Characterisation of European Marine Sites

Poole Harbour

Special Protection Area

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Aerial view of Poole Harbour,
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Site Characterisation of the South West European Marine Sites

Poole Harbour SPA

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A study carried out on behalf of the Environment Agency and English Nature

by the Plymouth Marine Science Partnership

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Plate 1: Some of the operations/activities which may cause disturbance or deterioration to key interest features of the SPA

1: (left) Discharges to Holes Bay

2: (above) Wytch Farm oil field is the largest onshore oilfield in Europe with eleven well sites producing oil and gas

3: (left) Large and small shipping in Poole Harbour

4: (below) Agricultural practices

Photographs:
1: Getmapping plc **2:** KCA Deutag
3&4: Ian Britton (freefoto.com)

Plate 2: Some of the Interest Features and habitats of Poole Harbour SPA

1: (left) Avocet *Recurvirostra avosetta*

2: (below right) One of the Common Terns *Sterna hirundo* nesting on the man-made tern islands in the lagoon at Brownsea Island (**3** below right)

4: (left) Cordgrass *Spartina anglica*
5: (right) Saltmarsh, Holes Bay

Photographs:
1: Eric Isley
2 & 3: Helen Baines
4: CEH Dorset
5: English Nature (www.oursouthwest.com

1. EXECUTIVE SUMMARY

The Environment Agency and English Nature are currently undertaking investigative work in order to review permissions required under regulation 50 of the Conservation (Natural Habitats &c.) Regulations, 1994. Phase 1 of this exercise is the characterisation of designated European marine sites. In the South-West these sites include the marine areas of the Poole Harbour Special Protection Area (SPA).

This project, undertaken by the Plymouth Marine Science Partnership (PMSP), (comprising Marine Biological Association (MBA), University of Plymouth (UoP) and Plymouth Marine Laboratory (PML)), has two main objectives. Firstly, to characterise the site in terms of water quality over recent years (up to 2002), and to identify areas where conditions might result in effects on habitats and species for which the site was designated. Secondly, to consider permissions, activities and sources, either alone or in combination which have, or are likely to have, a significant effect on the site.

Site characterisation has been accomplished by review of published literature and unpublished reports, together with interrogation of raw data sets, notably that of the EA (this does not include recent compliance data and other forms of self-monitoring for Integrated Pollution Control sites, which was not available). Some of the key findings areas follows:

It is evident from accounts of the flora and fauna in Poole Harbour, and from various diversity indices, that biodiversity is relatively low in comparison with other UK estuaries and coastal sites. This may be due, partly, to the restricted range of habitats in the Harbour and, partly, to ongoing problems of hypernutrification and contamination. Many intertidal areas are characterised by a poor fauna with dominance of certain species (notably cirratulid or capitellid polychaetes and oligochaetes) which is indicative of contaminated sediments.

Non-native species feature high on lists of most common epibenthic species, notably slipper limpet *Crepidula fornicata*, tunicate *Styela clava* and jap weed *Sargassum muticum*. Saltmarsh areas are dominated by the cord-grass *Spartina anglica*, a hybrid resulting from a cross between an introduced and a native species. However, *Spartina* marsh is currently declining within the Harbour and consequently saltmarshes are undergoing considerable change.

Metals have historically been a concern, particularly in Holes Bay. Trade discharges (now ceased) and STW have led to concentrations in sediments which exceed sediment quality guidelines widely, but most consistently in the upper, eastern part of Holes Bay. The metals most likely to be of significance in terms of bioaccumulation and toxicity are Cd, Hg, Ag, Cu, Zn and Se, and juvenile bivalves are amongst the most sensitive groups. Sub-lethal indicators, notably metallothionein induction in Poole molluscs, appear to provide sensitive and selective measures of response to metals and may be useful in future assessment of biological effects.

Current chemical compliance monitoring of tidal waters is restricted to the outer Harbour, where water quality generally conforms to statutory limits. Nevertheless, Zn levels in water (and seaweed) from the Lake Pier area appear to be anomalously high and may reflect additional sources from antifouling and sacrificial anodes.

In the late 1980s TBT was present in high concentrations in parts of Poole Harbour, particularly in marinas and secondary embayments along the northern shoreline. As a result, molluscs were affected at some sites. Since legislation in 1987 there has been a significant reduction in TBT levels in water, but a more gradual decline in sediment concentrations. The prognosis for the recovery of mollusc populations in parts of Poole Harbour might be expected to be slow in view of the persistence of TBT in sediments. It is not known whether TBT concentrations have returned to no-effects levels, though this seems unlikely at some locations. The effects of other endocrine disruptors have not been studied at the site.

A number of organic compounds including PCBs and PAHs may be present (in Holes Bay at least) at concentrations which are of biological relevance, particularly in sediments. Further work is needed to assess the extent of their biological significance.

Much of the marine site, notably the North Harbour and secondary embayments, is subject to eutrophication. Although the majority of nutrient inputs in the system may be due to diffuse sources such as agricultural run-off, sewage discharges constitute additional loading and result in chronic contamination of the affected areas. Poole STW is implicated as the major offender. Diffuse inputs from tributary rivers and streams are significant and may also be enhanced by STW discharges. Nutrient-associated water quality problems have been recorded for several decades and include macroalgal, and to a lesser extent, microalgal blooms and periodic oxygen sags. These manifestations have been the subject of a series of public complaints and are implicated in shellfish mortalities and occurrence of ASP and DSP toxins in shellfish (leading to shellfishery closures and near collapse of the local industry). Effects to other biota are largely unresearched.

The problem of hypernutrification has lead to the designation of Poole Harbour as a Sensitive Area (Eutrophic) and Polluted Waters (Eutrophic) and its catchment area a Nutrient Vulnerable Zone (NVZ), which, it is hoped, will herald long-recommended reductions in nutrient loadings from both point, and diffuse sources.

Various bioassays confirm that water (and sediment) quality of the SPA is relatively good in the outer Harbour with some deterioration in ‘biological condition’ inside Holes Bay. No specific compound is implicated; in reality, it is probably due to a combination of the contaminants listed above, and possibly others.

Thus, despite its high conservation value, the threats to the biota of Poole Harbour from contaminants are not unimportant. Concerns focus on Holes Bay and are probably dominated by inputs from Poole STW, though diffuse sources, notably sediments, contribute to the impact on biota. Dredging activities and other disturbance will inevitably remobilise and redistribute some of this loading. The persistence and behaviour of sediment-bound contaminants, and their potential combined effects, therefore gives rise to the greatest uncertainty, and will require better characterisation.

These principal findings are discussed in detail in the following report, together with implications for key habitats and species. A major challenge for the future is to establish a more reliable integrated means of assessing changes in the biology and chemistry of the marine site. Recommendations are made which may improve understanding of the system and assist Regulatory Authorities in their statutory responsibilities to ensure the favourable condition of the site and its features.

CONTENTS

2. INTRODUCTION

This review considers the characteristics of the marine areas of the POOLE HARBOUR SPA and how the status of the site is influenced by existing permissions and activities, either alone or in combination. Also considered is possible impact from other factors such as unconsented activities, diffuse sources and natural processes. This includes activities and consents outside the site itself. The purpose is thus to collate and interpret information relevant to the assessment of water quality impacts and risks to the marine component of the SPA, to ensure that EA and EN are fully informed when making decisions in relation to the scope of appropriate assessment. The opinions expressed are made on the basis of available information (up to 2002). We have emphasised areas where information is lacking, or where we see an opportunity to improve implementation and monitoring to comply with the requirements of the Habitats Directive and to provide a better means of establishing the status of the site.

To achieve this goal, specific objectives were:

- To prepare comprehensive reference lists of previous investigations and existing datasets, including published research and unpublished reports, relevant to an assessment of the effects of water quality on the marine sites and interest features identified.
- To review the existing information, pinpoint key studies, collate and summarize their findings.
- To identify site-specific models predicting pollutant concentrations and their links to impact.
- To prepare a summary of existing datasets (spatial and temporal) on water and sediment quality (e.g. determinands and summary statistics where available).
- To integrate and evaluate biological information, with specific reference to water/sediment quality.
- To conclude if there is any evidence that existing water (or sediment) quality is causing impact and highlight limitations of the available data.
- To identify and recommend further research which will address the limitations of current information and establish cause/effect relationships.

English Nature has provided advice on the Poole Harbour Marine site, given under Regulation 33(2) of the Conservation Regulations 1994 (English Nature, 2000). A summary of the interest (or qualifying) features, and conservation objectives, for the site is given in Annex 1. The table below is a summary of the operations which, in the opinion of English Nature, may cause disturbance or deterioration to these interest features. In terms of the current project's emphasis on consents, we will focus on the vulnerability to toxic contamination and non-toxic contamination unless any of the other threats are seen as highly relevant.

Table 1. Summary of the operations, which, in ENs opinion may cause disturbance or deterioration to key interest features of the Poole Harbour SPA. Toxic and non-toxic contamination are the principal threats considered in the current project. (Table adapted from English Nature, 2000)*

Standard list of operations which may cause deterioration or disturbance	INTEREST FEATURES*		
	Internationally important:		
	Annex 1 birds	populations of regularly occurring migratory species	assemblage of waterfowl >20,000
Physical loss			
Removal (e.g. harvesting, coastal development)	✓	✓	✓
Smothering (e.g. artificial structures, disposal of dredge spoil,)	✓	✓	✓
Physical damage			
Siltation (e.g. run-off, channel dredging, outfalls)	✓	✓	✓
Abrasion (e.g. boating, anchoring, trampling)	✓	✓	✓
Selective extraction (e.g. aggregate dredging,)			
Non-physical disturbance			
Noise (e.g.boat activity)	✓	✓	✓
Visual presence(e.g. recreational activity)	✓	✓	✓
Toxic contamination			
Introduction of synthetic compounds (e.g. TBT, PCB's,)	✓	✓	✓
Introduction of non-synthetic compounds (e.g. heavy metals, hydrocarbons)	✓	✓	✓
Introduction of radionuclides			
Non-toxic contamination			
Changes in nutrient loading (e.g. agricultural run-off, outfalls)	✓	✓	✓
Changes in organic loading (e.g. mariculture, outfalls)	✓	✓	✓
Changes in thermal regime (e.g. power station)			
Changes in turbidity (e.g. run-off, dredging)	✓	✓	✓
Changes in salinity (e.g. water abstraction, outfalls)	✓	✓	✓
Biological disturbance			
Introduction of microbial pathogens			
Introduction of non-native species and translocation			
Selective extraction of species (e.g. bait digging, wildfowl, commercial and recreational fishing)	✓	✓	✓

***Note: Key habitats (subfeatures) of the site which support these internationally important birds include shallow inshore waters (including lagoons), intertidal sediment communities, saltmarsh communities and reedbeds. See Annex 1 for more detailed descriptions.**

The key questions, which we have tried to incorporate into our considerations of site characteristics are in line with the Agency's Management System i.e.

- Is there a potential hazard mechanism by which the consent/activity could affect the interest features of the site (directly or indirectly)?
- Is there a probability that the consent/activity could affect the interest features of the site (directly or indirectly)?
- Is the scale and magnitude of any effect likely to be significant[1]?

Clearly if the answer to all three questions is positive a more detailed assessment is likely to be required.

We have also kept in mind similar criteria which EA/EN may need to apply during the review process as outlined in their *Guidance for the Review of Environment Agency Permissions: Determining Relevant Permissions and 'significant effect'* (March 1999):

A. The designated feature is in favourable condition and there is no evidence to suggest existing consents are currently having a significant effect.

B. The designated feature is in favourable condition but there is concern that a water quality problem caused by a consented discharge may be threatening that condition and/or causing a decline in it.

C. The designated feature is in unfavourable condition, but this can be attributed to a factor unrelated to water quality, e.g. vegetation management, and there is no evidence to suggest relevant consents are currently having a 'significant effect'.

D. The designated feature is in unfavourable condition and poor water quality may be or is likely to be responsible.

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[1] Examples of 'significant' effects criteria:
- Causing change to coherence of the site
- Causing reduction in area of the habitat
- Causing change to the physical quality and hydrology
- Altering community structure (species composition)
- Causing ongoing disturbance to qualifying species or habitats
- Causing damage to size, characteristics or reproductive ability of qualifying species (or species on which they depend)
- Altering exposure to other impacts
- Causing a reduction in resilience against other anthropogenic or natural changes
- Changing stability of the site/feature
- Affecting a conservation objective

3. REFERENCE LISTS AND SOURCES OF INFORMATION

- A full list of publications in the open literature has been assembled using the Aquatic Sciences and Fisheries Abstracts (ASFA) and Web of Science information retrieval systems. The NMBL in-house data base ISIS has provided additional listings (see accompanying electronic database).

- Unpublished reports and data-bases: Environment Agency, Joint Nature Conservancy Council (JNCC) Coastal Directories Reports, Centre for Environment, Fisheries and Aquaculture Science (CEFAS); (see accompanying electronic database).

- Information, monitoring data and summary statistics provided by the Environment Agency up to 2002, extracted from WIMS (this does not include recent compliance data and other forms of self-monitoring for Integrated Pollution Control sites, which was not available).

- The Plymouth Marine Science Partnership (PMSP) laboratories (MBA, PML, and UoP) have already undertaken a number of studies in the Poole Harbour system on modelling of nutrient loads, bioaccumulation of metals, TBT and ecology of benthic organisms. Comparative data for other UK estuaries, including south-west marine sites (e.g. Severn, Tamar, Exe, Fal) have been used to draw comparisons.

Section 4 of this report describes the main physical, chemical and biological features which shape the character of the site and summarises some of the perceived threats to its favourable status. A review of published information on contaminants is the focus of **Section 5**, which contains sub-sections on toxic contamination (metals, TBT, petrochemicals, pesticides, PCBs, radionuclides) and non-toxic contamination (nutrients, turbidity, dissolved oxygen). Studies which describe the biology and ecology of benthic communities within the site, many of which support the bird populations for which the site is designated, are discussed in **Section 6**.

Section 7 presents summary statistics of previously unpublished water quality data, in relation to Environmental Quality Standards and guidelines (listed in Annexes 2-5). This draws on available information provided by the Environment Agency (extracted from WIMS). The section again includes considerations of toxic and non-toxic contamination. A synthesis of available information on sediment quality, based mainly on MBA metals data and mapping routines, is given in **Section 8.**

A brief description of modelling exercises of direct relevance to the environmental quality status of the site is provided in **section 9**. These exclusively concern nutrient levels.

Concluding remarks (**section 10**) include a summary of evidence for impact in the Poole Harbour marine site, together with recommendations for future monitoring and research requirements.

4. THE SITE: FEATURES AND THREATS

In addition to its SPA status under the European Commission Directive on the Conservation of Wild Birds (79/409/EEC) parts of Poole Harbour are designated as SSSI/ASSI in the British context (Arne, Holton and Sandford Heaths, Studland and Godlingston Heaths, The Moors, Wareham Meadows).

Poole Harbour was also listed as a Ramsar site under the International Convention on Wetlands of International Importance. The Ramsar margins include fen meadows, wet pasture with ditches and transitions to peatland mires which are of exceptional conservation importance.

The Harbour is one of 37 natural saline lagoons in England and Wales (most of which are situated along the south and east coasts), with a further five behind artificial barriers. However, its importance lies in its size: Poole occupies all but a quarter of the total area of 3,300ha represented by this habitat.

The boundaries of the Poole Harbour marine site, which essentially incorporates the entire marine component of the SPA, are shown in figure 1. Detailed maps of communities and features within the site can be found in greater detail elsewhere (English Nature, 2000). The site does not include a substantial subtidal component: some 80% of its area is composed of inter-tidal fine-grained muds, sands and salt marshes.

The Harbour, which extends some 10km from Wareham to Bournemouth, occupies a shallow depression towards the south-western extremity of the Hampshire Basin which has flooded over the last 5,000 years as a result of rising sea levels. The area was formerly the upper course of the Solent River a major Pleistocene drainage system. Extensive erosion of the underlying tertiary formations (Eocene sands) has resulted in the present shape of the Harbour. The geology of the catchment is dominated by these deltaic sands and seams of clay of the Bagshot Beds.

The Harbour is a bar-built estuary of nearly 4,000ha.with an unusual double-high water, micro-tidal regime. The tidal range (mean springs) is only of the order of 1.7m. Most of the Harbour is shallow with depths varying between 0.5m below and 2.5m above chart datum. This increases to 10m below chart datum near the Harbour mouth at Sandbanks. The north channel and middle ship channel are regularly dredged.

The narrow opening to Poole Bay means that a significant body of water may be retained throughout the tidal cycle. The proportion of the water body exiting the harbour at neap tides is 22% at neap tides, increasing to 45% at springs. The Harbour therefore exhibits many lagoon-like characteristics with relatively low tidal energy. Tidal currents peak at around 1m sec^{-1} near the Harbour mouth. Some large waves may enter the Harbour entrance when the wind is SE, though these dissipate rapidly with distance. Holes Bay, the largest secondary embayment entering Poole Harbour also has a narrow entrance, compounding restrictions to flow still-further: the tidal prism here is about 40% greater than in the outer Harbour. Consequently, a sheltered nature and reduced currents, particularly in the upper bay, encourage siltation.

There are extensive intertidal mud-flats throughout the marine site and to the south and west extensive areas of saltmarsh and reedbed. Intertidal flats are comprised of two major habitat types; firstly fine silts and clays (usually in sheltered embayments and creeks) and, secondly, sediments with a wider range of grain size (subjected to greater wave action and tidal currents). As there is virtually no natural hard substrate apart from rocky outcrops on the islands and mainland promontories, diversity is somewhat limited and dominated by Annelida and occasionally other detritivores such as *Abra tenuis* or the crustacea *Corophium volutator*. However, the northern shoreline to the east of Lytchett Bay is urbanised, with piers and sea walls offering additional habitats. Benthic communities are discussed in greater detail in section 6.

As a whole, the Harbour supports important numbers of waders in winter and is also an important breeding site for terns and gulls, hence its designation as an SPA (see annex 1). Nationally important aquatic species include the peacock worm *Sabella pavonina* and the sponge *Suberites massa,* whilst regionally important species include two bryozoans *Anguinella palmata* and *Farella repens*.

The Harbour is separated from Poole Bay by the Studland Dunes (part of the Dorset Heaths [Purbeck and Wareham] and Studland Dunes SAC) and the SPA includes Littlesea, a large oligotrophic dune-slack lake of importance for wintering wildfowl. Parts of the Harbour, especially along the western and southern shores, adjoin the Dorset Heathlands SPA. Where the two areas meet, there are unusual transitions from saltmarsh and reedbed to valley mire and heath habitats.

Rivers

There are two major freshwater sources to Poole Harbour, the Rivers Frome (7.35[*]) and Piddle (3.027*), together with much smaller inputs from the River Sherford (0.54*), Corfe River (0.47*) and a number of smaller tributaries such as Holton Heath stream (figure 1). These river valleys support grazing marsh that contribute to the importance of the SPA for wintering birds.

The Frome rises in West Dorset and flows east towards Dorchester and on to Poole Harbour. It is joined by a number of tributaries including the Cerne, Sydling Water and Hooke Stream. The Piddle rises near Buckland Newton and then flows south-east towards Poole Harbour. Its' major tributaries include the Devil's Brook and the Bere Stream. Migratory salmon and trout spawn in both the Frome and the Piddle, and both species have been recorded in tidal reaches of the Sherford. Bass are also found in the tidal reaches of all three rivers.

The catchment is predominantly rural but experiences some urban influences at Dorchester and downstream at Wareham. The town of Poole and suburbs of Bournemouth which border the north east of the Harbour provide potentially more direct sources of urban and industrial contamination.

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[*] These are relative ratios scaled using micro-low flow means (Murdoch and Randall, 2001).Total mean river flow ~9 m^3s^{-1}.

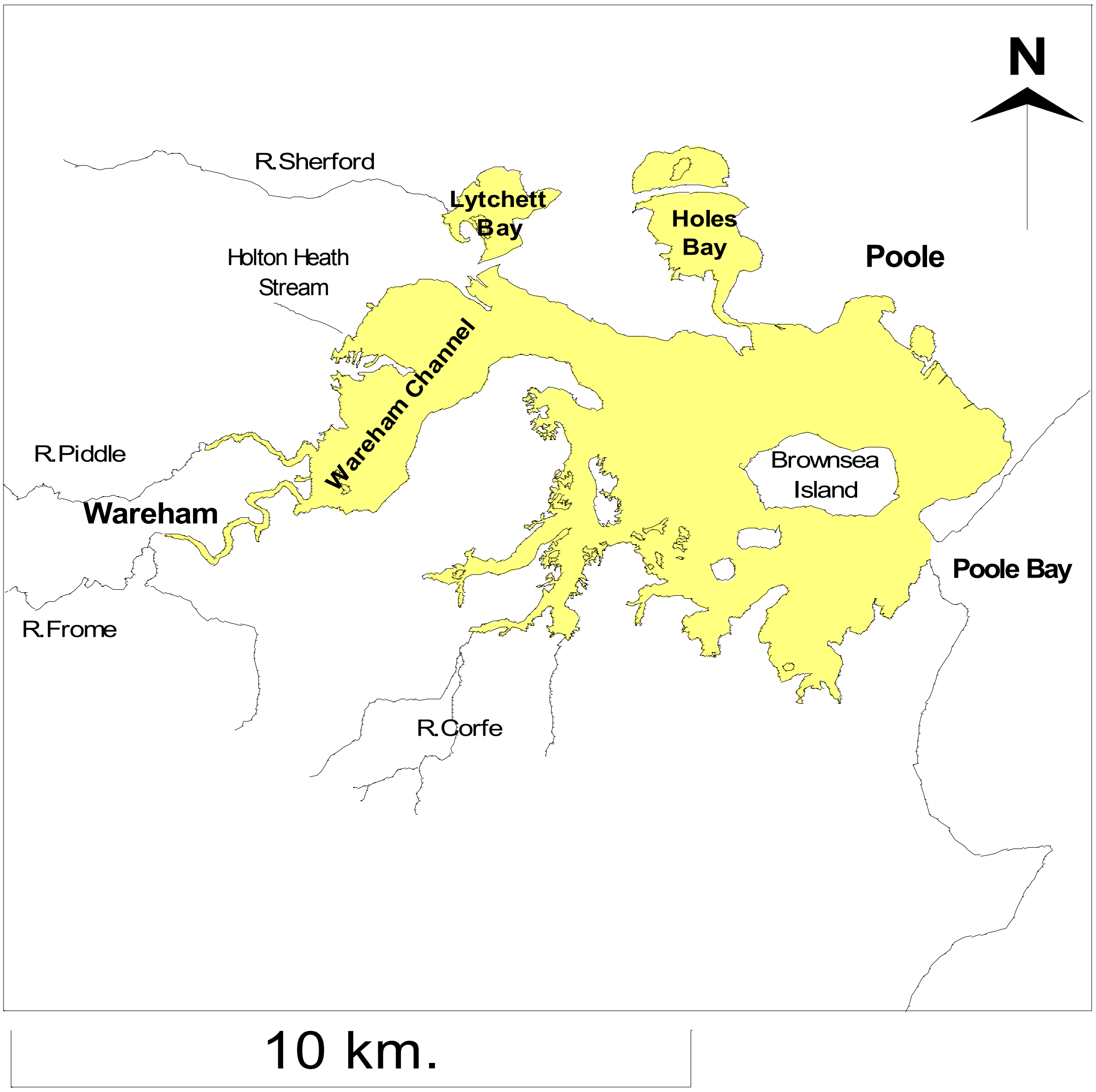

Figure 1: Poole Harbour SPA showing boundaries of marine site

Threats

The pressures of urbanisation in Poole have resulted in some loss of intertidal estuarine habitat in recent years. In particular Holes Bay has undergone a degree of infilling to accommodate industrial developments, trunk road construction in the east and marina expansion.

Physical loss through development would reduce both availability of roosting habitats and food-supply. Physical damage may also arise, intermittently, through siltation and dredging. Associated high turbidity could affect invertebrates, fish and algal productivity, and theoretically could reduce visibility of prey items to the common tern. Noise and disturbance are inevitable in such a populated area and has obvious direct impact on roosting, feeding and nesting. Objectives of The Poole Harbour Aquatic Management Plan and Poole Harbour Management Policy include limitation of such disturbances e.g. through the zonation of potentially disturbing activities.

The enclosed nature of Poole Harbour and its secondary embayments make it vulnerable to the effects of chemical contamination which would disperse extremely slowly due to restricted tidal exchange. Contingency plans have been drawn up for oil spills ("Poolspill"[1]), which could be potentially serious for birds in terms of smothering, direct toxicity and bioaccumulation through the food chain. The same threat of bioaccumulation exists with respect to a number of other chemicals which could be introduced to the system through accidental spillage, rivers, road run-off and discharges. Some of these threats are discussed in more detail in sections 5, 7 and 8, in relation to published and EA data, respectively. Nutrients and organic loadings ('non-toxic contaminants') may have a two-edged effect: in small amounts they may enhance productivity and appear beneficial, but in excessive amounts lead to the formation of nuisance blooms and algal mats whose eventual decay can deplete oxygen levels, increase hydrogen sulphide in sediments, and, in extreme cases, produce toxins which eliminate prey items. Areas most susceptible to the threat of eutrophication are the secondary embayments of Holes Bay, Lytchett Bay and Blue Lagoon (sections 5 and 7). There have been red tides reported in the Poole Harbour area in recent years (South West Environmental Observatory website, 2002).

Boating and shipping

Poole Harbour is one of the world's largest natural harbours and its 10,000 acres attract a large number of leisure vessels which are provided for by a number of marinas along the northern shoreline. Around 5000 yachts visit Poole each year, in addition to the resident leisure fleet. Ten boatyards around the Harbour cater for repair, maintenance, storage and launching facilities.

There is an active fishing fleet - particularly for shellfish, crabs and lobsters. Around 100 boats land some 2,800 tonnes of fish and shellfish each year.

Some 60 acres are devoted to commercial Port operations. Since its peak in the 19th century the activity of the commercial port has declined, though the last decade has seen an increase in cross-Channel traffic. Cargoes include continental RO-RO, conventional cargoes, off-shore/on-shore oil support and bulk cargoes such as steel, sand, gravel and fuel.

Diffuse Sources

Diffuse pollution, particularly from agricultural land runoff, is seen as an important issue in the South Wessex Area. Intensive agricultural practices can increase soil erosion. Resultant run-off from eroded land can lead to water quality problems (siltation, eutrophication, pesticide residues and River Quality Objectives compliance issues). Increased run-off may reduce infiltration to ground, compounding low flow problems. Farm animal waste and fuel oil storage facilities are a potentially significant source of pollution to rivers feeding the Poole Harbour SPA.

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[1] www.phc.co.uk (Poole Harbour Commissioners, 2002)

Point source discharges and other activities of potential significance

There are a number of trade and sewage discharges of varying sizes (see list as supplied by EA in the accompanying electronic database). Siting of some of the more important (by volume) discharge consents to the site are shown in figure 2. Consented sewage discharges (which may contain some trade wastes) are moderate to large, amounting to some 134,399 m^3d^{-1} for the SPA as a whole (not including inputs upstream of the tidal limits). Specific trade discharges are comparatively small accounting for an estimated 6167 m^3d^{-1} (Allen *et al.*, 2000) although they include discharges from Wytch Farm oilfield.

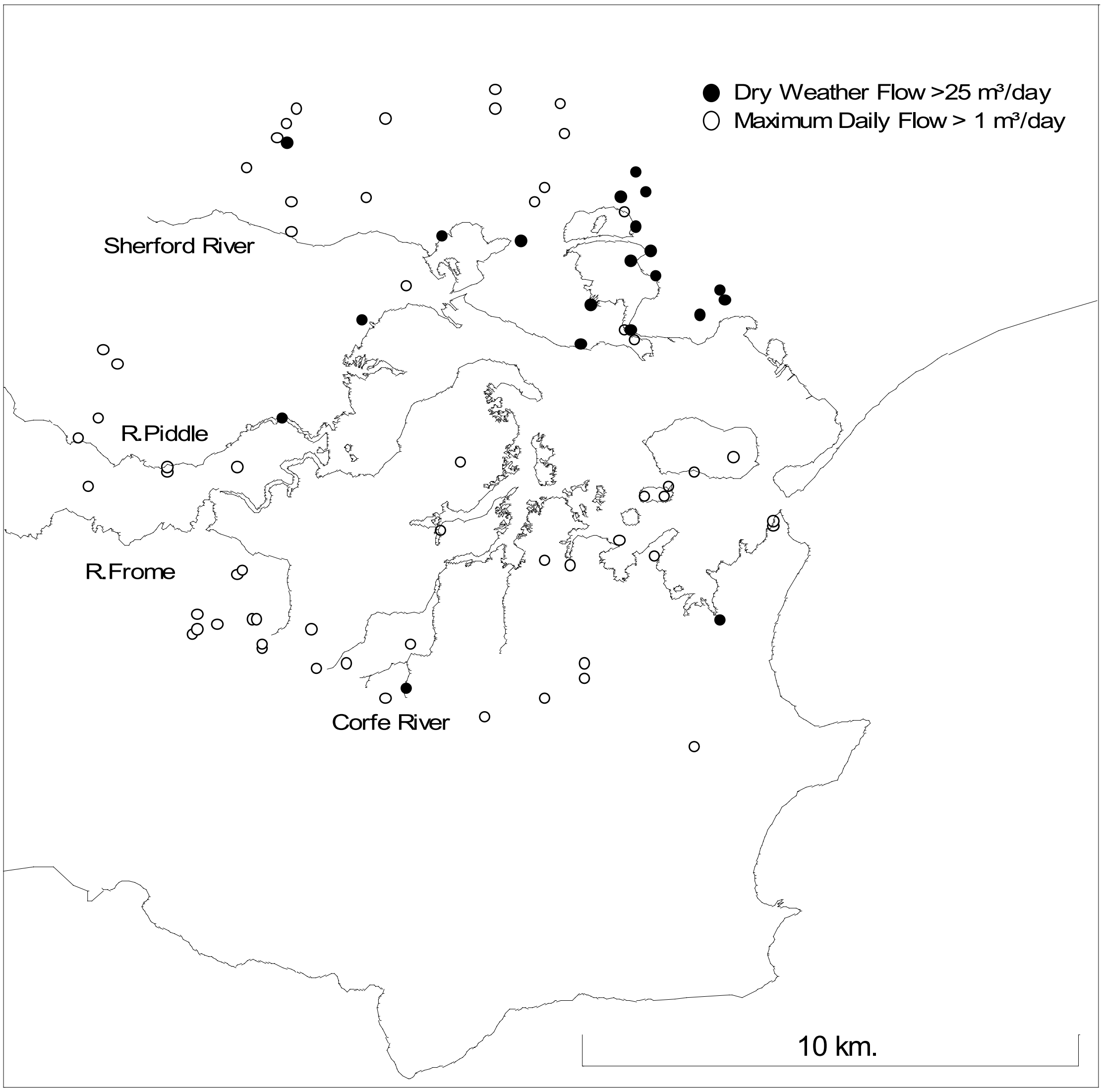

Figure 2. Locations of some of the larger discharge consents to the Poole Harbour system. Consents for the discharge of sewage are based on Dry Weather Flow (values > 25 m^3/d shown). Trade consents, and miscellaneous sources of effluents are expressed as Maximum Daily Flows (values >1 m^3/d shown). (From data supplied by the Environment Agency, South West Region).
NB No distinction has been made between continuous and intermittent discharges and details of specific discharges should be clarified with the Environment Agency

The list of potential chemical pressures includes metals, organometals (e.g.TBT), hydrocarbons, nutrients, volatile organics, mineral acids, biocides, fungicides, and pesticides. The decline in industry and introduction of pollution control in the later part of the 20th century, however, has seen a general downward trend in inputs entering the Harbour from key discharges. An inventory of principle loadings in recent years is summarised in Annex 7. These are based on available values from the Environment Agency's (South-West Region) website.

One of the most significant and sensitive discharges arises from the STW at Poole (47,000m^3d^{-1} dry weather flow). This partly arises because of the enclosed and sheltered nature of the receiving environment in the north east corner of Holes Bay. On a number of occasions in recent years the sewage effluent discharging into the Fleetsbridge Channel, a tributary of Holes Bay, has exceeded its ammonia limit by over 20 per cent. The high ammonia levels were caused by problems with the sewage treatment plant. Wessex Water had identified the need for improvements at the works in May 2001 and these were completed in November. Light industries discharge to this plant and surface drainage from industrial estates may contribute small amounts to the bay. Until recently, direct discharges into Holes Bay included the Merck chemical plant (stopped in 1998), and electroplating works. Probably because of a combination of pressures, the biodiversity of infauna in Holes Bay is generally considered to be lower than expected, though in fact abundance of individuals may be high - dominated by opportunists with relatively short life cycles.

Other major STW discharges are at Wareham (Keysworth - 2,500 m^3d^{-1} dry weather flow) and Lythchett Minster (1,200 m^3d^{-1} dry weather flow) (Murdoch and Randall, 2001).

Wytch Farm, operated by BP Amoco (now trading as BP Exploration Operating Company Ltd), is Europe's largest on-shore oil field, producing 90% of the onshore oil in the UK. The Wytch Farm field extends under the Harbour and into Poole Bay and is estimated to contain 53 million tonnes of oil of which approximately half has been used to date. The original plan was to develop the field on an artificial island offshore, however the high capital cost and lack of oil production during platform construction forced a more economic alternative.

Inevitably, dredging has to be undertaken to keep navigational channels open, amounting to an average displacement of 70,000m^3 sediment per year. Smaller quantities may be dredged for access to marinas. Future demands may include widening the main channels to improve access for larger vessels which could include capital dredging of 3.4 million m^3 (over several phases). At present most dredged material is disposed of at the licensed dumpsite at 'Old Harry' off the Dorset Coast, though it is believed that some material from the Middle Ship Canal and Hamworthy will be used in 'beach nourishment' schemes at Sandbanks.

A report by Posford Duvivier (1992) commissioned by English Nature and Poole Harbour Commissioners reviews the possible advantages and disadvantages of capital and maintenance dredging, for nature conservation. Possible advantages include creation of new habitats elsewhere; disadvantages include destruction of the dredged habitat itself and adjacent areas, increased turbidity, erosion and slumping, and changes in hydrodynamics which can effect biota. An additional concern over dredging relates to the potential re-release and bioavailability of sediment-bound contaminants. As a general rule, oxygenation of sulphide-rich sediments during dredging will result in a change of chemical equilibria: metals (and nutrients) may become oxidised and solubilised, initially, before becoming re-sequestered with oxyhydoroxides. There is potential for increased bioaccumulation during this process but precise impact is difficult to predict and may vary according to the nature of the sediment.

The report by Posford Duvivier included a case study of Holes Bay sediment (from the south east corner) - described as chemically “clean” sandy silt – which, it was suggested, would be suitable for habitat creation elsewhere. However, the present study indicates that these sediments may not have been typical: much of the silt in Holes Bay could not be accurately classed as ‘chemically clean’.

Disturbances through bait digging and fishing are also potentially damaging to feeding birds, however there is little quantitative information on these threats.

Bathing beaches and shellfish waters

All of the six designated beaches in the Poole Harbour region complied with the Bathing Water Directive (76/160/EEC) imperitive standards 1998-2002. Shore Road Sandbanks, Poole Sandbanks car park, and Shell Bay (N) complied with the more stringent guideline standards during this period. Poole Harbour Sandbanks and Poole Harbour Lake bathing waters have also achieved the guideline standards since 2000. Under the Shellfish Water Quality Directive, Poole failed to meet statutory requirements in 1997. As of September 2001, under the shellfish hygiene directive, waters throughout the Harbour (except the Wareham Channel) were designated as class B bivalve production areas for Pacific and native oysters, manila clams, mussels and cockles; Wareham Channel has a class C designation for clams. In addition, for native oysters, Poole Bay is a class B production area and South Deep, within the Harbour is a designated relaying area.

Improvements to treatment of wastes under the Urban Waste Water Treatment (UWWT), Shellfish and Bathing Water Directives are on-going. Between 7 and 9 storm overflows to Poole Harbour have received attention between 2001/2. Anticipated water company improvements in the near future include addition of UV disinfection to the secondary treatment at Poole, Wareham and Lytchett Minster Sewage Treatment Works by 2003. The storm tanks at these works are also due for improvement. A storm overflow in Upton is due for improvement by April 2003 (see annex 8).

Abstraction

The abstraction of water can have an unacceptable environmental impact on wildlife and amenity by reducing river flows. Low flows on the Piddle are caused by abstraction of groundwater by Wessex Water. In response, a statement of intent to restore acceptable flows by March 2003 has been agreed by the Environment Agency, Ofwat, English Nature and Wessex Water.

5. SUMMARY OF KEY STUDIES ON CONTAMINANTS

Overview of contaminant loadings and sources

There are relatively few published statistics available as to the relative contributions from different sources into the SPA itself. Where published information exists (mainly for nutrients) this is included in the relevant sections on individual contaminants.

The Agency provides data for OSPAR on a regional basis. The returns for the region encompass the coastline between Bournemouth and Lyme Regis and do not present loadings for Poole separately. Only a broad impression of relative inputs for selected contaminants can be gained. Figure 3 distils the information for 1999 as to the relative contributions to the seas arising from rivers and sewage. No separate figures for industrial discharges are entered. Principal sources for all the determinands considered are rivers, though there are sizeable proportions from sewage for suspended particulate matter, Zn, Cu and, notably, orthophosphate. Contributions of Hg and γHCH (lindane) from sewage appear to be negligible.

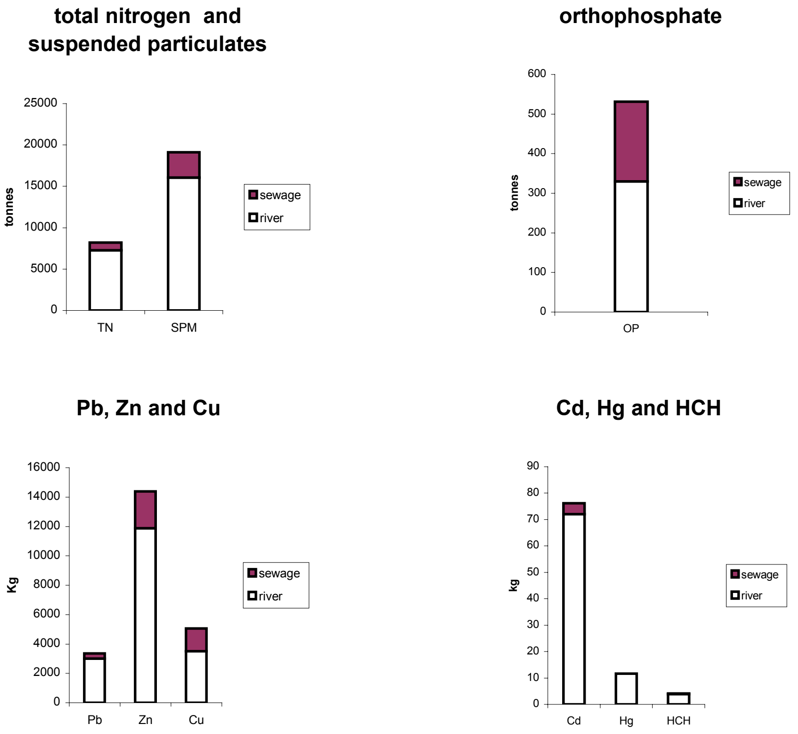

Figure 3. Relative loadings for OSPAR determinands from rivers and sewage discharging to the sea between Bournemouth and Lyme Regis, 1999. Data source EA.

NB Principal rivers are sampled just upstream of their tidal limits to assess freshwater discharges into marine waters therefore riverine sources may also contain an indirect sewage and industrial component.

In sections 5.1 and 5.2, below, we discuss published information for Poole Harbour on toxic- and non-toxic contaminants, respectively.

5.1 Toxic Contaminants

5.1.1 Metals

Water: Inputs from chemical industries based in Poole focused much interest on metal contamination in the site in earlier years. Data from the 1970s and 1980s suggested that metal concentrations were elevated, particularly in Holes Bay, and were responsible for poor performance of oyster larvae in a newly-located hatchery near Poole Bridge, at the mouth of Holes Bay (Boyden, 1975). At that time average concentrations of Cd and Zn at the hatchery intake were 5.7 and 177 µg l^{-1}, respectively - high enough to affect sensitive oyster larvae. Chemical plant, numerous light industries and the STW at Poole were all considered likely sources.

In the 1970s, dissolved metal concentrations in Holes Bay could exceed values considered typical for the English Channel by 30-40 fold for Cu, Ni and Zn, and more than 100-fold for Cd and Pb (table 2). They were also higher than in the Wareham Channel in the outer Harbour. As with most data for water, there was substantial temporal variation, even between hourly samples; highest concentrations occurred at low water, consistent with land-based sources (Boyden 1975).

Table 2. Historic data for dissolved metals (µg l^{-1}) Poole Harbour[1] and background concentrations in seawater[2,3]

site	Cu	Zn	Pb	Cd	Mn	Fe	Ni	Ref
Holes Bay (HW)	2.5	9	-	0.44	11	12	13	1
Holes Bay (LW)	18	68		7.4	24	9	8	1
Poole Hatchery	4	25	2.5	3.5	12.1	4.7	10.9	1
Wareham Channel	2	7	-	0.16	6	5.6	5	1
E.Channel (1970)	0.46	2		0.06	0.32	0.3	0.38	2
OSPAR Background (2000)	0.05 - 0.36		0.005 - 0.02	0.005 - 0.025				3

[1]Boyden , 1975 [2] Preston, 1972; [3] OSPARCOM (2000)

The data in table 2 are of course now almost 30 years old and indicate historic levels of metals in water. It is *likely* that concentrations have decreased since Boyden's study (see section 7.1.) though no direct comparisons can be made due to the absence of recent data for Holes Bay. Whereas sources in the past would probably have been comprised of a significant component from the chemical industry, data in annex 7 imply that the loadings from STW are now likely to be the only significant input.

Sediments: Geological background levels of metals in the sediments of streams and rivers entering the estuary are anticipated to be generally low in rural areas (table 3), with a small number of anomalies in catchment areas containing metal-rich sedimentary rocks (mainly in the west of the region). In urbanised tributaries, enrichment with Ag, Cu, Cd, As, Zn and Pb has been observed, downstream of industrial areas (Thornton *et al.*, 1975).

Table 3. Average metal concentrations (µg g^{-1}) in sediments (<200µm) from tributaries of Poole Harbour (from Thornton *et al.*, 1975)

tributaries draining:	As	Cd	Cu	Cr	Fe	Pb	Zn
urban areas	6	4	40	59	16000	223	184
metalliferous rocks	25	1	49	46	113000	146	149
other (rural) areas	4	<1	5	41	8000	29	38

Particulate metals can form an important part of the loading discharged to estuaries. Scavenging of dissolved metals also occurs, so that the sediments of the estuary provide an integrated record of contamination history. Because of their larger surface area and greater density of organic and oxyhydroxide binding sites, contamination loadings will be highest in fine fractions and lowest on coarser substrates. Cundy and Croudace (1995) show that, at the higher energy sandier sites close to the mouth of Poole Harbour, highest metal concentrations (Cu, Pb and Zn) are associated with fractions <8µm. At Wytch Farm (mainly clay-rich fines) these metals are more widely distributed across size fractions, though some enrichment is seen in the <8µm fraction. Distributions of Cu, Pb and Zn correlate highly with Fe and Mn oxyhydroxides indicating that adsorption onto these coatings is particularly important. However, there is also some association of metals with organic phases, notably so in coarser sediments near the Harbour mouth; this organic material may represent debris from adjacent salt marsh (Cundy and Croudace, 1995).

The importance of organic and oxyhydroxide coatings in metal cycling is determined partly by their relative abundance. This, in turn, will govern the ability of the sediment to act as a sink for contaminants and the extent to which they are remobilised. Sequential extractions suggest that a significant fraction of the metal loading in Poole sediments may be labile and thus has potential for remobilisation (Langston, 1982, 1985; Cundy and Croudace, 1995).

Because of the sheltered, low-energy conditions in Poole Harbour, sediments may remain relatively undisturbed. As a result, contamination arising from discharges tends not to be dispersed over a large area and may lead to some heterogeneity (unlike the Severn, for example). This is particularly so for the secondary embayments such as Holes Bay which contains the highest sediment loadings. Metal concentrations in the upper, eastern part of the Bay are amongst the highest in the Harbour (partly due to proximity to Poole STW and partly due to enriched organic and oxyhydroxide coatings). Here, values of Cr, Cu, Pb, Ni and Zn, for example, are between 31 and 74% higher than in the upper western sector of Holes Bay, and 95-167% higher than in the lower part of Holes Bay, below the railway bridge (figure 4, from data in White, 1991). Enrichment for this particular suite of metals decreases in the sequence Cu>Zn>Pb>Cr >Ni. Most elements decrease towards the Harbour mouth as finer muds become diluted with coarser, less polluted, sediment of marine origin.

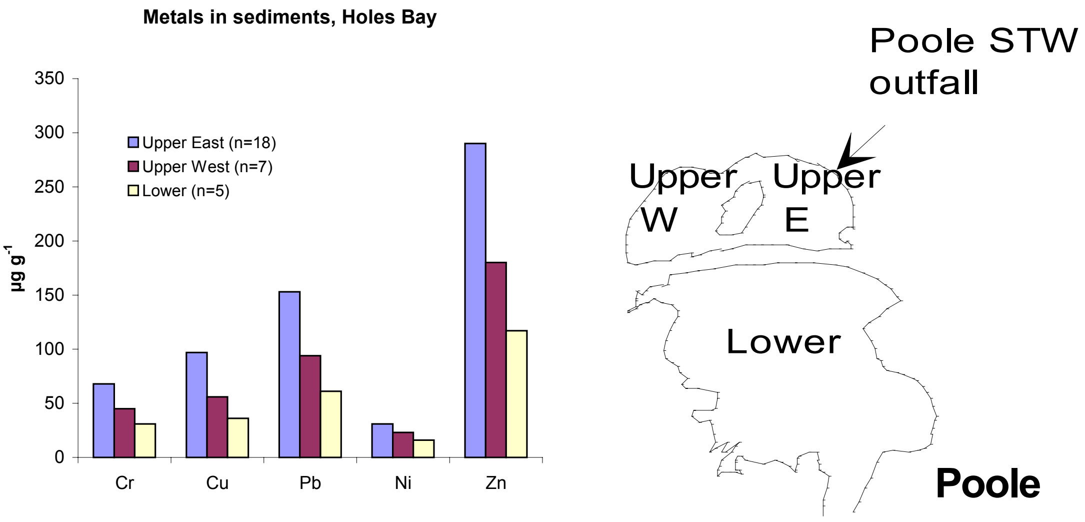

Figure 4. Metal concentrations in Holes Bay sediments indicating enrichment in the upper part of the embayment, notably in the East, nearest Poole STW discharge. (Drawn from data in White, 1991; n= number of samples in each sector).

Table 4 compares average sediment-metal data for Poole Harbour with other estuaries in the south -west, including the highly contaminated Restronguet Creek (Fal), the Severn, and the Avon in Devon (little anthropogenic input). Clearly, for metals such as As, Cu and Zn, concentrations in Poole sediments pale in comparison with those derived from mining sources in Restronguet Creek. However, enhancement in Poole sediments is observed for most metals, relative to the 'baseline' represented by the Devon Avon. This is most significant for the pollutant metals Ag, Se, Cd and Hg (six- to ten-fold higher, typically, but this enhancement may increase to twenty-fold or more in localised hotspots in Holes Bay). In contrast, there is no evidence of enrichment in As or Mn.

Table 4. Metals in intertidal sediments (µg g^{-1} dry wt): typical values for Poole Harbour and other sites in south west England.

site	Ag	Cu	Zn	Pb	Cd	Mn	Fe	As	Hg	Se	Ref
Poole	0.82	50	165	96	1.85	185	29290	14.1	0.81	1.51	1
Tamar		145-545	221-605	19-239	0.5	105-1500	21000-49000	25-236	0.2-1.5		2
Severn	0.42	35	242	84	0.63	672	26805	8.4	0.44	0.23	3
Restronguet	1.37	1690	1540	684	3	1030	54000	1080			4
Avon (Devon)	0.06	19	98	39	0.3	417	19400	13	0.12	<0.1	3

[1] Bryan and Langston, 1992; [2] Ackroyd *et al* 1986 [3] own unpublished data; [4] Aston *et al* 1975

Further discussion of the distribution of metals in intertidal sediments of Poole Harbour is presented in section 8.1 in relation to ecotoxicological guideline values.

Dredging of sediments from the lower part of Holes Bay and at Wareham is carried out to maintain navigational channels. This material is analysed by CEFAS as part of its legislative responsibility to prevent disposal of hazardous materials to sea. Results for 1996 and 1997 indicate that metal concentrations in the dredged materials were within accepted guidelines and that the metal component should pose little direct toxic threat following disposal (table 5). However, TBT levels in the same samples may be of more concern (see next section).

Table 5. Metals in dredged materials (µg g^{-1} dry wt), Poole Harbour (Data source (CEFAS, 2000)

site	n	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Wareham 1996	4	5.73	0.23	17.8	12.2	0.1	8.58	20.5	53
Holes Bay 1996	8	5.4	0.23	14.3	8.6	0.14	7.65	17.1	42.9
Holes Bay 1997	22	5.6	0.47	17.9	13.3	0.15	8.19	19.4	49.7

n = number of measurements

Biota: Although measurements of metals in sediments and water are a useful guide to environmental contamination, ultimately it is the impact on biota which is of most concern. Unfortunately, studies specifically designed to evaluate the issue of bioavailability in inter-tidal flats of Poole Harbour, using appropriate bioindicators, are few.

The distribution of key indicator species (e.g. *Nereis diversicolor*, *Scrobicularia plana*, *Mytilus edulis*, *Cerastoderma edule*, *Littorina* spp. *Fucus* spp,) and the bioavailability and impact of metals in UK estuaries, including Poole Harbour, have been the subject of research at the MBA over a period spanning three decades (see Bryan *et al.,* 1980, 1985; Langston *et al*., 1994). Though much of the data for Poole are comparatively old, they nevertheless provide a useful background to assess characteristics of the site and some of the issues surrounding metal bioavailability. They also act as a valuable baseline for future changes.

Comparative data for metals in *Nereis diversicolor* and *Scrobicularia plana*, from Poole and other UK sites, give an indication of the relative scale of concentrations (tables 6 and 7). These data are for Holes Bay and therefore probably represent ‘worst-case’ conditions in the Harbour.

Polychaetes are among the most widespread inhabitants of contaminated and uncontaminated sediments. Nereids such as *Nereis diversicolor* accumulate a number of metals in amounts which reflect bioavailability in their sedimentary environment (Bryan *et al.,* 1980; 1985; Langston, 1980, 1982). Tolerance to a wide range of salinity also makes *Nereis* extremely useful for monitoring in estuaries and *N. diversicolor* is relatively abundant throughout most of the SPA, including Holes Bay. It should be noted, however, that Zn and Fe are partially regulated by *Nereis* and therefore body burdens can underestimate contamination with these metals. It is important to recognise also that sediment conditions can modify availability somewhat (e.g. high sediment Fe and organics can reduce uptake of As, and Hg, respectively: Langston*,* 1980; 1982).

Table 6. *Nereis diversicolor*. Metal concentrations (µg g^{-1} dry weight) in Poole Harbour (Holes Bay), other sites in the south-west, and UK baselines. (Langston, Burt and Chesman, unpublished data)

	Ag	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Poole	**0.49**	**0.64**	**0.6**	**12**	**337**	**11.5**	**8.9**	**3.6**	**165**	**0.24**	**7**	**1.1**
Severn	8.01	3.79	0.52	54.4	396	14.3	4.94	3.56	264	1.42	12.8	0.31
Avon (Devon)	0.1	0.14	0.5	19	564	11.8	3.3	5.4	163	0.07	8	0.09
UK min	0.06	0.02	0.03	7.69	210	4.03	0.63	0.16	87.8	0.02	3.22	0.05
Poole ÷ UK min	8	32	20	2	2	3	14	23	2	12	2	22

na- not analysed

Table 6 shows metal concentrations in *Nereis diversicolor* from Holes Bay, Poole (mean values spanning ~25 years, therefore incorporating the recent history of contamination rather than current status). For comparison, also shown are equivalent data from the Severn pSAC, the Avon (Devon), together with the average of the lowest ten sites in our UK data set (encompassing the entire range of estuaries in England and Wales: UK min).

The degree of enrichment for Poole worms, relative to baselines, is also included in table 6 and decreases in the order Cd>Pb>Sn>Cr>Ni>Hg>Ag>>Mn>As, Cu, Fe, Zn. The relatively low enrichment of the latter two metals is almost certainly a result of the ability of *Nereis* to regulate body burdens of these essential elements.

The deposit-feeding clam *Scrobicularia plana* has also proved to be a valuable indicator species, particular in terms of understanding trends in sediment metal bioavailability. Though less widespread in distribution in Poole than *Nereis,* and in places much less abundant, it is in many respects a better accumulator and indicator of metals (with the exception of Cu), and has the advantage of not regulating Zn. Its range in the main Harbour extends from Shell Bay and Brands Bay near the mouth, to the upper reaches of the Wareham Channel. Its ability to survive in upper estuaries is presumed due to the buffering influence of its burial behaviour in sediments (upto 25cm) and an ability to isolate itself, through shell closure, from extreme low salinities. It is also present in localised populations in Holes Bay and Lytchett Bay.

Table 7 shows summary statistics for metals in *Scrobicularia plana* in Holes Bay, Poole, in comparison to the Severn, Avon and, for comparison, the average of the lowest ten sites in our UK data set (UK min). These indicate that degree of enrichment is generally of similar order to that displayed by *Nereis,* with some slight variation on ranking: Ag>Cd>Hg>Cr>Pb>Sn>Ni>Fe>Cu,Zn>As, Mn.

Both species highlight, consistently, the increased bioaccumulation of Ag, Cd, Hg, Cr, Ni, Pb and Sn in Holes Bay. As none of these metals is naturally enriched in sediments of the area, they are presumed to be of anthropogenic origin. It is stressed that these represent a historical perspective of contamination. Nevertheless, they illustrate that the use of a validated suite of indicator species is a useful way of estimating bioavailability around the system. Valuable baseline information is already in place regarding the distribution of these species (and others such as *Fucus, Littorina*, *Mytilus* and *Cerastoderma*), together with their metal burdens. The current status of the SPA needs to be more extensively defined in similar fashion, and at intervals in the future, to ensure bioavailability does not increase.

Table 7. *Scrobicularia plana*. Metals concentrations (µg g^{-1} dry weight) in Poole Harbour (Holes Bay), other sites in the south-west, and UK baselines. (Langston, Burt and Chesman, unpublished data).

	Ag	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Poole	22.5	12	5.8	46	1490	6	10.7	18	878	1.08	13	1.44
Severn	8.37	7.18	3.68	47.4	1271	69	6.44	43.5	775	0.64	20	0.39
Avon (Devon)	0.24	0.93	2.7	61	796	41	8.2	23	1078	0.9	23	0.26
UK min	0.05	0.13	0.26	9.5	226	4.8	0.89	3.79	193	0.04	5.7	0.08
Poole ÷ UK min	450	92	22	5	7	1	12	20	5	27	2	18

Residue data for other biota (*Fucus, Littorina*) generally follow similar patterns indicating that bioavailability of pollutant metals decreases away from the sewage and trade discharges in Holes Bay (own unpublished data). Boyden *et al*., (1975) described levels of Cd, Cu, Fe, Ni, Pb and Zn in *Ulva lactuca* from Holes Bay as being 2-5 times higher compared with samples from the Wareham Channel; metal concentrations in indigenous shellfish (cockles and mussels) exhibited similar gradients, decreasing out towards Brownsea Island (table 8).

Table 8. Metals (µg g^{-1}) in shellfish from Holes Bay and Brownsea Island (From Boyden, 1975).

	Cd	Cu	Fe	Mn	Ni	Pb	Zn
			Cerastoderma edule				
Holes Bay	16.9	9	502	5	174	14	271
Brownsea	1.5	4	185	3	51	5	117
			Mytilus edulis				
Holes Bay	65.4	11	154	5	12	19	162
Brownsea	3.7	7	87	3	5	7	94

It would seem that the principal areas of concern regarding metals can be sourced to Holes Bay. The metals most likely to be of significance toxicologically are Cd and Hg and perhaps locally Ag, Cu and Se. Sn levels also tend to be high in Poole samples due to the relatively high levels of organotin which is much more biologically available than inorganic Sn (Langston *et al.,* 1990, 1994; see also section 5.1.2).

Commercial fish and shellfish grounds are restricted to the outer Harbour and body burdens in these species will therefore not exhibit contamination gradients to the same extent as indicator species sampled in Holes Bay. Oysters, however, have been used in the past to demonstrate bioavailability of metals throughout Poole Harbour.

Using transplanted Pacific oysters, *Crassostrea gigas*, Thornton *et al.,* (1975) showed a significant increase in accumulation of Cd (4 to 26.7 µg g^{-1} dw) and to a lesser extent Cu (155 to 396µg g^{-1} dw) and Zn (1190 to 2670µg g^{-1} dw) after an eight month period. These oysters were laid in the north of the Harbour, presumably close to sources in Holes Bay.

Observations on the native oyster *Ostrea edulis* have also demonstrated the increased bioavailability of metals in Holes Bay (table 9). Tissues of these oysters display considerable enhancement in Ag, Cd, Pb and Hg, relative to specimens from near Brownsea Island. Copper and zinc concentrations are also elevated, though oysters are exceptional natural accumulators of these two metals. This is illustrated in comparisons with body burdens in *O. edulis* from the Fal, whose Cu and Zn contents exceed even those in Holes Bay by a significant margin. Likewise Cu and Zn concentrations in *Crassostrea,* from Poole (sampled from the outer Harbour only), are substantially lower than those in Fal oysters (table 9).

Table 9. Oysters *Ostrea edulis* and *Crassostrea gigas*: comparison of metals concentrations (µg g^{-1} dry weight) in samples from sites in Poole Harbour and the Fal Estuary.

	As	Ag	Cd	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg
Ostrea edulis												
Poole Harbour												
Holes Bay, 1983	3.2	17.2	29.9	0.6		278	219	10		1.2	3736	0.49
Brownsea Isl., 1983	3.8	2.85	1.56	0.32		40	140	8.0		0.35	1500	0.16
Fal Estuary												
Weir Point, 1992	12.4	0.05	3.25	0.53	2.04	1575	282	12.7	3.33	2.12	8723	
St Just, 1997	5.85		1.84			518	227	12.1			2974	
Crassostrea gigas												
Poole Harbour												
Brownsea Isl., 1983	12.9	1.6	4.8	0.5		61	341	27		2.5	1925	0.26
Fal Estuary												
Weir Point*			11.8			2500	365	17	5.4	9.1	7114	

Data sources: MBA unpublished results except * Boyden (1977)

EA/CEFAS sample oysters from the commercial beds as part of their monitoring programme and data for *O. edulis* up to 1990 indicate Cd and Hg have exceeded OSPAR upper guidelines in Holes Bay. In oysters from the outer Harbour and Poole Bay, Cd levels were consistently at the upper guideline level of 1 $\mu g\ g^{-1}$, whilst Hg levels sometimes exceeded the lower Hg guideline of 0.1$\mu g\ g^{-1}$ but never the upper limit of 0.3 $\mu g\ g^{-1}$ (White, 1991).

