# Characterisation of European Marine Sites
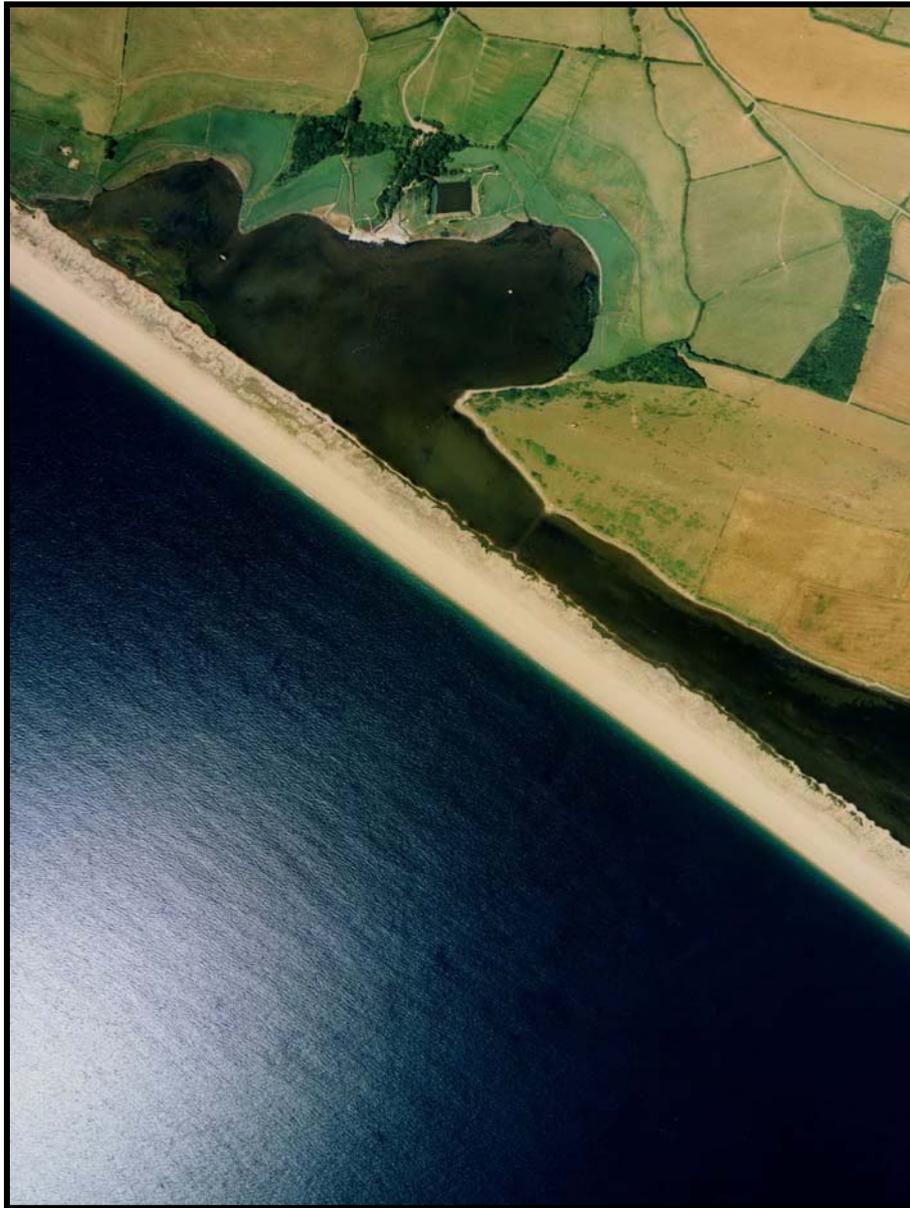
Chesil and the Fleet

(candidate) Special Area of Conservation
Special Protection Area

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Site Characterisation of the South West European Marine Sites

Chesil and The Fleet cSAC, SPA

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by the Plymouth Marine Science Partnership

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ACKNOWLEDGEMENTS

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

1: Agricultural practice

2: Agricultural land bordering the northeast banks of the Fleet Lagoon

3: (above) Fuel bunker terminal on the banks on the shores of Portland Harbour
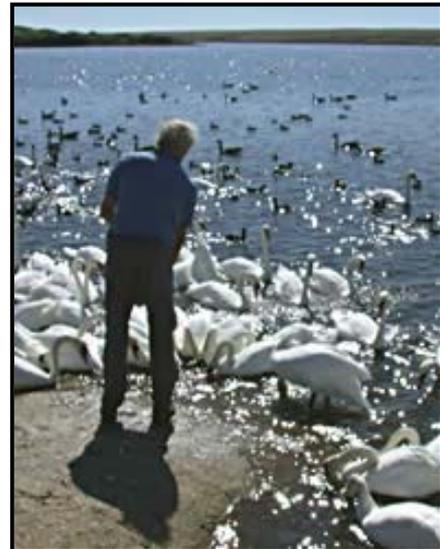
4 & 5: (right) Feeding the swans at Abbotsbury: up to 850 swans are fed daily

Photographs:
1 & 3: Ian Britton Freefoto.com
2: A.F. Lack South West Coast Path Association (http://www.swcp.org.uk)
4& 5: John Houston, Abbotsbury Tourism

Plate 2: Some of the Interest Features and habitats of Chesil and the Fleet cSAC, SPA

1: (left) Seagrass and green algal meadow within the central lagoonal basin (includes eel grasses *Zostera marina* and *Zostera noltii*, and the tasselweeds *Ruppia cirrhosa* and *Ruppia maritima*).

2: (below) Foxtail stonewort *Lamprothamnium papulosum*

3: Specialised flowering plant: the yellow-horned poppy *Glaucium flavum*

4: Little Tern **Sterna albifrons**

5: Widgeon *Anas penelope*

Photographs:
1: Peter Dyrynda   **2:** S Scott   **3**: Stan Beesley   **4** : Martin Cade   **5**: Stephane Moniotte

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 Chesil and The Fleet candidate Special Area of Conservation (cSAC), Special Protection Area (SPA).

This project, undertaken by the Plymouth Marine Science Partnership (PMSP), has two main objectives. Firstly, to characterise the site in terms of water and sediment 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. This 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 follow:

In common with other saline lagoons, **biodiversity** is somewhat limited in the Fleet compared with other marine habitats. Species diversity and abundance both peak in the Langton Hive area where the salinity is generally 20–35‰, and the subtidal lagoon bed is dominated by mud and mixed stands of seagrass and tassleweed. The fauna of the Fleet is characterised by brackish water species and there are a number of specialist lagoonal species including the starlet sea anemone *Nematostella vectensis*, lagoon sandworm *Armandia cirrhosa*, and lagoon sand shrimp *Gammarus insensibilis*. The specialised conditions in the link channel have led to an exceptional abundance of some of the more common marine species, notably snakelocks anemone *Anemonia viridis* and cushion star *Asterina gibbosa*. Vegetation in the lagoon includes the rare charophyte *Lamprothamnium papulosum* (foxtail stonewort) together with beds of the aforementioned eelgrasses (*Zostera spp*) and tasselweeds (*Ruppia spp*). There have been changes in the distribution of *L. papulosum* which appear to be due to elevated phosphate levels. The lagoon provides important habitat for waterfowl, marshland- and sea-birds. A herd of mute swans is managed at Abbotsbury and has been a feature of the lagoon since the 1300's. The birds nest within the Abbotsbury embayment in spring and early summer, and forage along the length of the lagoon. These birds and their feed constitute an important source of nutrients, locally.

It is evident that much of the western lagoon, notably the Abbotsbury embayment is subject to eutrophication. Although the majority of **nutrient** inputs in the system may be due to diffuse sources such as agricultural run-off, localised enrichment from STWs and the swannery may also be significant. The poor flushing characteristics of the lagoon, coupled with its shallow depths, extremes of temperature, and high pH, compound the enrichment problems of the affected areas. Nutrient-associated water quality problems include macroalgal, and to a lesser extent, microalgal blooms and periodic oxygen sags. The problem of hypernutrification has lead to the recent designation of the Fleet Lagoon as a Polluted Water (Eutrophic) (under the Nitrates Directive 91/676/EC), and the catchment area for the Fleet as a Nitrate Vulnerable Zone (NVZ), which, it is hoped, will herald significant reductions in nutrient loadings. For the effectiveness of any remediation measures to be accurately monitored, regular nutrient analysis in the lagoon has been reinstated.

At present there is little unequivocal evidence, from chemical data, that modifications to biota of the European marine site have occurred, or would be expected to occur, as a result of **toxic contaminants**. However, the available information, both in the literature and in unpublished data-sets, is very scarce. For example, there have been no measurements of **TBT** in the Fleet, making it impossible to provide an up-to-date and accurate assessment of the cSAC/SPA itself. At nearby sites in Portland Harbour, TBT levels in water are sometimes above the EQS and it is possible they may impinge to some extent on the east Fleet. At Portland Bill the dogwhelk population, affected by imposex in the early 1990's, has been eliminated, whilst shellfish from Portland Harbour contain TBT residues above the OSPAR guideline. Clearly, a more detailed survey of TBT is needed to establish current status and trends for TBT in the Fleet. For **metals**, monitoring of tidal waters of the Fleet lagoon is limited to a single site (the Narrows) where there have been no cases of EQS exceedences in recent years. Metal concentrations in sediments of the Fleet are considered typical for the region and according to guideline criteria, no harm to biota would be predicted (all below probable effects level), though at some sites chronic effects cannot be excluded (concentrations exceed the ISQG for Pb, Zn, Cu, As and Hg). Bioindicator studies with infaunal species such as worms and clams are recommended, as there is little information on metal bioaccumulation other than occasional compliance monitoring of commercial shellfish (Fleet oysters may contain elevated levels of Cu and Zn). Similarly, most **hydrocarbon/PAH** data are for sediments, and, in the Lyme Bay area, highest concentrations are associated with muds of Portland Harbour and the Fleet. The total hydrocarbon composition at Portland is consistent with contamination dispersing from shipping or from runoff from Weymouth. Sediment PAHs are dominated by pyrogenic components indicative of fallout from incomplete combustion of fossil fuels (perhaps concentrated in urban runoff). PAH concentrations in Fleet sediments are above the ISQG but below the PEL. Although the perceived threat to the Fleet is probably low this should be verified by further sampling of Fleet sediments. Since there are few body burden data for PAHs, measurement of PAHs in infaunal bioindicator species should also be considered. Most **synthetic organic compounds** in tidal waters of the region are below EA detection limits and appear to comply with EQS standards. Effluent from Weymouth long-sea outfall increases the bioaccumulation of some synthetic organics (PCBs, OP' DDE) near to the discharge, but risks to biota within the European Marine Site, from aqueous sources, are likely to be low. There are no records of synthetic organic contamination of Fleet sediments. Concentrations of γ-HCH in nearby Portland Harbour sediments were above PEL whilst the majority of other compounds were below detection limits. Bioaccumulation data for organic contaminants are restricted to commercial shellfish in the Fleet (Abbotsbury oysters) and Portland Harbour (mussels). The predominance of '<' values in these samples (with the exception of γ-HCH and PCBs) supports the notion that most organics pose little threat in the region, though more extensive, targeted sampling in the Fleet is needed for confirmation. The possibility of endocrine disruption from synthetic organics should also be addressed.

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 that the site can be maintained in favourable condition.

CONTENTS

2. INTRODUCTION

This review considers the characteristics of the marine areas of CHESIL AND THE FLEET cSAC, SPA and how the status of the site is influenced by existing permissions and activities, either alone or in combination. Also considered are possible impacts 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 cSAC, 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 European marine site, given under Regulation 33(2) of the Conservation Regulations 1994 (English Nature, 1999). 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 Chesil and The Fleet cSAC, SPA. Toxic and non-toxic contamination are the principal threats considered in the current project. (Table adapted from English Nature, 1999)[1]

Standard list of operations which may cause deterioration or disturbance	INTEREST FEATURES				
	Lagoon[2]	Annual vegetation of drift lines	Mediterranean and thermo-Atlantic halophilous scrub	Internat. important Annex 1 birds	Internat. important migratory species
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)	✔	✔	✔	✔	✔
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)	✔			✔	✔

[1] Note: more recently, additional interest features have been submitted to the EU as reasons for recommendation as a cSAC. These include Perennial Vegetation of Stony Banks and Atlantic Salt Meadows. See Annex 1 for more detailed descriptions.

[2] The Lagoon feature is a Priority Habitat under the Habitats Directive: Priority habitats are those considered in danger of disappearance and for the conservation of which the European Community has particular responsibility. Sites submitted for priority habitats to the European Commission are automatically classed as Sites of Community Importance (one step on from being identified as a candidate SAC in the SAC selection and designation process).

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, Fleet Study Group, 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 undertaken a number of generic studies on modelling of nutrients, bioavailability of metals, TBT impacts and the ecology of benthic organisms. Comparative data for other UK estuaries, including south-west European marine sites (e.g. Exe, Severn, Poole, Fal, Plymouth Sound and Estuaries) have been used to draw comparisons with the Chesil and Fleet cSAC, SPA, wherever possible.

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. 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 5.**

Section 6 discusses published information on toxic contamination (metals, TBT, petrochemicals, pesticides, PCBs, volatile organics) and non-toxic contamination (nutrients, turbidity, dissolved oxygen). Section 6 also 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).

A synthesis of available information on sediment quality is given in **Section 7.**

A brief description of modelling exercises of direct relevance to the environmental quality status of the site is provided in **section 8**.

Concluding remarks (**section 9**) include a summary of evidence for impact in the Chesil and Fleet European marine site, together with recommendations for future monitoring and research requirements.

4. THE SITE: FEATURES AND THREATS

The Chesil and the Fleet European marine site has been recommended as a candidate Special Area of Conservation (cSAC) because it contains habitat types and/or species which are rare or threatened within a European context. In addition to its cSAC status, the area is a Special Protection Areas (SPA) - designated under the European Commission Directive on the Conservation of Wild Birds (79/409/EEC), and an SSSI. The SSSI citation describes the special interests for which the site was notified in the British context (although this does not include the sublittoral). The site is also a Ramsar Site (wetland of international importance designated under the Ramsar convention), a nature reserve (encompassing the Chesil Bank, the Fleet lagoon and Portland Harbour) and an Area of Outstanding Natural Beauty (AONB).

