The bacterial population was subjected to minimal UV stress from only 1.6 h of direct insolation during the day. The conditions were therefore considered ideal to study axial dispersion from south to north through the unrestricted harbour entrance. Bottom counts of all indicator bacteria increased sequentially from site 10 at Fishers Nose through to site 2 in Sutton Marina concordant with the tidal shunting mechanism. Surface counts increased from site 10 to site 9 south of the harbour mouth but decreased from site 9 to site 8 north of the harbour mouth. Then, counts of TC and FS increased whilst counts of TFC decreased from site 8 to site 2. This distribution of bacterial counts suggests a greater settling of bacteria associated with suspended particles inside the harbour (presumably due to lower water velocities) and is corroborated by surface counts greater than bottom counts outside the harbour and the opposite condition inside. An effort to support this hypothesis with tentative microbiological determinations of the surface sediments (Parker, 1993) was inconclusive, revealing FS counts of 4, 4 and 7 cfu g^{-1} at sites 2, 8 and 9 and zero counts of TC or TFC throughout. It would appear that final settling to the sediments is therefore slower than mortality rates for faecal bacteria in the marine environment. The literature partially supports this view with values for coliform-specific sedimentation velocities of 1.4 m d^{-1} in Lake Onondaga, N.Y. (Auer and Niehaus, 1993) and 0.39 to 1.8 m d^{-1} in the Clyde Estuary (Milne *et al.*, 1986) and bacterial mortality rates of 0.24 to 0.79 d^{-1} (Servais *et al.*, 1985) up to a maximum of 0.14 h^{-1} (Garcia-Lara *et al.*, 1991) in the Belgian coastal zone. However, the sediments are still considered an important sink and secondary source of faecal bacteria and pathogens (Martinez-Manzanares *et al.*, 1992), particularly where the nutrient concentrations in the sediments are greater than in the overlying water (Chan *et al.*, 1979). Investigation of the role played by the sediments in the fate of faecal indicator bacteria was not specified in the aims of the present study, but it is acknowledged that it may be of greater significance to similar studies in the future.

4.5 Trace Metals

The analysis of particulate trace metals was hindered throughout the study by an inability to collect sufficient suspended particulate matter (SPM) to raise the concentration of each determinand above that detected in the combined reagent/filter blanks, despite scrupulous sample handling, preparation and analysis under clean conditions. It was hypothesised that the SPM must have high metal loadings (Millward, 1990) to account for the elevated trace metal concentrations observed in the sediments (Section *5.3.2*). However, the surveys conducted in May 1990 and November 1991 revealed SPM metal concentrations that were below the detection limits. Samples from May 1992 and November 1992, selected according to weight of SPM, contained only detectable concentrations of Zn. These were only slightly greater than the blanks and were therefore underestimated. Selected analyses of samples from the May 1993 and November 1993 surveys were more successful, and these results are summarised in Table 4.12. The concentrations of Cu, Hg, Pb and Zn were in good agreement with those in the <63 μm fraction of the sediments (Section *5.3.2*) and supported the hypothesis of a source of trace metals to the fine sediment fraction. The high concentrations of Cd coupled with the non-detection of Zn in May 1993 are symptomatic of the low mass of SPM digested, leading to high blank to sample concentration ratios and exaggerated interference in analysis. Insufficient SPM was collected in May 1994 or November 1994 to allow useful analyses of metal content.

Table 4.12. Results of selected particulate trace metal (HNO_3) analyses from May 1993 and November 1993. Mean ± 1 SD; n = 4.

Date	Location	Particulate trace metal, μg g^{-1}				
		Cd	Cu	Hg	Pb	Zn
May 1993	Inside	18.5 ± 5	155 ± 76	0.9 ± 0.3	302 ± 216	ND
	Outside	12.0 ± 1.4	164 ± 35	0.8 ± 0.6	206 ± 43	ND
November 1993	Inside	76 ± 62	161 ± 90	1.5 ± 0.7	194 ± 28	168 ± 60
	Outside	71 ± 9	195 ± 99	1.1 ± 0.3	171 ± 87	158 ± 5

ND - below limit of detection.

The solution to the problem of low SPM concentrations is to filter a greater volume of water until sufficient SPM is collected, but this must be optimised against the exponential increase in time required for filtration of greater volumes. With respect to trace metal determinations, this can only be effected using techniques currently employed in shelf sea studies, using large volume Go-Flo sampling bottles and pressure filtration through Teflon filter presses (Williams *et al.*, 1994) under ultra-clean shipboard conditions (Morley *et al.*, 1988). Clearly these techniques would be unwieldy and inappropriate when translated to inshore surveys and were beyond the scope of the present study.

Table 4.13 contains estimates of the percentage mass transport of trace metals in the particulate phase inside the harbour. It is evident that Hg is transported mainly in the particulate phase. This would be expected because Hg has a high particle reactivity due to organic complexation and metal hydroxide co-precipitation (Olsen *et al.*, 1982). Pb is also a particle-reactive metal and shows a relatively high transport in the particulate phase. The remainder of the metals studied have relatively low particle reactivity and were seen to be transported mainly in the dissolved phase. These observations have implications for the relative accumulation of trace metals in the harbour sediments. The tracer studies of water renewal in the harbour (Section *3.1.2*) suggested that pollutants entering the harbour in the dissolved phase are effectively flushed out in the surface waters due to low vertical exchange, whereas pollutants in the particulate phase settle to the bottom waters where flushing is less effective and are ultimately deposited to the sediments. This would suggest that Hg and Pb are transported to the sediments at a greater relative rate than Cd, Cu or Zn.

Table 4.13. Estimated percentage transport of metals in the particulate phase from estimated particulate trace metal concentrations and mean dissolved metal concentrations.

Trace metal	Particulate metal ($\mu g\ l^{-1}$)[a]	Dissolved metal ($\mu g\ l^{-1}$)[b]	Particulate mass transport (%)
Cd	0.092	0.32	22
Cu	0.79	5.1	13
Hg	6×10^{-3}	0.37×10^{-3}	94
Pb	1.24	1.7	42
Zn	0.84	15.4	5

[a]Assuming a mean SPM concentration of 5 mg l^{-1}; [b]Mean from Table 4.14.

In the absence of extensive particulate trace metal data, it is difficult to make a more detailed assessment of the fate of trace metals in the water column of Sutton Harbour. Moreover, Balls (1989) concluded that the constant throughput of water and particles in the coastal environment blurred any signal when monitoring trends in dissolved metal concentrations. Meaningful data can only be obtained for metals with low particle reactivity, such as Cd. Therefore, the discussion of dissolved metal concentrations has been restricted to an evaluation using current EQS values as a frame of reference. Table 4.14 contains an overall summary of the dissolved metal concentrations inside and outside Sutton Harbour, together with the relevant EQS values and additional data from the Plym and Tamar Estuaries. It is immediately evident that mean concentrations of Cd, Pb and Zn were considerably lower than the EQS values both inside and outside the harbour during the present study. Concentrations of Hg were consistently below the LOD of 0.37 ng l^{-1} and well below the EQS of 0.3 $\mu g\ l^{-1}$. This was not surprising since Hg shows strong

Table 4.14. Comparison of dissolved heavy metal concentrations in Sutton Harbour between May 1990 and November 1994 with adjacent estuaries and Environmental Quality Standard (EQS) values.

Trace metal	EQS $\mu g\ l^{-1}$	Dissolved concentration, $\mu g\ l^{-1}$ (mean ± 1 SD, min - max or <max) Inside harbour (SH1 to SH3)[f]	Outside harbour (SH4)[g]	Plym Estuary -Laira Bridge	Tamar Estuary ($S > 20 \times 10^{-3}$)
Cd	2.5[a]	0.32 ± 0.75	0.55 ± 1.21	0.14 ± 0.02[b]	0.04 - 0.12[d]
Cu	5[a]	5.1 ± 9.6	3.4 ± 5.9	3.5 ± 2.0[b]	0.5 - 4.0[e]
Hg	0.3[a]	$<0.37 \times 10^{-3}$	$<0.37 \times 10^{-3}$	<0.03[c]	<0.03[c]
Pb	25[a]	1.7 ± 6.9	1.1 ± 2.1	<5.0[b]	2.0[c]
Zn	40[a]	15.4 ± 19.8	14.7 ± 31.5	9.3 ± 2.6[b]	5.0 - 12.0[e]

[a]Campbell *et al.*, 1985; [b]Millward, 1993; [c]SWW, 1978; [d]Morris *et al.*, 1986; [e]Ackroyd *et al.*, 1986; [f]n = 192; [g]n = 56.

removal from the dissolved phase in estuaries (Cossa *et al.*, 1988). Mean concentrations of dissolved Cu were marginally above the EQS value of 5.0 $\mu g\ l^{-1}$ inside the harbour, but remained below this outside. Enhancement may be linked to the proliferation of copper-based antifouling paints in the wake of legislation against the use of TBT, and corroborates the findings of Claisse and Alzieu (1993) in the Bay of Arcachon (SW France), a semi-enclosed bay with a high density of small craft in marinas and at moorings. Concentrations of Cd, Cu and Zn were slightly greater than in the adjacent estuaries, whilst those of Hg and Pb were reasonably similar. From the current data, it does not appear that Sutton Harbour water is seriously affected by trace metals or that it is a significant source of dissolved trace metals to adjacent systems.

Table 4.15 shows the results of correlation analysis between dissolved concentrations of Cd, Cu, Hg. Pb, Zn, and salinity. With the exception of Pb, the dominant source of trace metals in coastal waters is thought to be riverine (Balls, 1989). However, correlation of salinity with the five dissolved species studied did not reveal any significant explanation of the variations in metal concentrations. Furthermore, there are many other sources of trace metals in heavily urbanised areas, such as domestic sewage inputs, industrial discharges and commercial and recreational activities, all of which may be important. The strongest significant correlations inside and outside the harbour were between Cu and Zn. It is feasible that a contribution to this correlation came from the concurrent use of Cu-based antifouling paints and dissolution of Zn sacrificial anodes on the high density of small craft using the marinas inside and immediately outside the harbour. No other significant strong correlations ($r > 0.5$) existed between the dissolved species of the metals studied, suggesting that a wide variety of diffuse sources were responsible.

Table 4.15. Matrix of correlation coefficients (r) between salinity (S), dissolved Cd, Cu, Hg, Pb and Zn in all samples (surface and bottom; spring and neap) inside and outside Sutton Harbour.

Inside harbour (SH1 to SH3) from May 1990 to November 1994 (n = 192).						Outside harbour (SH4) from November 1991 to November 1994 (n = 56)					
	Cd	**Cu**	**Pb**	**Hg**	**Zn**		**Cd**	**Cu**	**Pb**	**Hg**	**Zn**
S	NS	NS	NS	NS	NS	**S**	NS	NS	NS	NS	NS
Cd		0.39[a]	NS	NS	0.22[a]	**Cd**		0.39[a]	0.30[b]	NS	0.30[b]
Cu			NS	0.20[a]	0.53[a]	**Cu**			0.38[a]	NS	0.87[a]
Pb				NS	NS	**Pb**				NS	0.49[a]
Hg					0.22[a]	**Hg**					NS

NS - not significant at $p < 0.05$; $^{a}p < 0.005$; $^{b}p < 0.025$.

The variation in dissolved metal concentrations between surface and bottom and springs and neaps was examined in further detail. Mean concentrations are shown on logarithmic scales in Figs. 4.33a to d for Cu, Pb and Zn and in Figs. 4.34a to d for Cd and Hg. It is evident that Cu, Zn and Cd were the only metals of which concentrations exceeded the EQS values at any time during the present study. These excesses were all observed during the surveys in May 1992 and November 1992 (Figs. 4.33a, b and c and Fig. 4.34c), the analyses of which were performed using ICP-MS. The drawbacks of this technique have already been discussed (Section *2.3.1*), casting doubts upon the precision of these data relative to the samples from May 1993 onwards. However, it should also be noted that the greatest physical disturbance to the sediments was observed during the period from May 1992 to November 1992, with dredging and blasting at the harbour mouth and the creation of a dry-dock for the lock chamber construction in the eastern arm of the harbour. Water pumped from behind the coffer dam into the harbour near SH3 to maintain the dry-dock was sampled by the NRA TWIU on 30/07/92 (Sharp, pers. comm.), and high dissolved metal concentrations were observed (40 μg-Cu l^{-1}; 140 μg-Pb l^{-1}; 520 μg-Zn l^{-1}). It is conceivable therefore that metal-rich porewaters and sediments were re-entrained and resuspended throughout this period, thus increasing the dissolved metal concentrations in the water column.

No consistent differences were observed between inside and outside the harbour or between springs and neaps from May 1993 onwards, although a fractional enhancement in surface over bottom concentrations of Cu, Pb and Zn was noted at springs and neaps, suggesting that the surface inputs (aeolian Pb, riverine, sewage/industrial discharges, commercial/recreational activities) were more significant than resuspension and porewater entrainment processes. There were no consistent differences in dissolved metal concentrations before and after impoundment either, and therefore it must be concluded that impoundment had no effect. However, future episodes of benthic anoxis may enhance

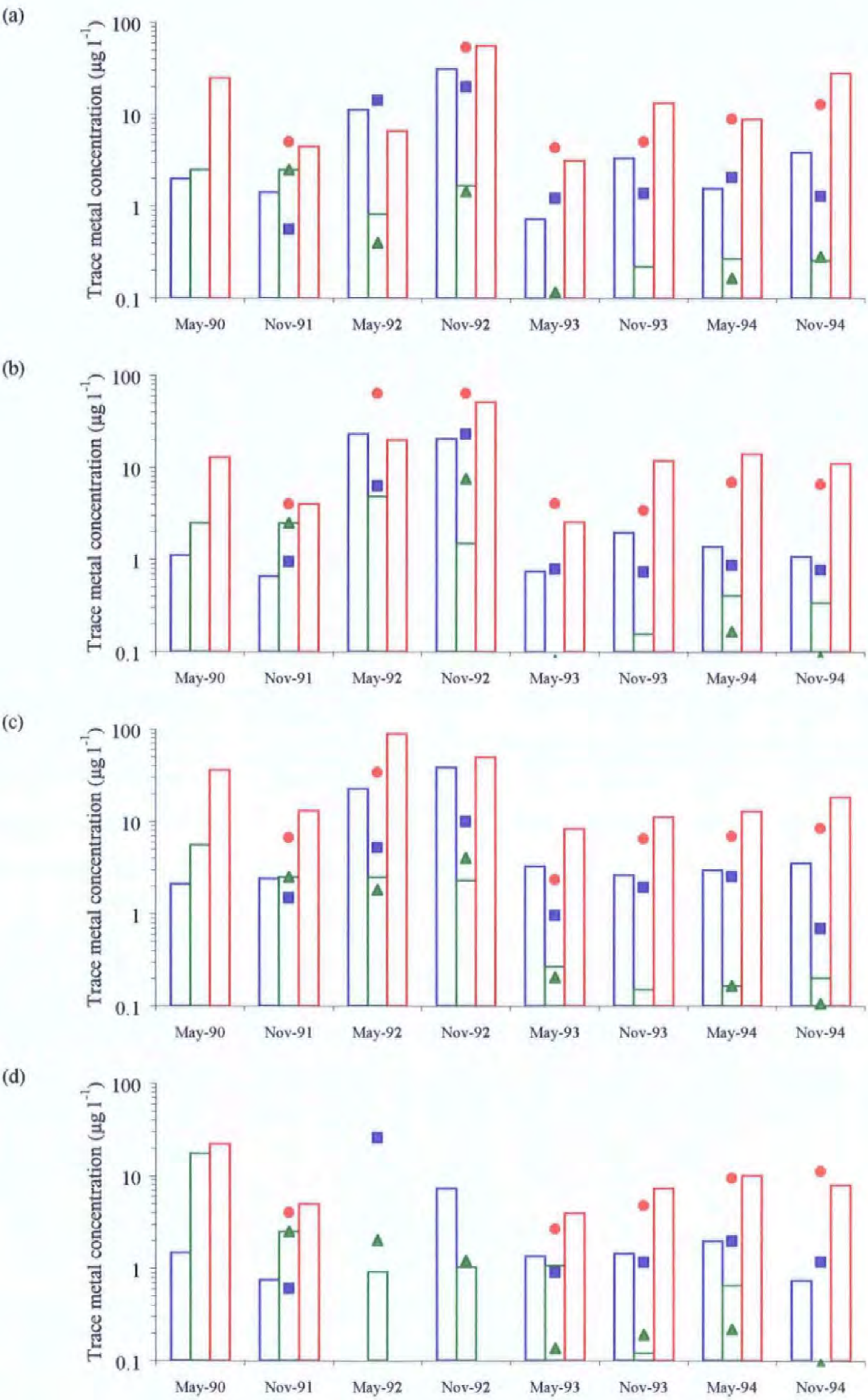


Fig. 4.33. Dissolved trace metal concentrations at (a) surface and (b) bottom at springs and (c) surface and (d) bottom at neaps. Bars represent mean of SH1 to SH3 and symbols represent SH4. Cu is represented by blue bars and squares, Pb by green bars and triangles and Zn by red bars and circles.

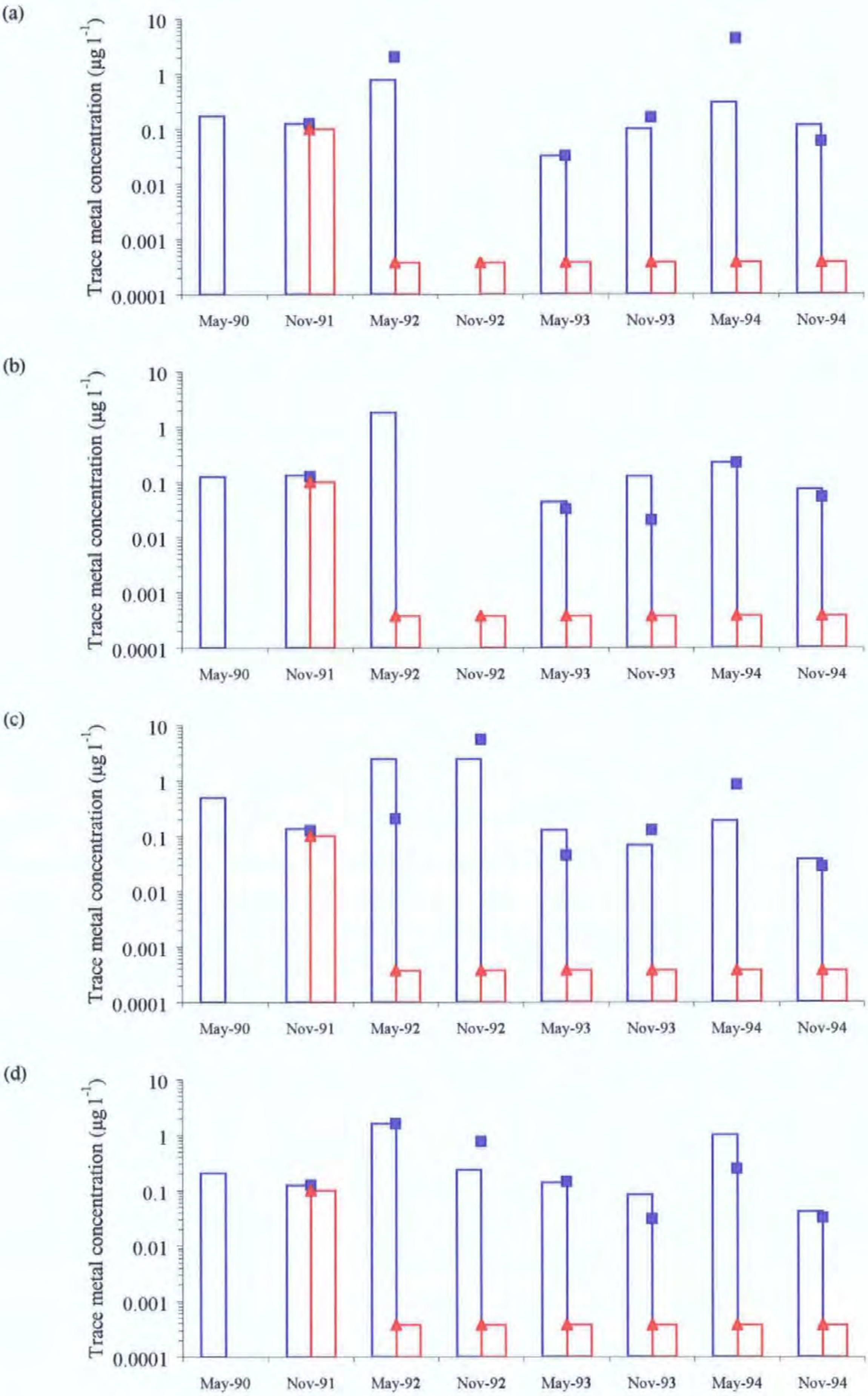


Fig. 4.34. Dissolved trace metal concentrations at (a) surface and (b) bottom at springs and (c) surface and (d) bottom at neaps. Bars represent mean of SH1 to SH3 and symbols represent SH4. Cd is represented by blue bars and squares and Hg by red bars and triangles.

the reducing conditions in the sediments, allowing dissolved metals to diffuse along a concentration gradient into the overlying water (Olsen *et al.*, 1982). In that event, reduced flushing may lead to a build-up of dissolved metal concentrations that exceed current EQS values and the harbour may become a source of dissolved metals to the adjoining areas. The trace metal status of the harbour sediments is fully discussed in Section *5.3.2*. Over the entire study period from May 1990 to November 1994, only one clear trend emerged: a general decrease in dissolved Pb concentrations both inside and outside the harbour. This might be indicative of the decline in use of alkyl-Pb additives in petrol, resulting in decreased secondary Pb inputs from the atmosphere and road runoff and reduced primary inputs from vessel exhausts, or from spillage through better management of refuelling operations.

4.6 Visual Aesthetics

The visual surveys conducted between May 1992 and November 1994 revealed a wide variety of issues that are important to the management of water quality inside the harbour. The main findings over three years and 32 surveys are summarised in Fig. 4.35 which depicts the circumstances before and after impoundment of the harbour. Three main phenomena affect the visual characteristics of the harbour waters: direct and indirect inputs of land-based litter from the public and commercial areas; ingress of litter from outside through the harbour entrance; and commercial and recreational activities of harbour users. These categories will be discussed in turn.

Litter from the quayside. The most important factor controlling distribution and density of floating litter in the harbour was (not surprisingly) wind direction. Litter tends to accumulate in several key areas in the north, south, east or west of the harbour, such that similar amounts gave varying impressions of cleanliness from the shore. Local knowledge of areas of litter concentration was used to good effect after impoundment by Sutton Harbour Company in order to periodically remove large volumes of floating litter. However, this effort could be reduced, by employing public awareness notices in order to discourage direct inputs of litter. It may be that indirect inputs are more significant, a situation that could perhaps be remedied by installing more bins around the harbour perimeter and by ensuring that during busy periods existing bins do not overflow. Public awareness could be drawn to the results of Denyer (1993) who examined the degradation of common litter items by storing samples below the water surface in a mesh sack tied to a pontoon in the marina and periodically sub-sampled them in order to obtain dry weights. The experimental mass half-lives for each item are given in Table 4.16. The considerable longevity of many of these common items may make the public think again before littering.