Because commercial oyster beds are situated towards the Harbour mouth, at some distance from sources of metals in Holes Bay, it seems unlikely that changes in consents will have any impact on oyster populations in terms of their metal burdens. It would be helpful to confirm this with more recent information on oyster body burdens and water quality.

Results of EA monitoring of farmed mussels (1983-1992) confirm that metal (Ag, As, Cd, Cr, Cu, Ni, Pb, Hg, Zn) levels at the outer harbour shellfish beds were below guideline values, though increases in Zn and possibly As were observed over this period[1]. In order to help meet standstill requirements under the EC Dangerous Substances Directive, the EA, in 1996, included Poole Harbour among its monitoring sites along the Wessex coastline, as part of its regular Bioaccumulation Programme[2]. This is essentially a rocky shore survey, therefore the species used as indicators were restricted to hard-substrate epibionts including, *Mytilus edulis, Fucus* spp and *Patella* spp. Sampling of these species was consequently confined to the outer Harbour. For *Fucus* spp (the principal indicator used in Poole Harbour), metal concentrations were not considered as being 'substantially elevated', though they appear to be above background, for the Wessex region, at one or more sites (As, Cu, Pb, Hg, Zn and Cr). In particular, Zn levels at Rockley were relatively high (see also section 7, which indicates anomalous concentrations of dissolved Zn in this area). The concentrations of Hg in seaweed from Brownsea Island, though not exceptional, were also the highest recorded along the Wessex coast in 1996. In terms of temporal data, seaweed samples from Sandbanks indicate a slight upward trend in Hg between 1993 and 1996.

Since *Fucus* is likely to accumulate metals from water, principally, these observations largely relate to bioavailability of dissolved metals. Clearly, this is important information for the Wessex coastal zone as a whole, but in terms of the Poole SPA, specifically, monitoring of detritivores and other infaunal types should be incorporated as a priority in long-term surveillance programmes, as suggested above. This is essential to address the issue of bioavailability of sediment metals. Soft-substrate habitats occupy most of the harbour and are becoming increasingly important, relatively, as diffuse sources, particularly in the more contaminated areas in secondary embayments such as Holes Bay (where rocky shore indicators are atypical or absent). Many infaunal species are also important prey items for predators including the birds for which the SPA is designated.

To date, there have been few investigations of the effects caused by metal pollution in Poole Harbour. Dyrynda *et al.,* 1998 monitored a variety of immune parameters in the marine mussel, *Mytilus edulis*, transplanted from a clean reference site in Poole Bay to Holes Bay. Compared to the reference site, haemocytes of transplanted mussels showed enhanced superoxide production and reduced activity of degradative

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[1] Ponting, J, Shellfish Bioaccumulation Report to the Biology Unit, Blanford, 1993

[2] EA report south Wessex region (Biology). Bioaccumulation of metals in seaweeds, limpets and mussels along the south Wessex Coast (1980-1996).

enzymes - effects which may result in tissue damage or have implications for the efficient destruction of invading microorganisms. Significant reductions in the binding of certain monoclonal antibodies to haemocyte epitopes were also observed. However, it was not possible to attribute effects to any one contaminant, which may indicate a more general stress response.

The physiological significance of metal burdens, has been examined more specifically by determining metallothionein (MT) induction and intra-cellular metal-binding patterns. Metallothionein is a metal-binding protein induced in direct response to metal contamination. Molluscs such as *Littorina* and *Mytilus* spp were tested in this capacity in preliminary trials in Poole Harbour, conducted in the mid 1980's, and resulting data illustrate the value of this approach.

The simplest example is that is shown in figure 5, which compares sub-cellular Cd binding patterns in mussels from Holes Bay, Poole, and an open coastal reference site at Godrevy in North Cornwall (own unpublished data). Induction of Cd-binding MT is significantly enhanced in the Poole mussels, in response to the elevated levels of bioavailable metal. Most of the excess Cd here appears to be sequestered by MT protein in an attempt to detoxify the metal (preventing interaction with enzymes and other vital molecules).

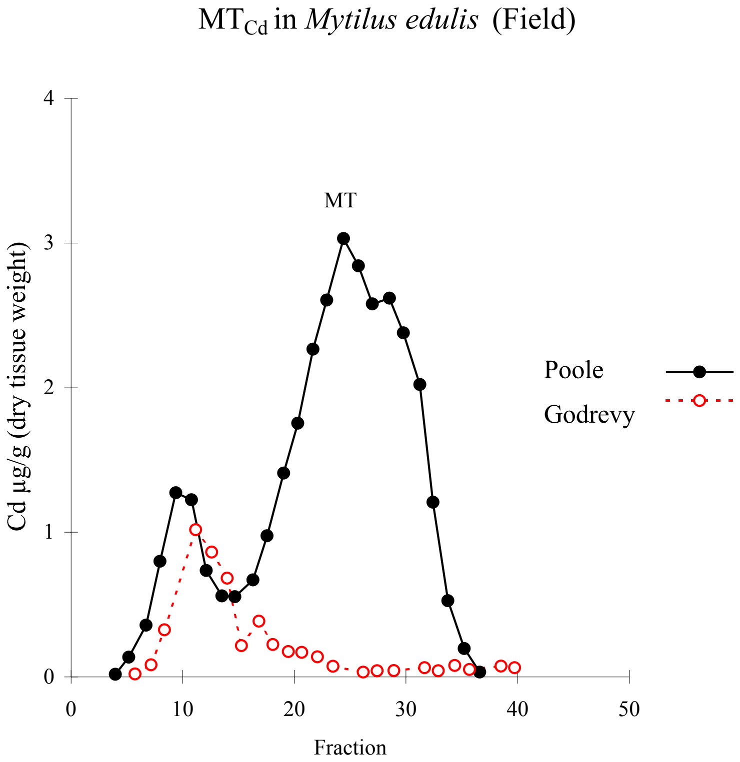


Figure 5. Size-exclusion chromatograms of cadmium-binding in *Mytilus edulis* from Holes Bay (Poole) and a reference site at Godrevy, showing induction of metallothionein (MT) at the former site (own unpublished data , 1985 samples).

Cadmium-binding patterns in *Littorina* from Holes Bay (body burden 31 µg g^{-1} Cd), Lytchett Bay (3.9µg g^{-1}) and a control site at Penzance (1.2µg g^{-1}) are compared in figure 6. These chromatograms reveal a more complicated 'fingerprint' of exposure history. MT induction in Poole *Littorina* is again signified in response to increasing levels of contamination (Holes Bay> Lytchett Bay> Penzance). However, there is also a potentially more significant 'signal' in the appearance of excess Cd in 'abnormal' metal fractions in Poole samples. Cd becomes associated with HMW (High Molecular Weight) ligands (e.g. enzymes) as a result of saturation of MT. In the more heavily contaminated Holes Bay gastropods, Cd is also present in LMW (Low Molecular Weight) fractions which includes amino acids and any free (potentially toxic) metal. This pattern signifies saturation of MT - the so-called 'spillover effect'- and may be associated with the onset of deleterious events.

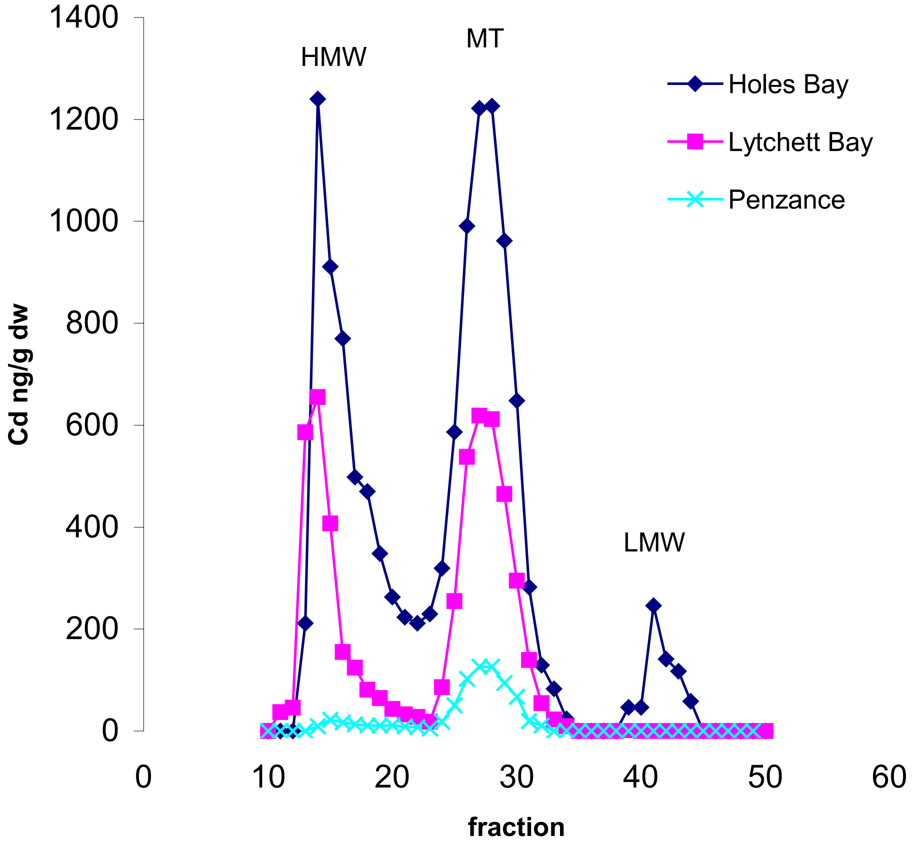


Figure 6. Cadmium-binding patterns in *Littorina littorea* from Holes Bay and Lytchett Bay (Poole), and a reference site at Penzance (adapted from Langston and Zhou, 1986). MT= metallothionein; HMW, LMW= high and low molecular weight pools, respectively.

The above results illustrate the value of the MT assay as an 'early-warning' indicator of metal impact, providing an insight into the toxicological significance of metal burdens. Quantification of the MT protein is now possible, adding to the possibilities for mapping the spatial extent of biological response to metal contamination. There is also scope to deploy the assay to show temporal trends; i.e. to demonstrate improvements arising from planned schemes to reduce metals, standstill provisions of the Dangerous Substances Directive, or as may be required under the Habitats Directive to achieve Favourable Condition.

The extent of ecological impact due to metal contamination in Poole Harbour is largely unknown, but is probably restricted to parts of Holes Bay. White (1991) established that, within Holes Bay, infaunal communities varied between upper (east and west) and lower areas of the Bay, as did a number of sediment parameters, including a small suite of metals (see above); however, multivariate statistics did not pick out a dominant controlling factor. It seems likely that a combination of features (organic enrichment and anoxic conditions in sediments of the inner bay, large salinity variations, and a cocktail of toxic and non-toxic contaminants) - largely determined by the STW inputs - are of significance in modifying communities in the Bay.

Nereis diversicolor appears to be able to out-compete most other infaunal species, close to the STW discharge, perhaps because of it's known ability to tolerate metals. It is important to stress here that *Nereis* is an important dietary item for birds and fish, and therefore is a major vector for the transfer of metals and other contaminants along food chains. By comparison, juvenile bivalves (particularly sensitive to Cu and Zn) are much reduced in numbers towards the outfall.

In recent years, the impact of metals on biota in Holes Bay may have been overshadowed by the effects of TBT, and nutrient enrichment. However, there is scope for more research to assess this issue.

5.1.2 TBT

Use of tributyltin (TBT) antifouling on boats less than 25m in length was prohibited in 1987, though larger vessels (the commercial fleet and Navy) are still entitled to use them, at least until 2003 when recommendations from IMO for a total ban should be implemented. The classic TBT indicator *Nucella lapillus* is not a native of Poole Harbour (unsuitable substrates and other physical and chemical constraints), though imposex has been observed in dog-whelks from nearby Swanage. Closer to Poole (Studland), there are indications that the species may have been eliminated by TBT in the late 1980s (Crothers, 1975, had previously reported them here a decade earlier).

Prior to 1987 attempts to grow *Crassostrea gigas* at shellfish beds near Brownsea island proved abortive as TBT-induced shell-thickening resulted in abnormal and unmarketable oysters. Two years after the ban, in 1989, there was evidence of reversion to more normal growth in the remnants of this stock, and in freshly laid two-year old oysters deposited here and in the South Deep - both outer Harbour locations with low boat traffic (Dyrynda, E.A., 1992). However, animals layed in Holes Bay (an area of high boat density) continued to show abnormal shell-thickening after legislation, consistent with delayed return of TBT concentrations to no-effects levels (see below).

Time-series studies at a variety of TBT-impacted sites were initiated by MBA staff in the 1980s, and included Poole Harbour. After TBT legislation in 1987, seawater concentrations of TBT typically declined in small-boat dominated areas, such as Poole. However, sediment particles scavenge TBT from the water column and provide a longer-term reservoir of the compound. Rather than reducing TBT bioavailability, sediments may provide a major route for uptake (and toxicity) for some species. This appears to be the case for a number of sensitive molluscs including deposit-feeding clams such as *Scrobicularia plana,* which were common in the 1970s but had declined considerably by the 1980s. Some of the data describing these events, between 1987 and 1994, are outlined below. (Further details can be found in Langston *et al.,* 1987,1994b and Langston and Burt, 1991).

Prior to restrictions in 1987, Poole Harbour was considered to be a possible worst-case situation with regard to contamination from tin-based anti-fouling paints. The entire water-body is sheltered and so utilised by several thousand leisure vessels. The narrow harbour entrance and limited tidal range impart lagoon-like characteristics and result in relatively restricted currents throughout much of the primary basin. Similar features produce an even less energetic tidal regime in the secondary basins such as Holes Bay. As a consequence of the high density of vessels and sheltered conditions, elevated TBT residues were encountered in various parts of the Harbour during the 1980's.

Other than the cross-channel ferry service and a small number of commercial vessels, large ships (which are still entitled to use TBT-based paints) seldom frequent the harbour. Since boat-traffic is therefore dominated by small-vessels which are prohibited from using such paints, Poole was considered an ideal location in which to examine distributions in relation to boating patterns and, subsequently, to judge the effectiveness of TBT legislation.

A field-data set was initiated at nine sites around the Harbour (table 10) incorporating TBT and DBT analysis of sediment, water and a variety of species (*Scrobicularia plana, Mya arenaria*, *Nereis diversicolor*, *Fucus vesiculosus* and *Littorina littorea,* where available). Marinas and high-density moorings are predominantly situated along the northern part of the shoreline, between Lytchett Bay and the Harbour mouth. In contrast the southern shoreline is relatively undeveloped, although extraction and processing of oil occurs in the vicinity of Wytch Farm.

Up to 1994, TBT contamination throughout the Harbour was still clearly related to the densities of moorings and routes of major boat traffic. Highest concentrations were encountered at sites in the northern sector of the main harbour and in Holes Bay (up to 229 ng l^{-1} in water from lower Holes Bay, for example). At sites along the southern shoreline of the Harbour, and at Sandbanks (just outside the Harbour mouth), TBT concentrations were considerably lower (0.2-2.8 ng l^{-1}), though still close to, and sometimes exceeding the EQS value of 2ng l^{-1} TBT. A summary of TBT levels in water is shown in table 10.

Table 10. Summary statistics TBT in water Poole Harbour 1989-1992 (From Langston *et al.,* 1994b)

		TBT ng/L as Sn		
site	site number *	mean	minimum	maximum
Wytch Farm	1	0.87	0.2	2.8
Brands Bay	2	0.94	0.2	2.0
Jerry's Point	3	1.09	0.4	3.0
Sandbanks	4	1.20	0.2	2.7
Marina	5	22.60	7.4	58.0
Parkstone	6	14.09	6.3	28.6
Sterte	7	9.06	1.3	22.5
Power Station	8	51.01	10.4	229.0
Lytchett Bay	9	7.74	0.6	24.7

The success of TBT regulations, in reducing inputs to the water column from leisure vessels, can be assessed from samples taken close to marinas (e.g site 5, table10). Here, TBT concentrations in water up to 1300ng/L were recorded prior to 1987, with evidence of distinct seasonal inputs as freshly painted boats were re-installed at their moorings in summer months. Since then, seasonal releases were on a very much reduced scale and the mean concentrations in the early 1990s (22.6 ng/L), though still much higher than the EQS, were some 50-fold lower than earlier maxima (figure 7).

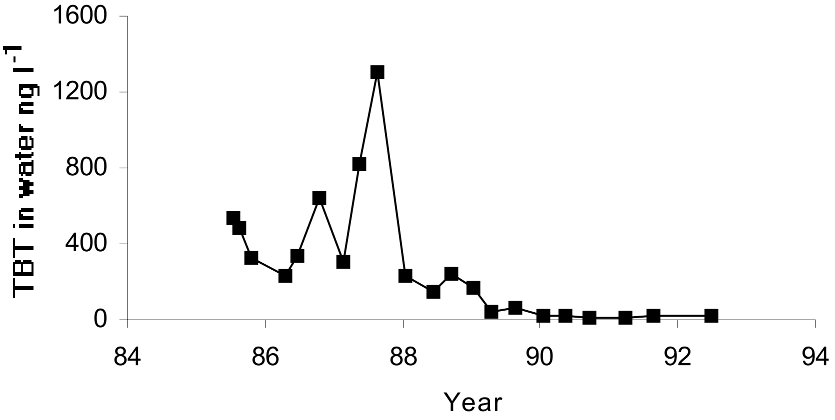


Figure 7. Temporal trend in TBT in water at a marina site, Poole Harbour (1985-1992)

Although legislation has had obvious and immediate effect in stemming new inputs of TBT from smaller craft, there were indications that the rate of decline in TBT concentrations in water was slowing down by 1994. Similar trends were observed at other sites, though half-lives were variable (13–35 months) and, occasionally, trends were reversed. The reasons for such variability are not yet known. It is possible that 'new' TBT was still being introduced illegally into the system, causing transient pulses of TBT in water. However, a more general explanation for the slow disappearance of contamination at some sites seems to be related to the retention and persistence of TBT bound to particulates: Sediments may be acting as reservoir and a source for the continuing gradual release of TBT, even in the absence of fresh inputs.

Between 1989 and 1992, mean concentrations of TBT and DBT in surface sediments ranged from low values of 0.014 and 0.027 mg/kg (as Sn), respectively, at southern sites, to 0.52 and 0.24 mg/kg at Parkstone in the north (note the provisional ecotoxicological guideline value for TBT is set as 0.00005 mg/kg – OSPARCOM, 2000). As expected, the distribution of TBT in sediments closely resembles that described for water, with severest contamination being restricted to areas associated with high boating activity.

Sediment-water partition coefficients (K_p), the ratio between TBT in sediment (mg/kg) and water (mg/L), varied between 2.2 and 170 x 10^3 (mean 29.5 x 10^3). These indicate relatively high affinity of TBT for the particulate phase. Removal of TBT to sediments is therefore important and explains why high concentrations tend to be retained close to the major inputs in the sheltered environment of Poole Harbour.

As a result, reductions in TBT concentrations in the water column do not appear to be matched by equivalent reductions in sediment-bound TBT, indicating that the latter is the more persistent form. The long-lasting nature of TBT (and other contaminants) in sediments is, clearly, likely to be most pronounced in the depositional environments of the secondary embayments.

For the least contaminated sites (generally in the south), temporal trends in TBT (and DBT) contamination in sediment follow a common pattern (as indicate for Wytch Farm in figure 8). Significant reductions occurred at the time restrictions on the use of TBT-based paints came in to force, though the rates of loss have subsequently declined. The overall half-time for TBT at these southern sites is 53.7 months - much longer than in overlying water. Temporal trends in sediment TBT concentrations at the more contaminated stations in the northern part of the harbour (e.g Parkstone figure 8) are difficult to determine. Slopes used to calculate half-times for sediment-TBT, at four sites in the more contaminated northern sector, could not be established as being significantly different from zero.

The fact that DBT is present in sediments (in proportion to TBT levels) suggests that some degradation of the parent compound occurs and that that the persistence of DBT may be similar to that of TBT. Estimations of half-times for DBT, range from 44.4 months to 80.9 months.

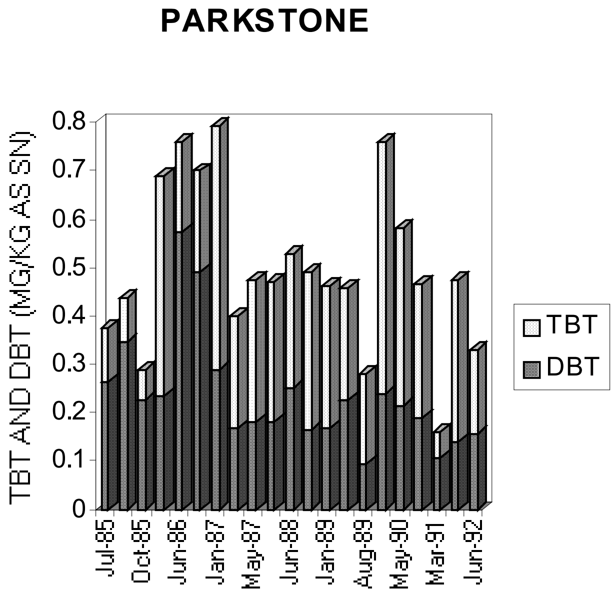
Re-survey of these sites is recommended in order to update these effective half-times for organotins and to re-evaluate the prognosis for recovery.

Body burden data were assembled for several bioindicators, from the same range of sites in Poole Harbour as indicated above. Although different bioindicators display comparable spatial and temporal patterns, their ability to accumulate TBT can vary considerably depending on uptake route and capacity to metabolise the parent compound. Sediment-dwelling molluscs tend to accumulate the highest body burdens, most notably *Mya arenaria* (Langston *et al*., 1987;1994b). Results for the clam *Scrobicularia plana* serve to illustrate the general trends in bioavailability of organotins.

As with sediments, concentrations of both TBT and DBT in *S.plana* vary by more than an order of magnitude between "clean" sites in the south (e.g Wytch Farm, figure 9) and the more contaminated sites in the north (such as Parkstone, figure 9). It should be noted however that, up until 1992, clams could not be found at some of the most polluted stations (e.g in Holes Bay), where, judging from numbers of shells, they had previously been common.

Significant relationships were observed between organotins in clams and the level of environmental contamination (water and sediment). The slightly higher r-value for sediments appears to confirm other field and experimental evidence which suggests that particulates are a more important source of TBT to clams than water (Langston and Burt, 1991).

Organotin (TBT and DBT) levels in clams declined appreciably following the introduction of restrictions in 1987 (figure 9) though there were indications that, during the last three years of observations (1989-1992), losses had slowed considerably, mirroring trends for sediments. Calculated half-times for TBT in *S. plana* range from 25.5 months at sites low in contamination to 57 and 59 months at the more impacted sites. These numbers are close to equivalent estimates for sediments, and are considerably longer than respective TBT half-times in water.



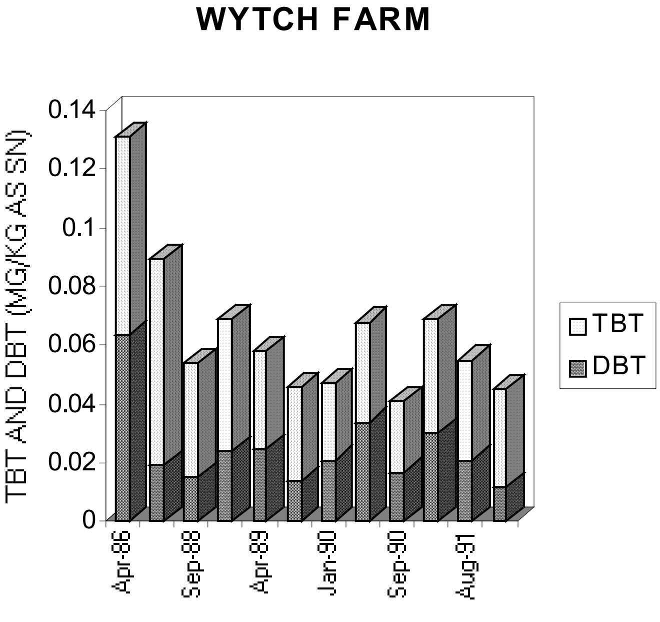
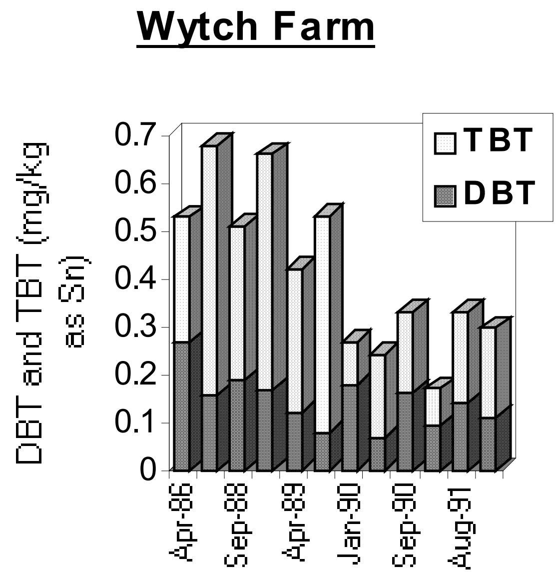


Figure 8. TBT and DBT in sediments Poole Harbour, 1986-1992 (from Langston *et al.*, 1994b).



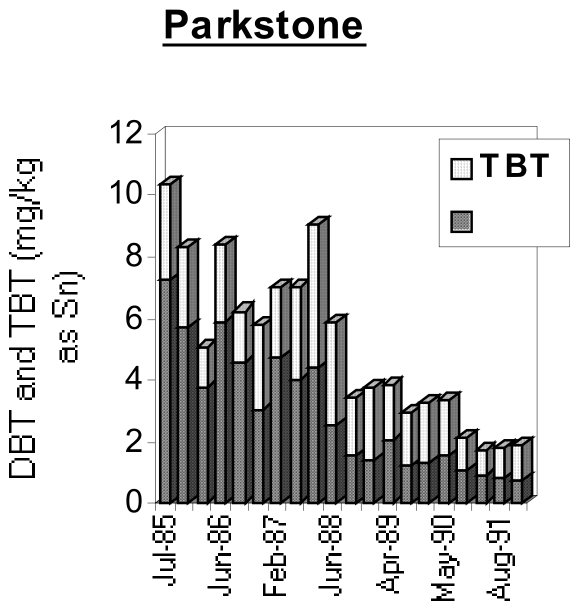


Figure 9. TBT and DBT in clams *Scrobicularia plana,* Poole Harbour, 1986-1992 (from Langston *et al*., 1994b).

Attempts were made to quantify the effects of TBT on infaunal bivalves, since it had been observed in Poole Harbour and other popular boating areas that populations of some species, such as *S.plana,* were in decline (Langston *et al* 1990; 1994b). Conventional random quantitative sampling has proved to be of limited value in assessing the abundance of adult infaunal clams, at some sites, due to their patchy distributions and low numbers. A better impression is gained simply from timing the collection of fixed numbers, usually achieved by searching for siphon marks. These semi-quantitative estimates appear to establish an inverse correlation between TBT levels and the occurrence and abundance of adult clams. Quite often, moderately impacted sites sampled in this way were characterised by low numbers of older clams, perhaps settled prior to the surge in TBT usage in the mid 1980s (and followed by low recruitment since that time). Experiments have confirmed that concentrations of sediment-bound TBT in the range 0.1-0.3 µg g^{-1} (equivalent to levels in the more contaminated Poole sediments) would almost certainly contribute to the decline of clams numbers (Langston and Burt, 1991).

Quantitative sampling, e.g. by coring, is more representative for monitoring the settlement of juvenile bivalves. Results from sites in Poole Harbour suggest that spatfall occurs widely, but at contaminated sites these juveniles do not appear to survive long enough to establish the type of age-frequency structure encountered in "normal" (TBT-free) populations such as those found in the Exe (figure 10). Low numbers of post-juvenile stages, and resulting abnormal size-frequency distributions for clam populations have been a feature at a number of other TBT-contaminated locations.

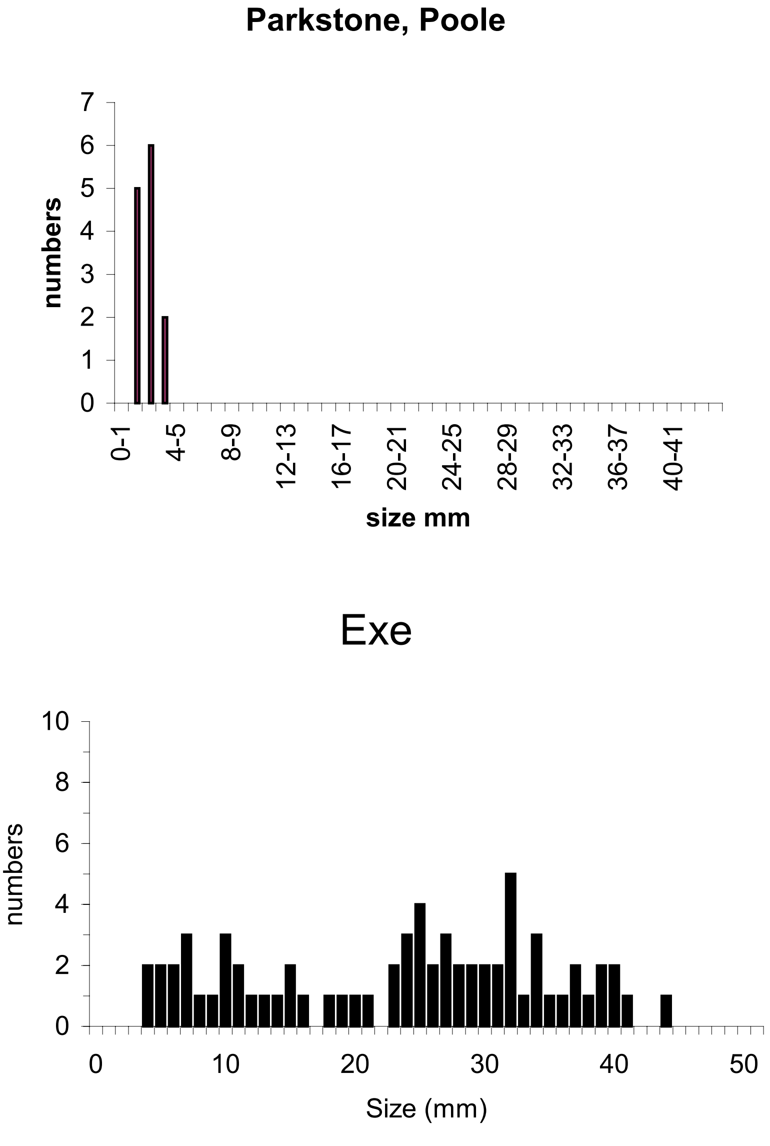


Figure 10. ***Scrobicularia plana*****. Size frequency histogram of populations from Parkstone, Poole Harbour (sediment TBT 0.27 µg g^{-1}) and from Lympstone, Exe Estuary (<0.01 µg g^{-1}). From Langston *et al.*, 1990.**

Ad hoc sampling by MBA staff has continued in Poole Harbour (biological and chemical) but it is now a decade since a detailed survey has been performed. It is likely that waters at some locations still exceed the EQS. Extremely slow recovery of mollusc populations in parts of Poole Harbour might be expected in view of the persistence of TBT in sediments. It would be timely to conduct such a study now to monitor the progress towards recovery and to establish better predictions for the future.

Poor sediment quality in parts of Poole Harbour, attributable partly to TBT, has also been inferred from surveys undertaken for the National Monitoring Programme (MPMMG, 1998). In recent years CEFAS have analysed dredge spoil samples in the SPA (from Holes Bay and Wareham) as part of the procedure for licensing for disposal at sea (CEFAS, 2000). TBT concentrations at these locations are indicated in table 11 for the period between 1994 and 1997. These are above ecotoxicological guidelines recommended by OSPAR and, bearing in mind these represent *average* levels in dredge material, are relevant, ecologically. TBT concentrations in dredged sediments near heavy boat traffic or moorings can be variable, even over a small area, which could be due to the heterogenous distribution of paint particles. At one Poole boatyard, for example, dredged sediments ranged from 3.03-14.4 µg g^{-1} TBT. Using bioassays with *Arenicola marina*, *Corophium volutator* and *Tisbe battagliai,* toxicity was observed in some of these samples, though not all (Thain *et al*., 2000). Significant reduction in cast production in *Arenicola*, a sign of reduced feeding activity, was also observed in sediments collected from Brownsea and Holes Bay in the mid 1990's (CEFAS, 1998). Although mortalities of *Arenicola* (and *Corophium*) were relatively low, results were interpreted as indicating diminished sediment quality. Acute toxicity is probably not to be expected, except close to contaminant sources, though it is possible that chronic sediment impairment is more widespread. In addition to TBT, mud from Holes Bay may contain elevated levels of Cu (273µg g^{-1}) that may contribute to the overall toxicity of these sediments.

Thus, studies with Poole sediments indicate there are substantial reservoirs of TBT, albeit localised to areas near marinas and boatyards in the north, which should raise caution, particularly if they are to be dredged. Processes including physical resuspension and bioturbation could remobilise these sinks. Furthermore, TBT in such contaminated sediments is likely to be available and potentially harmful to deposit-feeders and infauna (Langston and Burt, 1991).

Table 11. TBT concentrations in dredged sediments collected between 1994 and 1997. (Data source: CEFAS, 2000)

location	year	number	TBT µg g^{-1} wet wt
Wareham	1996	4	0.13
Holes Bay	1997	21	0.15
Holes Bay	1996	5	0.19
Holes Bay	1995	10	1.35
Holes Bay	1994	6	0.8

5.1.3 Hydrocarbons (Oil, Petrochemicals, PAHs)

Oil

Oil pollution is a continual threat to all inshore marine habitats, and is particularly pronounced in Poole Harbour due to its enclosed and sheltered nature. Risks include small leaks, spills and discharges, as well as the possibility of a major accident.

There are a number of ways in which oil could potentially impact on the interest features of the SPA. Inter-tidal habitats are under greatest threat from the physical effects of oil pollution: the most vulnerable of these are inter-tidal sand and mudflats and salt-marshes of the inlets and bays (see reviews of vulnerability of shores to oil damage by Gundlach and Hayes, 1978; Elliott and Griffiths, 1987). In extreme events lethal effects would induce community changes.

The direct effects of oil on shellfish beds are potentially serious. In the event of moderate spillages, significant mortalities of bivalves would be expected.

Birds would be affected by consumption of contaminated food and damage to plumage.

Oil pollution may result in hydrocarbons becoming incorporated into sediments and buried. Heavily contaminated sediments are likely to have acute effects on populations of bottom-dwelling fish such as plaice and sole (in extreme cases, such as the Amoco Cadiz spill, whole year classes were wiped out over large areas of northern Brittany). Sub-lethal changes could be manifested as increased bioaccumulation and tainting, induction of components of the mixed function oxidase (MFO) enzyme system (e.g. ethoxyresorufin-O-deethylase – EROD - activity), and higher order changes in productivity, fecundity and behaviour. Olfactory responses in crustacea can affect their searching, feeding and grooming responses. Fish behaviour, including migration, is also known to be affected by oil spills with most fish avoiding areas of heavy contamination.

Sensitivity of *Zostera* beds to chronic exposure to oil (refinery effluent) may not be very high (Hiscock, 1987). The likely impact of acute exposure (oil spillage) will be influenced by the type of oil, the degree of weathering and the nature of the habitat and in general, it is the associated faunal communities that are more sensitive to oil pollution than the *Zostera* plants themselves (Jacobs, 1980, Zieman 1984, Fonseca, 1992). As is often the case, dispersants are likely to be more harmful to *Zostera* than oil, and coated plants should be left untreated.

Eggs and planktonic larval stages of fish, molluscs and crustacea are also vulnerable to contact with oil in surface waters. Because of the physically contained nature of the Poole Harbour marine site, recruitment could be threatened over relatively long time scales.

The hydrocarbons present in crude oil can range from aliphatic (straight chain) compounds to more complex aromatic (containing a benzene ring) and polynuclear aromatic (containing two or more benzene rings) compounds. Processed products include petrol and diesel and a range of petrochemicals, e.g. propylene, acetylene, benzene, toluene and naphthalene. In addition to shipping, sources also include river-borne discharges, (including road runoff and licensed and unlicensed discharge to sewers) diffuse discharges from industrialised municipal areas, oil production sites (e.g. accidental release from the Wytch Farm site) and the atmosphere (PAH's).

Surprisingly few data on total petroleum hydrocarbon (THC) concentrations in Poole Harbour could be found in the literature. MAFF data from 1992 report levels in water of the outer Harbour ranging from 1.1µg l^{-1} near Brownsea Island up to 6.4µg l^{-1} at Lake (MAFF, 1994). These are relatively low in comparison to the range of concentrations (17-64µg l^{-1}) found in industrial estuaries such as the Tees.

In order to provide pre-operational (1985) baselines prior to commissioning of the Wytch Farm site, BP[1] commissioned THC measurements in inter-tidal sediments of the outer Harbour, mainly along the southern and western shoreline (where highest levels were found, generally), and around Furzey Island (lowest levels). Concentrations varied between 1.4 and 127 µg g^{-1} and were considered as being intermediate between pristine and heavily industrialised sites. The composition of aliphatic hydrocarbons in these Poole sediments was fairly typical of inshore sediments (substantial proportion derived from terrestrial plant matter) but most sites also had a signature consistent with a degree of (weathered) petroleum contamination (manifested as an unresolved complex mixture). N-alkanes were measured in the range 0.4-16.6 µg g^{-1}. Likely sources of these aliphatic components were proposed to be inputs from boating and shipping, and sewage. At the time of sampling most of these anthropogenic hydrocarbon signatures indicated chronic long-term accumulation. At two sites more recent inputs were suggested.

Interestingly, THC levels in these sediments were highly correlated with metals: both groups of chemicals appear to have common geochemical controls on their overall distribution e.g. organic matter content, oxyhydroxides and % fines. (Note: though distant from the contaminated sites in the northern sector of Poole Harbour, the metal content of these sediments from the south and west shorelines was considered noticeably higher, for Cu, Pb, V and Zn, than baselines from Sullom Voe inshore sediments).

THC concentrations in unspecified invertebrates are reported by Allen *et al* (2000) and shown in table 15; on face value these are relatively low, though without any indication of species name or location, and in view of the small sample size, little further comment can be made.

PAH's

PAH's are ubiquitous environmental contaminants, estimated to constitute some 8% by weight of the total hydrocarbon composition (Kirby *et al.*, 1998). Although they can be formed naturally (e.g. oil seeps, forest fires) the predominant source of PAHs is often anthropogenic emissions, and the highest concentrations are generally found around urban centres (Cole *et al.*, 1999). In the aquatic environment, PAHs are generally highest in sediments, intermediate in biota and lowest in the water column. They are of particular concern in the marine environment as the lower molecular weight PAHs are toxic to marine organisms, and metabolites of the higher molecular

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[1] Report for BP by Heriot-Watt University (Institute of offshore Engineering): Biological and chemical intertidal survey of Poole Harbour, June 1985. IOE/85/351.

weight PAH's are carcinogenic (Law *et al*., 1997). PAH concentrations in sediments may be persistent (particularly where tidal action is restricted, and degradation limited by anoxia). Elevated levels have been linked to liver neoplasms and other abnormalities in demersal fish (Malins *et al.,* 1988). In addition, some PAHs have been identified as endocrine disruptors (Anderson *et al*., 1996a,b; Kocan *et al.,* 1996).

There are few published papers relating to PAHs in waters of Poole Harbour. Law *et al*. (1997) report the results of a survey of PAH's in coastal waters around England and Wales, undertaken as part of the UK National Monitoring Programme (NMP). This indicates the presence of PAH contamination in Holes Bay (table 12). The total concentration of 15 PAHs measured in unfiltered water from Holes Bay, taken over two consecutive days, was 16-70 ng l^{-1} and the major compounds were three- and four-ring PAHs, phenanthrene, fluoranthrene and pyrene. Naphthalene, a more volatile and water soluble 2-ring PAH was the major component of a filtered sample (ΣPAH= 35 ng l^{-1}) from the same site (table 12).

Table 12. Concentrations of PAHs in water, Poole Harbour (adapted from Law *et al.,* 1997)

	Concentrations of PAHs in unfiltered water (ng l^{-1})							Concentrations of dissolved PAHs (ng l^{-1})
	Poole Bay	Poole Harbour, Holes Bay	Poole Harbour, Brownsea Island	Poole Bay	Poole Harbour, Brownsea Island	Poole Harbour, Stakes buoy no. 55	Poole Harbour, Holes Bay	Poole Harbour, Holes Bay
Date	17/06/93	17/06/93	17/06/93	18/06/94	18/06/94	18/06/94	18/06/94	17/06/93
Naphthalene	<15	<15	<15	<6	<6	<6	<6	23
Acenaphthene	<3	<3	<3	<1	<1	<1	<1	<3
Fluorene	<1	4	<1	<2	<2	<2	<2	<1
Phenanthrene	<8	10	<8	<3	<3	<3	<3	<8
Anthracene	<1	1	<	<1	<1	<1	<1	1
Fluoranthene	<1	13	<1	<2	<2	<2	9	8
Pyrene	<1	13	1	<1	<1	<1	7	3
Benzo[a]anthracene	<2	6	1	<6	<6	<6	<6	<2
Chrysene	<2	5	<2	<4	<4	<4	<4	<2
Benzo[e]pyrene	<1	<1	<1	<2	<2	<2	<2	<1
Benzo[b]fluoranthene	<1	5	<1	<2	<2	<2	<2	<1
Benzo[k]fluoranthene	<1	2	<1	<4	<4	<4	<4	<1
Benzo[a]pyrene	<1	5	<1	<4	<4	<4	<4	<1
Dibenzo[a,h]anthracene	<1	1	<1	<12	<12	<12	<12	<1
Benzo[g,h,i]perylene	<1	5	1	<17	<17	<17	<17	<1
Total PAH	ND	70	2	ND	ND	ND	16	35

ND: no PAH compounds detected
NR: no result

In comparison with concentrations in industrialised estuaries (e.g River Tees, up to 10724ng l^{-1}), PAHs in waters of Poole Harbour, particularly outside Holes Bay, appear to be relatively low (Law *et al.* 1997) and would not be expected to exhibit direct acute toxicity. ΣPAH concentrations fall below the threshold of 1 μg l^{-1} considered by Cole *et al*., (1999) to be environmentally important and below the EU maximum threshold of 200ng l^{-1} to protect human consumers of aquatic life (though not the equivalent US criteria of 30 ng l^{-1}). Individual PAHs generally fall within tentative upper 'safe-levels' such as those proposed by OSPARCOM (1994).

However, although water may not be acutely toxic, a contribution towards possible chronic (sub-lethal) biological effects in Holes Bay is tentatively suggested for non-polar and semi-polar organics, including total hydrocarbons and PAHs. Using unconventional bioassays with the copepod *Tisbe battaglia,* CEFAS tested toxicity following hexane extraction of water samples and subsequent serial concentration procedures (Kirby *et al*., 1998). Although contaminants in offshore waters (e.g. the Western Approaches) often require almost 1000-fold concentration before eliciting toxicity, those in estuarine waters typically require only 10-300-fold concentration. Samples collected in 1993 from Poole Harbour were fairly toxic by comparison (requiring concentration factors of 71-155 at Brownsea and Holes Bay; 126-318 offshore in Poole Bay) - surpassed only by samples from the Tees, Tyne, Wear, Mersey and Blyth. In the previous year a concentration factor of Holes Bay water of only 13 was required to induce acute toxicity (CEFAS, 1998).

Using this technique in samples taken over a tidal cycle, highest toxicity in Poole waters occurred in samples taken at low water - indicative of terrestrial and shore-based origins. However, whilst hydrocarbons were considered as likely contributors to the response, the cause of the majority of the toxicity was not identified. As TBT is concentrated by this procedure it may well be a likely candidate: assuming a background of only 2 ng l^{-1} in Poole samples (probably a conservative estimate in some areas –see section 5.1.2), the concentration procedure would produce a level of 142-310 ng l^{-1} TBT, almost certainly contributing towards effects. It is therefore possible that the (weak) correlation between toxicity and hydrocarbon levels in these assays reflects the fact that the latter act as a general surrogate for other groups of contaminants, whose distributions co-vary (Kirby *et al*., 1998; CEFAS, 1998).

One general concern over a number of PAHs (benzo[a]anthracene, dibenz[a,h]anthracene, benzo[b]fluoranthrene, benzo[k]fluoranthene, chrysene, benzo[a]pyrene) is that the mixed function oxidase (cytochrome P450) system can be triggered, producing carcinogenic and mutagenic metabolites in fish. However, the risk of carcinomas is probably low at environmental concentrations of 1µg l^{-1} or less, as present throughout much of Poole Harbour (Payne *et al*., 1988; Law, *et al.,* 1997).

Based on observed environmental behaviour, physical and chemical properties, microbial degradation rates and statistical analyses, PAHs are divisible into two groups: *Group 1* or low molecular weight (≤200) PAHs (including naphthalene, phenanthrene and anthracene) have a low affinity for particulates and are subject to microbial degradation. Their solubility and vapour pressure is higher than group 2 PAHs, and photo-oxidation and air-water exchange are important in estuaries. Consequently group 1 PAHs tend to have comparatively shorter residence times and often exhibit a complex distribution pattern. In contrast, *group 2* or high molecular weight (≥200) homologues (including benzo(a)pyrene, fluoranthrene, pyrene and chrysene), are readily adsorbed onto particulates. They are often correlated with suspended solids along estuaries and, due to the high particulate affinity and microbial refractivity, the principal fate of group 2 PAHs may be sediment burial.

Since many PAHs (especially group 2) have such an affinity for particulates, concentrations in estuarine sediments tend to be much higher. Data from a pilot study of PAHs in sediments around the UK are reported as part of the National Monitoring Programme (MPMMG, 1998; Woodhead *et al*. 1999). Concentrations range from undetectable at offshore sites (such as the central English Channel) to 43,470µg kg^{-1} dry weight on the River Tyne at Hebburn. Within this range total PAH concentration in sediments from some Poole sites would classify as moderate. Again, concentrations in Holes Bay (Cobbs Quay marina) are elevated (1694µg kg^{-1}) relative to the outer Harbour (table 13).

Table 13. Concentrations of PAHs in sediment, Poole Harbour (adapted from Woodhead *et al.,* 1997)

Concentrations of PAHs in surface sediment (µg kg^{-1} dry wet)

	Holes Bay Cobbs Quay	Poole Harbour Stakes Buoy	English Channel	Threshold effect level* TEL	Probable effect level* PEL
Naphthalene	28	<17	<13	34.6	391
Acenaphthene	15	<4	<4	6.7	89
Fluorene	52	<15	<15	21.2	144
Phenanthrene	71	55	<11	86.7	544
Anthracene	20	16	<1	46.9	245
Fluoranthene	185	72	<4	113	1494
Pyrene	263	102	<2	153	1398
Benzo[a]anthracene	101	35	<2	74.8	693
Chrysene	90	21	<2	108	846
Benzo[e]pyrene	270	78	<3		
Benzo[b]fluoranthene	198	58	<3		
Benzo[k]fluoranthene	87	24	<1		
Benzo[a]pyrene	152	45	<2	88.8	763
Dibenzo[a,h]anthracene	12	78	<3		
Benzo[g,h,i]perylene	150	40	<5		
Total PAH	1694	624	nd	1684	16770

ND: no PAH compounds detected
* Thresholds from Macdonald *et al*., 1996

The PAH enrichment in Holes Bay sediment is probably consistent with an anthropogenic origin. The source of these PAHs may include airborne particulates derived from combustion products (close to the site of Poole power station, no longer in operation), urban run-off, various trade and domestic inputs, and marina activity.

Although PAH concentrations in the Poole SPA are not as high as those recorded in highly industrialised estuaries, the reported levels represent a potential contribution to the threat of harmful effects on benthic organisms in Holes Bay. As indicated above, carcinogenic and mutagenic intermediates and the formation of PAH-DNA adducts may arise in fish through the actions of cytochrome P450. Parts of this enzyme system (e.g ethoxyresorufin-O-deethylase – EROD – activity) can be induced in fish exposed to sediments containing ΣPAH concentrations between 1000-3000 µg kg^{-1}dry weight, and possibly lower (Payne *et al* 1988; Woodhead *et al.*, 1999) – i.e. within the range encountered in Holes Bay. Similar levels have also been shown to reduce growth of larval fish, presumably weakening their resistance to disease and other stressors (Misitano *et al.,* 1994). Less is known of the toxicity to invertebrates, though sub-lethal effects on crustacea, perhaps exacerbated by UV, have been observed at 1000-6000 µg kg^{-1}dry weight (Alden and Butt, 1987; Ankley., *et al* 1994).

Concentrations in Holes Bay are just above the Threshold Effect Level (ΣPAH) of 1684 µg kg^{-1} dry weight, proposed by Macdonald *et al.*, (1996) but below the Probable Effects Level of 16770 µg kg^{-1}dry weight. The same applies also for a number of individual PAHs (table 13). Any effects, if they occur, are likely to be chronic rather than acute. However, in view of the uncertainties over these guidelines, further work will be needed to assess the actual biological consequences of PAHs for this part of the marine site: the presence of elevated levels of other contaminants is a complicating feature here and demonstrates the need to integrate biological and chemical monitoring.

Exposure pathways to most biota probably consist of both water and dietary sources including sediments. Feeding habit and lifestyle will be significant factors in modifying the bioavailability of PAHs, as with most contaminants. Because of their stronger affinity for sediments, high molecular weight PAHs are thought to be accumulated more strongly by deposit-feeders, rather than by those types which process overlying waters.

Irrespective of assimilation pathway, PAHs, because of their affinity for organic matter, have a tendency to be accumulated by organisms. Set against this, the MFO system of certain taxonomic groups is capable of metabolism and elimination of some of the smaller molecules (though as indicated above this does not make the hosts immune to effects). Fish and polychaetes are among the most efficient PAH metabolisers (tissue concentration factors generally less than 500) whilst in crustacea and bivalves metabolism is slow, particularly for larger PAHs (concentration factors 3000-6000).

Bivalves are therefore probably better indicators of PAH distributions (though not necessarily biological effects) compared with fish, and are of particular relevance in Poole Harbour in view of its designation as a shellfish water. To date measurement of PAHs has been limited to mussels near Brownsea Island (table 14). Over the three years for which records are available there appears to be considerable variation in tissue concentrations of some compounds, notably naphthalene.

Table 14. PAHs in whole *Mytilus edulis* (µg kg^{-1} wet weight); Brownsea (EA data)

	15-Mar-99	**28-Feb-00**	**28-Mar-01**
Anthracene	1.32	1.80	2.15
Benz-[A]-Anthracene	2.58	5.90	12.83
Benzo-[A]-Pyrene	0.65	2.13	0.65
Benzo-[GHI]-Perylene	1.20	3.00	5.46
Chrysene	6.35	10.60	31.33
Fluoranthene	9.27	15.77	41.43
Indeno (1,2,3-CD)-Pyrene	0.25	1.70	2.88
Naphthalene	17.07	222.00	0.33
Phenanthrene	8.43	22.70	0.25
Pyrene	9.88	14.70	39.23
Total PAH	57.00	300.30	136.55

It is somewhat difficult to put these results in context as there are no UK standards for PAHs in shellfish, though the German Federal Health Department has a cautious guideline of 25 μg kg^{-1} in fish oil. A Canadian guideline value of 1-4 μg kg^{-1} (depending on the level of fish/shellfish consumption) has been established for benzo (a) pyrene, again based on human health concerns, rather than in an ecotoxicological context. Two out of the three Poole mussel samples in table 14 would meet these recommendations.

According to NOAA's mussel-watch programme in the United States, a ΣPAH concentration of 1100μg kg^{-1} dry weight in molluscs would classify as being high - equivalent to 220μg kg^{-1} wet weight (assuming a wet:dry weight ratio of 5). According to this criterion, the year 2000 sample from Poole would fall into the 'upper' category, whilst the other years are more comparable to average values encountered in the mussel watch programme. Clearly, more widespread monitoring of PAHs in Poole biota may be advisable.

5.1.4. Pesticides, Herbicides, PCBs, VOCs and Endocrine Disruption.

This section deals with pesticides, herbicides, PCBs and volatile organic compounds (VOCs) in the Poole SPA, together with any evidence of their involvement in endocrine disruption. Other potential endocrine disrupting chemicals, some of which have been described in detail above (e.g. metals and TBT), are also briefly included in this general discussion.

CEFAS monitors levels of selected pesticides and other potential endocrine disrupting chemicals in bivalves from UK designated shellfish waters. Analyses of cockles, mussels and oysters made between 1995 and 1996 imply that pesticide body burdens in most shellfish waters in England - including, presumably, Poole Harbour - were close to or below detection limits. Typically concentrations on a wet weight basis were; α,γ HCH - 0.003 mg kg^{-1}; dieldrin 0,001 - 0.007 mg kg^{-1}; DDT - 0.001-0.003 mg kg^{-1}. Maximum concentration of PCBs (Σ25 congeners) was 0.07 mg kg^{-1} wet wt (CEFAS, 2001). Concentrations of pesticides and PCBs in bivalves from UK designated shellfish areas are therefore considered of little toxicological importance. Body burdens in oysters, mussels and cockles fall well within guideline values for human consumers. Very little published data exists for biota in other parts of Poole SPA.

Some specific, if fragmentary, data on pesticide, PCB (and THC) concentrations - in Poole mussels, unspecified invertebrates and sediments - are listed in table 15. These data are extracted from WIMS (mussels), and from a preliminary evaluation of potential endocrine disrupting chemicals in English SACs and SPAs undertaken by CEFAS on behalf of English Nature (Allen *et al.,* 2000). Concentrations of most compounds are low. However, because the data are extremely sparse, both numerically and geographically, a comprehensive risk assessment for endocrine disruption is not possible and, consequently, Poole SPA has been identified by Allen *et al.* as a high priority for future research on this subject. The prioritisation is presumably based on a combination of factors, including the presence of large volumes of discharges entering Holes Bay, combined with a restricted mixing regime and hence perceived susceptibility to contamination by endocrine disrupting chemicals.

Table 15. PCBs, THC and pesticides in whole *Mytilus edulis* from Brownsea (EA data[1]) and unspecified sediments and invertebrates of the Poole SPA (Allen *et al.*, 2000[2]). Units are µg kg^{-1} wet weight for tissues and µg kg^{-1} dry weight for sediment

	[2]sediments	[2]'invertebrates'		[1]*Mytilus edulis* n=3	
	(n=6)	(n=2)	Mar-99	Feb-00	Mar-01
Aldrin			0.05	1.25	1.45
DDE (PP')		0.45	0.72	0.75	3.70
DDT (OP')		1.6	0.05	0.75	0.12
DDT (PP')			0.57	0.50	0.47
DDD		0.56			
Dieldrin		0.53	0.05	1.25	4.41
Endrin			0.05	1.25	0.33
HCH-alpha		0.35			
HCH-beta		0.54			
HCH-Gamma		0.35	0.13	1.07	0.63
Hexachlorobenzene (HCB)		0.3	0.14	0.50	0.25
Hexachlorobutadiene			0.13	0.50	0.15
Isodrin			0.05	1.25	2.23
THC		3800			
PCB NO.101			0.45	0.50	2.79
PCB NO.118			0.62	0.50	2.66
PCB NO.138			0.81	0.67	4.00
PCB NO.153			1.30	1.40	6.87
PCB NO.180			0.14	0.50	0.84
PCB NO.28			0.09	0.50	0.38
Total PCB*	345	1	3.39	4.07	17.6

* for mussels this is the sum of the congeners shown

On the basis of the limited 'evidence' in table 15, the levels of pesticides and PCBs do not appear to represent a serious threat to the SPA. Much more extensive, targeted sampling will be needed to test this thoroughly, and should incorporate samples from Holes Bay, where effects, if they occur, would be expected to be most prevalent. The discussion below describes some of the generalised *potential* threats from groups of known endocrine disrupting compounds, which may help to target future sampling.

Organochlorine pesticides (OCs)

OCs of relevance include agricultural pesticides dichlorodiphenyltrichloroethane (DDT) and its metabolites DDE and TDE; chlorinated cyclodiene insecticides such as aldrin, dieldrin and heptachlor (most widely used as seed dressings and soil insecticides); and hexachlorocyclohexanes (HCHs), such as lindane, (used against pests and parasites of farm animals and also in insecticidal seed dressings). Although most of these compounds have been banned in the UK, they may persist in environmental samples. Several organochlorine pesticides have been identified as endocrine disrupting substances (e.g dieldrin, aldrin, endrin, lindane, endosulfan, DDT and its metabolites). Many OCs are toxic List I contaminants.

Once in the environment OCs are persistent contaminants. They are stable and degrade very slowly, some taking 100 years to break down completely into harmless chlorides, whilst others do not degrade to any appreciable extent. Moreover when OCs do break down, the products are often more toxic and hazardous than the original substance. In general, these compounds have low water solubility and are therefore likely to sorb strongly to suspended solids and sediments. The majority of OCs are lipophilic, dissolving readily in fats, and tend to accumulate in the fatty tissues of living organisms. Invertebrate and fish species accumulate OCs in their tissues which can be transferred and magnified along the food chain, resulting in very high concentrations of OCs in upper trophic levels such as birds and marine mammals.

With the exception of isolated cases of exposure to concentrated compounds, the effects of OCs on marine life tend to be chronic rather than acute, with different OC compounds having similar effects and possibly acting synergistically (Leah *et al.,* 1997).

DDT

DDT and its residues interfere with calcium metabolism and were responsible for the well-documented phenomenon of eggshell thinning in sea and land birds during the 1960's when many eggs did not survive incubation, and a number of species were threatened with extinction. In general, environmental concentrations of the parent compound DDT are now lower than its metabolites and, like other organic substances, preferentially absorb onto sediments, particularly where these are fine-grained and/or contain a high proportion of organic carbon (Cole *et al*., 1999). No published papers on levels of DDT in sediments or water of Poole Harbour could be found. It is likely that the majority of pesticides present in the environment nowadays are predominantly at low levels in the water column, and that sediments, together with biota, form the major reservoirs for these compounds. Allen *et al*., (2000) report low DDT concentrations in 'invertebrates' of the SPA, as indicated in table 15 (note however, n= 2, and neither the site nor the species are described). EA mussel samples also contain low levels of DDT (table 15) - well below the recommended 'no-effects' guideline of 100 µg kg^{-1} set by ADRIS (Association of Directors and river Inspectors in Scotland) under the EU Shellfish Growing waters directive, to protect shellfish and their larvae (ADRIS, 1982). This suggests that the sampled 'invertebrates' are not at risk from DDT.

Dieldrin

Dieldrin, along with other 'drins' (aldrin, endrin, isodrin), is another endocrine-disrupting OC pesticide of potential concern in the SPA (Allen *et al.,* 2000). Dieldrin is highly toxic to fish and other aquatic animals and is said to be largely responsible for the dramatic decline of the otter population in the UK during the '50s and '60s. Dieldrin, used in sheep dips and seed dressings, leached into water systems and became concentrated in the fatty tissues of fish such as eels, which are a major component of the otter diet. The result was a dramatic decline, which reached its nadir, nationally, in the early 1970s, when otters were restricted to a handful of upland tributaries on the cleanest rivers[1].