The site is largely natural and undeveloped, principally because the lagoon bed and the majority of the shore has been owned and managed privately by the Ilchester Estate for over 400 years. The boundaries of the cSAC, which covers an area of 1631.63ha, are shown in figure 1. Detailed maps of biological communities and features within the site can be found elsewhere (English Nature, 1999).
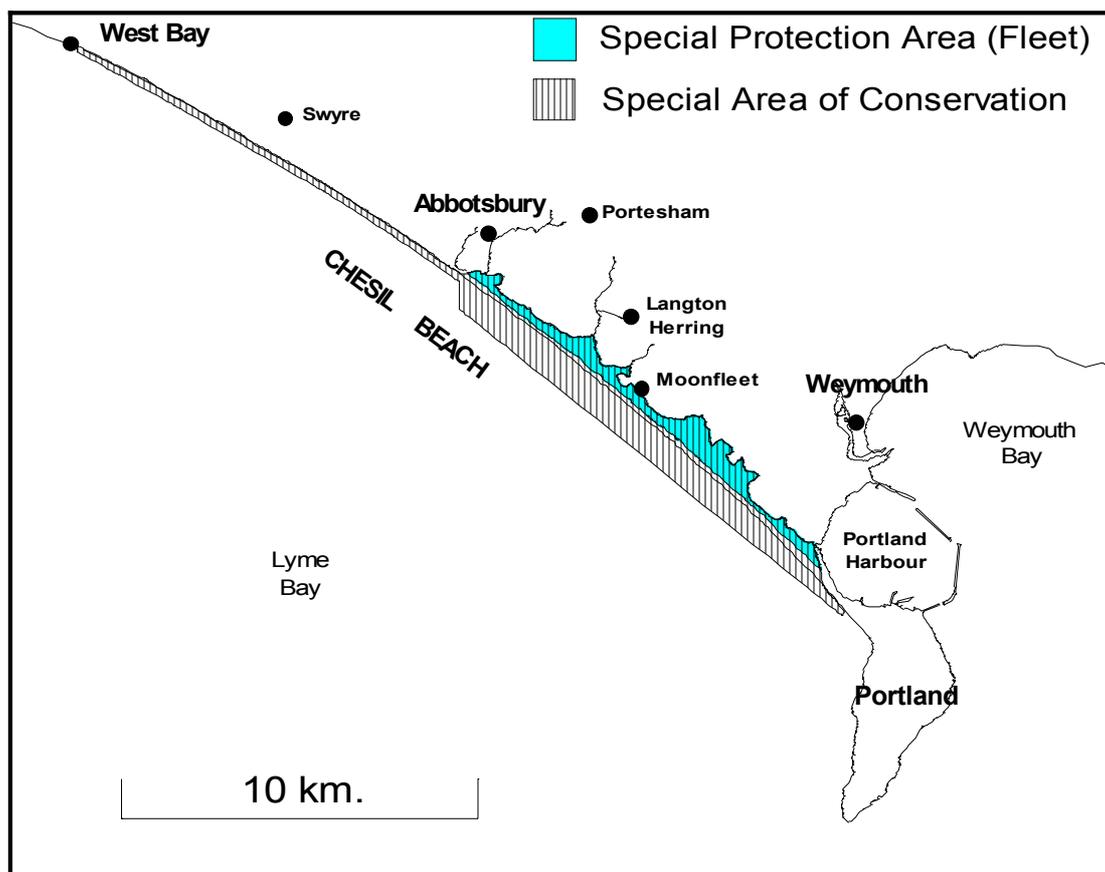

Figure 1: Chesil and the Fleet European marine site showing boundaries of the cSAC and SPA

The Chesil Bank is situated on the West Dorset coast, stretching 29km from West Bay to Portland, and is one of the five largest shingle beaches in Britain. It faces the storm waves, driven by the prevailing south-westerly winds up the English Channel, from the Atlantic Ocean. The bank varies in height, generally increasing from the west, but is at its maximum (~14m) at Ferrybridge (Moxom, 1993). On the seaward side, the shingle extends to a depth of 11m below low-water mark (some 270m offshore) at West Bexington and also at Abbotsbury, and to about 18 m at Wyke Regis (also to ~270 m offshore) and to 15 m at Portland. On the landward side, however, the shingle rests on a bed of clay 1 to 1.2m below low water mark.

The overall grading of the beach is from pea-gravels at West Bay to cobbles at Chiswell, Portland (although there are local variations that may be significant). The pebbles are predominantly locally derived flint and chert with some exotic material, notably quartzite pebbles from Budleigh Salterton in Devon. In geomorphologic terms Chesil Bank is known as a tombolo - a coastal feature above sea-level for most of the time, formed when a belt of sand and/or gravel is deposited between an island and the mainland. There is no full agreement on the exactly how the beach was formed. The traditional view is that Chesil Beach was driven onshore with rising sea levels at the end of the last Ice Age. As the sea advanced, material from what is now Lyme Bay was swept up to form the beach, trapping the Fleet lagoon in the process. More recently it has been proposed that a large volume of the beach was derived from the landslides of East Devon and West Dorset (Bray, 1997). A succession of weakly consolidated strata underlie the mainland shores of the Fleet, the erosion of which has released shingle, gravels and sands on to much of the lagoons shores (Bird, 1972).

Sandwiched between Chesil Bank and the impounded mainland coast, the Fleet Lagoon is the largest regular tidal lagoon in Britain covering 500ha (1200 acres) at high tide. The lagoon is very shallow, with a nominal depth of one third of a metre at the western extreme, and around 5 metres under Ferrybridge to the east. At its widest point (Littlesea) the lagoon is 900m wide; this varies to 65m at the narrows. It stretches 13km from Abbotsbury in the west to a point where tidal waters enter the lagoon through a small opening at its south-eastern end (Small Mouth). There is also some small-scale saline intrusion via intertidal saline springs and seepage through the shingle of the bank (Whittaker, 1978).

The Portland sea area is microtidal with a range of only 1.5m at spring tides. This is combined with an unusual double low cycle causing prolonged low water stands of up to 4 hours. The effect within the Fleet lagoon is an attenuated and phase-lagged tidal cycle, which is increasingly eccentric toward the western reaches, where tidal influence is weakest (Whittaker, 1978). There is very little wave exposure within the lagoon; any wave action that occurs is generated by north-westerly, or south-easterly winds along the length of the estuary (Bird, 1972). Freshwater inputs to the Fleet from inland streams and land run-off result in brackish water. The lagoon generally shows salinity gradient from Small Mouth to Abbotsbury although this can be variable according to rainfall[1]. Values in excess of 30ppt are typical of East Fleet and can occur in the West Fleet, but are usually lower. Salinities as low as 10ppt have been recorded at Abbotsbury during high rainfall periods (Dyrynda and Cleator, 1995). The shallow nature, and generally poor flushing characteristics of the Fleet

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[1] For more detail see section 8 (Models)

result in exaggerated maxima and minima water temperatures in comparison with other south coast inlets. Extensive icing can occur in cold weather, whereas during hot spells, temperatures can exceed 20°C throughout the lagoon, and up to 30°C has been recorded in West Fleet (Seaward, 1994).

The lagoon is divisible into two ecologically distinct zones determined by its shape and tidal regime: the 'lagoonal basin', and the 'inlet channel'. Both are highly sheltered against wave action, but the two differ in terms of tidal energy. The 'lagoonal basin' experiences a minimal tidal range, weak currents and very poor flushing. Salinity is low and variable. The substratum of the basin is mainly soft organic muds supporting extensive meadows of sea grasses and algae during summer months. Abbotsbury Embayment, at the blind end of the lagoon, is home for a highly unusual, ancient colony of mute swans, historically (since at least the 14th century) farmed for the table but now the focus for a wildlife reserve, the Abbotsbury Swannery. A wide variety of other wildfowl also inhabit the lagoon.

The 'inlet channel' occupies the south-easterly quarter of the lagoon from Small Mouth to the Narrows (a 1 km rapids system) and extends through a tidal basin to the man-made lagoon entrance at Small Mouth. This section experiences a pronounced tidal rise and fall, strong currents and better flushing than the western reaches. The currents and scour have resulted in significant erosion and recession of the mainland shore in the Narrows region, and exposed areas of oolithic corallian limestone in the subtidal, where coarse sediments, bedrock, hard clay, shingle or sand dominate (Dyrynda and Farnham, 1985). The channel bed supports an unusual and diverse assemblage of algae and sedentary invertebrates, particularly within the Narrows. Rare, or scarce species include the red alga *Gracilaria bursa-pastoris,* the sponge *Suberites massa*) and the southern black-faced blenny *Tripterygion delaisi*. A specialist lagoonal polychaete worm, *Armandia cirrhosa,* is found on subtidal sediment flats (Dyrynda and Cleator, 1995).

Extensive meadows of eel-grass *Zostera spp,* and wigeongrass *Ruppia spp*. carpet much of the subtidal. These support a rich and unusual invertebrate fauna and provide food for the resident mute swan population and a variety of waders and wildfowl (Dyrynda, 2002; Fleet Study Group [FSG] website).

Saltmarsh areas, best developed around inlets and bays at either end of the Fleet, are distributed around the shores and freshwater marsh is found adjacent to some of the saltmarsh areas e.g the Abbotsbury embayment.

As a whole, the site is important to a large number of bird species. Brent and Canada Geese overwinter in the Fleet and many rare species such as Little Auk, Fosters Tern and Great Northern Diver are blown into shore or pass through. The area is also important to a lesser extent for breeding species. The Chesil is one of the few sites in the southwest with populations of Little Tern, Common Tern, and Ringed Plover.

The catchment area for the Fleet is predominantly rural and relatively small (28km^2) and drained by several freshwater flows. Table 2 shows freshwater source mean flows calculated by Murdoch (1999) for use in modelling studies and serves to illustrate the relative importance of each source. None of the streams are continuously gauged and so the flow profiles have been estimated using Micro Low

Flows (a piece of software developed by the Institute of Hydrology, particularly for catchments subject to artificial influences such as impoundment, discharges and abstraction, to allow the estimation of natural low flows at ungauged sites).

Table 2. Freshwater source mean flows

Stream	Micro Low Flow[1]
Coward's Lake	0.03
Mill Stream	0.086
Horsepool	0.026 (0.035)
Rodden Stream	0.053
Herbury	0.023
West Fleet	0.018
East Fleet	0.027
Diffuse[2]	0.03

NB. The figure in parenthesis was used in modelling (Murdoch, 1999)

Average annual rainfall for the coastal area surrounding the Fleet is 701-750mm, rising to 800mm in the higher ground from which the tributaries that feed the Fleet at Abbotsbury and Roddon Hive originate (EA, 1997).

Threats

There are a few moorings for small boats within the Fleet (e.g. Langton Hive Point). A warden (for the Chesil Bank and Fleet Nature Reserve) is present on site and is largely responsible for controlling human disturbance (JNCC website, 2000). Two military training camps (Wyke Regis and Chickerell) use areas of Chesil bank for training and there is a small arms range, with a safety area which extends across the Beach and out to sea.

The original natural entrance to the lagoon at Small Mouth was considerably widened in 1824 by a great storm which caused a tidal surge along the Fleet and resulted in the destruction of small villages and loss of life.   Extensive scouring out of organic silts is thought to have occurred over subsequent years, before material was deposited along the causeway to artificially restore Small Mouth Passage to its former dimensions and the mud-flats began to re-shape (Moxom, 1993).

The breakwaters of Portland Harbour, built in the late 1800s, subsequently altered the force and direction of tides in the harbour area and afforded some protection to the Small Mouth entrance (Green, 1981; Moxom, 1993). Until recently the Harbour was a major naval base.

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[1] These are relative ratios scaled using micro-low flow means. The method is to use standard mean flow statistics from Micro Low Flows to provide the ratios of stream daily flows to the daily flows at the nearby Gauging Station at Broadwey. A time series of stream flows is then formed by multiplying the gauged record by the flow ratios. (Murdoch, 1999).

[2] The diffuse flows were assumed uniformly distributed between Abbotsbury and East Fleet and amount to approximately 13% of the total flow. (The contribution beyond East Fleet was ignored, because whilst MLF apportions it to the Fleet, it may flow elsewhere. Additionally the main concern is nutrient levels in the upper part of the Lagoon.)

There was concern for the ecology of the lagoon in 1986, when the Small Mouth entrance was relocated to an adjacent site due to the erection of a new road bridge over the entrance channel. However, the impact appears to have been minimal (Dyrynda and Cleator, 1995) (see section 5). It is proposed to build a new jetty to the east of the oyster farm, although minimal impact is again predicted, as the adjacent seabed has no features of biological interest.

The most recent survey (Bunker *et al.,* 2002) reports some 'disturbed' areas of cobble seabed in the vicinity of the narrows which gave the appearance of being dredged and were comparatively bare of epibiota (in comparison to other adjacent cobble areas). The cause of this disturbance is not known and could be the result of a number of different factors e.g. boat anchoring.

Non-native algae *Sargassum muticum* occurs in the Fleet, and is monitored, cut and removed annually by the army, although re-growth is rapid. There appear to be no deleterious effects to other biota directly attributable to *Sargassum* cutting.

Fisheries

The Fleet is a MAFF designated bass nursery area and fishing from boats is prohibited at all times. This prohibition does not apply to the shore however and small-scale netting for bass has been reported in this area. There is also a prawn fishery off the old Ferrybridge and a private fish farm within the Fleet, the latter is not governed by the Sea Fisheries Committee restrictions. The EA issues eight licences (for 10 fyke nets each) annually for eel fishing on the Fleet. Reported annual catch varies from a minimum of <500 kg to over 2500 kg (EA, 1997). Other fish in the Fleet include large shoals of small grey mullet, sandsmelt, sandeels, flounder, rockling, blenny, wrasse, pouting, and stickleback.

Shellfish farming is the largest single commercial fishery on the Fleet. Natural populations of oysters *Ostrea edulis* have been harvested since the 15th century and there were commercial operations in the 18th and 19th century. Currently the pacific oyster *Crassostrea gigas* is farmed on a relatively large scale upstream of the narrows, where trestles are situated along the margins of the channel. Annual production of up to 100 tonnes has been reported (Copperthwaite, 1993). Under the Shellfish Harvesting classifications for 2002[1], Fleet beds were designated as class B bivalve production areas for *C. gigas* - a deterioration from an A classification in 1995. Potential impacts associated with this culture include increased sedimentation beneath the trestles and related changes in the composition of the benthos (see section 5).

The seaward shore of Chesil bank is popular with anglers for the variety of fish usually caught from boats - bass, cod, spurdog, skate, smoothhound, turbot, brill, dogfish, sole conger and more conventional beach-caught fish like pollack, pout and whiting. Small-scale commercial fishing from the bank has taken place for hundreds of years and is mentioned in the Domesday Book. Seine netting using lerrets, small traditional double bowed-boats, for mackerel, whiting, herring and sprat still takes place, and there is a small local fishing industry.

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[1] The EC shellfish hygiene Directive (91/492/EEC) defines standards for shellfish quality required in the end product, and classifies shellfish harvesting areas into 4 categories according to the concentration of bacteria found in the shellfish flesh

A limited amount of bait digging takes place on the intertidal mudflats, this is on a small scale and not considered a problem although there is little quantitative information available.

Bathing waters

There are no designated bathing waters within the Fleet Lagoon, however the nearby Portland Harbour, Weymouth, and Chesil designated bathing waters passed guideline standards in 2001 indicating that seawater quality in the area is generally good.

Nutrients

There have been concerns regarding the nutrient status of the lagoon for some years: Algal blooms were identified in The Fleet during 1994 which may have been a result of diffuse agricultural inputs (JNCC website, 2002). The Fleet Lagoon has recently been designated as Polluted Waters (eutrophic) and the catchment as an NVZ.