(a)

(b)

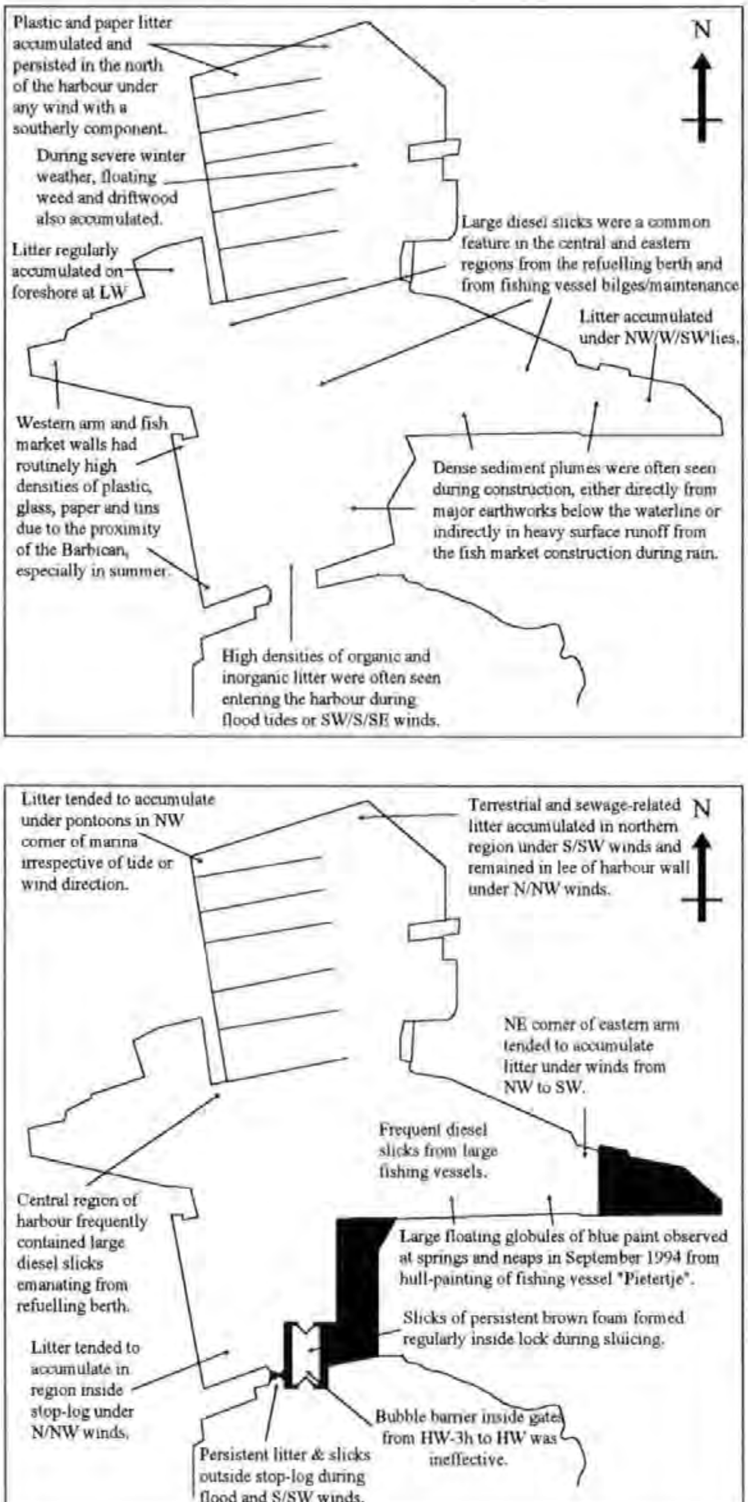


Fig. 4.35. Summary of key issues of visual aesthetics inside Sutton Harbour during (a) pre-impoundment and construction periods and (b) post-impoundment period of study, based on observations made on 32 occasions over a 3 year period.

Litter from outside the harbour. An ingress of litter through the harbour entrance was often observed before the lock was positioned and high concentrations of litter and organic slicks continue to collect outside the stop-log gate during the early flood whilst the lock remains closed. The bubble barrier that was installed across the bed of the lock chamber has only been seen in operation once during the routine surveys and appears to be

Table 4.16. Experimental mass half-lives of seven common litter items submerged in Sutton Harbour during Spring 1993 (Denyer, 1993).

Litter item	Mass half-life (days)	Observations
Apple quarters	21	Complete loss after 80 d.
Sanitary towels	24	Complete loss of absorbent lining after 30 d. Plastic backing showed little decay after 112 d.
Orange quarters	36	Two-stage decay: flesh followed by peel.
Plain card	46	Complete loss after 80 d.
Cigarette butts	72	Two-stage decay: paper/tobacco followed by filter.
Latex condoms	78	Initial resilience, body began to decay after 30 d and base ring showed little decay after 112 d.
Waxed card	120[a]	Two-stage decay: wax followed by card.

[a]Extrapolated - 50 % reduction in mass not observed after 112 days.

ineffective against the ingress of litter from HW-3h to HW when the lock is open to free flow. However, its use may still be beneficial in oxygenating the inflowing water. There would seem to be no viable technology which prevents litter from entering the harbour, although it is apparent that the presence of the lock has effectively reduced this source by half.

Litter/contamination from harbour users. Aspects of the use of the harbour as a centre for commercial and recreational seagoing activities may conflict with the visual quality. The main relevant example is the frequency and extent of the diesel slicks observed inside the harbour before and after impoundment. The main source is thought to be spillage from the fuelling berth near Sutton Pier, but slicks were frequently also seen coming from fishing vessels when refuelling and pumping bilges, particularly the larger craft regularly berthed in the eastern arm of the harbour. Although a certain amount of diesel spillage is unavoidable in a working harbour, the frequency and extent of the observations during this study suggest that more care should be taken by all parties when refuelling. Steps should also be taken to ensure that oily bilges are not routinely pumped into the harbour waters, in the first instance by polite official request to the vessel operators. Other examples are entirely avoidable: On several occasions, large fish carcasses have been seen floating in the harbour, and vessel owners have been observed emptying ashtrays and throwing fast food wrappers overboard. Most notably, during the spring and neap surveys of September 1994, the eastern arm of the harbour was covered with globules of fresh blue paint as a result of the hull-painting activities of the crew of "Pietertje". These few examples and many others illustrate a lack of care for the water quality of the harbour by those using it on a daily basis. Again, an awareness campaign should target these people in an effort to reduce the visual pollution of the harbour.

In general terms, the visual aesthetics of the harbour are deemed to have improved following the commissioning of the impoundment scheme. This has been achieved through a combination of reduced inputs *via* the harbour entrance and a periodic litter-picking operation by Sutton Harbour Company. However, further improvements are possible and these can be achieved through greater public awareness and greater vigilance.

4.7 Summary

The main source of TON and the major source of orthophosphate to the harbour waters is the River Plym, but there is strong evidence to suggest that the main source of ammonium and a minor source of orthophosphate are the sewage outfalls outside and the sediment porewaters inside. The sediment porewaters appear to have become a more significant source after impoundment. Distinct seasonality was exhibited in TON and orthophosphate concentrations concordant with variations in river flow and primary production, causing maxima in winter and minima in summer perturbed only by intermittent meteorological events and not by the impoundment of the harbour. The surprising minima in winter observed during dry periods could be used effectively to flush the harbour with low nutrient water, when the preceding LW exhibits sufficiently high salinities outside the harbour. Ammonium and UIA concentrations exhibited less seasonality than TON or orthophosphate, consistent with the main source being the sewage outfalls. UIA concentrations showed a possible tendency to increase inside the harbour in summer due to the increased temperatures caused by impoundment. However, all UIA concentrations were below the EQS (21 $\mu g\ l^{-1}$) and future inputs should be reduced when the Plymouth Area Sewage Scheme is commissioned.

The capability of modelling the input of contaminants from the Plym Estuary to Sutton Harbour has been demonstrated using the ECoS modelling shell. Ammonium was shown to behave conservatively in the Plym and is independent of land runoff. The refined ECoS model has the potential for use as an aid to water quality management in the harbour, predicting when and under which environmental factors the optimum conditions for beneficial harbour flushing prevail, given the availability of contemporary river flow data, and contaminant flux data from the catchment and the estuarine anthropogenic discharges.

Primary production in the summer period was particularly dependent upon the amount of direct insolation and therefore exhibited high inter-annual variability. Phytoplankton bloom events observed before impoundment were short-lived, showing almost complete decay within one week. The harbour waters appear able to sustain an intense bloom for considerably longer after impoundment, even when general conditions are not favourable for phytoplankton growth outside the harbour. The winter period in the harbour produced persistent oxygen sags that were considerably deeper after impoundment. Low BOD_5 values at this time suggested that the oxygen demand was created at the sediment surface due to the combustion of epibenthic phytoplankton deposits. The risk of eutrophication in the impounded harbour is enhanced by the mineralisation of the greater phytoplankton biomass during the winter period and will further increase under greater winter precipitation and summer insolation than was observed in 1994. The extended period of sub-oxic conditions observed between August 1993 and May 1994 is therefore likely to deepen in the impounded harbour, leading to increased accumulation of trace metals in the sediments *via* the mechanism described. This has implications for the removal and disposal of these sediments under future dredging requirements.

The major sources of faecal indicator bacteria to the harbour appear to be localised (sewage outfalls) rather than estuarine in origin. Strong correlation between TC and TFC suggested a redundancy of TC data in studies of this nature. Counts of all bacteria before impoundment were generally greater outside the harbour at LW due to sewage discharges and greater inside the harbour at HW due to the tidal shunting mechanism. These distributions were modified by the bacterial mortality factors, particularly insolation during the summer period. They were also reversed on occasions by the presence of internal sources such as stormwater overflows, yacht heads and the possibility of a damaged sewer in the NW corner of the harbour. The impoundment of the harbour has produced a considerable improvement in microbiological water quality due to the physical barrier to contaminant influx. However, the internal sources must be addressed in the run-up to commissioning of the Plymouth Area Sewage Scheme.

Particulate trace metal data, where obtained, were in good agreement with total metal concentrations in the fine sediment fraction. However, the dataset was extremely sparse due to low SPM concentrations observed throughout the study. All dissolved metal concentrations were below the respective EQS values throughout the current study apart from in May 1992 and November 1992 when some Cd, Cu and Zn concentrations exceeded these. However, this may have been due to the effects of sediment disturbance that were maximised at this time. No effects of impoundment were observed, although it is acknowledged that increased benthic anoxis could lead to diffusion of reduced metal species into the water column. This may reach a dynamic equilibrium with the processes returning oxidised species to the sediments, or may result in a net export of dissolved metals to the surrounding region. Overall, dissolved metal concentrations, particularly Hg, gave no cause for concern during the study period.

The visual aesthetics of the harbour are adjudged to have improved after impoundment, due to cleaning activities by Sutton Harbour Company and reduced ingress of litter *via* the lock. The bubble barrier was ineffective in this respect. The main issue of visual aesthetics continues to be diesel slicks, and this should be addressed through a harbour user awareness campaign. Other issues of general litter and littering should also be monitored and addressed when necessary, particularly during the summer months.

Chapter Five

Benthic Ecology

and

Sediment Heavy Metals

Chapter 5 - Benthic Ecology and Sediment Heavy Metals

The fine anoxic sediments of Sutton Harbour support a community of benthic macrofauna comprising infaunal and epifaunal primary and secondary consumers. The initial environmental assessment of Sutton Harbour undertaken in May 1990 (Houston, 1990; Millward, 1990) demonstrated non-uniformity in the community structure. Stations SH1 and SH3 were almost azoic, with respective abundances of only five and two individual cirratulids (*Tharyx marioni*) and spionids, whilst stations SH2 and SH4 were relatively rich in comparison, with 13 and 29 taxa (mainly cirratulids) respectively. Species richness varied throughout the harbour and was lower than outside. Therefore, it is important to note that the aim of this part of the study was to use changes in species diversity and community similarity to monitor the effects of the construction and operation of a harbour impoundment scheme. In this particular case, the benthic macrofauna were more valuable as indicators of environmental change (Bilyard, 1987) than as components of the food chain.

The data obtained in May 1990 have been included for completeness where possible, but it should be noted that they are not strictly comparable to the present study, owing to slight differences in site location. It must be noted at the outset that the samples collected during May 1990 and November 1991 were obtained using different techniques from those used from May 1992 onwards (Section *2.5.1*). All data were normalised to 1 m^2, an approach deemed satisfactory in correcting abundance. However, a question of differing species composition arises from the depth of sediment sampled and this point is worthy of further discussion. The samples in May 1990 were obtained by Day grab, from which five replicate cores of area 2.5×10^{-3} m^2 were extracted to a depth of 10 cm. The samples collected in November 1991 were obtained by diver, removing an area of 0.4 m x 0.4 m of sediment to a depth of *circa* 3 cm. The May 1992 to November 1994 samples were recovered using a spring-loaded Shipek grab, with a semi-cylindrical bucket, that samples an area of 0.2 m x 0.2 m to a central depth of 10 cm. Thus, in May 1990 an equal area of sediment was sampled between 0 cm and 10 cm depth, but in November 1991 an equal area was sampled between 0 cm and 3 cm. From May 1992 onwards an unequal area was sampled, ranging from a maximum at 0 cm to a minimum at 10 cm. It would appear that a differing species composition might have been sampled by the three techniques if the sediments were inhabited throughout this 10 cm depth. However, from redox profiles recorded during May 1990 (Table 5.1), it is evident that the sediments were anoxic below a thin surface layer of recent sediment, and visual inspection confirmed that they remained so throughout the present study. It is unlikely that infauna would penetrate these sediments to a depth greater than between 2 cm and 3 cm. It is therefore concluded that the three sampling techniques were sufficiently comparable in the context of a long-term, multi-participant ecological study.

Table 5.1: Redox potentials, Eh (mV), for the top 5 cm of Sutton Harbour sediments in May 1990 (after Houston, 1990).

Sediment depth	Redox potential, Eh (mV)			
(cm)	SH1	SH2	SH3	SH4
1	-18	-19	-146	-3
2	-30	-46	-182	-27
3	-34	-70	-199	-38
4	-50	-78	-204	-57
5	-67	-99	-213	-68

5.1 Univariate Analyses of Benthic Macrofauna

The effects of construction and impoundment upon the status of the benthic macrofauna of Suton Harbour were examined through a study of changes in species diversity. The concept of species diversity as defined by Odum (1983) has two components: (i) richness, based on the total number of species present and (ii) evenness, based on the relative abundance of species and the degree of dominance exhibited. Either or both of these components may change with time if an established community is subjected to anthropogenic stress from physical disturbance or pollution.

5.1.1 Analytical methodology

The two components of species diversity were examined using graphical techniques (species-numbers plots and dominance-diversity curves) and the following species diversity indices: (sample) species richness index (Margalef, 1958; Odum, 1983),

$$D = \frac{S-1}{\mathrm{Log_e} N} \qquad \text{Eq. 5.1.}$$

where S = total number of species and N = total number of individuals; the Shannon-Wiener species diversity index, (Shannon and Weaver, 1949; Pielou, 1975),

$$H'e = -\sum_{i=1}^{S} n_i/N(\mathrm{Log_e}\, n_i/N) \qquad \text{Eq. 5.2.}$$

where n_i = number of individuals of i^{th} species; and Pielou's evenness index, (Pielou, 1966 and 1975),

$$J' = \frac{H'e}{\mathrm{Log_e} S} \qquad \text{Eq. 5.3.}$$

The species richness index (Eq. 5.1) yields a zero value if only one species is present, such that a larger number indicates a greater species richness. However, it does not take into account relative abundance (Spellerberg, 1991), since in two hypothetical samples consisting of five species and 100 individuals, the richness indices would be identical despite compositions varying between 20:20:20:20:20 and 96:1:1:1:1. This further information is revealed by Pielou's evenness index (Eq. 5.3), which would generate a maximum value (+1) for the first sample where there is equitability between the species present and a minimum score (0) for the second sample where one species is clearly dominant. The components of diversity are expressed in combination in the general species diversity or Shannon-Wiener index (Eq. 5.2), where the highest value would be returned if each individual was a member of a different species and the lowest if all individuals belonged to the same species. In order to give meaning to the values obtained for this index, the theoretical maximum and minimum values were calculated for the harbour (Table 5.2) using the highest and lowest species numbers, S, and the highest abundance per m^2, N, of all replicates from May 1990 to November 1994.

Table 5.2. Theoretical maximum and minimum values of the Shannon-Wiener diversity index in Sutton Harbour.

Parameter	Value
Maximum N for all replicates	301,000
Maximum S for all replicates	41
Minimum S for all replicates	2
Theoretical maximum H'e	3.71
Theoretical minimum H'e	4.5×10^{-5}

5.1.2 Variation in species diversity (species richness and species evenness)

The five main taxa (species where possible, family where further identification was deemed ambiguous) from pooled replicates at each site are shown in Table 5.3 in terms of percentage abundance per m^2. It is evident that in most cases, cirratulid worms were the dominant members of the benthic fauna, comprising up to 96 % abundance. Where these were not so numerous (particularly in May 1992), *Ophryotrocha* spp. was the main taxon and was otherwise generally coexistent with the cirratulids. However, after May 1993, *Ophryotrocha* spp. disappeared from all sites, along with tubificids and *Capitella capitata*. Presence of *C. capitata* is directly linked to organic content of the sediments (Pearson and Rosenberg, 1978; Chareonpanich *et al.*, 1993). Tubificids are also detrital feeders, so that their absence may signify a reduction in the organic content of the harbour sediments (although absence from SH4 might suggest that this was a more general feature). The reappearance of *Tubificoides spp.* at SH1 in November 1994 may further indicate a return to organic rich sediment conditions after impoundment. *Nephtys hombergi* was present in modest numbers at almost all sites throughout the study period but was not the most numerous species, even in the absence of cirratulids.

The species-abundance plot in Fig. 5.1 indicates the existence of a spatial and temporal spread in species richness throughout the harbour system. Several trends emerge. Before construction in May 1990 and November 1991 the spread was maximised with abundance

Table 5.3. Table of top five ranked taxa (percentage abundance / m^2) from May 1990 to November 1994.

SH1		SH2		SH3		SH4	
			May 1990				
Cirratulidae spp.	80.0	Cirratulidae spp.	52.4	Cirratulidae spp.	50.0	Cirratulidae spp.	42.6
Spionidae spp.	100.0	*Ophryotrocha* spp.	97.4	Spionidae spp.	100.0	*Ophryotrocha* spp.	85.0
		Tubificoides swirencoides	98.4			*Tubificoides benedini*	92.2
		Nephtys hombergi	98.8			*Tubificoides swirencoides*	95.6
		Sphaerosyllis tetralix	99.2			*Melinna palmata*	96.3
			November 1991				
Cirratulidae spp.	38.5	Cirratulidae spp.	83.7	Cirratulidae spp.	64.1	Cirratulidae spp.	73.8
Capitella capitata	55.8	*Tubificoides* sp.	90.1	*Capitella capitata*	78.3	*Tubificoides* sp.	86.7
Jassa falcata	71.2	*Exogone naidina*	92.7	*Ophryotrocha* spp.	85.6	*Ophryotrocha* spp.	89.2
Nephtys hombergi	80.8	*Trypanosyllis coelica*	93.8	*Jassa falcata*	88.6	*Tubificoides benedini*	91.5
Spionidae spp.	86.5	*Ophryotrocha* spp.	94.9	Spionidae spp.	90.7	*Cyathura carinata*	93.1
			May 1992				
Ophryotrocha spp.	58.0	*Ophryotrocha* spp.	55.4	*Ophryotrocha* spp.	85.1	*Ophryotrocha* spp.	65.6
Cirratulidae spp.	87.1	Cirratulidae spp.	74.9	Ampharetidae sp.	88.9	Spionidae spp.	91.4
Spionidae spp.	98.4	Spionidae sp.	93.6	*Nephtys caeca*	92.3	*Pariambus typicus*	94.3
Temora longicornis	99.1	*Ampelisca brevicornis*	95.6	Cirratulidae spp.	93.8	*Modiolus* sp.	96.5
Nephtys hombergi	99.4	*Exogone* sp.	96.6	*Arenicolides ecaudata*	94.8	*Retusa obtusa*	97.8
			November 1992				
Cirratulidae spp.	76.1	Cirratulidae spp.	86.9	Cirratulidae spp.	81.5	Cirratulidae spp.	71.7
Nephtys hombergi	99.1	*Lumbrineris gracilis*	92.0	*Lumbrineris gracilis*	90.2	*Nephtys hombergi*	80.3
Tubificoides sp.	100.0	*Nephtys hombergi*	95.3	*Nephtys hombergi*	98.3	*Apseudes latreillii*	85.3
		Manayunkia aestuarina	96.5	*Tubificoides* sp.	99.4	*Melinna palmata*	88.9
		Tubificoides sp.	97.6	*Manayunkia aestuarina*	100.0	*Nephtys* sp.	92.4
			May 1993				
Heteromastus filiformis	49.6	Cirratulidae spp.	27.3	*Ophryotrocha puerilis*	91.5	Cirratulidae spp.	59.1
Cirratulidae spp.	88.6	*Lumbrineris gracilis*	43.1	Cirratulidae spp.	94.6	*Ophryotrocha puerilis*	71.7
Nephtys hombergi	92.3	*Ophryotrocha puerilis*	58.3	*Capitella capitata*	95.9	*Heteromastus filiformis*	79.9
Capitella sp. indet.	95.1	*Heteromastus filiformis*	67.1	*Heteromastus filiformis*	97.1	*Nephtys hombergi*	83.4
Capitella capitata	96.7	*Nephtys hombergi*	75.9	Ostracoda indet.	97.8	*Exogone gemmifera*	86.9
			November 1993				
Cirratulidae spp.	95.8	Cirratulidae spp.	80.8	Cirratulidae spp.	67.2	Cirratulidae spp.	95.0
Melinna palmata	97.7	*Melinna palmata*	89.8	*Lumbrineris gracilis*	89.3	*Melinna palmata*	98.4
Nephtys hombergi	99.6	*Nephtys hombergi*	96.2	*Nephtys hombergi*	95.9	*Nephtys hombergi*	99.3
Mytilus edulis	99.8	*Lumbrineris gracilis*	98.5	*Melinna palmata*	99.2	*Hydrobia* sp.	99.7
Retusa obtusa	100.0	*Retusa obtusa*	99.4	*Nereis diversicolor*	100.0	*Modiolus* sp.	99.8
			May 1994				
Cirratulidae spp.	90.8	Cirratulidae spp.	79.1	*Lumbrineris gracilis*	78.7	Cirratulidae spp.	93.4
Lumbrineris gracilis	98.7	*Lumbrineris gracilis*	88.3	Cirratulidae spp.	96.7	*Melinna palmata*	97.2
Melinna palmata	99.5	*Melinna palmata*	94.7	Spionidae spp.	99.0	*Nephtys hombergi*	98.3
Nephtys hombergi	99.9	*Nephtys hombergi*	100.0	*Nephtys hombergi*	99.4	*Retusa obtusa*	99.2
Mytilus edulis	100.0			*Melinna palmata*	99.7	*Mytilus edulis*	99.6
			November 1994				
Cirratulidae spp.	91.2	Cirratulidae spp.	91.2	Cirratulidae spp.	93.4	Cirratulidae spp.	88.9
Nephtys hombergi	96.3	*Nephtys hombergi*	97.6	*Nereis pelagica*	96.3	*Melinna palmata*	95.3
Tubificoides sp.	100.0	*Melinna palmata*	99.7	*Lumbrineris gracilis*	98.8	*Nephtys hombergi*	97.0
		Retusa obtusa	100.0	*Nephtys hombergi*	99.5	*Mytilus edulis*	97.8
				Nereis diversicolor	100.0	*Anaitides mucosa*	98.4

and species number increasing from SH1 and SH3 through SH2 to SH4. This general trend was also evident with respect to increasing species numbers during the construction period between November 1992 and November 1993, although in May 1993 and November 1993 the abundances at SH2 were lower than at SH1 or SH3 as a possible consequence of the proximity to the construction site. The spatial species-abundance patterns differed from the general trend in May 1992 at the start of construction and in May 1994 and November 1994 after impoundment. and the spread between stations was at a minimum during these periods. In May 1992 station SH4 outside the harbour had the lowest species numbers and abundances, with a notable absence of cirratulids and *Nephtys hombergi* (Table 5.3). It is known that *N. hombergi* favours finer substrates (McLusky, 1989) but no increase in grain size was concomitant with its absence (Section *5.3.1*). In May 1994 station SH2 had the lowest species numbers and abundances. The population of annelids was similar to all other stations except for an absence of spionids, but there was an absence of all other taxa, notably *Mytilus edulis* which was present at all other stations in this period. In November 1994 station SH2 maintained similarly low species numbers and abundances whilst station SH1 had become more impoverished. Station SH4 exhibited a considerable increase in species numbers over the stations inside the harbour, possibly signalling a more rapid recovery from the effects of construction towards November 1991 conditions.

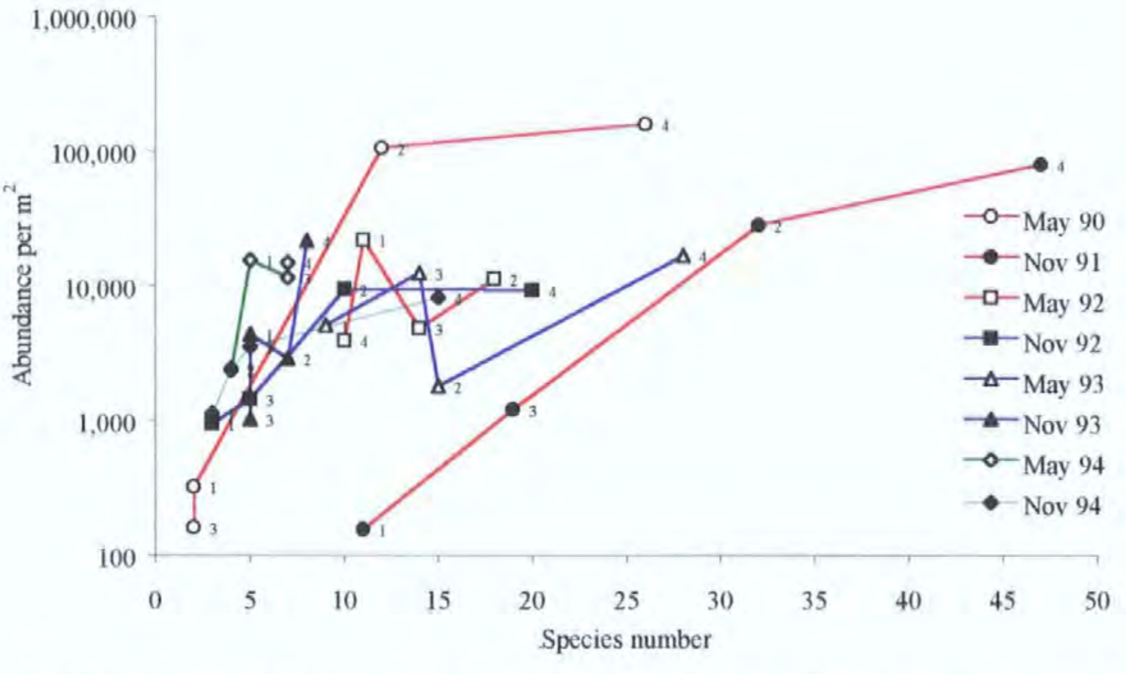


Fig. 5.1. Semi-log plot of abundance per m^2 *versus* species number for pooled samples (n = 2 to n = 5) between May 1990 and November 1994. Lines indicate increasing trend in species number for each survey period. Red lines represent pre-construction period, blue lines represent construction period and green lines represent post-construction period.