Of the 'drins' group of pesticides, dieldrin tends to be the most common in biological samples, as other 'drins' tend to revert to dieldrin in the natural environment and are unlikely to be detected unless the organism has been recently exposed (MPMMG, 1998). Very little information on dieldrin concentrations in Poole Harbour could be found. Allen *et al*. (2000) report low concentrations in invertebrates of the SPA, as indicated also in the EA mussel data (table 15). Despite evidence of some temporal variation in mussels, all dieldrin concentrations are well below the recommended 'no-effects' guideline of 100 $\mu g\ kg^{-1}$ set by ADRIS, suggesting that the sampled invertebrates are not at risk.

γ-HCH (lindane)

Due to its toxicity and endocrine-disrupting effects, the use of lindane is currently being phased out in Europe following an EU decision in 2000 to ban it. However, its use on food crops (especially cocoa) imported from other counties results in lindane residues in sewage effluent. Lindane and dieldrin were amongst five pesticides listed as most frequently exceeding $0.1\mu g\ l^{-1}$ in estuaries and coastal waters of the south west during 1993 (NRA 1995), but in the current analysis of EA data, γ-HCH was below the limit of detection in more than 50% of water samples from the Poole SPA (see section 7.1.3). Little published information exists on concentrations of γ-HCH (or other HCH isomers) in Poole Harbour. Allen *et al*., (2000) report very low concentrations in 'invertebrates' of the SPA. This is also confirmed by EA mussel data (table 15): γ-HCH concentrations are well below the recommended 'no-effects' guideline of 30 $\mu g\ kg^{-1}$ set by ADRIS, suggesting that the sampled invertebrates are not at risk.

Endosulfan

Endosulfan is a mixture of two isomers, endosulfan a, and b, and is one of the few organochlorine pesticides which is still in use in the UK. It is a 'red list', and list II compound linked to fatal poisoning incidents in West Africa (Ton *et al.,* 2000). The high toxicity of endosulfan has led to its ban in many countries. Endosulfan has been identified as an endocrine disruptor, and is toxic to algae and invertebrates (particularly crustaceans) at concentrations above the EQS of $0.003\mu g\ l^{-1}$ (Cole *et al*., 1999). There is no published information available regarding endosulfan levels in the SPA though in the current evaluation of EA data, the majority of water samples were below limits of detection ($< 2ng\ l^{-1}$) and below the EQS (see section 7.1.3). Relatively small amounts of endosulfan are discharged by STW (annex 7). Greve and Wit (1971) found that 75% of the endosulfan in the River Rhine was associated with particulate matter (mud and silt), therefore sediment levels in the SPA may be more relevant, toxicologically. However, the ultimate fate of endosulfan in the marine environment is not known.

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[1] http://www.nfucountryside.org.uk/wildlife/home.htm 2002

Organophosphate pesticides (OPs)

OPs were first introduced for use in insecticides and fungicides in the 1950s, but remained second choice pesticides behind organochlorines until concerns over the environmental persistence of these compounds (notably DDT) began to surface in the 1970s. As the use of organochlorines tailed off, OPs succeeded them, and use in the UK increased during the mid 1980s. Throughout this time OPs became widely used both in livestock and arable farming. However, changes in the regulations on sheep dipping have meant that use of OPs in the livestock sector have declined in recent years. Overall, OPs now account for some 38% of total pesticide use globally, although the figure for western Europe is somewhat lower than this (~26%). Organophosphate (and carbamate) pesticides have the potential to exhibit neurotoxic activity at low concentrations. Zinkl *et al.* (1991) cite examples of median lethal concentrations of OPs (parathion and azinphos-methyl) to fish as low as 10µg l^{-1}. Sub-lethal effects on olfactory function in Atlantic salmon were observed after exposure to the OP diazinon at concentrations as low as 1 µg l^{-1}, and significantly reduced levels of reproductive steroids in mature male salmon parr resulted from exposure to 0.3µg l^{-1} diazinon (Moore and Waring, 1996).

Organophosphates enter the marine environment via spillage, industrial effluents, spray-drift and run-off from agricultural land. Several OPs are on list II water quality standards for the protection of marine life. Principal OP compounds which have been identified as of potential concern in the marine environment include; azinphos-methyl, malathion and fenitrothion (Cole *et al.,* 1999).

No information could be found in the literature for levels of individual OPs in the SPA. Similarly, the EA database contains no information on OPs in tidal waters, though freshwater samples from the Rivers Piddle and Frome have been analysed for azinphos (-ethyl and-methyl) fenitrothion, malathion and parathion (-ethyl and-methyl). Virtually all samples were below detection limits (see section 7). In earlier years these DLs were sometimes above EQS values, expressed as annual averages (10ng l^{-1} for azinphos-methyl and fenitrothion, 20ng l^{-1} for malathion). For the most recent samples however, detection limits have been well below the EQS indicating riverine inputs are unlikely to be of biological significance in Poole Harbour. Azinphos was withdrawn from the UK market between the late 1980s and early 1990s, whilst parathion is thought not to have been used extensively.

Simazine and Atrazine

The *s*-triazine family of herbicides to which atrazine and simazine belong have been used in large quantities (several hundred tons annually) in the UK to control weeds on croplands, roads and railways. Both atrazine and simazine are on the UK red list of toxic compounds with a combined EQS of 2µg l^{-1}. Though toxic, they are not accumulated significantly by organisms. They have also been identified as endocrine-disrupting substances (Allen *et al*., 2000).

Because of their major usage and high water solubility they are widespread in aquatic systems. In 1992 and 1993, elevated levels of atrazine and simazine were found in groundwater, freshwater and estuarine water of the southwest region. Since 1993, however, they have been banned from non-agricultural use: run-off from treated land should therefore be the main source to coastal waters. Unfortunately, no published information could be found relating to concentrations in Poole Harbour. EA data are limited to monitoring on the two major rivers entering the harbour, both of which comply with EQS in (section 7). There are no listed consented discharges from STW (annex 7). It seems unlikely, therefore, that triazine herbicides represent a significant threat to the marine site.

Polychlorinated Biphenyls - PCBs

PCBs have low water solubility and a high affinity for suspended solids, especially those with high organic carbon content. Therefore, in the aquatic environment, PCBs are usually found in much higher concentration in sediments, where they are amongst the most persistent of environmental contaminants. The sediment quality guideline value for (total) PCBs in sediments is 21.5μg kg^{-1} dw, and the probable effect level (PEL) is 189μg kg^{-1} dw (CCME 1999 see annex 5). The Poole sediment indicated in table 15 (from Allen *et al*. 2000) would exceed the PEL, though the precise location of this sample is not known.

Dredge spoils, including those from Poole Harbour, are monitored for PCBs before they are approved for disposal at sea (CEFAS, 2001). There appears to be considerable variability between samples (as also indicated for metals in section 5.1.1). Reported PCB values (medians ± SD for Σ25 congeners) for the Poole region in 1997 were 10.9μg kg^{-1} ± 8.4 μg kg^{-1} whilst in 1998 these were more than an order of magnitude higher, 345.4μg kg^{-1} ± 229.4μg kg^{-1}, suggesting relatively elevated concentrations are present in some parts of the system. The latter data (which exceed the proposed PEL level) would appear to be the same as those shown in table 15 (from Allen *et al*., 2000). The source of these PCBs cannot be ascertained from current discharge information. It should be stressed however that these samples do not necessarily reflect background concentrations for the region: dredge spoil samples often represent sediments from the most heavily impacted areas and might be expected to contain elevated levels of contaminants. More comprehensive data on spatial and temporal trends are needed to resolve the status of PCB contamination.

EA data on PCBs in sediments are rather scarce and variable. In 1996 all data for individual isomers were reported to be below detection limits. Expressed as Arochlor 1260, a commercial mixture, values ranged from below detection limits to 290 μg kg^{-1} (dry weight) at the mouth of the Frome; up to 740 μg kg^{-1} at the mouth of the Piddle; up to 817μg kg^{-1} at Wytch Farm; and up to 579 μg kg^{-1} at Holton Heath. These are not extreme values compared to industrialised estuaries, but again appear to exceed CCME guidelines.

Like the majority of organochlorine substances, PCBs are lipophilic, dissolving more readily in fats than in water; therefore they tend to accumulate in the fatty tissues of living organisms. The data for PCBs in table 15 indicate rather low levels in mussels from Brownsea. According to NOAA's mussel-watch programme in the United States, a ΣPCB concentration of 430 μg kg^{-1} dry weight in molluscs would classify as being high - equivalent to 86 μg kg^{-1} wet weight (assuming a wet:dry weight ratio of 5). This compares with a maximum of 18μg kg^{-1} wet wt in Brownsea mussels.

However, in view of the relatively high PCB levels in some Poole sediments, it is unfortunate that more data on infauna are not available to assess the issue of PCB bioaccumulation at a wider range of sites. Sediment-dwelling organisms are obviously the most vulnerable of estuarine biota in terms of direct exposure, and PCB's accumulated in the tissues of invertebrates can be transferred and magnified along the food chain resulting in high concentrations in upper trophic levels, particularly birds and mammals.

With the exception of isolated cases of exposure to concentrated compounds, the effects of PCBs on marine life tend to be chronic rather than acute. PCBs are implicated in endocrine disruption and linked to eggshell thinning and deformities in seabirds (Allen and Thompson 1996). Biomagnification of PCBs may result in impaired reproductive success in fish and seals (von Westernhagen *et al.,* 1981, Reijnders 1986), and also immunosuppression in seals (Brouwer *et al.,* 1989) which in turn has been linked to the phocine distemper epizootic of 1988 (Hall *et al.,* 1992). PCBs are also carcinogenic and on the red list of dangerous substances.

A number of volatile organic compounds (VOCs) are, potentially, endocrine disruptors, as well as being toxic directly. Several are discharged (in relatively small amounts) into Poole Harbour (see annex 7). These include:

Chloroform (trichloromethane)

Chloroform, a List I substance is an industrial solvent used in the UK in the production of fumigants and anaesthetics. It is also a principal transformation product of chlorine-based biocide products used in cooling water systems: chloroform is the major organohalogen formed at sites using freshwater sources, whereas bromoform predominates where water is taken from estuarine and marine sources (Cole *et al.,* 1999). By-products formed during chlorination of power plant cooling water may have adverse effects on the growth of marine invertebrates during their larval stages (Stewart *et al.,* 1979). It has been known for some time that reproductive tissues, especially sperm, and the immature stages of organisms are sensitive to very low concentrations of organohalogens (Davis and Middaugh, 1978).

There is no published information available regarding chloroform levels in the SPA, though in the current analysis of Poole water data (EA), the majority of samples were below limits of detection and below the EQS of 12µg l^{-1} (see section 7.1.7).

Carbon tetrachloride (Tetrachloromethane)

Carbon tetrachloride is a list I substance mostly produced for use in the manufacture of chlorofluorocarbons (CFCs). Carbon tetrachloride is also used as a chemical intermediate in the manufacture of pharmaceutical and pesticide products. Carbon tetrachloride production in the United Kingdom has recently ceased and its major use (production of CFC-11 and CFC-12) is now in decline. For UK marine waters, Willis *et al* (1994) reported levels to be between <0.1 – 44µg l^{-1}, with higher levels in source-dominated areas. Concentrations measured in the open ocean were generally much lower, at around 0.5 ng l^{-1}. There is no published information available regarding levels of carbon tetrachloride in the SPA though in the current review of EA data concentrations in the majority of water samples were below limits of detection and below the EQS of 12µg l^{-1} (see section 7.1.7).

Chlorinated Ethylenes (trichloroethylene, tetrachloroethylene [perchloroethylene])

These are List I substances produced in large quantities and widely used in industry in the production of food packaging, synthetic fibres and industrial solvents. MPMMG (1998) indicate that concentrations of chloroethylenes in UK coastal and estuarine waters appear unlikely to exceed relevant EQS (10µg l^{-1} annual average) derived for the protection of saltwater life. This appears to be true for Poole Harbour. Analysis of EA data on trichloroethylene indicates the majority of values were below limits of detection and below the EQS (see section 7.1.7).

Trichloroethane

Trichloroethane is a list II substance used as an industrial solvent. Potential sources of contamination include direct discharge of wastewaters, accidental spillages and deposition from the atmosphere. However, EA data indicates that the compound seldom occurs in measurable quantities in Poole water samples (see section 7.1.7). Concentrations are invariably well below the EQS of 100µg l^{-1} (annual average).

Metals and organometals

There are indications that a number of metals, notably Cd, Pb and Hg may cause endocrine disrupting effects. Experimentally, Cd (1 mg l^{-1}) has been shown to induce vitellogenin production in female fish, to increase the secretion of gonad-inhibiting hormones in fiddler crabs and, at 25µg l^{-1}, alter hormone titres in sea-stars (Thomas, 1989; Rodriguez *et al*., 2000, Besten *et al*., 1991). However, concentrations used to demonstrate these effects were substantially higher than those found in Poole Harbour.

Likewise for Pb and Hg, although it is known from mammals that ED action can occur at the level of the hypothalmic pituitary unit or on gonadal steroid biosynthesis, evidence of comparable activity on estuarine and marine biota, exposed chronically in the field, is not available (Allen *et al.,* 2000). Experimental studies on freshwater crayfish have, nevertheless, indicated that Cd and Hg at a concentration of 0.5µg g^{-1} body weight can arrest ovarian maturation due to inhibition of gonad stimulating hormone and serotonin, respectively (Reddy *et al*., 1997). In view of enhanced bioaccumulation of these and other metals at a number of sites (tables 6 – 9), similar reactions in marine crustaceans and other organisms from the SPA cannot be ruled out.

Chronic stress can lead to elevation of cortisol, following ACTH secretion in the pituitary. This is a normal adaptive response for mobilising the energy needed to deal with stress, and is not strictly-speaking, endocrine disruption. However, prolonged chronic stress can suppress the normal response, due to exhaustion of the pituitary-kidney feedback mechanism. In North America, metal-exposed sea trout (*Salmo* *trutta*) populations have been shown to exhibit symptoms of inhibition of the ACTH/cortisol response to acute stress (Norris *et al.,* 1999). Similar effects have been seen in catfish exposed experimentally to Hg (Kirubagaran and Joy, 1991). Possible knock-on effects on energy metabolism and salinity adaptation are likely. As yet, however, it is not known whether metal exposure in the Poole Harbour (Holes Bay) results in similar chronic effects.

One observation of note is that, in the past at least, Holes Bay contained locally elevated levels of Se. Concentrations up to 29µg l^{-1} and 1.5µg g^{-1} have been measured in water and sediments, respectively, resulting in bioaccumulation in invertebrates (Bryan and Langston 1992, and own unpublished results). Inputs were probably introduced from the local chemical industry (no longer manufacturing). The toxicology of Se is interesting in that there is a narrow margin between levels considered as essential for life and those viewed as harmful. It has been shown that where Se contamination exists, consumption of benthic organisms with elevated tissue burdens may result in elevated levels in tissues of predators, including birds (Ohlendorf and Fleming, 1988). Resultant reproductive effects and embryo deformities in migratory birds and waders nesting at Se-contaminated ponds in the US have been attributed to this metalloid (Ohlendorf *et al*., 1986). It may be that Se poses no serious problem in the SPA; nevertheless, in view of the ability of sediment-dwelling organisms to accumulate Se, additional studies on distribution, and the potential for reproductive abnormalities in birds, are warranted.

Among organometallic compounds, the androgenic impact of TBT on marine neogastropods is widely documented. Levels of TBT in Poole are certainly sufficient to result in effects (see section 5.1.2), though no specific studies have been undertaken in the SPA. The mud snail *Hydrobia ulvae* (an important dietary component of a number of important wildfowl) may well be susceptible to TBT in sediment.

Alkylphenols

Alkylphenolpolyethoxylates (APEs) are a major component in surfactants in detergents and their presence in sewage effluent and adjacent freshwaters has been associated with both ED and even acute toxicity in extreme cases. APEs have also been used in large amounts by the oil and gas industry as a component in rig washes (now being phased out), in paints and cosmetics, and as a spermicide (Blackburn *et al*., 1995). There is now a voluntary ban on their use in the UK domestic market.

In fresh water, concentrations up to 180µg l^{-1} of the degradation product nonylphenol have been measured near to STW (Blackburn and Waldock, 1995). There is a new proposed EQS for nonylphenol - 1µg l^{-1} (Annual Average) and 2.5µg l^{-1} (Maximum Allowable concentration) (EA *pers comm*). With the exception of industrial estuaries, such as the Mersey and Tees (5 µg l^{-1} nonylphenol), concentrations of alkylphenols in tidal waters are generally much lower. As indicated in table 16, the majority of nonylphenol species in samples from Poole Harbour were below detection limits and below thresholds where estrogenic or toxic effects might be anticipated (20µg l^{-1}; Blackburn and Waldock, 1995). Even in Holes Bay concentrations were only slightly above detection limits.

Though these substances are relatively lipophilic and will be sequestered by particulates, concentrations in Poole sediments were also generally below detection limits (table 16, from Blackburn *et al.*, 1999). This implies alkylphenols pose no significant problem for most of the SPA. As yet there is no published information on levels of alkylphenols in Poole biota.

Table 16. Alkylphenols in Poole Harbour water and sediments. (Adapted from Blackburn and Waldock, 1995; Blackburn, Kirby and Waldock, 1999).

site	date	Total extractable nonylphenol (µg/l)	Dissolved nonylphenol (µg/l)	Total extractable [NPEO + NP2EO] (µg/l)	Dissolved [NPEO+NP2EO] (µg/l)
			water		
Anchor	July 1995	<0.2	<0.2	<0.6	<0.6
No.42 buoy	July 1995	<0.2	<0.2	<0.6	<0.6
Stakes buoy	July 1995	<0.2	<0.2	<0.6	<0.6
Holes Bay	July 1995	0.1	0.1	<0.6	<0.6
Hales buoy	Oct-93	<0.8	ns		
Brownsea buoy	Oct-93	<0.8	ns		
Poole Anchor	Oct-93	<0.8	ns		
		sediment (all µg g^{-1} dry weight)			
Poole Anchor	July 1995	<0.1		<0.5	
No 42 Buoy	July 1995	<0.1		<0.5	
No 55 Buoy	July 1995	<0.1		<0.5	
Holes Bay	July 1995	<0.1		<0.5	

NPEO nonylphenol monoethoxylate
NP2EO nonylphenol diethoxylate
ns- not sampled

In summary, monitoring data for endocrine disrupting chemicals are very limited, but sources in Poole Harbour, particularly Holes Bay, are potentially important for some. TBT, hydrocarbons, PCBs and metals may well be toxicologically significant, especially if their influence is additive. Therefore given the cocktail of contaminants in sediments in parts of Holes Bay, and the added inputs from discharges (annex 7), it is possible that non-specific adrenocorticoid effects are occurring in sensitive species (i.e. through the exhaustion of the general cortisol stress response). The combined pressure caused by this mixture of contaminants could also be contributing to other, more specific, forms of endocrine disruption.

Further study is needed to evaluate current risks to both invertebrates and higher organisms. Food items of various sorts will be the most likely route of exposure to birds, many of which are extremely sensitive to ED substances such as pesticides, PCBs, PAHs and metals (reviewed by Fry, 1995). Consumption of prey species such as molluscs (which are excellent bioaccumulators of contaminants), or accidental ingestion of sediment by waders feeding on mud-flats, represent unquantified risks within the SPA (Allen *et al.,* 2000, CSL, 2000). This has led Allen *et al* (2000) to designate Poole Harbour a high priority for ED research.

5.1.5 Radionuclides

The main source of anthropogenic radionuclides in sediments of Poole Harbour, other than atmospheric deposition (Cs, Pu, Am, Sr, I etc.), has been waste discharge via a long outfall into Weymouth Bay from the prototype steam generating heavy water reactor at Winfrith. Authorised discharges of treated effluent containing tritium, principally, but also some metallic activation products such as ^{60}Co and ^{65}Zn (more important radiologically) were made to sea between 1970 and 1990, when the reactor was closed. Discharges of low level liquid waste have continued at reduced levels and environmental radionuclides are now monitored under the auspices of FSA and SEPA (e.g. RIFE-6, 2001). Shellfish are sampled in Poole Harbour, as they represent the most likely pathway to humans. As indicated in table 17 and figure 11 concentrations of radionuclides are present at extremely low levels in these samples and appear to have been declining in recent years; the radiation dose is currently less than 0.5% of the allowable dose limit. A decline in radionuclide levels in environmental samples, following reactor closure, has also been reported by Cundy *et al.* (1999). Nevertheless, measurements in Poole Harbour show that, despite the extensive dilution in the sea off the Dorset coastline, transport and re-concentration in the SPA has occurred in the past and may continue, albeit at very reduced levels. ^{65}Zn is relatively soluble and some dispersal in dissolved form is to be expected. For ^{60}Co, dispersal in a mixture of dissolved and complexed forms is likely (Leonard *et al.,* 1993).

^{60}Co, transported eastwards along the Dorset coast, has been shown to be deposited in sediments within Poole Harbour (Cundy and Croudace, 1995). Particle size fractionation indicates that, at a sandier site close to the harbour mouth, most of the ^{60}Co is associated with fines (<8μm). At Wytch Farm (mainly clay-rich) ^{60}Co is more widely distributed across size fractions though some enrichment is seen in the <8μm fraction (Cundy and Croudace, 1995). Distributions of ^{60}Co appear to correlate well with Fe and Mn oxyhydroxides, indicating adsorption onto these coatings. However, there is also some association with organic content in coarser fractions, particularly in sediments close to the Harbour mouth, perhaps representing debris from adjacent salt marsh.

Sequential extractions suggest that a significant fraction of the ^{60}Co in Poole sediments may be labile, and has potential for remobilisation. Therefore, sediments may not provide a permanent sink for radionuclides and other stable elements.

Table 17. Radioactivity in shellfish, Poole Harbour and Weymouth Bay, 2000 (from RIFE, 2001)

Material	**Location**	**Mean radioactivity concentration (wet) Bq kg^{-1}**						
		^{60}Co	^{65}Zn	^{137}Cs	^{238}Pu	^{239}Pu+ ^{240}Pu	^{241}Am	^{243}Cm+ ^{244}Cm
Pacific oysters	Poole	0.06	<0.06	<0.02			<0.06	
cockles	Poole	0.8	<0.12	0.08			<0.23	
whelks	Poole	0.42	<0.18	<0.06	0.0029	0.01	0.0024	0.00002
whelks	Weymouth	0.23	<0.12	<0.04			<0.05	

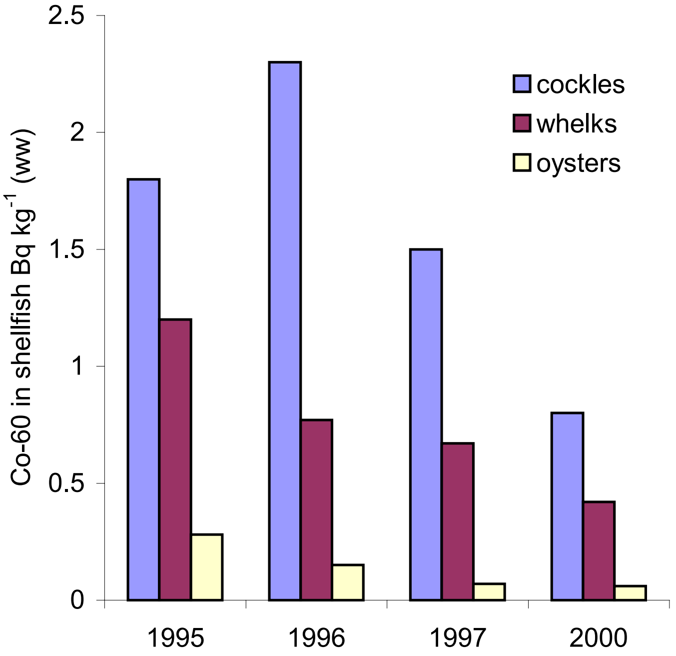

Figure 11. ^{60}Co concentrations in shellfish, Poole Harbour 1995-2000 (data source: RIFE)

^{55}Fe has also been detected in sediments, macroalgae and molluscs (cockles, oysters, whelks and winkles) of Poole Harbour and the adjacent coastline. However, the present dose in shellfish which is attributable to the ^{55}Fe discharge from Winfrith is probably no greater in scale than that arising from atmospheric fallout, and is not of significance, radiologically (Warwick *et al*., 2001). As with ^{60}Co and ^{65}Zn, levels of ^{55}Fe in the discharge have dropped dramatically since 1990 and this is reflected in most biota. Half-lives for the majority of invertebrates monitored are of the order of one year.

5.2 Non-Toxic Contaminants

This section deals with non-toxic contamination in Poole Harbour. Concentrations of non-toxic substances are an important issue in marine sites although they do not appear on priority lists. Areas of concern, identified by the nature conservation agencies include: nutrients (nitrogen, phosphorus and silicon), organic carbon, oxygen depleting substances (BOD and COD), pH, salinity, temperature (thermal discharges) and turbidity (Cole *et al*., 1999).

5.2.1 Nutrients

Water quality with regard to nutrients is primarily assessed in terms of the trophic status, or degree of nutrient enrichment in estuaries and near shore waters. 'Nutrient enrichment' generally refers to nitrogen and phosphorus species which are elevated beyond background levels, as these are the two leading causes of poor water quality.

Nitrogen and phosphorus enter the estuarine environment via point or diffuse sources. Point sources are generally consented discharges and a direct result of human activities including; sewage effluent from sewage treatment works (STW), discharges from some industrial processes (including detergents) and cage fish farm installations. Diffuse inputs originate from both natural and anthropogenic sources. These comprise run-off/leaching from the land catchment (either directly into estuaries and coastal waters or via rivers and groundwater), atmospheric deposition, imports from off-shore waters and nitrogen fixation by plant life.

In a tidal estuary, or a semi-enclosed lagoon such as Poole Harbour, point source inputs may be of greater importance. The potential for nutrient enrichment and localised effects will be determined by physico-chemical and biological characteristics of the estuary such as flow, seasonal variability, flushing, tidal regime, primary production and rates of remineralisation.

The principal effect of extreme nutrient enrichment is eutrophication, defined as 'the enrichment of natural waters by inorganic plant nutrients, which results in the stimulation of an array of symptomatic changes' (EA, 1998). These changes include an increase in phytoplankton growth that is reflected by an increase in chlorophyll α concentrations. Dissolved oxygen concentrations in the water column fluctuate during the growth phase of a phytoplankton bloom and there is a potential for depletion of dissolved oxygen concentrations in the water column and sediments as a result of microbial activity following the die-off of phytoplankton blooms. pH may be affected. The bloom may contribute to increased turbidity in the water column reducing light availability.

Nutrient enrichment can also result in proliferation of the larger macroalgae. In comparison to effects caused by phytoplankton blooms, macroalgae are considered harmful due to dense overgrowth that can occur in localized areas, or coastal embayments receiving excessive nutrient loading (Valiella *et al*., 1997). Accumulations can be so high as to cover the bottom of a region, excluding other biota as well as creating an environment in which high oxygen consumption and the associated anoxic conditions accompany decomposition of the accumulated or displaced biomass (Raffaelli *et al,* 1998). These problems can be compounded by ammonia release from sediment; Owens and Stewart (1983) showed that ammonia release from sediment is greatest when macroalgae biomass declines, due to increased ammonifying microbial activity.

Some of these changes are quantifiable and, in addition to nitrogen, phosphorus and ammonia, a range of other parameters can be measured for determination of water quality in relation to nutrients. These include dissolved oxygen (DO), biological oxygen demand (BOD), chlorophyll *a*, suspended solids and turbidity. Nitrogen levels can be monitored as nitrate, nitrite and ammonium concentrations in tidal waters which, when added together, produce total inorganic nitrogen (TIN), an approximation of bioavailable nitrogen. Phosphorus is present in the aquatic environment in both inorganic and organic forms, although the principal inorganic form is orthophosphate and is measured as dissolved orthophosphate (soluble reactive phosphate SRP), or as total reactive phosphate (TRP) by measuring phosphate in unfiltered samples.

Parr *et al* (1999) report a wide range of nutrient levels in UK coastal waters and estuaries[1]; concentrations of 0.07 – 1.85mg l^{-1} TIN and 0.007 – 0.165mg l^{-1} TRP are found in coastal waters, whilst the upper reaches of estuaries have nitrogen concentrations similar to those in river water, 0.1 - 15mg l^{-1} TIN. TRP in upper estuaries, as in rivers can also be variable, 0 – 11.4mg l^{-1}.

Poole Harbour is designated as a Sensitive Area (Eutrophic) & "Polluted Waters" (Eutrophic) and nutrient-related problems have been manifest for several decades) (EA, 2001). During the late 1960's, the growth of green algae in Holes Bay was identified as a problem when the cooling water intake screens of the Power Station were blocked, and there were complaints locally about the smell of rotting seaweed. Investigations showed that the growth of *Enteromorpha* and *Ulva* were stimulated by high levels of both ammonia and nitrate. Ammonia was taken up by the macroalgae although nitrate was utilized almost as effectively. Significant reductions in the concentrations of nitrate in Holes Bay were recommended.

Aerial photographs taken in 1980, and in follow-up surveys, showed that the problem had spread and there were substantial growths of macroalgae in the Southeast of the Harbour (south of Brownsea Island) in addition to those of Holes Bay and Blue Lagoon. Detailed surveys and nutrient budgets were carried out in an attempt to address the problem.

The reports (Wessex Water Authority, 1981) included estimates of the nutrient input to Poole Harbour over 3-4 days. Figure 12 shows nutrient loads from major sources in the Harbour, plotted from estimated data. The calculated loads are acknowledged to be approximate but give an indication of the relative load from each source. The major source of inorganic nitrogen appeared to be rivers, particularly the Frome, and the load from this source responded directly to increased river flow (figure 12a). Following a period of rain, the concentration of oxidized nitrogen in the rivers increased slightly. A sizeable N input from Poole STW is also evident.

The phosphate load was considered relatively stable, and the majority of phosphate was derived from sewage effluents (Poole, Wareham and Dorchester) with land run-off contributing very little (figure 12b). In contrast to N loads, phosphate loads decreased slightly during the period of high flows. Phosphate concentrations in tidal waters reached a peak at low water. Elevated phosphate levels were observed in one of the surveys which may have be due to remobilisation of phosphates from sediments following spring tides.

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[1] Note: It is generally assumed that an N:P ratio of 10:1 is ideal for plant growth. At N:P ratios >10:1 (mainly in freshwater), P is thought to be limiting, and at N:P ratios < 10:1 (mainly in seawater) N is thought to be limiting. However there are 3 large coastal areas in the UK where P may be limiting – from the Solent to Dartmouth; around the Severn from Padstow to Oxwich, and from the Humber to Essex (Parr *et al,,* 1999).

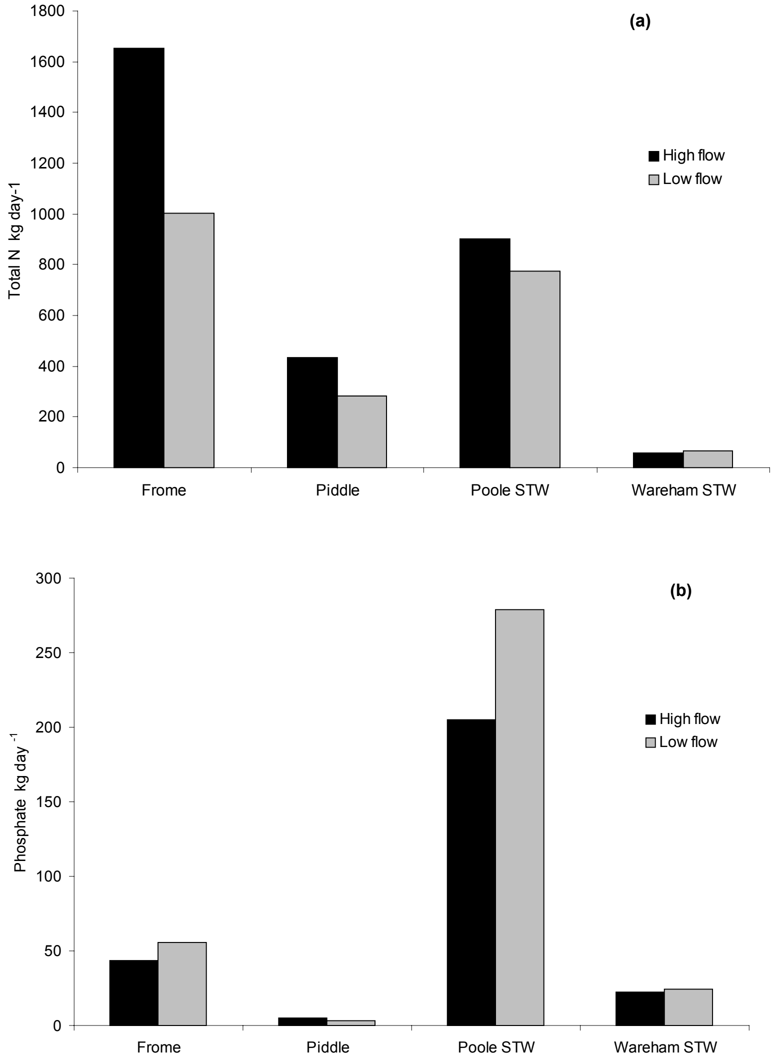

Figure 12. Estimated loads of nutrients from major sources in Poole Harbour under high and low flow conditions. (a) Total N (b) Phosphate. (Plotted from data in WWA, 1981)

Poole sewage works was also considered to be the largest ammonia source in the Harbour, and the ammonia concentration in waters reached a peak at low water.

Principal findings from follow-up surveys (Shucksmith, 1982) are as follows:

- Total N input to Poole Harbour falls (5000 to 1800kg day^{-1}) between April and September due to reduced river flow.
- N input from Poole STW is relatively constant (900kg day^{-1}) April – September and is therefore relatively more significant in September.
- Ammoniacal N (NH_3 as N) concentrations at Poole Bridge (low water) were significantly higher than in the main harbour due to inputs from Poole STW (esp Poole Eastern Works, where concentrations in the final effluent of >25mg l^{-1} were frequently observed)
- 80% (~280kg day^{-1}) of the total input of phosphate to the Harbour is from Poole STW – hence maximum PO_4 concentrations occur in Holes Bay.

Falconer (1983) applied a numerical model to Poole Harbour and Holes Bay to simulate the existing total N levels in the Harbour arising from a number of inputs, and hence predict the corresponding nitrogen levels as a result of removing the input from Poole STW. Results from this study suggested that the nutrient levels in Poole Harbour itself would not be reduced to any great extent by the removal of the N input, but that N concentrations in Holes Bay could be reduced significantly. Falconer subsequently developed and modified the numerical model (Falconer, 1986). Simulations resulted in a similar outcome:

- N concentrations predicted for April simulations suggested that removal of nitrogen input from Poole STW would result in the mean N concentration being reduced by ~10% for both the whole, and lower harbour.
- September simulations for nitrogen removal resulted in overall reductions of about 10% for TON removal only, or 25% for removal of TON and NH_3.
- Findings again indicated that N removal from Poole STW discharge would not have a dramatic effect on N concentrations in the outer Harbour, but would reduce concentrations in Holes Bay by 50% or more.

Recent export coefficient modelling (Hanrahan *et al.,* 2001) calculated the total phosphorus loading in the Frome catchment area and predicted that diffuse sources (land use, animals, and septic tanks) made the most significant contribution to the total load (65%) with 35% coming from STWs. Application of the model to investigate the potential effect of EU Directives on improved P extraction at STWs indicated that, for the Frome catchment, reducing the P levels in discharged waste waters to 2 mg l^{-1} would reduce total P loading to the catchment by at least 15%. (see section 9 - models).

During the 1980's, eutrophication became less of an issue because several of the most noticeable effects had disappeared; Poole power station went onto intermittent production during the 1970's and was finally closed down in early 1980, and intertidal areas of Sterte Bay, within Holes Bay, where algal growth and anoxic sediments were particularly evident, were infilled as part of a road building development (Dyrynda, 1987). However, recent aerial photography (1998) indicates that there is still a problem in Poole Harbour with macroalgal blooms, due to excessive nutrient inputs: eutrophication is manifested in increased growth of algae (*Enteromorpha* and *Ulva*) on intertidal mudflats. Detailed investigations using a higher resolution sensor, Compact Airborne Spectrographic Imager (CASI) were used to identify areas of dense algal growth, which are evident at low water (figure 13). Principal areas of this dense growth are Blue Lagoon, Holes Bay, Lytchett Bay, Keysworth Point, and Ower and Brands Bays (S of Brownsea Island).

Nutrient concentrations need not be elevated by a large margin before algal proliferation commences (Parr, 1999). Hypernutrification in seawater may occur when maximum winter TIN[1] values exceed 0.144mg l^{-1} (provided P>0.006mg l^{-1}), (see section 7.2.1). The Agency report average (winter) TIN concentrations of 1.36mg l^{-1} in the east Harbour area, 5.28mg l^{-1} in Wareham Channel and 2.89mg l^{-1} in Holes Bay, far in excess of these guidance values (EA, 1997). Poole STW was reported to be the major influence on TIN in Holes Bay, and the riverine inputs of the Frome and Piddle the major influences on TIN concentrations in the Wareham Channel.

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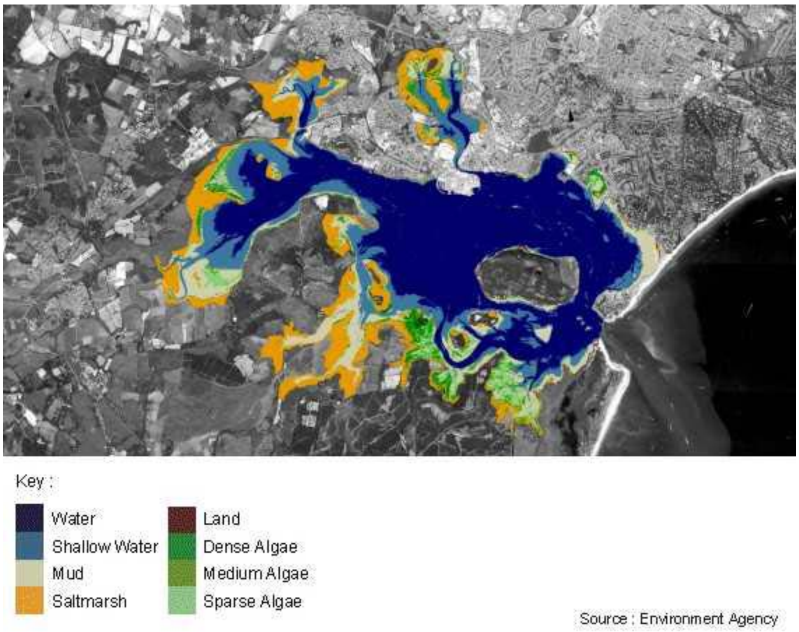
[1] (TIN is an approximation of bioavailable nitrogen)



Source : Environment Agency

Figure 13. CASI (Compact Airborne Spectrographic Imager) imagery of algal growth on intertidal mudflats, Poole Harbour, August 1998 (www.environment-agency.gov.uk)

A recent Agency report has summarised the eutrophication problem in Poole Harbour (EA, 2001). Key findings are:

- Chemical and biological data collected over the period 1997-2000 indicated that the Harbour, and in particular Holes Bay, is eutrophic, or at risk of becoming eutrophic.
- Winter DAIN levels show that hypernutrification occurs, with the overall highest levels recorded at Wareham Channel and at the entrance to Holes Bay (figure 14a).
- 95% of winter samples (n=80), and 71% of summer samples (n=120) for DAIN in tidal waters exceeded CSTT[1] standards (12 mmol m^3 DAIN in the presence of 0.2mmol m^3 DAIP).
- Nutrient load analysis indicates that Poole STW and the Rivers Frome and Piddle are the major nutrient inputs to the Harbour.
- The results of modelling are reiterated: nutrient removal at Poole STW would reduce nutrient levels in Holes Bay by 50% on average, and by 62% in the summer (Murdoch and Randall, 2001, see section 9).
- There has been a significant increase in the (intertidal) area of Holes Bay covered by macroalgae since 1995 – this is supported by several complaints by concerned public, and the problem has been raised at Government level.
- It is unclear whether there has been an increase in macroalgal cover for the rest of the Harbour although there is evidence of dense algal mats in sheltered areas.
- Shellfish mortalities 1995 – 1997 have been linked to excessive macroalgal growth.
- The shellfishery was closed twice during 2000 due to ASP and DSP toxin contamination – this was said to demonstrate that eutrophication has resulted in an 'undesirable disturbance' which brought the industry to the brink of closure.

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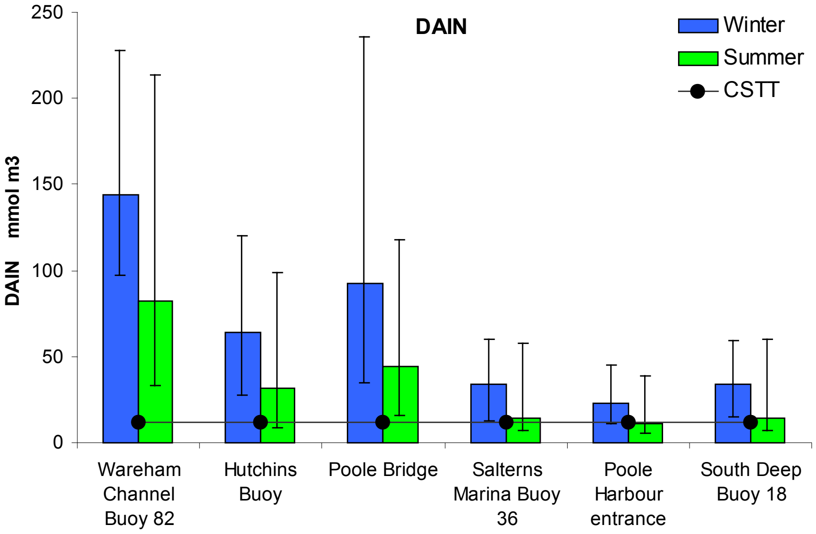
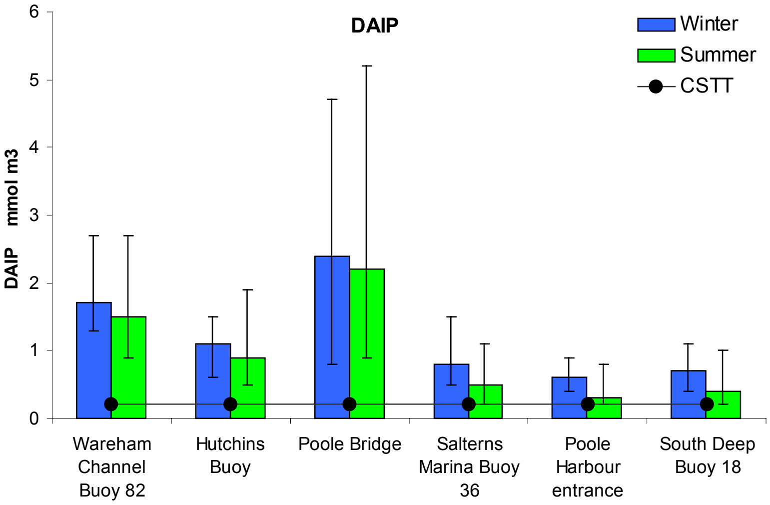
[1] CSTT: Comprehensive Studies Team of the Marine Pollution Monitoring Management Group (CSST, 1997)

(a)



(b)



CSTT standard 12 mmol m^3 DAIN in the presence of 0.2mmol m^3 DAIP (CSTT, 1997)

Figure 14. Mean values for (a) DAIN and (b) DAIP (mmol m^3) in winter (October to March) and summer (April to September) for tidal waters of Poole Harbour 1997 – 2000. Error bars show min and max concentrations (from EA, 2001).

The Agency report recommended that the Harbour be designated as a Sensitive Area (Eutrophic) (SAE) and a Polluted Waters (Eutrophic) (PWE) in order to provide the framework which will address the problem and reduce both point and diffuse source nutrient inputs to the Harbour (EA, 2001)[1].

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[1] If a site is designated as a Sensitive Area (Eutrophic) under the Nitrates Directive, the objective is to "reduce water pollution caused or induced by nitrates from agricultural sources" and "prevent further such pollution". Therefore, improvements to STW are not an automatic result of designation. STW improvments may be instigated if the STW is considered to be an indirect nutrient source (ie, discharges upstream of the designated site) and serves a population greater than 10000 population equivalent.

At the time of the EA report, in January 2001, principal discharges (Poole, Wareham and Dorchester STWs) were subject to secondary treatment. Water company improvements are planned under AMP3 to uprate secondary treatment at Poole, Wareham and Lytchett Minster STWs, by the addition of UV disinfection by 2003 (see annex 8). However, this is to satisfy the Bathing Waters, and Shellfish Waters Directive and will not alleviate the eutrophication problem. Future improvements under AMP 4 (2005-2010) are expected to include tertiary treatment and to concentrate on nutrient removal due to the SA(E) and PW(E) designation. The area is also designated a nutrient vulnerable zone (NVZ) therefore diffuse sources of N will be addressed by measures such as defining the amount of N applied to land in the catchment (P. Jonas, *pers comm.*).

5.2.2 Turbidity

There is little information in the literature regarding turbidity in Poole Harbour. Poole Harbour Commissioners (PHC) suggest that the natural turbidity of the water is often quite high due to the hydrodynamic regime in the harbour. However, suspended sediment readings carried out by PHC in the harbour entrance and Wareham Channel indicated low sediment loads throughout the water column, except seawards of Russel Quay where concentrations of suspended solids reached 60 – 90mg kg^{-1} in small areas (Howard and Moore, 1988). No further references to turbidity in the Harbour are available.

Regular maintenance dredging is carried out in a number of areas by, or on behalf of PHC, to ensure that the channels remain navigable and that there is no build up of material alongside the quays or in marinas. Dredged material varies depending upon its location within the harbour. The majority of the harbour consists of fine sands and silt, but around the entrance where currents are strong, all fine material is removed leaving a mobile pebble structure. Outside the harbour mouth in the Swash Channel leading out through the offshore sandbanks, deposits of coarse sand collect. Previously large volumes dredged from outside the harbour mouth have been used for beach replenishment at Bournemouth. The port attempts to minimise dredging activity both for practical and environmental reasons. Mobilisation of the sediments during the dredging operation can have physical and chemical impact on habitats although this is not considered by PHC to be a great problem. Mobile fish populations are unlikely to be significantly affected, whereas sessile invertebrates may be more vulnerable. Mussel cultivation is an established feature of Poole Harbour although most of the harbour's shellfish sites are not in close proximity to the areas where dredging occurs[1].

5.2.3 Chlorophyll *a* and micro-algae

Elevated levels of chlorophyll *a* are one of the primary symptoms of increased nutrient inputs to estuarine waters and as such are another response variable measurement, although it is important to distinguish between natural blooms and those induced by "artificial" causes. It is pronounced or persistent blooms which cause concern. Chlorophyll *a* is the molecule mediating photosynthesis in almost all

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[1] http://www.phc.co.uk/Environment.doc.

green plants, including phytoplankton. Rapid proliferation or blooms of phytoplankton, as reflected in elevated chlorophyll *a* levels, can occur throughout the ocean but are typically associated with temperate coastal and estuarine waters, notably in restricted exchange environments such as Poole Harbour. During winter months. growth of phytoplankton populations are at a minimum because of reduced temperature, light availability, and water column stability, and chlorophyll-*a* levels generally remain low. Monitoring of chlorophyll *a* is undertaken more often in spring and summer months when estuarine concentrations may exceed 50-80µg l^{-1}, under optimum growing conditions (Monbet 1992).

In the UK, the indicator (mean) value for suspected eutrophic conditions is set at 10µg l^{-1} chlorophyll *a* (Dong *et al.,* 2000; also the DoE criterion, (EA, 2001)). Values greater than this, coupled with cell densities of 5 x 10^5 cells l^{-1} (500 cells ml^{-1}) are considered indicative of phytoplankton blooms (EA, 1997).

There is little information in the literature regarding chlorophyll *a* in tidal waters of Poole Harbour. EA (1997) report that winter concentrations were generally low in 1995, with a maximum of 4µg l^{-1} at Wareham Channel. In January 1996, higher levels were recorded, up to 16µg l^{-1} in the Wareham Channel, with elevated levels at Lake Pier, Rockley Point and South Deep. These higher winter levels indicated a winter bloom, which is unusual unless potentially eutrophic conditions exist. Summer values for chlorophyll *a* exceeded 10µg l^{-1}at many sites during 1995 and 1996. Highest concentrations were recorded in Holes Bay and Wareham Channel (maximum 79, and 45µg l^{-1} respectively) in August, 1996.

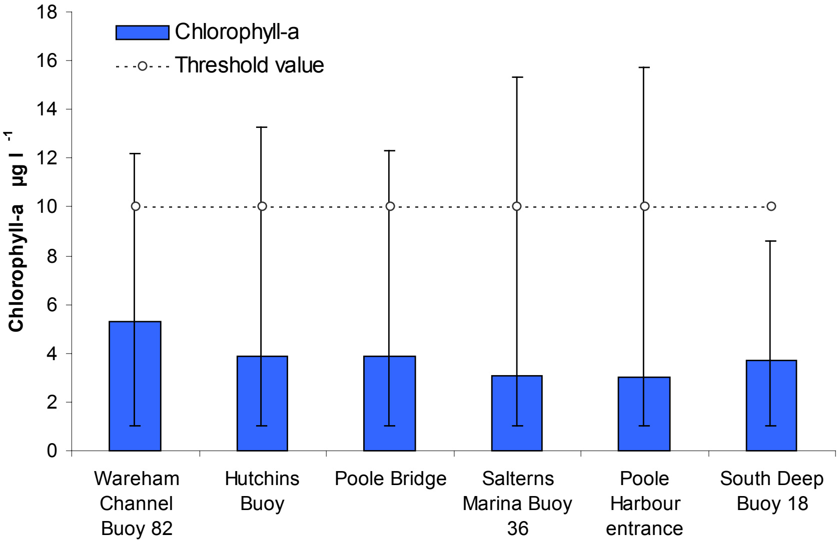

Figure 15. Chlorophyll *a* in tidal waters of Poole Harbour (means 1998-2000; error bars represent min and max concentrations). (From EA, 2001)

In a brief account of chlorophyll *a* distributions in Poole Harbour between 1998-2000, the Agency indicate that there were no exceptional micro-algal blooms, and no observations of scums or foams, although the shellfishery was closed twice during 2000 due to ASP and DSP toxin contamination (EA, 2001). ASP (Amnesic Shellfish Poisoning) toxins associated with *Pseudo-nitzchia* spp., were detected in *Ensis* (razor clams) and *Pecten* (scallops) at Ramshorn shellfish bed during March 2000. At the same site in August 2000, *Mytilus* (mussels) were found to be DSP (Diarrhetic Shellfish Poisoning) positive. DSP toxins are associated with Dinoflagellate species.

Between 1998 and 2000, mean concentrations of chlorophyll *a* in the main body of the Harbour (excluding secondary embayments) were reported to decrease with distance from Wareham Channel and with increasing salinity (EA, 2001). However, maximum concentrations in individual samples were recorded at the Harbour entrance and Salterns Marina (15.7 and 15.3µg l^{-1}, respectively) (figure 15). Approximately 7% of all samples (n=106) exceeded the 10µg l^{-1} threshold value.

5.2.4 Dissolved Oxygen

The Agency report continuous monitoring of DO (% saturation) at three sites (Wareham Channel, Hutchins Buoy and Harbour Entrance) in the Main Harbour during summer 1995 (EA, 1997). Concentrations showed significant diurnal variation which was particularly pronounced in the Wareham Channel and, to a lesser extent, at Hutchins Buoy. This pattern suggested that algal activity had a considerable influence on DO levels, artificially boosting DO during the hours of daylight and resulting in O_2 supersaturation of the water column, whilst at night severe depletion of DO occurred due to respiration.

The diurnal variation continued throughout the last week of June but became less exaggerated during early July, when % saturation generally dropped. This was particularly noticeable at Hutchins Buoy, where DO levels 'tailed off', dropping to almost 0% saturation on July 5th, compared to a minimum of ~60% at Wareham Channel and ~80% at the Harbour entrance. These fluctuations can cause problems for fish and invertebrate communities. During bloom die-offs (macro- and micro-algae) microbial decomposition of algal cells may lead to an increase in oxygen demand and acute DO depletion, which can result in lethal and sub-lethal effects to fish and invertebrate communities. The Agency report the retrieval of 14 dead salmon from the tidal reaches of the River Piddle in August 1994. The cause of these deaths was not positively established but high water temperature and low levels of DO were suspected (EA, 1997).

Parts of Poole Harbour may be especially vulnerable to oxygen depletion, as these changes in water quality are likely to be greatest in semi-enclosed bodies of water with long retention times, and where stratification of the water column occurs (Cole *et al.,* 1999). Reduced DO levels could therefore be anticipated in some areas of the marine site, such as the secondary embayments of Holes Bay and Blue Lagoon, where nutrient levels are shown to be high.

6. STUDIES ON BIOLOGICAL COMMUNITIES

The Poole Harbour Subtidal surveys (Dyrynda, 1984, 1987) provide useful background information on the SPA, including a full species list for the subtidal environment in the harbour, whist the Report on the sediment shores of Dorset, Hampshire and the Isle of Wight (Holme and Bishop, 1980) gives a comprehensive site-by-site species guide to the distribution of intertidal organisms. Further information on studies relating to key organisms, or groups of organisms are reviewed below.

Saltmarsh

Poole Harbour and Holes Bay support extensive *Spartina* marshes for which there is a history of investigation, e.g Stapf, (1914) to Raybould, (2000). Prior to 1870, the small cord-grass S. *maritima* was the only *Spartina* species in Great Britain. It was never common, but formed a significant community of the low marsh around the Wash, Thames basin and the Solent. Accidental introduction of *S. alterniflora*, a native of America, to Southampton Water and its subsequent crossing with the native plant resulted in the appearance of a fertile common cord-grass *S. anglica*, which is now the most widespread *Spartina* species in the UK (Davidson *et al.,* 1991).

Reasons for its success are numerous, and include a suite of biological properties related to its hybrid origin, and notably, its occupancy of a formerly vacant niche on intertidal mudflats to seaward of the previous limit of perennial vegetation (Gray, 1986). The earliest record of *Spartina* in Poole Harbour is given as 1899 (Oliver, 1925), *Spartina* reached its zenith in the Harbour during the 1920s and then went into sharp decline. Studies indicate that the lower limit of *Spartina* has gradually retreated upward on the shore resulting in an overall extensive reduction in *Spartina* marsh area in the Harbour from a maximum of 775ha in 1925 to around 415ha in 1980. Recent figures are available for Holes Bay only and show that *Spartina* is still being lost (figure 16)

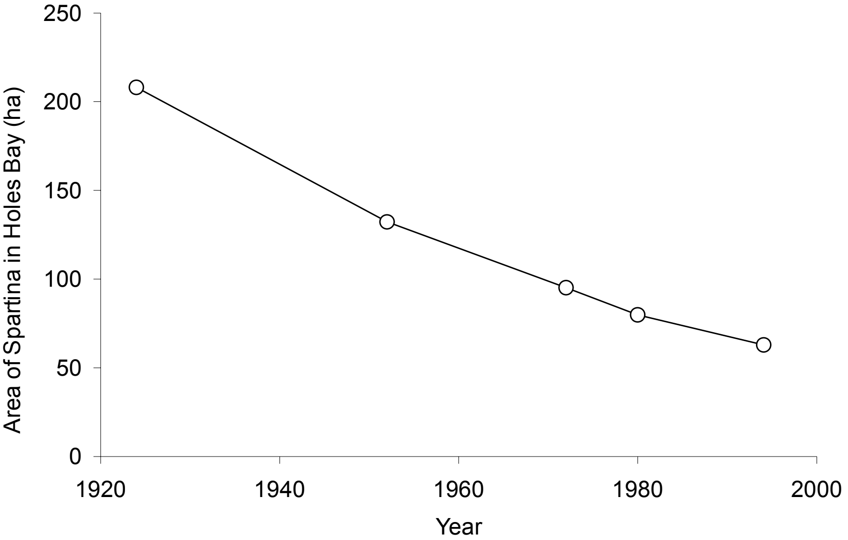

Figure 16. Changes in the area of *Spartina* in Holes Bay, Poole Harbour (plotted from Raybould, 2000)

The cause of the dieback, which was also widespread in southern Britain and Europe, was thought to relate to highly anaerobic soil conditions which prevail in dieback areas (Gray *et al.*, 1991*)*. The phenomenon was viewed in the longer term as part of the process of niche adjustment - by causing rapid accretion of large volumes of sediment, and irreversibly changing the physiographic and hydraulic characteristics of the lower vegetated zones, *Spartina* had paved the way for its own destruction and a gradual narrowing of the species' realised niche was predicted (Gray *et al.,* 1995). This process may be expedited by the genetic uniformity of *Spartina* and its consequent vulnerability to pathogens such as the ergot fungus *Claviceps pupurea* (Raybould, 2000: Raybould *et al.,* 1998), and the effects of projected climatic and sea-level changes which are difficult to predict in detail (Long, 1990).

Spartina marsh in Poole Harbour has also been lost by the invasion of other saltmarsh species such as *Phragmites* in areas of low salinity (Ranwell *et al.*, 1964: Gray, 1986), and *Atriplex portulacoides, Puccinellia maritima* and *Elymus pycnanthus* (Gray, 1986) although these may be acting opportunistically in colonising intertidal areas modified by *Spartina.*

One of the principal effects of the spread of *Spartina* was to significantly alter the sedimentation regime of the Harbour (detailed accounts in Bird and Ranwell.1964: Hubbard and Stebbings, 1968). Initially, *Spartina* caused accretion and consolidation of sediment by rhizome growth; up to 1.8m of sediment (1 million tonnes) was estimated to be trapped by *Spartina* in the upper Harbour over 55 years (Hubbard and Stebbings, 1968). Raybould (2000) considered that the principal hydrographical effect of the expansion of *Spartina* was the stabilisation of previously mobile sediments and mudflats, rather than accretion of new material, which in turn led to a deepening of many navigational channels in the Harbour. Subsequent die back and erosion of *Spartina* released large amounts of sediment into the Harbour very quickly, resulting in shoaling in all channels in the main Harbour (Green, 1940).

Spartina has also had important ecological effects: its initial spread was implicated in the decline of wader populations (particularly dunlin, *Calidris alpina*) in many British estuaries (Goss-Custard and Moser, 1988). Recent analysis of wader numbers in Poole Bay (1969-1994) revealed a highly significant upward trend ($r = 0.63$, $p < 0.01$) which could be related to *Spartina* loss; a significant relationship was also found between loss or gain of *Spartina* and increase or decrease in wader numbers in Poole and other estuaries in the last 25 years (Raybould, 2000).