Diffuse sources; Diffuse pollution, particularly from agricultural land runoff, is seen as an important issue in the South West Region. Intensive agricultural practices are susceptible to soil erosion and leaching[1]. 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 also reduce infiltration to ground, compounding low flow problems. Farm animal waste can be a significant potential source of pollution to rivers feeding the Fleet, and the area is targeted under the Countryside Stewardship Scheme to implement buffer zones to reduce runoff into the reserve.

Point sources; There is little industrial activity in the area and few consented discharges to the lagoon or its tributaries. Siting of some of the more important discharge consents to the site are shown in figure 2.

Two Wessex Water Services consented STW discharges exist to tributaries of the Fleet, at Abbotsbury[2] and at Langton Herring - no consented DWF, max. daily flow only). A further four private sewage treatment works with consented maximum discharge volume of >5 m^3day^{-1} discharge to the Fleet at Abbotsbury (<10 m^3day^{-1}), Langton Hive (26-50 m^3day^{-1}), Chickerell (>50 m^3day^{-1}) and the west end of the Narrows (<10 m^3day^{-1}) (EA, 1997).

New treatment facilities (a large sewage farm) serving Weymouth and Portland have been sited close to the headworks at Martleaves, Wyke Regis, and discharge into Lyme Bay, via a long sea outfall pipe.

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[1] Soil erosion may contribute to phosphorus loading to lagoon but the dominant source of nitrogen is thought to be leaching of nitrate from soil.
[2] There is a consented DWF of 140 m^3 day^{-1} at Abbotsbury STW although this likely to have been exceeded for many years. Wessex Water has calculated a theoretical DWF of 243 m^3 day^{-1} based on the connected population, and this is predicted to increase to 290 m^3 day^{-1} by 2020 (EA pers comm.).

1 :- SEAWAY LANE AND SMUGGLERS COTTAGE
2 :- SWANNERY RESTAURANT STW FE
3 :- ABBOTSBURY PUMPING STATION
4 :- ABBOTSBURY STW FINAL EFFLUENT
5 :- LAND AT REAR OF KINGS ARMS
6 :- MOONFLEET MANOR HOTEL
7 :- STABLE/GROOMS QUARTERS
8 :- HIGHFIELD
9 :- 1 & 2 BUTTER STREET
10 :- R E T C CHICKERELL
11 :- RETC BRIDGING HARD
12 :- FURZEDOWN FARM
13 :- FERRYBRIDGE PS OVERFLOW
14 :- FERRYBRIDGE FISHERMANS STORES
15 :- NO 1 FISHERMANS QUAY
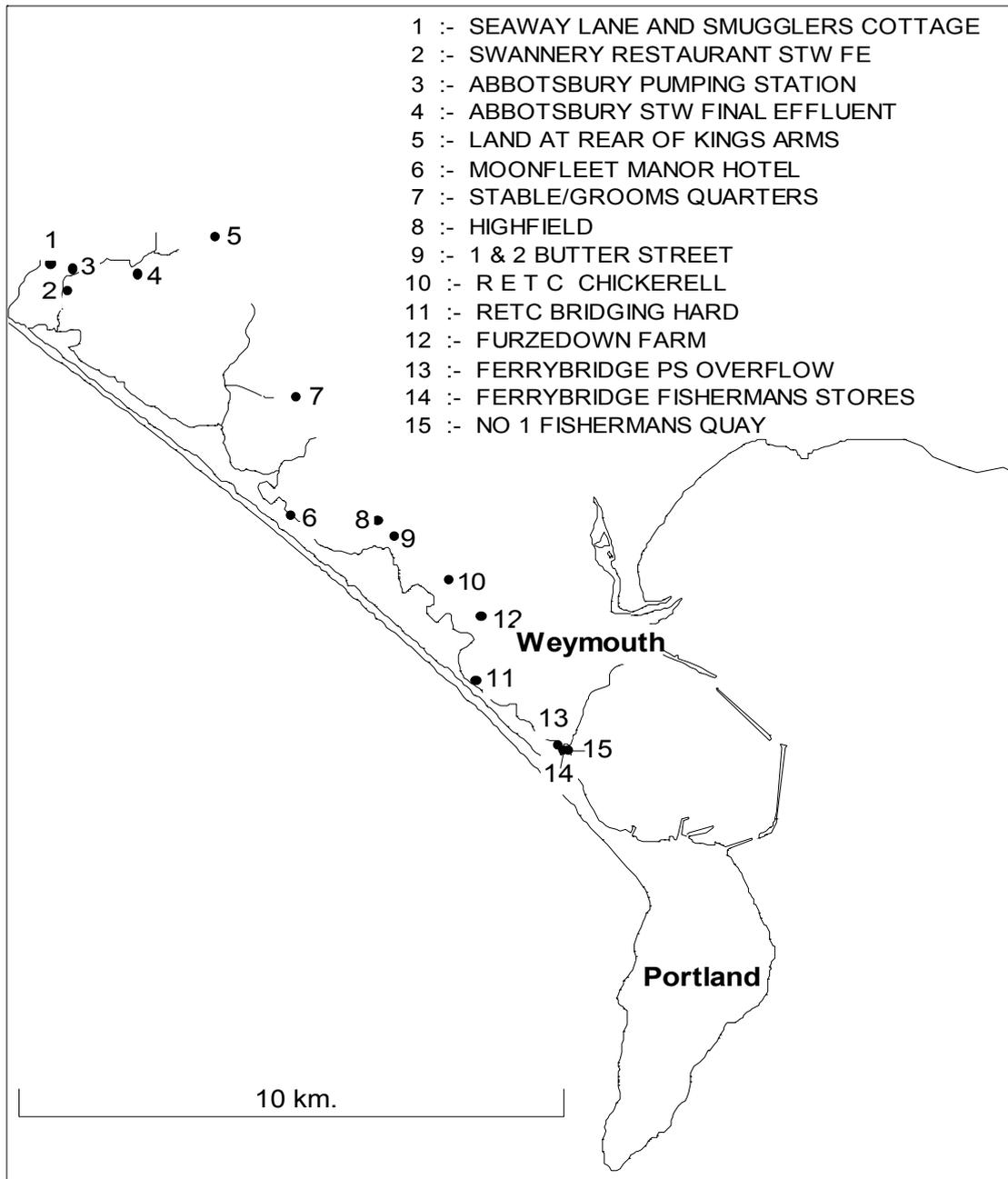
Figure 2. Locations of discharge consents to the Fleet Catchment. (From data supplied by the Environment Agency, South West Region).
NB No distinction has been made between continuous and intermittent discharges and details of specific discharges should be clarified with the Environment Agency

The point source discharges from long sea outfalls at Weymouth and West Bay are shown in figure 6. Although these are unlikely to impact on the Fleet their potential consequences for the Chesil Beach component of the cSAC are considered in section 6.

There is one groundwater abstraction licence for public water supply within the Fleet catchment. This allows removal of between 0.5 and 5.0 Ml d^{-1} from tributaries at Portesham near Abbotsbury.

5. STUDIES ON BIOLOGICAL COMMUNITIES

The ecological divisions of the Fleet Lagoon (see section 4) largely determine the distribution of the aquatic fauna and flora. Vegetation is strongly seasonal, with seagrasses growing on the lagoon bed from late spring to autumn, accompanied by dense swards of green algae until mid-summer. The flora dies back in late autumn, and plant debris litters the exposed mud on the lagoon bed. Dyrynda (1997) identified a gradient of decreasing vegetation and invertebrate numbers, from mid lagoon to shore, in all areas, with permanently submerged central areas supporting the highest densities of vegetation and invertebrates.

Whittaker (1978) studied the lagoon and its vegetation, and constructed a list of 150 algal species found in the Fleet between 1975 and 1977. More recently, Dyrynda and Cleator (1995) provided a characterisation of the habitats and benthic communities in the Fleet Lagoon together with comprehensive lists of the aquatic fauna and flora present. The work also reviewed previous studies and information, encompassing physical and ecological parameters.

The Fleet Study Group (FSG), established in 1975 for management and protection of the Fleet lagoon and Chesil Bank, holds an accessible archive of work[1] carried out by members of the Fleet Study Group and others, concerning, primarily, the biological and historical interest of the Fleet.

Saltmarsh

Saltmarsh areas around the Fleet are not extensive in comparison with some other southwest saltmarsh sites (e.g Poole Harbour) but are distributed all around the shores. The coverage varies considerably according to tidal range and shoreline shape, but best developed areas occur around the inlets and bays at either end of the lagoon. Around promontories such as Herbury, the saltmarsh comprises only a narrow fringe, and on the beach itself, salt marsh areas are patchy. At Ferrybridge saltmarsh borders the tidal creek and ponds which were created when shingle was extracted to make concrete in the early 20th century.

Shrubby sea-blite, *Suaeda fruticosa*, and Sea Purslane, *Halimione portulacoides*, are the two most obvious saltmarsh species, and stiff saltmarsh-grass *Puccinellia rupestris* and Borrer's saltmarsh-grass *P. fasciculata* are amongst more uncommon plants. FitzPatrick (1981) recorded 34 saltmarsh species on the shores of the Fleet, including *Lathyrus japonica, Glaucium flavum, Crambe maritima* and *Trifolium scabrum*. Freshwater marsh species (*Phragmites sp., Scirpus maritimus, Iris pseudacorus* and *Aster tripolium*) grow alongside those of the saltmarsh. The Fleet is the only site in Dorset where the marshmallow *Althea officinalis* occurs.

Saltmarsh is important inasmuch as it provides cover for much of Chesil's mammal population and its extensive and tight rooting system allows other plant species, such as the greater sea-spurrey *Spergularia media*, and little robin *Geranium purpureum* to

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[1] held at Weymouth College Library, Dorset

take hold. It also minimises the shingle surge towards the Fleet, thus increasing the beach's effectiveness as a sea defence (FitzPatrick, 1981).

Macroalgae

Distribution of macroalgae in the Fleet can be related to the two major regions: East Fleet, from Butterstreet Cove to Ferrybridge and Small Mouth (an area strongly affected by tides, featuring large expanses of exposed sand and mud which are crossed by channels at low tide), and West Fleet, west of Butterstreet Cove to Abbotsbury. This stretch is less affected by tides and generally has reduced salinity (Burrows, 1981).

Whittaker (1978) described algal communities and reported that green algae are particularly abundant, notably in the lower salinity areas of West Fleet, where *Ulva lactuca*, *Cladophora*, *Chaetomorpha* and *Enteromorpha* spp. form dense mats on *Zostera* beds and cover large tracts of the waters surface and the mudflats in spring. The majority of *Ulva* was said to die back after releasing its spores around May, whilst the other species were reported to persist until late autumn. Brown seaweeds were principally represented by *Fucus* species, *F. serratus and F. vesiculosus,* although *F. spiralis* was occasionally found. The distribution of the fucoids was also determined by salinity: *Ascophyllum nodosum* occurred near the tidal inlet at Small Mouth, and *F.vesiculosus* was confined to the east of Butterstreet Cove on the landward shore. *Laminaria saccharina* was the only representative of the kelps to be found in the Fleet, its distribution appeared to be depth-controlled and restricted to the deepwater channels of the Narrows and Littlesea. Red algae were found in the deeper and more marine areas of the Fleet, east of Chickerell Hive, although the filamentous *Ceramium* occurred in large floating masses in spring and was driven by wind into West Fleet. Overall, Whittaker listed 36 green algal species, 63 red and 33 brown, and noted the presence of a rare charophyte *Lamprothamnion papulosum*, and several species of blue-green algae. Later, Holmes (1993) reported that the nationally rare alga *L. papulosum* had extended its range and abundance, and was found to be thriving better in 1991 than it had on previous surveys, although it was absent from the Abbotsbury embayment. In all surveys, this plant was found associated with coarse sandy/gravel substrata where minimal amounts of organic matter were present.

L. papulosum was common in the Abbotsbury embayment during the last century, samples collected at that time, and held at the Natural History Museum, are labelled as being taken from Abbotsbury[1]; This has recently been confirmed by changes in oospore concentrations in cores taken from the embayment (Martin and Carvalho, in press). Its disappearance from this part of the West Fleet could be linked to nutrient enrichment: studies indicate that *L. papulosum* appears to be absent from potentially suitable sites when levels of soluble reactive phosphate are greater than $30\mu g\ l^{-1}$ P, and is most frequently found where levels are less than $10\mu g\ l^{-1}$ P (Martin, 1999). Mean levels of orthophosphate (as elemental P) in West Fleet in the Abbotsbury embayment exceeded $20\mu g\ l^{-1}$ in 1998 whilst at other sites in the lagoon mean levels were well below $10\mu g\ l^{-1}$ P (see section 6.2.1). The most recent survey in July 2002 (Bunker *et al.*,2002) found that *Lamprothamnion* occurred mainly in embayments and

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[1] Marine SACs website http://www.ukmarinesac.org.uk/

shallow water from Lynch Cove to Rodden Hive. No *Lamprothamnion* was encountered northwest of Rodden Hive.

Holmes (1983) described the co-dominance of *Ulva* and *Chaetomorpha* sp. in the Abbotsbury embayment, whereas elsewhere, *Enteromorpha* and *Cladophera* spp. were more common. More recently, Saunders-Davies (1995) reported thick mats of chlorophytes (including *Ullothrix* and *Enteromorpha* spp.) growing below the high tide level along most of the length of the Fleet on the landward side.

In the deeper channels (2-5m) of East Fleet, sublittoral algae, including *Acrothrix gracilis, Cordylecladia erecta, Gloisiphonia capillaris* and *Gracilaria* spp (all three British species) occur. Other algae grow as epiphytes on *Zostera* leaves (e.g. *Cladosiphon zosterae),* or entangled around the bases (e.g. *Entermorpha flexuosa* and *Cladophora* spp.).

Exaggerated growth of *Ulva* and *Enteromorpha* is generally a sign of nutrient enrichment and occurs in response to eutrophication. Macroalgal blooms can result in reduced diversity of communities and species: 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), and blanketing of sediment can have major impacts on the underlying sediment, bringing bivalves to the surface (see also section 6.2.3). Other potential ecological impacts of green macroalgal blooms are discussed in Raffaelli *et al.* (1998) and include exclusion of surface deposit feeders and reduced abundance of invertebrate prey for fish and shore birds. Burrows (1981) recognised a nutrient enrichment problem in the Fleet, manifested by the large populations of *Ulva lactuca*, and noted that the whole system could eventually be affected and the situation should be watched. Burrows recommended surveillance of the chemical composition of the water in particular.

Several non-native species of macroalgae also occur in the Fleet, namely *Grateloupia filicina* var. *luxurians, Solieria chordalis* and *Sargassum muticum* (Whittaker and Farnham, 1983). The latter two species were thought to have migrated into the Fleet from Weymouth Bay on mobile substrates such as stones and shells.