The values of Margalef's species richness index (Eq. 5.1) for the pooled replicates at each site are plotted against survey period in Fig. 5.2. A general trend of increasing species

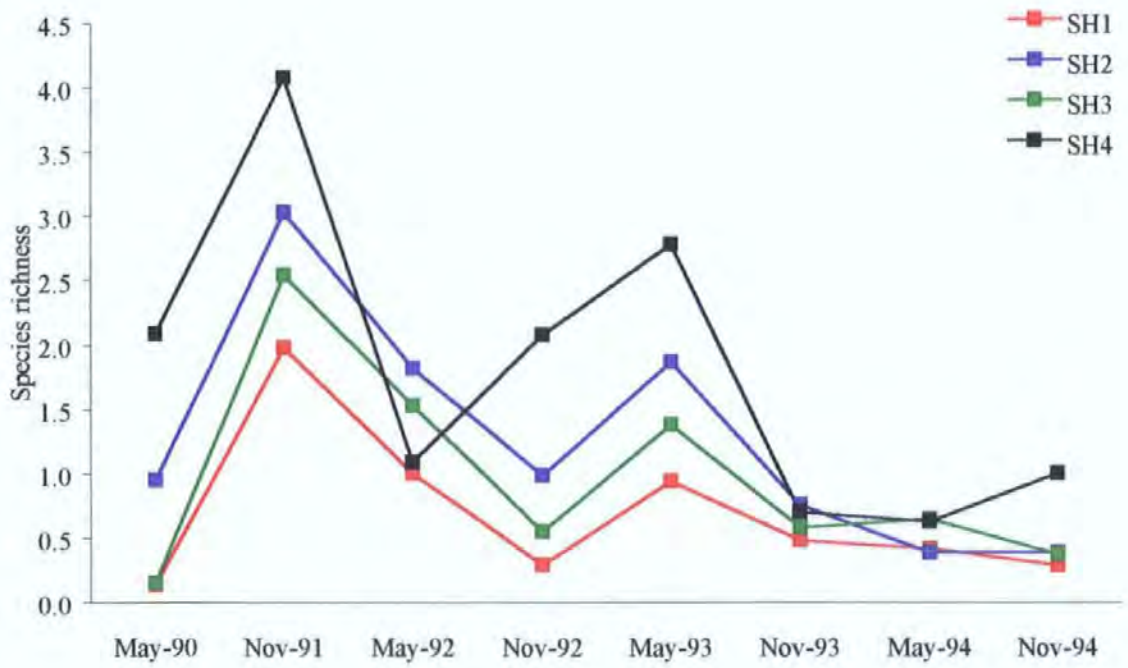


Fig. 5.2. Site-aggregated species richness (Margalef's Index) at sites SH1 to SH3 (inside harbour) and SH4 (outside harbour) from May 1990 to November 1994.

richness from SH1 to SH3 to SH2 to SH4 was evident, again with the exception of SH4 in May 1992 and SH2 in May 1994. Maximum species richness was observed at SH4 in November 1991 and minimum species richness (with the exception of SH1 and SH3 in the initial assessment in May 1990) was observed at SH1 in November 1992. Over the study period from November 1991, there was a decreasing trend in species richness both inside and outside the harbour.

The evenness of the species-abundance distribution at each site as revealed by Pielou's index (Eq. 5.3) is shown for each survey period in Fig. 5.3 and exhibits different trends to those in Fig. 5.2. The high richness at stations SH4 and SH2 in November 1991 was not accompanied by a high evenness. The reason for this was dominance by cirratulids and tubificids (Table 5.3). Conversely, the lowest richness at SH1 in November 1992 exhibited a relatively high evenness due to a significant number of *Nephtys hombergi* reducing the cirratulid dominance. The highest degree of evenness (apart from stations SH1 and SH3 in May 1990) was exhibited at station SH2 in May 1993, when *Lumbrineris gracilis, Ophryotrocha puerilis, Heteromastus filiformis* and *Nephtys hombergi* all moderated the dominance of cirratulids. The lowest degree of evenness was exhibited at station SH4 in November 1993, with a 95 % dominance by cirratulids. Overall, there were no clear trends in evenness during the study period.

The dominance-diversity curves in Fig. 5.4 are a graphical representation of diversity at each site from the pooled replicates. The higher and flatter curves indicate greater diversity and the steeper curves indicate lower diversity and greater dominance by one or a few species. Environmental stress, whether it is of a physical or pollutant nature, tends to

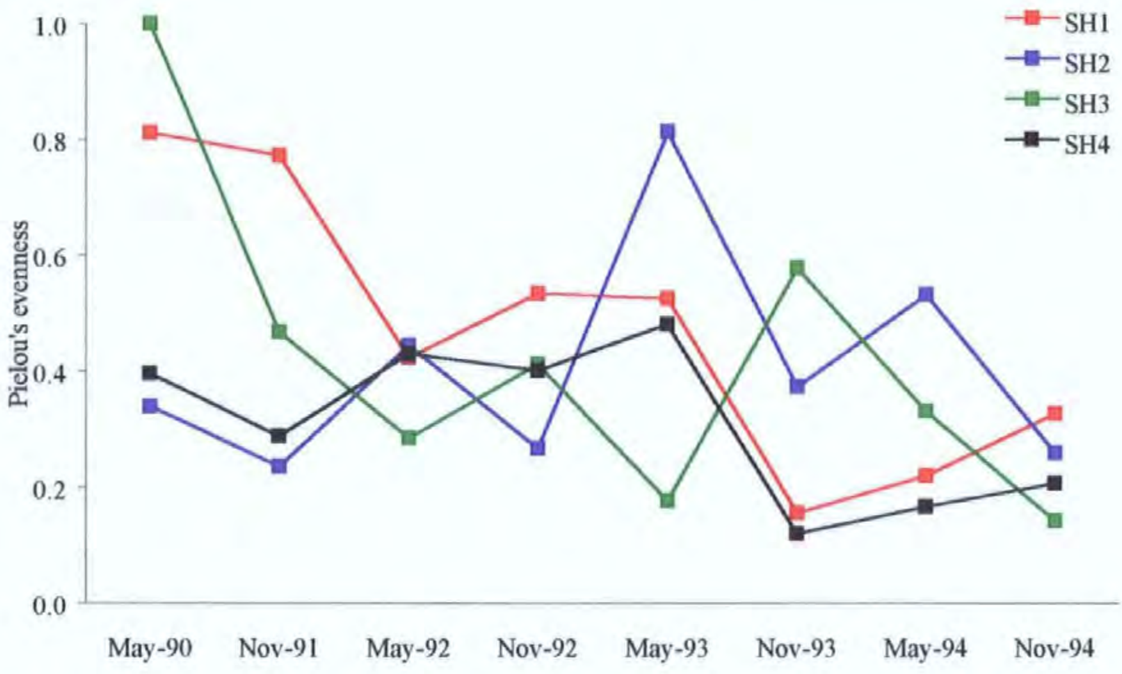


Fig. 5.3. Site-aggregated species evenness (Pielou's evenness) for each site from May 1990 to November 1994.

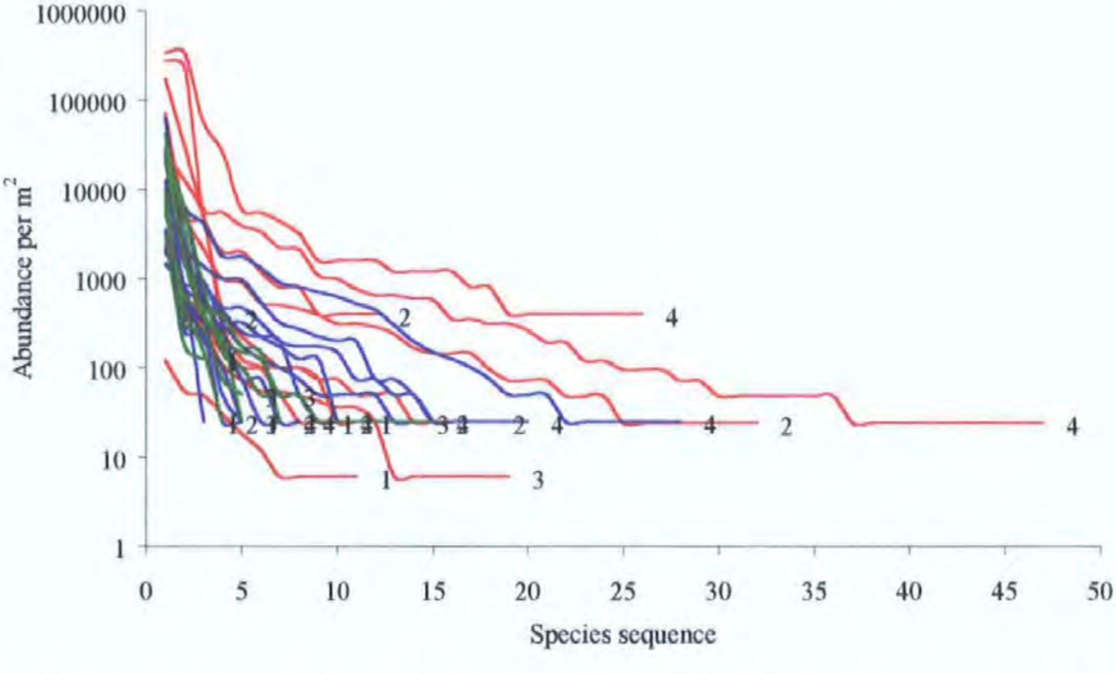


Fig. 5.4. Dominance-diversity curves for site-aggregated data. Red lines represent pre-construction period (May 1990 to May 1992), blue lines represent construction period (November 1992 to November 1993) and green lines represent post-construction period (May and November 1994). Curve tails are annotated with site number.

steepen the curves (Odum, 1983). This analysis clearly showed two trends in diversity although the spread of data was large, particularly with respect to the latter trend. Firstly, a spatial trend in diversity was evident in the pre-construction period, with station SH4 exhibiting the greatest diversity, closely followed by SH2 and then by SH3 and SH1 both exhibiting low diversity. Secondly, a temporal trend was evident, such that diversity (and the spatial difference in diversity) decreased with each successive phase from pre-construction through construction to post-impoundment.

The analysis was repeated using only May data to remove any influence of seasonality on the changes in diversity. However, on this occasion, the replicates from each site were not pooled. The results are presented in Fig. 5.5 and again several trends emerge. The spatial trend of increasing diversity from inside to outside before construction was evident as before. Temporally, there was a marked decrease between May 1990 and May 1992 before construction, a recovery of diversity in May 1993 outside the harbour during construction, then a general decrease throughout following impoundment in May 1994.

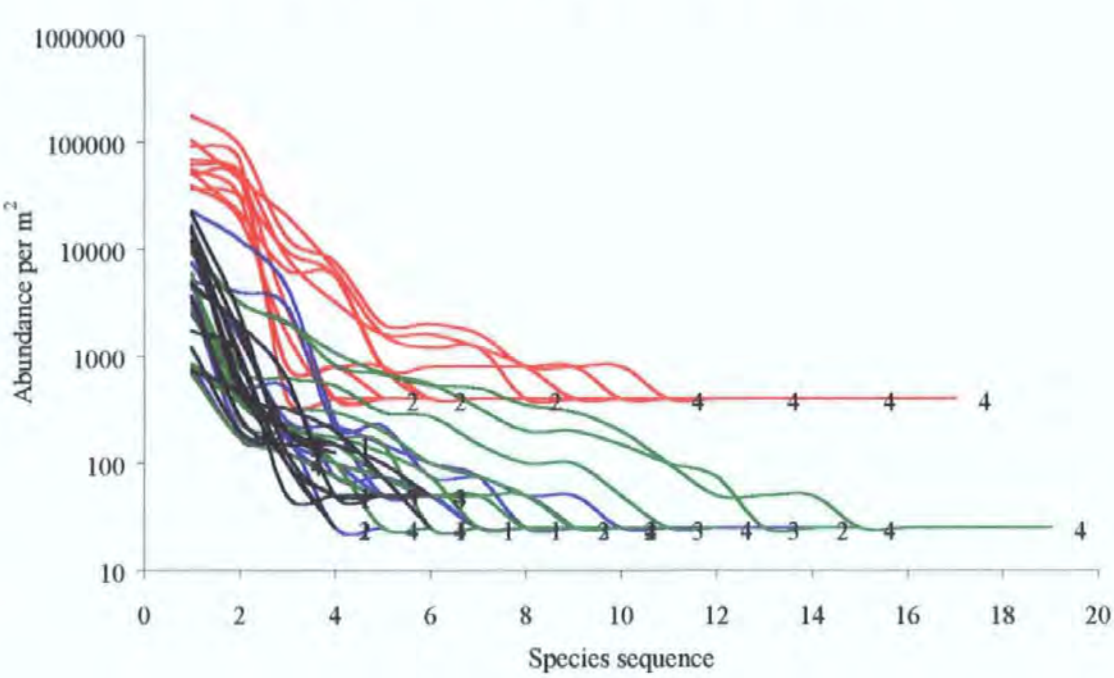


Fig. 5.5. Dominance-diversity curves for all replicates (May only). Red lines represent May 1990, blue lines represent May 1992, green lines represent May 1993 and black lines represent May 1994. Curve tails are annotated with site number.

The graphical observations were not entirely borne out by the numerical analysis of species diversity, the Shannon-Wiener index (Eq. 5.2). These values for the pooled replicates are plotted against survey period in Fig. 5.6. Any spatial trend was indistinct, with station SH4 outside and SH1 inside the harbour having the joint lowest diversity at the end of the construction in November 1993 (H'_e = 0.25) and after impoundment in May 1994 (H'_e = 0.35). The lowest diversity overall was observed at SH3 in November 1994 (H'_e = 0.2) whilst the highest diversity was seen at SH2 in May 1993 (H'_e = 2.2), considerably less than the theoretical maximum (H'_e = 3.7) because this was more a result of reduced dominance by cirratulids than a high number of species. The diversity at stations SH2 and SH3 returned to May 1990 values in May 1994 after considerable variability before and during construction, but decreased again in November 1994. The diversities at stations SH1 and SH4, furthest from the construction site and outside the

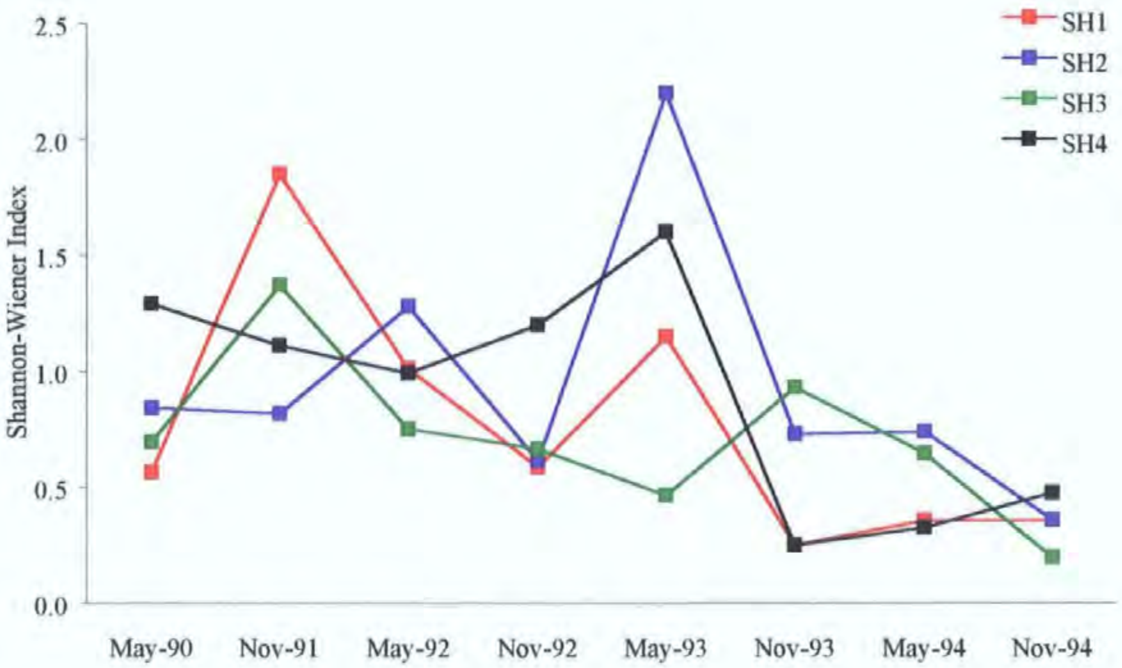


Fig. 5.6. Site-aggregated species diversity (Shannon-Wiener Index) for each site from May 1990 to November 1994.

harbour respectively, were also lower in May 1994 and November 1994 than in May 1990. This was particularly so for SH4, the control station outside the harbour. This may mean that the variability in diversity was a general feature that could not be attributed to construction activities or impoundment, or more probably that the control station was situated too close to the construction site and was therefore equally affected. In either event, the traditional univariate analyses proved inconclusive in the study of the effects of construction and impoundment, because they did not adequately describe the subtle changes in community structure that may have occurred.

5.2 Multivariate Analyses of Benthic Macrofauna

The effects of major construction and subsequent impoundment on the benthos were more effectively examined using a holistic approach, interpreting species number and abundance as community structure and identifying changes to that structure. Similar techniques have recently been used to great effect in a variety of perturbed ecosystems: tropical coastal regions (Agard *et al.*, 1993), temperate shelf seas (Craig *et al.*, 1993) and UK estuaries (Somerfield *et al.*, 1994a and 1994b).

5.2.1 Analytical methodology

All data were analysed using the suite of computer programs PRIMER (Plymouth Routines in Multivariate Ecological Research) developed by workers at Plymouth Marine Laboratory (Clarke, 1993). The macrofauna data were examined in terms of community structure using the normal q-type analysis described by Field *et al.* (1982) with ordination by non-parametric multi dimensional scaling or MDS (Kruskal and Wish, 1978). The abundance data in the samples/species matrix were root-root-transformed in order to reduce the statistical prominence of the more abundant species. The Bray-Curtis measure of dissimilarity (Bray and Curtis, 1957) was calculated between every permutation of sample pairs, generating a triangular similarity matrix which was then classified by group-average sorting to produce a dendrogram showing clusters within the dataset. The matrix was also subjected to MDS analysis in order to produce a two-dimensional ordination of relative similarities, such that the separation of one sample from all others was directly proportional to the similarity between it and the rest. The efficiency of reducing the complex relationships between samples from many dimensions to just two was tested using Kruskal's stress formula (Kruskal and Wish, 1978), checked against the rule of thumb proposed by Clarke (1993). One-way ANOSIM (analysis of similarity) was used to test the formal significance of differences between sites before, during and after construction (Clarke and Green, 1988). This was effected by calculating the difference between the average rank similarity of samples within the same survey period, and that of that of those collected different survey periods, according to the following equation (Clarke, 1993):

$$R = \frac{(\bar{r}_B - \bar{r}_W)}{(M/2)} \qquad \text{Eq. 5.4.}$$

Where $\bar{r}_B$ is the average rank similarity among all pairs of replicates between periods, $\bar{r}_W$ is the average rank similarity among all pairs of replicates within periods, $M = n(n - 1) / 2$ and n is the total number of samples.

A global R value of 0 indicated no temporal differences and +1 indicated that all samples in a survey period were more similar to each other than to any sample in any other survey period. Using random relabelling of samples in the similarity matrix, the R statistic was recalculated for 1,000 of the many possible permutations and these values were compared to the original R statistic to determine the significance level. If none of these random permutations were greater than or equal to the original R, the null hypothesis could be rejected with a significance level P <0.001. The species contributing to dissimilarities between pre- and post-construction samples were investigated using the similarity percentages or SIMPER procedure (Clarke, 1993).

A comparable multivariate analysis technique was applied to a subset of the sediment variables in Section 2.4 (viz. percentage fine (silt + clay) fraction, total and bioavailable concentrations of Cd, Cu, Hg, Pb and Zn) to produce ordinations of spatial and temporal variability between sites, using correlation-based principal components analysis (PCA). The BIOENV routine was used to provide harmonic or weighted Spearman Rank correlations between the macrobenthic similarity matrix and combinations of variables from the 'environmental' euclidean distance matrix underlying the PCA ordination (Clarke and Ainsworth, 1993), in order to identify possible controls upon the benthic community structure.

5.2.2 Variation in benthic community structure (similarity)

The similarity matrix generated by PRIMER from the site-aggregated species-abundance data collected between May 1990 and November 1994 was subjected to one-way ANOSIM in order to test for differences between replicates before and during construction, and after impoundment (Table 5.4), using 1000 randomised permutations in each case. The global test revealed a slight significant difference between groups, whilst the pairwise tests generated strong significant differences between all sites both before and during construction, and before and after construction. However, no significant difference between all sites during and after construction was identified, suggesting that any major changes in community structure occurred during the construction period. Classification of the similarity matrix for site-aggregated samples is shown in dendrogram form in Fig. 5.7. Several groups emerged at low levels of similarity. At 25 %, the data clustered in two main groups, the first comprising samples collected during May 1990, November 1991 and May 1992 (before construction) and the second comprising all samples taken in November 1992, May 1993, November 1993 (during construction), and May 1994 and November 1994 (following impoundment), together with two semi-azoic sites sampled in November 1991. The first group divided into two further groups at 30 % similarity, reflecting a high degree of temporal and spatial variability between samples before construction.. The first of these contained stations SH2 and SH4 in May 1990 and November 1991, and the second all stations in May 1992. Stations SH1 and SH3 were almost azoic in May 1990 as previously discussed and could not be included in the classification owing to their overpowering effect on the subsequent ordination. The second group at 25 % similarity remained largely intact at 30 %, apart from the removal of semi-azoic sites SH1 and SH3 in November 1991.

Table 5.4. Results of one-way ANOSIM test on the rank similarities between site-aggregated samples before construction (May 1990, November 1991 and May 1992), during construction (November 1992, May 1993 and November 1993) and after impoundment (May 1994 and November 1994).

Test	*R* value	Significance level, *P*
Global: before, during, after	0.43	<0.001
Pairwise: before, during	0.67	<0.001
Pairwise: before, after	0.62	<0.001
Pairwise: during, after	0.01	NS

NS - Not significant.

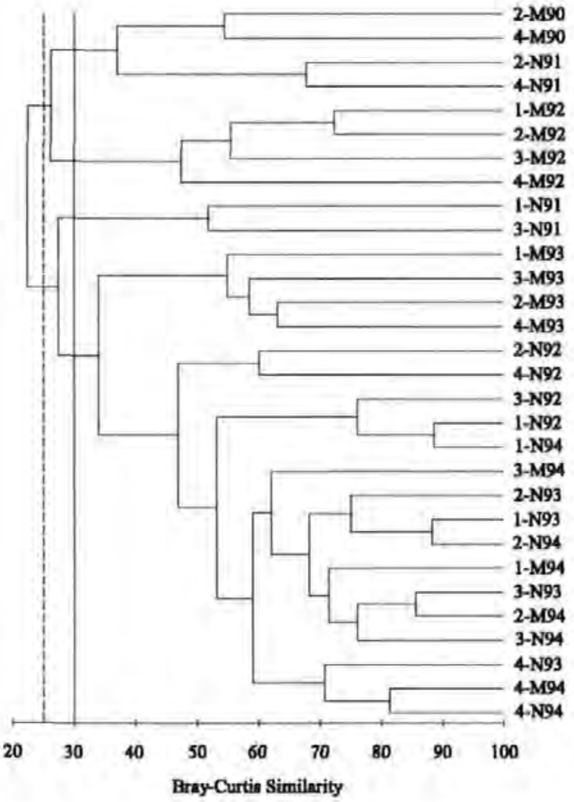


Fig. 5.7. Classification dendrogram of site-aggregated abundance data (root-root-transformed). Vertical dashed line at 25 % similarity separates two clusters and vertical solid line at 30 % similarity separates two further clusters from each of the first. Annotation on right is in the form 'station number - survey date'.