Gray (1985) noted that the 'upper salt marsh' of Poole Harbour consists of at least three recognisably different communities: the most seaward of these is dominated by *Plantago maritima* but may contain many species. Brackish and rush communities are dominated by *Juncus maritimus, J. gerardii* or *Scirpus maritimus,* these communities tend to be species rich, particularly where transitions to unimproved grazing marsh occur. A more uniform salt marsh, characterised by *Elymus pycnathus,* develops on locally well-drained areas. Where chlorinities are below 1.2%, dense areas of brackish marsh are dominated by *Phragmites australis*

Zostera

Zostera beds are reported to have once been widespread in Poole Harbour, but by 1980 appeared to be confined to the East side of Saltern Pier, main channel (Holme and Bishop, 1980). There is no direct evidence as to the cause of this disappearance, although green algal blooms are implicated: a thick blanket of *Enteromorpha*, such as that occurring in Poole Harbour is considered to have eradicated *Zostera* in Langstone Harbour further along the south coast (den Hartog, 1994).

There are no recent reports of *Zostera* within the Harbour, although extensive beds, in Studland Bay, noted by Holme and Bishop (1980) to be eroding, were later mentioned by Light (1994) with a brief description of associated molluscan species, notably *Rissoa membranacea, Hydrobia ulvae, Tragula fenestrata* and *loripes lucinalis.* This Bay is just outside the marine site but may be affected by the water quality of the Harbour as a result of flushing and mixing processes.

Macroalgae

Sargassum muticum is a large brown macroalga, its fronds, which can grow to over 3m in length, have spherical gas bladders which render them relatively buoyant. Since it was first discovered on the Isle of Wight in 1973, *S. muticum* has spread along the south coast to the Isles of Scilly and along the north Cornish coast to Lundy. Populations have also become established in Strangford Lough in Northern Ireland. It is now common on the South coast of England, dominating low shores with a broken stone or boulder substratum. Because of its rapid growth and reproductive capacity it can compete with native species such as *Zostera* and is considered a nuisance in harbours, beaches and shallow waters. However, Withers *et al*. (1975) reported a rich epiphytic community associated with *S. muticum* collected from the east Solent, suggesting that native epiphytic species are not particularly affected, indeed species richness may even be enhanced (Critchley *et al*., 1990).

As a fouling organism, *S. muticum* does have major economic impacts: it is reported to interfere with recreational use of waterways, particularly when it becomes detached from holdfasts and floats off forming large masses (Dyrynda, 1987: Farnham *et al.,* 1981). It can block propellers and intakes of boats and is a nuisance to commercial fisheries: *Sargassum* can proliferate on oyster beds obstructing dredges and even attach to live oysters and 'steal' them by floatation and tidal transport. The long fronds can also foul fishing nets (Critchley *et al.,*1983, 1986).

Sargassum was first recorded in Poole Harbour in 1978. It formed copses within areas not previously noted for seaweeds, including shallow navigational channels where the long fronds accumulated in sufficient densities to cause nuisance to small vessels. *Sargassum* has increased in abundance since 1978, and is now one of the three most abundant algal species in the harbour, partially displacing various native organisms. Although it is a locus for substantial biological productivity, the resulting copses are less species-rich than other major epibenthic communities (Dyrynda 1987). Dense copses of *Sargassum* are reported to act as sediment traps on the harbour bed, accumulating small banks of sediment within their vicinity. However, these sediment accretions are remobilised during the winter months when the plants die back and shed their fronds. Middle and upstream channels are not affected by *Sargassum* to the same extent as the lower reaches of the Harbour; reports suggest that reduced salinities, coupled with the turbidity and resultant light limitation, precludes *Sargassum* establishment and only isolated plants are swept to these areas (Dyrynda, 1987).

Ulva lactuca and *Enteromorpha intestinalis* in Poole Harbour, and in particular Holes Bay, have been the subject of several studies in relation to eutrophication and the effects of discharge of sewage and industrial effluents (Howard and Moore, 1988: Savage, 1971). In the late 1960's, CEGB (Central Electricity Generating Board) was experiencing difficulties with its cooling water intake screens (located in Holes Bay) becoming blocked by *Ulva* and *Enteromorpha*. Investigative reports recommended reducing the nutrient concentrations in the water column (see section 5.2.1) and modelling exercises indicated that the inherently restricted flushing characteristics of the Harbour exacerbated the eutrophication problem. Dyrynda (1987) found that *Ulva* occurred throughout the harbour and was the fourth most common epibenthic species in term of % biomass (table 18). Holes Bay and Blue Lagoon were reported to be particularly affected by abnormal proliferation of green algae. (Howard and Moore, 1988) noted that exaggerated growth of *Ulva and Enteromorpha* completely blanketed large areas of Holes Bay resulting in a very low diversity of communities and species. It is also possible that green algal blooms may have contributed to the disappearance of *Zostera* in Poole Harbour; there is compelling evidence that an *Enteromorpha* bloom is responsible for the suffocation of a *Zostera* bed in Langstone Harbour further to the east (den Hartog, 1994). Blanketing of sediment can have major impacts on the underlying sediment biota, bringing bivalves to the surface. Other ecological impacts of green macroalgal blooms are discussed in Raffaelli *et al.* (1998) and can include exclusion of surface deposit feeders and reduced abundance of invertebrate prey for fish and shore birds.

Proliferation of macroalgae in Poole Harbour is apparently an ongoing problem. There appears to have been an increase in macroalgal cover in Holes Bay since 1995, leading to a number of complaints form the public, also macroalgae have been implicated in shellfish mortalities in 1995-7 (EA, 2001) (See also section 5.2.1).

Microalgae

Phaeocystis pouchetti was found in samples from throughout Poole Harbour in the spring of 1990, and a bloom of *P. pouchetti* was observed in late summer of the same year (NRA, 1990). This species is probably the most common bloom-forming alga in British coastal waters, and creates clouds of white, cream or brown-coloured foam. The formation of this foam is promoted by high N:P ratios, in particular, high nitrate:P (EA, 1997). Bloom numbers of *Phaeocystis* occurred at various sites throughout Poole Harbour during 1996, notably Harbour entrance, Hutchins Buoy and Holes Bay (EA, 1997).

Diatom blooms, apparent as scum, along much of Poole Quay in April 1995, was found to consist principally of benthic diatoms *Nitschi*a and *Navicula* spp. In the same year, scum samples (benthic algal mat) collected at Hutchins Buoy consisted of *Nitschi*a (50000 ml^{-1}) *Navicula closterium* (20000 ml^{-1}) and *Pleurosigma* (1000 ml^{-1}).

A follow up survey after the 1995 bloom found the dominant species were *Thalassiosira* (<40 ml^{-1}), with other species present including; *Scripsiella, Gyrodinium* (Dinophyceae)*, Chaetoceros, Ditylum, Pleurosigma* (Bacillariophyceae) and fragments of *Phaeocystis* (Haptophyceae) (EA, 1997).

Potentially toxic algae have also been reported in Poole Harbour (EA, 1997). *Gyrodinium aureolum* was found in low levels at three sites in the Harbour in early 1995 and summer 1996, with highest counts in Holes Bay. *Gyrodinium* can cause fish kills and mortalities of other marine organisms, either directly by exposure to toxins, or indirectly through the food chain (Steidinger and Vargo, 1988). *Alexandrium tamarense* was also found at two sites in the Harbour during 1996, Harbour entrance and Town Bridge (EA, 1997). This species causes paralytic shellfish poisoning (PSP) and is responsible for 'Red tides' which occur in the British coastal waters, notably the Fal Estuary in Cornwall.

Macrofauna

Table 18. Most common epibenthic species of Poole Harbour (from Dyrynda, 1987)

Species	Common name	% Biomass*
Crepidula fornicata †	Slipper limpet	31.8
Ascidiella aspera	Sea squirt	22.4
Sargassum muticum †	Jap weed, wire weed, strangle weed	16.1
Ulva lactuca	Sea lettuce	6.2
Styela clava †	Korean, or leathery sea squirt, clubbed tunicate	6.1
Halichondria bowerbanki	Atlantic sponge	4.2
Ostrea edulis	Native oyster	3.0
Sabella pavonina	Peacock worm	2.8
Mytilus edulis	Common mussel	2.4
Laminaria saccharina	Oarweed	1.4
Haliclona oculata	Eyed finger-sponge	1.3
Hymeniacidon perleve	Orange breadcrumb sponge	1.3
Halichondria panicea	Green breadcrumb sponge	0.5
Cliona celata	Boring sponge	0.4
Suberites ficus	Fig sponge	0.3

* Wet weight values according to dredge survey
† denotes non-native species

Dyrynda, (1987) listed the 15 most common epibenthic species of Poole Harbour (table 18). It is notable that three non-native species feature high on the list. The Harbour is a high-risk environment with respect to non-indigenous species as it has served as a sea port for cargo vessels since Roman times and has historical links with North American ports spanning several centuries. It has long been a centre for molluscan aquaculture, with deposits of native oysters and mussels being augmented historically by stocks of exotic shellfish imported directly from continental Europe and North America.

The gastropod mollusc *Crepidula fornicata* is known to have been introduced to Essex between 1887 and 1890 from North America, in association with the American oyster *Crassostrea gigas* (Crouch 1894, 1895; Orton 1912; Fretter and Graham 1981) and spread fairly rapidly (Franklin and Pickett 1974). Its success in this country is probably due to a lack of predators and the unusual method of reproduction (which relies upon individuals settling upon each other to form breeding 'stacks' as they develop from males to females). A pelagic larval stage aids the spread of the species, once introduced.

Reports suggest that high densities of *C. fornicata* can modify the nature and texture of sediments in some bays (Ehrhold *et al.,* 1998) and where *Crepidula* stacks are abundant, few other bivalves or other filter-feeding invertebrates can live amongst them. This is due to spatial competition, trophic competition and alteration of the substratum (the pseudofaeces of *C. fornicata* smother other bivalves and render the substratum unsuitable for larval settlement) (Fretter and Graham, 1981; Blanchard, 1997). In this way, *C. fornicata* has become a serious pest on oyster beds and has caused many traditional oyster fisheries to be abandoned (e.g. in the Norman Gulf, France) (Blanchard, 1997). However, De Montaudouin *et al*. (1999) showed that *Crepidula* had no major influence on the local density or diversity of smaller coexisting macroinvertebrates and did not affect the growth of 18 month old oysters.

There is no doubt that the slipper limpet is a serious invader in Poole Harbour, forming very dense beds within former oyster beds. It has become the most common sub-tidal species and the major community-former within the harbour. Dense *Crepidula* beds occur in most channels of the Harbour with the exception of the Upper Holes Bay and Lytchett Bay channel. Aggregations generally accumulate on relatively level channel beds or in holes, as clusters of *Crepidula* can be moved around by tidal currents. *O. edulis* numbers are known to have declined significantly, and the inability of populations to recover may be largely attributed to the space competitive superiority of C*repidula*, whose preferred habitat overlaps that of *O. edulis.* Thus, *C. fornicata* has brought about a major change in the natural ecology of Poole Harbour with unfavourable consequences in the suppression or elimination of native species. However effects are not wholly detrimental, *Crepidula* beds are amongst the most species rich in the harbour, and the vast number of dead shells add to the number of small hard substrates available for colonisation by sessile epibenthic species Dyrynda, (1987). With the trend of rising temperatures *Crepidula* could become even more widespread; Minchin *et al.,* (1995) indicate that temperatures may be an important limiting factor in its ability to develop extensive populations.

Elminius modestus, is an Australasian immigrant that reached south Devon in 1948. Its numbers increased during the 1950s and it has displaced most of the native intertidal barnacles and now dominates the intertidal hard substrata (natural and artificial) of Poole Harbour.

The ascidian *Styela clava* was first introduced in Plymouth in the early 1950s (Carlisle 1954; Houghton and Millar, 1960). It has spread to a number of estuarine habitats in Southern Britain including Poole Harbour, where it is present in sufficient numbers to feature in terms of biomass as one of the 'lesser' dominant species (Dyrynda, 1987). It occurs on hard substrate (e.g *Crepidula* shells, subtidal sea walls) across much of the harbour, but is not thought to constitute a major threat to native species, although there may be competition for space and food where *Styela* and *Ciona intestinalis* co-occur.

Tapes philippinarium, the Manila clam, is a non-indigenous species introduced as hatchery reared spat into Poole Harbour for cultivation in 1988 by Othniel Shellfisheries. *T. philippinarium* grows faster than the native *T. decussates* (the Palourde) and therefore was a more appropriate choice for cultivation. There were concerns that the Manila clams could spawn during warm summers, especially as the waters of Poole Harbour are sheltered, as an established community may affect the natural ecology of the area (Nicholson and Barry, 1995), but it was assumed that the clams would not reproduce successfully in the wild. However, the farmed Manila species did spawn successfully and created a naturalised population which has been farmed commercially within Poole Harbour (Morgan 2000). Manila sites include Arne Bay, Ower Bay, Seagull Bay and Giggers Island. There is no further information on the Manila clam, or the effects of its presence in Poole Harbour.

There are several other deliberately introduced species, notably *Crassostrea gigas*, the Portuguese oyster. This species is farmed (from hatchery reared spat) in the Harbour, and although it was originally thought that temperature of UK waters would preclude spatfall, 30 years of extensive cultivation has provided opportunities for establishment in the wild. Under favourable conditions for larval development*, C. gigas* has the capacity to spread substantial distances (Spencer *et al.,* 1994). *C. gigas* was the subject of experiments in Poole Harbour between 1989 and 1991, when 200 hatchery-reared oysters were deployed at three sites in the harbour to determine the effects of restrictions on TBT use (Dyrynda 1992, see section 5.1.2).

(Howard and Moore, 1988) describe the nature conservation importance of Poole Harbour and note that several species present have been rarely recorded elsewhere in Britain, notably mollusc *Aeolidiella sanguinea,* sponge *Suberites massa,* and the bryozoans *Anguinella palmata* and *Farella repens. S. massa* was the major component of an assemblage of eight sponges in the entrance channel to Holes Bay. The nationally worm *Sabella pavonina* forms extensive beds in Poole Harbour.

Other key findings of Howard and Moore (1988) are as follows:

- Diversity of habitats and communities was generally low – highest diversities were found in the downstream areas of the harbour associated with the beds of *Sabella pavonina*.
- Lowest species diversity was found in Holes Bay
- Productivity of sediment infauna was high and dominated by oligochaetes and polychaetes.

Harris (1983) recorded macrofauna, meiofauna and diatoms in sediments of Holes Bay. The only macrobiota found were *Hydrobia ulvae, Ulva lactuca, Enteromorpha sp., Scrobicularia plana, Hediste diversicolor, Mya arenaria, Littorina littorea, Neanthes virens* and a few *Vaucheria sp.* Meiofaunal diversity was also low and there were very few nematodes and harpacticoid copepods. 69 species of epipsammic diatoms were recorded with 12 to 20 species at most sites. Harris concluded that Holes Bay was impoverished in terms of diversity and density of organisms.

Another detailed study of Holes Bay was carried out in the same year (Dyrynda. 1983). The impoverished flora and fauna recorded was considered to be a result of the restricted flushing characteristics of the Bay, combined with the effects of discharges which rendered the Bay eutrophic.

Fish

A small number of fish are harvested within the Harbour in substantial (commercial) numbers, although many species are recorded within the Harbour. Table 19 shows fish recorded in the harbour prior to 1987.

Table 19. Fish species recorded within Poole Harbour (adapted from Dyrynda, 1987)

Species	Common name
Agonus cataphractus	Pogge
Anguilla anguilla	Common or European Eel
Belone belone	Garfish
Blennius gattorugine	Tompot Blenny
Blennius pholis	Common Blenny
Clupea harengus	Herring
Conger conger	Conger Eel
Cotus bulbalis	Long Spined Sea Scorpion
Cyclopterus lumpus	Lumpsucker
Gadus morhua	Cod
Gobiusculus flavescens	Two-spotted Goby.
Gobius niger	Black Goby
Morone labrax	Bass
Mugil labrosus	Thick Lipped Grey Mullet
Mullus surmulletus	Red Mullet
Nerophis lumbriciformis	Worm Pipefish
Pltichthyes flesus	Flounder
Pleuronectes platessa	Plaice
Pollachius pollachius	Pollack
Pomatoschistus minutus	Sand Goby
Salmo samar	Salmon
Salmo trutta	Sea Trout
Scomber scomber	Mackerel
Scopthalmus maximus	Turbot
Scopthalmus rhombus	Brill
Solea solea	Dover Sole
Spondyliosoma canthrus	Black Sea Bream
Syngnathus acus	Greater Pipefish
Trachurus trachurus	Scad

The principal commercially fished species include the **common eel**, which occurs in the rivers Frome and Piddle, and within the Harbour, where eels have historically been recorded in large numbers and are the subject of a small fyke-net fishery. However, eel numbers are reported to be declining in Poole Harbour, also in the rivers Frome and Piddle. This downward trend is consistent with the decline currently observed throughout Europe, and attributed to marine factors (EA, 2000). Also fished commercially are **grey mullet**, a common grazer within all parts of the harbour but particularly in the upstream areas flanked by mudflats, **bass**, also common in the downstream sections where they are caught by seine and gill-netting and by rod and line, and flatfish – **flounder, sole and plaice**, which are trawled.

Migratory salmon and trout spawn in both the Frome and the Piddle , and both species have been recorded in tidal reaches of the Sherford. Bass are also found in the tidal reaches of all three rivers.

An artificial reef complex, constructed from blocks made of stabilised coal-fired power station waste materials was deployed in Poole Bay in 1989 (Collins *et al.,* 1990). Monitoring of the growth of organisms on the reef and chemical analyses showed that the material was suitable for use in the marine environment. This is outside the marine site, but merits inclusion, firstly, for the constructive use of waste products, previously dumped as raw pulverised fuel ash with subsequent infaunal mortality (Eagle *et al,* 1979; Bamber, 1984), secondly, as a fishery enhancement exercise (Collins *et al.,* 1990), and thirdly as a long-term monitoring study (Baldock *et al.,* 2001). Initial results came within hours of deployment when shoals of post-larval fish, bib (*Triscopterus luscus*) and pollack *Pollarchius pollachius*) were attracted to the site. After three weeks, lobsters (*Hommarus gammarus*) were recorded at the reef, subsequently lobster movement studies were initiated using tagging techniques with the aim of investigating behaviour, and the potential for stock enhancement schemes, (Jensen *et al.,* 1994). Lobsters on the artificial reef and natural reefs in the area were studied intensively for 10 years including electronic tracking of up to 20 animals every minute of the day for nearly two years. In 1998 the reef experiment was doubled in size with the deployment of scrap tyre modules, again to study the marine environmental impact of the material. Colonisation was rapid and biota showed no significant uptake of tyre compounds. The potential use of scrap tyres in coastal defence structures is also being investigated.

Overall, results from this exercise have been promising, and suggest that artificial reefs may be useful in the creation of fisheries where none exist at present, and to supplement habitat of existing fisheries (Collins *et al.,* 1990), this will undoubtedly prove to be benefit all marine life of the area and not only target species.

Birds

The mudflats and salt marshes in and around Poole Harbour are of great ecological value for feeding and roosting birds. Full description of their distributions is beyond the scope of the current report. However, over winter, the area regularly supports around 30000 individual waterfowl including those listed in table 20 and for which Poole Harbour qualifies as an SPA.

Table 20. Bird populations of European importance in Poole Harbour (from www.jncc.gov.uk)

Species	Common name	Notes
Tringa totanus	Redshank	
Numenius arquata	Curlew	
Calidris alpina	Dunlin	
Vanellus vanellus	Lapwing	
Mergus serrator	Red-breasted Merganser	
Bucephala clangula	Goldeneye	
Aythya ferina	Pochard	
Anas clypeata	Shoveler	
Branta bernicla bernicla	Dark-bellied Brent Goose	
Phalacrocorax carbo	Cormorant	
Limosa limosa islandica	Black-tailed Godwit	‡
Tadorna tadorna	Shelduck	‡
Recurvirostra avosetta	Avocet	
Egretta garzetta	Little Egret	†
Sterna hirundo	Common Tern	*
Larus melanocephalus	Mediterranean Gull	*
Acrocephalus paludicola	Aquatic Warbler	†

* During the breeding season; † On passage; ‡ Migratory

Recently the British Trust for Ornithology has carried out a review of species trends in SPAs over the last 5, 10 and 25 year time periods (up to 2000) using data collected as part of the Wetland Bird Survey (WeBS). SPAs where species are declining at a rate of greater than 25% over a specified time period when the larger-scale regional or national trends indicate stable or increasing population sizes are targeted as being of concern. Population declines of between 25% and 50% are flagged as 'Medium Alerts' and declines of greater than 50% as 'High Alerts'. Alerts are intended as advisory measures triggering further investigation. The report, produced for the Environment Agency, English Nature and the Countryside Council for Wales summarises statistics for twelve evaluated species in the Poole SPA: Cormorant, Dark-bellied Brent Goose, Shelduck, Shoveler, Pochard, Goldeneye, Red-breasted Merganser, Avocet, Dunlin, Black-tailed Godwit, Curlew, Redshank (Armitage *et al* 2002).

A 'High Alert' was triggered for Pochard over the 10 year period and a Medium Alert for the latest 5 years. Since 1985/86 numbers of this species have fluctuated markedly but are nevertheless considerably higher (280 in 1999/2000) than in the mid-1960s (27). A 35% decline in the nationally important Shelduck population in the latest 5 year period has led to a 'Medium alert' although, again, the long term trend for this species is not clear. Other bird populations have either remained stable or increased. In particular, Avocet (a species of national importance) have become much more numerous in the last 15 years. On balance, therefore, the trends in bird numbers and possible adverse factors (cited by Armitage *et al.* as including recreational disturbance and over-exploitation of shellfish stocks) were not considered sufficiently important to trigger further investigations into the causes of population changes ('Level 2' assessment).

7. WATER STATUS AND QUALITY STANDARDS

In this section we examine EA data (from WIMS) on determinands which may influence Poole Harbour SPA. Summary statistics have been drawn up by the Agency (based on monitoring since 1990), and raw data analysed in an attempt to establish further evidence as to whether or not water quality is likely to cause impact. Where relevant, temporal trends are discussed - otherwise only the most recent data are shown.

It should be noted that much of the data from monitoring surveys are often several years old, and may be for the purpose of compliance monitoring only. Detection limits are often set with that specific intention in mind, such that the data may be of limited value for environmental behaviour studies. Nevertheless (half) detection limits have usually been included in summary statistics since it allows at least a crude assessment of water quality issues. With this caveat in mind we have scrutinised summary statistics supplied by the EA for a number of determinands. These statistics are broken down in to:

1) **Discharges** – to gauge the importance of specific industrial and trade effluent point sources. Here another major caveat has to be introduced, since information for a number of discharges is not available. Calculation of fluxes is beyond the scope of the current project, therefore only available concentration data are discussed for most contaminants (with regard to potential threat to the site).
2) **Harmonised monitoring points** (HMP) or the equivalent freshwater site immediately above the tidal limit (to characterise riverine input). Again data are not always available at each site – the majority of information relates to the Frome and Piddle, the two major sources of freshwater to Poole Harbour.
3) **Tidal waters** – a review of data within the SPA. Because the EA data set do not contain widespread information on contemporary values, entries recorded over the last ten years have been summarised to provide a more integrated picture of water quality issues, and to make comparisons with Environmental Quality Standards.

The majority of List I and List II (Dangerous Substances) determinands have been screened here, together with other water quality parameters such as nutrients and DO. In the absence of extensive site-specific biological effects information, comparisons of water-monitoring results with Environmental Quality Standards (EQS) are used in order to gain a first-order approximation of possible impact on biota. Thus, in the context of the current project, descriptions of 'threat' or 'risk' to the site from individual contaminants are scaled against the relevant EQS, assuming this to be an appropriate threshold for the protection of aquatic life.

For a number of reasons this is an uncertain supposition. The compliance limits for contaminants and other water quality parameters are themselves based on reviews of general toxicity data for aquatic life, coupled with a safety margin below the lowest reliable adverse effects concentration. The assumption is that below the EQS, adverse biological and ecological effects *are unlikely*. Above the EQS, effects *might be expected to occur* though this will depend on the magnitude and duration of the exposure. The application of EQS values involves uncertainties arising from limited toxicity data, differential responses between chronic and acute toxicity, inter-species variation in sensitivity, and modifying factors within each individual ecosystem (notably, the issue of synergy and additivity discussed below). Sensitivity may also vary between different levels of biological organisation; lower-order effects (molecules and cells) are likely to occur at lower levels of contamination, and in advance of, community and ecosystem-level response. Often this involves a high degree of precaution in setting standards and could give rise to an apparent mis-match between chemical data and measured biological responses, particularly at the level of biological diversity. Conversely, it is also possible that subtle effects may occur at concentrations below the EQS, giving rise to a failure to protect the system. Compliance/non-compliance patterns are therefore not necessarily synonymous with ecological implications: at present the latter can only be gauged by considering a wider array of ecosystem characteristics. EQS values are used here merely help to prioritize some of those sites and contaminants which merit closer investigation. They do not necessarily assure Favourable Condition.

Another drawback to the EQS approach is that it considers the toxicology of contaminants individually, assuming that each is acting independently of others. In reality, some of the more significant discharges contain a range of contaminants which, though they may individually pass the 'EQS test', may pose a greater threat to nature as a result of additive toxicity. The question of synergistic/ antagonistic interactions from outfalls should be a priority for future research.

7.1 Toxic Contaminants

7.1.1 Metals

Results are discussed here on a metal by metal basis, based on EA statistics for freshwater, estuarine water and outfall data, collected over the last ten years.

Arsenic

The EQS for As in fresh waters is 50µg l^{-1}. The only riverine sources into Poole Harbour for which data are available are for the River Frome at Holme Bridge and the River Piddle at West Mills. The majority of values here are close to the detection limit of 1µg l^{-1} and can be considered as background values for As in rivers. No obvious temporal trends can be discerned from the data.

The pattern of dissolved As in estuarine water for six sites around Poole Harbour are plotted in figure 17. Annual averages are invariably below the EQS for tidal water (25 µg l^{-1}) by an order of magnitude. Even highest concentrations are only marginally above background. These occur at Poole Bridge[1].

Discharge data (annex 7) indicate small inputs from STW and Merck. An estimated 15kg As was discharged to the Harbour from STW in 2001 (compared to a riverine flux of approximately 140kg, assuming an average total arsenic concentration in fresh-water of 0.5µg l^{-1}).

There are no obvious temporal trends for As in tidal waters. Highest values tend to be found in some of the earlier data (early 1990s) though this is probably a function of

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[1] There are no EA monitoring sites within Holes Bay. The sampling site at Poole Bridge may, arguably, provide a general view of the water quality in Holes Bay, as the Bay effectively empties at low tide, with water mostly restricted to low water channels. However, it is difficult to say if they are reflective of Harbour water, or Holes Bay water without further detailed analysis in relation to salinity.

higher detection limits in the past. For the sites monitored dissolved As poses little threat.

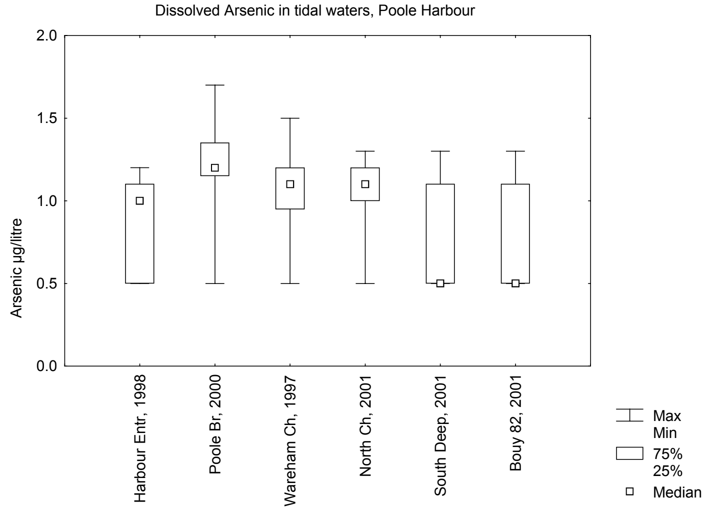

Figure 17. Concentrations of dissolved As µg l^{-1} in estuarine waters, Poole Harbour. Data source EA.

Cadmium

The riverine sources into Poole Harbour for which most data are available are the River Frome (Holme Bridge) and the River Piddle (West Mills, Wareham). As indicated in figure 18, the majority of values for dissolved Cd are close to the detection limits: respective means in 2001 were 0.051 and 0.01 µg l^{-1} respectively. Higher values were recorded (in 1993) in samples from Holton Heath Stream, both upstream and downstream of the STW. No obvious temporal trends can be discerned from the data (other than detection limits have decreased during the recording period: 90% of freshwater samples were below DLs).

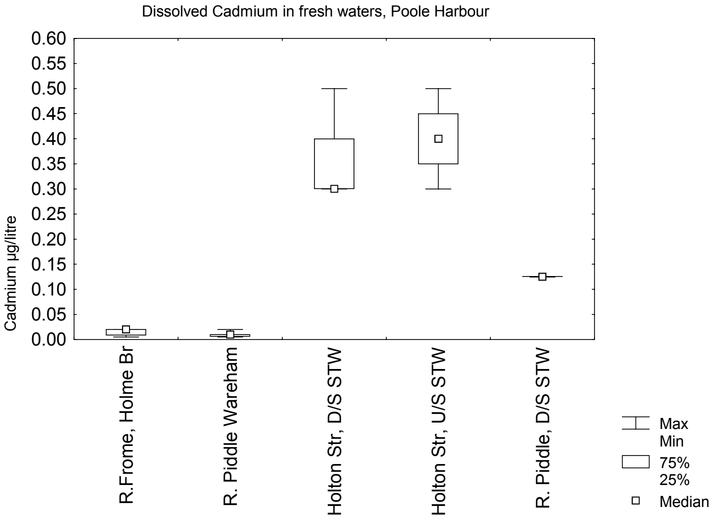

Figure 18. Concentrations of dissolved Cd (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream (1993). Data source EA.

The EQS for Cd in fresh waters is 5µg l^{-1} and relates to 'total' rather than dissolved metal. Clearly dissolved Cd at all the above sites fall within this standard value.

Long-term summary statistics for total Cd were available for the River Frome (Holme Bridge) only (figure 19). Occasional values of 0.5µg l^{-1} have been recorded in earlier years though these appear to be becoming less frequent. Annual averages have been at 0.1µg l^{-1} or lower for more than a decade, indicating compliance with the EQS and that there are no untoward inputs from this source. The same is true in 2001 for the River Piddle (0.03µg Cd l^{-1}).

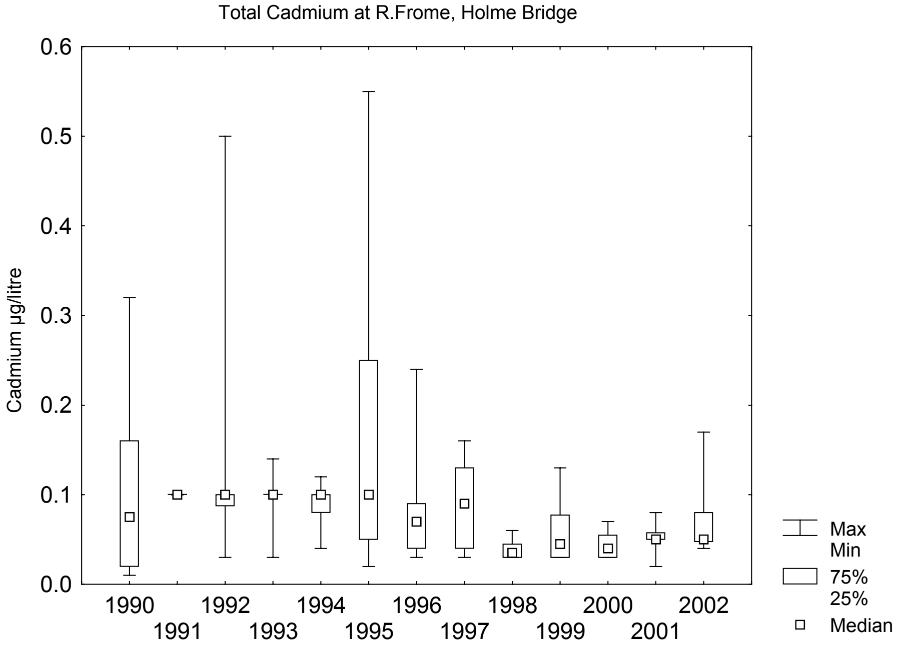

Figure 19. Concentrations of total Cd (µg l^{-1}) in the River Frome (Holme Bridge). Summary statistics 1990-2002. Data source EA.

The pattern of dissolved Cd in estuarine water around Poole Harbour is plotted in figure 20. Some of the unusual summary statistics in this 'box and whisker' plot signify the influence of detection limits (more than 80% of tidal water values were below DL). Nevertheless, annual averages are invariably below the EQS for saline waters (2.5 µg l^{-1}) by between one and two orders of magnitude. Highest concentrations were those at Poole Bridge.

Discharge data (annex 7) indicate small inputs from STW and, historically, Merck. An estimated 9.6kg Cd was discharged to the Harbour from STW in 2001 (compared to a riverine flux of approximately 14 kg, assuming an average total Cd concentration in fresh-water of 0.05µg l^{-1}). The annual mean Cd concentration in Poole STW final effluent in 2001 was 0.05± 0.02µg l^{-1} (max 0.15 µg l^{-1}).

Unfortunately there is no tidal data for Holes Bay, which is probably the most contaminated area (see section 5.1.1). However, at the above tidal water sites in the outer Harbour, there is little evidence to suggest Cd concentrations would be acutely harmful.

No temporal trends are discernible in the data.

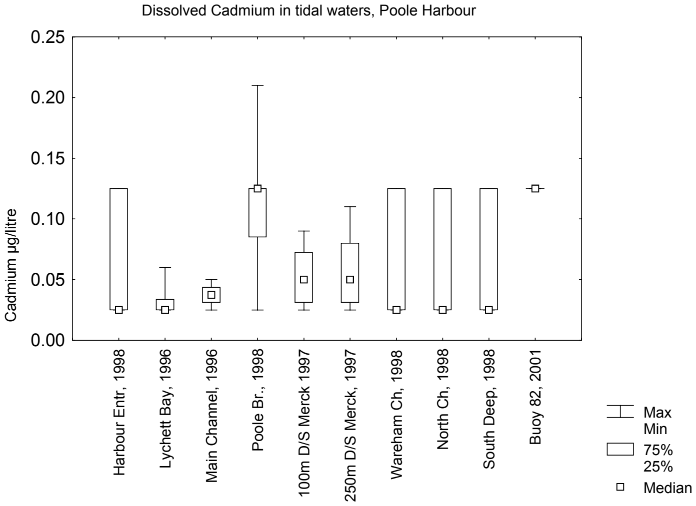

Figure 20. Concentrations of dissolved Cd µg l^{-1} in estuarine waters, Poole Harbour. Annual summary statistics. Data source EA.

Chromium.

The riverine sources into Poole Harbour for which most data are available are the River Frome (Holme Bridge) and the River Piddle (West Mills, Wareham). The majority of dissolved Cr values here are close to the detection limit (figure 21). Based on ½ DL values annual averages in 2001 were 0.3 µg l^{-1} for both sites. Higher values were recorded (in 1993) in Holton Heath Stream, both upstream and downstream of the STW, and also downstream of the STW at Wareham (figure 21). The EQS for Cr in fresh waters suitable for salmonids ranges between 5 and 50µg l^{-1} depending on hardness. All the points examined above would comply with even the lowest standard.

No obvious temporal trends can be discerned from the data (40% of freshwater samples were below DLs).

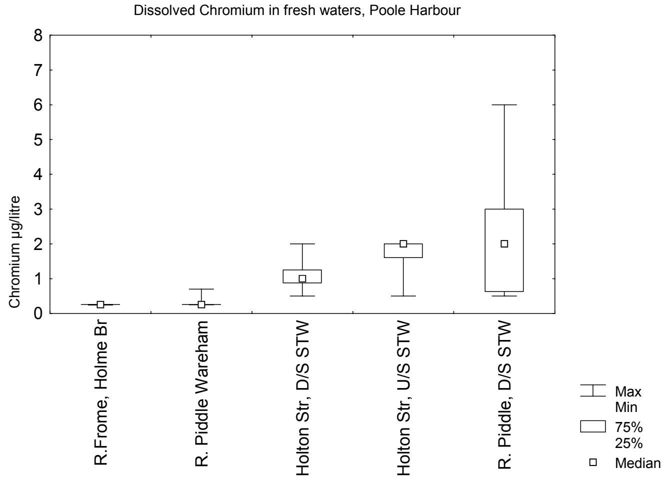

Figure 21. Concentrations of dissolved Cr (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream (1993). Data source EA.

The pattern of Cr in estuarine waters of Poole Harbour is shown in figure 22. Median values for all sites are lower than the EQS (15 µg l^{-1}) by more than an order of magnitude (almost 90% of data for tidal waters are below DL). There are occasional elevated levels of dissolved Cr at South Deep near Furzey Island. The source of this is not clear. Sporadic elevated levels are also recorded at Poole Bridge, presumably reflecting sources in Holes Bay.

Discharge data in annex 7 indicates inputs to Holes Bay from STW and, previously, Merck). An estimated 120 kg Cr was discharged to the Harbour from STW in 2001 (compared to a riverine flux of approximately 100 kg, assuming an average total Cr concentration in fresh-water of 0.35µg l^{-1}). The annual mean Cr concentration in Poole STW final effluent in 2001 was 3.2 ± 2.9µg l^{-1} (max 10.4 µg l^{-1}).

No temporal trends are discernible in the data.

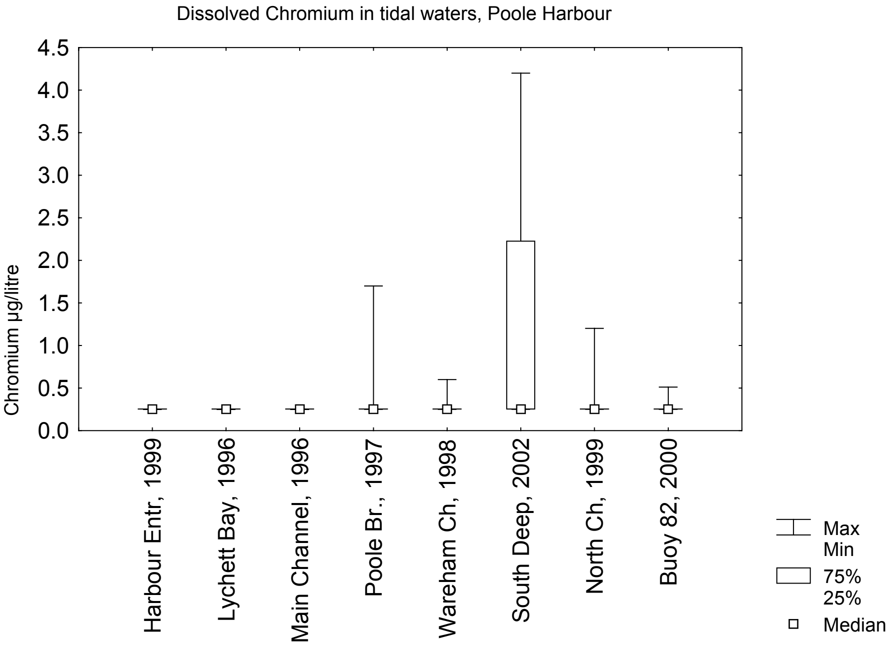

Figure 22. Concentrations of dissolved Cr µg l^{-1} in estuarine waters, Poole Harbour. Annual summary statistics for year shown. Data source EA.

At outer Harbour sites there is little evidence to suggest, from annual averages, that Cr concentrations in tidal waters would be acutely toxic; however some forms of Cr are toxic to invertebrates and fish at concentrations close to the EQS for saline waters (15 µg l^{-1}). Cr is also considered genotoxic. More comprehensive data are needed to assess possible sources in Holes Bay and South Deep and any sublethal impact on biota.

Copper

The riverine sources into Poole Harbour for which most data are available are the River Frome (Holme Bridge) and the River Piddle (West Mills, Wareham) where the majority of values are close to the detection limit (means for 2001, 1.0 and 0.8 µg l^{-1}, respectively). This is also the case for the River Sherford, River Piddle downstream of the Keysworth STW (figure 23) and the Corfe River (not shown). Higher values were recorded in Holton Heath Stream, both upstream and downstream of the STW.

The EQS for Cu in freshwater is in the range 1 - 28 µg l^{-1} depending on hardness. Since most of the catchments examined probably have a hardness towards the upper end of the range a typical EQS value of 10 µg l^{-1} might be applicable. In this case only the Holton Heath stream might be of concern. Trends for annual average dissolved Cu levels in the early 1990s however have shown a significant downward trend (figure 24).

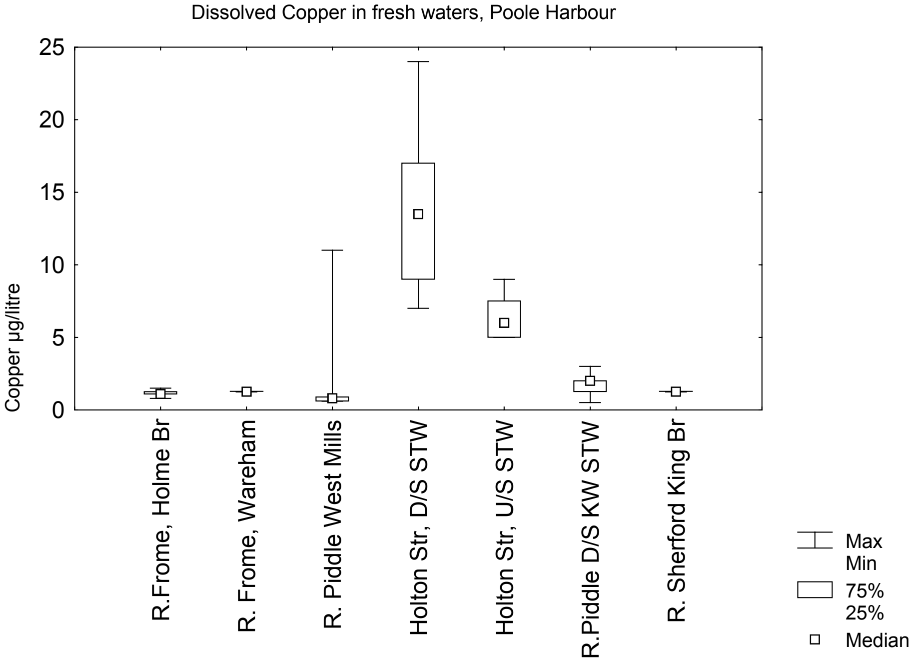

Figure 23. Concentrations of dissolved Cu µg l^{-1} in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream and Keysworth (KW) STW (1993). Data source EA.

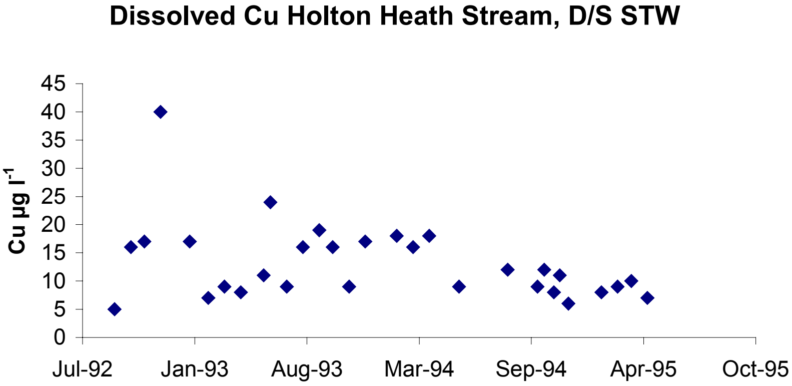

Figure 24. Trends in concentrations of dissolved Cu at Holton Heath stream (D/S STW). Data source EA.

The pattern of dissolved Cu concentrations in estuarine waters of Poole Harbour is shown in figure 25. Median values for all sites are below the EQS (5 µg l^{-1}). At the majority of sampling sites ~1 µg l^{-1} dissolved Cu appears to be the normal. However elevated levels are recorded at Poole Bridge, again implying sources in Holes Bay.

Discharge data in annex 7 indicate inputs to Holes Bay from STW and, historically, Merck. Antifouling paints and sediments may also be a source of some Cu.

An estimated 190 kg Cu was discharged to the Harbour from STW in 2001 (compared to a riverine flux of approximately 450 kg, assuming an average total Cu concentration of 1.6µg l^{-1}). The annual mean Cu concentration in Poole STW final effluent in 2001 was 8.3±3.2 µg l^{-1} (max 24 µg l^{-1}).

More comprehensive geographical and spatial data in Holes Bay are needed to assess possible influences on estuarine water quality and sublethal impact on biota, since Cu is potentially one of the most toxicologically significant metals.

No temporal trends are discernible in the data.

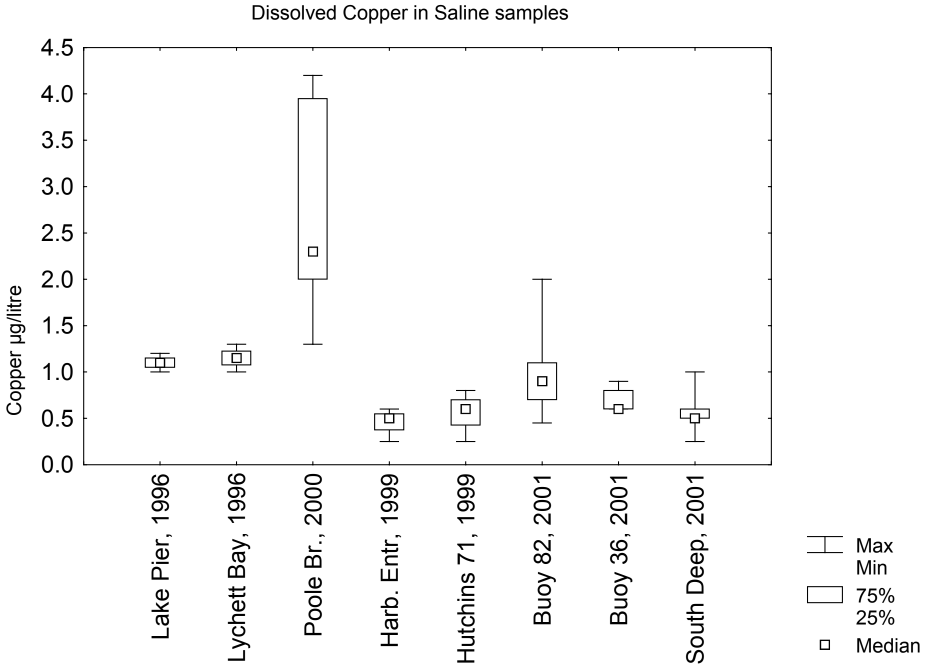

Figure 25. Concentrations of dissolved Cu µg l^{-1} in estuarine waters, Poole Harbour. Annual summary statistics for most recent year. Data source EA.

Iron

The riverine sources into Poole Harbour for which most dissolved Fe data are available are the River Frome (Holme Bridge) and the River Piddle (West Mills, Wareham). As indicated in figure 26, the majority of values here are of the order of 100 µg l^{-1} or less. (The same is true for waters in the river Piddle downstream of Keysworth STW at Wareham). Higher values were recorded (in 1992, when last determined) in the Holton Heath Stream, downstream of the STW. These values are close to the EQS for dissolved Fe (1000µg l^{-1}). No obvious temporal trends can be discerned from the data.

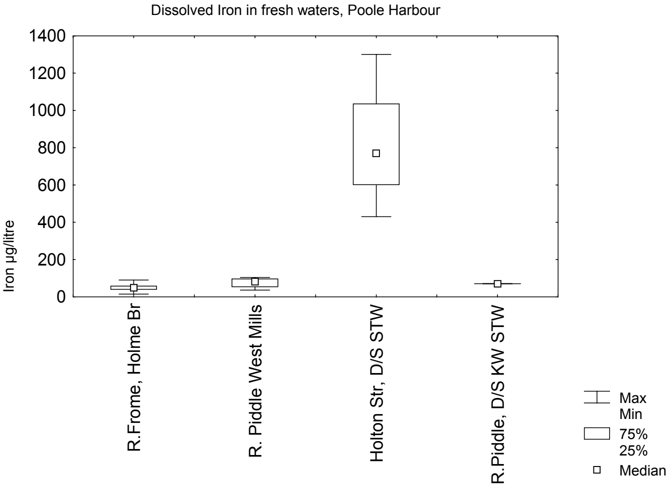

Figure 26. Concentrations of dissolved Fe (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream and Keysworth (KW) STW (1992). Data source EA.

The pattern of dissolved Fe concentrations in estuarine waters of Poole Harbour is shown in figure 27. Median values for all sites are well below the EQS (1000 µg l^{-1}). It is therefore unlikely that Fe would represent a threat to marine biota at the site. Occasional elevated levels are recorded at Poole Bridge, and it would appear that the incidence of these high values has increased in recent years (figure 28). The source of this Fe is not known (no values for Fe in discharges are available), but presumably originates in Holes Bay. No distinctive long-term trends can be seen for dissolved Fe at other estuarine sites.

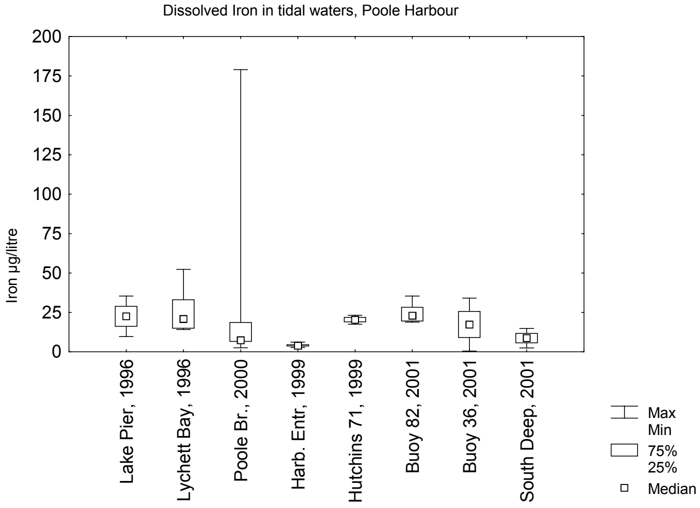

Figure 27. Concentrations of dissolved Fe (µg l^{-1}) in estuarine waters, Poole Harbour. Annual Summary statistics for most recent year. Data source EA.

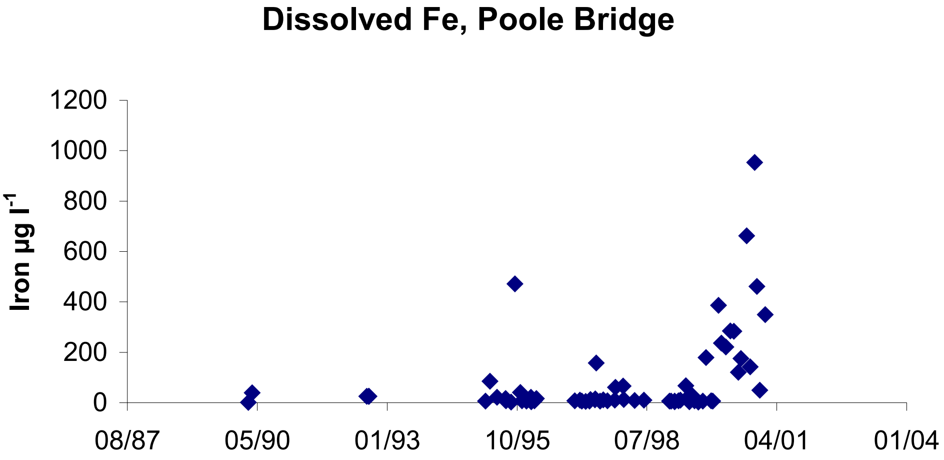

Figure 28. Trends in concentrations of dissolved Fe at Poole Bridge. Data source EA.

Nickel

Virtually all 2001 dissolved nickel data for the River Frome (Holme Bridge) and the River Piddle (West Mills) are at or below detection limits (mean values, based on ½ DL, of 1.5-2.5 µg l^{-1}). There appears to be little anthropogenic contribution from these rivers. Higher values are indicated for other sites (figure 29) although medians rarely exceed 5µg l^{-1} and are below the EQS for dissolved Ni which ranges between 50 and 200 µg l^{-1} depending on hardness. There are no unequivocal temporal trends for Ni probably because many of the data involve detection limit-derived values.

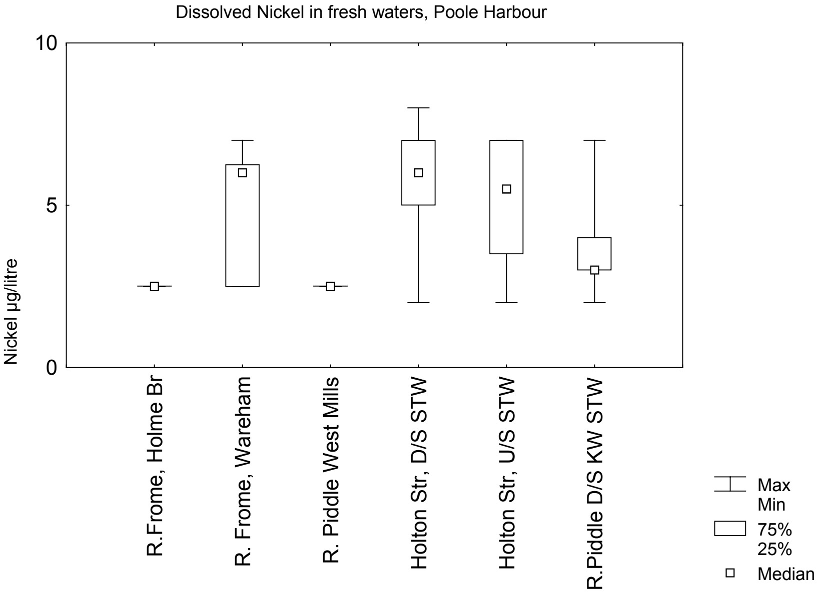

Figure 29. Concentrations of dissolved Ni (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream and Keysworth (KW) STW (1993). Data source EA.

The pattern of dissolved Ni concentrations in estuarine waters of Poole Harbour is shown in figure 30. Median values for all sites are well below the EQS (30 µg l^{-1}). It is therefore unlikely that Ni would represent a threat to marine biota at the site.

Again occasional elevated levels are recorded at Poole Bridge though these are not much higher than outliers seen at some of the other sites.

An estimated 610kg Ni was discharged to the Harbour from STW in 2001 (compared to a riverine flux of approximately 700 kg, assuming an average total Ni concentration of 2.5µg l^{-1}, based entirely on ½DL values). The annual mean Ni concentration in Poole STW final effluent in 2001 was 26.7±10µg l^{-1} (max 59µg l^{-1}).

Not surprisingly, lowest recorded concentrations are those at the Harbour mouth. No distinctive long-term trends can be seen for dissolved Ni again probably because many of the data involve detection limit-derived values.

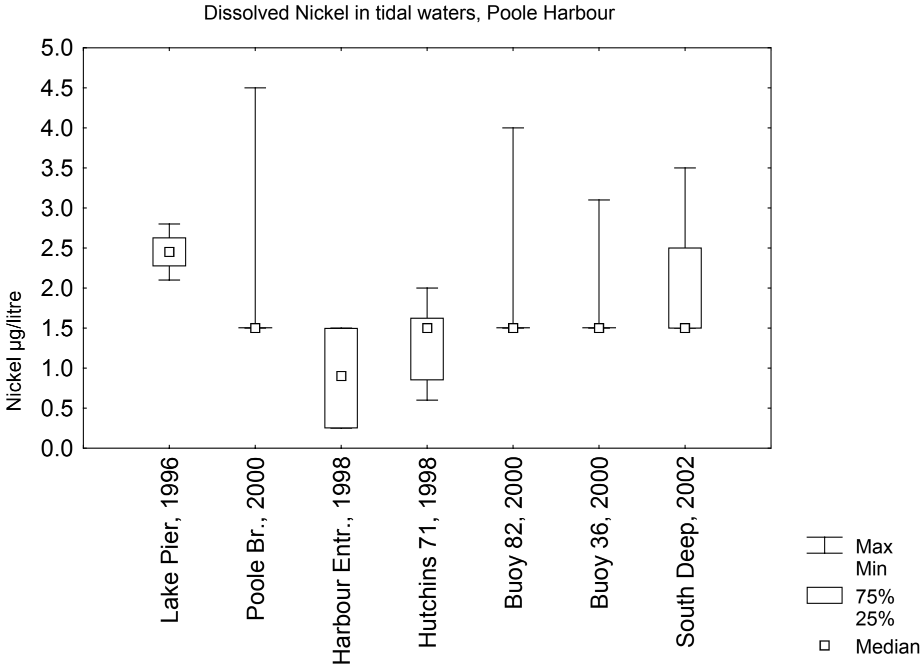

Figure 30. Concentrations of dissolved Ni (µg l^{-1}) in estuarine waters, Poole Harbour. Annual summary statistics for most recent year. Data source EA.

Lead

Virtually all 2001 data for dissolved lead for the River Frome (Holme Bridge) and the River Piddle (West Mills) are at or below detection limits (0.1-2 µg l^{-1}, most recently 0.1µg l^{-1}). There appears to be little anthropogenic contribution from these rivers (figure 31). Higher values are indicated for Holton stream in the early 1990s (no recent data) although medians do not exceed 4µg l^{-1} (equivalent to the lowest EQS for dissolved Pb which ranges between 4 and 20 µg l^{-1} depending on hardness). There are no unequivocal temporal trends for Pb because many of the data involve detection limit-derived values.

An estimated 60kg Pb was discharged to the Harbour from STW in 2001 (compared to a riverine flux of almost 300kg, assuming an average total Pb concentration of 1µg l^{-1}). The annual mean Pb concentration in Poole STW final effluent in 2001 was 1.8± 1.4 µg l^{-1} (max 6µg l^{-1}).

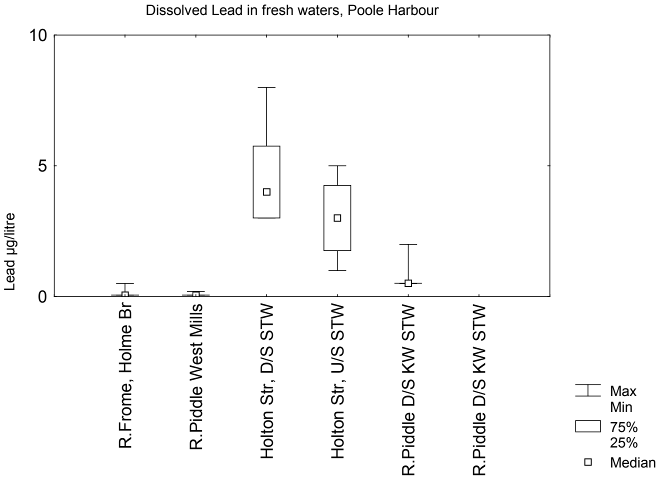

Figure 31. Concentrations of dissolved Pb (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Holton Heath stream and Keysworth (KW) STW (1993). Data source EA.