Of these, *S. muticum* is potentially the more problematic. It is a large brown macroalga with fronds that can grow to over 3m in length and 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, the effect of its spread may not be all negative: 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).

The spread of *S. muticum* in the Fleet is monitored, and it is cut and removed annually from the narrows region by the army, although regrowth occurs quickly. A recent

study (Bunker *et al.,* 2002) found no deleterious effects to directly attributable of *Sargassum* cutting. Effective methods for the permanent removal of *Sargassum*, in the Fleet or elsewhere, have not yet been found (Farnham *et al*. 1981; Critchley *et al*., 1986).

Zostera and *Ruppia spp.*

The marine angiosperm *Zostera* is represented in the lagoon by two of the three species[2] that occur in the UK, *Z. angustifolia* and *Z. noltii* (Whittaker, 1978). The third, *Z. marina* is found in the adjacent Portland Harbour, and Weymouth Bay (Whittaker, 1981). All of these are considered nationally scarce. The plants stabilise the substratum, and are an important source of organic matter and a rich source of food, particularly for wildfowl (wigeon, Brent goose). Eelgrass beds also provide shelter and surface attachment for many species, such as fish (flatfish, cuttlefish, squid) and molluscs (*Rissoa membranacea, Akera bullata*) as well as algae, diatoms, anemones and stalked jellyfish *Haliclystus auricula*).

There have been several surveys carried out to establish the distribution of *Zostera* and spiral tassleweed *Ruppia cirrhosa* in the Fleet (Holmes, 1983, 1985, 1986, 1993; Whittaker, 1978,1981).

Whittaker (1981) reported extensive meadows of *Zostera* over the bed of the Fleet, from Lynch Cove, north of the narrows, almost to the Abbotsbury embayment. *Z. angustifolia* was the more common species, particularly in the easternmost *Zostera* beds. *Z. noltii* was reportedly much rarer and occurred in the lowest salinities of West Fleet. *Ruppia*, or 'wigeon grass' occurred only in West Fleet from Works Cove, Herbury, to Shipmoor Point, at the mouth of the Abbotsbury embayment, and was found mainly in association with *Zostera*. There was some evidence of a decline in both *Zostera* and *Ruppia* in West Fleet between 1968 – 1981 as Whittaker had previously mapped *Zostera* and *Ruppia* in the Fleet and found both in the Abbotsbury embayment (Whittaker, 1978). Whittaker (1981) suggested that this decline might be a result of overfeeding by the bird population, although this was acknowledged to be conjecture.

Holmes (1993) reported the distribution of *Zostera* and *Ruppia* in August 1991. The west-east and north-south zonation of species reported from previous surveys (August 1983 and 1985) was confirmed. *R. cirrhosa* was dominant west of Rodden Hive Point, but then gave way gradually to *Zostera* eastwards down the Fleet. *Z. noltii* appeared to thrive best along the northern coves, whilst *Z. angustifolia* dominated along the shore in the shadow of Chesil and the main body of the Fleet. Holmes considered that the decline of *R. cirrhosa* recorded in 1985 was shown by the 1991 surveys to have been a temporary phenomenon, *Z. noltii* and *Z. angustifolia* were all thriving in the Fleet in very similar proportions to those previously reported. However, no *Zostera* species were recorded in the Abbotsbury embayment, indicating that the seagrass had not re-established since Whittaker's earlier observation (Whittaker, 1978).

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[2] *Z. angustifolia* is considered by some workers (den Hartog 1989, Brenchley & Probert 1997) to be an ecological/environmental variation of *Z. marina*.

Table 3. Natural events and human activities which may contributing to the decline of *Zostera spp.* (adapted from Davison and Hughes (1998)

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Natural events

-*Zostera* beds are spatially dynamic, and subject to a number of naturally-occurring factors which can cause changes in coverage at a range of scales.
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- Extreme weather conditions such as violent storms or heavy floods can denude eelgrass beds over wide areas. Plants can also be killed or damaged by severe frosts.

- Wasting disease is the most important factor observed to cause long-lasting declines in the number and extent of *Zostera* beds. The most severe outbreak of this disease took place in the early 1930s, and recovery from this is still incomplete. The disease-causing agent is the fungus *Labyrinthula macrocystis*. This is probably continually present at low levels, but undergoes occasional epidemic outbreaks for reasons which are not fully understood. *Labyrinthula* does not appear to cause disease if salinity is low, so that the intertidal/estuarine *Zostera* species (*Z. angustifolia* and *Z. noltii*) are much less susceptible than *Z. marina*, which prefers subtidal marine conditions.

- Widlfowl grazing can remove a high proportion of the available *Zostera* biomass (over 90% in some cases), but beds can normally withstand this grazing pressure unless under stress from some other factor.

- Declines in populations of epiphyte grazers can indirectly affect the health of *Zostera* beds by allowing increased growth of fouling algae. Nutrient enrichment or other forms of anthropogenic pollution are the factors most likely to bring about such changes.

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Human activities

- A large proportion of the UK's population lives on or adjacent to the coast. As a result, pollution, development and recreation pressures are increasingly affecting the coastal environment, and their impacts can be especially acute in the shallow bays, estuaries and lagoons where *Zostera* biotopes most commonly occur.
</div>

- Coastal development can have adverse effects on *Zostera* beds by causing increased sediment erosion or accretion (depending on the nature of development), and by causing increases in water turbidity.

- There is little evidence of harm caused by heavy metals or antifoulants, but runoff of terrestrial herbicides has been shown to affect growth and survival of *Zostera* plants.

- Eelgrass beds are not highly sensitive to chronic oil pollution (eg. refinery effluent). However, when exposed to major oil spillages, the associated fauna appear to be more susceptible to damage than the *Zostera* itself. The chemical dispersants used to control oil spills are more harmful to *Zostera* than the oil alone, and should not be used in these biotopes.

- Excessive nutrient enrichment can cause damage to eelgrass beds by a variety of mechanisms, the most important of which are metabolic imbalance, proliferation of phytoplankton, epiphtyic or blanketing algae, and increased susceptibility to wasting disease.

- Eelgrass beds are not physically robust biotopes, and can be degraded by trampling, mechanical bivalve harvesting, dredging and other forms of disturbance.

- Two non-indigenous plants, the cord-grass *Spartina anglica* and the brown alga *Sargassum muticum* have colonized eelgrass beds in the UK, mainly in the south of England. To date, there is no firm evidence of either species competing significantly with *Zostera* or displacing it in the absence of other adverse environmental factors.

- Disturbance by wildfowlers may cause local increases in numbers of ducks and geese on *Zostera* beds, and hence higher grazing pressure on the eelgrass.

- Human-induced climate change may have significant long-term effects on the distribution and extent of *Zostera* beds. Possible significant effects include higher temperatures and increased frequency and severity of storms.

Overall, the evidence suggests that in general, the seagrass and tassleweed beds in the Fleet are flourishing. Changes in distributions may have occurred with time, although these are likely to be natural changes, associated with hard winters and ice damage, or recovery of *Zostera* from the wasting disease prevalent in the 1930's. The decline of *Zostera* in the Abbotsbury embayment may be important, however, and could be linked to several factors. Continuation of the monitoring programme would seem prudent as a reduction in the biomass can be an early indication of stress in seagrass beds and it is important to identify possible stressors. Its decline nationally may have serious consequences for the rich and diverse fauna often associated with beds of *Zostera* and fine algae, and for the waders and wildfowl which feed amongst the beds. This is particularly relevant in SPAs and SACs, where rare and nationally important species occur. Davison and Hughes (1998) have produced a comprehensive overview of dynamics and sensitivity characteristics of *Zostera* in UK SACs, and although it is difficult to speculate on the exact cause of the decline, table 3 summarises natural events and human activities which may be contributing factors.

Although *Zostera* species are fast-growing and relatively short-lived, they can take a considerable time to recover from damaging impacts - if recovery is possible at all. Holt *et al.,* (1997) estimated that *Zostera* species recoverability is within the range of five to ten years but, in many cases, recovery may take longer. This is borne out by the slow or apparent lack of recovery from the 1920s to mid-1930s wasting disease epidemic. Table 4 summarises the key factors which may limit or facilitate seagrass bed recovery in marine SACs and elsewhere.

Table 4. Summary of major factors believed to influence the capacity of *Zostera* beds to recover after disturbance or destruction (from Davison and Hughes 1998).

Factors that may limit bed recovery	Factors that may facilitate bed recovery
Removal of habitat	Artificial transplantation
Unstable substrata	Stable substrata
Fragmenting and destabilized *Zostera* beds, caused by factors such as changes to coastal processes, physical damage or stochastic weather events	Stable *Zostera* beds
Reduced rhizome growth, seed production, germling success and seedling development into patches	Increased rhizome growth, seed production, germling success and seedling development into patches
Reduced light penetration, caused by increased turbidity, eutrophication, some forms of pollution, or epiphyte smothering	Improvements in light penetration, caused by reductions in turbidity, eutrophication, pollution, epiphyte and algal smothering
Nutrient enrichment	Reductions of, or limited increases to, nutrient inputs
Declines in epiphyte grazer populations. Unusual increases in wildfowl grazing pressure	Healthy and stable epiphyte grazer populations. Wildfowl grazing activities may prevent excessive sediment build up in *Zostera* beds
Competition with non-native species, *Spartina* sp. and *Sargassum muticum*	Absence of non-native species, *Spartina* sp. and *Sargassum muticum*
Environmental stress, (e.g. extreme temperatures or pollutants), which may increase the susceptibility to wasting disease infection	Absence of environmental stresses and low populations of *L. macrocystis*, the causative fungal pathogen for wasting disease

Macrofauna

Dyrynda (1997) reported the results of monitoring in the Fleet lagoon 1995-1996; there are a variety of invertebrates, including some rare species, within the lagoonal basin. Some of the invertebrates are zoned in abundance both along and across the lagoon. The fauna comprises both common and rare lagoonal specialists, and estuarine brackish species. Lagoonal cockles *Cerastoderma glaucum* are widespread, as is the crustacean *Idotea chelipes* and gastropod molluscs *Hydrobia ventrosa, Rissoa membranacea* and *Littorina saxatilis*. A dwarf variety of the nudibranch mollusc *Akera bullata* (var. *nana*) is also common. Common species were relatively evenly distributed whilst other lagoonal specialists were much more localised, e.g. the starlet anemone *Nematostella vectensis* and the crustacean *Gammarus insensibilis*, both of which were found principally within the upper, more brackish reaches of the West Fleet basin adjacent to Clouds Hill. The more abundant brackish water species include the polychaete worm *Scoloplos armiger* and the nemertine *Lineus viridis* (Dyrynda 1997).

Principal findings of the subtidal benthic survey of Dyrynda and Cleator (1995) are;
- The fauna at Ferrybridge is distinctly more marine in nature than that of the Narrows and includes e.g. *Cancer pagurus* (edible crab) and *Maia squinado* (spider crab), the hydroid *Sertularia cupressa* and the bryozoan *Scruparia ambigua.*
- There is exceptionally high abundance of some of the more common marine species within the highly specialised conditions of the link channel, e.g. snakelocks anemone *Anemonia viridis* and cushion star *Asterina gibbosa*.
- Sponge *Halichondria bowerbanki* grows to an exceptional size in waters of East Fleet.
- First record of warm water pycnogonid *Tanystylum conirostre* in UK mainland (recorded at Small Mouth)
- Hydroids *Monotheca oblique* and *Sertularia distans* were recorded in the Fleet for the first time (in the Narrows).
- Infaunal communities of West Fleet are typical of estuarine muddy sediments and include a number of lagoonal species such as lagoon sand shrimp *Gammarus insensibilis*, lagoon cockle *Cerastoderma glaucum* and starlet sea anemone *Nematostella vectensis* which are all nationally rare and protected under the Countryside Act (1981).
- Large numbers of chironomid larvae present in the western end of the lagoon are indicative of the lower salinity of this region, but may also be a result of organic enrichment.
- According to Dyrynda (1997), overall, the most obvious characteristic of the data collected was species abundance, rather than composition. Diversity and abundance maxima peaked at Langton Hive Point and declined toward the east.

Ladle (1986) edited a collection of papers describing the biology of the Fleet, which included a report describing the changes observed during the Ferrybridge reconstruction when the old channel was infilled and a new one constructed (Seaward, 1986). Molluscs, including the intertidal rare sea slug *Aeolidiella alderi*, appeared to be unaffected by the development, although Seaward notes that the local population

of *A. alderi* was reduced by severe weather in January 1985. During the infilling, a small subtidal community (which included the sea slug *Doto millbayana*) under the old bridge was destroyed. Changes to the sand flats in Portland Harbour adjacent to the new channel were also observed, but otherwise, no significant impacts were reported.

Thompson and Seaward (1986) described the nationally scarce opisthobranch mollusc *Akera bullata* (subspecies *nana*) in the Fleet, together with aspects of its taxonomy. The Fleet population of *A. bullata* has existed in the lagoon since around 1850 and individuals rarely exceed 20mm in overall body length. It is ecologically distinct from larger *A. bullata* (subspecies *farrani*) found elsewhere. Densities of up to 120 per m^2 were recorded by Seaward (1978). Other rare species recorded in the lagoon include lagoon sand worm *Armandia cirrhosa*, lagoon snail *Caecum armoricum*, and the starlet anemone *Nematostella vectensis*. Also present are the gastropod *Paludinella littorina* and the lagoon sea slug *Tenella adspersa*, both of which are nationally rare and protected.

The non-native (far-eastern) ascidian *Perophora japonica* was observed for the first time at Pirates Cove in 2000 and, in the early summer of 2001, was found in a dense mass near to the south-eastern end of the Fleet between the Narrows and Small Mouth (Baldock and Bishop, 2001). *P. japonica* was present here at depths of between 2.2 and 3.8m occupying up to 10% of the available substrate (mixed muddy sand with pebbles and shells to which the ascidian attaches). By late summer however the species appeared less abundant. The only other site where this species has been recorded previously is Plymouth Sound, in 1999. It is unlikely that the ascidian spread the 130km from Plymouth in this short period of time and therefore its occurrence in the Fleet is more likely due to shipping in the adjacent waters of Portland Harbour. Monitoring of this potentially prominent invasive species and its occupation of subtidal habitat is recommended.

Native oysters *Ostrea edulis* have been collected from the lagoon since the 11th century although the first record of farming oysters was not until the mid-1700s. Pacific oysters *Crassostrea gigas* are currently farmed in the waters of the Fleet. There were 10,000 racks on the shores in the early 1990s, which supported annual production of 100 tonnes of oysters. One effect of this cultivation is the creation of an 'artificial reef' which in turn attracts many young fish using the reef for shelter and protection (Copperthwaite, 1993).

Seaward (1994) monitored water temperature in the Fleet during 1993 and 1994 as part of an investigation into the possibilities of the introduced pacific oyster breeding in British waters. Peak water temperature (17-18ºC) occurred in June-August, and minimum water temperature (6ºC) in February, indicating that during 1993-94, temperature was not a limiting factor to spawning and settlement of *Crassostrea gigas*, although successful spawning of this species had not been observed up to that time. Whittaker (1978) had previously indicated that more extreme temperatures may occur, up to 26-28ºC in summer and minima in January/February of below freezing, when the water surface has frozen.