The relative similarities were easier to visualise after MDS ordination (Fig. 5.8), in which the clusters at 25 % and 30 % similarity were transposed from the classification dendrogram. A high degree of spatial variability before construction was evident, particularly in November 1991. This tended to decrease with time to a minimum in May 1994 and November 1994, in concordance with the univariate observations in Section *5.1.2*. The temporal variability between adjacent survey periods also decreased with time. November 1993, May 1994 and November 1994 samples were very similar, suggesting that the benthic community was achieving a new, stable equilibrium. It must be noted that all sites were sublittoral throughout the study; the physical effects of impoundment were restricted to an increase in the mean depth at sites inside the harbour and an alteration to the tidal currents. Therefore, similarity between sites at the end of construction and shortly after impoundment might be expected, particularly since the most profound changes in community structure, as indicated by spatial separation in the ordination, occurred during the early stages of construction between May 1992 and November 1992. However, Warwick (1993) lists among the disadvantages of using soft-bottom macrobenthos for impact studies that the response time to a pollution event is slow. Whilst the impoundment was not strictly a pollution event, a rapid loss of certain species might not be replaced for one or two years. It is therefore difficult to predict the long-term effect of impoundment upon the macrobenthos from the current dataset. It is interesting to note the temporal similarity of samples from the same site within the construction and post-

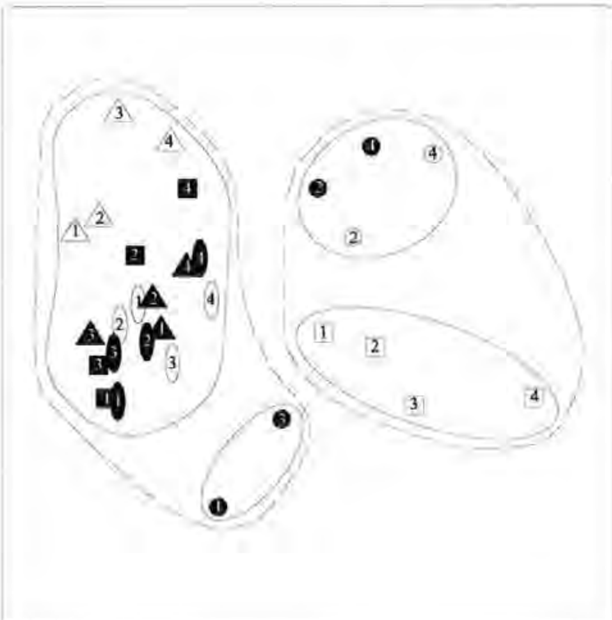


Fig. 5.8. MDS ordination (stress = 0.17) of site-averaged macrofaunal abundance data (root-root-transformed) from May 1990 (open circles), November 1991 (filled circles), May 1992 (open squares), November 1992 (filled squares), May 1993 (open triangles), November 1993 (filled triangles), May 1994 (open ellipses) and November 1994 (Filled ellipses). Each symbol is marked with station number. Clusters from Fig. 5.7 are delineated by solid lines (30% similarity) and dashed lines (25% similarity). Axes are arbitrary and therefore not shown.

impoundment cluster to those within the pre-construction cluster. SH1 and SH3 are located towards the bottom of the cluster near to the corresponding stations in November 1991 and May 1992, whilst SH2 and SH4 tend towards the top of the cluster near to those in May 1990 and November 1991. However, the stress coefficient of 0.17 for the ordination was approaching the upper limit of 0.2, above which interpretation of the intra-sample relationships becomes increasingly misleading (Clarke, 1993) and a higher-dimensional plot might reveal different features. Therefore, little emphasis should be placed on these. It is the larger-scale patterns previously discussed which are important.

The analysis was repeated for May samples only, in an effort to remove the influence of natural seasonal variability on community structure. The decision to use May samples in preference to November was arbitrary except that they constituted the longer time series. The iterative nature of the MDS algorithm is extremely computer-intensive, so a reduced number of survey periods also allowed the inclusion of all replicates in the analysis, rather than aggregated data. One-way ANOSIM was performed on the similarity matrix in order to test for differences between replicates before and during construction, and after impoundment (Table 5.5) using 1000 randomised permutations in each case. It is again evident that there were strong significant differences in community structure before, during and after construction, with the greatest change occurring between survey periods before and during that episode.

Table 5.5. Results of one-way ANOSIM test on the rank similarities between all May survey replicates before construction (May 1990 and May 1992), during construction (May 1993) and after impoundment (May 1994).

Test	R value	Significance level, P
Global: before, during, after	0.67	<0.001
Pairwise: before, during	0.78	<0.001
Pairwise: before, after	0.65	<0.001
Pairwise: during, after	0.71	<0.001

The classification of the similarities between samples during May-only surveys using abundance data for all replicates is shown in dendrogram form in Fig. 5.9. Again, two groups clustered together at 25 % similarity, the first comprising all replicate samples from May 1990 and May 1992 (before construction) and the second comprising all replicate samples from May 1993 (during construction) and May 1994 (after impoundment). The first group subdivided into two further groups between 25 % and 30 % similarity, entirely separating May 1990 from May 1992, whilst the second group subdivided into two further groups at a slightly greater similarity of 30 %. One of these groups contained the majority of replicates from May 1993, the other contained all replicates from May 1994 together with the remaining May 1993 replicates, one from each of stations SH1, SH2 and SH4. The relative similarities were again easier to visualise after MDS ordination (Fig. 5.10), in which the clusters at 25 % and 30 % similarity were transposed from the dendrogram.

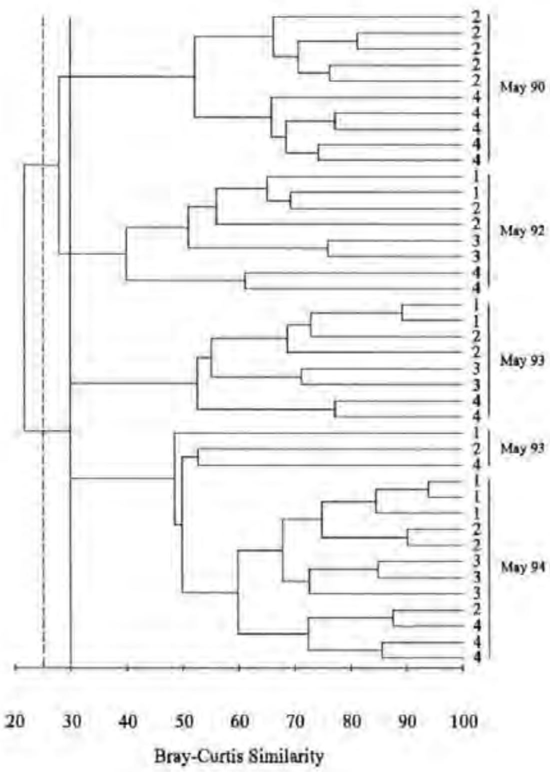


Fig. 5.9. Classification dendrogram of macrofaunal abundance data (root-root-transformed) for May only (all replicates). Vertical dashed line at 25 % similarity separates two clusters and vertical solid line at 30 % similarity separates two further clusters from each. Station numbers on right are bracketed by date.

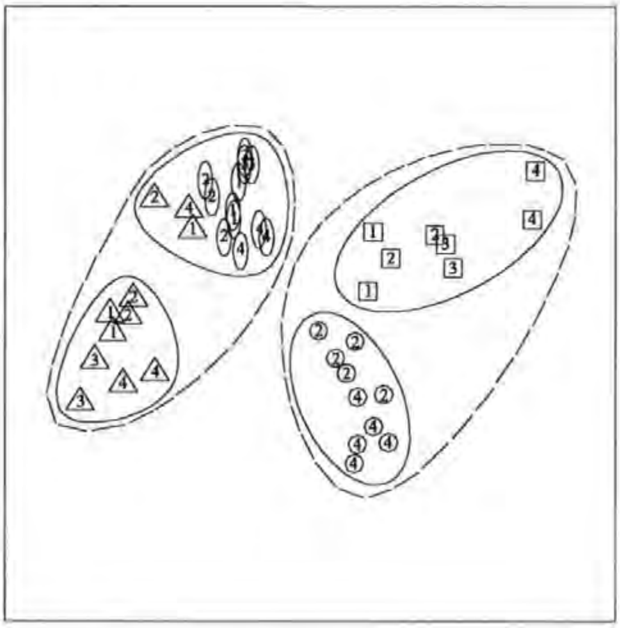


Fig. 5.10. MDS ordination (stress = 0.17) of macrofaunal abundance data (all replicates; root-root-transformed) for May 1990 (circles), May 1992 (squares), May 1993 (triangles), and May 1994 (ellipses). Clusters from Fig. 5.9 are delineated by solid lines (30% similarity) and dashed lines (20% similarity). Each symbol is marked with station number.

The distribution of replicates from before construction on the right of the ordination indicates that spatial variation within each survey period was less than temporal variation between survey periods both inside and outside the harbour. The spread of May 1993 replicates and their inclusion in different clusters suggests a degree of intra-site variability during construction not seen in any other May period, contrasting particularly with the tight grouping of all replicates from May 1994. Removal of seasonal factors from the community structure did not therefore increase the sample similarity between survey periods. Once again, the greatest changes in community structure occurred during the early stages of construction, between May 1992 and May 1993. SIMPER analysis of transformed species data for the May samples identified those contributing most to the dissimilarity between pre-construction and post-construction associations (Table 5.6). The macrofauna communities in the post-construction samples are separated from those from before construction by a general reduction in the abundance of several polychaete taxa including *Ophryotrocha* spp., Cirratulidae spp. and Spionidae spp. Furthermore, the previously abundant oligochaete genus *Tubificoides* was absent from post-construction samples, although it should be noted that it reappeared at SH1 during November 1994 (Section *5.1.2*). A few taxa, such as the polychaetes *Heteromastus filiformis* and *Capitella capitata*, increased in abundance following construction of the lock. Polychaete abundance has also increased in areas of the North Sea suffering from eutrophication (Turkstra *et al.*, 1991), whilst *C. capitata* is a species particularly associated with the colonisation of perturbed environments (Grassle and Grassle, 1974).

Table 5.6. Summary of similarity terms (SIMPER) analysis. Difference in mean abundances (individuals m^{-2}) of species contributing to >3 % of dissimilarities between pre- (May 1990 and May 1992) and post-construction (May 1993 and May 1994) samples.

Species	Abundance				Percentage dissimilarity
	Post-construction (n=24)		Pre-construction (n=18)		
	Mean	SD	Mean	SD	
Polychaeta					
Ophryotrocha spp.	1308	3666	34929	42138	26.9
Cirratulidae spp.	5804	5881	35332	36169	12.7
Lumbrineris gracilis	1432	4464	0	0	5.8
Spionidae spp.	36	131	685	1352	5.6
Melinna palmata	157	204	400	629	5.6
Nephtys hombergi	178	213	147	231	4.3
Heteromastus filiformis	542	1219	22	94	3.6
Ampharetidae sp.	0	0	115	217	3.1
Capitella capitata	72	136	8	19	3.0
Oligochaeta					
Tubificoides swirencoides	0	0	1800	2714	5.2
Tubificoides benedini	0	0	3200	5892	4.9
Crustacea					
Pariambus typicus	0	0	107	216	3.0

5.3 Analysis of Changes in Environmental Variables

An investigation of the changes in sedimentary environmental variables was made in order to test whether any of these factors were controlling the benthic community structure.

5.3.1 Spatial and temporal variations in sediment particle size

Changes in particle size may affect benthic macrofauna community structure in several ways. Many species of infauna are adapted to certain substrates in which their burrows and siphon holes remain stable with minimal maintenance, or their respiratory and feeding mechanisms are unclogged (McLusky, 1971). Therefore, physical changes in the substrate may aid or hinder survival of certain species. Distribution of particle size may also affect the total load of contaminants in the sediments, with implications for the survival of infauna. For example, the sorptive capacity for trace metals of the finer fractions is greater (De Groot *et al.*, 1976). Also, finer sediments will be richer in organic material and poorer in oxygen, thereby favouring succession of certain taxa such as *Capitella capitata* and spionids (Pearson and Rosenberg, 1978). Fig. 5.11 shows the percentage distribution of particle size (by weight) in the Sutton Harbour

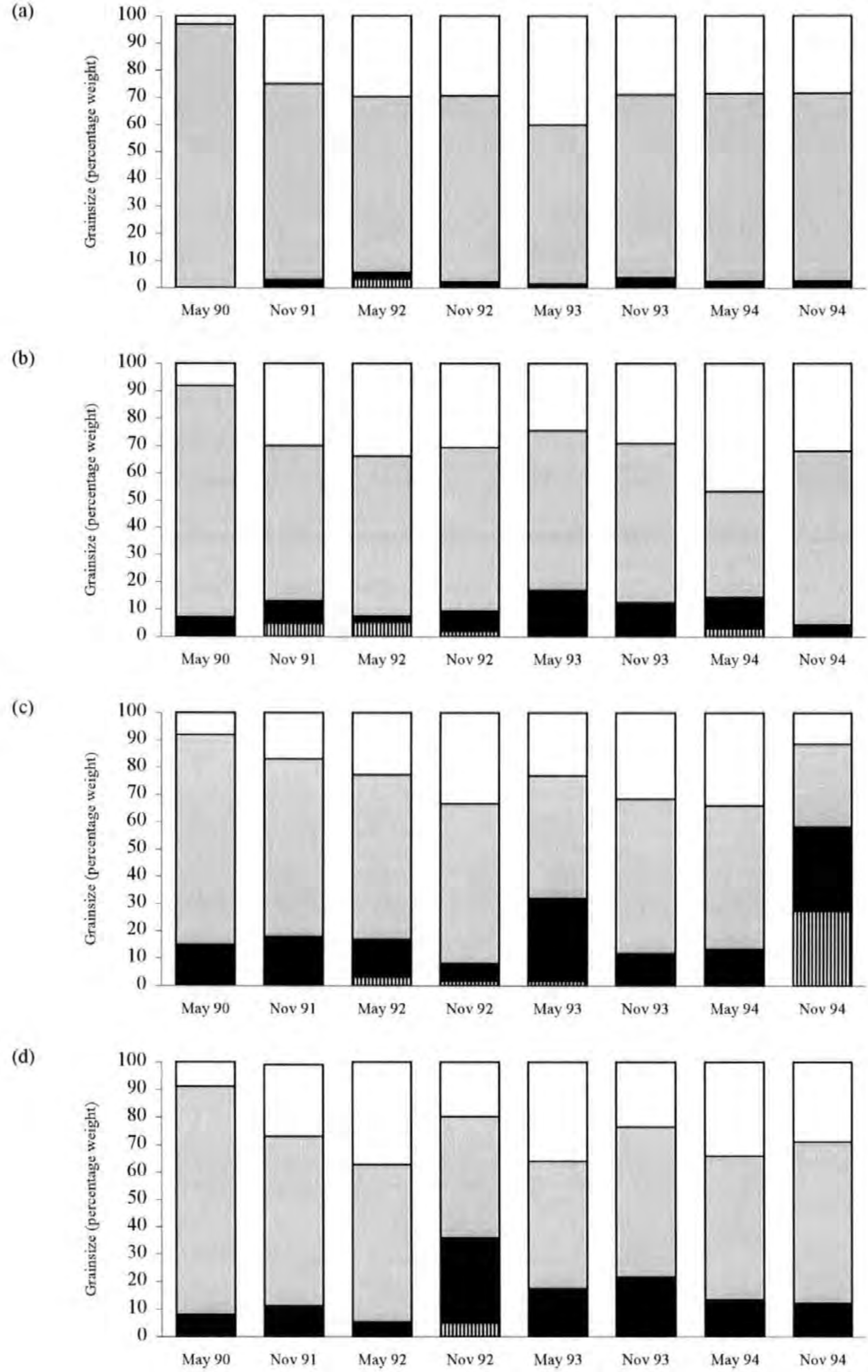


Fig. 5.11. Percentage grainsize (by weight) at stations (a) SH1, (b) SH2 and (c) SH3 inside the harbour and (d) SH4 outside the harbour. Vertical hatching represents gravel (>2 mm), black represents sand (>63 μm), grey represents silt (<63 μm) and white represents clay (<4 μm).

sediments between gravel, sand, silt and clay fractions from May 1990 to November 1994. It is evident that the sediments at SH1 (Fig. 5.11a) furthest from the location of construction activities were the least perturbed during the study period and were predominantly fine muds. Notable changes were a permanent increase in the clay fraction in November 1991 and a further temporary increase in May 1993. Station SH2 (Fig. 5.11b) was characterised by slightly coarser sediments than SH1 as a result of increased current shear near the harbour mouth. Sand content was greatest, concordant with low clay content in May 1993 when the harbour mouth had been restricted by the lock structure but the gates were not yet in place, probably resulting in the greatest current shear during the study. By November 1994, station SH2 appeared to be returning to the starting condition in May 1990 as a possible result of a balance between increased tidal shear and decreased flow duration. Station SH3 (Fig. 5.11c) was characterised by the greatest variability in particle size distribution during the study. This was undoubtedly due to the proximity of the coffer dam construction and subsequent land reclamation and quay wall construction. The particle size distributions do not fully portray the extent of the perturbation at this station; observations made during sampling and analysis showed that the coarser fractions were dominated by non-indigenous building materials. The patchy nature of the sediments is highlighted by the unusually coarse distribution in November 1994. The sediments outside the harbour at station SH4 (Fig. 5.11d) appeared as perturbed as stations SH2 and SH3, supporting the contention that this control station was located too near to the construction activities. The considerable changes in November 1992 coincided with the period of intense dredging and blasting activity at the harbour mouth and may reflect a displacement of coarse material. A return to stability was evident from May 1993 to November 1994 during the latter stages of construction and the early stages of impoundment. It does not appear that closure of the harbour mouth had a significant effect upon the sedimentation regime outside the harbour.

5.3.2 Spatial and temporal variations in total and bioavailable heavy metals

Changes in heavy metal concentrations in the harbour sediments could have an effect on the community structure of the benthic macrofauna because of the chronic and acute toxic effects of these elements. Heavy metal concentrations in sediments are often between three and five orders of magnitude higher than in the overlying water, so the bioavailability of a small fraction is often enough to produce biological effects, particularly with infauna (Bryan and Langston, 1992). Table 5.7 compares the total (HNO_3 digest) metal concentrations in the sediments inside and outside the harbour during the present study with those obtained in adjoining systems. Concentrations of Cd were in good agreement with measurements in the Plym and Tamar made 15 to 20 years ago, but showed some elevation above more contemporary observations. However, the bioavailability of Cd tends to be controlled by free Cd ions in the sediment porewaters and is reduced in saline waters by complexation with chloride ions (Langston, 1986).

Table 5.7. Comparison of heavy metal concentrations in Sutton Harbour sediments (<63 μm HNO_3 digest) from May 1990 to November 1994 with literature values for adjacent estuaries.

Metal	Sediment metal concentration, μg g^{-1} (mean ± 1 SD, mean or min-max).				
	Inside harbour (SH1 to SH3)	Outside harbour (SH4)	Plym Estuary	Tamar Estuary	Lynher Estuary
Cd	2.3 ± 2.4[a]	2.7 ± 3.0	3.6[b]	2.3[c]	0.71[c]
			2.5 - 9.3[c]	0.96[f]	
			1.1 ± 0.4[d]		
Cu	172 ± 63[a]	138 ± 42	68 ± 30[d]	330[f]	274[g]
Hg	1.5 ± 0.9[a]	0.8 ± 0.3	0.02 - 2.6[e]	0.83[f]	2.1[g]
			0.3 ± 0.2[d]		
Pb	212 ± 81[a]	154 ± 63	-	235[f]	150[g]
Zn	318 ± 165[a]	228 ± 88	256 - 358[c]	673[c]	570[c]
			166 ± 62[d]	452[f]	317[g]

[a]Current study (n=24 inside; n=8 outside); [b]Bryan *et al.*, 1980; [c]Bryan and Hummerstone, 1973; [d]Millward, 1993; [e]Millward and Herbert, 1981; [f]Bryan and Langston, 1992; [g]Bland *et al.*, 1982.

Concentrations of Cu in the Plym were four to five times lower than in the Tamar and Lynher estuaries. This is probably due to the more metalliferous nature of the Tamar and Lynher catchment areas and the proximity of old mine workings (Bryan and Hummerstone, 1973). However, Cu concentrations in the harbour sediments were more than twice those observed in the Plym, and therefore indicated localised inputs. A similar situation was observed with Zn: Tamar concentrations were three times greater than Plym concentrations, yet Sutton Harbour Zn concentrations were up to twice those observed in the Plym. A strong significant correlation between Cu and Zn in the sediments inside the harbour (Table 5.8) supports the hypothesis made in Section 4.4 that these inputs are from the same source, namely the high density of vessels inside the harbour using both Cu-based antifouling paints (Claisse and Alzieu, 1993) and Zn sacrificial anodes. These elements enter the water column in the dissolved phase, but mutual strong significant correlations with Fe (Table 5.8) suggest that they are scavenged by Fe oxyhydroxide surface coatings on particulate matter and then deposited to the sediments.

Concentrations of Hg in the harbour gave good agreement with early measurements in the Plym, but as with Cd were enhanced relative to more recent measurements in both the Plym and Tamar Estuaries. Estuarine contamination with Hg stems primarily from anthropogenic (industrial and sewage) inputs (Bryan and Langston, 1992) so the reducing concentrations in the Plym may represent an improvement in the pollution status of this estuary from 1981 to 1991, whilst elevated Hg concentrations in the harbour could be due to historic contamination from Fishers Nose outfall and from materials such as discarded

Table 5.8. Matrix of correlation coefficients (r) between Fe, Mn, Cd, Cu, Hg, Pb and Zn concentrations (<63 μm; HNO_3 digest) in sediments inside and outside Sutton Harbour.

	Inside harbour (SH1 to SH3) from May 1990 to November 1994 (n = 24).						Outside harbour (SH4) from May 1990 to November 1994 (n = 8)					
	Mn	**Cd**	**Cu**	**Hg**	**Pb**	**Zn**	**Mn**	**Cd**	**Cu**	**Hg**	**Pb**	**Zn**
Fe	0.37[a]	NS	0.85[c]	0.48[b]	0.59[c]	0.81[c]	0.78[b]	-0.6[a]	NS	NS	-0.8[b]	0.84[c]
Mn		NS	NS	NS	0.44[a]	NS		-0.7[a]	NS	NS	NS	NS
Cd			0.37[a]	NS	NS	0.37[a]			NS	NS	NS	NS
Cu				NS	NS	0.79[c]				NS	NS	0.62[a]
Hg					0.54[c]	NS					NS	NS
Pb						NS						NS

NS - not significant at $p < 0.05$; [a]$p < 0.05$; [b]$p < 0.01$; [c]$p < 0.005$.

batteries. However, because of the formation of strong covalent bonds between Hg and organic molecules, the bioavailability of Hg is inversely proportional to the organic content of the sediments (Langston, 1986). Although no data are available, the Sutton Harbour sediments are thought to be organically enriched owing to the presence of high numbers of nematodes. These were considered indicative of organic enrichment in the Ems-Dollard before reductions in organic loading (Essink and Romeyn, 1994). Furthermore, although the more toxic methylated Hg (MeHg) species are biomagnified 100 to 150 times in estuarine benthic macrofauna, they typically represent <0.2% of the total Hg concentration in sediments (Bryan and Langston, 1992). Concentrations of Pb in the harbour are similar to those in the Tamar and, in the absence of data from the Plym, are thought to be greater than outside. They exhibited a strong, significant correlation with Fe, and were probably associated primarily with Fe oxyhydroxide particle coatings, whilst a strong significant correlation with Hg suggested an (unmeasured) association with organic matter. Organic species of Pb, chiefly alkyl Pb compounds as petrol additives, probably cause the augmented concentrations of Pb in the harbour sediments. Although dissolved alkyl Pb behaves conservatively in estuarine profiles suggesting that its particle reactivity is low (Langston, 1986), the high hydrocarbon content of the Sutton Harbour sediments (from field observations) suggests retention of inputs from surface runoff, bilge pumping and refuelling spillage.

The histograms shown in Figs. 5.12a and c and 5.13 a and c record total concentrations (HNO_3 digest) of metals in the coarse and fine fractions of the harbour sediments from May 1990 to November 1994. Figs. 5.12b and d and 5.13b and d contain bioavailabilities of these metals ($[X]_{HOAc}/[X]_{HNO_3}$) as a percentage.