The distribution of Pb at estuarine water sites in of Poole Harbour is shown in figure 32, though little can be gleaned from this data regarding sources or trends because the majority (~90%) of values are reported as below detection. Nevertheless, by incorporating ½DL values, the plots indicate that average Pb concentrations throughout Poole Harbour are invariably more than an order of magnitude below the EQS (25 µg l^{-1}) and therefore probably of little concern for biota.

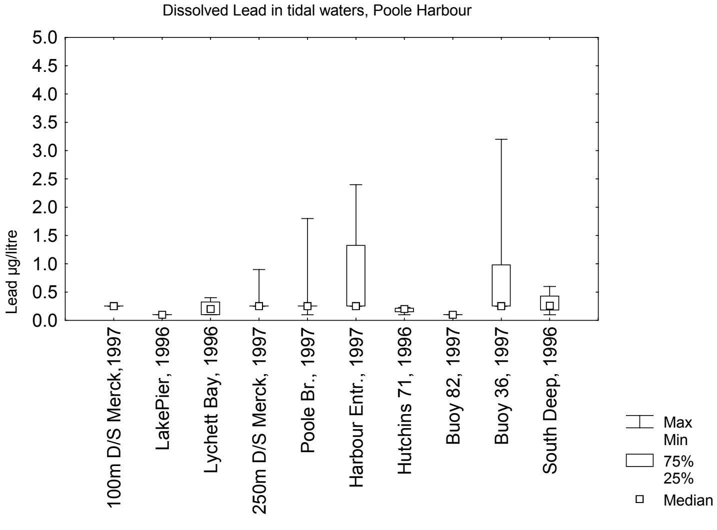

Figure 32. Concentrations of dissolved Pb (µg l^{-1}) in estuarine waters, Poole Harbour. Annual summary statistics 1996-7. Data source EA.

Zinc

The only freshwater sites with substantial data-sets for dissolved Zn are the River Frome (at Holme Bridge) and the River Piddle (at West Mills and downstream of the Keysworth STW). Median values, as shown in figure 33, are of the order of 10 µg l^{-1} (slightly higher downstream of the STW).

The EQS (based on total Zn) in fresh waters (suitable for salmonid fish), ranges from 8 to 120 µg l^{-1} depending on hardness. Since both the Frome and Piddle are relatively hard waters a likely standard value would be towards the upper end of the range – i.e. 75µg l^{-1}). The annual average total Zn at Holme Bridge (R Frome) in 2001 was 13.3µg l^{-1}, and at West Mills (Piddle), 14 µg l^{-1} – both slightly higher than dissolved Zn concentrations, but both compliant with an EQS of 75µg l^{-1}.

In 2001 mean total Zn concentrations in the Corfe and Sherford Rivers were 7µg l^{-1} and 12 µg l^{-1}, respectively. In view of their smaller size, inputs of Zn will be scaled down accordingly.

There are no observed temporal trends for Zn in any these rivers which appear to be a constant low-level source: an approximation based on an average Zn concentration of 13 µg l^{-1}, and total riverine flow of 9.3 m^3s^{-1} implies an annual input of almost 4000 kg.

Figure 33. Concentrations of dissolved Zn (µg l^{-1}) in freshwaters feeding Poole Harbour. Data are for 2001 except Keysworth (KW) STW (1994). Data source EA.

The pattern of dissolved Zn concentrations in estuarine waters of Poole Harbour is shown in figure 34. Median values for all sites appear to fall below the current Zn standard (though the EQS of 40µg l^{-1} is for *total* Zn, with possible revision to 10µg l^{-1}). It is therefore unlikely that Zn would represent an acute threat to most marine biota at the site. However median values at Lake Pier, the entrance to Lytchett Bay, and

Poole Bridge are elevated (~15µg l^{-1}), and would not meet the proposed[1] revised standard, indicating sources in this region, and a potential threat to sensitive species. Some Zn discharges into Holes Bay are indicated in Annex 7. The annual mean Zn concentration in Poole STW final effluent in 2001 was 69 ± 59µg l^{-1} (max 347µg l^{-1}), equivalent to an input of approximately 1000kg annually (compared to the to a riverine flux of 4000kg). There could also be a small Zn input to the water column from sediments.

Other possible sources of Zn in the Harbour include marinas and the ferry terminal. Zn pyrrithione is now used as booster biocide to replace TBT in antifouling paints and the use of sacrificial anodes on boats and structures, to reduce corrosion, can be expected to lead to a measurable increase in concentrations of dissolved Zn in marinas and nearby estuarine waters. Studies by Bird *et al*., (1996) which incorporated sites in Poole Harbour, have indicated concentrations may increase up to 21µg l^{-1} in enclosed marinas (compared to mean concentrations of around 2 µg l^{-1} in the Wareham Channel and Sandbanks). Zn concentrations in marina sediments may also be double that in adjacent 'control' sediments (representing an increase of up to ~100 µg g^{-1} and taking them above probable effects levels in sediment –see section 8). Estimated amounts of Zn used annually in some marinas (of the order of 1000kg where anodic Zn is used on marina structures) are equivalent in scale to that discharged to Holes Bay in 2001 from STW.

No distinctive long-term trends can be seen for dissolved Zn in tidal waters.

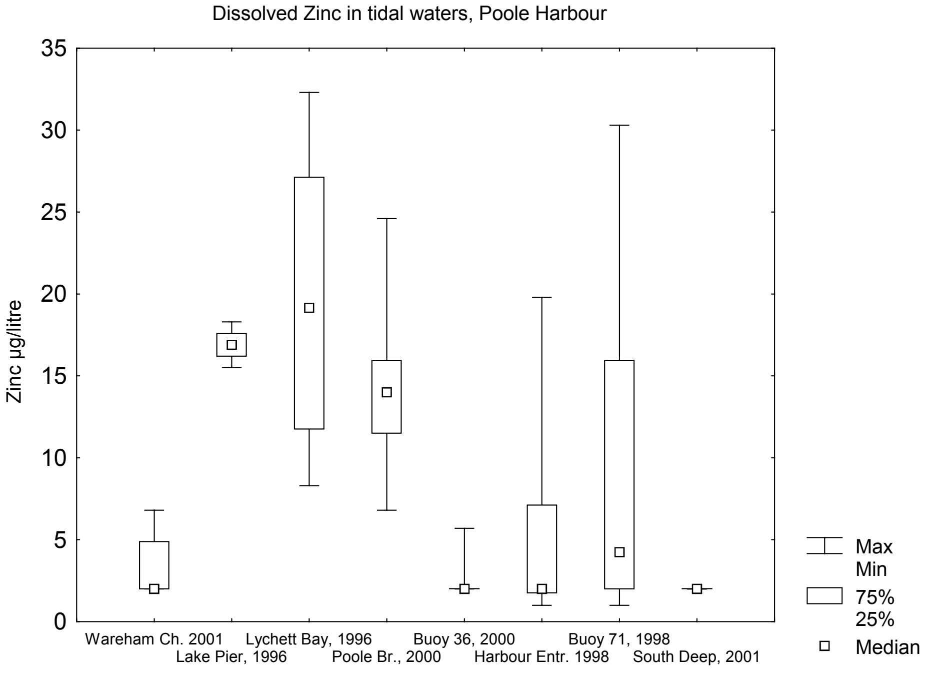

Figure 34. Concentrations of dissolved Zn (µg l^{-1}) in estuarine waters, Poole Harbour. Annual summary statistics. Data source EA.

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[1] Following a review of more recent toxicity data, Hunt and Hedgecott (1992) proposed a more stringent EQS to DoE of 10 µg l^{-1}, based on the lowest, most reliable NOECs (7 – 20µg l^{-1}) though this has yet to be adopted.

Mercury

The EQS for Hg in all fresh waters, based on total metal, is 1µg l^{-1}. The only riverine source where extensive data for total Hg are available are for the River Frome at Holme Bridge (figure 35). Only two values exceed 1µg l^{-1} (none since 1997). Annual averages fall significantly below the EQS here (the majority of samples are in fact recorded as below the limit of detection and comply with the Hg standard by almost two orders of magnitude). At West Mills on the River Piddle the annual average for 2001 (0.008 µg l^{-1}) was also significantly below the EQS.

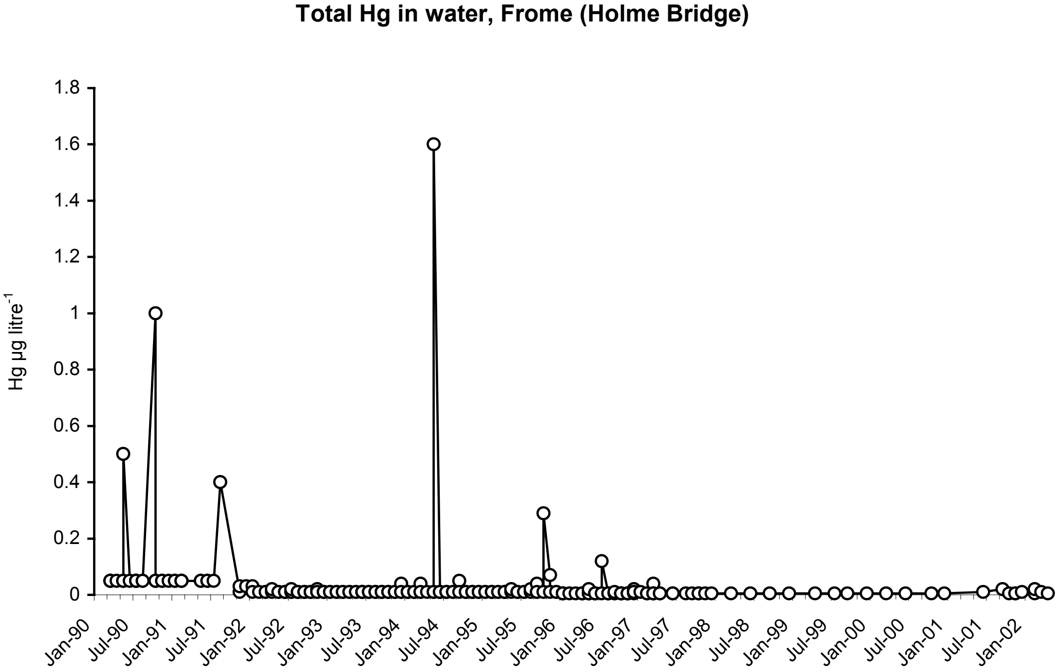

Figure 35. Concentrations of total Hg in the River Frome Holme Bridge. Data source EA.

The pattern of dissolved Hg in estuarine water for six sites are plotted in figure 36. Annual averages are invariably below the EQS (0.3 µg l^{-1}) by almost two orders of magnitude. The extreme Hg value at South Deep is considered an outlier. Elsewhere concentrations were highest at Poole Bridge reflecting potential sources from Holes Bay. Discharge data (annex 7) indicate small inputs from Poole STW and, previously, Merck. In 2001 STW discharged an estimated 0.42kg Hg (compared to an estimated riverine flux of 2.58kg, assuming a riverine concentration for total Hg of 0.009µg l^{-1}). The annual mean Hg concentration in Poole STW final effluent in 2001 was 0.018 ± 0.011 µg l^{-1}.

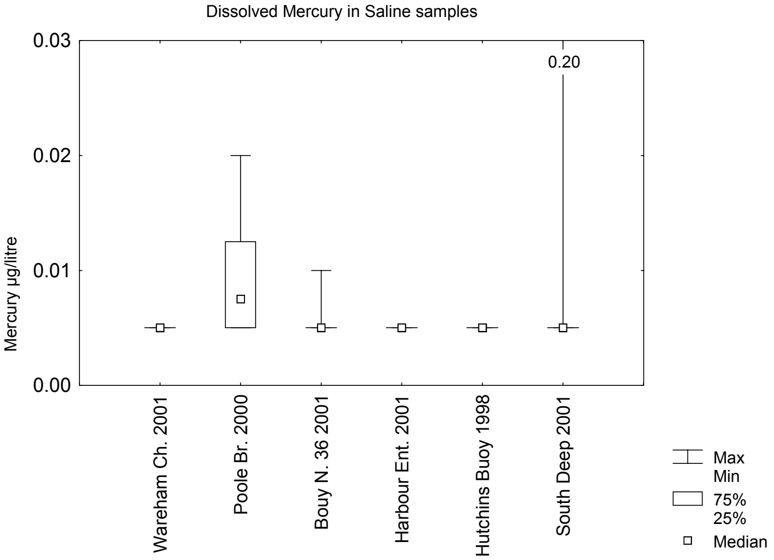

Figure 36. Summary statistics for concentrations of dissolved Hg (µg l^{-1}) in estuarine waters Poole Harbour. Data source EA.

Detection limits for Hg in water have been declining over time masking any genuine long-term temporal trends for Hg.

For the sites monitored dissolved Hg poses little threat. Hg however has a strong affinity for particulate matter and the sediment data shown in sections 5 and 8 indicate there are hotspots for this metal, particularly in Holes Bay.

Vanadium and Boron

These list two metals are not usually considered as causing problems in the marine environment but have been included in the discussion since some information is available for fresh-waters and trade effluents. There are, however, no data for tidal waters.

Boron is used in fire retardants, as a component of enamels, and in the photographic, cosmetic, leather, paint, textile and wood-processing industries. Borax, a major boron compound, is used as a cleaning compound and may occur in domestic and/or industrial effluents. Because of its wide variety of industrial uses, there are many potential pathways for entering the aquatic environment.

The EQS for dissolved B in fresh waters is 2000µg l^{-1} (annual average). Concentrations in the River Frome at Holme Bridge and the River Piddle at West Mills were invariably below detection limits of 100µg l^{-1}. The coastal and estuarine EQS for boron is 7mg l^{-1} (derived by applying an arbitrary factor of 10 to the lowest, rather limited toxicity data). This equates to a level approximating to normal ambient concentrations of boron in seawater. The highest values in effluent samples were all below this concentration, therefore even within the mixing zone it is unlikely that EQS would be exceeded.

Vanadium is used in non-ferrous metals and widely as a catalyst in the chemical industry, especially for the production of sulphuric acid and plastics. Small quantities of vanadium are used in a variety of other applications. Burning of fossil fuels probably causes the most widespread discharge of vanadium into the environment.

The EQS for dissolved V in fresh waters is in the range 20-60µg l^{-1} (hardness related). Concentrations in the River Frome at Holme Bridge and the River Piddle at West Mills were invariably below detection limits of 2µg l^{-1} and therefore not considered a problem. The coastal and estuarine EQS for vanadium is 100 µg l^{-1}. The highest values in trade effluent samples have exceeded the EQS value on occasions (table 21) though outside the immediate mixing zone it is unlikely that EQS would be exceeded. The bioconcentration factor for this relatively insoluble metal (and boron) in most organisms is relatively low (<100)[1] therefore neither bioaccumulation nor toxic effects would seem likely. However, it may be advisable to investigate this further due to the occasional high concentration in trade discharges.

Table 21. Highest values for Vanadium in discharges to Poole Harbour. (Data source EA).

Site	Date	vanadium µg l^{-1}
PILKINGTON HOLES BAY	07/05/92	1,000
PILKINGTON HOLES BAY	18/06/99	580
PILKINGTON HOLES BAY	06/05/94	400
PILKINGTON HOLES BAY	30/08/00	169
PILKINGTON HOLES BAY	25/08/99	140
PILKINGTON HOLES BAY	05/05/98	120
PILKINGTON HOLES BAY	09/05/96	110
MERCK LTD COOLING WATER	02/10/91	101

7.1.2 Organotins

There are no data for tributyltin (TBT) and triphenyltin (TPT) in rivers or effluent streams entering the Harbour. Analysis for both compounds was performed on tidal water samples from Poole Bridge and the Harbour mouth on 14-15 occasions between 1993 and 1995. Concentrations were consistently below detection limits of 28ng l^{-1} and 13 ng l^{-1}, respectively (apart from one anomalously high value of 123 ng TBT l^{-1} at Poole Bridge). It should be noted that these detection limits are higher than the EQS for TBT and TPT (of 2 and 8 ng l^{-1}, respectively, as a maximum concentration). It is therefore not possible to make an assessment of risk from organotins based on this data. More work should be done to resolve the current status, and to follow up some of the *ad hoc* trend data described in section 5.1.

7.1.3 Pesticides and Herbicides

Data for pesticides and herbicides in tidal waters of the SPA are relatively few in number.

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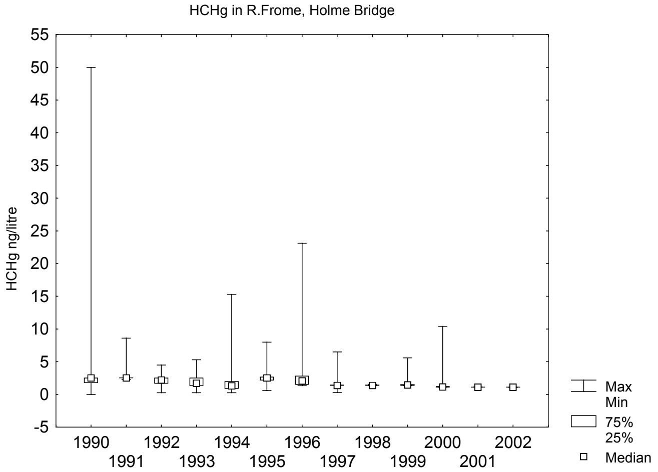
[1] Ascidians and holuthurians are somewhat exceptional in accumulating V for essential purposes NB as a blood pigment

Hexachlorocyclohexanes - γ-HCH (lindane) and other isomers.

The EA database contains information on total hexachlorocyclohexanes (HCH), and individual isomers notably γ-HCH (lindane). γ-HCH is principally discussed here as representative of the distribution of this group of pesticides. Since half the recorded values are below detection limits, data interpretation should be viewed with caution.

Summary statistics for concentrations of γ-HCH in the major rivers entering the SPA (Frome and Piddle) are shown in figure 37. Annual median values, were consistently less than 3ng l^{-1} – below the EQS of 100 ng l^{-1} for fresh-water. Total HCH values in the Frome and Piddle were of a similar order, eg ~ 2.5 ng l^{-1} in 1999. Individual samples may contain higher concentrations of γ-HCH, though incidences of these are rare and appear to be declining. The use of lindane is currently being phased out in Europe following an EU decision in 2000 to ban it, however its use on food crops imported from other counties could result in γ-HCH residues in sewage effluent. There is also the possibility of illegal usage.



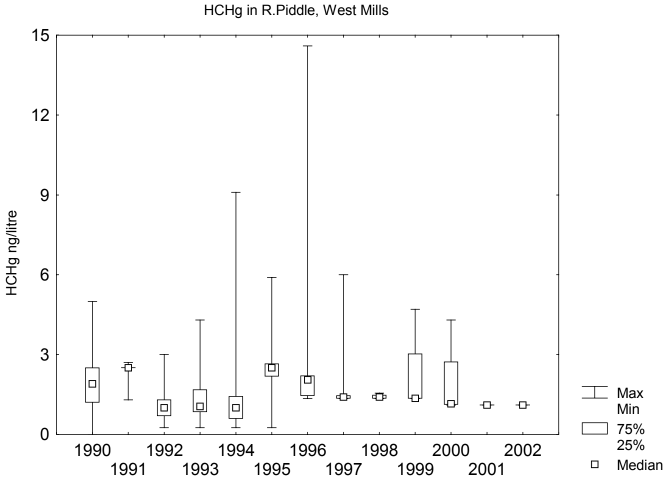


Figure 37. Concentrations of γ-HCH in the Rivers Frome and Piddle; summary statistics 1990-2002. Data source EA.

γ-HCH concentrations in trade discharge samples are only for one site in the data set provided (Merck, West Quay; now ceased). For information, available summary statistics for this site are shown in figure 38. At this particular location, we can speculate that risks to biota are likely to have been low, since apart from 1991, γ-HCH concentrations in the discharge were below the standard for the receiving water.

It would appear from the discharge data (annex 7) that small amounts of γ-HCH (and other isomers) may enter Holes Bay via STW. The annual mean concentration in Poole STW final effluent in 2001 was 12.3 ng l^{-1}. Levels of other isomers of HCH (α,β,δ,ε) were generally at or below detection limits of ~5ng l^{-1}. Summed together however HCH (all isomers) concentrations in the discharge averaged 33 ng l^{-1}± 9.8 ng l^{-1}. The EQS for marine waters is 20ng l^{-1}.

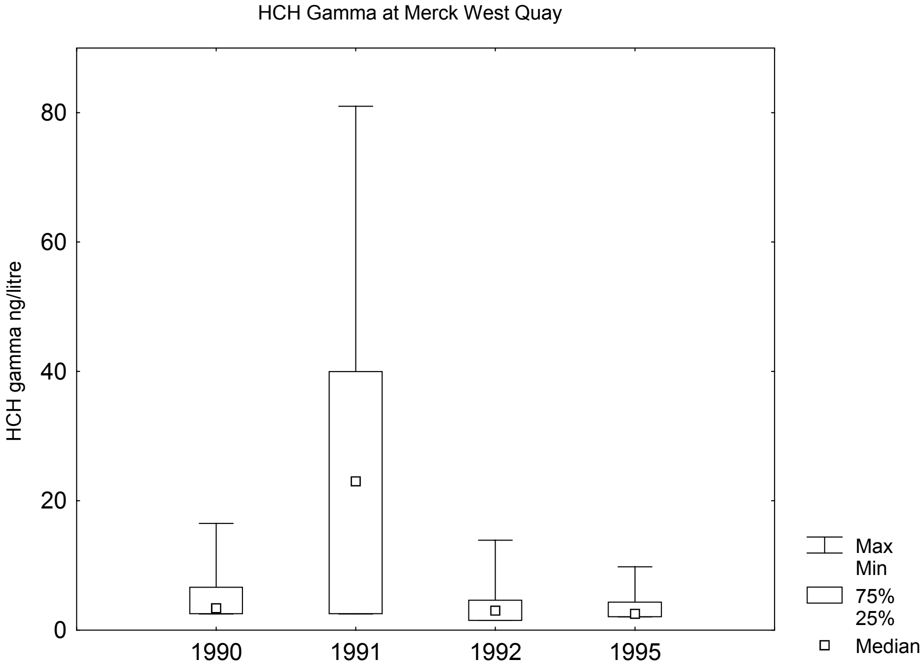


Figure 38. γ-HCH **concentrations in trade discharge, Holes Bay (most recent available annual averages) .**

Overall distributions of γ-HCH in tidal waters of Poole Harbour are shown in figure 39, Median values are below the EQS. Highest concentrations are restricted to one or two values at the Poole Bridge site presumably reflecting inputs to Holes Bay. On this evidence however the threat from γ-HCH appears relatively small.

The data for total HCH in tidal waters are few in number but the relatively elevated levels at Poole Bridge appear to confirm the input from Holes Bay (figure 40).

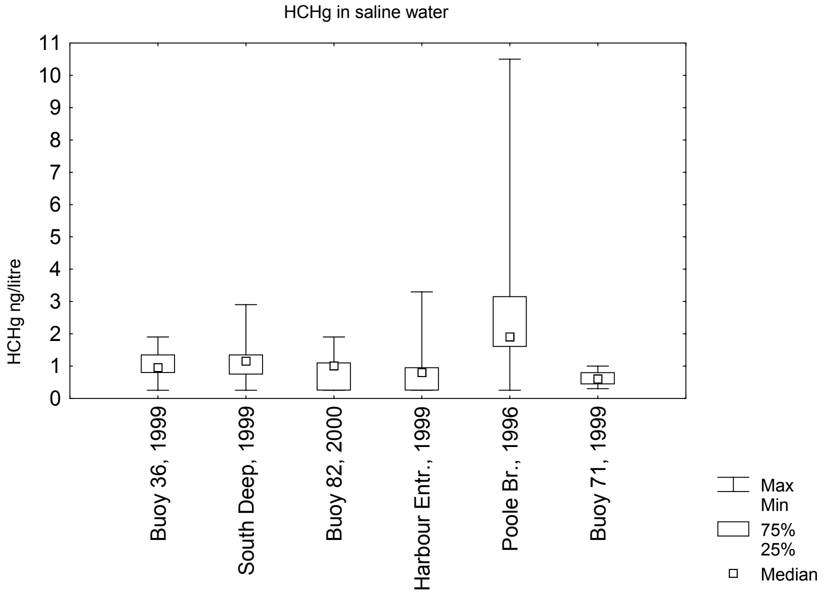


Figure 39. Lindane (γ-HCH) in estuarine waters Poole Harbour. Data source EA.

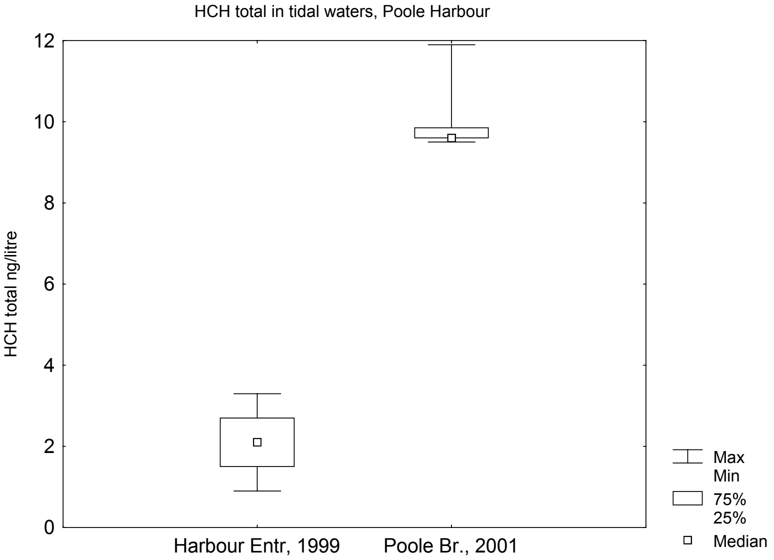


Figure 40. HCH total (all isomers) in estuarine waters Poole Harbour. Data source EA.

There is an indication of a decreasing trend for γ-HCH in estuarine waters in recent years, though this is somewhat equivocal due to the large number of values below (variable) detection limits.

DDT

Over the last decade, concentrations of para, para-DDT (pp-DDT) in freshwaters entering the SPA via the Frome and Piddle have, for the majority of samples (99%), been below detection limits (1ng l^{-1} in 2001 - an order of magnitude below the EQS of 10ng l^{-1}). Maximum measured values are 15.7 ng l^{-1} at Holme Bridge and 4.8 ng l^{-1} at West Mills. All values for op-DDT were below the limit of detection. On this evidence, riverine sources of DDT do not appear to be a significant issue. This is evidently also the case for trade and sewage discharges which appear to be negligible (annex 7). Concentrations in Poole STW final effluent in 2001 were consistently below DL (calculated mean 2.55 ng l^{-1} based on ½ DL). As a result impact on the marine site is minimal: none of the tidal waters sampled during the last decade had concentrations above detection limits (currently 0.4ng/l).

Dieldrin

Over the last decade, concentrations of dieldrin in freshwaters entering the SPA via the Frome and Piddle have mostly (95%) been below detection limits (0.3 – 5 ng l^{-1}, most recently ~1 ng l^{-1}) – i.e. below the EQS of 10 ng l^{-1} (fresh- and salt-water). Those values above detection limits were in the range 0.6-2.3 ng l^{-1}. On this evidence, riverine sources of dieldrin are probably not a significant issue. This is also the case for monitored trade and sewage (Poole STW) discharges, which appear to be negligible (annex 7). Concentrations in Poole STW final effluent in 2001 were, with the exception of one sample, below DL (calculated mean 2.85 ng l^{-1} based on ½ DL). As a result, impact on the marine site is likely to be minimal; none of the tidal waters sampled during the last decade had concentrations above detection limits (0.5 - 5ng/l, most recently 1 ng l^{-1}).

Aldrin

Over the last decade, concentrations of aldrin in freshwaters entering the SPA via the Frome and Piddle have been below detection limits (1ng l^{-1} in 2001 - an order of magnitude below the EQS of 10ng l^{-1}). On this evidence, riverine sources of aldrin are not a significant issue. This is also the case for trade and sewage (Poole STW) discharges, which appear to be negligible (annex 7). Concentrations in Poole STW final effluent in 2001 were consistently below DL (calculated mean 2.5 ng l^{-1} based on ½ DL). As a result, impact on the marine site is minimal; none of the tidal waters sampled during the last decade had concentrations above detection limits (currently 0.5-1ng/l).

Endrin

Over the last decade, concentrations of endrin in freshwaters entering the SPA via the Frome and Piddle have been below detection limits (0.2 –5 ng l^{-1}, most recently ~1 ng l^{-1}) – i.e. below the EQS of 5 ng l^{-1} (fresh- and salt-water). On this evidence, riverine sources of endrin are probably not a significant issue. This is also the case for monitored trade and sewage (Poole STW) discharges which appear to be negligible (annex 7). Concentrations in Poole STW final effluent in 2001 were consistently below DL (calculated mean 2.7 ng l^{-1} based on ½ DL). As a result, impact on the marine site is likely to be minimal; none of the tidal waters sampled during the last decade had concentrations above detection limits (0.5-5ng/l, most recently 1 ng l^{-1}).

Endosulphan (A and B)

Over the last decade, concentrations of endosulphan (A and B) in freshwaters entering the SPA via the Frome and Piddle have been below detection limits (0.6-6.5 ng l^{-1} for Endosulphan A; 1-15 ng l^{-1} for Endosulphan B - compared with an EQS of 3 ng l^{-1}). The most recent data (2002) - for Endosulphan B at Holme Bridge on the Frome - indicate values <2ng l^{-1}. On this evidence, riverine sources of Endosulphan are probably not a significant issue, though an accurate assessment is clearly not possible since detection limits are sometimes above the EQS.

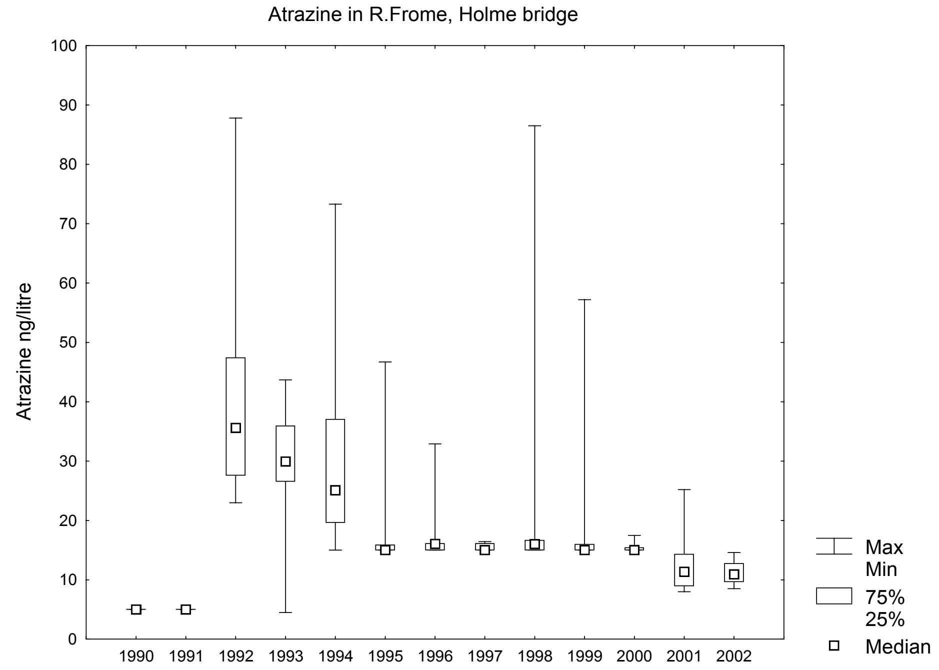
This is also the case for trade and sewage discharges. The only information is for Poole STW final effluent where the annual average concentration for Endosulphan B in 2001 was 3.9 ng l^{-1} (based on ½ DL; all values below DL). Inputs from this source appear to be negligible, consistent with discharge figures for 2000 (annex 7).

Up to the mid 1990s when routine endosulphan A monitoring of tidal waters appears to have ceased, virtually all values were below detection limits (0.6-6.5 ng l^{-1} for Endosulphan A; 1.5- 13 ng l^{-1} for Endosulphan B – compared with an EQS of 3 ng l^{-1}). The most recent of these data indicate values were < 2 ng l^{-1}. It is likely that impact on the marine site is minimal though, again, an accurate assessment is not possible since detection limits are sometimes above the EQS.

Atrazine and Simazine

Atrazine and Simazine are *s*-triazine herbicides and are on the UK red list of toxic compounds with a combined EQS of 2µg l^{-1} (annual average) or 10 µg l^{-1} (maximum allowable concentration).

Summary statistics for atrazine are shown in figure 41. The majority of values for both the Frome and Piddle are below detection limits (76%). Both data sets are typified by occasional elevated values (more frequent in the Frome) though these appear to be decreasing in frequency. The high value for 1993 in the Piddle represents a single value and may be an outlier.



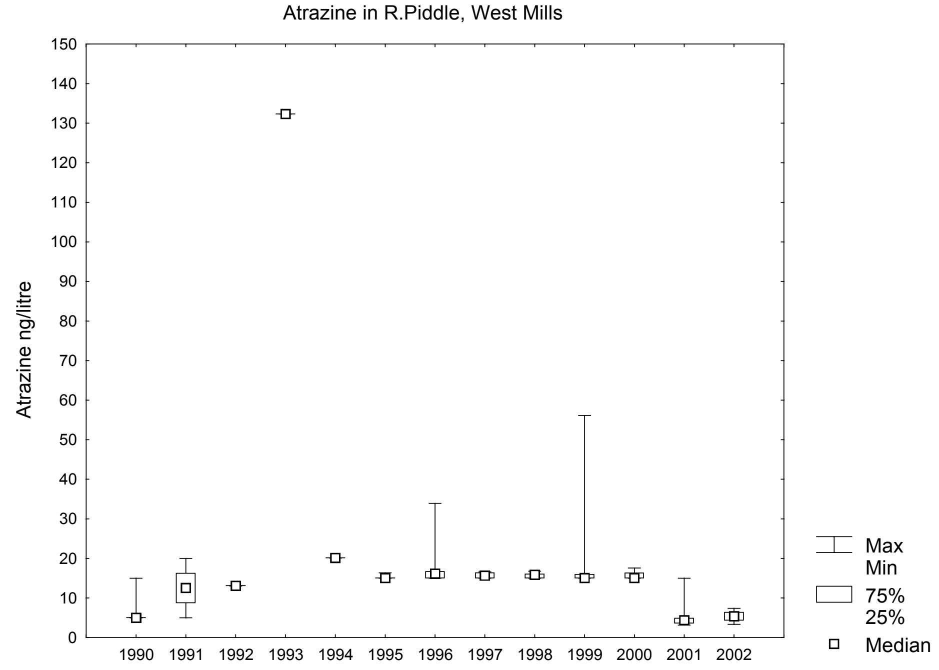


Figure 41. Atrazine in Rivers Frome and Piddle. Summary statistics. (EA data)

Summary statistics for simazine are shown in figure 42. The majority of values for both the Frome and Piddle are again below detection limits (87%). Both data sets are typified by occasional elevated values (more frequent in the Frome, peaking in 1997) though these appear to be decreasing in frequency.

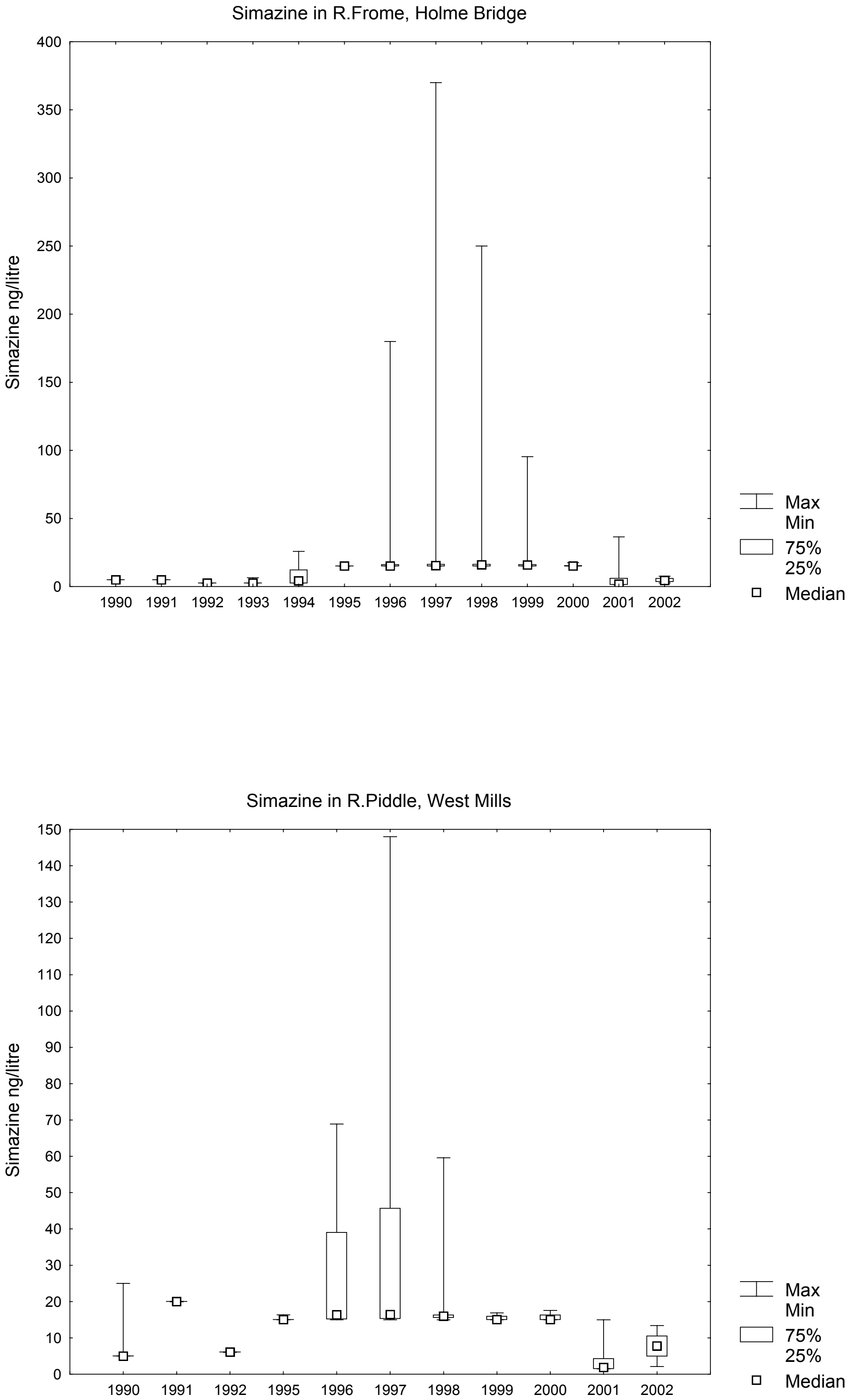


Figure 42. Simazine in Rivers Frome and Piddle. Summary statistics. (EA data)

Though atrazine and simazine are present in these systems, sporadically at high levels perhaps following agricultural application, exceedences of the EQS (2 µg l^{-1}, annual) did not occur, even when the two are summed. The most recent annual averages (2001) for atrazine were 12.6 ± 5.1 ng l^{-1} and 5.5± 3.89 ng l^{-1} for the Frome and Piddle, respectively. Corresponding means for simazine were 7.1±11.2 ng l^{-1} and 4.2±4.8 ng l^{-1}. Maximum concentrations of each of these herbicides in 2001 were in the range 30-40ng l^{-1} for both rivers.

It seems unlikely, therefore, that triazine herbicides represent a significant threat to the marine site. Unfortunately, no data could be found for tidal waters to test this assertion.

Organophosphate insecticides

Samples from the River Frome and Piddle have been analysed for several OPs including azinphos (-ethyl and-methyl) fenitrothion, malathion and parathion (-ethyl and-methyl). Virtually all samples were below detection limits (see table 22). In earlier years these DLs were sometimes above EQS values, expressed as annual averages. For the most recent samples however, detection limits have been well below the EQS indicating riverine inputs are unlikely to be of biological significance. Since these are likely to be the only sources, the threat to the from OPs seems negligible.

Table 22. Organophosphates in Rivers Frome and Piddle

OP	**River**	**values below DL**	**DL range 1990-2002* (ng l^{-1})**	**EQS ng l^{-1}** **annual mean**	 **maximum**
parathion-ethyl	Frome	100%	4*- 22	ns	ns
parathion-methyl	Frome	100%	3* -18	ns	ns
azinphos-ethyl	Frome	100%	6* - 52	ns	ns
azinphos-methyl	Frome Piddle	100%	3* - 51	10	40
malathion	Frome Piddle	99%	2* - 20	10	500
fenitrothion	Frome Piddle	100%	1* - 50	10	250

*lower detection limits (DL) apply to the latest (2001/2002) data

ns – no standard applicable

7.1.4 PCBs

PCB 153, typically one of the most abundant PCBs in environmental samples, was used as being indicative of distributions of this group of organochlorines. This congener was analysed in freshwater samples from the River Frome at Holme Bridge between 1990 and 1994. All samples were below detection limits (<5ng l^{-1}) implying little acute threat.

There are very few other data for PCBs in water to assess site characteristics or spatial trends. In view of their affinity for, and persistence in, sediments, information on PCBs in this phase is probably the most appropriate for such an assessment. However at present sediment PCB data are also extremely limited.

7.1.5 Pentachlorophenol

Pentachlorophenol, used as a wood preservative and pesticide, is a List I compound with an EQS of 2µg l^{-1} (annual average) in all waters.

More than 90% of values for the Rivers Frome and Piddle were below detection limits. These limits have ranged from 0.04- 0.1µg l^{-1} in recent years. A similar proportion of values were below detection limits in tidal waters (ranging from 0.005-0.1µg l^{-1}). Thus, although occasional exceptional values of up to 0.8µg l^{-1} occur in the database (Poole Bridge) the majority of samples are substantially below the EQS. Small amounts are present in consented STW discharges (annex 7) and, historically, trade discharges from chemical manufacturers in Holes Bay.

7.1.6 Hydrocarbons (Oil, PAHs)

There are no EQS values for hydrocarbon oils in estuarine waters *per se*. Two directives list criteria which can be used as general guidance; the Bathing Waters Directive, under the heading organic substances: 300µg l^{-1} as the 90th percentile (non-routine sampling prompted by visual or olfactory evidence of hydrocarbon presence); and the Shellfish Waters Directive listed under organic substances, which states that 'hydrocarbons must not be present in such quantities as to produce a visible film on the surface of the water and/or a deposit on the shellfish, or to have harmful effects on the shellfish'. Also under the Shellfish Waters directive, hydrocarbon contamination is (presumably) included in 'general physico-chemical parameters' – tainting substances – where 'the concentration of substances affecting the taste of shellfish must be lower than that liable to impair the taste of the shellfish'.

These EQS guidelines for Shellfish waters are obviously difficult to quantify, however tainting (an odour or flavour foreign to the product) can occur in commercial species contaminated with crude and refined oils. Species with a high body fat content such as salmon or herring are more easily tainted and retain the taint for longer than lean-muscle species. GESAMP (1993) report studies detecting taints in fish and macro-crustaceans resulting from exposure during acute incidents, chronic discharges and in experimental studies. There are no accepted permissible standards in organisms. In some instances, hydrocarbons may be present at well above background levels, even though no taint can be detected. Conversely fish can be tainted where analysis indicates that contamination is only at background levels. Experimental studies indicate that taints can be detected when fish are exposed to concentrations of oil in water in the range 0.01 to 1mg l^{-1}. Tainting can occur very rapidly on exposure (within a few hours at concentrations of oil above 1 mg l^{-1}), and fish have been shown to lose their taint within 1 to 4 days (experimental study on cod). However, field studies have indicated that fish may still be tainted days or weeks after a spill of fuel oil (GESAMP 1993). Because fine sediments absorb and retain oil, infaunal species such as clams and burrowing shrimps, and some demersal fish may be at risk of tainting on a more prolonged basis.

The EA database provided no information on 'hydrocarbon oils' for rivers or estuarine waters in the SPA. A small number of discharge water samples relating to the Wytch Farm gathering station are shown in figure 43. Their significance is difficult to assess with respect to the above guidelines. There are occasional high values, as might be anticipated from process waters, though median values suggest they are likely to be of little significance to the SPA as a whole.

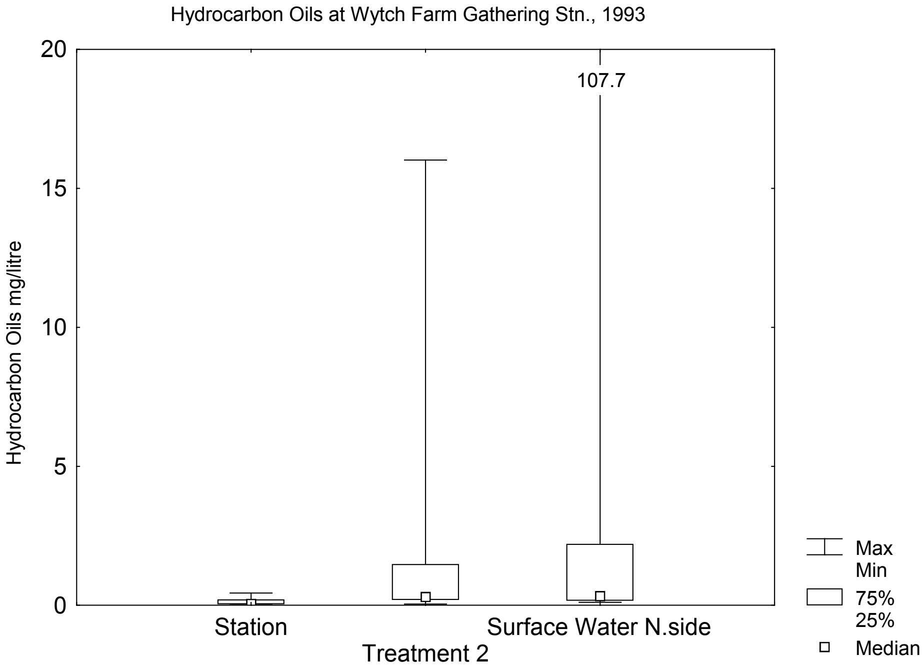


Figure 43. Hydrocarbon oils in discharge and process waters, Wytch Farm gathering station, 1993. (Data source EA).

The only other data for hydrocarbons relates to Hamworthy Stream, presumably subjected to a discharge prior to 2000, when relatively high levels were present (figure 44). Since then concentrations have declined substantially and are currently at, or close to, the detection limit of 0.2mg l^{-1}.

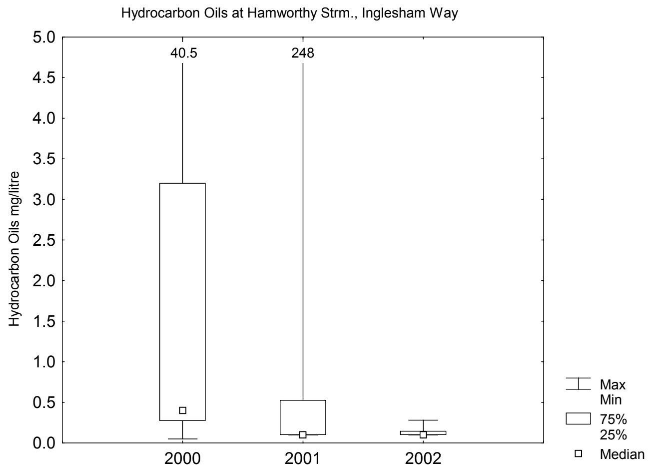


Figure 44. Hydrocarbon oils (mg l^{-1}) in Hamworthy Stream. Data source EA.

In the absence of recent data for concentrations in waters of the SPA, we cannot comment, realistically, on the current water quality status with regard to hydrocarbons or their potential impact on estuarine biota and shellfish beds.

Likewise, there is no information in the EA data-set for total PAH concentrations in water or discharges in Poole Harbour.

Data on a number of individual PAHs were extracted from the database to try and indicate distributions and trends: these included benzo(k)fluoranthrene and benzo(ghi)perylene. However, the only data available concerns freshwater samples from the Frome and Piddle, taken in 1997: mean concentrations were 2.97 ng l^{-1} and 1.19 ng l^{-1}, respectively, for benzo(ghi)perylene, and 1.57 ng l^{-1} and 0.8 ng l^{-1} for benzo(k)fluoranthrene. These are considered to be of little biological significance[1]. No data could be found for PAHs in tidal waters therefore it is not possible to characterise the impact of PAHs in waters across the Harbour.

7.1.7 Volatile organics (solvents, freons).

Trichloromethane (chloroform)

Chloroform is a List I compound with an EQS of 12µg l^{-1} (annual average) in all waters.

Summary statistics for fresh-water are shown in figure 45. The majority of values for chloroform were below detection limits (>75%). These DLs have ranged from 1-0.1µg l^{-1} over the last decade, and most recent values are at the lower end of this

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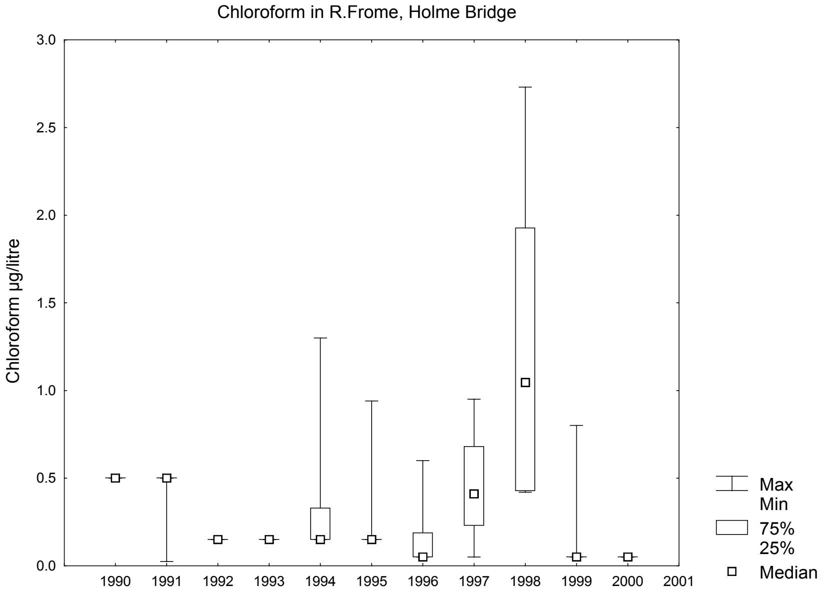
[1] Maximum permissable concentrations in Dutch waters are 100 ng l^{-1} and 20 ng l^{-1} for benzo(k)fluoranthrene and benzo(ghi)perylene, respectively. There are no UK standards for these compounds. The EQS for naphthalene is 500ng l^{-1} annual average; 80µg l^{-1} maximum.

range. Both rivers are typified by occasional elevated values (notably in the Frome, peaking in 1998). However concentrations are invariably below the EQS implying riverine sources are not a major threat.

Some discharges of chloroform occur from STW (annex 7) though concentrations in effluent from Poole STW in 2001 (0.61 ± 0.52 µg l^{-1}) do not appear to be a direct toxicological concern.



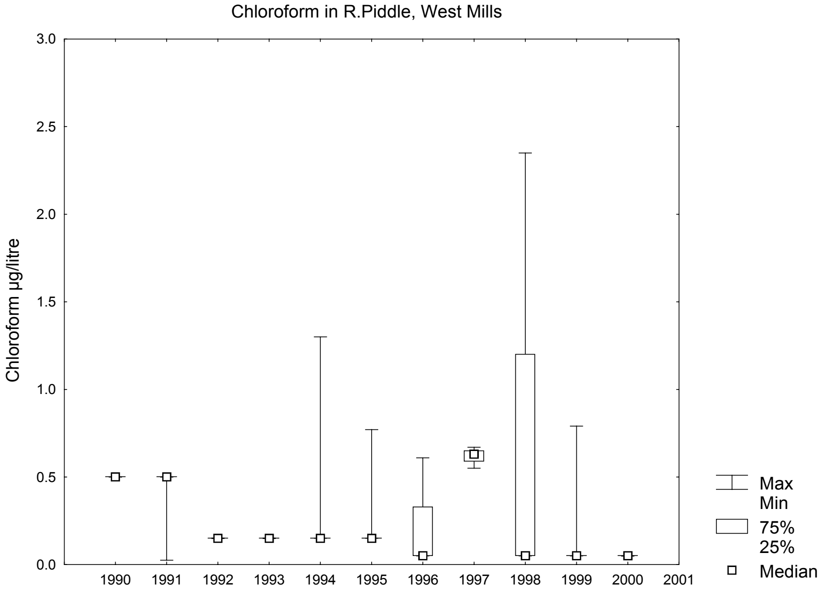


Figure 45. Chloroform in Rivers Frome and Piddle. Summary statistics. Data source EA

The pattern of chloroform concentrations in estuarine water at four sites in the outer Harbour are plotted in figure 46. Occasionally concentrations were highest at Poole Bridge reflecting potential sources from Holes Bay. However annual averages were invariably quite low and below the EQS (12 µg l^{-1}) at all sites.

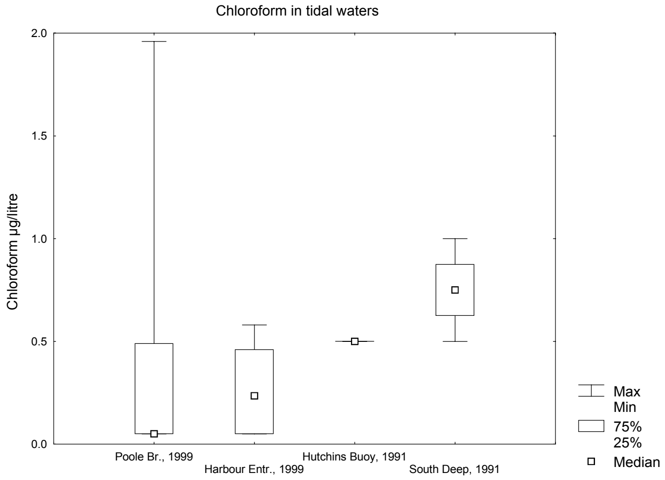


Figure 46. Chloroform in tidal waters Poole Harbour. Summary statistics. (EA data)

Tetrachloromethane (carbon tetrachloride)

Carbon tetrachloride is another List I compound, also with an EQS of 12µg l^{-1} (annual average) in all waters.

More than 95% of values for fresh- and tidal-waters, at the sites described above for chloroform, were below detection limits. These limits have ranged from 1- 0.05µg l^{-1} over the last decade: most recent values are at the lower end of this range. Concentrations are invariably below the EQS, usually by more than an order of magnitude, implying little threat.

Trichloroethene (trichloroethylene)

Trichloroethene, another List I compound, has an EQS of 10µg l^{-1} (annual average) in all waters.

More than 95% of values for fresh- and tidal-waters at the sites described above, were below detection limits. These limits have ranged from 0.1- 0.05µg l^{-1} over the last decade. Concentrations are invariably below the EQS, usually by more than an order of magnitude implying little threat. Some discharges of trichloroethylene occur from STW (annex 7) though concentrations in effluent from Poole STW in 2001 (0.17 ± 0.9 µg l^{-1}) do not appear to be a direct toxicological concern.

Trichloroethane

1,1,1 Trichloroethane is a List II compound with an EQS of 100µg l^{-1} (annual average) in all waters.

More than 90% of values for fresh- and tidal-waters were below detection limits. These limits have ranged from 0.3- 0.1μg l^{-1} over the last decade. Concentrations are invariably below the EQS, usually by almost two orders of magnitude.

It seems unlikely, therefore, that volatile organics represent a significant threat to the marine site.

7.2 Non-Toxic Contaminants

7.2.1 Nutrient Quality Criteria.

Nutrient concentrations vary with salinity, therefore measurements collected simultaneously from different regions within the estuary, or from the same region but at different states of the tidal cycle, may show considerable differences and not be truly representative of water quality. To compound this difficulty, nutrient concentrations also vary throughout the year with freshwater flow. As yet there are no statutory water quality standards for nutrients in the UK and determination of the nutrient status of estuaries, and the ecological consequences, remain a notoriously contentious issue. To quote from the Agency's Technical Guidance for Water Quality: Review of Permissions to Discharge and New Applications (Habitats Directive) - 'Generally, it is impossible to calculate permit conditions in the absence of water quality standards…' and ' it is not easy to make a case or refuse or reject an application in the absence of such standards'. Therefore, judgement of nutrient status in Poole Harbour, as elsewhere, consists largely of subjective assessment of monitoring information concerning the primary variables, coupled with contextual information on the site characteristics and condition. The primary variables are generally considered to be nitrogen and phosphorous (though there is still great scientific debate as to which forms to measure). It is usually considered essential to monitor these parameters alongside initial biological response indicators such as chlorophyll-a (a measure of primary production), dissolved oxygen and, for example, Secchi depth (a measure of turbidity). These data may then be fed into models to develop criteria for the selection of numerical water quality objectives.

Although no statutory standards exist for N and P in estuarine and marine SACs or SPAs, a number of 'guideline values' have been established which could be of relevance for assessment of the status of nutrients in the catchment of Poole Harbour and for initiating management responses:

- EU nitrates directive 91/676/EEC, on the protection of all waters against pollution caused by nitrates from agricultural sources, calls for the identification of all waters that contain **50mg l^{-1} nitrate**.
- The USEPA is still in the process of arriving at their national nutrient strategy but has for many years proposed a limit of **10mg l^{-1} nitrate** nitrogen for the protection of domestic water supplies (against over enrichment and impacts on human and animal health). A phosphorous criterion was reported some years ago in the EPA 'Red Book' as **0.1μg l^{-1} (as P)** to protect estuarine and marine organisms against

the consequences of bioaccumulation (EPA, 1976). However, this was not established as threshold for eutrophication and is currently under review.

- The North Sea Status report stated that hypernutrification in sea water exists when winter (maximum) **TIN values exceed 0.144mg l^{-1}** (provided P>0.006mg l^{-1}), implying that nutrient concentrations need not be elevated by a large margin before algal proliferation commences (Parr, 1999). In estuaries however it seems likely that thresholds will be higher.

- Based on work in 2 eastern USA estuaries, Deegan *et al.,* (1997) have suggested that a DIN value of ~ **1mg l^{-1} DIN** or more might lead to poor habitat quality for fish populations, which may be due in part to cloaking effects of macroalgal mats on *Zostera* beds.

- There is a proposed EQS of **0.021mg l^{-1} un-ionised ammonia** (NH^3 N) for the protection of saltwater fish and shellfish, although due to the technical difficulties in measuring the unionised form, total ammonium is usually monitored and NH^3 calculated. However, even calculations can be difficult as the relative proportion of ionised and un-ionised ammonia depends on salinity, temperature and pH.