Meiofauna

The meiofauna of the lagoon has been little reported. It is a vital link in many of the complex food webs of benthic communities (Thorson, 1966) and includes harpacticoid copepods, ostracods, foraminifera, and polychaete and oligochaete worms. Humphrey (1986) carried out a survey of the meiofauna of the Fleet as part of an MSc project. Sediment samples taken at low water from five sites within the Fleet were analysed. Greatest numbers of individuals was found at Moonfleet (table 5) as was highest diversity and abundance of species. High numbers of individuals were also recorded at Abbotsbury, where sediment sorting was greatest (although still relatively poorly sorted). Nematodes were the major taxa overall, and at most sites: the exception was Morkhams Lake, where the greatest abundance of animals were foraminifera. The Fleet was therefore considered to be rich in terms of meiofaunal abundance and number of species, especially for harpacticoids and foraminiferans.

Table 5. Meiofaunal abundance in fleet sediments (from Humphrey, 1986)

Site	Number/$10cm^2$
Chickerell Hive Pt	6062
Moonfleet	10122
Langton Hive Pt	7078
Morkhams Lake	4335
Abbotsbury embayment	9517

The sediment samples were analysed for temperature, particle size and organic content. Salinity and temperature of the overlying water at the sampling sites was also taken. Highest water temperatures were recorded at Moonfleet. pH measurements were unusually high in the Abbotsbury embayment (up to 9.9 compared with the 'normal' seawater pH of 8). The high pH in the upper Fleet was later attributed to photosynthetic activity of seagrasses (Saunders-Davies, 1995). The same author considered the phenomenon to be partly responsible for the preservation of numerous empty Foraminifera tests which were observed in the samples.

Saunders-Davies (1995) also examined the distribution of benthic and littoral rotifers in the Fleet and described a new species which was named after the lagoon - *Proales fleetensis*. Two rotifer species, *Aspelta clydona* and *A. harringii* were the first recorded in Britain. A significant population of rotifers was discovered living amongst the algae, and species richness and total abundance shown to increase along the decreasing salinity gradient between Ferrybridge and Abbotsbury. A strong negative correlation was found between salinity and total abundance (r=0.94, p<0.05), and to a lesser extent between salinity and number of species (r=0.81, p=0.1) which indicated a greater number of fresh/brackish water species. During the period of the survey (June to Sept 1993) an explosive growth of the sulphur bacterium *Thiopedia roseola* occurred at Morkhams Lake, which covered the benthic algae. Rotifers then became abundant in the mats of filamentous algae floating clear of the bottom.

Fish

Ladle (1981) noted that at least 23 species of fish had been recorded in the Fleet, the most abundant forms being euryhaline species such as the eel *Anguilla anguilla* and flounder *Pleuronectes flesus.* Later, Ladle (1986) studied the fish of the Fleet, in particular, bass *Dicentrarchus labrax*, to establish whether this species spawns within the lagoon or whether, as it was generally believed, spawning occurred chiefly in offshore situations and larvae migrate inshore. Surveys were conducted in May-October 1983 and provided no evidence of bass spawning in the Fleet, although the status of the lagoon as a nursery area was confirmed. The population was described as substantial, with young fish relatively quick growing for a species nearing the northern limits of its geographical distribution. Other fish recorded in the study included large shoals of mullet (*Crenimugil labrosus* and *Liza aurata*), which are fished commercially by local fishermen, sandsmelt *Atherina presbyter*, sandeels *Ammodytes tobianus*, post-larval and adult flounder *P. flesus,* rockling *Ciliata mustela*, blackface blenny *Tripterygion delaisi*, wrasse (ballan, goldsinny and corkwing), pouting, and three- and fifteen-spined stickleback.

Gobies *Pomatoschistus spp.* were the most abundant fish during the survey, with sandsmelt the second most abundant (Ladle, 1986). A the end of May, sandsmelt eggs were found in the plankton on the ebb tide, attached to algal fragments, indicating that they are benthic rather than planktonic. This has rarely been observed in other British waters. Sandsmelt in the Fleet were also the subject of a study by Bamber and Henderson (1985) which established that the population exhibited a high infection rate of diplostomiasis, a parasitic fish eye-fluke disease which appears to be limited to the mid English Channel. Analysis of postlarvae and juveniles from the Fleet showed that infection can occur at 1 week old and verified the hypothesis that the scales of older fish inhibit settlement of the early stages of this parasite. Circumstantial evidence suggested that *Hydrobia ventrosa* may be the first vector host for this parasite.

Other fish recorded in the Fleet include greater-, lesser- and snake-pipefish, butterfish, gunnel, blennies, gobies and short-spined sea scorpion (Cook, 1969, Dyrynda and Cleator, 1995).

Birds

Recently, the British Trust for Ornithology (BTO) 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 (Armitage *et al*., 2002) summarises statistics for Chesil and the Fleet cSAC, SPA.

The one cited species present at a level of international importance is the Dark-bellied Brent Goose. The WeBS count site includes part of Portland Harbour and the Radipole Lake and Lodmoor RSPB sites, which are outside of the SPA, but do not hold significant numbers of geese. The majority of Dark-bellied Brent Geese counted utilise The Fleet. Numbers increased during the 1970s and 1980s but have decreased from a peak of 1600 in 1991/2 to 650 in recent winters. This mirrors regional and national trends and therefore no alerts have been triggered. Any adverse factors such as pollution from domestic sewage and agriculture, and disturbance from military activities, were not considered sufficient to warrant additional investigation into population trends at the site (Armitage *et al*., 2002). However, in view of the suspected sensitivity of the site to nutrients, and the largely unknown status of many toxic contaminants, the acquisition of any further information which addresses possible links between environmental quality and biological consequences seems desirable.

6. TOXIC AND NON-TOXIC CONTAMINANTS

There are relatively few published statistics available as to the relative contaminant loadings from different sources into Chesil and the Fleet cSAC, SPA. Where published information on sources 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 the Fleet/Chesil Beach, specifically, therefore 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.
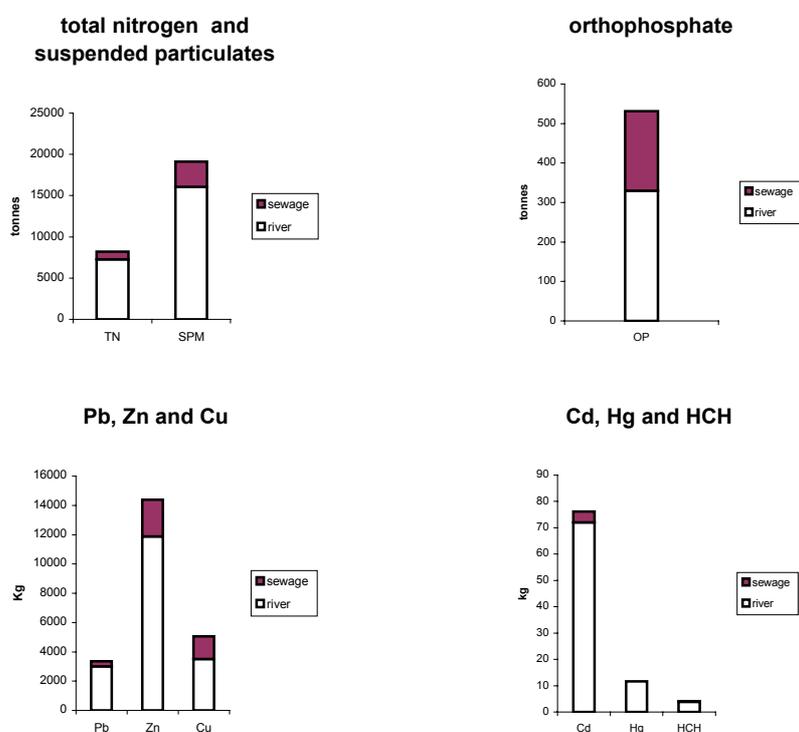
Figure 3. Relative loadings for OSPAR determinands from rivers and sewage discharging to the sea between Bournemouth and Lyme Regis, 1999. Data source EA.
NB Principal rivers are sampled just upstream of their tidal limits to assess freshwater discharges into marine waters therefore riverine sources may also contain an indirect sewage and industrial component.

In sections 6.1 and 6.2, below, we discuss published and unpublished information for toxic- and non-toxic contaminants, respectively. Before doing so, however, we outline briefly the rationale and limitations of the current assessment of environmental quality in the European marine site.

Water quality and environmental standards

Because of the paucity of contaminant studies on Chesil and the Fleet cSAC, SPA, the assessment of environmental quality draws heavily on data for key determinands supplied by EA. Summary statistics have been drawn up by the EA (based on monitoring since 1990), and the raw data analysed in an attempt to establish further evidence as to whether or not existing 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. These statistics are broken down in to:

1) **Discharges** – to gauge the importance of specific point sources (mainly STWs). The locations of consented discharges to the Fleet are shown in figure 2. 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) **Fresh waters** - usually sites immediately above the tidal limit (to characterise riverine input). Data are not always available at each site, for each determinand – the majority of information relates to nutrients in tributaries of the Fleet (figure 4).

3) **Tidal waters** – a review of data for saline waters within the Fleet itself and the adjacent tidal waters of Portland Harbour. The location of sample sites monitored by the Agency within the Fleet (and Portland Harbour) are shown in figure 5.

   Water quality along Chesil Beach, and how this may be influenced by STW discharges at West Bay (Bridport) and Weymouth is also considered briefly (figure 6 shows location of sampling points and outfalls). Again data are not always available at each site, for each determinand.

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 trends in water quality, and to enable comparisons with Environmental Quality Standards.
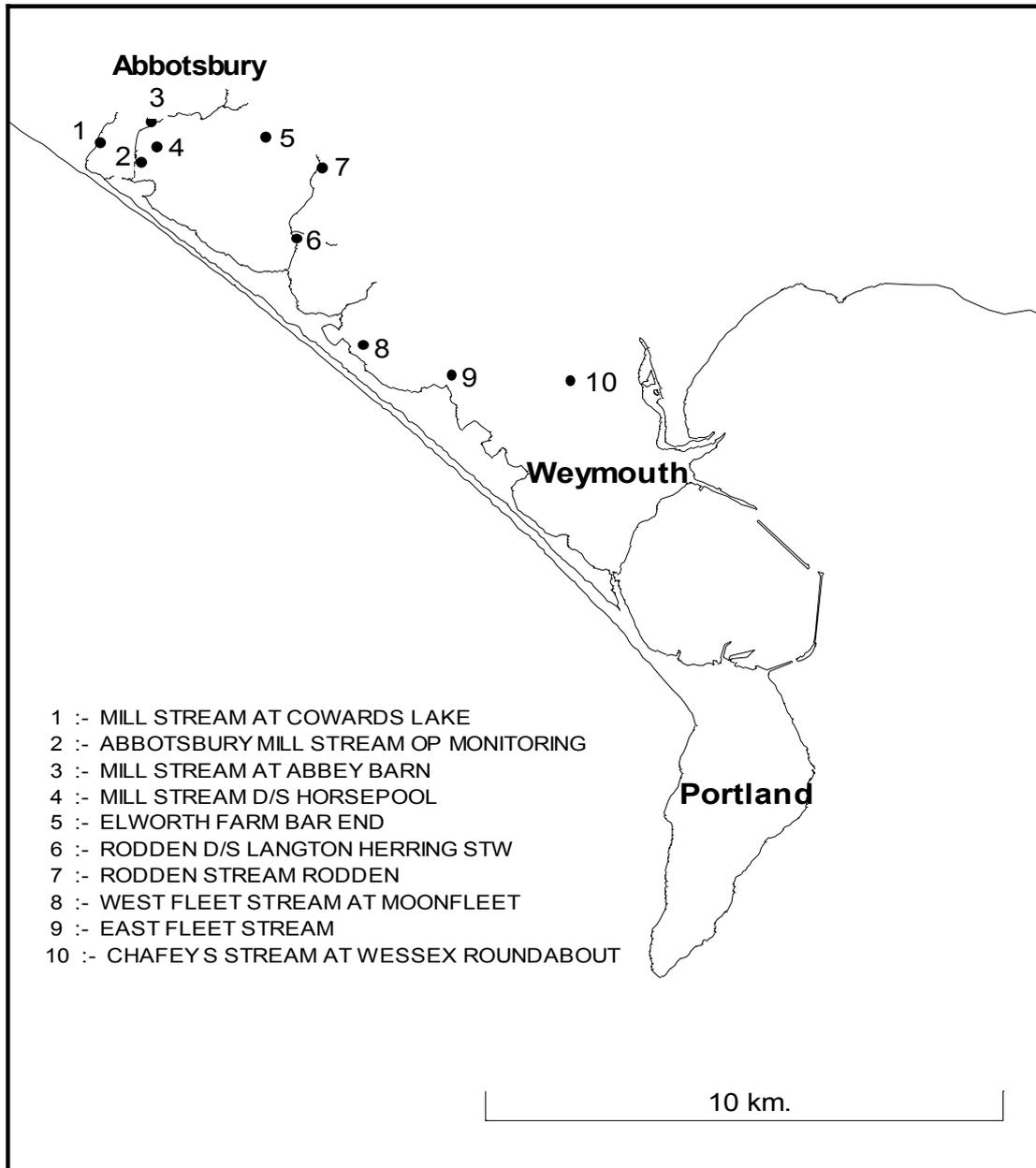
Figure 4. Location of the Environment Agency sampling sites (fresh waters) in the Fleet for which data have been supplied.