Fine fraction (<63 μm). It is evident that Cu, Pb and Zn concentrations in the fine fraction follow similar trends over the entire study period (Fig. 5.12a): Values increased from May 1990 to November 1991/May 1992, then declined to May 1993/November

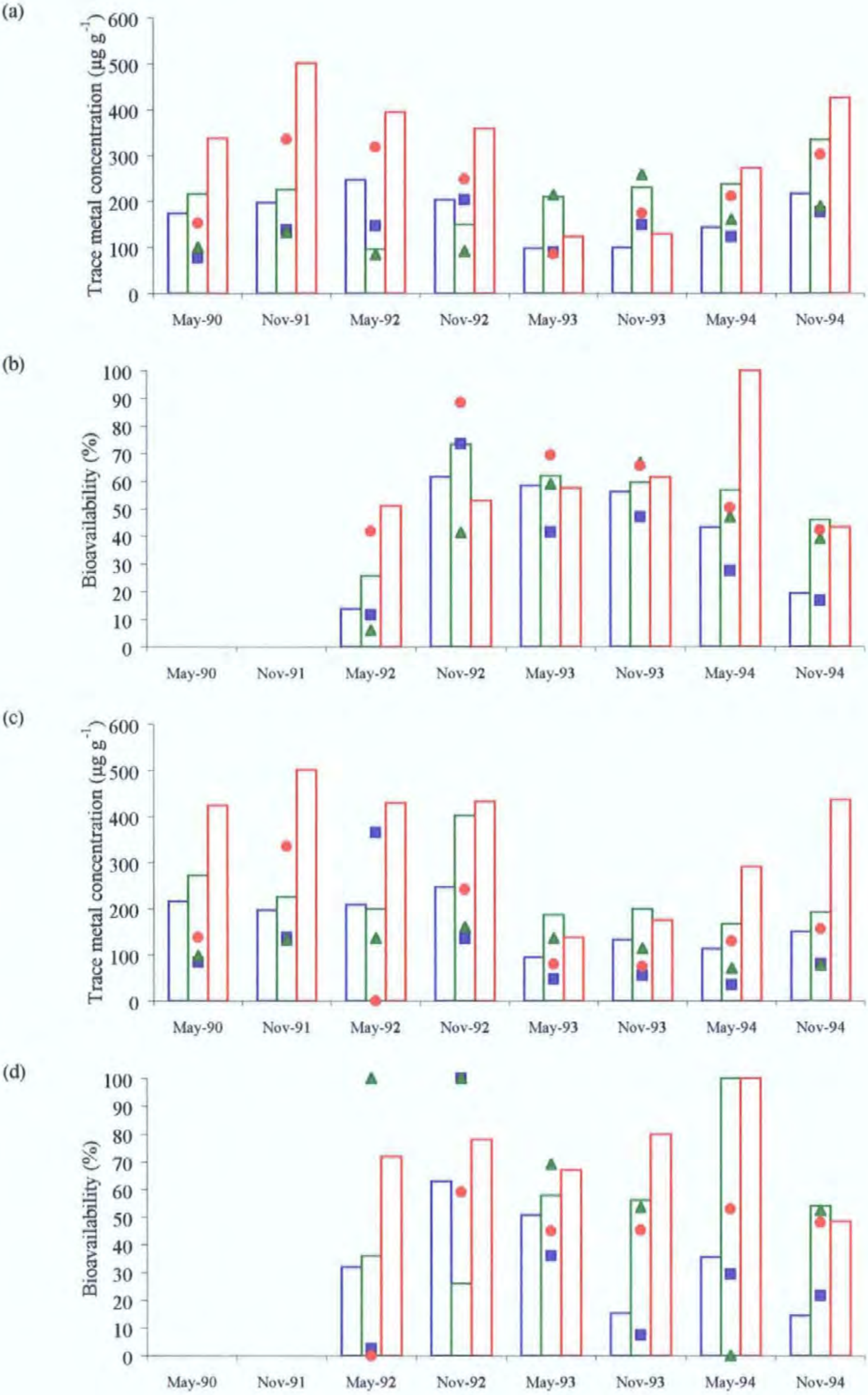


Fig. 5.12. Sediment trace metal concentration (HNO_3) and percentage bioavailability ($[X]_{HOAc}/[X]_{HNO_3}$) in (a,b) <63 µm and (c,d) >63 µm fractions. Bars represent mean of SH1 to SH3 and symbols represent SH4. Cu shown as blue bars and squares, Pb as green bars and triangles and Zn as red bars and circles.

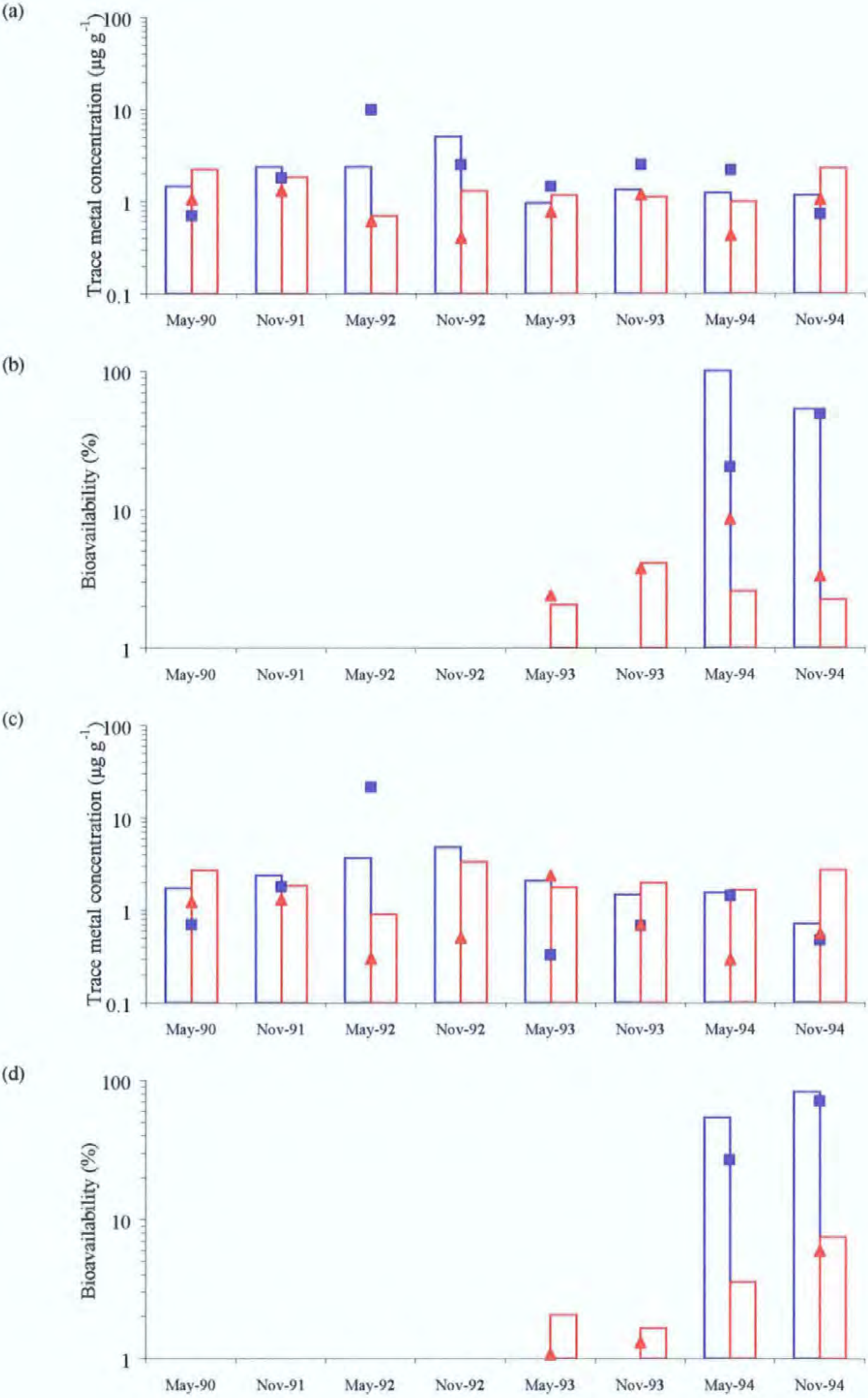


Fig. 5.13. Sediment trace metal concentration (HNO_3) and percentage bioavailability ($[X]_{HOAc}/[X]_{HNO_3}$) in (a,b) <63 µm and (c,d) >63 µm fractions on a logarithmic scale. Bars represent mean of SH1 to SH3 and symbols represent SH4. Cd shown as blue bars and squares and Hg as red bars and triangles.

1993 and then increased again to November 1994. These changes may be linked to the redistribution of artificially resuspended sediment from dredging or blasting. Resuspended sediments may have become depleted in heavy metals through oxidisation processes before settling elsewhere, thus causing the observed minimum in metal concentrations in May 1993, directly after the major period of sediment disturbance. The relatively rapid increases to November 1994 support the hypothesis of contaminant accumulation in the bottom waters and sediments after impoundment. Concentrations of Cd and Hg (Fig. 5.13a) exhibited no such clear trends, although Hg concentrations were depleted between November 1991 and May 1994 relative to the initial (May 1990) and final (November 1994) concentrations of *circa* 2 $\mu g\ g^{-1}$, whilst concentrations of Cd were augmented during construction in May 1992 and November 1992 relative to the initial and final concentrations of *circa* 1.5 $\mu g\ g^{-1}$. The short-term elevation of Cd, particularly in November 1992, was possibly due to the use of detonators containing its fulmonate during the bedrock blasting activities.

Coarse fraction (>63 μm). It is evident that the trace metal concentrations in the coarse fraction (Figs. 5.12c and 5.13c) are equivalent to those in the fine fraction. This is unusual given the greater sorptive capacity of the fine fraction (De Groot, 1976) and is probably symptomatic of the high degree of anthropogenic modification of the harbour sediments: discrete flakes and fragments of metal were observed in many samples during collection and sieving. However, the bioavailability of these metals was probably lower than those associated with the fine fractions because of the lower specific surface area and relatively lower intake by deposit feeders. The concentrations of Zn in the coarse fraction exhibited an almost identical trend to those in the fine fraction, with an increase to November 1991, a decrease to May 1993 followed by an increase to November 1994. Concentrations of Cu and Pb exhibited a two-stage trend, with relatively high concentrations (200 to 250 μg-Cu g^{-1} and 200 to 400 μg-Pb g^{-1}) from May 1990 to November 1992 and lower concentrations (90 to 140 μg-Cu g^{-1} and 160 to 200 μg-Pb g^{-1}) from May 1993 to November 1994. Concentrations of Cd and Hg showed similar trends to those in the fine fraction, with the highest concentrations of both in November 1992 probably due to sediment disturbance. Given the high degree of anthropogenic modification, it is difficult to draw any conclusions from these data, other than to state that the trace metal concentrations in the coarse fraction have not increased overall due to construction or impoundment.

Bioavailability. The data for percentage bioavailability are not available for May 1990 and November 1991. Several trends emerge from May 1992 onwards. The bioavailability of Cu and Pb in the fine fraction (Fig. 5.12b) and Cu in the coarse fraction (Fig. 5.12d) were highest in November 1992, and then decreased steadily to a low in November 1994 both inside and outside the harbour. Bioavailability of Zn in the fine fraction and Pb and Zn in the coarse fraction was at a maximum (100 %) in May 1994. Bioavailability of Hg was extremely low throughout the study, ranging from <1 % to 4 % in the fine fraction (Fig. 5.13b) and from <1 % to 7 % in the coarse fraction (Fig. 5.13d). This is not surprising, since organically-bound Hg is resistant to all but the strongest chemicals (such as HNO_3) and is therefore unlikely to bioaccumulate (Langston, 1982; 1986). Bioavailability of Cd was extremely low (<1 %) in May 1992 and November 1992, but very high in May 1994 (100 % fine; 60 % coarse) and November 1994 (50 % fine; 90 % coarse). The data for May 1993 and November 1994 are unavailable. It is difficult to rationalise any of these variations, but it must be noted that the HOAc leach may considerably overestimate the bioavailable fraction in such anthropogenically modified sediments, dissolving far greater masses of discrete metals than would be available in the guts of benthic organisms. Conversely, Bryan and Langston (1992) summarised that HNO_3 was the chemical leachate that best predicted the bioavailability of Cu, Hg and Pb. For this reason and the uncertainties and sparsity of the HOAc dataset, the concentrations of metals in the HNO_3 digest of the fine fractions were selected for BIOENV correlation.

5.4 Identification of Controls on Benthic Community Structure

5.4.1 Principal Component Analysis of environmental variables

The PCA analysis of total sediment metals (HNO_3 digest; <63 μm fraction) and percentage fine fraction is represented by the ordination in Fig. 5.14. Although primarily produced as an input to the BIOENV procedure, the ordination also serves to illustrate the complex variability in the combined environmental variables.

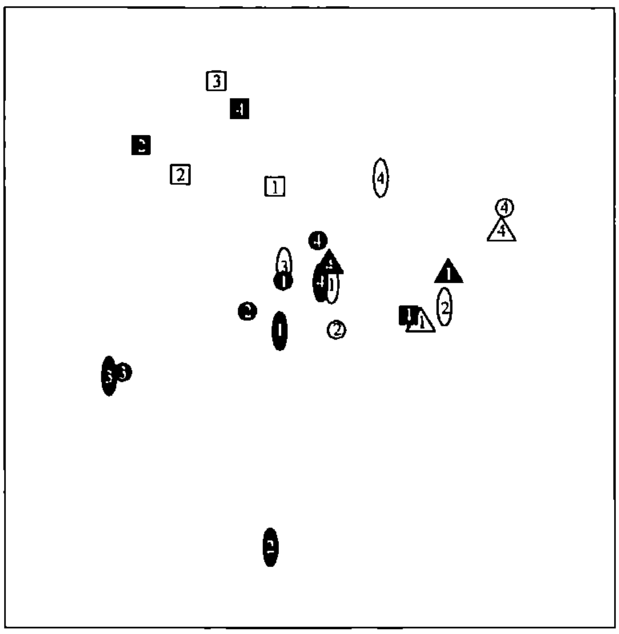


Fig. 5.14. PCA ordination of sediment trace metals (Cd, Cu, Hg, Pb and Zn) and percentage fine sediment fraction at stations SH1 to SH4 from May 1990 (open circles), November 1991 (filled circles), May 1992 (open squares), November 1992 (filled squares), May 1993 (open triangles), November 1993 (filled triangles), May 1994 (open ellipses) and November 1994 (Filled ellipses).

As might be anticipated by their relative distance from the disturbance, a certain amount of temporal consistency is apparent at stations SH1 and SH4, whilst greater spread is evident at stations SH2 and SH3.

5.4.2 Correlation between community structure and environmental variables

The results of the BIOENV procedure are shown in Table 5.9. It is evident that the highest correlation between the sediment euclidean distance matrix and the macrofaunal similarity matrix was provided by a combination of Pb and Zn concentrations. Inorganic Pb has been shown to accumulate biologically in marine organisms and to undergo alkylation to acutely toxic tetraethyl and tetramethyl Pb species *via* chemical and bacterial mechanisms (Maddock and Taylor, 1980; Bryan and Langston, 1992) and it features in the primary correlations of 1, 2 and 3 variable combinations. However, Zn is not generally thought to be especially toxic to marine organisms, particularly certain polychaetes which are able to regulate their body burden (Bryan and Hummerstone, 1973; Renfro, 1973). It is surprising therefore that Zn should feature in the strongest correlation ($\rho_w = 0.49$). This may be due to association with an undetermined sediment component of greater toxicity. Concentrations of Cu, thought to be the third most toxic metal to marine organisms after Hg and Ag (Bryan, 1976), also featured highly in the correlations with sets of one, two and three variables. Rygg (1985) observed a strong negative correlation between macrofauna diversity and Cu concentration in the sediments of Norwegian fjords. Of the fifty most frequently occurring species, twenty were significantly absent ($p < 0.05$) from stations with Cu concentrations of >200 $\mu g\ g^{-1}$, lowering diversity to 50 % of the maximum observed value. Concentrations of Hg did not feature in any of the strongest correlations, suggesting that the production of MeHg was below that required for biological effects to be evident, particularly when other toxic metals were present in considerably greater concentrations.

Table 5.9. Output from BIOENV for three highest weighted Spearman Rank correlations (ρ_w) between macrofaunal and environmental similarity matrices (May data only) for combinations of k sediment variables.

k	1° correlation		2° correlation		3° correlation	
	Parameter	ρ_w	Parameter	ρ_w	Parameter	ρ_w
1	Pb	0.42	Cu	0.30	Zn	0.30
2	Pb/Zn	0.49	Cu/Pb	0.44	Cd/Pb	0.36
3	Cu/Pb/Zn	0.45	Cd/Pb/Zn/	0.44	Cd/Cu/Pb	0.41

5.5 Summary

The univariate analyses of benthic macrofauna in Sutton Harbour shows that cirratulid worms were by far the most abundant taxa. A general pattern of increasing species number and abundance (species richness) was evident from SH1 and SH3 through SH2 (inside the harbour) to SH4 (outside) before and during the early stages of construction that was perturbed in the later phases. After impoundment, recovery was only evident outside the harbour at SH4. Measures of species evenness confirmed the dominance of a few taxa (mainly cirratulids) at sites with higher species richness. Diversity measures before construction record the same increasing trend as species richness, followed by a general decrease in diversity throughout the study. Only at station SH4 was an increasing trend in species diversity recorded at the end of the study, suggesting recovery of the benthic community outside the harbour, and establishment of a new impoverished equilibrium inside the harbour.

When analysed with multivariate techniques, both as site-aggregates from all survey dates and all replicates from May surveys only, benthic macrofauna community structure exhibited significant changes throughout the study. In both cases, the greatest dissimilarities were identified between samples recovered before, and during, construction work. The impact upon the benthic macrofauna varied from general decline in abundance of many species and complete disappearance of one, to increases in abundance of other opportunistic taxa.

The heavy metal content and grainsize distribution of sediments also underwent changes during the study period. The most stable sediment composition was observed at station SH1, furthest from the lock, whilst the sediments at stations SH2, SH3 and SH4 all underwent significant changes in grain size distribution, particularly during the construction period. Concentrations of Cd, Cu, Hg, Pb and Zn were all enhanced inside the harbour compared with contemporary measurements in the Plym Estuary, as a combined result of historical inputs of sewage and waste materials, and more recent use respectively of Cu, Zn and Pb in antifouling paints, sacrificial anodes and fuel additives. Concentrations of Cu, Pb and Zn exhibited a significant increase in the fine fraction over the entire study, following considerable fluctuations during construction. Concentrations of Cd and Hg, although experiencing fluctuations during the study, did not increase overall from May 1990 values. Concentrations of Pb and Zn (and possibly Cu) provided the best explanation of the changes in community structure. The construction work and impoundment appear to have had an impact upon benthic faunal community structure, possibly as a result of sediment disturbances which changed the speciation of toxic heavy metals, leading to increased bioavailability.

Concisely, the benthic macrofauna community in Sutton Harbour was richer and more diverse before impoundment than after impoundment and shows little evidence of recovery within the first year of the new hydrodynamic regime. The significance of this observation is slight in a local context due to the initial paucity of the macrofauna. However, the observation is crucial to those concerned with ecosystem health in other systems requiring impoundment that may sustain a richer, more diverse benthic community. Future practitioners can now predict with some confidence that construction and operation of such a scheme will have a deleterious effect upon the benthos and the subsequent food chain.

Chapter Six

Conclusions, Management Strategies and Recommendations

Chapter 6 - Conclusions, Management Strategies and Recommendations

This study provides the first comprehensive statement that water and sediment quality in Sutton Harbour varies on short (semi-diurnal and diurnal), medium (spring-neap and seasonal) and long (inter-annual) timescales. Perturbations of these natural variations are a result of construction and operation of a major civil engineering project. The study is thought to be unique in this respect, in that it provides high quality data spanning the period before, during and after construction. An account of the actual ecological impact of harbour impoundment is given, rather than the mere assessment of environmental risks that would be provided by a statutory EIA. The original hypotheses were validated and revised in the light of the database, leading to a thorough understanding of the harbour processes at a high level of confidence. The recommendations will facilitate management decisions that must be made with respect to water and sediment quality in the immediate harbour area. It is hoped that it will contribute to the local knowledge held by the NRA and SWW plc, and provide useful ancillary information relevant to assessment of the Plymouth Area Sewage Scheme, to be commissioned before the end of the decade. It is further hoped that the study is adopted as a model for future monitoring of the impact of such impoundment schemes wherever they become necessary, providing, from the general findings, both an efficient mode of study and a predictive tool. With these points in mind, this final chapter contains an executive data summary, in which conclusions from the foregoing chapters are presented, followed by a study critique pertinent to future practitioners of long-term marine monitoring programmes. The study is concluded with strategic planning recommendations in order to facilitate future water quality maintenance in Sutton Harbour.

6.1 Executive Data Summary

The conclusions drawn from analysis of the extensive Sutton Harbour database are set out below, together with certain conceptual figures arising from observed changes in the seasonal variability of key variables. These are useful for reference in the management of the impounded harbour water, and of its internal sediment quality.

6.1.1 Water renewal times and hydrodynamic processes

The increases in water renewal time due to impoundment were quantified theoretically using volume exchange calculations and experimentally with a tracer study:

- The 95 % water renewal time theoretically increased from 28 h (>1 d) to 129 h (>5 d) at spring tides and from 65 h (<3 d) to 437 h (>18 d) at neap tides.

- Experimentally-determined 95 % water renewal times increased from 45 h in May 1990 to 72 h in July 1994.

- Since water renewal could not physically be more efficient than as predicted by volume exchange, a combination of tracer decay and settling coupled with considerable non-mixing was adjudged to have caused the discrepancy.

- Vertical mixing was reduced by a factor of 5 after impoundment. Thus, the surface waters were renewed over several tidal cycles whilst the bottom waters were retained for over a week.

The hydrodynamic markers temperature and salinity helped to identify several important processes that were perturbed by impoundment:

- Short-term (semi-diurnal and diurnal) variability driven by the tidal cycle and environmental conditions revealed a tidal shunting mechanism that prior to impoundment introduced contaminated water from the Plym into the harbour during the first half of the flood tide. This process was largely curtailed by the operation of the lock gates post impoundment.

- Mean water temperatures increased by 1 °C in the summer after impoundment. However, the increased mean volume buffered changes in water temperature that previously occurred on a weekly (spring-neap) timescale. Similar buffering of salinity variations occurred after impoundment due to the decreased tidal range. These processes reduced the potential for weekly water quality variations.

• The limited winter data available suggest that depth-averaged salinities were reduced after impoundment. There are implications for the retention of contaminants in the harbour bottom waters and particularly for the biogeochemical cycling of nutrients (Section *6.1.2*).

6.1.2 Biogeochemical cycling of nutrients between water and sediments

Nutrient fluxes to Sutton Harbour come from several different sources which were identified and partially characterised during the study:

• The Plym Estuary is the main source of TON, and a major contributor of orthophosphate to the harbour. Concentrations of these nutrients are inversely correlated with salinity, and therefore exhibit a strong seasonality, with winter maxima and summer minima separated by 2 to 3 orders of magnitude.

• Strong evidence suggests that the sediment porewaters within the harbour, and the localised sewage outfalls at Fishers Nose and West Hoe, are the main sources of ammonium and are minor contributors of orthophosphate to the harbour. Ammonium and UIA exhibited less seasonality, which is consistent with their source as sewage effluent.

• Speciation of ammonia demonstrated evidence of perturbation after impoundment. There is possibly a tendency for UIA concentrations to increase, owing to the effect on equilibrium of increased temperatures and decreased salinities. However, this problem should be self-correcting after commissioning of the Plymouth Area Sewage Scheme.

Pathways of biogeochemical cycling of nutrients between water column and sediments are shown in Fig. 3.1. These mechanisms are important, in that retention of a greater quantity of less saline water in the bottom layer of the harbour will increase the concentration gradients which drive storage of nutrients in the porewaters during winter. These are then released in summer. However, if discharges of organic-rich materials decrease under the Sewage Scheme, the role of the sediments as a nutrient source may decline.