Recognising the dilemma in arriving at standards, there have been attempts in recent years to develop and test General Quality Assessment (GQA) schemes for nutrients in estuaries and coastal waters which may be adopted nationally and internationally. One such scheme is proposed for the EA by the WRc as part of their General Quality Assessment (GQA) scheme (Gunby *et al.,* 1995). For nitrogen, this method uses the combined concentrations of nitrate, nitrite and ammonium concentrations in tidal waters (total inorganic nitrogen, TIN), as an approximation of bioavailable nitrogen. Assuming conservative behaviour for TIN and a standard concentration in marine waters, allows the TIN concentration in the freshwater input to be calculated, provided salinity data are available. For phosphorus, Total Reactive Phosphate (TRP - phosphate in unfiltered samples) is measured and, as for nitrogen, the concentration in freshwater calculated. Estuaries are then be grouped according to the following class boundaries (table 23):

Table 23. TIN and TRP classification criteria for estuaries (based on Gunby *et al.*, 1995)

	Class	Median projected TIN (mg l^{-1})	Class	Median projected TRP (mg l^{-1})
	A/B	5.3	A/B	0.087
	B/C	8.1	B/C	0.35
	C/D	11.1	C/D	1.00
Mean for UK estuaries*	**C**	**9.5**	**D**	**2.5**

* Mean of 33 UK estuaries (based on projections by Cole *et al* 1999)

In view of the hydrodynamic differences between estuaries, together with seasonal and other site-specific factors, it is not known how these thresholds would apply to Poole, or how valid they may be. Nevertheless, in the absence of site-specific guidelines they at least represent benchmarks as to the potential threats, against which to draw comparisons. Based on these criteria, and published data from other estuaries, it is possible to attempt a brief analysis of nutrient monitoring observations supplied by the Agency including;

- determination of background (reference) values and 'hotspots' for the area
- examination of historical data and trends in Poole Harbour and comparisons with other areas
- validity of guideline values and classification schemes

We have used measurements of total inorganic nitrogen (TIN), nitrate, total reactive phosphate and ortho-phosphate as markers of nutrient status in the SPA. (Nitrate typically makes up the largest proportion of TIN inputs to estuaries, with nitrite and ammonia usually accounting for < 10%).

Table 24. TIN and TRP in freshwaters entering Poole Harbour based on observations 2001. (Data source: EA)

River	Median TIN conc. (mg l^{-1}) in freshwater	GQA TIN class	Median TRP conc. (mg l^{-1}) in freshwater	GQA TRP class
Sherford River (King Bridge)	7.75	B/C	0.169	B/C
Hamworthy Stream Hamworthy)	3.19	A/B	0.024	A/B
Upton Stream (Hamworthy)	1.59	A/B	0.028	A/B
River Frome (Holme Bridge)	5.53	B/C	0.157	B/C
Piddle (Wareham)	7.09	B/C	0.041	A/B

EA data (where available) for TIN and TRP in some of the freshwater of rivers and streams entering the SPA recently (2001) allows waters to be classified according to the scheme of Gunby *et al* 1995 (table 24). These figures represent concentrations, and not loadings, but suggest that waters of the Sherford, Frome and possibly the Piddle may contain the highest concentrations of nutrients although generally, levels are not elevated in comparison to the average for UK estuaries derived form projections (table 23). However, these classifications result from actual measurements as opposed to modelled values which can only be used as a rough guide.

There are other schemes which estimate the nutrient status from freshwater load inputs, thus encompassing point source discharges to rivers. Dong *et al.,* (2000) calculate estuarine nutrient loads by multiplying annual averages of all nutrient concentration measurements for contributing rivers, by the annual freshwater flow, however there is scope for error in that diffuse freshwater sources entering directly into the estuary will not be accounted for; likewise estuarine sources may be not be accounted for and make this type of estimate unreliable.

The issue of whether or not to focus on nutrient concentrations in the tidal waters or loading criteria has been a contentious one among both scientists and managers. Historically, sources of P are considered to be industrial effluents, rivers and streams and domestic sewage, and for N, rivers (predominantly) with a significant component from discharges (Owens, 1984).

Murdoch and Randall (2001) calculated annual average loads of TIN from the major discharges and tributaries of Poole Harbour for 1998 to 2000 (figure 47).

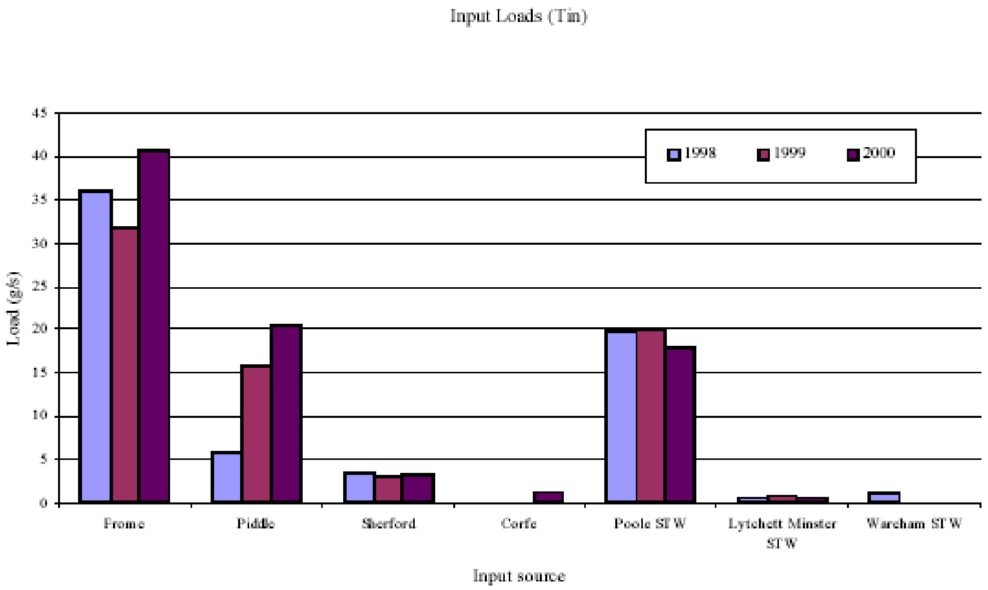


Figure 47. Discharge loads of TIN (g/s) for tributaries and point source discharges into Poole Harbour (from Murdoch and Randall, 2001). Loads were calculated by multiplying determinand observations and inferred flows. For tributaries, this was done by multiplying monthly mean flow and monthly spot sample data and taking the mean. For point sources, mean annual concentration from the monitoring data was multiplied by 1.5 times the STW's dry weather flow.

Murdoch and Randall (2001) considered that the Frome represents the most significant source of TIN overall, and of the STWs, Poole is by far the largest load. Wareham STW discharge appears to contribute little compared to the other loads and would therefore be expected to have little impact (although Murdoch and Randall note that the Wareham data were incomplete - none for 1999 or 2000 – the available data from 1998 suggests that the load is low).

As noted above, the characteristics of estuaries differ significantly, and therefore nutrient sources, their fate and effects in the estuarine environment are not easily predicted. Rather than relying on a classification scheme for the estuary as a whole it may be more beneficial to investigate the distribution of key determinands in finer detail:

7.2.2 Phosphate

Concentrations of orthophosphate in freshwaters entering the SPA are summarised in figure 48. The River Piddle at West Mills, and the Frome at Holme Bridge are HMPs (Harmonised Monitoring Points). Concentrations do not appear to be particularly high in the rivers and streams in comparison with the freshwater sources of other SW estuaries i.e orthophosphate at HMPs on the Severn are in the range 0.05 – 0.7mg l^{-1} and the Tamar, 0.02 – 0.08mg l^{-1}. Median values for Poole fall somewhere between these two. Annual average concentrations over the past 10-12 years show gradual reductions in orthophosphate concentrations for several of these sites (Sherford, Upton Stream) whilst the remainder are generally unchanged.

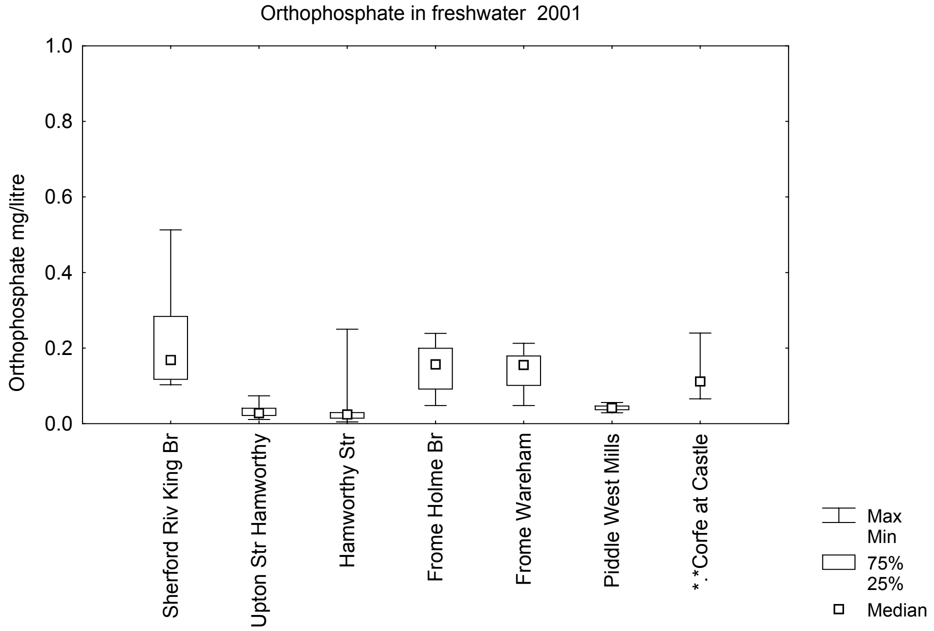


Figure 48 Orthophosphate in freshwaters feeding Poole Harbour 2001. Data source EA. *.* Mean, min and max concentrations only available for Corfe.

Data for orthophosphate concentrations in some of the discharges to Poole Harbour are summarised in figure 49. The available up-to-date values are site means, therefore mean, min and max concentrations are presented. Highest values are for Brownsea, Corfe and Studland STWs (10.5, 9.7 and 8.2mg l^{-1} respectively). No information on temporal trends for orthophosphate from these three STWs is available, although there have been gradual reductions of orthophosphate in discharges from Poole (see figure 53), Keysworth and Lytchett STWs over the past decade. *It is stressed that the data are for concentration only and do not take into the account volume discharged.*

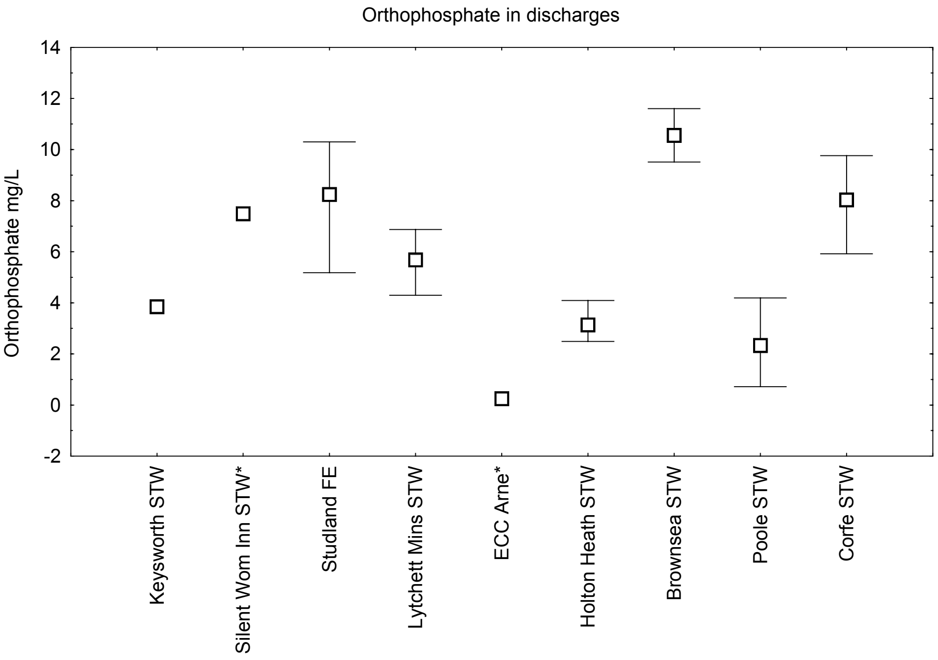


Figure 49. Mean Orthophosphate in discharges to Poole Harbour and tributaries. Data source EA. * denotes non-water company or combined discharge**.** Data are for 2001 except Brownsea, Silent Woman – 2000, Keysworth, ECC Arne – 1999. Error bars show min and max concentrations where n = >1

Orthophosphate in tidal waters is shown in figure 50. Highest concentrations are recorded at Poole Bridge, at the mouth of Holes Bay, reflecting inputs from Poole STW to this semi-enclosed Bay, and Hutchins Buoy 71, Wareham Channel (E). Both sites are in the north of Poole Harbour.

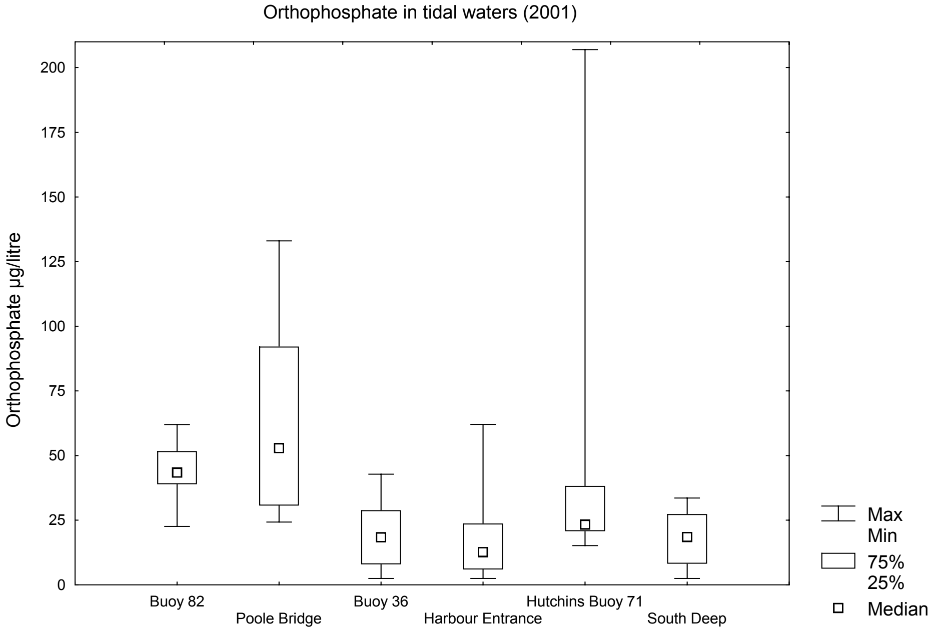


Figure 50. Orthophosphate in tidal waters of Poole Harbour. Data are for 2001. Data source EA.

Calculated as elemental P, the approximate background for the tidal waters (25th percentile) is in the range 2.01 – 12.75µg l^{-1} (table 25), invariably above the 0.1µg l^{-1} criteria set by the EPA(US) to protect estuarine and marine organisms, but in the lower range reported by Parr *et al* (1999) for coastal waters (7 – 165µg P l^{-1}). The more elevated background levels of orthophosphate are recorded at Wareham Channel buoy 82, and other north Harbour sites (Poole Bridge, Hutchins Buoy 71).

Table 25. Elemental P: 25th percentile for Poole Harbour sites 2001. Data source:EA

Site	Elemental P 25th percentile µg l^{-1}
Poole Harbour 1 Wareham Channel Buoy 82	12.75
Poole Harbour Four (Poole Bridge)	10.06
Poole Harbour Six (Near Buoy No 36)	2.64
Poole Harbour Ten (Harbour Entrance)	2.01
Poole Harbour Three (Hutchins Buoy 71)	6.82
Poole Harbour Twelve (South Deep)	2.72

7.2.3 Nitrate

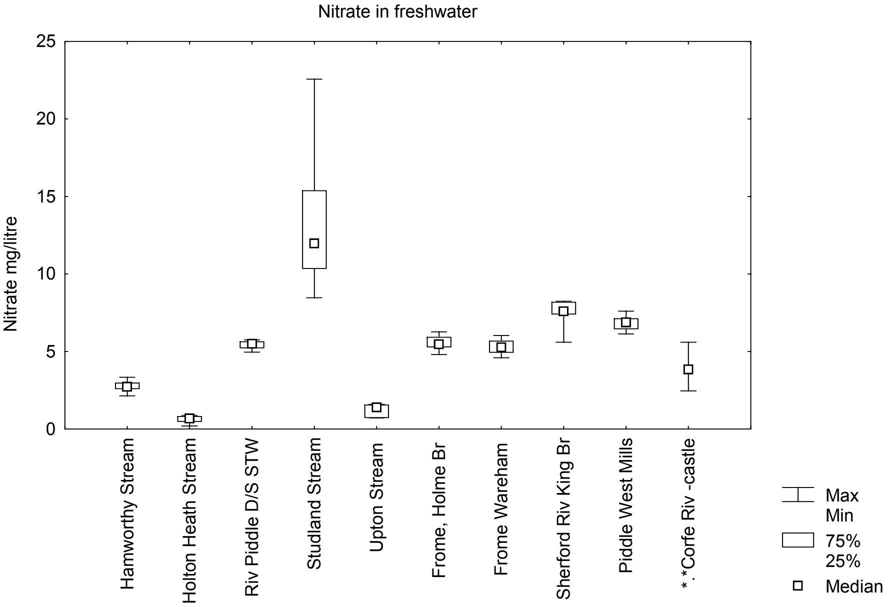


Figure 51. Nitrate in freshwaters feeding Poole Harbour. Data source EA. Data are for 2001 except Holton Heath and Studland Streams (1995) River Piddle d/s STW (1999). . *.* Mean, min and max concentrations only available for Corfe

Recent values for nitrate in freshwaters entering the Harbour are shown in figure 51. Concentrations appear to be elevated at several sites although with the exception of Studland stream, median values (and 25th and 75th percentiles) are below the lower threshold of 10mg l^{-1}. Since 1990, average values for nitrate in several freshwater sources have been steadily increasing, notably, the Frome at Wareham and Holme Bridge, the Sherford at King Bridge, Upton Stream, and the Piddle at West Mills. Mean annual values for nitrate at other sites remain relatively unchanged.

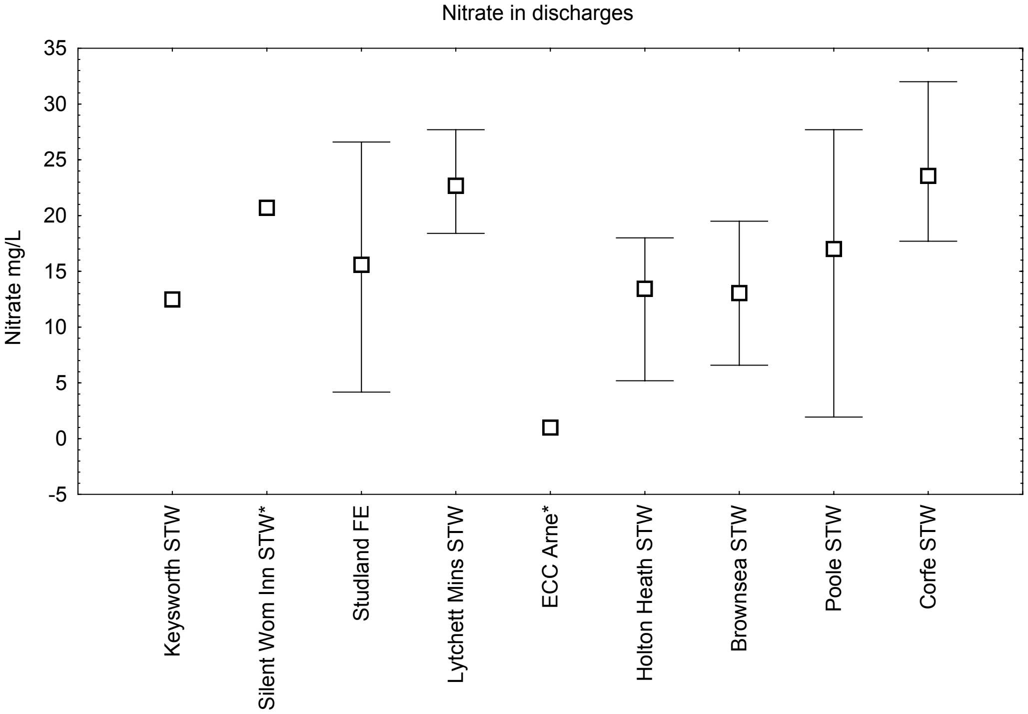


Figure 52. Nitrate in discharges to Poole Harbour (mean annual). Data source EA. * denotes non-water company or combined discharge**.** Data are for 2001 except Brownsea, Silent Woman – 2000, Keysworth, ECC Arne – 1999. Error bars show min and max concentrations where n = >1

Figure 52 summarises data for nitrate in a selection of discharges within the Poole Harbour catchment. *Again, it is stressed that the data are for concentration only and does not take into the account volume discharged.* There have been increases of nitrate concentration in several STW discharges over the past decade, including Brownsea, and notably Lytchett Minster where annual average concentrations rose from 9.9 to 25mg l^{-1} between 1990-1999 and have subsequently remained at roughly 20mg l^{-1}. Increases in nitrate concentrations in discharges are partly attributable to a revised ammonia standard which was introduced for discharges to this environmentally sensitive area. The aim of this standard is to reduce ammonia in tidal waters in order to comply with the proposed EQS of 0.021mg l^{-1} un-ionised ammonia (NH_3 N) for the protection of saltwater fish and shellfish. The effect of the ammonia reduction is a net increase in nitrate concentration exemplified in figure 53 below, which shows temporal trends for nitrate, ammonia and orthophosphate in discharges from Poole STW.

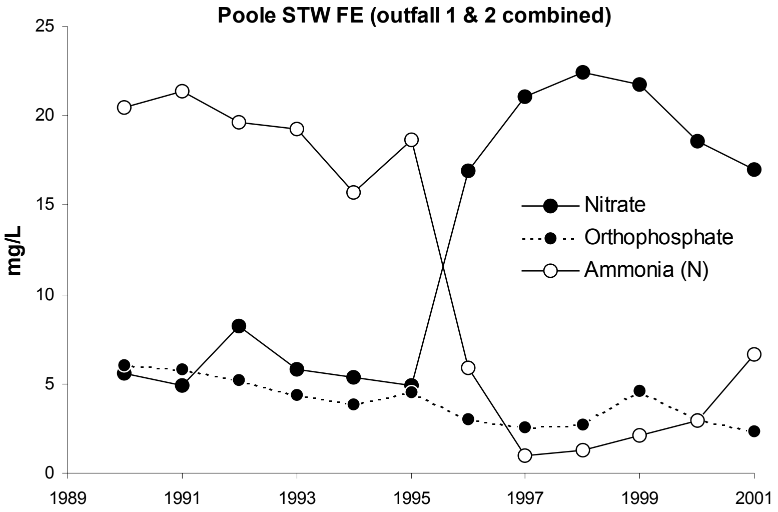


Figure 53. Temporal trends for mean annual concentrations of nitrate, orthophosphate and ammonia (N) in discharges from Poole STW. Data source:EA

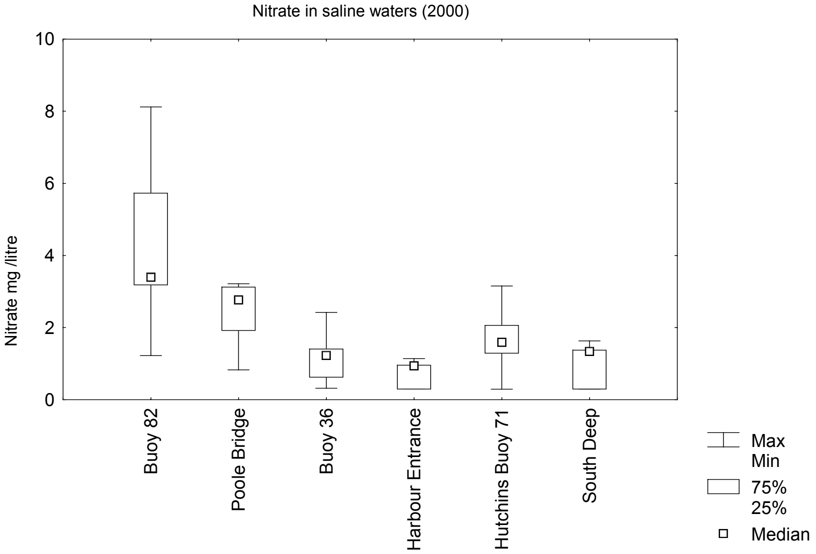


Figure 54. Nitrate in tidal waters of Poole Harbour. Data are for 2000. Data source EA.

Recent values for nitrate in tidal waters of Poole Harbour are shown in figure 54. The more elevated levels are recorded in (west) Wareham Channel to the north of the Harbour (buoy 82), probably reflecting nitrate input from the Rivers Frome and Piddle coupled with additional inputs from STWs (Keysworth and Wareham), and Poole Bridge at the mouth of Holes Bay due to discharges from Poole STW. Annual average values for nitrate (expressed as N) in tidal waters are shown in table 26. Again, values are above 0.144mg l^{-1} N, which indicates hypernutrification (dependant on P levels) as suggested in the North Sea Status Report (see above), although only at Wareham Channel and Poole Bridge do they approach or exceed (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations.

Table 26. Nitrate as N: Mean annual for 2000.

Region	Mean annual mg l^{-1}
Poole Harbour 1 Wareham Channel Buoy 82	1.41
Poole Harbour Four (Poole Bridge)	0.77
Poole Harbour Six (Near Buoy No 36)	0.39
Poole Harbour Ten (Harbour Entrance)	0.24
Poole Harbour Three (Hutchins Buoy 71)	0.55
Poole Harbour Twelve (South Deep)	0.32

7.2.4 Ammonia

Some forms of ammonia are toxic to marine life, whereas the effects of nutrient enrichment tend to be indirect. Ammonia is present in all natural waters, even if only at very low concentrations. It is derived either from the breakdown of organic nitrogen (mineralisation) or by the reduction of nitrate (a process known as denitrification). Ammonia as an intermediate stage in nitrogen fixation (conversion of atmospheric N_2 to fixed nitrogen and subsequent incorporation into microbial proteins, etc) is a relatively unimportant source in comparison to mineralisation (Cole *et al.,* 1999).

However, anthropogenic sources are generally more important in estuaries, notably sewage treatment effluent and, in some situations, run-off from agricultural land (Seager *et al* 1988). In tidal waters, the primary source of ammonia is direct discharge from Sewage Treatment Work (STW) outfalls. The toxicity of ammonia can therefore be a cause for concern in estuarine European marine sites and close to sewage outfalls in coastal waters.

The un-ionised form of the ammonium ion (NH_3) is the most toxic although total ammonia as N is more commonly monitored. The toxicity of ammonia to aquatic life is affected by temperature, pH, dissolved oxygen and salinity. In general, it increases with higher temperature and pH, and lower levels of dissolved oxygen and salinity. Of these three factors, salinity is the least important

Table 27. Proposed ammonia classification criteria for estuaries in England and Wales (Nixon *et al.*, 1995)

Ammonia (as N) (mg l^{-1}) 90th percentile	Class
0.86	A/B
4.7	B/C
8.6	C/D

A proposed GQA classification scheme for ammonia in the estuaries of England and Wales (Nixon *et al* 1995) is a tiered system (table 27), with class boundaries derived from a review of toxicity data. This scheme has not been implemented in England and Wales so far but provides a rough guide for comparison of ammonia levels in the marine site.

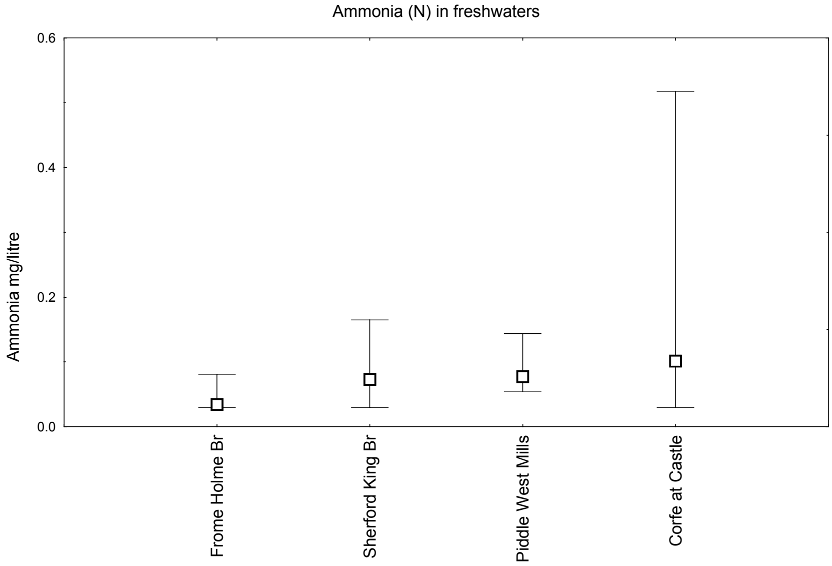


Figure 55. Ammonia (N) in freshwater entering Poole Harbour (mean annual). Data source EA. Data are for 2001. Error bars show min and max concentrations where n >1

Recent data for ammonia (N) in freshwater sources is summarised in figure 55. Values indicate that concentrations in riverine inputs to Poole Harbour are relatively low in comparison to concentrations in discharges (figure 56). *Again it is stressed that concentrations do not necessarily reflect ammonia loadings.*

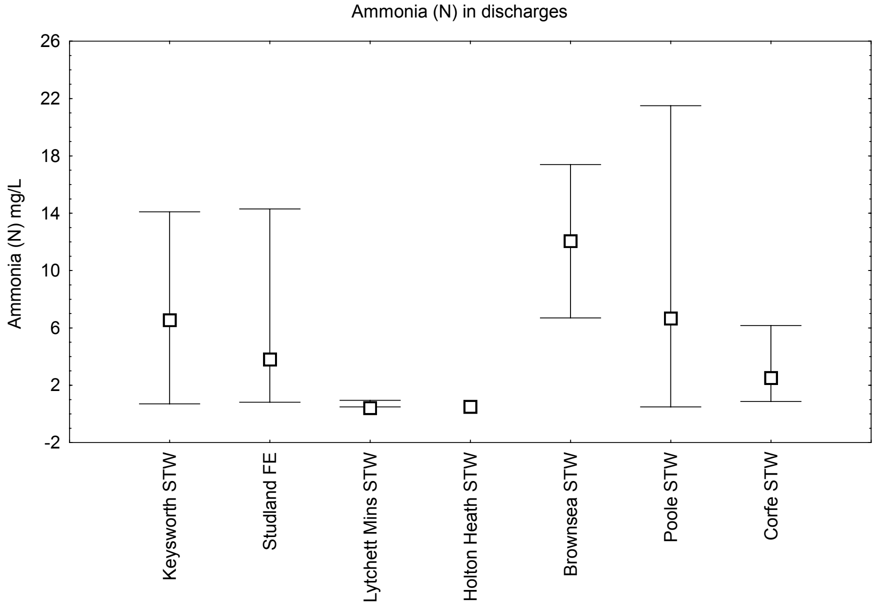


Figure 56. Ammonia (N) in discharges to Poole Harbour (mean annual). Data source EA. Data are for 2001. Error bars show min and max concentrations where n >1

18% of samples (17 out of 94) from Poole STW discharges failed to comply with the ammonia standard (for discharges) of 10mg l^{-1}, 95th percentile) in 2001, and the maximum concentration of ammonia recorded was 21.5mg l^{-1} in August 2001. Ammonia in discharges from Brownsea are also elevated (mean 12mg l^{-1}), and although no compliance failures are reported for 2001 this could be due to the low number of samples taken (n = 2).

Information on temporal trends for ammonia in discharges from Poole STW indicates a significant reduction between 1995 and 1997, presumably coinciding with the designation of the Poole Harbour as a sensitive area, and the nitrification of principal discharges. Subsequently, ammonia concentrations have been gradually increasing (figure 53).

Mean annual concentrations of ammonia in discharges from Lytchett STW have remained low since 1997 (<0.5mg l^{-1}). However, in discharges from Keysworth, mean values for ammonia also appear to be increasing (from 1.7 to 6.65mg l^{-1} between 1998 and 2001). There is no information available on temporal trends for Brownsea STW discharges.

Recent values for ammonia in tidal waters of Poole Harbour are shown in figure 57. highest concentrations are recorded in waters at Poole Bridge, at the mouth of Holes Bay, and Wareham Channel buoy 82. (max 0.37 and 0.22mg l^{-1} respectively). Again, these elevated values probably reflect STW inputs from Poole and Keysworth to the more enclosed areas of the Harbour. Note that the ammonia data are totals, and values for unionised ammonia, NH_3 (N), would need to be calculated from the total data, taking account of pH, temperature, and salinity. As a rough guide; for a pH of 8.2, a temperature of 20°C, and a salinity of about 30, 0.44 mg l^{-1} total ammonia (N) relates to about 0.021mg l^{-1} NH_3 (N), which is the proposed EQS for the protection of saltwater fish and shellfish.

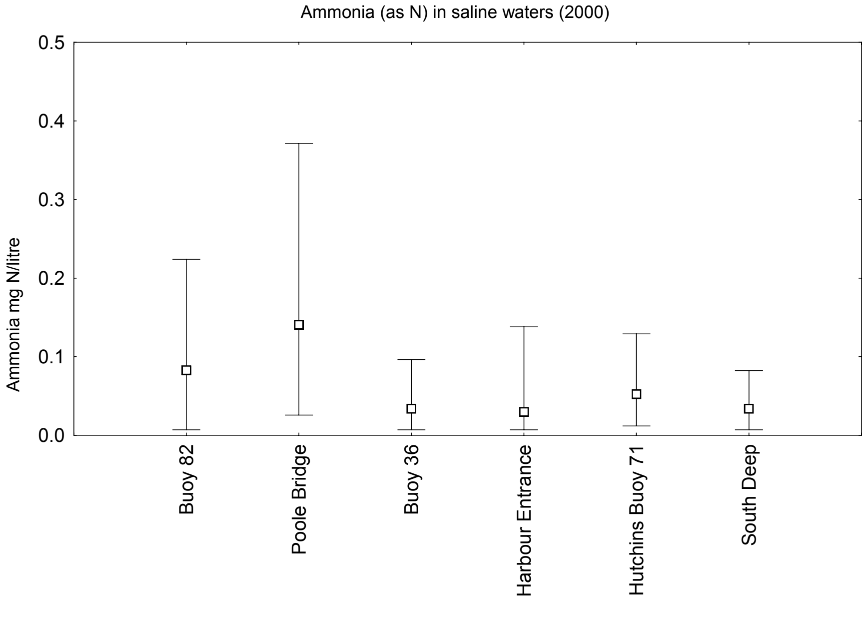


Figure 57. Ammonia (filtered as N) in tidal waters of Poole Harbour (annual mean). Data are for 2001. Data source EA. Error bars show min and max concentrations.

Mean annual values for ammonia in tidal waters show that concentrations have been steadily increasing over the past 4 years at all sampling sites (figure 58). This is likely to be partly due to increases in ammonia concentrations in some discharges, and also possibly related to releases from sediment during macroalgal decline (see section 5.2.1). More work is needed to establish the contribution from this source.

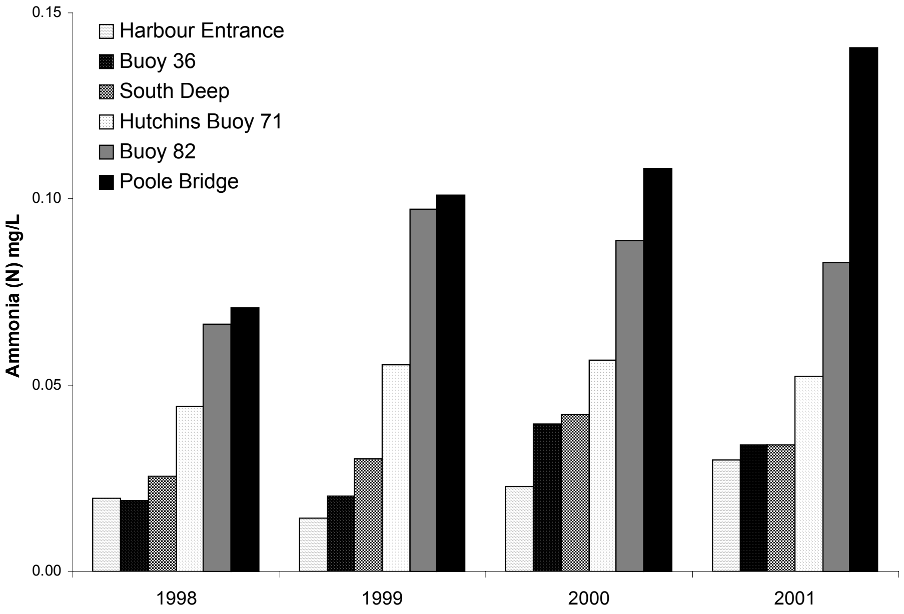


Figure 58. Temporal trends for mean annual ammonia (filtered as N) concentrations in tidal waters of Poole Harbour. Data source EA.

A review of the effects of ammonium on estuarine and marine benthic organisms is given in Nixon *et al* (1995). Toxicity data are presented for shrimps, mysids and lobsters (in which ammonia appears to interfere with the ability of lobsters to adjust to different salinities). Estimated 96-hour LC50s for juvenile school prawns *Metapenaeus macleayi* and leader prawns *Penaeus monodon* are 1.39 and 1.69 mg un-ionised ammonia NH_3 (N) l^{-1} (26.3 and 37.4mg l^{-1} total ammonia (N)) respectively (Allan *et al*., 1990). For the nauplius of the marine copepod *Tisbe battagliai*, Williams and Brown (1992) estimated a 96-hour LC50 of 0.787 mg NH_3 (N) l^{-1} (24.6mg NH_4 (H) l^{-1}), and tests on several life stages showed a No Observed Effect Concentration (NOEC) of 0.106mg NH_3 (N) l^{-1} (3.34mg NH_4 (N) l^{-1}). For invertebrates, toxicity appears to increase as salinity decreases (Miller *et al.,* 1990, Chen and Lin 1991), although more work is needed to establish whether this pattern is typical for all, or most, invertebrates (Nixon *et al.,* 1995). Several studies indicate that ammonia toxicity is greatest to early life stages of invertebrates.

Diverse invertebrate populations can survive, and flounder and salmonids pass through the Mersey Estuary with a mean unionised ammonia concentration of 0.008 mg NH_3 (N) l^{-1} (Cole *et al,* 1999). The majority of ammonium toxicity data relates to fish, although most of the species tested are freshwater species, with many coarse fish appearing to be as sensitive to ammonia as salmonids (Mallet *et al.,* 1992). Acute toxicity of ammonia to fish increases with low dissolved oxygen concentrations in both fresh and marine water environments (Seager *et al.,* 1988, Nixon *et al.,* 1995).

For this reason, the proposed GQA scheme for ammonia in estuaries was combined in a proposed joint scheme for dissolved oxygen and ammonia (Nixon *et al.,* 1995).

Ammonium toxicity to fish is also related to salinity, and reduced at lower salinity levels, gradually decreasing until it reaches a point similar to that found in freshwaters (Seager *et al*., 1998, Miller *et al*., 1990). This may be of relevance, especially in estuaries where DO sags occur at low salinities.

Ammonia does not accumulate in the sediments, although ammonifying microbial activity in sediments can result in ammonia release. This activity is greatest when large quantities of macroalgal biomass decline (Owens and Stewart, 1983), and may be a significant source in areas of Poole Harbour, and potentially toxic to sediment dwelling organisms and those organisms that use water in the boundary layer between the sediment and the water column for feeding or respiration (molluscs, crustacea and most annelids).

7.2.5 Turbidity and Suspended Solids

Turbidity is a measure of the attenuation of light in the water column and may be defined as the properties of water that cause light to be scattered and absorbed. Turbidity is be caused by particles and dissolved substances in water, including organic and inorganic particulate suspended matter, and dissolved substances that contribute to the colour of water. During blooms, the organic component can include significant amounts of algae.

The composition of particulate matter varies but is derived from: directly eroded material, sediments that have settled to the substratum and become resuspended during periods of high flow, dredging, suspended solids in discharges, chemical flocculation (at the salt/freshwater interface) and plankton. Increased or sustained turbidity in the water column may result in a reduction in algal (macroalgae and phytoplankton) growth rates due to reduced light availability. Subsequent adverse effects to zooplankton, benthic communities and fish populations (a general reduction in biodiversity) would be anticipated as particulates are suspended and re-deposited. An accompanying reduction in food availability may have secondary effects on higher trophic levels.

Methods for measuring turbidity vary, utilising different combinations of light transmission and scattering, water transparency (secchi disc), suspended solids (sample filtered and dried at 105°C or 500°C) or remote sensing. The results of these methods are not readily inter-convertible making comparisons problematic, and the only EQS appears to under the Bathing Waters Directive and relates to transparency using a secchi disc (guide value 90th percentile >2m, imperative 95th percentile >1m). These values are only applicable during the bathing season and may be waived in the event of ‘exceptional weather or geographical conditions’.

The principal method used by the EA for quantifying turbidity Poole Harbour is suspended solids (at 105°C) (units: mg l^{-1}). Rivers and streams entering the Harbour do not appear to comprise a significant source of suspended matter, mean annual values for 2001 are in the range 4.3 to 11.4mg l^{-1} (Upton Stream and Kings Bridge Sherford River, respectively). For discharges, recent data also indicate that levels are low (figure 59). On a temporal scale, mean annual values for suspended solids in discharges show reductions over the past decade, from several of these STWs (Brownsea, Poole and Studland). Data for other discharges, such as Cold Harbour, Holton Heath and Keysworth STWs show overall increases in suspended solids but are still relatively low and do not appear to constitute a significant source. Trade discharges from Pilkington (Holes Bay) have, in the past contained high concentrations - up to 15943mg l^{-1} (annual average) in 1994 - but there are no data for Pilkington since 2000.

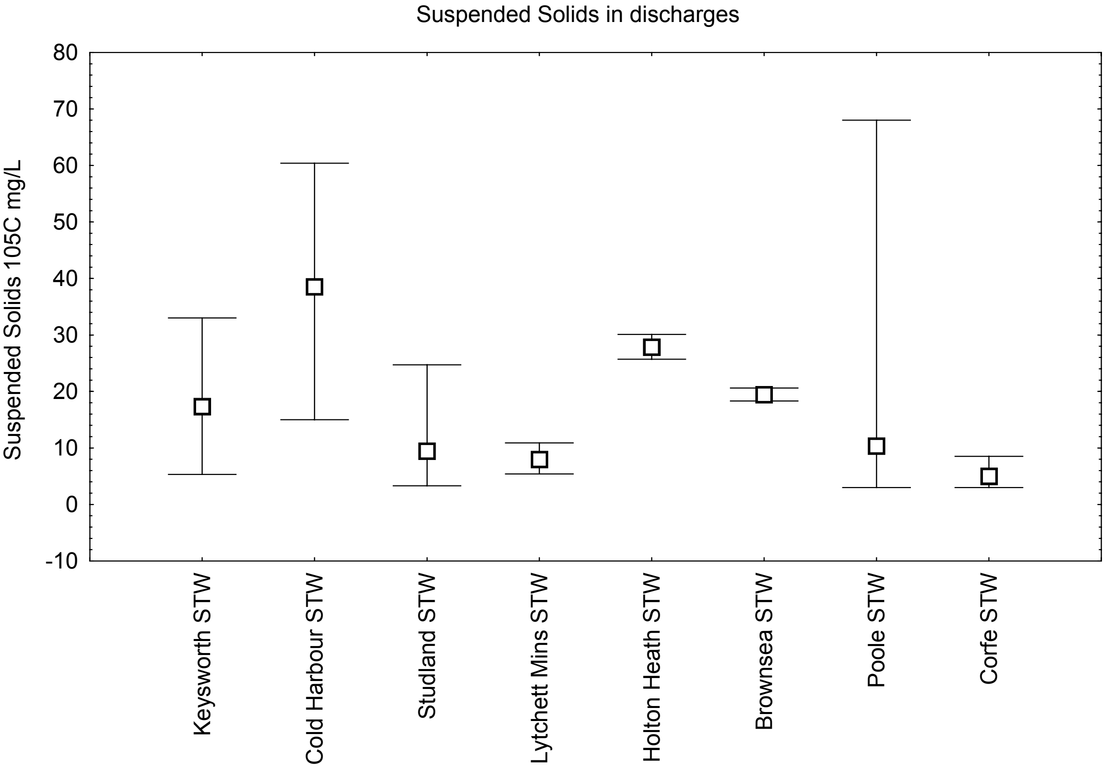


Figure 59. Mean levels of suspended solids @ 105°C in discharges to Poole Harbour. Data source EA. Data are for 2001. Error bars show min and max concentrations.

Recent data for suspended solids in tidal waters are summarised in figure 60. Concentrations are low (median values between 4.2 and 14.6mg l^{-1}) and not indicative of excessively turbid waters. Temporal trends for suspended solids in the tidal reaches of Poole Harbour are interesting in that all sites show gradual reductions between 1991 and 1999. Subsequently, these trends are reversed, with data for 2000, 2001 and early 2002 showing increases at all sites.

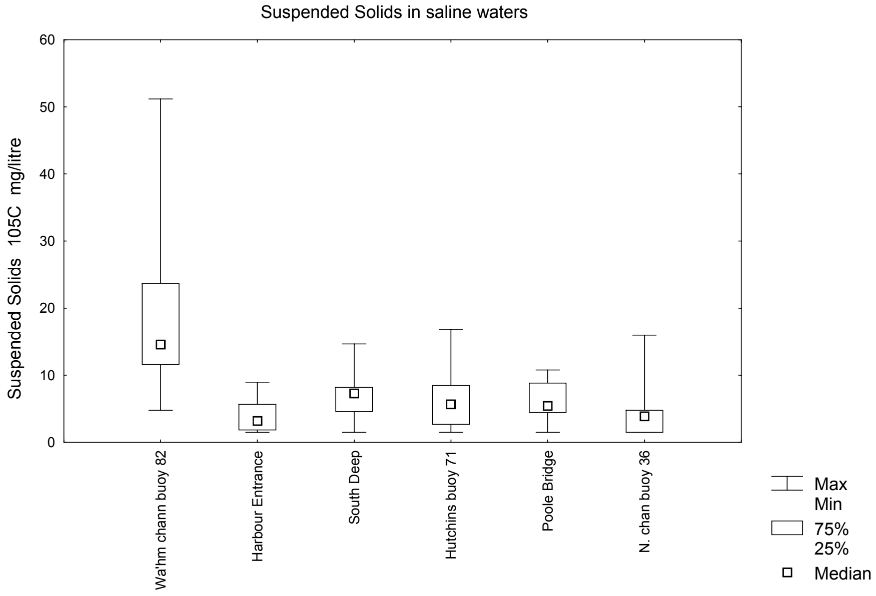


Figure 60. Suspended solids @ 105°C in tidal waters of Poole Harbour. Data source EA. Data are for 2001.

However, in comparison to concentrations of suspended solids in other UK estuaries, values for Poole waters are relatively low. To put these levels into some perspective, Cole *et al* (1998) cited typical mean annual values for suspended solids (105^{o}C) around the English and Welsh coast as 1-110mg l^{-1}, and suggested that anything >100 mg l^{-1} could be considered high. Annual average values for Poole tidal waters (2001) are in the range 4.1 to18.7 mg l^{-1}.

7.2.6 Chlorophyll *a*

It is important to distinguish between natural blooms and those induced by “artificial” causes. Levels of chlorophyll would be expected to increase in spring due to the natural spring bloom. It is pronounced or persistent blooms which cause concern. Elevated and prolonged spring and summer levels of chlorophyll *a* are one of the primary symptoms of increased nutrient inputs to estuarine waters and, as such, are another response variable measurement. Chlorophyll *a* is the molecule mediating photosynthesis in almost all green plants including phytoplankton. Rapid proliferation or blooms of phytoplankton, as reflected in elevated chlorophyll *a* levels, can occur throughout the ocean but are typically associated with temperate coastal and estuarine waters.

During winter months, growth of phytoplankton populations are at a minimum because of reduced temperature, light availability, and water column stability, and chlorophyll-*a* levels generally remain low. Monitoring of chlorophyll *a* is more often restricted to spring and summer months when estuarine concentrations in optimum growing conditions may exceed 50-80μg l^{-1} (Monbet 1992).

Data for chlorophyll *a* concentrations in tidal waters of Poole Harbour are summarised in figure 61. Generally, data are for April-September, although measurements taken in the winter months (Dec, Jan, Feb) are included for some sites

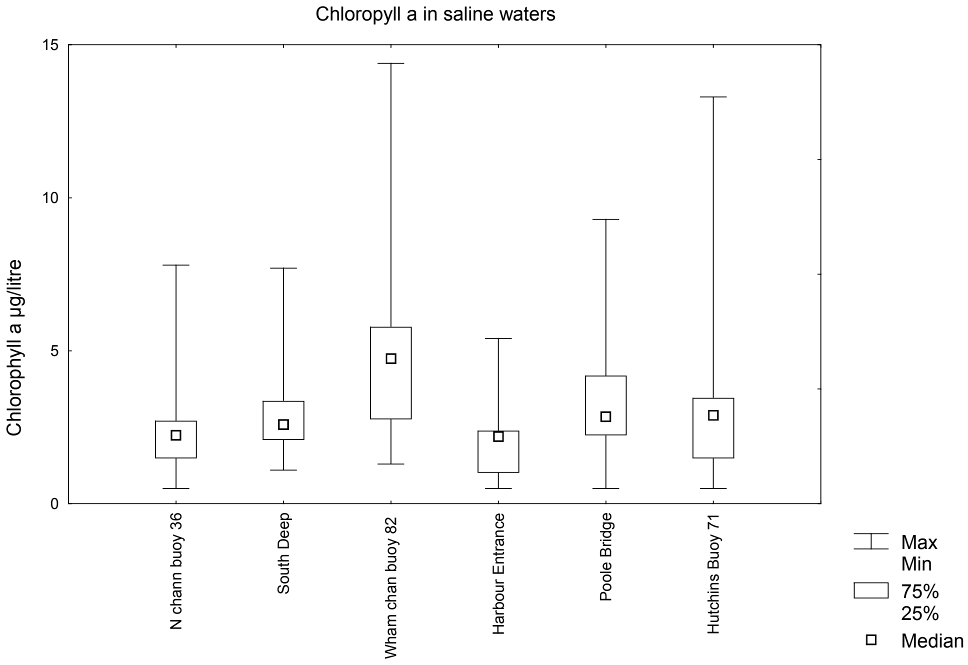


Figure 61. Chlorophyll *a* in tidal waters of Poole Harbour. Data source EA. Data are for 2001.

In the UK, an indicator (mean) value for suspected eutrophic conditions is set at 10µg l^{-1} chlorophyll *a* (Dong *et al.,* 2000). This value was exceeded in 2001 at north Harbour sites; Wareham Channel at buoy 82, and Hutchins buoy 71 (14.4 and 13.3µg l^{-1}, respectively) with maximum concentrations occurring in May. Temporal trends generally show a seasonal pattern for chlorophyll *a* in tidal waters, with the expected spring and summer blooms. The most pronounced of these was during August 1995 when elevated concentrations were recorded at most sites (maximum 50µg l^{-1} off Lake Pier).

A winter bloom during January, 1996 is indicated by unusually high concentrations of up to 17µg l^{-1} chlorophyll *a* recorded at many sites (Wareham Channel, Lytchett Bay. South Deep) (see section 5.2.3).

Recent (2001) values for chlorophyll *a* in freshwaters entering the Poole Harbour are shown in figure 62. Highest concentrations are recorded in the Corfe River with a maximum of 57µg l^{-1} in April (presumably freshwater phytoplankton species). Potential effects of elevated chlorophyll *a* concentrations introduced in freshwaters would be due to die-off of these species (release of DOC) if they entered more saline waters in any great numbers.

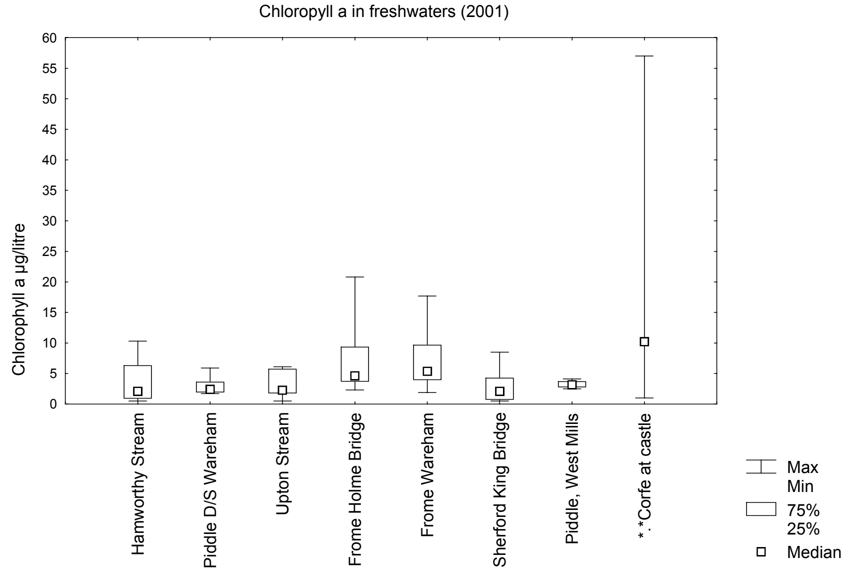


Figure 62. Chlorophyll *a* in freshwaters feeding Poole Harbour. Data source EA. Data are for 2001. *.* Mean, min and max concentrations only available for Corfe.

Overall, data for chlorophyll *a* are not indicative of intensive problematic phytoplankton blooms, although it appears that conditions may favour such an occurrence and continued careful monitoring of the situation is recommended.

7.2.7 Dissolved Oxygen

DO is measured in estuaries and coastal waters in terms of either a concentration (mg l^{-1}) or as a percent saturation (%). Table 28 shows recommended EQS values for saline waters derived from the review of Nixon *et al.,* (1995).

Table 28. Recommended EQSs for dissolved oxygen in saline waters (from Nixon *et al.,* 1995)

Saltwater use	EQS	Compliance statistic	Notes
Designated shellfishery	70% saturation 60% saturation 80% saturation	50%ile, mandatory standard Minimum, mandatory standard 95%ile, guideline value	EC Shellfish Water Directive
Saltwater life	5 mg l^{-1} 2 mg l^{-1}	50%ile 95%ile	
Sensitive saltwater life (e.g. fish nursery grounds)	9 mg l^{-1} 5 mg l^{-1}	50%ile 95%ile	
Migratory fish	5 mg l^{-1}3 mg l^{-1}	50%ile95%ile	Higher values may be required where fish have to traverse distances >10 km, or where high quality migratory fisheries are to be maintained

Various class thresholds for estuaries in England and Wales, based on DO over a continuous period of >1 hour were proposed by Nixon *et al.,* (1995) (see table 29) and although this scheme has not been implemented, the class thresholds are a useful indication of the levels of DO that are likely to affect organisms which are exposed for long periods.

Table 29. Proposed GQA class thresholds for dissolved oxygen in estuaries in England and Wales (from Nixon *et al.,* 1995)

GQA class boundary	Threshold value of DO (mg l^{-1})
A/B	8 mg l^{-1}
B/C	4 mg l^{-1}
C/D	2 mg l^{-}

The principal sources of DO in the marine environment are the atmosphere, via O_2 gaseous exchange across the air-sea surface, and *in situ* production by algae and aquatic plants during photosynthesis. DO levels vary with temperature, with lowest levels in estuaries occurring during the summer months. MPMMG (1998) reported summer and winter concentrations of DO at National Monitoring Programme sites in the UK in the range 4 to 11 mg l^{-1} expressed as a median, with lowest concentrations occurring in estuaries during the summer.

Increased levels of nutrients in estuarine waters can stimulate growth of both macro algae and phytoplankton (algal bloom), resulting in an intensification of both seasonal and diurnal variation in DO. Daytime photosynthetic activity may result in O_2 supersaturation of the water column; whilst at night severe depletion can occur due to respiration. These fluctuations can cause problems for fish and invertebrate communities. During bloom die-offs, microbial decomposition of algal cells leads to an increase in oxygen demand and acute DO depletion, which again can result in lethal and sub-lethal effects to fish and invertebrate communities.

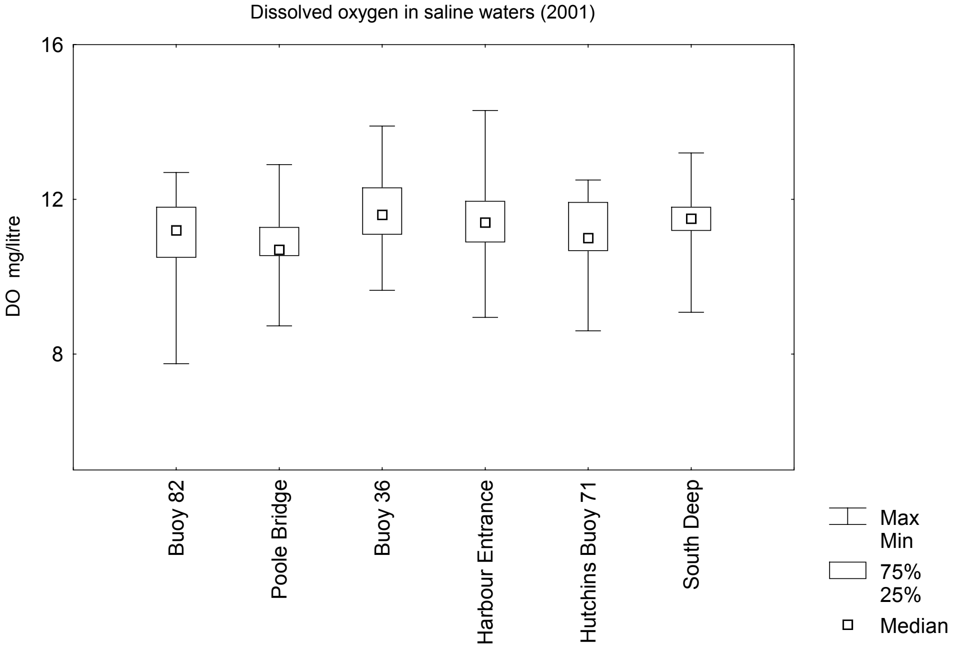


Figure 63. Dissolved oxygen in tidal waters of Poole Harbour. Data source EA. Data are for 2001.

Data for DO in tidal waters of Poole Harbour (2001) are summarised in figure 63. Overall, median values are in the range 10.7-11.6mg l^{-1}, and are above the 5mg l^{-1} (median) recommended EQS value for saltwater life and for migratory fish (see table 28) and minimum values for the year do not fall below this figure.

Temporal trends in mean annual values show an overall increase over the past 6 years for all tidal waters sites, although there appears to have been a sag during 2000. This trend is exemplified in figure 64 which shows mean annual DO concentrations in waters at North Channel buoy 36. Closer examination of the data shows that the lowest concentrations throughout the Harbour in 2000 occurred in September (minimum 6.02mg l^{-1} in South Deep).