(Note: Site 10 is not in the Fleet catchment (Wey). Site 5 is no longer sampled. The Herbury Stream at Under Cross Plantation (SY 6156 8146) is now also sampled although no data were available at the time of writing.)
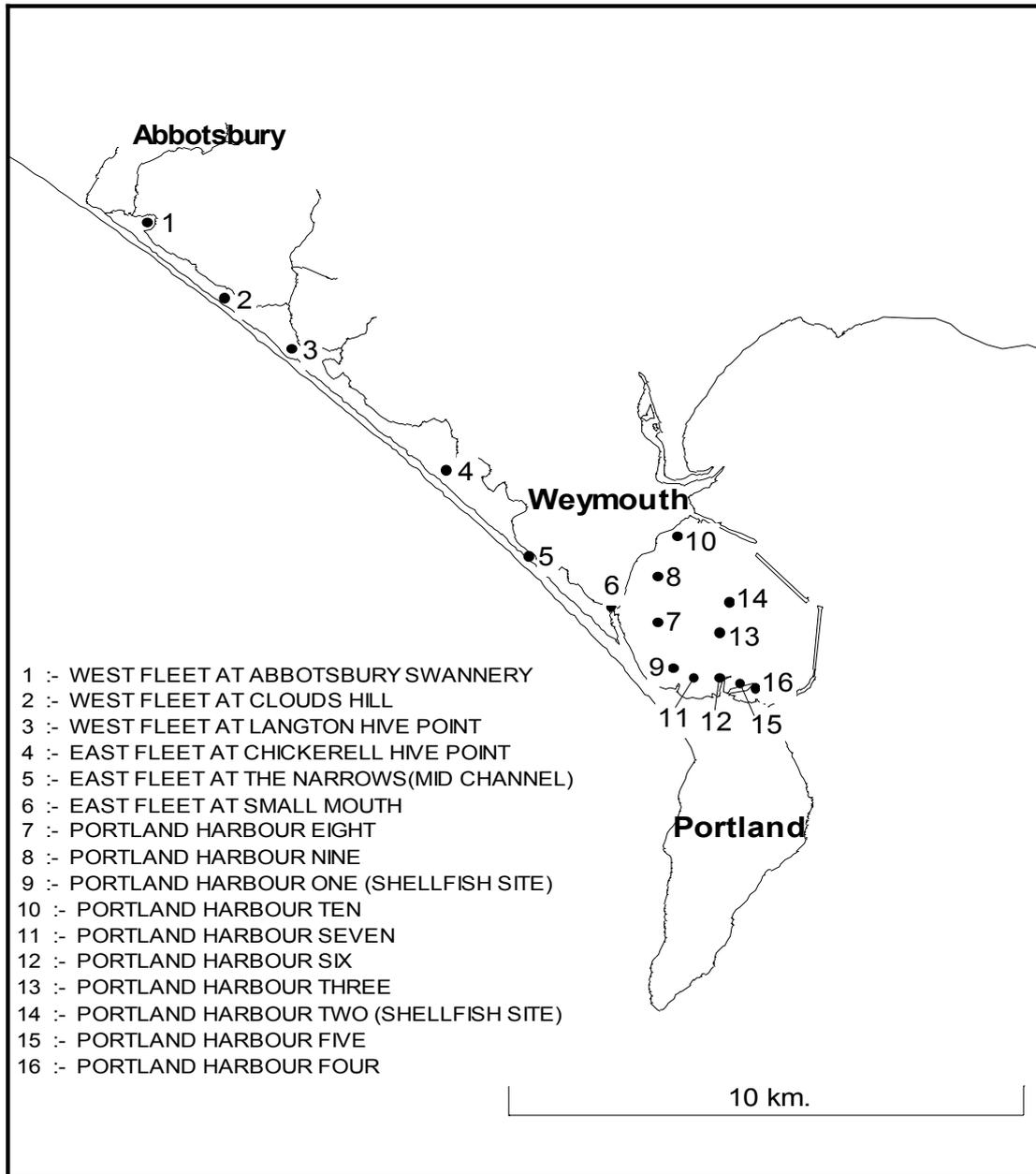
Figure 5. Location of the Environment Agency sampling sites (saline waters) in the Fleet and Portland Harbour.

Legend within figure:

1 :- WEST FLEET AT ABBOTSBURY SWANNERY
2 :- WEST FLEET AT CLOUDS HILL
3 :- WEST FLEET AT LANGTON HIVE POINT
4 :- EAST FLEET AT CHICKERELL HIVE POINT
5 :- EAST FLEET AT THE NARROWS(MID CHANNEL)
6 :- EAST FLEET AT SMALL MOUTH
7 :- PORTLAND HARBOUR EIGHT
8 :- PORTLAND HARBOUR NINE
9 :- PORTLAND HARBOUR ONE (SHELLFISH SITE)
10 :- PORTLAND HARBOUR TEN
11 :- PORTLAND HARBOUR SEVEN
12 :- PORTLAND HARBOUR SIX
13 :- PORTLAND HARBOUR THREE
14 :- PORTLAND HARBOUR TWO (SHELLFISH SITE)
15 :- PORTLAND HARBOUR FIVE
16 :- PORTLAND HARBOUR FOUR

10 km.

**Figure 6. Location of the Environment Agency sampling sites (saline waters) along Chesil Beach, and positions of the Weymouth and Bridport long-sea outfalls.**

In the absence of extensive site-specific biological effects information, results from water-monitoring are compared with Environmental Quality Standards (EQS) in order to gain a first-order approximation of possible impact on biota. 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.

The majority of List I and List II (Dangerous Substances) determinands have been screened within the marine site and its catchment area (section 6.1), together with other water quality parameters such as nutrients, suspended solids and DO ('non-toxic contaminants'- section 6.2).

6.1 Toxic contaminants

6.1.1 Metals

Water

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

Summary statistics for dissolved As concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 7. This figure also includes sites in Portland Harbour, including shellfish sites (=Portland 1 and 2) for comparison. Annual averages are invariably below the EQS for tidal waters (25 µg l^{-1}), by more than an order of magnitude. Even highest individual concentrations are only marginally above background. There is no indication that As concentrations in the Fleet, or adjacent waters, pose a threat for biota.

There are no obvious temporal trends for As in tidal waters.

Dissolved Arsenic in saline waters
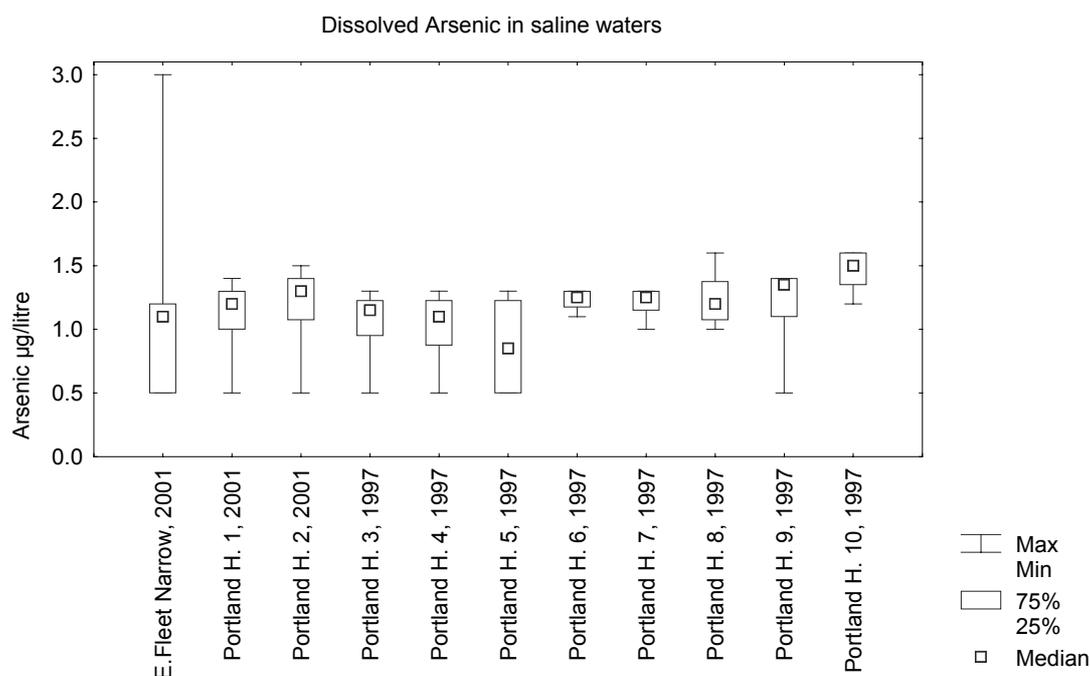
Figure 7. Concentrations of dissolved As (µg l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.

The EQS for As in fresh waters is 50µg l^{-1}.

Dissolved As concentrations have not been measured in any of the streams or tributaries entering the Fleet. The River Brit, flowing into the sea at Bridport, immediately to the west of the Chesil Beach, contains only background concentrations of dissolved As (figure 8).

The only discharge data for dissolved As are for Weymouth STW. Annual median concentrations in 1990 and 1997 were below 1 µg l^{-1}. An estimated 8.1 kg were released to controlled waters in 2001 (annex 7).

In view of the background concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges are significant sources of As to the cSAC/SPA.
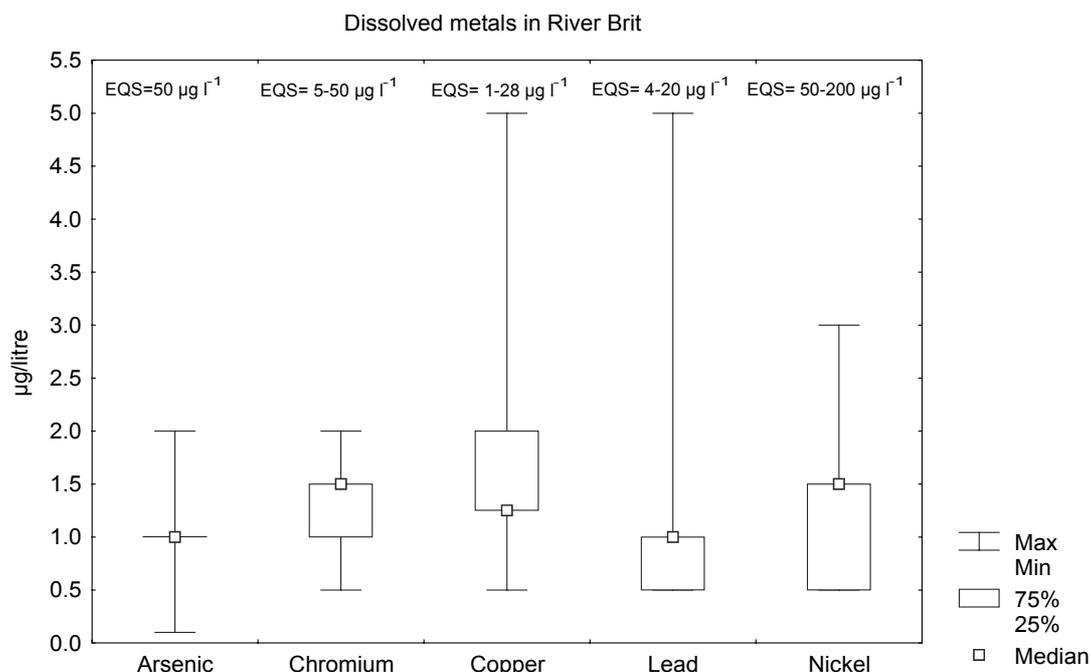
Figure 8. Summary statistics for dissolved As, Cr, Cu, Pb and Ni in the River Brit 1990-2002 (Data source EA).

Cadmium

Dissolved Cd concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 9. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison.

The apparent distribution of dissolved Cd in tidal waters is largely determined by the influence of detection limits (more than 90% of tidal water values were below DL). Nevertheless, annual averages are invariably below the EQS for saline waters (2.5 µg l^{-1}) by at least one order of magnitude as indicated in figure 9.

There is no indication that Cd concentrations in the Fleet, or adjacent waters, pose a threat for biota.

There are no obvious temporal trends for Cd in tidal waters.

Dissolved Cadmium in saline waters
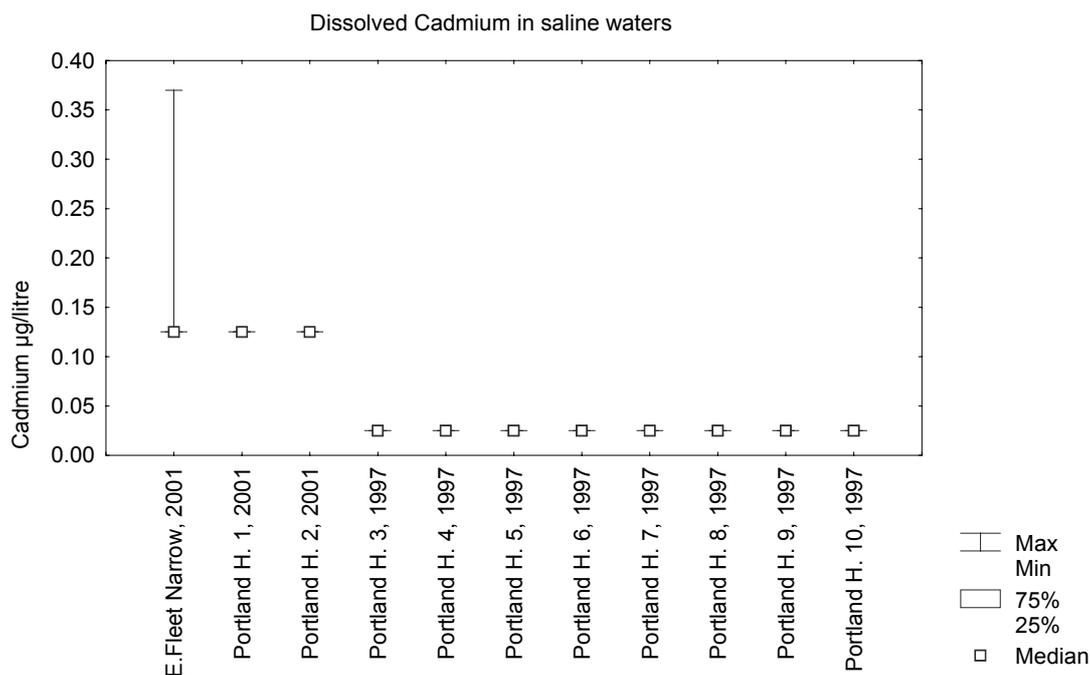
Figure 9. Concentrations of dissolved Cd ($\mu g\ l^{-1}$) in tidal waters, Fleet and Portland Harbour. Data source EA.

The EQS for Cd in fresh waters is $5\mu g\ l^{-1}$ ('total' metal).

Dissolved Cd concentrations have not been measured in any of the streams or tributaries entering the Fleet.

The only discharge data for dissolved Cd are for Weymouth STW. The annual average concentration in 1997 was $0.1 \pm 0.08\ \mu g\ l^{-1}$. It is estimated that <1kg were released to controlled waters in 2001 (annex 7).

In view of the low concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges are significant sources of Cd to the cSAC/SPA.

Chromium

Summary statistics for dissolved Cr concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 10. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison.

Median values for all sites are essentially equivalent to (half) detection limits, throughout the area (95% of all values were <DL). These are lower than the EQS (15 $\mu g\ l^{-1}$) by more than an order of magnitude.

There is no indication that dissolved Cr concentrations in the Fleet, or adjacent waters, pose a threat for biota.