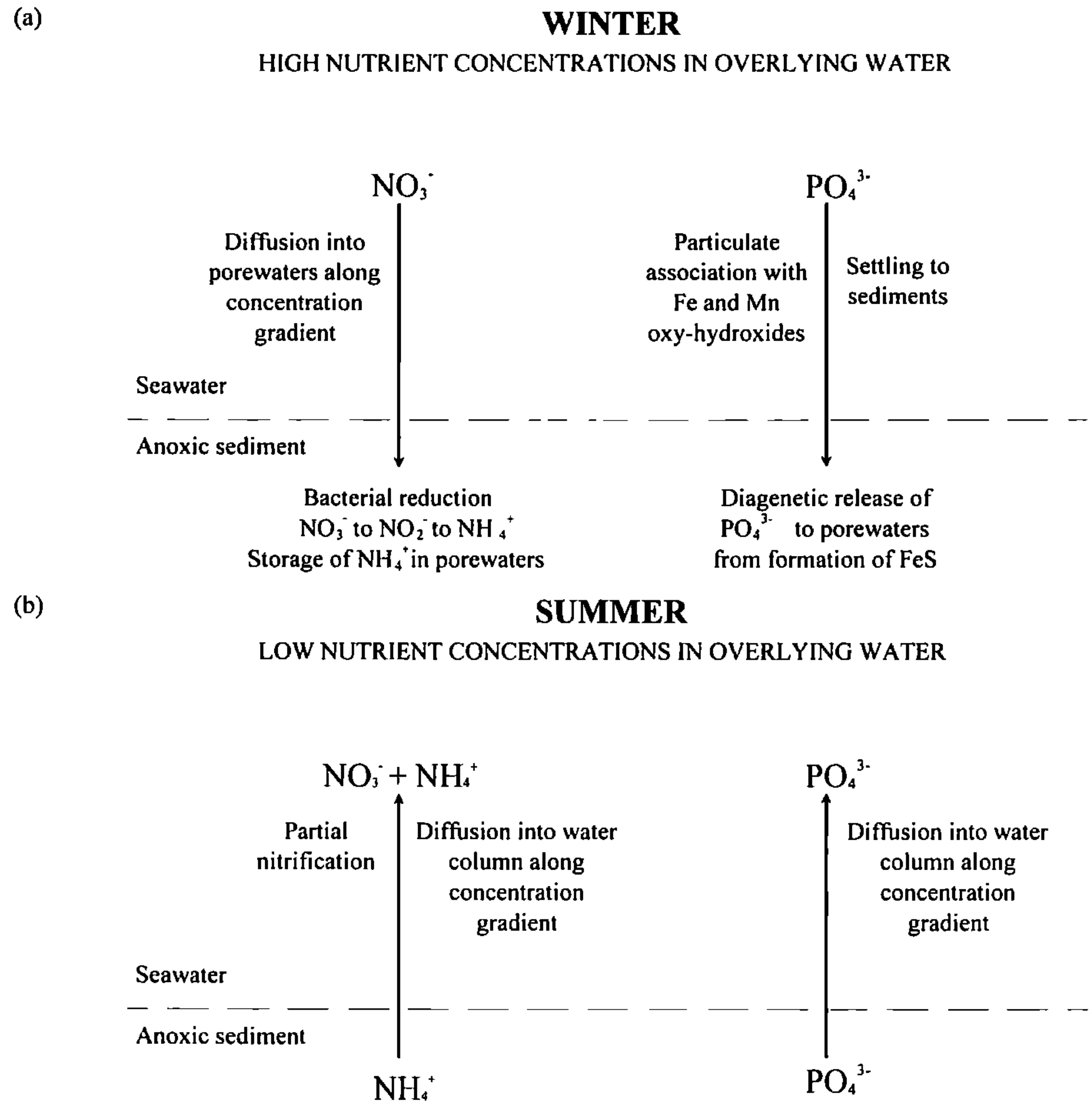


Fig. 6.1. Pathways of net biogeochemical cycling of dissolved nutrients between water and sediments in (a) winter and (b) summer.

6.1.3 The potential for and impact of eutrophication

This study has shown that primary productivity in the harbour follows an established pattern of seasonality, although the intensity of phytoplankton blooms exhibits considerable inter-annual variability, depending upon total insolation, and nutrient availability:

- Phytoplankton blooms occurred between May and September, and before impoundment were intense but short-lived, generally decaying or disappearing from the water column within one week.

- Limited summer data after impoundment show that the harbour is able to sustain a considerable bloom for longer when general conditions do not favour a bloom outside the harbour.

- Oxygen sags persisted in the harbour from August to April before impoundment, resulting from the degradation of organic matter in the epibenthic layer.

- Post-impoundment data suggest that the oxygen sag deepened, possibly due to the combustion of a greater phytoplankton biomass deposited to the sediments during the summer, coupled with the increased renewal time of the water column. The deeper oxygen sag has implications for the biogeochemical cycling of trace metals (Section *6.1.5*).

- Whilst the potential for primary production was undoubtedly greater inside the harbour during the first summer after impoundment, preliminary data from an additional period of monitoring between January 1995 and July 1995 revealed nutrient enrichment in winter but did not show enhanced primary production or nutrient enrichment during the spring and summer. Tentative comments are made in Section *6.3.3*.

6.1.4 Human health risks from microbiological water quality

The current study has shown unequivocally that the microbiological water quality of Sutton Harbour has improved as a direct consequence of impoundment:

- The main sources of contamination were shown to be the local sewage outfalls at Fishers Nose and West Hoe rather than the STW discharges in the Plym Estuary.

- The tidal shunting mechanism transported water from the area of Fishers Nose, with a high degree of sewage contamination at LW, to the harbour during the flood tide.

- The lock gates effectively prevented the influx of this microbially contaminated water which is now thought to be directed into the lower Plym Estuary, allowing less-contaminated water from the Sound to flow into the harbour during the late flood when the gates are opened to free flow. Bacterial counts were entirely within EC Bathing Water (I) limits after impoundment.

• Intermittent sources of sewage contamination were identified inside the harbour that were significant enough to reverse the gradient of bacterial counts from inside to out on occasions. One source may be what is thought to be a storm overflow beneath Sutton Pier and another appears to be a damaged foul sewer or storm overflow that leaks effluent through the quay wall in the NW corner of the harbour. Whilst these sources were not evident in the data collected after impoundment, they will continue to be active during periods of heavy rainfall and the effects will be exacerbated by impoundment.

6.1.5 Trace metal status of the water column and sediments

This study has revealed a significant reservoir of trace metals in Sutton Harbour, as a legacy of historic anthropogenic waste inputs and contemporary sewage and industrial discharges to the Plym Estuary:

• Concentrations of SPM in the harbour water column were low throughout the study, in the range 0 to 30 mg l^{-1}. Winter values lay in the upper half of this range and summer values lay in the lower half. However, the time series were perturbed by construction activities that resuspended considerable plumes of sediment giving manifold increases over background SPM values.

• Low SPM values resulted in a sparsity of particulate metal data. Where these were obtained, they gave good agreement with concentrations in the fine sediment fraction and supported the hypothesis of continued metal transport to the sediments.

• Dissolved concentrations of Cd, Cu, Hg, Pb and Zn were below the EQS values throughout the current study except for in May 1992 and November 1992 when sediment disturbances during construction may have caused this perturbation. The harbour water quality remains entirely satisfactory with regard to dissolved trace metals, although increased benthic anoxia may in future cause diffusion of reduced metal species from the sediments into the water column.

- The sediments have undergone considerable physical disturbance during the study, proportional to the proximity of the harbour works. Since impoundment, the sediments have stabilised to an extent and there is no evidence to suggest a significant accretion of fine sediment.

- Concentrations of Cd, Cu, Hg, Pb and Zn are all enhanced in the harbour sediments compared with contemporary measurements made in the Plym, and Cu, Pb and Zn have shown an increase in the fine fraction during the present study from 180, 210 and 330 $\mu g\ g^{-1}$ to 210, 320 and 410 $\mu g\ g^{-1}$, respectively, on top of considerable semi-annual variability.

- UK red list metals Cd and Hg did not increase in concentration over the entire study period, but varied in the range 0.8 to 5.0 $\mu g\text{-}Cd\ g^{-1}$ and 0.9 to 3.5 $\mu g\text{-}Hg\ g^{-1}$.

The deepening oxygen sags observed suggest by implication that reduced Fe and Mn are released by the sediments into the water column where they form oxy-hydroxide particle surface coatings that scavenge dissolved trace metals from the water column and deposit them to the sediments. However, this slow accumulation may take place over many years.

6.1.6 Condition of the benthic macrofauna

The benthic macrofauna in Sutton Harbour have been perturbed by the construction of the impoundment scheme as revealed by the semi-annual assays between November 1991 and November 1994. The database was analysed with traditional univariate analyses and more contemporary multivariate techniques that are becoming established in the literature. The main findings are detailed below:

- Univariate analyses showed that cirratulid worms dominate the soft-bottom macrofauna in the harbour. Species richness and diversity increased from the extremes of the harbour through the mouth to outside before and during the early stages of construction. Species diversity decreased at all sites inside and outside the harbour during the construction and early impoundment period and only showed an upturn outside the harbour at the end of the study, suggesting a partial recovery there but a new impoverished equilibrium inside the harbour due to impoundment.

- Multivariate analysis revealed significant changes in benthic macrofauna community structure that were greatest between samples taken before and during construction. The impacts were a general decline in species abundance, complete loss of more sensitive taxa such as Ampharetidae sp. and *Pariambus typicus* and abundance increases in opportunistic polychaete species such as *Heteromastus filiformis, Lumbrineris gracilis* and *Capitella capitata*.

- Correlation between the macrofauna similarity matrix and a PCA matrix of the environmental variables showed that a combination of Pb and Zn concentrations best explained the changes in community structure. Sediment disturbances during construction leading to blanketing of the benthos did not appear in the strong correlations, but may instead have increased the metal bioavailability to the benthic macrofauna.

6.1.7 Visual aesthetics

The shared use of the harbour as a centre for sea-going and land-based commerce and recreation was seen to be deleterious to the visual amenity of the harbour waters after thirty-two surveys over a period of three years:

- Litter from the quayside collects in certain areas depending upon wind direction. Clean-up operations observed after impoundment used this phenomenon to good advantage and must continue to ensure the visual amenity for all.

- Consideration should be given to preventative measures such as the provision of more litter bins and/or more regular emptying of the existing bins to prevent overflow. Furthermore, a localised anti-litter campaign could be organised around the findings of a recent study of litter decay in Sutton Harbour (Denyer, 1993).

- Litter was often observed entering the harbour *via* the harbour entrance. The installed bubble barrier was ineffective in this regard, although it may provide beneficial oxygenation. Whilst the lock operation may theoretically halve the ingress of litter, the ponding of litter by the stop-log gate during the early flood provides a concentrated litter source to the harbour during the late flood. This litter could be collected as an extension of the routine clean-up inside the harbour.

• Diesel slicks are a frequent and extensive problem which may have worsened following impoundment. Vessel operators should be asked to take more care when refuelling inside the harbour, and the routine pumping of oily bilges should be actively discouraged. More generally, commercial and recreational vessel operators should be encouraged to have more thought for their surroundings before emptying ashtrays, discharging heads, discarding fish, littering and spilling paint into the harbour waters.

• Overall, the visual amenity of the harbour has improved following impoundment, not least because of the permanent submergence of previously intertidal mud flats, but also the clean-up operations and the reduced ingress of litter. However, further improvements are possible through public awareness campaigns and greater vigilance.

6.1.8 Conceptual representation of the environmental impact of impoundment

The time series presented in Figs. 6.2a to c and 6.3a to d are intended as an aid to water quality management in Sutton Harbour and are based on the hypothesis that inter-annual variability is generally slight. They were compiled by depth- and spatially-averaging all data collected inside the harbour before and after impoundment, and therefore provide an instant and simplified indication of anticipated mean values of key variables at any point during the year. They also illustrate the principal perturbations to the established seasonal cycles observed. Data obtained between January 1995 and July 1995, during an extension to the present study, have been included in the post-impoundment averages for completeness. However, it should be noted that post-impoundment data points for September, November, January and March still only contain values for one year:

• Temperature (Fig. 6.2a) increased in the harbour after impoundment, with higher summer values developing by July, which then persisted throughout the winter months until the following March, when temperatures again began to rise. The temperature elevation recorded in November after impoundment, of 2 °C over the mean value before, occurred despite the mediocre summer of 1994. This may reveal a change in the heat budget of the impounded system, caused by partial isolation from the surrounding water.

• Salinity (Fig. 6.2b) has decreased during the winter months after impoundment, with lower values persisting between November and May. This has important implications for

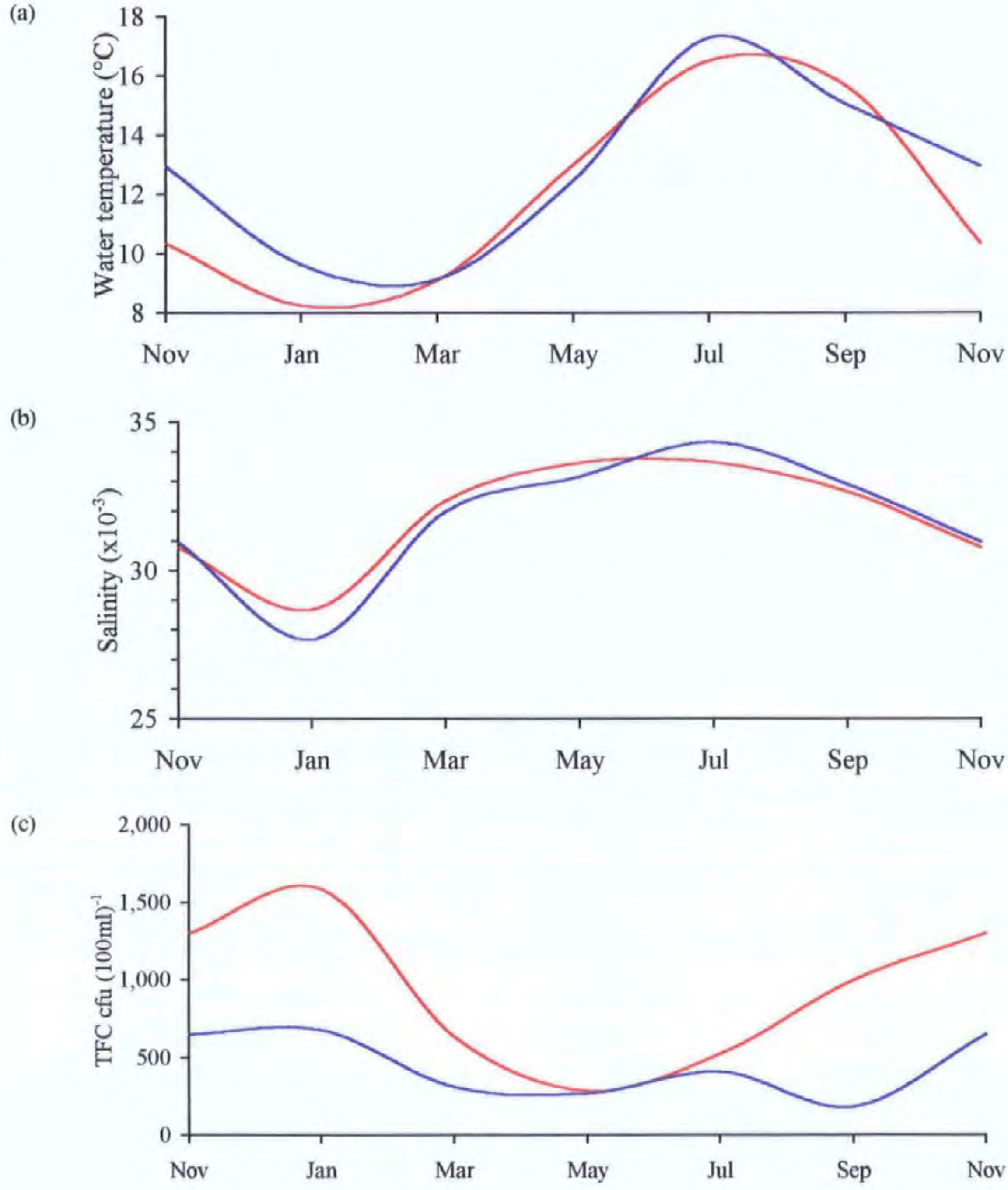


Fig. 6.2. Conceptual time series of (a) water temperature, (b) salinity, (c) SPM and (d) faecal indicator bacteria (TFC) inside Sutton Harbour before impoundment (red line) and after impoundment (blue line).

the enhanced storage of nutrients in the sediments as previously discussed. The opposite condition persists between June and November.

- Faecal indicator bacteria (Fig. 6.2c) represented by TFC have shown an unequivocal decrease after impoundment particularly during the winter months.

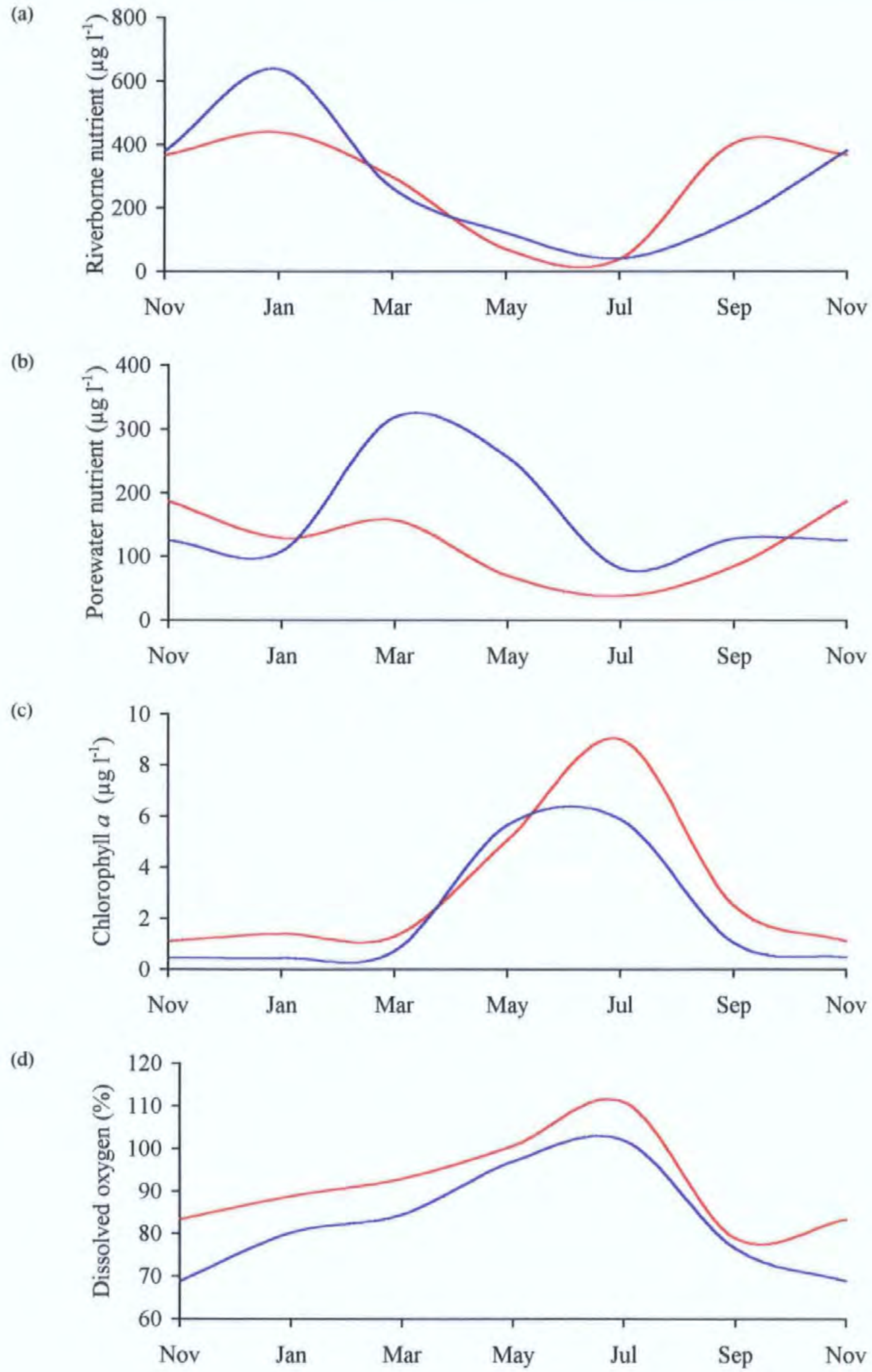


Fig. 6.3. Conceptual time series of (a) riverine nutrient (b) porewater nutrient, (c) chlorophyll *a* and (d) dissolved oxygen inside Sutton Harbour before impoundment (red line) and after impoundment (blue line).

• Levels of riverborne nutrients (Fig. 6.3a) represented by TON exhibit an increase in concentration during winter and are largely the same as before impoundment during the spring and early summer. They now do not appear to increase again until later in the year.

• Porewater nutrients (Fig. 6.3b) represented by ammonium also exhibit increased concentrations during the winter possibly as a result of *in situ* reduction of higher concentrations of TON. Values are greater in summer owing to increased release of ammonium from the sediment porewater reservoir.

• Chlorophyll *a* concentrations (Fig. 6.3c) were generally lower than before impoundment, although it must be noted that favourable bloom conditions were not encountered inside or outside the harbour after impoundment and that therefore these data are not necessarily representative of the general case. The modest blooms that were observed after impoundment now appear sustainable over a longer period.

• Dissolved oxygen saturation after impoundment (Fig. 6.3d) has shown a marked decrease throughout the seasonal cycle and is particularly low from September to March, the harbour only becoming fully saturated for a brief period during July. This reflects an increased retention of organic matter after impoundment, particularly in the bottom waters and at the sediment surface.

6.2 Study Critique

The three year multi-disciplinary study formed by this thesis represented *circa* 1 % of the cost of the Sutton Harbour scheme, but was considered essential by the NRA in order to ensure that impoundment did not prove deleterious either to the harbour, or the controlled waters of the Plym Estuary and Plymouth Sound. In the course of the study, many factors became apparent which might be used to design more effective survey and monitoring plans, enabling further study of the key processes affecting the ecology of the harbour.

6.2.1 Survey strategy

Except for TBT determinations, the initial survey strategy (Fig. 2.1) conformed rigorously with the original NRA requirements (NRA, 1991) and was enhanced after extensive consultation during the project. Design criteria in terms of the original NRA requirements and the enhanced survey strategy *versus* the ideal strategy, that are equally applicable to any such study, are examined in Table 6.1. Each issue is discussed below in detail:

- The spatial resolution achieved by monitoring three stations within the harbour was adequate under the relatively homogeneous conditions encountered, thus facilitating spatial averaging of the data and the subsequent treatment as surface and bottom water masses. However, greater heterogeneity in other systems would require more sampling stations.

- The temporal resolution achieved through the timing strategy highlighted a variety of short term processes (daily and weekly) and longer term processes (seasonal and inter-annual). However, the database would have benefited from the detection of processes between these timescales. The unifying factor in all processes was time and it is considered essential to tie in the sampling strategy with the different frequencies of events in the system. A summary of the major processes is presented in the form of an ecological spectrum in Fig. 6.4, together with suggestions for the optimal timing of observations. Once-daily measurements of master variables (particularly salinity, dissolved oxygen and chlorophyll *a*), taken with water monitors throughout the study period inside and outside the harbour would have provided a better characterisation of inter-diurnal trends (such as rapid growth and decay/disappearance of phytoplankton blooms) and intra-seasonal trends (such as successive periods of growth and decay) that were observed in the continuous water monitoring database between February 1993 and September 1994. These would have been complemented by a more responsive survey framework to detect and characterise intermittent events, although this is logistically difficult with the mobilisation of human and physical resources. It is not practical to have resources dedicated to such a project because of expense. However, it would be beneficial when planning the survey pairs to remain flexible in the timing of the latter survey dependent on the observations of the former. The frequency of the paired surveys could be decreased from bimonthly to trimonthly, representing a significant saving of cost and effort whilst still maintaining the required resolution of seasonal cycles in concert with the daily automated monitoring. Furthermore, consideration could be given to decreasing the frequency of benthic monitoring from semi-annually to annually coupled with increasing the number of sites or the amount of replication for physical, chemical and biological assays. This would remove the natural seasonal variations but retain the long-term trends and enhance the data quality.

Table 6.1. A comparison of the original and enhanced survey strategies employed in Sutton Harbour *versus* an ideal strategy devised with hindsight but with a similar cost.

Original Strategy	Enhanced Strategy	Ideal strategy
• Four stations (three inside; one outside).	• Five stations (three inside; two outside).	• Sufficient stations to give good spatial coverage particularly under heterogeneous conditions. • Two control stations (one near-field and one far-field) to ensure availability of true control data.
• Bimonthly water column and semi-annual sediment surveys. • Provision for extreme weather surveys as necessary.	• Additional higher resolution microbiological surveys and a tracer study of water renewal post-impoundment. • Extended period of continuous water monitoring at key sites inside harbour.	• Additional continuous water monitoring throughout the study at a key station (inside) and a control station (outside), but at a lower frequency (for example, daily) to limit data overload. • Responsive survey framework to characterise intermittent events such as river spates and blooms.
• Chemical and microbiological sampling at surface and bottom coupled with *in situ* profiling measurements at surface, bottom and intermediate depths.	-	• *In situ* measurements need only be recorded at surface and bottom (in concert with samples) in shallow (<10 m) regions. • However, intermediate depth measurements do not entail extra effort or cost and may be useful.
• Tracer study of water renewal conducted (pre-construction) as a component of the May 1990 environmental assessment. • Initial spore dosing at LW preceding the baseline HW.	• Tracer study of water renewal post-impoundment to verify volume exchange calculations. • Initial spore dosing during impoundment preceding the baseline HW.	• Initial spore dosing at HW preceding the baseline HW (or surface and bottom dosing at LW preceding the baseline HW) to ensure adequate mixing and dispersion of the spores.
• Determination of faecal indicator bacteria TC, TFC and FS.	-	• Omission of redundant TC determination (where it is not required under EC Bathing Water Directive) in favour of increased spatial resolution of TFC and FS or determinations of *Salmonella* or viruses.