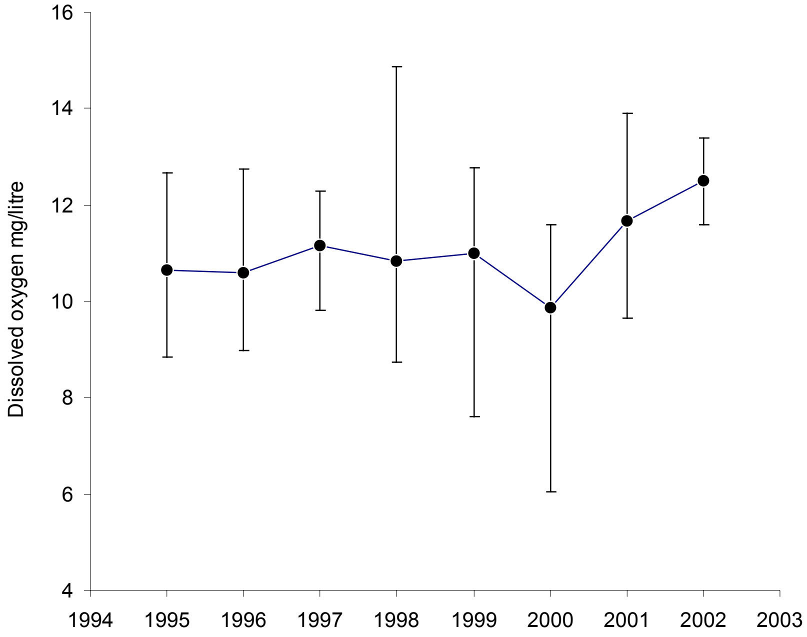


Figure 64. Temporal trends for mean annual concentrations of dissolved oxygen in tidal waters (North Channel) at Buoy 36. Data source EA. Error bars indicate minimum and maximum levels

Oxygen depletion also occurs in freshwater streams and rivers entering Poole Harbour. Lowest concentrations during recent years have been recorded in Holton Heath (1995), Hamworthy (2000) and Upton (1996) Streams (2.9, 3.8 and 4.3mg l^{-1}, respectively), and Hamworthy Stream continues to have relatively low levels of DO (7.5, 5.1 and 10.7mg l^{-1}, mean, min and max for 2001). However, for the principal rivers (Frome, Piddle, Sherford), annual mean DO concentrations have remained relatively constant with no indication of severe DO depletion.

There is no information available for nutrients, or levels of nutrient-related determinands in Holes Bay, where conditions, and published literature indicates that the greatest impact of hypernutrification occurs.

8. SEDIMENT STATUS AND QUALITY GUIDELINES

8.1 Metals

In general, levels of metal contamination in estuarine sediments decrease significantly towards offshore sites, partly due to distance from major inputs, and partly due to changing characteristics of the sediments. The progression from fine silts rich in binding sites in the upper estuaries, to coarser sediments offshore is usually accompanied by decreasing contaminant loading. Thus, distributions will be governed to a large extent by the hydrodynamic regime in the system and the sorting and redistribution of fines. Sieving and normalisation procedures[1] are required to compensate for such granulometric and geochemical effects, to allow meaningful comparison of contamination levels (Langston *et al*., 1999; MPMMG, 1998). In the current report we have used our own data, for sediments sieved at 100µm, to examine sediment quality in Poole Harbour. Metals discussed are Hg, Cd, As, Cr, Cu, Pb and Zn.

At present there are no statutory standards for sediments in the UK. Several guidelines on sediment quality are emerging, and CEFAS has cautiously recommended the Canadian/US effects-based approach (CCME ,1999; Long *et al*., 1995). Threshold Effects Levels (TELs - affecting the most sensitive species) and Probable Effect Levels (PELs - likely to affect a range of organisms) are derived from published toxicity data for a variety of substances in sediments (laboratory and field exposures). TELs are proposed as an Interim Sediment Quality Guideline (ISQG) value. As yet these guidelines have not been validated in the UK, though for many List I substances of the Dangerous Substances Directive a 'standstill' provision applies whereby the concentration of the substance in sediments (and organisms) must not increase with time. Sediment quality is also important under the remit of the Habitats Directive (attainment of Favourable Conservation Status - FCS) which may require improvements to sediments at the site in order to secure long-term sustainability.

There are a number of further caveats to the application of these guidelines, as discussed by Grimwood and Dixon (1997) in the context of List II metals. Foremost are the possible influences due to fundamental differences in sediment geochemistry (as discussed in the footnote below)[1] and the use of non-indigenous test species in

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[1] The need to standardise/normalise sediment measurements: This stems from the fact that chemical composition varies according to the sediment type, irrespective of anthropogenic influence. Thus muds and silts naturally have higher metal loadings than coarse sands because of their larger surface area and more extensive oxyhdroxide and organic coatings (capable of sequestering other chemicals). There are various ways in which this granulometric variance can be overcome, including normalisation to geogenic elements such as Al and Li: this may be particularly useful when comparing sediments of totally different geological backgound. An alternative and more direct technique to minimise the influence of grain size in comparisons is to select particles of similar size – hence the use of particles <100µm for the examples shown on the following pages in the current exercise. A study of microwave-digested Irish Sea sediments conducted in our own laboratory has shown that, following sieving at this mesh size, further normalisation confers no significant additional advantage when comparing contaminant trends. Sieving fulfils a further function - to place emphasis on particles which are accepted by benthic organisms. Sieving at 100µm was the preferred option for comparisons made in this project. It is stressed that this is only one of the options for classifying sediments, others may be equally suitable; the point is that some adjustment has to be made for grain size otherwise comparisons are uncontrolled and of little value.

deriving thresholds. Nevertheless, in the absence of any UK standards, interim guidelines adopted by Environment Canada (CCME 1999; see Annex 5) serve as a rough indication of the risk to biota from sediment contaminants. Hence, their application will help to identify instances where efforts should be made to minimise further inputs of these substances to the SPA.

The status of metals in sediments from Poole Harbour is based on average values from MBA's own published and unpublished records. These data have been collected at intervals over the last 30 years in connection with various research projects and were not intended as a monitoring programme. A number of the earlier analyses were carried out before formal quality assurance practices were introduced. Nevertheless, the quality of the data are considered to be good. Methodologies have been successfully validated in numerous intercalibration exercises (Langston *et al.,* 1994a). The sediment measurements described here are for the <100µm fraction, without further normalisation (*see footnote on previous page for details*), and are in concentrated nitric acid digests.

With the possibility of biological effects in mind, we have represented data for metals in inter-tidal sediments from around Poole Harbour in map form, classifying sites according to the interim sediment guideline criteria for each metal (figures 65 and 66). Green bars denote sites where no harm to biota is predicted (below ISQG's), grey bars denote sites where effects cannot be excluded (between ISQG's and PEL's) and red bars represent sediment concentrations where harmful effects might be expected (above PEL's).

Chromium levels are generally low, particularly in the main body of the Harbour where most values fall below the ISQG value (green bars in figure 65). Slightly higher concentrations occur at a number of sites in Holes Bay (grey bars – biological effects cannot be excluded), though none exceed the PEL value where effects would be expected.

Arsenic and copper sediment concentrations generally fall between ISQG and PEL values i.e. within the range where effects ‘ cannot be excluded' (designated as grey bars in figure 65) but would not be expected. The concentration of As at one site was above the PEL, according to CCME criteria (red bar), though this represents a single sample close to a freshwater input and cannot be considered typical.

For the majority of sites Cd and Zn values, also fall between (grey) or below (green) ISQG and PEL values and are lowest to the south, and close to the Harbour mouth (figure 66). At one site in the north of Holes Bay, Cd and Zn concentrations are above the PEL threshold (red bar). This site is close to the stream carrying discharges from Poole STW which may be a contributory factor to the sediment loading (see also section 5.1.1).

The majority of Hg and Pb values in the outer Harbour fall within the ‘grey zone' between ISQG and PEL guidelines (figure 66). At sites within Holes Bay, concentrations of Hg and Pb generally fall above the PEL, and are at levels where effects might be expected.

It is stressed that these are guideline values only. Where sediments exceed the PEL this is generally by a relatively small margin (usually less than a factor of two), rather than by orders of magnitude. Effects due to these metals, would largely be chronic rather than acute. Furthermore, much of the data are more than ten years old and may not be representative of conditions now. Re-survey is needed to establish the current status and evaluate temporal trends. This is seen as a particularly important issue in terms of meeting standstill provisions for sediments under the Dangerous Substances Directive, and attainment of Favourable Condition (Habitats Directive), and may partially drive the requirement to minimise further inputs via aqueous discharges.

Although these sediment classifications do not themselves constitute direct evidence for effects, the results nevertheless indicate that such diffuse sources (sediments) could be of significance in Holes Bay. The maps in figures 65 and 66 also clearly indicate the less contaminated nature of the main Harbour, particularly in the relatively undeveloped southern sector.

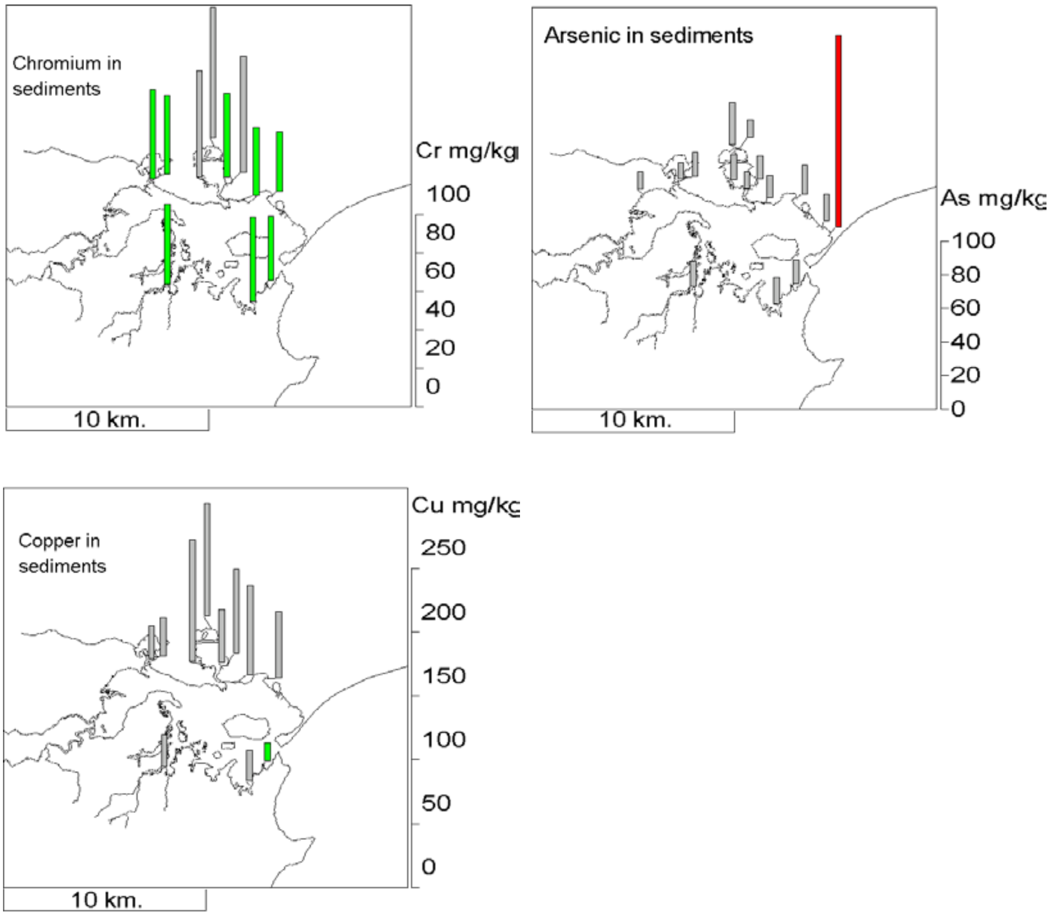

Figure 65. Chromium Arsenic and Copper in sediment (<100µm fraction). Classification of the Poole Harbour SPA based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source MBA)

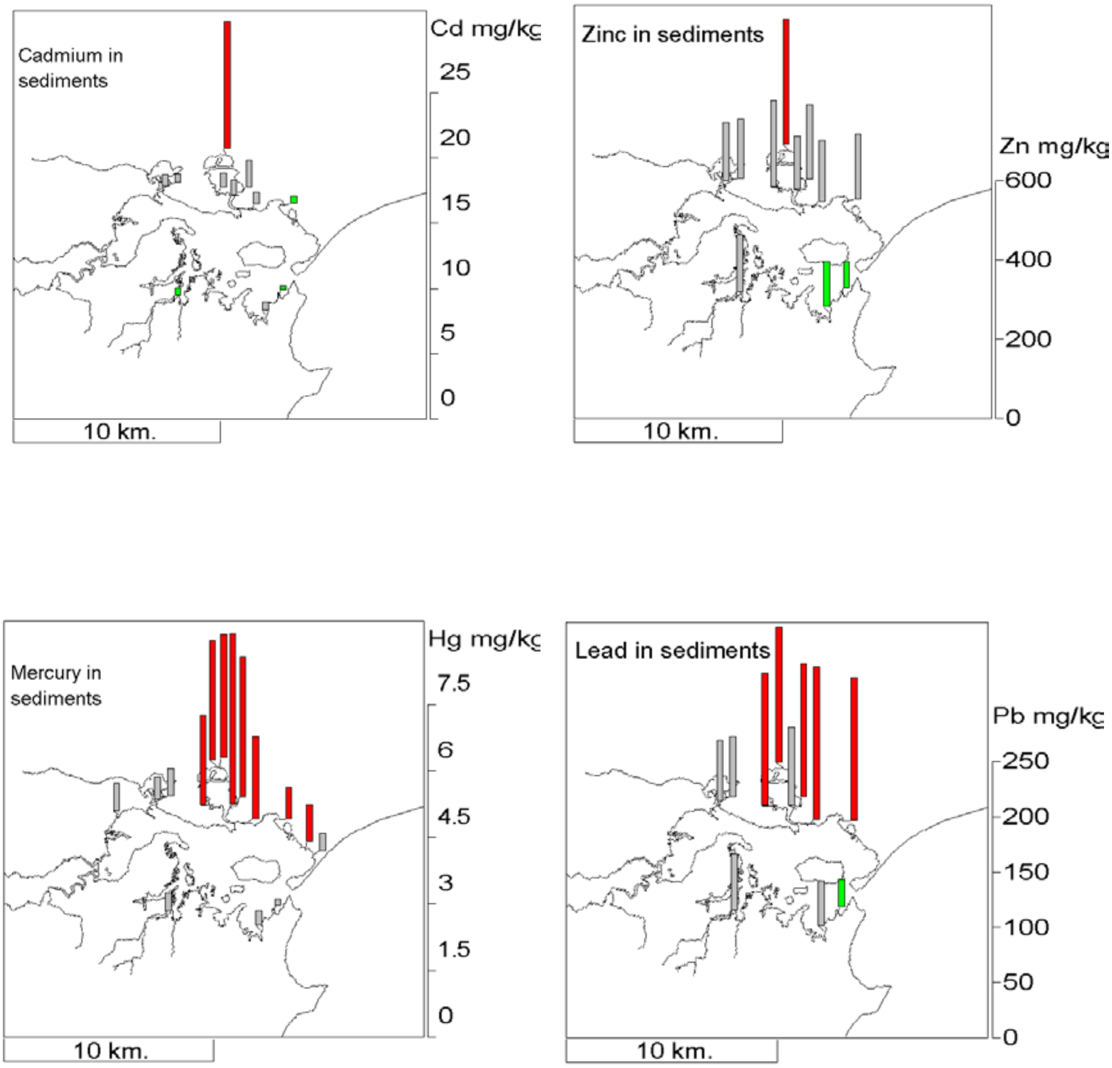

Figure 66. Cadmium, Zinc, Mercury and Lead in sediment (<100µm fraction). Classification of the Poole Harbour SPA based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source MBA)

8.2 Organic Substances –PCBs, pesticides, herbicides, PAHs, organotins

Time-averaged MBA data for TBT in sediments are plotted in figure 67. These concentrations (whose distributions have been described earlier in section 5.1.2) reflect patterns seen for metals, above; concentrations are much higher along the urbanised northern shoreline relative to the much less-disturbed south. There are no CCME guidelines for TBT, though OSPARCOM (1994;2000) has set a provisional ecotoxicological guideline value of 0.00005µg g^{-1} which is exceeded by all the values here. It is felt however that this guideline (which has no legal significance and is intended as a guide for further work) may be set too low. Whilst mollusc populations appear to have been affected by TBT along parts of the northern shoreline (section 5.1.2) TBT levels in sediments in the southern sector would be unlikely to pose any direct toxicological threat. Furthermore, as with metals, much of the data may not be representative of present-day conditions. Re-survey is needed to establish current status and trends.

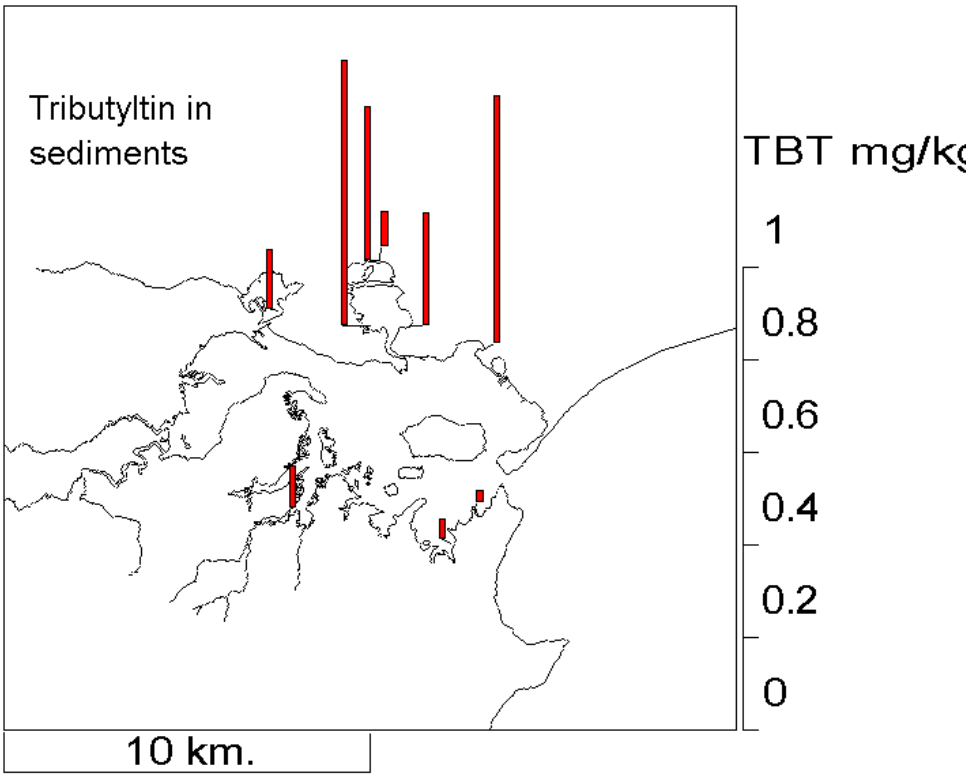

Figure 67. Tributyltin (TBT) in sediment (<100µm fraction). Classification of the Poole Harbour SPA based on interim marine sediment ecotoxicological assessment (OSPARCOM, 1994;2000). Red = effects expected (Data Source MBA)

Unfortunately, there is insufficient information on PCBs, PAH's, pesticides, (e.g. DDT, DDE, dieldrin) triazine herbicides to map distributions in Poole Harbour sediments or to evaluate the impact of these diffuse sources of contamination. Determinations appear to be limited to one site in the outer harbour (between Hamworthy and Brownsea Island). Concentrations of most organic contaminants here were predominantly below detection limits with the exception of γ-HCH. Values for the latter were above the probable effects level for lindane, though it is likely that the PEL may be overcautious (table 30). Endrin and dieldrin can be considered to be below thresholds. Even though the sum of DDT isomers appears to be above the PEL this is artifactual, due to the fact that detection limits are above the guideline value.

Table 30. Sediment concentrations (µg kg^{-1} dry weight) organic contaminants Poole Harbour (Data source EA). Interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) from CCME, (1999).

Compound	Mean[d] µg kg^{-1}	Values <DL %	ISQG[a] µg kg^{-1}	PEL[b] µg kg^{-1}
ALDRIN	1.40	96%		
DDT (PP')[c]	1.86	91%	1.19	4.77
DDT (OP)[c]	3.92	100%	1.19	4.77
DDE (OP')[c]	1.15	100%	2.07	3.74
DDE (PP')[c]	0.82	100%	2.07	3.74
DIELDRIN	1.45	91%	0.71	4.3
ENDOSULPHAN ALPHA	1.24	100%		
ENDOSULPHAN BETA	2.45	100%		
ENDRIN	0.98	100%	2.67	62.4
HCH GAMMA	5.99	4%	0.32	0.99
HEXACHLOROBENZENE	0.98	100%		
HEXACHLOROBUTADIENE	4.25	100%		
ISODRIN	2.94	100%		
PENTACHLOROPHENOL	20.00	100%		
TRIFLURALIN	4.48	91%		

[a] interim marine sediment quality guidelines (ISQG) and [b]probable effects levels (PEL) (CCME, 1999)

[c] guideline = sum of pp' and op' isomers;

[d]results with '<' sign are halved for calculation

It is recommended that future sampling programmes incorporate more information on sediment contaminants over a broader range of sites, including Holes Bay.

9. MODELS

Early work by Falconer is the source of much of the fundamental bathymetry used subsequently. However, there have been several models applied to Poole Harbour in attempting to determine nutrient levels, and predict changes resulting from various scenarios of 'nutrient removal/reduction from Poole STW'.

Falconer (1983) initially applied a numerical model to Poole Harbour and Holes Bay to determine the existing total N levels in the Harbour from a number of inputs, and hence predict the corresponding nitrogen levels as a result of removing the input from Poole STW. The model used was a two-dimensional depth integrated type and included the equations of momentum, in two mutually perpendicular directions, mass conservation, and solute mass balance. Included in the momentum equations were the convective accelerations, the Coriolis effect, shear forces on the waters' surface and bed as a result of wind action and bed friction, respectively, and a turbulence model based on a simple mixing length hypothesis.

The model was run with and without nitrate inputs from Poole STW so that direct comparisons could be made of the numerically predicted nitrate distributions across the Harbour. Each simulation was carried out for inputs of TON only, and for TON and ammonia, and for high and low riverine inputs. Results from this mathematical model study suggested that the nutrient levels in throughout the bulk of Poole Harbour would not be reduced by the removal of the N input, but that N concentrations in Holes Bay could be reduced significantly.

Falconer (1986) subsequently developed and modified the numerical model to include many of the complex flow features arising in Poole Harbour and Holes Bay. In particular these incorporated the tidal jet effect arising as a result of the relatively narrow harbour entrance, flooding and drying of extensive shallow water regions, and representation of the N input across the open seaward boundary for the flood tide. This hydrodynamic water quality model was applied to predict water elevations, depth-average velocity components and the depth mean N concentrations in Poole Harbour and Holes Bay. Simulations resulted in a similar outcome to the earlier model in that findings again indicated that N removal from Poole STW discharge would not have a dramatic effect on N concentrations in Poole Harbour, but would reduce concentrations in Holes Bay by 50% or more. Seasonal simulations (low and high flow scenarios) for April suggested that removal of nitrogen input from Poole STW would result in the mean N concentration being reduced by ~10% for both the whole, and lower harbour, whilst September simulations for nitrogen removal resulted in overall reductions of about 10% for TON removal only, or 25% for removal of TON and NH_3.

More recently, numerical modelling of environmental quality, using the estuarine simulating shell ECoS (Harris, *et al.*, 1991), has been tested by the EA in the context of nutrient loadings (nitrogen- as total inorganic nitrogen) and the Urban Waster Water Treatment Directive (Murdoch and Randall, 2001). Modelling was used to confirm whether reducing discharges from Poole STW would reduce the incidence of eutrophication (Poole Harbour is designated as a sensitive area in this context because of its low natural dispersion).

A basic three-compartment (Holes Bay, West Harbour, East Harbour) mass-balance design was used with the volume of each cell altering on a tidal basis, and river flows and discharges modified as required. After validation and audit the model was considered to be practical and capable of performing simulations over annual periods or longer, using routine monitoring data (Murdoch and Randall, 2001).

As indicated in section 5, TIN loadings from rivers (particularly the Frome and Piddle) generally exceed STW inputs (which account for 28% as an annual average, mostly from Poole). However, in summer Poole STW is estimated to contribute 45% of the load to the Harbour. Thus, the discharge is likely to have a significant local effect on Holes Bay water quality, at least in summer. Predictions from the ECoS model, arrived at by Murdoch and Randall, indicate that Poole STW inputs increase TIN concentrations in Holes Bay by 62% in summer (50% in winter). Effects are also seen in the east Harbour but are, as expected, much lower in magnitude (17 and 14% in summer and winter, respectively).

Turning to phosphorous, Hanrahan *et al* (2001) describe an export coefficient modelling approach to predict total phosphorus (TP) loading in the Frome catchment from point and diffuse sources. Output is provided on a seasonal (monthly) basis for 1998, and on an annual basis for 1990-1998. The model predicted an annual TP load of 25605 kg yr^{-1}, compared with an observed (measured) value of 23400kg yr^{-1}. Monthly loads calculated using the export coefficient model agreed well with monthly observed values except in months of variable discharge, when observed values were low, probably due to infrequent, and therefore unrepresentative, sampling. Comparison between filterable reactive phosphorus (FRP) and TP concentrations observed in the period 1990-1997 showed that trends in FRP could be estimated from trends in TP. A sensitivity analysis (varying individual export coefficients by plus or minus 10%) showed that STWs (3.5%), tilled land (2.7%), meadow-verge-semi natural (1.0%), and mown and grazed turf (0.6%) had the most significant effect (percent difference from base contribution) on model prediction. The model was also used to estimate the effect of phosphorus stripping at STWs in order to comply with a pending change in the European Union wastewater directive. Theoretical reduction of TP from the largest STW in the catchment (Dorchester) gave a predicted reduction in TP loading of 2174kg yr^{-1}, whilst reduction in TP from all STWs in the Frome catchment gave a predicted reduction in TP loading of 3902kg yr^{-1}.

The authors acknowledge however, that the export coefficient modelling approach is only as good as the quality of input data, and nutrient loading predictions exhibit some degree of variability. A further limitation in the model could be the reliability of livestock numbers within the catchment (collected from a 1998 census and assuming that the parish boundaries uniformly coincided with the Frome catchment).

Despite the limitations, the seasonal export coefficient model, ECoS, and others described here are obviously valuable as a management tools. They provide reasonable simulations of environmental behaviour arising from changes in levels of discharge (for nutrients at least). This should lead to informed decisions on which controls will be most practical and effective in the reduction of nutrient loads to receiving waters throughout the SPA.

10. CONCLUDING REMARKS: POOLE HARBOUR SPA

10.1 Biological Status

Poole Harbour is recognised to be of high conservation status by virtue of its ability to support substantial numbers of waders in winter, and because of its importance as a breeding site for terns and gulls. Nationally important aquatic species include the honeycomb worm *Sabella pavonia* and the sponge *Suberites massa,* whilst regionally important species include two bryozoans *Anguinella palmata* and *Farella repens*.

Generally, however, reports suggest that the Harbour as a whole supports a relatively impoverished diversity of fauna and flora, partly as a result of the restricted range of habitats. Essentially, the extensive intertidal mud-flats throughout are comprised of two major habitat types; firstly fine silts and clays (usually in sheltered embayments and creeks) and, secondly, sediments with a wider range of grain size (subjected to greater wave action and tidal currents). Diversity is particularly limited in muddy areas and dominated by Annelida, oligochaetes and, occasionally, bivalves such as *Abra tenuis* or the crustacea *Corophium volutator*. Higher diversity is indicated in the more heterogeneous sediment types. There is relatively little hard substrate apart from artificial structures along the urbanised north-eastern shoreline.

Diversity indices

Diversity indices are frequently used to assess the effects of environmental degradation on the biodiversity of natural assemblages of organisms. An intertidal survey was carried out (IOE, 1986) involving sediment particle size, hydrocarbon and trace element concentrations (see appropriate sections) and benthic macrofaunal analyses. Fifteen transects were sampled at high, mid and low tide levels. A total of 115 species were found, representing a moderately species-poor environment. The faunal community was dominated by Annelida (51 species representing 44% of total species) and detritus-feeding organisms, and considered typical of similar land-locked marine harbours on the South Coast of England. There were no significant differences in species diversity between the three tidal heights and major divisions were based on habitat type.

Abundance – The average abundance for UK estuarine sites is 6449 m^{-2}, and contaminated sites (such as the Tyne and Wear) contain highest densities (>10000 m^{-2}), usually dominated by a restricted range of principal species (*Polydora*, *Capitella* and *Ophryotrocha*) which probably indicates the effects of organic enrichment (MPMMG, 1998).

In Poole Harbour, lowest abundances, (105 and 510 m^{-2}) were found at mid- and low shore sites to the NE of the Harbour, (Sandbanks N and Parkstone Bay, respectively) whilst for high shore sites, the lowest abundance (1305 m^{-2}) was recorded at Nath Point in the mouth of the Corfe River

Highest abundances were found in the South of the Harbour; Brands Bay (low shore, 31870 m^{-2}), Ower Bay (mid shore, 17815 m^{-2}) and Wytch Channel (high shore, 21835 m^{-2}) (IOE, 1985).

Species richness is indicated by the number of taxa per unit area (e.g. 0.1m^2). Within Poole Harbour, Dyrynda (1987) examined trends in species richness using data from dredges, and overall survey data. Sites were then as identified as very high, high, intermediate or low (table 31).

Table 31. Harbour wide trends in species richness (from Dyrynda, 1987)

Very High	High	Intermediate	Low
Haven	Lower Wytch	Swash	Lower main channel
Lower South Deep	Blood Alley	Diver	Upper Wytch
Upper South Deep	Upper main channel	**Lower Wareham**	Upper Wareham
	Lower Holes Bay	Mid Wareham	
		Upper Holes Bay	

IOE (1986) also indicated that species richness was greatest (37 species) in Upper South Deep (low shore, Furzey Island E) but found least number of species (7) in the mouth of Parkstone Bay (low shore).

Shannon-Weiner diversity index (H^1) expresses the relationships between the occurrence of species and the apportioning of individuals among those species. Poole Harbour sites generally scored low with the lowest (0.17) for a high shore site - Furzey Island S. Highest scores were for mid shore sites, (up to 3.7, Furzey Island N) (IOE, 1986).

Evenness (Heips) is a complementary measure of the allocation of individuals across species: low values are associated with samples from sites numerically dominated by only one or two species which is generally indicative of stressed communities. The average is 0.738 for coastal waters (MPMMG, 1998). Evenness statistics for Poole Harbour sites are in the range 0.05 to 0.78 and almost invariably below average; the exceptions being mid shore sites at Furzy Island N, north Sandbanks and Parkstone Bay, although the evenness statistics are sill low (0.76, 0.78 and 0.77 respectively (IOE, 1986).

IOE (1986) concluded that indices indicated a poor fauna with dominance of certain species (notably cirratulid or capitellid polychaetes and oligochaetes, and occasionally the bivalve *Abra tenuis* or crustacean *Corophium volutator*) at several sites.

Few studies have described biodiversity in Holes Bay, although Howard and Moore, (1988) note that lowest diversity of communities and species was found in the Bay. Harris (1983) also concluded that Holes Bay was impoverished in terms of diversity and density of organisms. Dyrynda (1983) considered the impoverished flora and fauna was a result of the restricted flushing characteristics, combined with the effects of discharges which rendered the Bay eutrophic (see sections 6 and 5.2).

Abundance Biomass Comparison - Warwick *et al.,* (1989) used the Abundance Biomass Comparison (ABC) method (Warwick *et al.,* 1986) to analyse faunal distribution and community structure in several estuaries, including Poole Harbour, in an effort to establish whether observed patterns resulted from the effects of natural environmental variables, or whether they were affected by some unnatural disturbance such as chemical pollution, organic enrichment from sewage, frequent bait digging etc. This method depends on the fact that the distributions of biomass among species in marine macrobenthic communities show a differential response to disturbance, which can be demonstrated by the comparison of k-dominance curves for abundance and biomass. In Poole Harbour, only one (Sandbanks (N) of the seven sites sampled appeared to be completely undisturbed. Sandbanks (S), Brand's Bay (E), Newton's Bay and Holton were classified as moderately disturbed, whilst Brand's Bay (W, upper and lower) were grossly disturbed.

Thus, it is evident that biodiversity in parts of Poole Harbour is relatively low in comparison with other UK estuaries. Before summarising possible modifying effects of specific contaminants, it is worth citing several other general pieces of evidence indicating anthropogenic influence on biota.

Warwick *et al* (1989) noted that *Nereis diversicolor* from Poole Harbour populations were significantly heavier (for a given length) than *Nereis* from five other southwest estuaries (Southampton, Exe, Plym, Tamar and Severn). Polychaetes are among the most widespread inhabitants of contaminated and uncontaminated sediments and *Nereis* is relatively abundant throughout most of the Harbour, including Holes Bay. The observed differences in weight/length ratios for Poole *Nereis* may reflect organic and nutrient enrichment of sediments, due to STW discharges combined with the poor flushing characteristics of secondary embayments.

Perhaps a more serious manifestation of nutrient enrichment: prohibition orders were twice put in place in 2000 stopping the collection of shellfish due to ASP and DSP toxin contamination, and as a result the shellfishery came close to collapse.

Eel (*Anguilla anguilla)* numbers are declining in Poole Harbour and several of its tributary rivers including the Frome and Piddle, and is a cause for concern. At present this species is still subject to a fishery. White and Knights (1994) also note a reduced abundance of juvenile eels and elver in the Severn Estuary and comment on possible causes (poor water quality due to sewage effluent, turbidity, increased siltation and oceanic events), however there is no firm evidence that these factors can explain reduced catches and recruitment into freshwaters. The downward trend for eel numbers in UK estuaries is consistent with the decline currently observed throughout Europe. The impact of contaminants on Poole eels has not been studied extensively but could be useful in determining possible causes of decline. A study on metallothionein induction and 'metal fingerprinting', for example, has been shown to provide valuable information about the ecotoxicological status of eel populations, with regard to metals, in the Thames Estuary (Langston *et al.,* 2002; see annex 6).

As one means of assessing the potential biological impact of water quality, MPMMG regularly conduct oyster embryo assays using surface water samples from estuaries and offshore areas around the UK (MPMMG, 1998). These are incubated in the laboratory with a fixed number of freshly fertilised embryos of *Crassostrea gigas*.

The developing embryos integrate the adverse effects of the different contaminants in the sample and if the water quality is poor, they fail to develop normally to D-shaped larvae. Failure to reach the normal larval state can be caused by everything from lethal effects to subtle interferences with embryonic development. Water from Poole was tested in this assay in 1993, and the percentage net adverse response was low in samples from near the Harbour mouth (12% or lower). Percentage net response was higher in Holes Bay water samples, with 27% embryos failing to develop normally (CEFAS, 1998). Thus, water quality of the SPA appeared to be relatively good towards the seaward boundary with gradual deterioration in 'biological condition' inside Holes Bay. No specific compound is implicated as being responsible for the failure rate in oyster larvae, although, in reality, it is probably due to a combination of contaminants rather than any single compound.

10.2 Chemical Status

Despite its high conservation value, the threats from pollution in Poole Harbour are not trivial. Concerns focus on Holes Bay, dominated currently by inputs from Poole STW, though diffuse sources, (principally sediments containing a legacy from previous trade discharges) undoubtedly constitute a potential risk to biota.

Indeed, in terms of general vulnerability of intertidal habitats across the site as a whole, fine silts and clays (usually in sheltered embayments and creeks) are likely to be most in jeopardy. They are most effective at sequestering contaminants, which are likely to accumulate and persist as a result of low energy conditions. Sediments with a wider range of grain size (subjected to greater wave action and tidal currents) will be better flushed and therefore less vulnerable to contaminant build-up. Since these physical characteristics also influence the distribution of infaunal benthic communities in Poole Harbour (with sheltered embayments showing less diversity than the more open flats) it is always going to be difficult to isolate anthropogenic effects.

At present, evidence indicating that modifications to biota have occurred due to contaminants is fragmentary. To address this issue in the future a more subtle, targeted assessment of impact will be required in order to establish cause and effect (discussed in greater detail in section 10.3). Until more appropriate integrated chemical and biological effects monitoring is put in place to answer this challenge it is only possible to assess, subjectively, possible impact of individual contaminants, or groups of contaminants, based on available evidence. Some of the major water and sediment quality issues thought to be affecting benthic communities in Poole Harbour are discussed briefly, below, and summarised in table 32.

Table 32. A Summary of Water and Sediment Quality issues in Poole Harbour SPA. (Findings for each of the numbered 'contaminant categories' are explained in more detail in the accompanying text).

'contaminant'	Area	Potential Sources	Most vulnerable features/biota
1) Organotins (TBT, TPT?)	Probably localised to Holes Bay, docks and marinas in the north east	Mostly historic but possibly some continuing input from shipping. Sediments.	Molluscs
2) Metals (Cu, Cd, Hg, Zn, and perhaps Ag and Se)	Holes Bay and Hamworthy/Lake areas in main Harbour	Trade discharges (mostly historic?), major STWs, boats and ships, sediments	Invertebrates (primarily molluscs and crustaceans), species composition, larval fish and birds
3) Nutrients – esp.N and ammonia (Problem macroalgal blooms; microalgal blooms less problematic)	Whole Harbour, particularly the North, and West, esp. Wareham Channel, Holes Bay and secondary embayments	Sewage discharges (esp. Poole STW) Diffuse sources Sediments?	Invertebrates, fish (estuarine and migratory, esp. early life stages), seabirds, General diversity
4) Low dissolved oxygen	Wareham Channel River Piddle (tidal)	Algal blooms (possible nutrient enrichment, Keysworth STW)	Migratory fish, invertebrate communities
5) Hydrocarbons: PAHs, PCB	Poorly defined, probably localised–secondary embayments	Combustion, run-off, discharges, boats and ships, sediments	Benthic invertebrates and fish (NB those in contact with sediment)
6) Pesticides and herbicides and other endocrine disruptors	Direct acute toxicity improbable. Extent of endocrine disruption not tested but effects probably localised–secondary embayments	Discharges, sediment	Invertebrates (esp. crustacea), fish

1) Organotins

In the late 1980s TBT was present in high concentrations in parts of Poole Harbour, particularly in marinas and secondary embayments along the northern shoreline. As a result molluscs, particularly infaunal species, became scarce at some sites. Shell thickening of the Pacific oyster was evident even on the commercial beds in the outer Harbour. Since legislation in 1987 there has been a significant reduction in TBT levels in water and a more gradual decline in sediment concentrations. This is reflected in body burdens in a number of bioindicator species. Reversion to more normal growth has occurred in oysters from the outer Harbour, however, animals layed in Holes Bay continued to show abnormal shell-thickening after legislation, consistent with delayed return of TBT concentrations to no-effects levels.

There are preliminary indications of improved recruitment of mollusc species such as *Scrobicularia* and *Macoma* at impacted sites. Nevertheless, it is likely that waters at some locations still exceed the EQS. Furthermore, the prognosis for the recovery of mollusc populations in parts of Poole Harbour might be expected to be slow in view of the persistence of TBT in sediments.

OSPARCOM provisional ecotoxicological guideline values for sediments are exceeded at many sites. It is felt however that this guideline may be set too low. Whilst mollusc populations appear to have been affected by TBT along parts of the northern shoreline it seems unlikely that TBT levels in sediments in the southern sector continue to pose any direct toxicological threat. This requires further surveillance.

There may be localised reservoirs of TBT in sediments, particularly in Holes Bay and close to marinas, susceptible to remobilisation by processes including physical resuspension (e.g. dredging, erosion) and bioturbation. It is therefore important to map out potential hotspots and quantify the threats (i.e. increased bioavailability, toxicity) from these processes.

2) *Metals*

Data from the 1970s and 1980s suggested that metal concentrations were elevated, particularly in Holes Bay, and were responsible for poor performance of oyster larvae near Poole Bridge. Trade discharges and STW were considered as likely sources at the time, though the former have all but ceased. There is no indication that the major rivers entering the Harbour (Frome and Piddle) are a problem with regards metals though Holton Heath stream may have been the source of some localised enrichment.

Current monitoring of tidal waters by the EA is limited to the outer Harbour. No cases of EQS exceedences have been recorded in recent years, though for most metals elevated levels are reported near Poole Bridge. More comprehensive data are needed for Holes Bay to assess the possible influences on estuarine water quality, and sublethal impacts on biota. Zinc levels in water (and seaweed) from the Rockley area also appear to be anomalously high and may reflect additional sources from antifouling and sacrificial anodes. The consequences of this should be investigated further.

In general, metals in sediments exceed sediment quality guidelines widely, though exceedences of probable effects levels are restricted largely to Holes Bay (Hg and Pb). Here, low energy conditions restrict the dispersal of contaminated particulates. Urbanised tributaries will contribute some particulate metal directly; adsorption/precipitation reactions in estuarine waters will add to these loads.

Metal concentrations in the upper, eastern part of Holes Bay are amongst the highest in the Harbour (due to proximity to Poole STW, and to enriched organic and oxyhydroxide coatings which sequester metals). Most elements decrease towards the Harbour mouth as finer muds become diluted with coarser, less polluted, sediment of marine origin. The relative abundance of organic and oxyhydroxide coatings will govern the ability of the sediment to act as a sink for contaminants and the extent to which they are remobilised. Sequential extractions suggest that a significant fraction of the metal loading in Poole sediments may be labile and thus has potential for remobilisation. This may be important in view of the maintenance dredging needed in the Harbour.

Bioindicator studies with infaunal species such as worms and clams have demonstrated significant bioaccumulation of silver, cadmium, mercury, selenium and several other metals, above normal. This undoubtedly reflects anthropogenic origin. Again that the principal areas of concern regarding metals can be sourced to Holes Bay. The metals most likely to be of significance toxicologically appear to be Cd and Hg and perhaps locally Ag, Cu, Zn and Se. Sn levels also tend to be high in Poole samples due to the relatively high levels of organotin.

Commercial fish and shellfish grounds are restricted to the outer Harbour and these species will therefore not exhibit contamination gradients to the same extent as indicator species sampled in Holes Bay. Oysters, however, have been used in the past to demonstrate bioavailability of metals throughout Poole Harbour. In oysters from the outer Harbour and Poole Bay, Cd levels were consistently at the OSPAR upper guideline level of 1 $\mu g\ g^{-1}$, throughout the 1980s, whilst Hg levels sometimes exceeded the lower Hg guideline of 0.1$\mu g\ g^{-1}$ but never the upper limit of 0.3 $\mu g\ g^{-1}$.

The extent of ecological impact due to metal contamination in Poole Harbour is largely unknown, but is probably restricted to parts of Holes Bay. It seems likely that a combination of features, largely determined by the STW inputs, are of significance in modifying communities in the Bay. Opportunists such as *Nereis diversicolor* appear to be able to out-compete most other infaunal species, close to the STW discharge, perhaps because of their known ability to tolerate metals. By comparison, juvenile bivalves (particularly sensitive to Cu and Zn) are much reduced in numbers towards the outfall.

In recent years the impact of metals in Holes Bay may have been compounded, and perhaps overshadowed, by the effects of TBT and nutrient enrichment. There is scope for more research to establish the physiological and ecological significance of metal body burdens. Sub-lethal indicators, notably metallothionein induction, provide sensitive and selective measures of metal stress and can help map affected areas, as well as monitoring temporal trends.

3) *Nutrients*

Poole Harbour is regarded as being hypernutrified throughout and has recently been designated as a Sensitive Area (Eutrophic) and Polluted Waters (Eutrophic) (EA, 2001). The highest levels of orthophosphate, nitrate and ammonia occur in the north of the Harbour (Wareham Channel and Poole Bridge -mouth of Holes Bay) reflecting high nutrient loads from the Rivers Frome and Piddle, and Poole STW, respectively. Modelling exercises have shown that these three sources constitute the greatest (total) nitrogen loads to Poole Harbour.

It is apparent that high levels of nutrients from Poole STW, which discharges into a tributary of Holes Bay, impact upon the North of the Harbour, although there is no information on nutrient levels in Holes Bay itself.

A recently introduced standard to reduce the more toxic form of nitrogen (ammonia) in discharges was relatively successful (although exceedences still occur, e.g. Poole

STW, 2001). Contrary to the expected reduction of ammonia in tidal waters, concentrations are, in fact, increasing, and may be linked, in part, to the macroalgal blooms observed in the Harbour. In estuarine and marine ecosystems there is a complex interaction of nutrients between sediment and overlying water, which in the case of N, involves a range of processes including nitrification, denitrification, mineralisation, assimilation and fixation which may all vary spatially and temporally. It has been shown that ammonifying microbial activity is greatest when macroalgal biomass declines resulting in the release of ammonia from sediment (Owens and Stewart, 1983). The complexity of the nitrogen (and phosphorus cycle), and the significance of sediments, has been long appreciated, nevertheless monitoring still largely involves measurements of nutrients in water. Until more data becomes available for sediments, and the significance of related microbial activity, any attempt at evaluating the sediment as sources or sinks of nitrogen, ammonia and phosphorus is difficult.

Hence, the overall problem persists and total nitrogen levels in Poole Harbour remain unacceptably high. The effects of excessive nutrient inputs, in the form of extensive macroalgal blooms, and localised diatom blooms have been apparent for several decades. These manifestations have prompted a series of complaints (spanning almost 20 years) by concerned members of the public, regarding excessive macroalgal growth, and smells, foams and scums from rotting algae. The greatest densities of macro-algae occur in sheltered areas of the Harbour, notably Holes Bay, Blue Lagoon, Lytchett Bay, Keysworth Point, Ower Bay and Brands Bay. These blooms have been implicated in shellfish mortalities between 1995 and 1997. In 2001, amnesic- and diarrhetic shellfish poisoning was detected in shellfish from Poole Harbour, resulting in two closures of the shellfishery during the year and almost causing economic collapse of the industry.

The issue has finally been acknowledged in the designation of Poole Harbour as an SA(E) and PW(E), and the Frome catchment as a nutrient vulnerable zone (NVZ) This provides the framework to bring about the introduction of nutrient stripping processes for problem discharges, and to define application of nitrogen to the land in an attempt to reduce nitrogen loads from diffuse sources.

Measures to improve the situation will be instigated under Wessex Water's AMP 4 plans, which are scheduled for 2005 – 2010. Thus, the problem is finally being addressed, but it may be 50 years on from initial reports of excessive nutrient enrichment in the Harbour, before reductions are effective.

Effects of the high levels of nutrients on individual species in the Harbour are largely unresearched, overall diversity is impaired, particularly in secondary embayments, where abundance of individuals may be high but dominated by opportunists with relatively short life cycles. This situation is typical of impacted estuarine systems.

In view of the conservation importance of the Harbour, it is evident that any increase in nutrients should be avoided and changes to consents (quantities and location) should therefore be considered carefully to avoid the risk of further enrichment. Reductions in the nutrient content of discharges are long overdue. Furthermore, there are important lessons to be learned from Poole Harbour regarding the siting of outfalls; many of these problems could have been avoided by more careful planning, as the inherently restricted flushing characteristics of the Harbour (in particular Holes Bay) exacerbate hypernutrification and subsequent eutrophication and probably could have been foreseen.

4) *Dissolved oxygen*

Oxygen sags have been reported intermittently, both in the Harbour itself and in the tidal reaches of the River Piddle. Low DO levels can have a significant impact on migratory fish; due to the energetic demands of their upstream migration and the increased activity that takes place during spawning, the oxygen requirements of migratory teleosts are very high at these times in the life cycle (Claridge and Potter, 1975). Since the maximum amount of oxygen that will dissolve in water is inversely related to temperature, even in clean turbulent water, oxygen concentrations can fall to below the minimum level for survival. Consequently, any factors that lower oxygen tensions will have a significant impact on fish populations at these critical times (Bird, 2002) and should be minimized.

5) Hydrocarbons, PAHs, PCBs

There are very few recent data on distributions of THC, PAH and PCBs in the Poole SPA, however, the available evidence suggests moderate contamination in some samples (n.b. Holes Bay) with a potential contribution from these compounds to chronic effects (based on bioassays using concentrated extracts).

Sediments exhibit much higher concentrations than water and, in Holes Bay, may be above tentative threshold levels for PCBs and PAHs). Any biological effects are likely to be chronic rather than acute, nevertheless detritivores and other benthic species could be particularly susceptible. Therefore bioaccumulation and the threat from these compounds needs to be characterised more fully, in various parts of the system.

PAH and PCB exposure induces the production of detoxifying enzymes (P450-associated) which can be used as sub-lethal indicators of stress for these organic contaminants (in much the same way as metallothionein induction can be used to signify metal exposure). Levels of one of these enzymes, ethoxyresorufin-O-deethylase (EROD), should be investigated further in relation to bioavailability of these compounds.

Thus, although there is currently no overwhelming proof of impact from this type of contaminant, this is largely because of the lack of adequate evidence. There is no doubt, however, that hydrocarbon, PCB (and metal) contamination of sediments has implications for benthic (infaunal) marine organisms and any future increase could be potentially damaging because of the confined nature of the Harbour and depositional characteristics of some embayments. Many sediment-dwelling organisms are essential food items for the important bird species and assemblages for which the site was designated. The threat posed by bioaccumulation and food chain transfer of, for example, PAHs, PCBs and metals, needs to be quantified in these species and requires a much more extensive data set.

6) *Pesticides, Herbicides and other endocrine disruptors.*

Currently there is very little evidence of direct toxicological impact from the majority of these compounds. Only occasionally are elevated levels reported in rivers, and the risks from aqueous discharges are probably low for the compounds described in section 5.1.4. The general issue here is that there appears to be insufficient data available to characterise this threat fully. This is particularly so for sediments and body burdens of infauna at potentially impacted sites (Holes Bay?). Allen *et al.*, (2000) highlight the lack of information on endocrine disruption in Poole SPA, flagging, as a high priority, research into the presence and effects of pesticides and other potential endocrine disruptors.

In summary, Poole Harbour was severely impacted by TBT in the late 1980s but, following legislation, showed signs of improvement, at least initially. This urgently needs to be reassessed. Nutrients also continue to impact parts of the system. However, the current major 'contaminant issue' for biota of the Poole SPA is probably not the effect of any one individual chemical but the combined effects of several, notably in Holes Bay. The sheltered, depositional characteristics here mean that sediments can retain a number of contaminants over long time-scales and thus may act as a secondary source. Salinity fluctuations and low redox potentials add to the stresses faced by benthic organisms attempting to colonise the mud flats of the upper Bay.

10.3 Future Research Requirements

Better, more integrated information on the environmental chemistry, 'health' and biodiversity are obvious, generalised top-level needs to address the 'quality' of the Poole Harbour marine site, just as rigorous monitoring of habitats will be needed to provide estimates of 'quantity' and extent.

With regard to the latter, an appropriate surveillance strategy should be adopted to quantify the nature and extent of biotopes in order to fulfil Favourable Conservation Status[1] requirements encompassed in the Habitats Directive. Monitoring techniques are described in detail by Davison and Hughes (1998) in relation to surveillance of *Zostera* beds but are equally applicable to biotope monitoring in a broader context. These fall into four categories: *aerial remote sensing* (aerial and infrared photography, satellite and multi-spectral scanning imagery (CASI)); *sublittoral remote sensing* (sonar techniques such as RoxAnn^{TM} and side-scan sonar provide maps of substratum types which, with appropriate ground-truthing, can be linked to the distributions of habitats and communities); underwater video (e.g. remotely-operated vehicles (ROVs), towed video and drop-down video); conventional benthic

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[1] Favourable Conservation Status (FCS) for a given habitat/species requires that condition has to be characterised and, if considered necessary, brought up to a level where the habitat/species is sustainable in the long term. Under the Habitats Directive, Member States must report on the progress towards FCS for all nominated sites. English Nature has produced Favourable Condition tables to aid this process, which encompasses a number of attributes, including the extent and biological quality of the interest features (summarised in Annex 1 for Poole, SPA).

sampling (e.g. grabs and cores and observations arising from inter-tidal surveys and diving). The first three options have potential for large-scale generalised mapping of distribution and extent, whilst the fourth provides a more detailed synthesis of biotope structure and condition. Ideally the preferred monitoring scheme would probably consist of a blend of these approaches. The selection of techniques will of course have to take in to account questions of scientific objectives, practicality and cost.

The requirement to fulfil FCS and other drivers on water and sediment quality (such as the 'standstill' provision under the Dangerous Substances Directive) may be more difficult to monitor. The fragmented nature of much of the available environmental quality data - much of which has been collected for compliance rather than process-oriented purposes - prevents all but a first-order approximation of the status of the site. This needs to be addressed if we are to progress our understanding of how environmental quality, and in particular anthropogenic inputs, are affecting the biota and, hence, the status of the marine site.

A major issue central to the current project is how to monitor the health of the environment within the site i.e. to ensure that conditions are favourable for the survival of biota and, if they are not, to establish any cause and effect relationships. An obvious starting point is that, given the continual natural variation which occurs in ecological systems, critical assessments of human impacts (consented discharges, fishing, dredging, boatyards, tourism etc) can only be made against a time series of background data. The question is 'What data best serves this need'?

Some of the classes of contaminants which, in our opinion, should be prioritised in future Poole surveys are indicated above. Traditionally, surveillance of chemical parameters in water has often been carried out by the EA for the purposes of compliance monitoring and is not necessarily intended for survey purposes (or for the type of characterisation being undertaken here). This should perhaps be reviewed in future to maximise the value of the information. Accurate, up-to-date chemical data, on e.g. metals, PCBs, PAHs, organotins and pesticides, are needed to give better impressions of fluxes from rivers and discharges, and to provide details of their current distribution, sources and sinks within the SPA. It is particularly important that future sampling programmes incorporate more information on sediment contaminants and their role as diffuse sources.

TBT legislation has been effective in reducing levels of TBT in waters of Poole Harbour as might be expected in a small-boat dominated area. Sediments may have slowed down the recovery of benthic biota to an extent, though there have been indications in recent years of the return of some important molluscan species, notably in Holes Bay. It would be highly informative (and a useful model for other sites) to continue established time-series observations to determine the timescales for recovery of such impacted sites. These should include chemical monitoring of water, sediments and organisms together with complementary information on mollusc populations at established sites in the Harbour (see section 5.1.2). Selective coring of sediments would also help to map out potential TBT hotspots and define the threat from remobilisation and bioavailability in areas likely to require dredging. More work on the sensitivity of key organisms to sediment-bound TBT is also recommended.

For metals, the use of a validated suite of indicator species (which must incorporate species capable of reflecting sediment-bound contamination), at a selection of reference sites, is a useful way of estimating bioavailability around the system (section 5.1.1). Valuable baseline information for sites in Poole Harbour is already in place regarding the (historic) distributions of many of these species and their metal burdens. The current status of the SPA needs to be defined in similar fashion, and at intervals in the future, to ensure bioavailability does not increase. The same organisms may also be useful for the surveillance of organic pollutants and should be tested accordingly.

Metal concentrations in sediments presumably consist of both anthropogenic and geogenic components which have yet to be distinguished over much of the Harbour. Methods are becoming available which address this problem and the consequences for bioavailability and re-release. It is recommended that these be applied to the Poole SPA.

In recent years, techniques have been developed to assess sub-lethal biological impact in greater detail, which would allow targeted biological-effects screening of the SPA, including possible problem discharges. Also, an important priority is how to assess (and then minimise) the risk from diffuse sources. For the Poole site this applies particularly to the sediments of Holes Bay. Current sediment quality criteria are useful initial guidelines to impact, but are not seen as being entirely appropriate for all species or sediment types. Individual site conditions are likely to modify threats considerably, therefore accurate assessments need to be customised to the habitat in question. By selection of an appropriate suite of indicators/biomarkers, a sampling strategy could be tailor-made to establish the causes and extent of damage with greater certainty. This would ideally include; conventional quantitative ecological survey (for identifying changes in the abundance and diversity of species), chemical and biomonitoring procedures (for determining the concentrations and bioavailability of anthropogenic contaminants), and biochemical, physiological and behavioural biomarkers which signal exposure to, and in some cases, adverse effects of, pollution (see Annex 6 for further details).

When such procedures are used in combination in well-designed survey programmes, they can provide insights into which pollutants are responsible for environmental degradation. They may also be useful in addressing the long-standing problem of additivity/ synergism. A major criticism of many current statutory monitoring assessments, whether using comparisons with EQS values, or some other marker, is that they address only single contaminants at a time. Even if individual chemicals do comply with limit values (as most appear to do in Poole Harbour) it does not necessarily mean the environment is healthy. Biological effects may occur if several contaminants act together. Hydrocarbon residues alone consist a myriad of individual compounds, most of which have additive toxicity. The majority of outfalls and sediments contain a particular cocktail of chemicals whose true impact can usually only be assessed through a site-specific evaluation, taking into consideration the interactions that occur between different components. By incorporating biological-effects monitoring, alongside chemical surveillance, it may be possible to make substantial progress towards understanding and managing these complex environmental issues and would provide more reliable and objective site characterisations in the future. If such an integrated approach were put in place at an early stage, to provide baselines, it would clearly be amenable for measuring long-term trends e.g. in the assessment of recovery.

11. BIBLIOGRAPHY

Ackroyd, D. R., A. J. Bale, Howland, R.J.M , Knox, S., Millward, G.E. and Morris , A.W. (1986). Distributions and behaviour of dissolved Cu, Zn and Mn in the Tamar estuary. Estuarine, Coastal and Shelf Science. 23: 621-640.

ADRIS (Association of Directors and River Inspectors in Scotland). 1982. Standards for persistent pollutants in EEC-designated Shellfish Growing Waters. report by the Marine and Estuary Survey Group.

Allan, G.L., Maguire, G.B. and Hopkins, S.J. (1990). Acute and chronic toxicity of ammonia to juvenile *Metapenaeus macleayi* and *Penaeus monodon* and the influence of low dissolved oxygen levels. Aquaculture, 91:265-280

Alden, R.W. & Butt, A.J.(1987). Statistical classification of the toxicity and polynuclear aromatic hydrocarbon contamination of sediments from a highly industrialized seaport. Environmental Toxicology and Chemistry, 6, 673-684.

Allen, J.R. and Thompson, A. (1996) PCBs and organochlorine pesticides in shag (*Phalacrocorax aristotelis*) eggs from the Central Irish Sea: a preliminary study. Marine Pollution Bulletin, 32(12): 890-892

Allen, Y.T., Hurrell, V., Reed, J. and Matthiessen, P. (2000). Endocrine Disruptors and European Marine Sites in England. Centre for Environment Fisheries and Aquaculture Science (CEFAS). Contract C01042 for English Nature. 159pp

Anderson, M.J., Miller, M.R. and Hinton, D.E. (1996a) In vitro modulation of 17- beta -estradiol-induced vitellogenin synthesis: Effects of cytochrome P4501A1 inducing compounds on rainbow trout (Oncorhynchus mykiss) liver cells. Aquatic Toxicology 34(4):327-350

Anderson, M.J.,Olsen, H., Matsumara, F. and Hinton, D.E. (1996b) In vivo modulation of 17- beta -estradiol-induced vitellogenin synthesis and estrogen receptor in rainbow trout (Oncorhynchus mykiss) liver cells by beta-napthoflavone. Toxicology and Applied Pharmocology 137:210-218

Ankley, G.T., Collyard, S.A, Monson, P.D. and Kosian, P.A. (1994). Influence of ultraviolet light on the toxicity of sediments contaminated with polycyclic aromatic hydrocarbons Environmental Toxicology and Chemistry, 13, no. 11, pp. 1791-1796, 1994

Armitage, M. J. S., Burton, N. H. K., Atkinson, P. W., Austin, G. E., Clark, N. A., Mellan H. J. and Rehfisch, M. M. (2002) Reviewing the Impact of Agency Permissions and Activities on Bird Populations in Special Protection Areas: Level 1 Interpretation. BTO Research Report No. 296. 156pp. The National Centre for Ornithology, Thetford, Norfolk.