There are no obvious temporal trends for Cr in tidal waters.
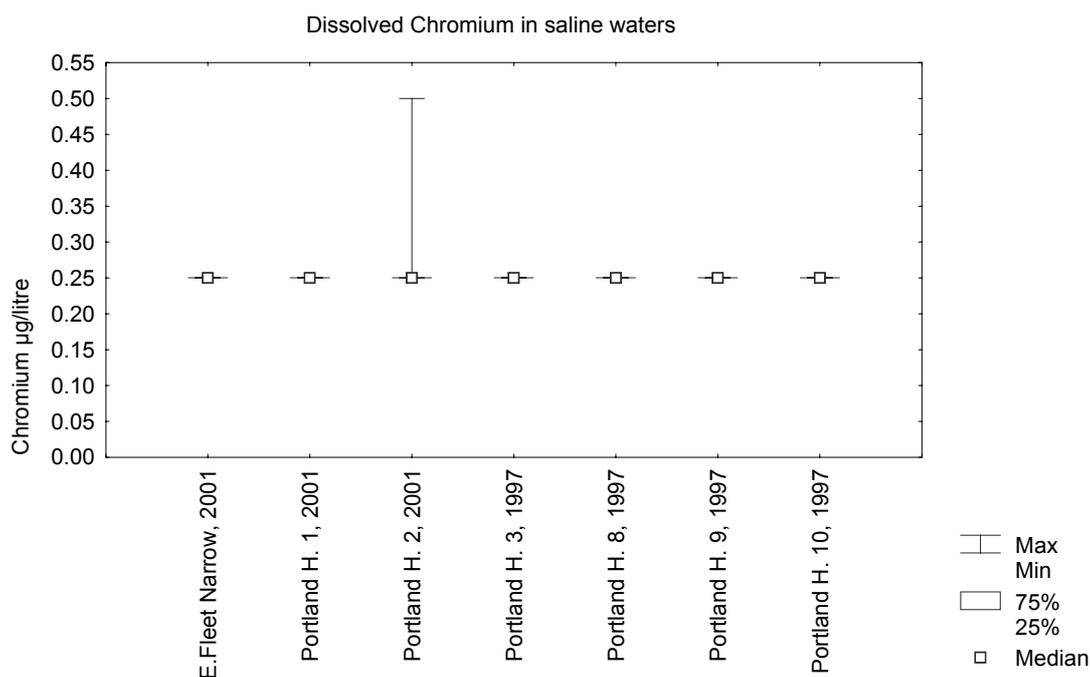
Figure 10. **Concentrations of dissolved Cr (μg l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.**

The EQS for Cr in fresh waters (suitable for salmonids) ranges between 5 and 50 μg l^{-1} depending on hardness. Dissolved Cr concentrations have not been measured in any of the streams or tributaries entering the Fleet. The River Brit contains only background concentrations of dissolved Cr (figure 8).

The only discharge data for dissolved Cr are for Weymouth STW. The annual average concentration in 1997 was 4 ± 3.5 μg l^{-1}. It has been estimated that <20 kg were released to controlled waters in 2001 (annex 7).

In view of the low concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges are significant sources of Cr to the cSAC/SPA.

Copper

Summary statistics for dissolved Cu concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 11. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison.

Just over half of all tidal waters samples were above detection limits for Cu. Annual median values for the sites shown range between 0.25 and 0.78 μg l^{-1} and are lower than the EQS (5 μg l^{-1}) by a significant margin. The maximum recorded value, in Portland Harbour, was 4.9μg l^{-1}. It is therefore unlikely that dissolved Cu concentrations in tidal waters would have deleterious effects on biota in the cSAC/SPA.

There are no obvious temporal trends for Cu in tidal waters.

Dissolved Copper in saline waters
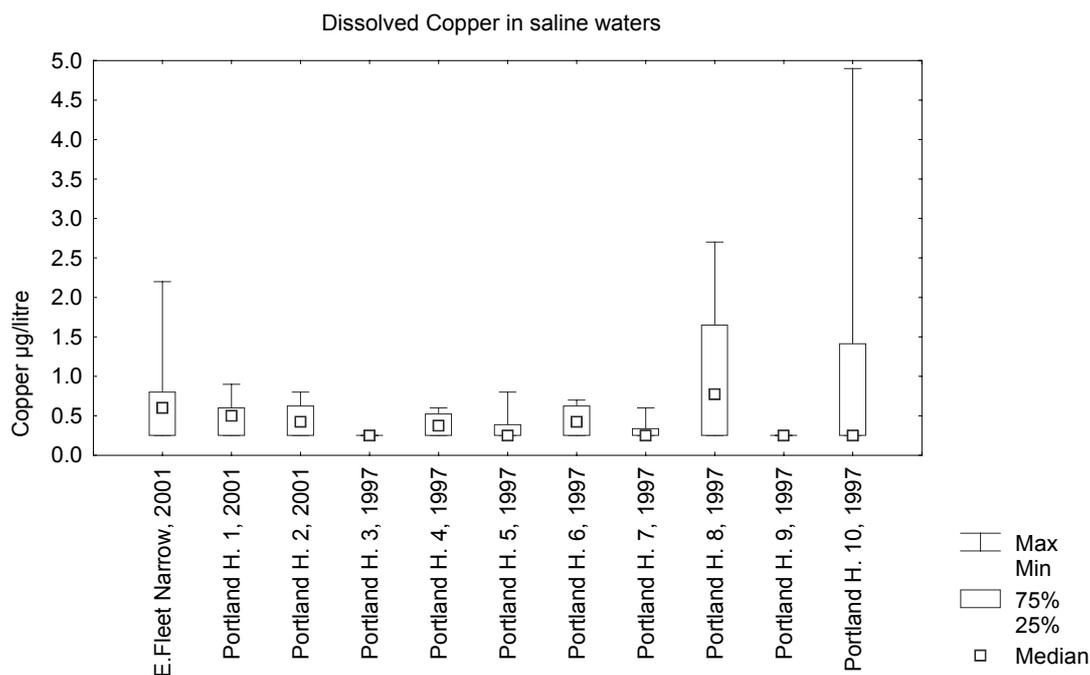
Figure 11. **Concentrations of dissolved Cu (μg l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.**

The EQS for Cu in freshwater is in the range 1 - 28 μg l^{-1} depending on hardness.

Dissolved Cu concentrations have not been measured in any of the streams or tributaries entering the Fleet. Summary statistics for the River Brit (figure 8) suggest median levels are slightly above the lowest EQS for dissolved Cu (1μg l^{-1} for waters with hardness between 0-50 mg/l CaCO$_3$), though this impression may be misleading since more than half the values (57%) are below detection limits. It is also likely that hardness, and hence the EQS, may be towards the upper end of the range. The River Brit is not likely to be significant source of Cu for the Chesil Beach site.

The only discharge data for dissolved Cu in the region are for Weymouth STW. The annual average concentration in 1997 was 75 ± 41 μg l^{-1}. An estimated 120 kg of copper were released to controlled waters in 2001 (annex 7). Clearly there is a small but measurable input of Cu from Weymouth STW, though this is likely to be rapidly diluted at sea.

The low Cu concentrations observed in tidal waters of the Fleet imply that it is unlikely that rivers or point source discharges have any significant effect in the lagoon.

Iron

There are no summary statistics for dissolved Fe concentrations in tidal waters of the Fleet. Data for sites in Portland Harbour are plotted in figure 12.. Highest concentrations were those at shellfish directive sites (=Portland 1 and 2). However, median values for all sites were well below the EQS (1000 μg l^{-1} dissolved Fe).

33

On this evidence it is unlikely that dissolved Fe concentrations in tidal waters would have deleterious effects on biota in the Fleet Lagoon.

There are no obvious temporal trends for Fe in tidal waters.

Dissolved Iron in saline waters
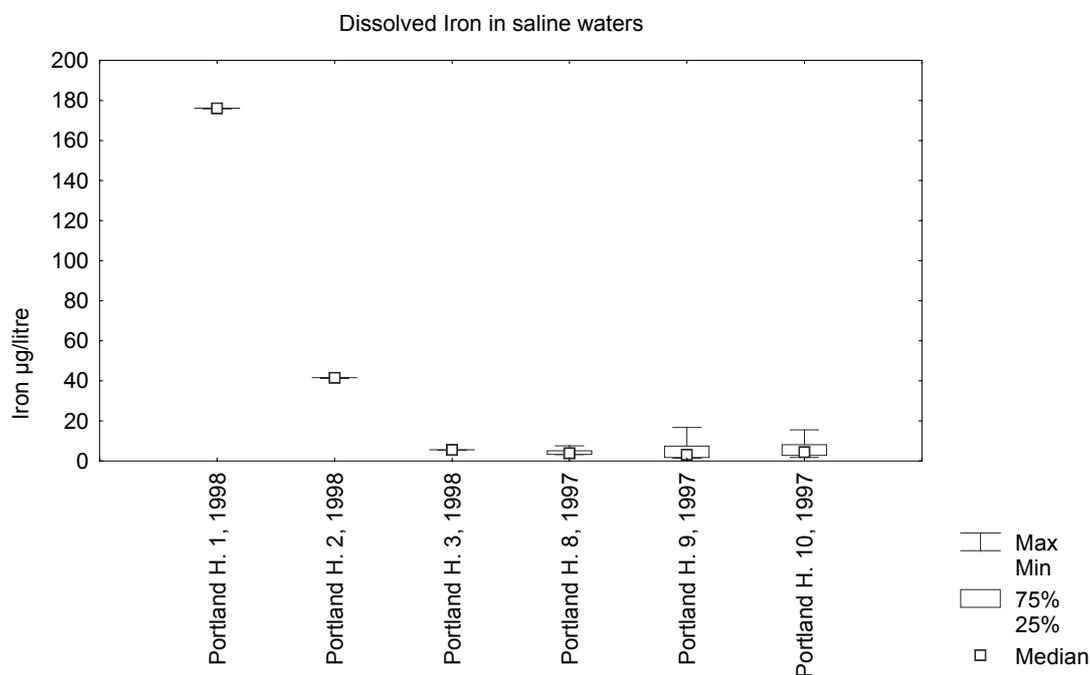
Figure 12. Concentrations of dissolved Fe (µg l⁻¹) in tidal waters, Fleet and Portland Harbour. Data source EA.

The EQS for Fe in freshwater is 1000µg l⁻¹.

Dissolved Fe concentrations have not been measured in any of the streams or tributaries entering the Fleet. The River Brit carries variable levels of dissolved Fe, sometimes above 1000 µg l⁻¹, though on average concentrations are below the EQS and unlikely to be of significance for the adjacent Chesil Beach site (median = 70 µg l⁻¹, figure 13). Iron is relatively insoluble in sea water and much of the river-borne Fe would precipitate rapidly during the early stages of mixing.

The only discharge data for dissolved Fe are for Weymouth STW. The annual average concentration in 1997 was 260 ± 200 µg l⁻¹ and clearly there is a small but innocuous input of Fe from Weymouth STW. However, in view of the relatively low concentrations of dissolved Fe observed in Portland Harbour it is unlikely that tidal waters represent significant sources of Fe to the Fleet.
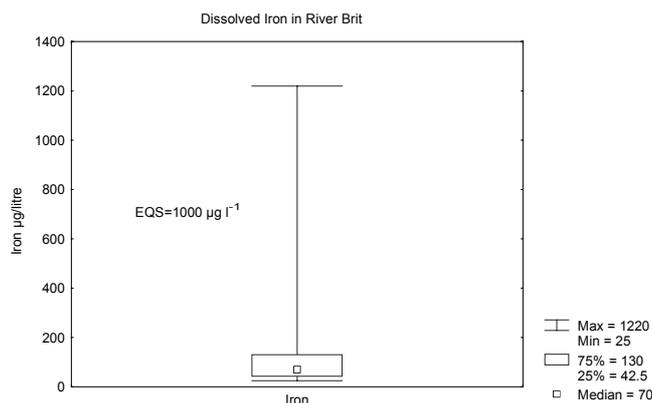
Figure 13.   Summary statistics for dissolved Fe in the River Brit, 1990-2002 (Data source EA).

Nickel

Summary statistics for dissolved Ni concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure14. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison.

Median values for all sites are essentially equivalent to (half) detection limits, throughout the area (96% of all values were <DL). These are lower than the EQS (30 $\mu g\ l^{-1}$) by more than an order of magnitude.

There is no indication that dissolved Ni concentrations in the Fleet, or adjacent waters, pose a threat for biota.

There are no obvious temporal trends for Ni in tidal waters.
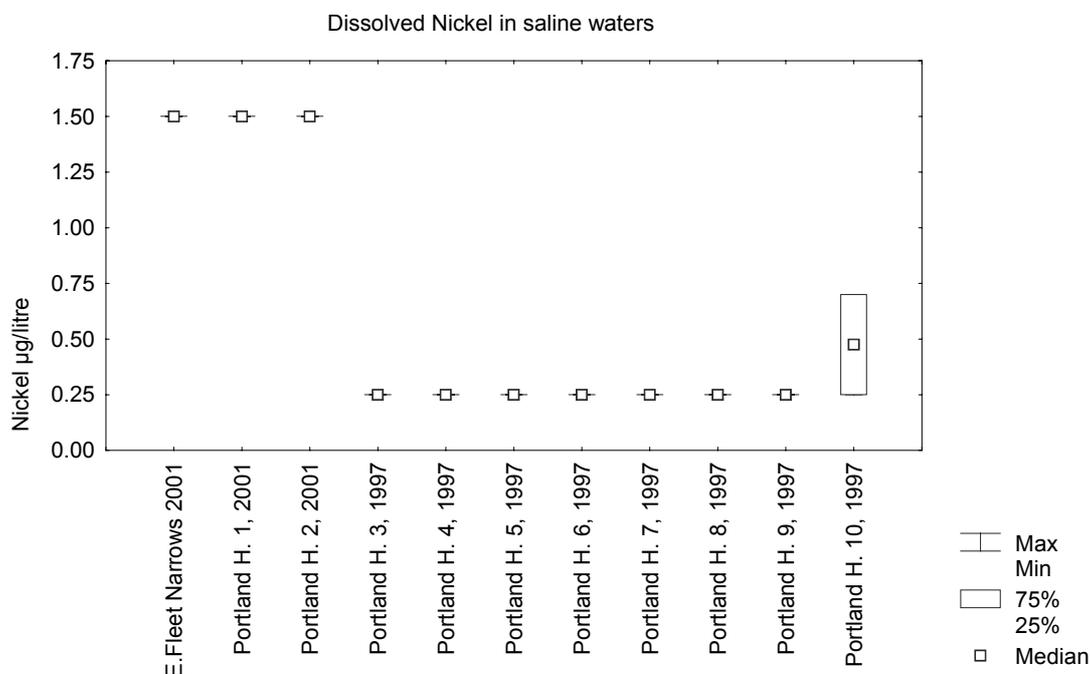
Figure 14.   Concentrations of dissolved Ni ($\mu g\ l^{-1}$) in tidal waters, Fleet and Portland Harbour. Data source EA.

35

The EQS for Ni in fresh waters (suitable for salmonids) ranges between 50 and 200 $\mu g\ l^{-1}$ depending on hardness .

Dissolved Ni concentrations have not been measured in any of the streams or tributaries entering the Fleet. Summary statistics for the River Brit (figure 8) indicate that median concentrations are significantly below even the lowest EQS for dissolved Ni (50 $\mu g\ l^{-1}$ for waters with hardness between 0-50 mg l^{-1} $CaCO_3$). The River Brit is therefore not likely to be significant of Ni for the Chesil Beach site.