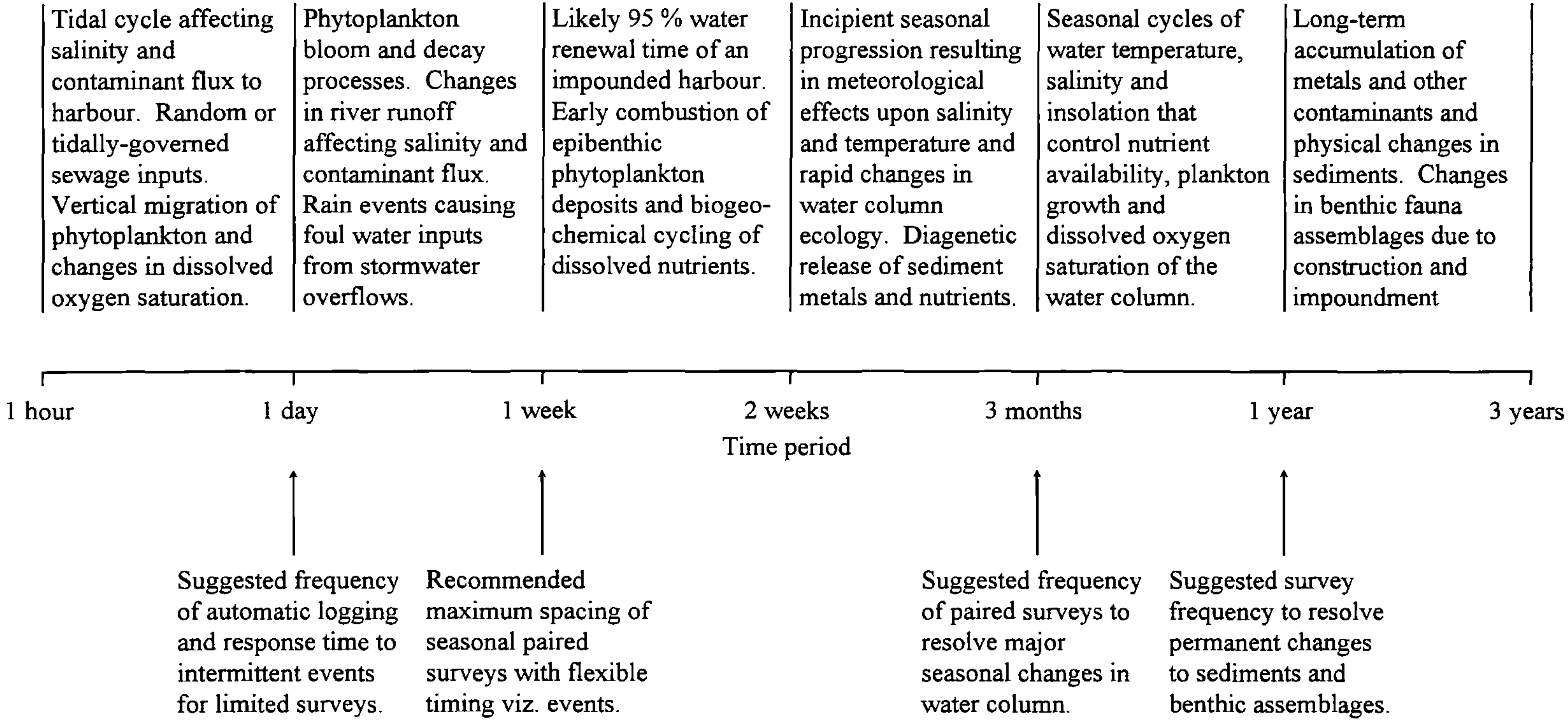


Figure 6.4. Ecological spectrum showing the timescales of the major processes affecting water and sediment quality in an impounded harbour and suggested survey timing to achieve the necessary temporal resolution for their study.

The surveys conducted in addition to the routine monitoring requirements provided valuable supplementary information on the altered flushing characteristics of the harbour and on additional sources of sewage contamination:

• The higher resolution bacteriological surveys provided valuable supplementary information on additional sources of sewage contamination and revealed hotspots that would otherwise have remained undetected despite three years of monitoring. However, this intense period of study still did not discern whether yacht heads are a significant source of sewage contamination. This information would only be obtainable by an intense investigation around the marina pontoons tantamount to snooping.

• Several issues arose from the study of water renewal, most importantly the non-conservative behaviour of the bacterial spore tracer, *Bacillus globigii*. Users should be aware that the spores exhibit varying rates of mortality in seawater depending upon intensity of insolation and a considerable settling rate in still water. Given sufficient depth of water (>2 m) into which the spores mix down away from UV rays, it was concluded from a combination of published data and laboratory experimentation that the spores would behave near-conservatively for up to ten days, although shorter timescales are recommended irrespective of the initial dosing concentrations. Improvements in the dataset from July 1994 over that from May 1990 showed that it is important to remain flexible in the timing of sampling runs and to adapt to the emerging flushing rate to ensure sufficient spread of data along the decay curve. Inadequacies in spore dosing were evident resulting in rapid tracer removal during the first ebb tide. Several solutions were postulated, including dosing at the preceding HW (HW_{-1}) before baseline measurements at HW_0 to allow a full tidal cycle of mixing and dispersion and simultaneous surface and bottom dosing at the preceding LW (LW_{-1}) combined with dosing at the harbour mouth between LW_{-1} and HW_0 during the flood tide. Of these, dosing at HW_{-1} is the most direct solution, but might require additional spore quantities to balance early losses, thereby incurring higher costs.

Several improvements could be made to the sampling methods and types of samples collected, improving the quality of the database whilst reducing the data burden:

• The *in situ* master variables were measured as water column profiles ($\Delta Z \leq 2$ m), resulting in two to five (usually four) sets of measurements for each station during each survey run. However, the chemical and microbiological samples collected along with these measurements were recovered near-surface and near-bottom. Interpretation of the samples in the context of the master variables thereforeled to data from intermediate depths (*circa* 50 % of the master variable dataset) being largely disregarded. Omission of the intermediate depths would not have speeded up survey operations, since these were measured whilst the first samples were being hand-filtered and/or decanted into bottles. However, it would have considerably reduced the size of the database and the burden of data input. Future workers might consider the worth of vertical profiles of master variables during large-scale ecological programmes, particularly in shallow inshore waters (<10 m) and where sampling is only conducted at surface and bottom.

• In addition to excessive *in situ* measurements, savings of time, effort and cost could also have been made during microbiological determinations. Counts of TC correlated strongly with counts of TFC in the present study, but they are not generally considered to be effective indicators of human health risk owing to a lack of correlation in epidemiological studies (Cabelli *et al.*, 1983). Future investigations where monitoring of TC is not obligatory might consider omitting this variable (representing one third of the microbiological effort in the present study) in favour of increased spatial resolution, of greater replication of TFC and FS determinations, or even measurement of *Salmonella*, in order to provide a better indication of human health risk.

• In the present study, standardisation of methods of sampling of benthic fauna was complicated by the involvement of three different organisations (in May 1990, November 1991 and May 1992 to November 1994). Uncertainties exist owing to the use of the Day grab, unit area sampling by diver, and the Shipek grab. Whilst area and hence abundance per m^2 could be standardised, species composition between epifauna, shallow- and deep-dwelling infauna could not be compared with certainty. Future studies should ensure application of a standardised technique (preferably Day grab; see Warwick, 1983) and constant replication (at least threefold) throughout the study in order to ensure data compatibility. Furthermore, the involvement of different personnel during identification raised further complications where difficult classifications to species level were attempted. The database was rigorously screened in order to try to counteract this effect. Any ambiguities at species level were resolved at genus or family. Moreover, future practitioners should weigh the cost of identification to species against the need to identify beyond family, particularly where multivariate analysis is proposed (Section *6.3.5*).

6.2.2 Database management

One of the great challenges of the present study was efficient management of an extensive database comprising *circa* 16 MB of raw data. Microsoft Excel (v. 4.0/5.0) was chosen for recording and manipulating data in spreadsheet form. Lateral thinking was required at times in order to overcome the bias of this software towards business applications, particularly in the production of certain hybrid figures. A specialised scientific graphing package may have provided more straightforward solutions, but this was compensated for by the powerful analysis suites, high data capacity, universal data import/export capabilities and compatibility with Microsoft Word offered by Excel. The chosen spreadsheet approach to data management afforded several advantages and disadvantages:

- Data were stored in a high-quality tabulated form which did not require further modification before direct reproduction in the routine survey reports.

- The spreadsheet format allowed automatic calculation of complex formulae (such as the Cox equation for seawater density and the Weiss gas solubility equation for dissolved oxygen saturation), and provided a convenient means of applying instrument calibrations to raw data.

- The infrastructure of each complex table was saved many times over, causing inefficient use of disk space. However, modern mass storage devices provided an adequate solution to this problem. Moreover, the database was conveniently broken down into manageable units which could easily be downloaded, backed up or copied.

- There was a high level of repetition and duplication of effort. Many different spreadsheets were produced which used the same data in slightly different formats or combinations. For example, the layout which facilitated tabulated data presentation was not, in many instances, suitable for data analysis routines such as correlation. Further layouts were then required for graphical presentation.

Future practitioners might take inspiration from the NERC North Sea Project CD-ROM (Lowry *et al.*, 1992) which provides 'kit-form' data in a series of flat ASCII files for export to a relational database management system (RDBMS). In an RDBMS, all data structures are reduced to two-dimensional tables consisting of rows (records) and columns (fields) and are interrogated using structured queries (Wass, 1989) to extract only the pertinent information for each data application. Each record would consist of a unique sample code followed by all data pertaining to that sample. This approach would have been ideal for the current study had it been adopted at the outset. However, the benefits of switching to this approach mid-way through the study could not have justified the time-consuming transfer process. The advantages and disadvantages in using an RDBMS for such an application are listed below:

- Any combination and permutation of variables could be extracted for statistical analysis, offering universal flexibility in data handling.

- The burden of data input could be passed on to technical or clerical staff with no need for specialist knowledge of individual spreadsheet complexities.

- The user would need to be certain that the data format would not change; a reasonable assumption in a long-term monitoring programme that could be ensured by waiting until several of the initial surveys were completed satisfactorily before designing the database.

- The central database file would be relatively large and unwieldy, requiring permanent storage on a dedicated PC with backup facilities other than floppy disks, such as a magneto-optical drive or tape streamer. The logistics of this may be prohibitive, depending on the circumstances of individual studies.

6.2.3 Data analysis and presentation

One of the major achievements of the present study was the synthesis of pertinent environmental information from a large multidisciplinary database, essentially the resolution of signals from noise. In all scientific studies, the rate of data collection must be optimised to provide the required resolution of measurement (spatial and temporal) without causing an overload of data. There is a temptation to collect more data than is necessary and whilst this is certainly better than not enough, the worker can be faced with time consuming data input and reduction techniques at the expense of data interpretation. Throughout the present study, several areas of potential data saving were identified and some innovative techniques were adopted to deal with data overload:

- During the early stages of the study the decision was taken to analyse data on semi-diurnal, diurnal, spring-neap, seasonal and inter-annual timescales. The short timescales were considered using unmodified data from each station at each tidal state, whilst the long timescales were analysed as time series generated from spatially- and tidally-averaged data to provide more stable comparative information. This approach to data reduction was justified because observed intra-harbour variability was considerably less than differences between inside and outside the harbour.

- Correlation analysis was employed extensively to test the dependence of key contaminants such as dissolved nutrients upon the variation of master variables such as salinity. In this way, important processes of contaminant flux to the harbour were statistically qualified. Correlation also suggested common major sources of contaminants, such as Zn and Cu due to marina operations and TC and TFC from local sewage outfalls.

- The size of the continuous water monitoring database (*circa* 30,000 records/rows) presented particular problems because Excel can process only 16,000 rows of data and 4,000 data points on a graph. Therefore, a novel technique was employed to reduce the data and produce the required length in the seasonal time series. A DOS command line programme, used to process electronic current meter data (Tyler, pers. comm.) was rewritten to process the database. The programme consisted of a Doodson X_0 filter that produced high pass and low pass filtered channels separating cycles either side of a 19 h period, such that the low pass data was effectively stripped of semi-diurnal variations and diurnal variations were significantly reduced. The resulting database was then condensed five-fold by averaging over the previous two and following two data points (2½ h) to produce a time series that retained significant trends but could be effectively manipulated in Excel.

- The decision to take advantage of the latest software, PRIMER, for multivariate analysis of the benthic macrofauna database provided a powerful technique of data reduction and analysis. The traditional univariate measures such as species richness, evenness and diversity were shown to have limited value and could not adequately explain the combination of subtle and dramatic changes in community structure resulting from construction and impoundment. These explanations were provided by multi-dimensional scaling, which reduced the complex inter-sample relationships between *circa* 100 species to a two-dimensional ordination that could be more easily interpreted. Furthermore, the multivariate analysis provided a means by which changes in community structure could be correlated with changes in the measured physical and chemical sediment variables, thereby suggesting a best explanation of the causative factors for community change. This approach is becoming established in the literature (Agard *et al.*, 1993; Craig *et al.*, 1993; Somerfield *et al.*, 1994a and 1994b) and is strongly recommended to future practitioners.

6.3 Strategic Planning Recommendations for Future Harbour Management

One of the most important findings of this study is that inter-annual variability in environmental master variables is relatively small in semi-enclosed marine and estuarine water bodies. Relatively little attention appears to have been paid to this subject in the literature, but the consistency of behaviour of these environments is crucial to the effective management of water quality in them. Those responsible for water quality are now able to confidently predict what conditions will prevail in any particular season and adapt operations to minimise the perturbation caused. The following sections detail the recommendations arising from this study and indicate those principles that will apply to other schemes in the future.

6.3.1 Frequency, optimal timing and duration of flushing events

It has been shown that the current lock operating regime provides less efficient flushing of the harbour waters, particularly near bottom, than before impoundment. The following recommendations are made to ameliorate any potential problems arising from this reduction. These are also presented as a flow chart in Fig. 6.5, coupled to conceptual models of general water quality and the nutrient/phytoplankton subsystem in order to illustrate their reliance upon the frequency of flushing.

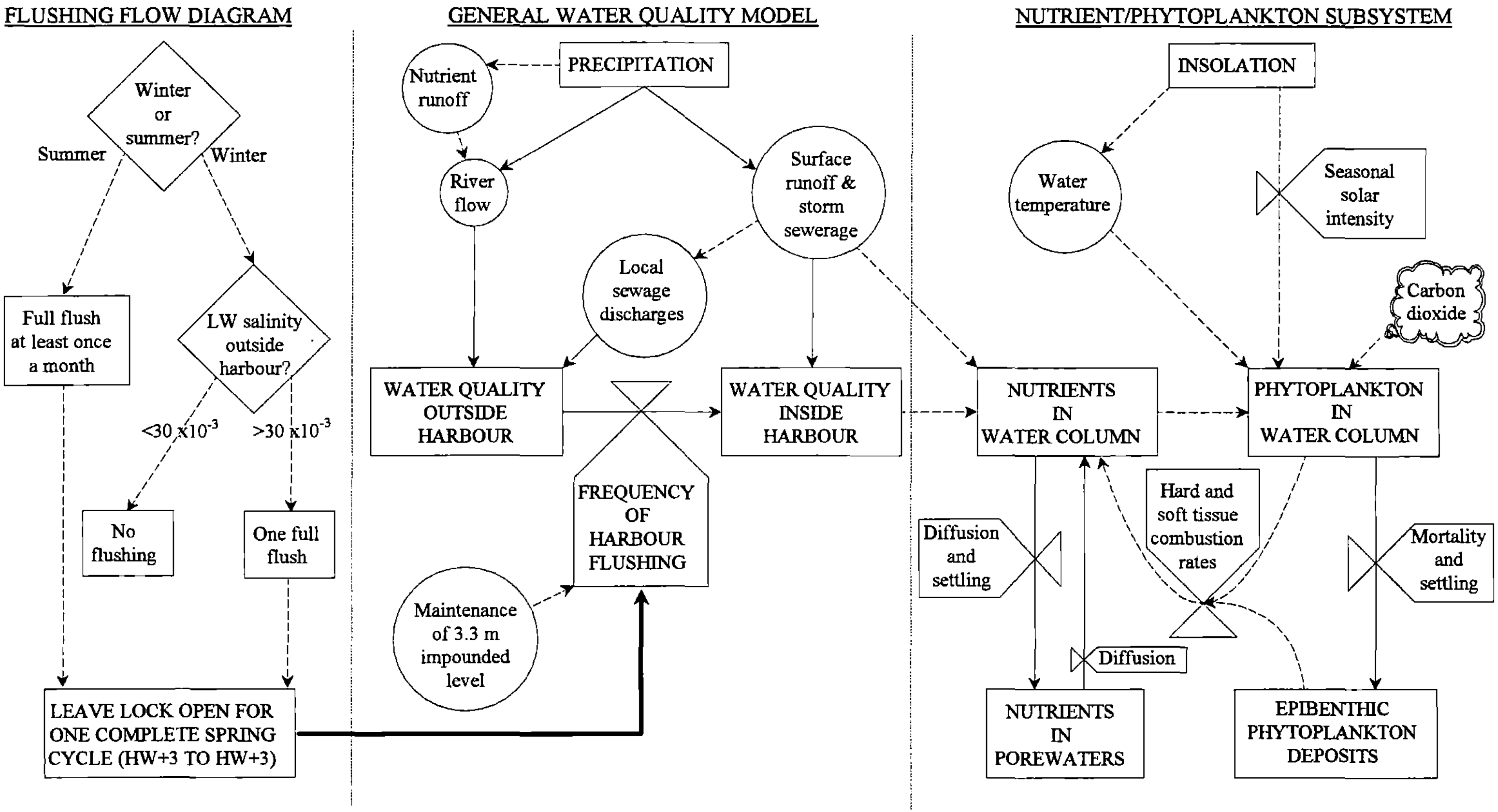


Fig. 6.5. Management strategy flow diagram for flushing linked to conceptual models of general harbour water quality and the nutrient/phytoplankton subsystem. System Dynamics notation is used for the conceptual models: rectangles are measurable quantities; valves are rates that affect these; circles are auxiliary variables; clouds are sources that are not affected by the model; solid arrows are fluxes; broken arrows are causal relationships. Pivotal regions of the model are denoted by capitals.

- More attention should be paid to maintaining the prescribed mean impounded level of 3.3 m in any 14 d period. The straightforward rule-of-thumb suggested in Section 3.1 would help in this regard.

- A minimum of one routine monthly flushing during the summer months (May to September), effected by opening the lock gates to free flow throughout an entire spring tidal cycle, would significantly improve the quality of the harbour bottom waters. This should obviously take place overnight to minimise disturbance to daytime harbour operations.

- Additional intermittent flushing events should be considered whenever a dry period occurs in the winter months. High salinity outside the harbour at LW would indicate low nutrient concentrations and flushing with this water would reduce the potential for storage of nutrients in the harbour sediments that become available during the bloom period.

- Any flushing during wet periods in the winter months should be actively discouraged, as this will reintroduce water with high concentrations of nutrients and sewage bacteria. In fact, proposals to restrict flushing still further during the winter months are discussed in Section *6.3.5*.

- The data suggest no current need to install the proposed system of gravity recirculation pipes to promote circulation in the harbour arms.

6.3.2 Modification of harbour operations to minimise contamination

This study has raised several issues with regard to the variety of human uses of the valuable harbour resource that are worthy of final comment and recommendations here:

- The storm sewer located under Sutton Pier and the suspected leak from a damaged sewer in the NW quay wall should be investigated and remedied as soon as practicable. These now pose a particular health risk due to their proximity to Sutton Marina.

- Public awareness campaigns should be mounted to restrict the ingress of litter from the harbour periphery, prevent the flushing of yacht heads within the confines of the harbour and promote careful handling of refuelling operations to limit the inputs of diesel.

- Vessel operators should be made acutely aware of the potential environmental damage that certain practices may cause. Activities such as hull painting that introduce significant contaminants to the harbour waters and ultimately to the sediments should be discouraged. Moreover, the historic use of the harbour as a repository for scrap that has undoubtedly contributed to the high heavy metal concentrations observed must be curtailed.

6.3.3 Potential long-term concerns for water and sediment quality

Conclusions drawn from the current database suggest that the harbour ecosystem has rapidly adapted to and absorbed the changes imposed upon it by the impoundment. The findings of this study indicate that the preconceptions of the author and the NRA, that the harbour water quality may suffer drastically, have been dispelled. The positive aspects of impoundment such as the improvement to visual aesthetics and microbiological water quality are to be welcomed. However, there remain some concerns that may evolve on a timescale longer than the current study, such as have occurred in the Haringvliet (SW Netherlands):

- The sediments are suspected to be accumulating contaminants, particularly trace metals, at an enhanced rate. Operational dredging has not been carried out since before construction began in 1992 and may therefore be necessary within a matter of years to maintain the benefits of the deep water facility. The disposal of these highly contaminated sediments will then pose a problem, probably requiring an expensive landfill solution. Some of the sediments could have been incorporated into the back-fill material used to reclaim the land during the construction, representing a permanent solution. All remaining options for disposal must be carefully considered.

- The potential for the development of eutrophic conditions (Section *1.1.1*) in the harbour does not appear to have been realised in the short term. However, nutrient concentrations are certainly higher during the winter months than before impoundment and it is reasonable to assume that the mechanisms for nutrient storage discussed are valid. It therefore remains to be seen whether eutrophication will occur in the longer term. In the event of eutrophication, the best solution will be an extended return to free flow conditions throughout the tidal cycle, in an effort to dilute nutrient concentrations and to disperse phytoplankton blooms. The NRA should make periodic inspections of the harbour during the high risk period from May to July so that immediate action can be taken by Sutton Harbour Company as necessary.

6.3.4 Applicability to future impoundment schemes

Water quality managers may now use this study as a model for future impoundment schemes which, depending on certain criteria, may apparently be constructed and operated without serious long-term consequences. However, considerable and lasting changes were observed in the sediments and benthos as a direct result of construction activity. Whilst this might not be vitally significant in a working harbour with historically-contaminated sediments and impoverished macrofauna assemblage, the consequences in a designated Site of Special Scientific Interest (SSSI), for example, would have legal implications enforcable by English Nature (Spellerberg, 1991).

In areas where there are known to be contaminated sediments, thought should be given to removing them before or during construction, possibly by incorporating techniques of contained aquatic disposal (CAD) and confined disposal facilities (CDF) described by Fraser (1993).

The current survey strategy (with the suggested improvements) and the various techniques of data management, analysis and presentation are to be commended to future practitioners as a cost-effective and efficient study regime. In particular, multivariate analysis is considered essential in order to enable coherent trend detection. However, if such a large scale study is not obligatory or feasible, then certain key investigations should still be made:

Tracer studies of water renewal should be conducted before and after impoundment in order to complement the volume exchange calculations from the design stage. This will facilitate management of lock operations in order to provide a level of flushing efficiency at least equal to the present case.

- Regular (perhaps daily) measurements of key parameters salinity, dissolved oxygen and chlorophyll *a* should be automatically recorded in the centre of the harbour and outside the entrance, ideally for at least a year either side of commissioning, to ensure that key features such as winter salinity minima, winter oxygen sags and spring/summer blooms are not dramatically perturbed by impoundment.

- External contaminant fluxes will almost certainly be reduced by impoundment. However, internal contaminant sources will become more significant and should be characterised and addressed to reduce the risks of human health risk and/or eutrophication. If internal sources are not significant, then spatial homogeneity may be assumed in a harbour of equivalent area to the present case and investigative monitoring may be conducted at one location only.

- The harbour sediments should be characterised, particularly with respect to List One substances. If application has been made to MAFF for a licence to dump dredged spoil, then this will be conducted as part of the licensing process.

6.3.5 Recommendations for further study of Sutton Harbour

Clearly there must be a finite limit upon the length of such monitoring programmes, although continued study may still reveal facets of the evolving ecosystem that were not apparent within the present timescale. Therefore, some measurements should be made on a routine basis, particularly around the period of major interest between May and July. Future work that could be conducted includes refinement of the ECoS Plym model with the inclusion of the harbour, and the benthic macrofauna database could be reanalysed by reducing all taxa to family level, as proposed by Warwick (1988), to test whether the same conclusions could be drawn from a dataset that would have been enumerated and identified far more rapidly and cost-effectively.

With the agreement of Sutton Harbour Company, it would be extremely interesting and possibly beneficial to conduct some *in situ* water quality and lock operation experiments during the winter period when vessel traffic is at a minimum, with a view to preventing the influx of dissolved nutrients. Over a period of *circa* one month, the free flow phase of the lock cycle could be reduced to a minimum by closing the lock gates at HW and only opening them on the following late flood tide to replace any losses. During this period, measurements could be made around the harbour to assess (i) the effect upon dissolved nutrient concentrations compared with those outside the harbour, (ii) the effect upon the dissolved oxygen sag and (iii) the definitive explanation of the alternative sources of sewage contamination. Favourable results might suggest a radical reworking of the lock operating regime to minimise flushing in the winter period and to maximise flushing in the summer period with considerable improvements in water and sediment quality as a result.

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Appendix One

Published Work

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MONITORING AND MANAGEMENT OF WATER AND SEDIMENT QUALITY CHANGES CAUSED BY A HARBOUR IMPOUNDMENT SCHEME

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A long-term monitoring programme of water and sediment quality is being undertaken before, during, and after the impoundment of Sutton Harbour, Plymouth (UK). The impoundment scheme protects an area of the city from periodic flooding, but harbour access is allowed via a lock. The water replacement times have increased from 45 h to 72 h, at springs, with a consequent effect on the retention of pollutants from internal and external sources. The monitoring programme involved determination of essential variables (viz temperature, salinity, dissolved oxygen, pH, chlorophyll *a*, and suspended particulate matter, SPM, concentration) coupled with nutrients, faecal bacteria, trace metals in the water column and sediments, and benthic fauna. The results show that before impoundment, the Harbour had an annual cycle in biogeochemical variables that was perturbed only by intermittent meteorological events. The Harbour is contaminated with heavy metals; the permanently anoxic sediments show relatively high concentrations of mercury (0.5-2.0 mg kg^{-1}) and cadmium (1.0-5.0 mg kg^{-1}). During construction, significant variations were observed in bacterial distributions, heavy metal concentrations, and benthic fauna populations. The trends are discussed in terms of a management strategy for maintaining water and sediment quality, especially those aspects that have more general applications to other harbour impoundment projects.