Aston, S.R.,Thornton, I.,Webb, J.S., Milford, B.L.and Purves, J.B. (1975). Arsenic in stream sediments and waters of South-west England. Science of the Total Environment, 4, 347-358.

Baldock, B.M., Collins, K.J., Jensen, A.C. and Mallinson, J.J. (2001) Long term monitoring of an artificial reef in Poole Bay, U.K. Newsletter of the Porcupine Marine Natural History Society. 15-19 pp

Bamber, R.N. (1984) The benthos of a marine fly-ash dumping ground. Journal of the Marine Biological Association of the United Kingdom, 64:211-226

Besten PJ den, Elenbaas JML, Maas JR, Dieleman SJ, Herwig HJ, Voogt PA (1991). Effects of cadmium and polychlorinated biphenyls (Clophen A50) on steroid metabolism and cytochrome P-450 monooxygenase system in the sea star *Asterias rubens* L. *Aquat Toxicol* 20: 95-110.

Bird, D. J (2002). Environmental Factors Affecting Migratory Fish in The Severn Estuary with Particular Reference to Species of Shad and Lamprey. Report for the Environment Agency Wales. Severn Estuary Research Group, University of the West of England, Bristol

Bird, P., Comber, S.D.W., Gardner, M.J. and Ravenscroft, J.E. (1996). Zinc inputs to coastal waters from sacrificial anodes. Science of the Total Environment, 181, 257-264.

Blackburn, M.A. & Waldock, M.J. (1995). Concentrations of alkylphenols in rivers and estuaries in England and Wales. Water Research, 29(7), 1623-1629.

Blackburn, M.A., Kirby, S.J. & Waldock, M.J. (1999). Concentrations of alkyphenol polyethoxlates entering UK estuaries. Marine Pollution Bulletin, 38(2), 109-118.

Blanchard, M. (1997). Spread of the slipper limpet *Crepidula fornicata* (L.1758) in Europe. Current state and consequences. Scientia Marina, 61(9):109-118.

Boyden, C.R. (1975). Distribution of some trace metals in Poole Harbour, Dorset. Marine Pollution Bulletin, 6(12), 180-186.

Boyden, C.R. (1977). Effect of size upon metal content of shellfish. Journal of the Marine Biological Association of the United Kingdom ., Vol. 57, no. 3, pp. 675-714.

Brouwer A, Reijinders PJH and Koeman JH (1989). Polychlorinated biphenyl (PCB)-contaminated fish induces vitamin A and thyroid hormone deficiency in the common seal (*Phoca vitulina*). *Aquat. Toxicol.* **15**: 99-106.

Bryan, G.W., Langston, W.J. (1992). Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. Environmental Pollution, 76(2): 89-131

Bryan, G.W., Langston, W.J., and Hummerstone, L.G. (1980) The use of biological indicators of heavy metal contamination in estuaries, with special reference to an assessment of the biological availability of metals in estuarine sediments from South-West Britain. Mar. Biol. Ass. U.K. Occasional Publication No. 1, 73 pp.

Bryan, G.W., Langston, W.J., Hummerstone, L.G., and Burt, G.R..(1985). A guide to the assessment of heavy-metal contamination in estuaries using biological indicators, Mar. Biol. Ass. U.K., Occasional Publication No. 4, 92 pp.

Carlisle, D.B. (1954) *Styela mammiculata* n.sp., a new species of ascidian from the Plymouth area. Journal of the Marine Biological Association of the United Kingdom, 33:329-334

CEFAS (1998) Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1995 and 1998. Aquatic Environment Monitoring Report, 51, 116pp

CEFAS (2000) Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1997. Aquatic Environment Monitoring Report, 52, 92pp

CEFAS (2001) Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1998. Aquatic Environment Monitoring Report, 53, 75pp

Chen, J-C. and Lin, C-Y. (1991). Lethal effects of ammonia and nitrite on *Penaeus penicillatus* juveniles at two salinity levels. Comparative Biochemistry and Physiology, 100C, 477-482.

Claridge P.N. and Potter I.C. (1975) Oxygen consumption, ventilatory frequency and heart rate of lampreys (*Lampetra fluviatilis*) during their spawning run. Journal of Experimental Biology 63, 193-206.

Cole, S., Codling, I.D., Parr, W. and Zabel, T. (1999). Guidelines for managing water quality impacts within UK European marine sites. 441p. Swindon: WRc, 1999.

Collins, K.J., Jensen, A.C. and Lockwood, A.P.M. (1990) Fishery enhancement reef building exercise. Chemistry and Ecology, 4(4):179-187

Cooke, M., Nickless, G., Povey, A. and Roberts, D.J. (1979) Poly-chlorinated biphenyls, poly-chlorinated naphthalenes and polynuclear aromatic hydrocarbons in Severn estuary (U.K.) sediments. Science of the Total Environment, 13:17-26

Critchley, A.T., Farnham, W.F., and Morrell, S.L. (1986). An account of the attempted control of an introduced marine alga *Sargassum muticum,* in southern England. Biological Conservation, 35: 313-332.

Critchley, A.T., Farnham, W.F., Yoshida, T., and Norton, T.A. (1990). A bibliography of the invasive alga *Sargassum muticum* (Yendo) Fensholt (Fucales; Sargassaceae). Botanica Marina, 33: 551-562.

Crothers, J.H. (1975). On variation in *Nucella lapillus* (L.): shell shape in populations from the south coast of England. Proceedings of the Malacological Society of London, 41, 489-498.

Cundy, A.B. and Croudace, I.W. (1995). Physical and chemicals associations of radionuclides and trace metals in estuarine sediments: an example from Poole Harbour, Southern England. J. Environ. Radioactivity, 29, 191-211.

Cundy, A.B., Croudace, I.W., Warwick, P.E., & Bains, M.E.D. (1999). Decline of radionuclides in the nearshore environment following nuclear reactor closure: A U.K. case study. Environmental Science and Technology, 33(17), 2841-2849.

CSL (2000) Environmental endocrine disruptors: Review of research on top predators. Report for DETR, 111pp

CSTT, (1997). Comprehensive studies for the purposes of Articles 6 and 8.5 of 91/271/EEC, the Urban Waste water Treatment Directive. Marine Pollution Management Group (MPMMG) Comprehensive Studies Team

Davies, W.P. and Middaugh, D.P. (1978). A revised review of the impact of chlorination processes upon marine ecosystems: update 1977. In: Jolley, R.L., (Ed). Water chlorination: environmental impact and health effects. 1:283-310. Ann Arbor Science Publications

Davison, D.M. and .Hughes, D.J. (1998). *Zostera* Biotopes (volume I). An overview of dynamicsand sensitivity characteristics for conservation management of marine SACs. ScottishAssociation for Marine Science (UK Marine SACs Project). 95 pp

den Hartog, C. (1994) Suffocation of a littoral Zostera bed by *Enteromorpha radiata* Aquatic Botany 47(1)21-28

De Montaudouin, X. and Sauriau, P.G., (1999). The proliferating Gastropoda *Crepidula fornicata* may stimulate macrozoobenthic diversity. Journal of the Marine Biological Association of the United Kingdom, 79(6):1069-1077).

Dyrynda, P. (1984) Poole Harbour subtidal survey - southern sector - 1984. Report to the Nature Conservancy Council, 115pp. Nature Conservancy Council, Peterborough.

Dyrynda, P.E.J. (1987) Poole Harbour subtidal survey - IV. Baseline assessment. Report to the NCC by the Marine Research Group of University College Swansea, pp. 195. Nature Conservancy Council, Peterborough.

Depledge, M.H. and Anderson, B.B. (1990). A computer-aided physiological monitoring system for continuous, long-term recording of cardiac activity in selected invertebrates. Comparative Physiology and Biochemistry, *96A*, 473-477.

Dyrynda, E.A. (1992). Incidence of abnormal shell thickening in the Pacific Oyster *Crassostrea gigas* in Poole Harbour (UK), subsequent to the 1987 TBT restrictions. Marine Pollution Bulletin, 24(3), 156-163

Dyrynda, E.A., Pipe, R.K., Burt, G.R., and Ratcliffe, N.A. (1998). Modulations in the immune defences of mussels (*Mytilus edulis*) from contaminated sites in the UK. Aquatic Toxicology,42, 169-185.

EA (1997) Candidate sensitive area (eutrophic) and polluted waters (eutrophic). Poole Harbour. Environment Agency, South West Region. (February, 1997)

EA (2000) Local environment agency plan – Frome & Piddle and Poole Harbour & Purbeck - Plan from November 2000 to October 2005. Environment Agency. 106p.

EA (2001) Poole Harbour. Candidate sensitive area (eutrophic) and polluted waters (eutrophic). Environment Agency, South West Region. (January 2001

Eagle, R.A., Hardiman, P.A., Norton, M.G. and Nunny, R.S. (1979) The field assessment of effects of dumping wastes at sea: 4. A survey of the sewage sludge disposal area off Plymouth. Fisheries Research Technical Report Series, MAFF, (50), 24pp.

Elliott, M. and Griffiths, A.H. (1987). Contamination and effects of hydrocarbons on the Forth ecosystem, Scotland. Proceedings of the Royal Society of Edinburgh, 93B, 327-342

English Nature (2000) Poole Harbour European marine site, English Nature's advice given under Regulation 33(2) of the Conservation (Natural Habitats &C.) Regulations 1994, 45pp .

Ehrhold, A., Blanchard, M., Auffret, J.P. & Garlan, T., 1998. The role of *Crepidula proliferation* in the modification of the sedimentary tidal environment in Mont Saint-Michel Bay (the Channel, France). Comptes Rendus de L'Acadamie des Sciences. Paris, 327, 583-588.

Falconer, R.A. (1983) Numeric Model of Nutrients in Poole Harbour and Holes Bay. Internal Report for WWA Dept of civil Engineering. University of Birmingham. UK 89pp

Falconer, R.A. (1986) Water quality simulation study of a natural harbour. Journal of Waterway, Port, Coastal and Ocean Engineering, 112:15-34.

Farnham, W., Murfin, C., Critchley, A., and Morrell, S. (1981). Distribution and control of the brown alga *Sargassum muticum*. In: Proceedings of the Xth International Seaweed Symposium, 277-282.

Fonseca, M.S., (1992). Restoring seagrass systems in the United States. In: Restoring the Nation's Marine Environment. Maryland Sea Grant College, Maryland, USA. 79-110 pp.

Fretter, V. and Graham, A., (1981). The Prosobranch molluscs of Britain and Denmark, part 6. Journal of Molluscan Studies, Supplement 9: 309-313.

Fry, D.M. (1995). Reproductive effects in birds exposed to pesticides and industrial chemicals. Environ. Health Perspect. 103, 165-171.

GESAMP (1993). Impact of oil and related chemicals and wastes on the marine environment. Reports and Studies of the Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), (50), ix,180p

Goss-Custard, J.D. and Moser, M.E. (1988). Rates of change in the numbers of dunlin, *Calidris alpina,* wintering in British estuaries in relation to the spread of *Spartina anglica.* Journal of Applied Ecology, 25:95-109

Gray, A.J. (1985). Poole Harbour: ecological sensitivity analysis of the shoreline. A report prepared for BP Petroleum Development Ltd. 36p. Huntingdon: Institute of Terrestrial Ecology.

Gray, A.J., Marshall, D.F.and Raybould, A.F. (1991). A century of evolution in *Spartina anglica*. Advances in Ecological Research, 21:1-62,

Gray, A.J., Warman, E.A., Clarke, R.T., and Johnson, P.J. (1995) The niche of Spartina anglica on a changing coastline. In Coastal Zone Topics: Process, Ecology and Management, pp. 29-34.

Gray, A. (1986) Do invading species have definable genetic characteristics? Phil. Trans. Royal Society London B314 655-674

Green, F.H.W. (1940) Poole Harbour. A Hydrographic Survey. London: Geographical Publications

Grimwood, M.J. and Dixon, E. (1997). Assessment of risks posed by List II metals to Sensitive Marine Areas (SMAs) and adequacy of existing environmental quality standards (EQSs) for SMA protection. Report to English Nature.

Gundlach, E.R. and Hayes, M.O. (1978).Vulnerability of coastal environments to oil spill impacts. Marine Technology Society Journal, 12:18-27

Hall, A.J., Law, R.J., Wells, D.E., Harwood, J., Ross, H.M., Kennedy, S., Allchin, C.R., Campbell, L.A. and Pomeroy, P.P. (1992) Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 phocine distemper epizootic. Science of the Total Environment, 115(1/2):145-162

Hanrahan, G., Gledhill, M., House, W.A. and Worsfold, P.J. (2001) Phosphorus Loading in the Frome Catchment, UK: Seasonal Refinement of the Coefficient Modeling Approach. Journal of Environmental Quality 30(5):1738-1746

Hiscock, K., (1987). The distribution and abundance of *Zostera marina* in the area of Littlewick Bay, Milford Haven, with an account of associated communities and hydrocarbon contamination of sediments. Surveys undertaken in 1986. Final report to the Institute of Petroleum., London. Field Studies Council, Pembroke, Wales. 34 pp

Holme N.A. and. Bishop, G.M (1980) Survey of the littoral zone of the coast of Great Britain. 5. Report on the sediment shores of Dorset, Hampshire and the Isle of Wight. Scottish Marine Biological Association and Marine Biological Association of the United Kingdom, Intertidal Survey Unit, 1980. (Nature Conservancy Council Contract Report) 180p

Howard, S. and Moore, J. (1988) Surveys of harbours, rias and estuaries in southern Britain: Poole Harbour. 35p. Peterborough: Nature Conservancy Council, (Unpub. NCC CSD Rep. 896)

Houghton, D.R.and Millar, R.H. (1960) Spread of *Styela mammiculata* (Carlisle). Nature, 185: 862

Hubbard, J.C.E.and Stebbings, R.E. (1968). *Spartina* marshes in southern England. VII. Stratigraphy of the Keysworth Marsh, Poole Harbour. Journal of Ecology, 56:707-722

Hunt, S. and Hedgecott, S. 1992. Revised Environmental Quality Standards for chromium in water, WRc report to the Department of the Environment DoE 2858/1.

Jacobs, R.P.W.M., (1980). Effects of the Amoco Cadiz oil spill on the seagrass community at Roscoff, with special reference to the benthic infauna. Mar. Ecol. Prog. Ser., 2: 207-212

Jensen, A.C., Collins, K.J., Free, E.K., and Bannister, R.C.A. (1994) Lobster (*Homarus gammarus*) movement on an artificial reef: the potential use of artificial reefs for stock enhancement. In Crustaceana, 67:198-211.

Kirby, M.F., Blackburn, M.A., Thain J.E. and Waldock, M.J. (1998). Assessment of water quality in estuarine and coastal waters using a contaminant concentration technique. Marine Pollution Bulletin, 38, 631-642.

Kirubagaran R and Joy KP (1991). Changes in adrenocortical-pituitary activity in the catfish, *Clarias batrachus* (L.) after mercury treatment. *Ecotoxicol. Environ. Safety*. **22**: 36-44

Kocan, R.M., Marty, G.D. Okihiro, M.S., Brown, E.D. and Baker, T.T. (1996) Reproductive success and histopathology of individual Prince William Sound Pacific herring 3 years after the Exxon Valdez oil spill. Canadian Journal of Fisheries and Aquatic Sciences. 53(10):2388-2393

Langston, W.J. (1980). Arsenic in U.K. estuarine sediments and its availability to benthic organisms. J.Mar.Biol.Ass.U.K., 60, 869-881.

Langston, W.J. (1982). The distribution of mercury in British estuarine sediments and its availability to deposit-feeding bivalves. J.Mar.Biol.Ass.U.K., 62, 667-684.

Langston, W.J. (1985). Assessment of the distribution and availability of arsenic and mercury in estuaries. In Estuarine Management and Quality Assessment (eds. Wilson, J.G. & Halcrow, W.), Plenum Press, New York, pp. 131-146.

Langston, W. J. and Burt, G. R. (1991) Bioavailability and effects of sediment-bound TBT in deposit-feeding clams, *Scrobicularia plana*. Marine Environmental Research, 32,61-77.

Langston, W.J. and Zhou, M. (1987) Cadmium accumulation, distribution and metabolism in the gastropod *Littorina littorea*: The role of metal-binding proteins. *J. mar.biol. Ass. U.K.*, 67, 585-601.

Langston, W.J., Burt, G.R. & Zhou M.(1987). Tin and organotin in water, sediments, and benthic organisms of Poole Harbour. Mar.Pollut.Bull., 18, 634-639.

Langston, W.J., Bryan, G.W., Burt, G.R. and Gibbs, P.E. (1990). Assessing the impact of tin and TBT in estuaries and coastal regions. Functional Ecology, 4, 433-443

Langston, W.J., Bryan, G.W., Burt, G.R.(1994a). Heavy metals in UK estuaries:PML data and mapping programme,R&D Note 280, National Rivers Authority, 85pp.

Langston, W.J., Bryan, G.W., Burt, G.R. and Pope, N.D. (1994b) Effects of sediment metals on estuarine benthic organisms, R&D Note 203, National Rivers Authority , 141pp

Law R.J., Dawes V.J., Woodhead R.J. & Matthiessen P. (1997) Polycyclic aromatic hydrocarbons (PAH) in seawater around England and Wales. *Marine Pollution Bulletin* **34,** 306-322.

Leah, R., Johnson, M.S., Connor, L. and Levene, C.F (1997). Polychlorinated biphenyls in fish and shellfish from the Mersey Estuary and Liverpool Bay. Marine Environmental Research. 43(4):345-358

Leonard, K.S., M^{c}cubbin, D & Harvey, B.R. (1993) A radiochemical procedure for the determination and speciation of radiocobalt in environmental waters. Science of the Total Environment, 130/131, 237-251.

Long, S.P. (1990). The primary productivity of *Puccinellia maritima* and *Spartina anglica*: a simple predictive model of response to climatic change. In: Beukema, J.J.,Wolff, W.J., Brouns, J.J.W.M., (Eds). Expected effects of climatic change on marine coastal ecosystems. p.33-39. Dordrect: Kluwer Academic Publishers, (Developments in Hydrobiology 57)

Macdonald, D.D., Carr, R. S., Calder, F.D., Long, E.R.; & Ingersoll, C.G. (1996). Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology, 5(4), 253-278

MAFF (1994) Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1992. Aquatic Environment Monitoring Report, 40, 83pp

Malins, D.C., McCain, B.B., Landahl, J.T, Myers, M.S., Krahn, M.M., Brown, D.W., Chan, S.L. and Roubal, W.T. (1988). Neoplastic and other diseases in fish in relation to toxic chemicals: an overview. Aquatic Toxicology 11:43-67

Mallett, M.J., Vine, S. Murgatroyd, C., Whitehouse, P., Jerman, E., Ashbycrane,R.E., Fleming, R., Wilson, K. and Sims, I. (1992). Toxicity of common pollutants to freshwater life. A review of the effects of ammonia, arsenic, cadmium, chromium, copper, cyanide, nickel, phenol and zinc on indigenous species. NRA R&D Note 82.

Miller, D.C., Poucher, S., Cardin, J.A. and Hansen, D. (1990). The acute and chronic toxicity of ammonia to marine fish and a mysid, Archives of Environmental Toxicology, 19:40-48.

Misitano, D.A., Casillas, E., & Haley, C.R . (1994). Effects of contaminated sediments on viability, length, DNA and protein content of larval surf smelt, *Hypomesus pretiosus.* Marine Environmental Research, 37, 1-21.

Moore, A. and Waring, C.P. (1996) Sublethal effects of the pesticide diazinon on olfactory function in mature male Atlantic salmon parr Journal of Fish Biology 48(4):758-775

Morgan, C. (2000). The ecology and fishery of the Manila Clam *Tapes philipparium* in Poole Harbour. Shellfish News 10 CEFAS

MPMMG (Marine Pollution Monitoring Management Group). (1998). National Monitoring Programme Survey of the Quality of UK Coastal waters. Marine Pollution Monitoring Management Group, Aberdeen, ISBN 0 9532838 36

Murdoch, N. and Randall, D. (2001). Poole Harbour Water Quality Model (Nitrogen). EA Report. 22pp

Nicholson, M. & Barry, J. (1995) Inferences from spatial surveys about the presence of an unobserved species. In Oikos, Vol. 72, pp. 74-78.

Nixon, S.C., Gunby, A., Ashley, S.J., Lewis, S. and Naismith, I. (1995). Developmentand testing of General Quality Assessment schemes: dissolved oxygen and ammonia in estuaries. NRA Project Record 469/15/HO.

Norris DO, Donahue S, Dores RM, Lee JK, Maldonado TA, Ruth T and Woodling JD (1999). Impaired adrenocortical response to stress by brown trout, *Salmo trutta*, living in metal-contaminated waters of the Eagle River, Colorado. *Gen. Comp. Endocrinol*. **113**: 1-8.

NRA (1990). Marine algal monitoring programme.

Ohlendorf, H.M. & Fleming, W.J. (1988) Birds and environmental contaminants in San Francisco and Chesapeake Bays. Marine Pollution Bulletin, 19, 487-495

Ohlendorf, H.M., Hothem, R.L., Bunck, C.M., Aldrich, T.W. & Moore, J.F. (1986). Relationships between selenium concentrations and avian reproduction. Trans. 51st N.A. Wildlife & Nat. Res. Conf. 330-342

Oliver, F.W. (1925). *Spartina townsendii*; its mode of establishment, economic uses and taxonomic status. Journal of Ecology 13:74-91

OSPARCOM (1994). Ecotoxicological Asessment Criteria for trace metals and organic microcontaminants in the North-East Atlantic. Oslo and Paris Commissions.

Raffaelli, D.G., Raven, J.A., and Poole, L.J. (1998) Ecological impact of green macroalgal blooms. In Oceanography and Marine Biology. Annual Review, 36:97-125.

Ranwell, D.S., Bird, E.C.F., Hubbard, J.C.F. and Stebbings, R.E. (1964). *Spartina* salt marshes in southern England. V Tidal submergence and chlorinity in Poole Harbour. Journal of Ecology, 52:627-641

Raybould, A.F. (2000) British saltmarshes. Hydrographical, ecological and evolutionary change associated with *Spartina anglica* in Poole Harbour (eds B.R. Sherwood, B.G. Gardiner and T. Harris), pp. 129-142. Linnean Society of London.

Raybould, A.F., Gray, A.J., and Clarke, R.T. (1998) The long-term epidemic of Claviceps purpurea on *Spartina anglica* in Poole Harbour: pattern of infection, effects on seed production and the role of *Fusarium heterosporum*. New Phytologist 138:497-505.

OSPARCOM (2000). Quality Status Report 2000. Oslo and Paris Commission, London.108pp

Owens, N.J.P.and Stewart, W.D.P.(1983) *Enteromorpha* and the cycling of nitrogen in a small estuary. Estuarine, Coastal and Shelf Science, 17:287-296

Payne, J.F., Kiceniuck, J., Fancey., L.L., Williams, U., Fletcher, G.L., Rahimtula, A. and Fowler, B. (1988). What is a safe level of polycyclic aromatic hydrocarbons for fish: subchronictoxicity study on winter flounder, (*Pseudopleuronectes americanus*). Can. J. Fish. Aquat. Sci. 45, 1983-1993

Posford Duvivier Environmental (1992). Capital and maintenance dredging – apilot case study to review the potential benefits for nature conservation. Report for English Nature and Poole Harbour Conmmissioners. March 1992, 82pp

Preston, A., Jefferies, D.F., Dutton, J.W.R., Harvey, B.R. & Steele, A.K. (1972). British Isles coastal waters: the concentrations of selected heavy metals in sea water, suspended matter and biological indicators - a pilot survey. Environmental Pollution, 3, 69-82.

Reijnders, P.J.H.(1986) Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature, 324:456-457

Reddy PS, Tuberty SR and Fingerman M (1997). Effects of cadmium and mercury on ovarian maturation in the red swamp crayfish, *Procambarus clarkii*. *Ecotoxicol. Environ. Safety* **37**: 62-65.

RIFE (2001). Radioactivity in Food and the Environment. RIFE-6. Food Standards Agency and Scottish Environment Protection Agency. Report produced by CEFAS: ISSN 1365-6414. 183p

Rodriguez EM, Lopez Greco LS and Fingerman M (2000). Inhibition of ovarian growth by cadmium in the fiddler crab *Uca pugilator* (Decapoda; Ocypodidae). *Ecotox. Env.Safety* **46**: 202-206.

Savage, P.D.V. (1971). The seaweed problem at Poole power station. Laboratory Note Central Electricity Research Laboratories, No. RD/L/N/ 263/71, 12p.

Seager, J., Wolff, E.W. and V.A. Cooper. (1988). Proposed environmental quality standards for List II substances in water. Ammonia. WRc report No TR26O.

Shucksmith, (1982) Poole Harbour Surveys (1982) Avon and Dorset Division Scientific Services. Wessex Water Authority

Spencer, B.E., Edwards, D.B., Kaiser, M.J., and Richardson C.A. (1994). Spatfalls of the non-native Pacific oyster (*Crassostrea gigas*) in British waters. Aquatic conservation: Marine and Freshwater Ecosystems, 4: 203-217.

Stapf, O. (1914) Report on *Spartina* vegetation in Poole Harbour. Unpublished report for Poole Harbour Commissioners

Steidinger, K.A.and Vargo, G.A. (1988) Marine dinoflagellate blooms: dynamics and impacts. In: Algae and human affairs. Lembi, C.A., Waaland, J.R.(Eds) p373-401. Cambridge University Press

Stewart, M.E., Blogoslawski, W.J., Hsu, R.Y. and Helz, G.R. (1979) By-products of oxidative biocides: toxicity to oyster larvae. Marine Pollution Bulletin, 10:166-169

Thain, J.E., Allen, Y., Kirby, S. & Reed, J. (2000). The use of sediment bioassays in monitoring and surveillance programs in the UK: a preliminary assessment. ICES Council Meeting Papers, C.M. 2000/S:10, 12p

Thomas P (1989). Effects of Aroclor 1254 and cadmium on reproductive endocrine function and ovarian growth in Atlantic croaker. *Mar. Env. Res.* **28**: 499-503.

Thornton, I. , Watling, H., and Darracott, A. (1975). Geochemical studies in several rivers used for oyster rearing. The Science of the Total Environment, 4, 325-345.

Valiela, I., J. McClelland, J. Hauxwell, P.J. Behr, D. Hersh, and Foreman. K. (1997). Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. Limnology and Oceanography 42:1105-1118.

von Westernhagen, H., Rosenthal, H., Dethlefsen, V., Ernst, W., Harms, Hansen, P.-D. (1981). Bioaccumulating substances and reproductive success in Baltic flounder Platichthys flesus. Aquatic Toxicology, 1: 85-99

Warwick, R.M. (1986). A new method for detecting pollution effects on marine macrobenthic communities. Marine Biology, 92: 557-562

Warwick, R. M., George, C. L., Pope, N.D. and Rowden, A.A (1989). The prediction of post-barrage densities of shorebirds. Volume 3. Invertebrates. Harwell, Department of Energy: 112pp.

Warwick, P.E., Cundy, A.B., Croudace, I.W., Bains, M.E.D. & Dale, A.A. (2001). The uptake of iron-55 by marine sediment, macroalgae, and biota following discharge from a nuclear power station. Environmental Science and Technology, 35(11), 2171-2177.

Wessex Water Authority (1981) Poole Harbour Nutrient Surveys Avon and Dorset Division Scientific Services. Wessex Water Authority

White, S.J. (1991). The effects of effluent from Poole sewage treatment works upon the intertidal macrofauna of Holes Bay, Poole Harbour. NRA (Wessex Region) report, SW/22, 11pp.

White, E.M. and Knights, B. (1994) Elver and eel stock assessment in the Severn and Avon. Applied Ecology Research Group, University of Westminster. Report to the NRA pp141

Williams, T.D. and Brown, B.R.H. (1992). The acute and chronic toxicity of ammonia to the marine copepod *Tisbe battagliai.* ICI Group Experimental Laboratory, Brixham. Report No.BL440/B.

Willis, B., Rea, J., Crookes, M., Howe. and Dobson. (1994). Environmental Hazard Assessment: Carbon Tetrachloride. BRE Report No TSD/21

Withers, R.G., Farnham, W.F., Lewey, S., Jephson, N.A., Haythorn, R.M., and Gray, P.W.G. (1975). The epibiota of *Sargassum muticum* in British waters. Marine Biology, 31: 79-81

Woodhead R.J., Law R.J. & Matthiessen P. (1999) Polycyclic aromatic hydrocarbons in surface sediments around England and Wales, and their possible biological significance. Marine Pollution Bulletin 38, 773-790.

Zieman, J.C., Orth, R., Phillips, R.C., Thayer, G.W., Thorhaug, A.T., (1984). The effects of oil on seagrass ecosystems. In: Restoration of Habitats Impacted by Oil Spills. Butterworth Publishers, Boston, Massachusetts, USA. pp. 36-64

Zinkl, J.G., Lockhart, W.L., Kenny, S.A. and Ward, F.J. (1991) The effects of cholinesterase inhibiting insecticides on fish. In: Mineau, P., (editor). Cholinesterase-inhibiting insecticides. Their impact on wildlife and the environment. .233-254. Elsevier

12. ANNEXES

Annex 1. Poole Harbour SPA: Summary Of The Interest Features

Poole Harbour SPA: Summary Of The Interest Features And Conservation Objectives (Adapted From English Nature, 2001)

Internationally important populations of regularly occurring Annex 1 bird species (avocet, mediterranean gull, common tern*)*

- Maintain numbers by avoidance (no increase) in disturbance to feeding , roosting and nesting areas: maintain absence of obstructions to view lines

Habitats

- Shallow inshore waters
  - maintain extent and distribution of habitat and food avilability (avocets - marine insects, crustaceans, molluscs, fish and worms; common tern – sandeel, sprat, crustacea and annelids; Med.gull –crustaceans, nnelids, gobies, molluscs)
- Intertidal sediment communities
  - maintain extent and distribution and thus food for avocets and gulls

- Saltmarsh
  - maintain extent and distribution and thus food supply for avocets and gulls
  - maintain vigitation characteristics(<10cm height) for roosting avocets, terns and gulls

Internationally important assemblage of waterfowl (dunlin, cormorant, dark-bellied brent geese, teal, goldeneye, red-breasted merganse, curlew, spotted redshank, greenshank, redshank, black-headed gull) including the internationally important populations of regularly occurring migratory species (shelduck and black-tailed godwit). The estuary supports over 20,000 wintering wildfowl

- Maintain numbers by avoidance (no increase) in disturbance to feeding , roosting and nesting areas: maintain absence of obstructions to view lines

Habitats

- Shallow inshore waters including lagoons
  - maintain extent and distribution of habitat and food avilability (includes mussels, cockles, *Hydrobia, Carcinus*, small fish, *shrimps, nereis*)

- Intertidal sediment communities
  - maintain extent and distribution of habitat and food availability (including *Nereis, Hydrobia, Corophium, Macoma, Cardium,Crangon, Carcinus, Littorina, Gammarus, Arenicola*, small fish, *Zostera* and *Enteromorpha*)*.
  - maintain absence of obstructions to view lines

- Saltmarsh
  - maintain extent and distribution of habitat and food availability * (above species, together with *Salicornia, Atriple, Spegularia,Puccinellia, Triglochin, Aster, Plantego*)
  - maintain vegitation characteristics (<10cm height) i.e. absence of obstructions to view lines

- Reedbeds
  - maintain extent and distribution of habitat
  - maintain presence and abundance of intertidal invertebrates (e.g. *Nereis, Corophium, Hydrobia*).

Annex 2. Water Quality Standards

List I (EC Dangerous Substances Directive) and list II substances (from Cole *et al.,* 1999, derived by WRc according to the methodology described in Grimwood and Dixon 1997)

List I substances

Parameter	**Unit**	**Water quality standard**		**Standstill Provision** [a]
		Estuary[b]	**Marine**	
Mercury	μg Hg l-1	0.5 DAA	0.3 DAA	yes[c]
Cadmium	μg Cd/l	5 DAA	2.5 DAA	yes
Hexachlorocyclohexane[d]	μg HCH/l	0.02 TAA	0.02 TAA	yes
Carbon tetrachloride	μg CCl_4/l	12 TAA	12 TAA	no
Dichlorodiphenyltrichloroethane				
(all 4 isomers, total DDT)	μg DDT/l	0.025 TAA	0.025 TAA	yes
(para, para-DDT)	μg ppDDT/l	0.01 TAA	0.01 TAA	yes
Pentachlorophenol	μg PCP/l	2 TAA	2 TAA	yes
Total □drins□	μg l-1	0.03 TAA	0.03 TAA	yes
Aldrin	μg l-1	0.01 TAA	0.01 TAA	yes
Dieldrin	μg l-1	0.01 TAA	0.01 TAA	yes
Endrin	μg l-1	0.005 TAA	0.005 TAA	yes
Isodrin	μg l-1	0.005TAA	0.005 TAA	yes
Hexachlorobenzene	μg HCB/l	0.03 TAA	0.03 TAA	yes
Hexachlorobutadiene	μg HCBD/l	0.1 TAA	0.1 TAA	yes
Chloroform	μg $CHCl_3$/l	12 TAA	12 TAA	no
1,2-Dichloroethane (ethylenedichloride)	μg EDC/l	10 TAA	10 TAA	no
Perchloroethylene (tetrachloroethylene)	μg PER/l	10 TAA	10 TAA	no
Trichlorobenzene (all isomers)	μg TCB/l	0.4 TAA	0.4 TAA	yes
Trichloroethylene	μg TRI/l	10 TAA	10 TAA	no

Notes: Substances are listed in order of publication of Directives.

D Dissolved concentration, ie usually involving filtration through a 0.45-μm membrane filter before analysis

T Total concentration (ie without filtration).

AA standard defined as annual average

[a] Most directives include, in addition to the standards for inland, estuary and marine waters, a provision that the total concentration of the substance in question in sediments and/or shellfish and/or fish must not increase significantly with time (the "standstill" provision).

[b] In the UK the standards for estuaries are the same as for marine waters - The Surface Waters (Dangerous Substances) (Classification) Regulations 1989

[c] In addition to a standstill provision applying to sediments or shellfish there is a further environmental quality standard of 0.3 mg Hg/kg wet flesh "in a representative sample of fish flesh chosen as an indicator".

[d] All isomers, including lindane

Annex 2 (cont.) Water quality standards for the protection of saltwater life.

List II substances

Parameter	Unit	WQS (see footnotes)	Uncertainties in the derivation : Details obtained from the relevant EQS derivation reports
Lead	µg Pb/l	25 AD[1,5]	The preliminary EQS was multiplied by a factor of 2 to account for overestimation of Pb toxicity in laboratory studies compared to the field environment. The EQS was considered tentative as a result of the paucity of reliable data, in particular for sub-lethal chronic studies with invertebrates and fish, and for field studies.
Chromium	µg Cr/l	15 AD[1,5]	There were limited data on the sub-lethal effect of Cr and long-term exposure to freshwater and saltwater life. Separate standards for different Chronium valences (Cr(VI) and Cr(III)) were not recommended as a consequence of the lack of data for Cr(III). In addition, a comparison of the toxicities of each oxidation state was not possible. Some data were available that indicated higher sensitivity of some saltwater organisms to low salinities. The EQS was based on data generated at salinities typical of normal seawater. Therefore, further research on the effect of Cr at lower salinities was recommended.
Zinc	µg Zn/l	40 AD[1,5]	The dataset available for the toxicity of Zn to saltwater life illustrated that at the EQS, adverse effects on algal growth had been reported. However, it was considered that there was currently insufficient evidence to suggest that the EQS would not adequately protect saltwater communities.
Copper	µg Cu/l	5 AD[1]	Further data were considered necessary on the sensitivity of early life stages and life-cycle tests to confirm the sensitivity of saltwater life.
Nickel	µg Ni/l	30AD[1]	Marine algae were reported to be adversely affected by Ni at concentrations as low as 0.6 _g l^{-1} which is below the EQS to protect saltwater life However, it was considered that there was insufficient evidence to justify a lower EQS based solely on results with algae and that further research into this area was desirable. There was also limited evidence to suggest that invertebrates in estuarine systems may be more susceptible to the effects of Ni than invertebrates in marine systems. Thus, an EQS to protect estuarine life may be needed in future when further data become available.
Arsenic	µg As/l	25AD[2]	Based on crab 96 hour LC50, and an extrapolation factor of 10 applied. Standards may need to be more stringent where sensitive algal species are important features of the ecosystem
Boron	µg B/l	7000 AT[1]	Few data available. However the standard was based on Dab 96 hour LC50, with an extrapolation factor of 10 applied
Iron	µgFe/l	1000AD[1,5]	The EQS for the protection of saltwater life was based on observed concentrations and general assessments of water quality. It was recommended, therefore, that the standard should be reviewed as soon as direct observations of water concentrations and biological status become available. Limited data did not allow an assessment of the importance of Fe species.
Vanadium	µgV/l	100 AT[1]	Data on the toxicity of vanadium on saltwater life were limited. As there were limited data for vanadium, it was not possible to recommend standards based on dissolved concentrations or separate standards for migratory fish. With regard to the latter, it may be necessary to base judgement of any risk in applying the EQS on knowledge of local risks and circumstances.

Tributyltin	µg l-1	0.002 MT[2]	The standards for TBT weres tentative to reflect a combination of the lack of environmental data, toxicity data or data relating to the behaviour of organotins in the environment.
Triphenyltin (and its derivatives)	µg l-1	0.008 MT[2]	The standards for TPT were tentative to reflect a combination of the lack of environmental and toxicity data or data relating to the behaviour of organotins in the environment.
PCSDs	µg l-1	0.05 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Cyfluthrin	µg /l	0.001 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values
Sulcofuron	µg /l	25 PT[1]	As a consequence of the general paucity of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values. The data for sulcofuron suggested that embryonic stages for saltwater invertebrates could be more sensitive than freshwater species and, therefore, the EQS for the protection of marine life, derived from the freshwater value, may need to be lower.
Flucofuron	µg /l	1.0 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were based on freshwater values.
Permethrin	µg /l	0.01 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Atrazine and Simazine	µg /l	2 AA[2] 10 MAC[4]	The EQSs for the protection of saltwater life were proposed as combined atrazine/simazine to take account of the likely additive effects when present together in the environment.
Azinphos-methyl	µg /l	0.01AA[2] 0.04 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Dichlorvos	µg /l	0.04 AA 0.6 MAC[2]	Based on data for sensitive crustaceans
Endosulphan	µg /l	0.003 AA[2]	There is little evidence on the ultimate fate of endosulfan and its metabolites or degradation products in sediments and on any effects on freshwater benthic organisms. Consequently, it is possible that some sediment-dwelling organisms, such as crustaceans, may be at risk.
Fenitrothion	µg /l	0.01 AA[2] 0.25 MAC[4]	As there were limited data with which to derive EQSs to protect saltwater life, the freshwater values were adopted. However, the annual average for the protection of freshwater life may be unnecessarily stringent in view of the uncertainties associated with the acute toxicity data used in its derivation. The uncertainties exist because the original sources were unavailable for certain studies. Lack of confirmatory data existed in the published literature and data for warm water species were considered in the derivation.
Malathion	µg /l	0.02AA[2] ;0.5MAC[4]	It was recommended that further investigation for both field and laboratory conditions into the effects of malathion on crustaceans and insects and on UK *Gammarus* species, in particular, should be carried out.

Trifluralin	µg /l	0.1AA[2] 20 MAC[4]	None mentioned with regard to the annual mean.
4-chloro-3-methyl phenol	µg /l	40 AA[3] 200 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2-chlorophenol	µg /l	50 AA[3] 250 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4-dichlorophenol	µg /l	20 AA[3] 140 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4D (ester)	µg /l	1 AA[3] 10 MAC[4]	For the EQS proposed for 2,4-D esters, comparison of the data and derivation of standards were complicated by the number of esters and organisms for which studies were available. In addition, the toxicity of the esters may have been underestimated in some of the studies due to their hydrolysis. There were limited data on the toxicity of 2,4-D ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
2,4D	µg /l	40 AA[3] 200 MAC[4]	There were limited data on the toxicity of 2,4-D non-ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
1,1,1-trichloroethane	µg /l	100 AA[3] 1000 MAC[4]	The 1,1,1-TCA dataset available for freshwater species contained comparatively few studies where test concentrations were measured and, consequently, comparison of studies using measured concentrations vs. those using nominal values indicated that data from the latter type of study could be misleading.
1,1,2-trichloroethane	µg /l	300 AA[3] 3000 MAC[4]	For 1,1,2-TCA, few data were available on chronic toxicity to freshwater fish. There were limited data on the toxicity of 1,1,2-TCA to saltwater life and, consequently, the EQS to protect freshwater life was adopted.
Bentazone	µg /l	500 AA[3] 5000 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Benzene	µg /l	30 AA[3] 300 MAC[4]	Limited and uncertain chronic data available.
Biphenyl	µg /l	25 AA[3]	The data available for marine organisms were considered inadequate to derive an EQS for the protection of marine life. However, the reported studies for saltwater organisms indicate that the EQS for freshwater life will provide adequate protection.
Chloronitrotoluenes (CNTs)	µg /l	10 AA[3] 100 MAC[4]	The dataset used to derive the EQS to protect freshwater life was limited. Toxicity data were available for comparatively few species and there was limited information on the bioaccumulation potential of the isomers. There were few chronic studies available to allow the assessment of the long term impact of CNTs. There were no reliable data for the toxicity to or bioaccumulation of CNTs by saltwater species and, therefore, the EQSs proposed for freshwater life were adopted.
Demeton	µg /l	0.5 AA[3] 5 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.

Dimethoate	µg /l	1 AA[3]	The available data for marine organisms were considered inadequate to derive an EQS for the protection of marine life. Crustaceans were considered to be the most sensitive organisms, but more data are required to confirm this. In view of the uncertainties associated with the marine toxicity dataset, the freshwater EQS was adopted. This was based on the toxicity of dimethoate to insects. Although there are no marine insects, there is some evidence that marine organisms are more sensitive than their freshwater counterparts.
Linuron	µg /l	2 AA[3]	In view of the lack of data for saltwater life, the EQS proposed for the protection of freshwater life was adopted until further data become available.
Mecoprop	µg /l	20 AA[3] 200 MAC[4]	There were limited data relating to the toxicity of mecoprop to aquatic life. The dataset for saltwater life comprised data for one marine alga, a brackish invertebrate and a brackish fish. Consequently, the freshwater values were adopted until further data become available.
Naphthalene	µg /l	5 AA[3] 80 MAC[4]	Limited and uncertain chronic data available.
Toluene	µg /l	40 AA[3] 400 MAC[4]	The dataset used to derive the EQS to protect saltwater life relied on static tests without analysis of exposure concentrations. Consequently, the derived values are considered tentative until further data from flow-though tests with analysed concentrations become available.
Triazophos	µg /l	0.005 AA[3] 0.5 MAC[4]	The dataset available for freshwater life was limited to a few studies on algae, crustaceans and fish. No information was available for the target organisms (insects), on different life-stages or on its bioaccumulation in aquatic organisms. There were no data on the toxicity or bioaccumulation of triazophos in saltwater organisms. Consequently, the EQSs to protect freshwater life were adopted until further data become available.
Xylene	µg /l	30 AA[3] 300 MAC[4]	Limited information available. Freshwater data used to § back up§ the standards.

Notes

Substances are listed in the order of publication of Directives.

A annual mean
D dissolved concentration, ie usually involving filtration through a 0.45-µm membrane filter before analysis
T total concentration (ie without filtration)
µg/ l micrograms per litre
AA standard defined as annual average
MAC maximum concentration
[1] DoE Circular in 1989 (Statutory standard)
[2] Statutory Instrument 1997 (Statutory standard)
[3] Statutory Instrument 1998 (Statutory standard)
[4] Non- statutory standard
[5] revised standards have been proposed but are not statutory

Annex 3. Quality Standards Stipulated In The Shellfish Waters Directive

(from Cole *et al.,* 1999)

Parameter	Unit	G	I
A. GENERAL PHYSIO-CHEMICAL PARAMETERS			
Colour			(a)
Dissolved oxygen	% sat	>80 T95	>70 TAA(b)
pH			7-9 T75
Salinity	g/kg	12-38 T95	40 T95(c)
Suspended solids			(d)
Tainting substances			(e)
Temperature		(f)	
B. METALS AND INORGANIC ANIONS			
Arsenic		(g)	(h)
Cadmium		(g)	(h)
Chromium		(g)	(h)
Copper		(g)	(h)
Lead		(g)	(h)
Mercury		(g)	(h)
Nickel		(g)	(h)
Silver		(g)	(h)
Zinc		(g)	(h)
C. ORGANIC SUBSTANCES			
Hydrocarbons			(i)
Organohalogens		(g)	(h)
D. MICROBIOLOGICAL PARAMETER			
Faecal coliforms	per 100 ml	300 T75(j)	

Notes:

G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)
D dissolved concentration ie usually involving filtration through a 0.45-µm membrane filter before analysis
AA standard defined as annual average
75 standard defined as 75-percentile
95 standard defined as 95-percentile
MA maximum allowable concentration
Pt/l concentration of platinum (Pt) determined photometrically on the Platinum/Cobalt scale as a measure of colour in water

[a] A discharge affecting shellfish waters must not cause an increase in colouration of more than 10 mg Pt/l compared to the waters not so affected (waters filtered in both cases). This standard is expressed as a 75-percentile.

[b] If an individual result indicates a value lower than 70% of saturation, the measurement must be repeated. Concentrations below 60% of saturation are not allowed, unless there are no harmful consequences for the development of shellfish colonies.

[c] A discharge affecting shellfish waters must not cause an increase in salinity of more than 10% compared to the water not so affected. This standard is expressed as a 75-percentile.

[d] A discharge affecting shellfish waters must not cause an increase in the concentration of suspended solids by more than 30% compared to the water not so affected. This standard is expressed as a 75-percentile.

[e] The concentration of substances affecting the taste of shellfish must be lower than that liable to impair the taste of the shellfish.

[f] A discharge affecting shellfish waters must not cause an increase in temperature of more than 2 °C compared to the water not so affected. This standard is expressed as a 75-percentile.

[g] The concentration of this substance or group of substances in shellfish flesh must be so limited that it contributes to the high quality of shellfish products.

[h] The concentration of this substance or group of substances in water or in shellfish flesh must not exceed a level which gives rise to harmful effects in the shellfish or their larvae. Synergistic effects must also be taken into account in the case of metal ions.

[i] Hydrocarbons must not be present in water in such quantities as to produce a visible film on the surface of the water and/or a deposit on the shellfish, or to have harmful effects on the shellfish.

[j] In shellfish flesh and intervalvular fluid. However, pending the adoption of a directive on the protection of consumers of shellfish products, it is essential that this value be observed in waters from which shellfish are taken for direct human consumption.

Annex 4. Bathing Waters Quality Standards

Quality standards for fresh and saline waters stipulated in the Bathing Waters Directive (from Cole *et al.,* 1999)

Parameter	Unit	G	I
A. INORGANIC SUBSTANCES AND GENERAL PHYSICO-CHEMICAL PARAMETERS			
Colour			(a, b)
Copper	mgCu/l		
Dissolved oxygen	% saturation	80-120 T90	
pH			6-9 T95(b)
Turbidity	Secchi depth m	>2 T90	>1 T95(b)
B. ORGANIC SUBSTANCES			
Floating waste(c)		(d)	
Hydrocarbons	μg l-1	300 T90(e)	(f)
Phenols	μgC_6H_5OH	5 T90(e)	50 T95(e)
Surfactants(g)	μg l-1 as lauryl sulphate	300 T90(e)	(k)
Tarry residues		(d)	
C. MICROBIOLOGICAL PARAMETERS			
Faecal coliforms	per 100 ml	100 T80	2 000 T95
Total coliforms	per 100 ml	500 T80	10 000 T95
Faecal streptococi	per 100 ml	100 T90	
Salmonella	per 1 l		0 T95
Entero viruses	PFU/10 l		0 T95

Notes

G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)80 standard defined as 80-percentile*
90 standard defined as 90-percentile*
95 standard defined as 95-percentile*

It is further stipulated that of the 20, 10 or 5% of samples from designated waters which exceed the standard, none should do so by more than 50% (except for microbiological parameters, pH and dissolved oxygen) and that "consecutive water samples taken at statistically suitable intervals do not deviate from the relevant parametric values" (Article 5 of CEC 1976).

[a] No abnormal change in colour
[b] May be waived in the event of exceptional weather or geographical conditions
[c] Defined as wood, plastic articles, bottles, containers of glass, plastic, rubber or any other substance
[d] Should be absent.
[e] Applies to non-routine sampling prompted by visual or olfactory evidence of the presence of he substance
[f] There should be no film visible on the surface and no odour
[g] Reacting with methylene blue
[k] There should be no lasting foam

Annex 5. Sediment Quality Guidelines

Interim marine sediment quality guidelines (ISQGs) and probable effect levels (PELs; dry weight) [1]: metals and organics (from Cole *et al.,* 1999)

Substance	ISQG	PEL
Inorganic (mgkg^{-1})		
Arsenic	7.24	41.6
Cadmium	0.7	4.2
Chromium	52.3	160
Copper	18.7	108
Lead	30.2	112
Mercury	0.13	0.70
Zinc	124	271
Organic (µgkg^{-1})		
Acenaphthene	6.71	88.9
Acenaphthylene	5.87	128
Anthracene	46.9	245
Aroclor 1254	63.3	709
Benz(a)anthracene	74.8	693
Benzo(a)pyrene	88.8	763
Chlordane	2.26	4.79
Chrysene	108	846
DDD[2]	1.22	7.81
DDE[2]	2.07	374
DDT[2]	1.19	4.77
Dibenz(a,h)anthracene	6.22	135
Dieldrin	0.71	4.30
Endrin	2.673	62.4[4]
Fluoranthene	113	1 494
Fluorene	21.2	144
Heptachlor epoxide	0.60[3]	2.74[4]
Lindane	0.32	0.99
2-Methylnaphthalene	20.2	201
Naphthalene	34.6	391
PCBs, Total	21.5	189
Phenanthrene	86.7	544
Pyrene	153	1 398
Toxaphene	1.5[3]	nd[5]

[1] from CCME, (1999)

[2] Sum of *p,p′* and *o,p′* isomers.

[3] Provisional; adoption of freshwater ISQG.

[4] Provisional; adoption of freshwater PEL.

[5] No PEL derived.

Annex 6. Examples of Recommended Biological Monitoring Techniques

Immunotoxicity Assays – these assay measures the immunocompetence of haemocytes from invertebrates, reflecting both the extent of exposure to immunotoxins and the general well-being of the test organism. Various immunological parameters (e.g. cell counts, generation and release of superoxide anions, phagocytosis, lysosomal enzyme activity) have proved useful in monitoring the status of shellfish in response to oil pollution and PAHs (Pipe *et al*, 2000; Raftos and Hutchinson, 1995). An example of the application of these assays in relation to Poole Harbour, is cited in section 5.1.1 (Dyrynda *et al*., 1998).

EROD (ethoxyresorufin-O-deethylase) is a marker for the activity of the mixed function oxidase (MFO) system, whose induction is usually associated with exposure to, and the detoxification of xenobiotics such as PAHs and PCBs. Occasionally these transformations may produce deleterious side effects due to the formation of carcinogenic or genotoxic compounds (e.g. formation of benzo(a) pyrene diol epoxide from the benzo(a) pyrene). Genotoxicicity assays (see below) may help to establish this possibility.

Metallothionein (MT) induction and associated changes in metal metabolism are specifically induced by metals and are sufficiently sensitive to be used to detect elevated levels of bioavailable metal in the field or arising from metals in discharges (e.g. Langston *et al*., 2002). The induction of MT protein, and associated metal-binding patterns can therefore be used to map spatial and temporal trends in biological respones to metals. Examples of the application of this assay in relation to Poole Harbour are indicated in section 5.1.1.

Genotoxicity-The Comet Assay - The single cell gel-electrophoresis (comet) assay is ideal for screening for possible genotoxicity associated with point-source and diffuse inputs to the system.

The CAPMON technique - Cardiac activity in bivalve molluscs and decapod crustaceans – Heart rate provides a general indication of the metabolic status of mussels and crabs. The CAPMON technique (Depledge and Anderson, 1990) permits the non-invasive, continuous monitoring of cardiac activity using infra-red sensors attached to the shell.

Tolerance Studies - More widespread investigations of community tolerance to establish their adaptation to contamination levels. Mapping the genetic composition of tolerant populations of individual species (*Hediste, Littorina* and others) in relation to induction of detoxification systems (such as metallothionein and EROD) should also be considered. This could add an interesting temporal dimension to biological monitoring – e.g. in determining the consequences of anticipated improvements in environmental quality (arising from planned schemes, standstill provisions of the Dangerous Substances Directive, or as required under the Habitats Directive to achieve Favourable Condition).

Toxicity Studies on sensitive species - Toxicity has been studied in a relatively small number of species to date. It would be useful to examine subtle sublethal-effects in some of the less-well represented and, perhaps, sensitive species. Also to include sediment bioassays to look at growth and survival of juvenile bivalves. Compare responses in Poole biota with those elsewhere to look for signs of adaptation.

Multivariate Statistical Analysis of benthic communities and environmental variables in order to examine spatial and temporal trends in communities in relation to contaminants (Warwick *et al*., 1998).

It is stressed that the above procedures have been selected primarily with regard to their ease of use, low cost and relevance to known environmental problems. Ideally, all components to the scheme need to be synchronised and run in tandem to achieve best value and to provide the most useful information on causal links and mechanisms. The results will assist environmental managers in identifying those consents and activities which most require attention and hopefully may help to decide on the best options for action.

Annex 7. Estimated[1] quantities of contaminants in some major discharges to Holes Bay, Poole Harbour

(from EA website).

Contaminant	Source	Quantity kg y^{-1}	Actual or Less than	Year
Arsenic	MERCK LTD	6	<	1998
Arsenic	Wessex Water	120		2000
Arsenic	Wessex Water	15		2001
Cadmium	MERCK LTD	1	<	1998
Cadmium	Wessex Water	2.5		2000
Cadmium	Wessex Water	9.6		2001
Chromium	MERCK LTD	20	<	1998
Chromium	Wessex Water	150		2000
Chromium	Wessex Water	120		2001
Copper	MERCK LTD	20	<	1998
Copper	Wessex Water	240		2000
Copper	Wessex Water	190		2001
Lead	MERCK LTD	20	<	1998
Lead	Wessex Water	130		2000
Lead	Wessex Water	60		2001
Mercury	MERCK LTD	0.13	<	1998
Mercury	Wessex Water	0.42		2001
Nickel	MERCK LTD	20	<	1998
Nickel	Wessex Water	690		2000
Nickel	Wessex Water	610		2001
Zinc	MERCK LTD	20	<	1998
Zinc	Wessex Water	2000		2000
Zinc	Wessex Water	1300		2001
1,1,1-Trichloroethane	Wessex Water	0.2	<	2001
Aldrin	Wessex Water	0.04		2000
Chloroform (Trichloromethane)	Wessex Water	19		2000
Chloroform (Trichloromethane)	Wessex Water	14		2001
DDT - All Isomers	Wessex Water	0.035		2000
Dieldrin	Wessex Water	0.061		2000
Dieldrin	Wessex Water	0.07		2001
Endosulfan	Wessex Water	0.078		2000
Endrin	Wessex Water	0.045		2000
Hexachlorobenzene	Wessex Water	0.041		2000
Hexachlorobutadiene (Hexachlorobuta-1,3-Diene)	Wessex Water	0.11		2000
Hexachlorocyclohexane (All Isomers)	Wessex Water	0.13		2000
Isodrin	Wessex Water	0.039		2000
Mecoprop	Wessex Water	1	<	2001
Pentachlorophenol And Its Compounds	Wessex Water	1.2		2001
Permethrin	Wessex Water	0.072		2001
Polychlorinated Biphenyls - Total	Wessex Water	0.002	<	2001
Tetrachloroethene(Tetrachlorethylene, Perchloroet)	Wessex Water	9	<	2000
Tetrachloroethene(Tetrachlorethylene, Perchloroet)	Wessex Water	9	<	2001
Trichlorobenzene - All Isomers	Wessex Water	0.41		2000
Trichloroethylene	Wessex Water	11	<	2000
Trichloroethylene	Wessex Water	11	<	2001
Trifluralin	Wessex Water	0.45		2000

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[1] These figures may be a substantial overestimate and the true figures may be much lower because of the method of calculation.

Annex 8. A summary of water company improvements in Poole SPA
(Information source - EA)

Asset Management Plans (AMPs) timescale

	Timescale
AMP1	1990 – 1995
AMP2	1995 – 2000
AMP3	2000 - 2005

Poole Harbour

Poole, Wareham and Lytchett Minster Sewage Treatment Works will be uprated by 2003, with the addition of UV disinfection to the secondary treatment under the Bathing Water Directive and the Shellfish Waters Directive in AMP3. The storm tanks at these works are also due for improvement.

A storm overflow in Upton is due for improvement by April 2003 under Bathing Water Directive and UWWT Directive.

Between 7 and 9 storm overflows to Poole Harbour are being improved under the Shellfish Waters and Urban Waste Water Directives during AMP3, the majority of which were completed between 2001/2.

Titles in the current series of Site Characterisations

Characterisation of the South West European Marine Sites: **The Fal and Helford cSAC**. Marine Biological Association of the United Kingdom occasional publication No. 8. pp 160. (2003)

Characterisation of the South West European Marine Sites: **Plymouth Sound and Estuaries cSAC, SPA**. Marine Biological Association of the United Kingdom occasional publication No. 9. pp 202. (2003)

Characterisation of the South West European Marine Sites: **The Exe Estuary SPA** Marine Biological Association of the United Kingdom occasional publication No. 10. pp 151. (2003)

Characterisation of the South West European Marine Sites: **Chesil and the Fleet cSAC, SPA**. Marine Biological Association of the United Kingdom occasional publication No. 11. pp 154. (2003)

Characterisation of the South West European Marine Sites: **Poole Harbour SPA**. Marine Biological Association of the United Kingdom occasional publication No. 12. pp 164 (2003)

Characterisation of the South West European Marine Sites: **The Severn Estuary pSAC, SPA.** Marine Biological Association of the United Kingdom occasional publication No.13. pp 206. (2003)

Characterisation of the South West European Marine Sites: **Summary Report**. Marine Biological Association of the United Kingdom occasional publication No.14. pp 112 (2003)

Aerial view Poole Harbour entrance
Photograph; EA (www.oursouthwest.com)

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