INTRODUCTION

Ports and harbours have an enormous commercial value when viewed on a global scale. They represent safe anchorages for vessels and for cargo handling, and they are often sites of property development (Couper 1989). In some cases, harbours may be impounded to improve their commercial value, while in others, there may be a need to protect the valuable land from flooding. Harbour impoundment schemes have been carried out at several locations in the U.K. There will be an increasing need for such defences, either at the current rate of sea level rise of 1-2 mm y^{-1} (Hansen 1985), or at the Intergovernmental Panel on Climate Change (IPCC) best-estimate of global mean sea level rise of 18 cm by 2030 (Warrick and Oerlemans 1990). However, such schemes have to be properly planned and implemented with regard to statutory requirements to maintain water quality. The National Rivers Authority (NRA) is the competent body in England and Wales that is charged, under the Water Act 1989 (DoE 1989), with ensuring compliance with EC water quality directives, such as the discharge of dangerous substances directive (CEC 1976a), the bathing water quality directive (CEC 1976b), and the urban waste water treatment directive (CEC 1991). Despite this legislation, there are relatively few reported studies of the impact of impoundment on harbour water. The main body of evidence comes from the Netherlands, where the Eastern Scheldt, as an integral part of the of the SW Netherlands Delta Plan, is the only widely-researched example of an impounded saline water body. Following disastrous flooding during a storm surge in February 1953, the decision was taken to dam five of the seven estuaries in the SW Netherlands (van Westen and Leentvaar 1988). However, following concerns over deleterious effects on the extensive aquaculture in the Eastern Scheldt (Elgershuizen 1981; Smaal and van Stralen 1990), plans for the impoundment were revised, and a permeable storm surge barrier was constructed at the mouth, with two compartment dams up-estuary that redirect seaward runoff elsewhere (Scholten et al. 1990). The reduced freshwater inputs resulted in the creation of a permanently saline water body, with decreased tidal exchange with the North Sea, a situation similar to impounded harbour schemes.

An opportunity to study a system before and after impoundment could lead to a better understanding of the environmental impact of both, the construction and the subsequent operation of the system. Such an opportunity arose in the case of Sutton Harbour, Plymouth, UK (Fig. 1). The Harbour covers an area of $2x10^4$ m^2, accommodates a total of 500 fishing vessels and leisure craft, and is adjoined by the commercial area with Plymouth City centre lying to the northwest. It is a centre for land- and water-based recreation and commerce that is vital to the local economy. The Plym Estuary to the east has a catchment of 80 km^2, with a significant pollutant potential from domestic sewage and industrial inputs (Millward and Herbert 1981). Brackish water from the Plym, containing dissolved and particulate pollutants, is only partially dispersed within Plymouth Sound on the ebb tide, and pollutants can be pumped into the Harbour by flood tides. Furthermore, additional pollutant sources such as crude sewage outfalls at Fishers Nose and inputs from the two marinas (Fig. 1) complicate the role and significance of pollutants for the Harbour. The commercial area to the west (the Barbican) was prone to periodic flooding during exceptional spring tides and episodic flooding caused by storm surges in the English Channel (George and Thomas 1976). Flood prevention measures finally became economically viable when the National Rivers Authority undertook to impound the Harbour, in conjunction with the Sutton Harbour Company's plans for a new fish market. Engineering work began on this £ 10 million joint project in May 1992. Figure 1 shows the Harbour plan after construction of the impoundment scheme. The scheme comprises a lock structure to seal the Harbour entrance, whilst allowing continual access to vessels, together with large areas of reclaimed land upon which the new fish market is situated. The two-year construction phase was completed in April 1994, at which point the Harbour was impounded, thereby changing the water and pollutant retention times.

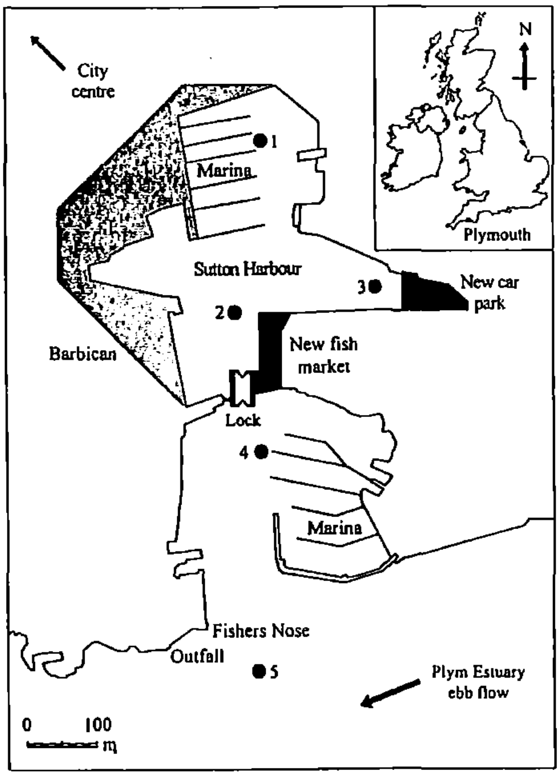


Fig. 1. Plan of Sutton Harbour, Plymouth (UK), after impoundment. The points 1 to 5 represent the routine monitoring stations. Black areas to east represent reclaimed land; shaded area to west represents region regularly inundated prior to impoundment.

The objective of this collaborative study was to acquire long-term data covering the period before construction, during construction, and after construction to design a management plan to improve and maintain water and sediment quality according to established criteria. The database is large. This paper will outline the monitoring strategy, representative data, and some pointers to the analysis of the data.

THE MONITORING PROGRAMME AND METHODS

The stations at which measurements were made and/or samples taken are shown in Fig. 1. Stations 1-3 represent conditions inside the Harbour, including the arms, station 4 represents conditions immediately outside the Harbour, whilst station 5 yields information on potential inputs from the Plym Estuary. The strategy adopted for the monitoring programme is shown in Fig. 2. Every two months, the primary variables were determined through the water column at high water, mid-ebb, low water, and mid-flood on consecutive spring and neap tides. Many of the measurements were made in situ using calibrated sensors (viz depth, temperature, salinity, dissolved oxygen saturation, pH, chlorophyll *a* fluorescence, and [SPM]). Near-surface and near-bottom samples for dissolved nutrients (ammonium, total oxidised nitrogen, and orthophosphate), faecal indicator bacteria, and biochemical oxygen demand were taken concurrently with the vertical profiles. Total coliforms, thermotolerant faecal coliforms, and faecal streptococci were determined using membrane filtration followed by incubation on selective media (DoE 1983). Un-ionised ammonia, which causes stress in marine life but cannot be determined directly, was calculated as a percentage of ammonium concentration from tabulated values dependent upon temperature, pH and salinity (Bower and Bidwell 1978).

At six monthly intervals, secondary variables in the water column were determined, to identify seasonal changes. These included dissolved and SPM concentrations of toxic metals Cd, Cu, Hg, Pb, and Zn. Dissolved metals were preconcentrated by a complexation-solvent extraction technique and analysed either by conventional atomic absorption spectro-photometry (AAS) or by graphite furnace AAS (Danielsson et al. 1978). The precision for all metals was in the range 10-15%. Particulate metals were determined in the same way following extraction in concentrated nitric acid. In concert with these samples, sediment samples were obtained by Shipek grab, and determinations of particle size distribution and benthic faunal abundance were undertaken. Sediment samples were fractionated across a nylon sieve (63 µm) and digested with concentrated ultrapure nitric acid to give total metal, and with 25% ultrapure acetic acid to yield a bioavailable metal fraction. Metal concentrations were determined by AAS and graphite furnace AAS, and the precision for all metals was <10%. Changes in the benthic faunal community structure were used in combination with the sediment metal data to assess the impact of the metal pollutants on the organisms. In addition to the routine monitoring programme, a continuous monitor was installed near the centre of the Harbour, which recorded temperature, salinity, pH, dissolved oxygen, and turbidity at half-hourly intervals. This enabled recording of short-term signals in the Harbour waters.

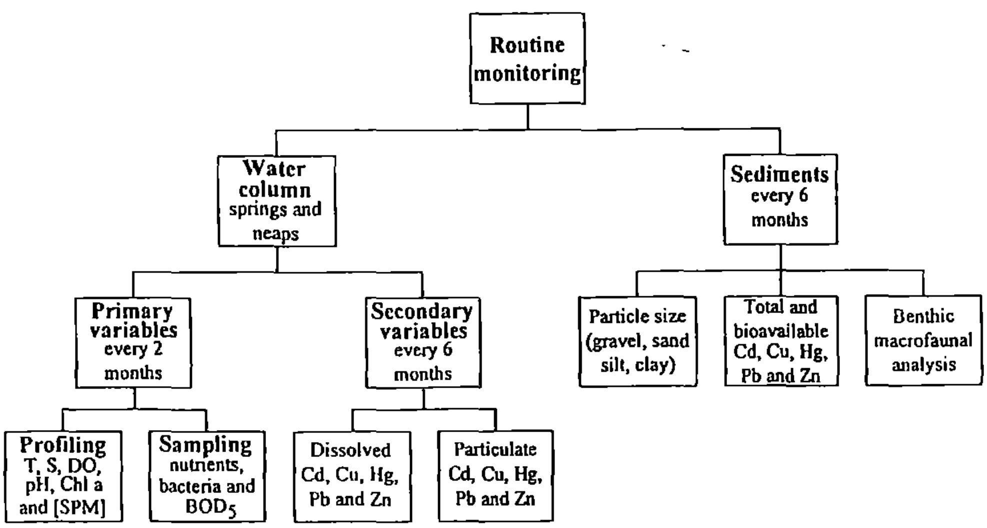


Fig. 2. Schematic of the routine monitoring programme.

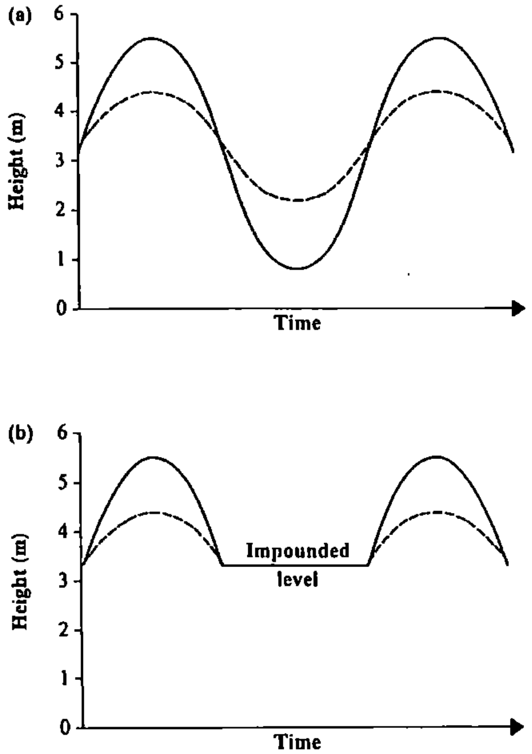


Fig. 3. Schematic tidal regimes within the Harbour: (a) pre-impoundment tidal regime; (b) post-impoundment tidal regime. Solid line indicates spring tide; dashed line indicates neap tide. Impounded level is maximum mean level retained over 14-d period.

RESULTS AND DISCUSSION

The study has generated a large database of water and sediment quality information. This paper will refer to specific sections of the database, which illustrate the value of monitoring to the management of impoundment schemes.

Flushing characteristics

Figure 3 shows the Harbour tidal regime's pre- and post-impoundment. The theoretical flushing times estimated from the water volumes exchanged were 25 h at mean springs and 62 h at mean neaps. Following impoundment, the water level is retained at 3.3 m reducing the water exchange per tidal cycle from 75% to 30% at mean springs and from 25% to 8% at mean neaps. Periods of free water exchange are only possible when the tidal height exceeds 3.3 m (Fig. 3). Consequently, the flushing time has increased to 136 h at mean springs and >300 h at mean neaps, allowing additional time for biogeochemical changes to occur in the Harbour waters and sediments, potentially involving pollutants. In order to test the water exchange, experiments were conducted with the conservative tracer *Bacillus globigii* before and after impoundment (Fig. 4). These showed that for 95% water renewal, the times were 45 h and 72 h, respectively, thus confirming the reduction in flushing due to impoundment. The management plan must consider the need for additional periods of free flow to enhance flushing despite the operational constraints of the Harbour. It must also evaluate the need for a gravity recirculation system that would draw water from zones of sluggish water movement within the Harbour to the outside via a pipeline.

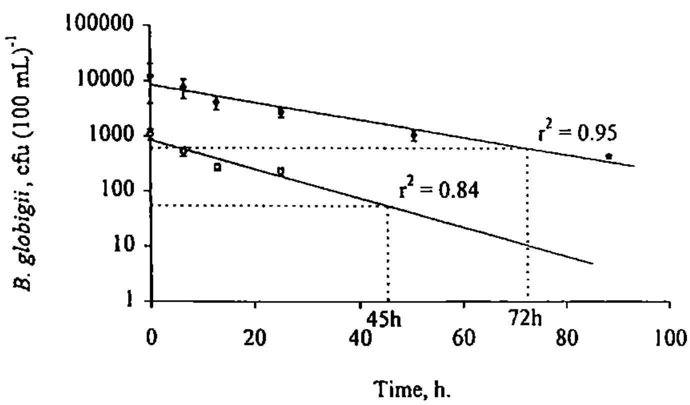


Fig. 4. Bacillus globigii concentrations (cfu (100 mL)$^{-1}$) versus time for the tracer experiments, May 1990 (open diamonds) and July 1994 (filled diamonds). Error bars indicate one standard deviation, r^2 quoted for each regression. Horizontal dashed lines represent 5% residual from initial mean concentration for each experiment.

Seasonal water column conditions

The tidally-averaged salinities within the Harbour (Fig. 5) show that the seasonal cycle is dominated by spikes of brackish water in the winter months, whilst the River Plym is in seasonal spate. The potential for the transport of far-field pollutants is enhanced under these conditions, and this is borne out by tidally-averaged dissolved orthophosphate concentrations from external sewage discharges and riverine sources, where a strong negative correlation exists between orthophosphate and salinity (r = -0.72 at surface; r = -0.68 at bottom). Orthophosphate concentrations contributed to pronounced phytoplankton blooms in the Harbour in spring, with dissolved oxygen saturations exceeding 170% and chlorophyll a concentrations exceeding 30 $\mu g\ L^{-1}$. These conditions were similar to those in the Eastern Scheldt between 1980 and 1984 (Wetsteyn et al. 1990). However, the phytoplankton blooms were short-lived in the Sutton Harbour due to nutrient depletion and sags in dissolved oxygen (ca. 60%) developed within 7-10 d, and may have contributed to benthic anoxia.

Although there is reduced water exchange following impoundment, it is still important to take into account the flux of pollutants from outside the Harbour. This suggests the management strategy should take account of careful timing of lock gate operation in sympathy with water quality outside the Harbour, especially in the spring bloom period when the accumulation of nutrients may lead to enhanced eutrophic conditions. This was illustrated in May 1994 when, directly after impoundment, an enhanced bloom was detected inside the harbour, with chlorophyll *a* concentrations of the order 10 $\mu g\ L^{-1}$ greater than outside the harbour for the first time during the study.

Health-related properties

In addition to the influx of far-field pollutants to the Harbour system, there is the possibility of inputs from sources within the Harbour. These are best illustrated by the distribution of total coliform bacteria counts on two separate occasions. Firstly, at HW springs (Fig. 6a), surface counts were below 5000 cfu $(100\ mL)^{-1}$, at all but one station both inside and outside the Harbour. The low count near the Fishers Nose outfall (see Fig. 1) was attributed to the crude discharge not taking place until LW. One week later at LW neaps (Fig. 6b), a markedly different distribution was seen. The Fishers Nose count was elevated to between 1 and 5 times the EU bathing water quality imperative limit of 10 000 cfu $(100\ mL)^{-1}$ (CEC 1976b), due to the LW discharge. These results must be interpreted with care, due to the inherent variability of the determination (Fleisher 1990). However, inside the Harbour, the mean counts were higher still, with two stations exceeding the EU limit by a factor between

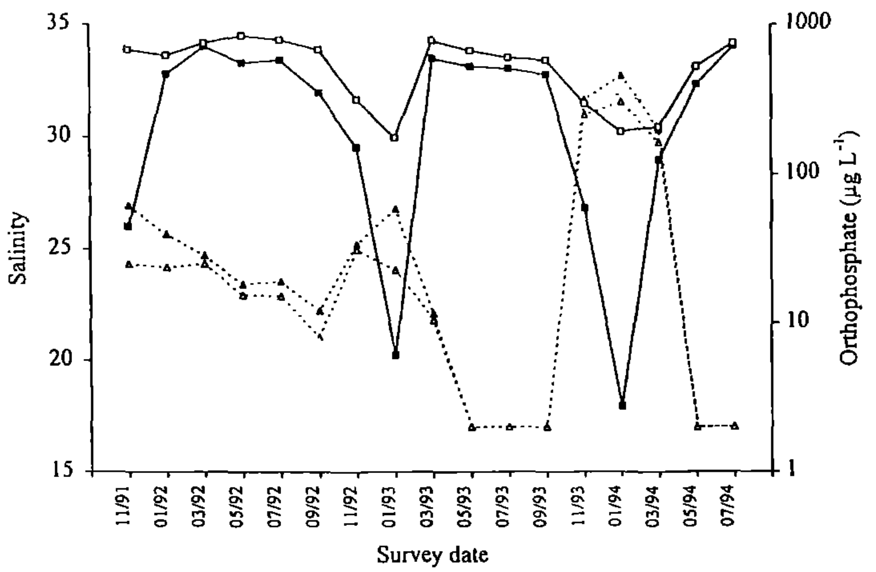
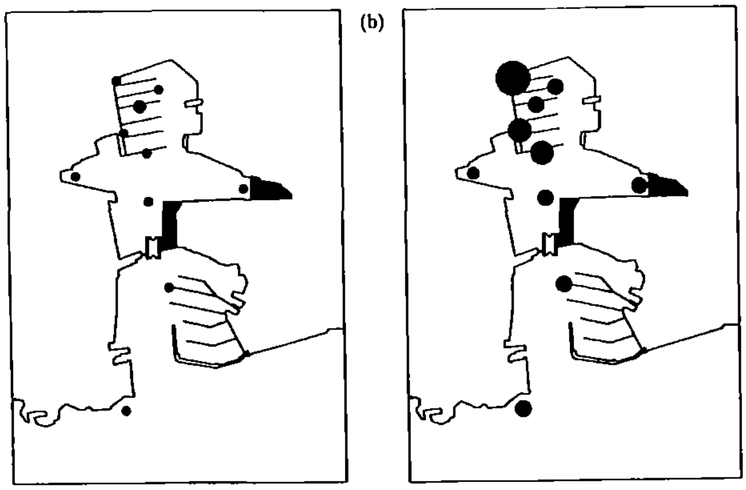


Fig. 5. Tidally-averaged salinity and dissolved orthophosphate concentration at neap tide inside Harbour from November 1991 to July 1994. Squares represent salinity, triangles represent orthophosphate; filled symbols are near-surface; open symbols are near-bottom.

(a) (b)



● $>1\times10^5$ cfu (100 mL)$^{-1}$; ● 5×10^4-1×10^5 cfu (100 mL)$^{-1}$; ● 1×10^4-5×10^4 cfu (100 mL)$^{-1}$;
● 5×10^3-1×10^4 cfu (100 mL)$^{-1}$; • $<5\times10^3$ cfu (100 mL)$^{-1}$.

Fig. 6. Distribution of total coliform bacteria: (a) at high water spring tide, 7 July 1993; and (b) at low water neap tide, 15 July 1993.

5 to 10, and one station exceeding by more than a factor of 10. An abundance of capitellid worms at these sites, indicative of sewage enrichment (Pearson and Rosenberg 1978), would suggest that there are internal crude sewage inputs in the Harbour, from land-based sources and/or from vessels in the Harbour. Mortality rates in these bacteria in the marine environment (Servais et al. 1985) are such that survival of these large numbers from the previous LW was extremely unlikely. These inputs must obviously be curtailed upon impoundment, since enteric viruses are known to outlive faecal indicator bacteria by many times (Borrego et al. 1987), and hence there may be an increased risk of illness (Balarajan et al. 1991) from the impounded Harbour waters, although little data exist on the minimum infective dose of each pathogenic organism (UNEP/WHO 1991). This is another important aspect of management, because, although the Harbour is not a designated bathing water, it is a recognised use area (Fiddes and Lack 1989) and should be included in integrated strategies for water quality.

Benthic macrofauna and sediment quality

The benthic macrofaunal community in the sublittoral zone of the Harbour, consisting mainly of polychaete worms, has undergone significant changes during construction of the scheme. These changes were primarily due to mechanical processes such as excavation and dredging causing smothering by resuspended sediment and are the subject of ongoing work. Following impoundment of the Eastern Scheldt, reduced currents led to increased sedimentation and increased transparency of the water column (Leewis and Waardenburg 1990), with consequences for the biota such as altered substrata and increased algal growth. Similar changes may be observed in the Harbour biota as the present study progresses.

Variations in heavy metal content were observed during the construction phase (Fig. 7), that were also correlated with changes in benthic community structure. Long-term increases in copper concentration from antifouling paint as a result of the decline in TBT have been reported (Claisse and Alzieu 1993). These increases may have contributed to the upward trend in the copper time series until May 1992, given the large number of small vessels using the Harbour. However, copper concentrations were reduced from November 1992. Similarly, zinc concentrations that were at a maximum in November 1991, decreased steadily during the construction period. The opposite trend is evident in the lead time series, with a minimum in May 1992 steadily increasing during the construction period. Cadmium and mercury are list I metals under the EU discharge of dangerous substances directive (CEC 1976a) and as such their discharge is now controlled. However, their concentrations in the Harbour sediment reservoir remain high relative to other locations in SW England, consistent with previous measurements in the nearby Plym Estuary (Bryan and Hummerstone 1973; Millward and Herbert 1981). Thus, there is a consid-

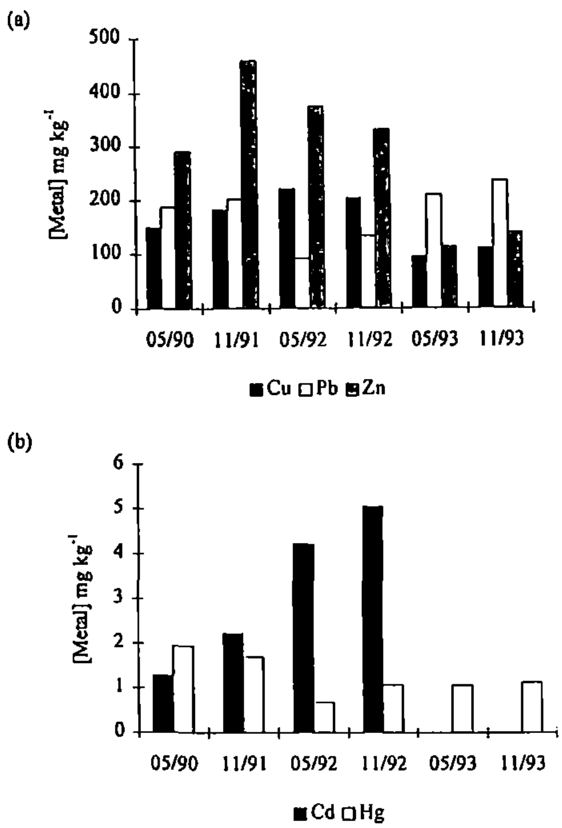


Fig. 7. Total heavy metal concentrations (HNO_3 digest) in the <63 μm fraction of Harbour sediments: (a) copper, lead and zinc; and (b) cadmium and mercury.

erable reservoir of metals in the sediments, and remobilisation of the metals takes place under the reducing conditions brought on by benthic anoxia. Any requirement to dredge these contaminated sediments must carefully consider all options of disposal (Fraser 1993).

CONCLUSION

This study has shown the value of a long-term water quality monitoring programme to the development of a management plan for impounded harbours. Thus, Sutton Harbour has a variety of water and sediment quality problems that may be exacerbated by reduced tidal flushing following the recent impoundment. The commercial and aesthetic benefits of impoundment must not take precedence over the informed ecological management of the Harbour. Monitoring will continue at least until November 1994 and may be extended if necessary in order to confirm any major alterations to the ecosystem. The completed management strategy will be reported elsewhere, and will be equally applicable to future impoundment schemes that become necessary as sea level rises.

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Appendix Two

Original Data

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