The only discharge data for dissolved Ni are for Weymouth STW. The annual average concentration in 1997 was 51 ± 43 $\mu g\ l^{-1}$. Clearly there is a small, if probably innocuous, input of Ni discharging into the eastern part of Lyme Bay from Weymouth STW. However, in view of the low concentrations observed in tidal waters of the Fleet it is unlikely that point source discharges or rivers are significant sources of Ni to the cSAC/SPA.

Lead

Summary statistics for dissolved Pb concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 15. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison.
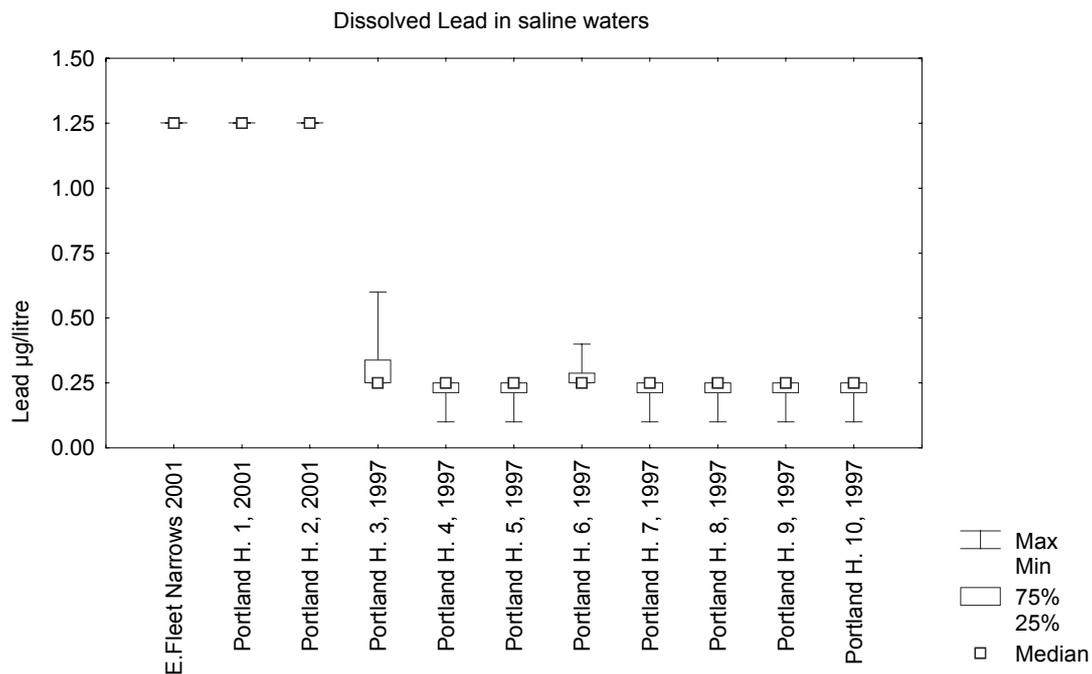
Figure 15. Concentrations of dissolved Pb ($\mu g\ l^{-1}$) in tidal waters, Fleet and Portland Harbour. Data source EA.

Little can be gleaned from this data regarding sources or spatial trends because median values for all sites are essentially equivalent to (half) detection limits, throughout the area (95% of all values were <DL). East Fleet, Portland 1 and 2 medians (for 2001) appear higher than other sites (sampled in 1997) purely because detection limits have increased in the intervening period. All calculated medians are lower than the EQS (25 $\mu g\ l^{-1}$) by more than an order of magnitude.

There is no indication that dissolved Pb concentrations in the Fleet, or adjacent waters, pose a threat for biota.

There are no obvious temporal trends for Pb in tidal waters.

The EQS for Pb in fresh waters (salmonids) ranges between 4 and 20 μg l^{-1}, depending on hardness.

Dissolved Pb concentrations have not been measured in any of the streams or tributaries entering the Fleet. Summary statistics for the River Brit (figure 8) suggest that occasional values are slightly above the lowest EQS for dissolved Pb, though the average is considerably lower - more than half the values (87%) are below detection limits. The River Brit is not likely to be a significant source of Pb for the Chesil Beach site.

The only discharge data for dissolved Pb are for Weymouth STW. The annual average concentration in 1997 was 4.7 ± 4.6 μg l^{-1}. An estimated 24 kg were released to controlled waters in 2001 (annex 7). However, in view of the low concentrations observed in tidal waters of the Fleet it is unlikely that point source discharges or rivers are significant sources of Pb to the lagoon part of the cSAC/SPA.

Zinc

Summary statistics for dissolved Zn concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 16. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison. Approximately 60% of values were below detection limits.
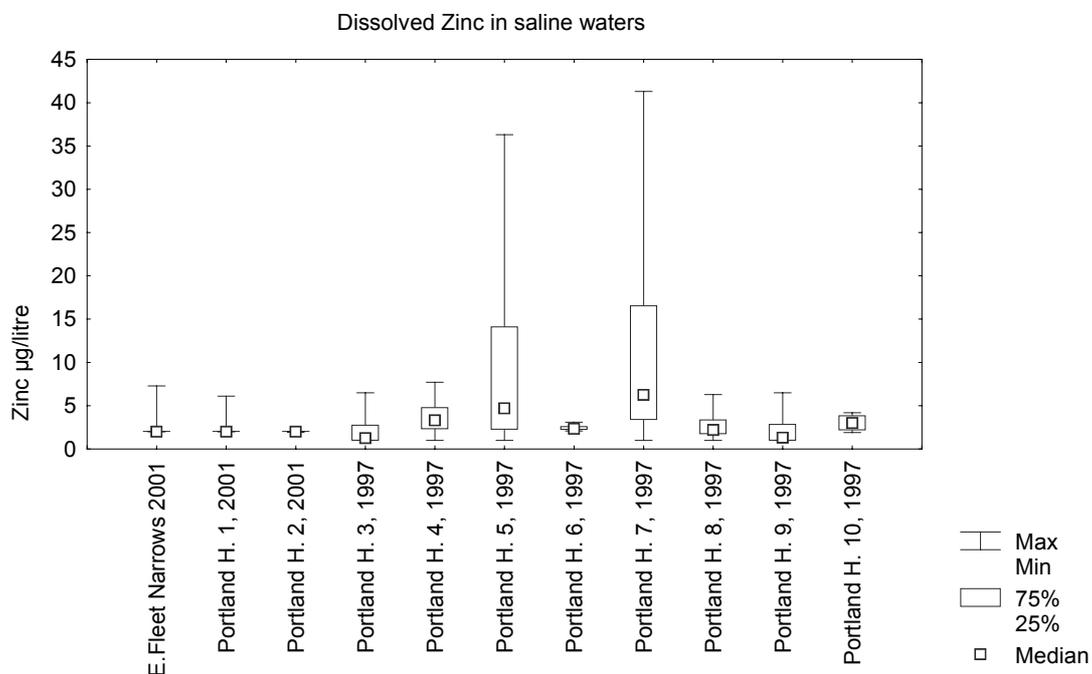

Figure 16. Concentrations of dissolved Zn (μg l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.

All calculated medians are lower than the existing EQS of 40 µg l^{-1} total Zn (and a proposed revision to 10µg l^{-1})[1], though, occasionally, individual elevated values (up to 41µg l^{-1}) are encountered in Portland Harbour. Sources may include run-off, STW inputs and antifouling measures (paints and sacrificial anodes). Nevertheless, dissolved Zn concentrations in the Fleet show no signs of enrichment and are considered unlikely to pose a threat for biota. There are no obvious temporal trends for Zn in tidal waters.

The EQS for Zn in fresh waters (salmonids) is thought to relate to 'total' rather than dissolved metal and ranges from 8 to 120 µg l^{-1} depending on hardness. However, dissolved Zn concentrations have not been measured in any of the streams or tributaries entering the Fleet. The only discharge data for dissolved Zn are for Weymouth STW. The annual average concentration in 1997 was 105 ± 65 µg l^{-1}. An estimated 170 kg were released to controlled waters in 2001 (annex 7). Thus, as for Cu and a number of other elements, there is a small input of Zn discharged regionally into the sea from Weymouth STW. However, in view of the relatively low concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges are significant sources of Zn to the lagoon part of the cSAC/SPA.

Mercury

Summary statistics for dissolved Hg concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 17. This figure also shows data for sites in Portland Harbour, including shellfish directive sites (=Portland 1 and 2), for comparison. Almost all (98%) of these values are below detection limits, therefore no details regarding sources or spatial trends can be provided.
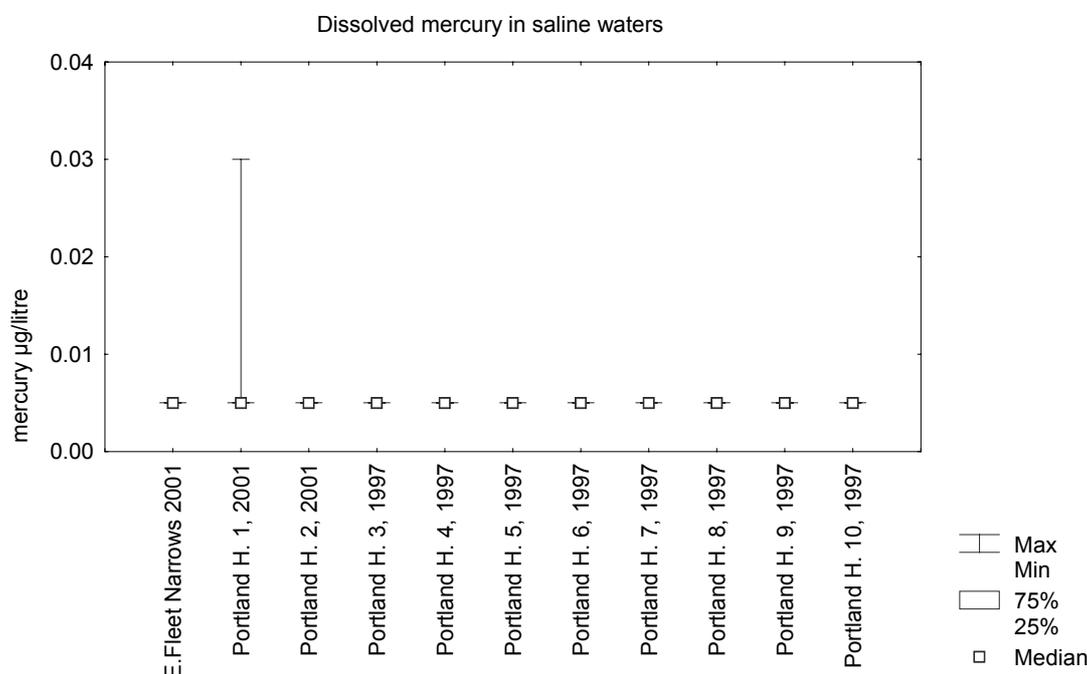

Figure 17. Concentrations of dissolved Hg (µg l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.

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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}).

All median values, and the maximum recorded concentration (0.03 µg l^{-1}), are lower than the EQS (0.3 µg l^{-1} total Hg) and therefore Hg is unlikely to pose a threat for biota in the Fleet. There are no obvious temporal trends for Hg in tidal waters.

The EQS for Hg in fresh waters, is 1µg l^{-1} (total). However, Hg concentrations have not been measured in any of the streams or tributaries entering the Fleet. The only discharge data for Hg (dissolved) are for Weymouth STW. The annual average concentration in 1997 was 0.03 ± 0.04 µg l^{-1}. An estimated 1.1 kg were released to controlled waters in 2001 (annex 7).

In view of the low concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges represent significant sources of Hg to the lagoon component of the cSAC/SPA.

In general, therefore, water quality is not compromised to any great extent by metals.

Sediments

The catchment of the Fleet is not generally considered to be heavily mineralised and deposition of particulates of terrestrial origin would not be expected to result in particularly high metal concentrations in lagoon sediments. Unfortunately, there appears to have been only one limited regional survey of metals in Fleet sediments, published by Nunny and Smith (1995). Table 6 compares average sediment-metal data from the Fleet, Portland Harbour and West Bay Harbour (Nunny and Smith, 1995) with that from other estuaries in the south-west, including the highly contaminated Restronguet Creek (Fal), Poole Harbour, the Severn, and the relatively unpolluted Exe and Avon Estuaries in Devon.

Table 6. Metals in intertidal sediments (µg g^{-1} dry wt): average values for the Fleet, Portland and West Bay Harbours, and other sites in south west England.

site	Cu	Zn	Pb	Cd	Fe	As	Hg	Ni	Cr	Ref
Fleet	32	104	15	nd	26683	15	0.06	23	27	1
Portland Harbour	21	88	43	nd	27560	11	0.17	32	27	1
West Bay Harbour	14	54	<9	nd	12500	7	<0.05	10	13	1
Exe (mid)	38	192	73	0.9	34400	12	0.18	36	44	2
Poole	50	165	96	1.85	29290	14.1	0.81	26	49	3
Tamar	330	452	235	0.96	35124	93	0.83	44	47	3
Severn	35	242	84	0.63	26805	8.4	0.44	33	55	2
Restronguet	2398	2821	341	1.53	49071	1740	0.46	58	32	3
Avon (Devon)	19	98	39	0.3	19400	13	0.12	23	28	2

Refs: [1]Nunny and Smith, 1995 (<90µm fraction); [2] own unpublished data and [3] Bryan and Langston, 1992 (<100µm fraction).nd- not detectable (<1.7 µg g^{-1} dry wt)

Nunny and Smith considered their regional estuarine sediment samples as being elevated in metals due to the influence of urban run-off and sewage (relative to those from off-shore in Lyme Bay). However, as indicated in table 6, for geologically common metals such as As, Cu, Pb, and Zn, concentrations in Fleet and Portland/West Bay sediments pale in comparison with those derived from mining sources (Restronguet Creek). Indeed all metals in the region compare favourably with sediments from both the Exe and Devon Avon (considered to be largely unaffected by metal inputs). It is interesting to note, however, that there is some minor enrichment

with Cu and Zn in sediments from the Fleet (principally at Ferrybridge at the mouth; see figures 53 and 54, section 7) relative to Portland and West Bay sediments.

Further discussion of the distribution and concentrations of metals is presented in section 7.1 in relation to ecotoxicological guidelines. These generally indicate there is likely to be little impact from metals in the European marine site. However, this should be verified by further sampling of Fleet sediments, particularly with a view to identifying sources of Cu, Zn and Pb anomalies and characterising bioavailability.

Biota

Although measurements of metals in sediments and water are a useful guide to contamination, ultimately it is the impact on biota which is of most concern. Unfortunately, published data on metal levels in the region are few. There are EA entries in WIMS of unspecified shellfish collected in Portland Harbour but these do not contain records for metals, presumably because there is no requirement under the Shellfish Hygiene Directive to measure metals in shellfish flesh. An internal regional report on shellfish monitoring between 1985 and 1992 indicates metals in mussels from Portland as being generally below expected/guideline values (Ponting, J., Blandford Biology Unit, 1993). The CEFAS database contains one sample of pacific oysters collected from the East Fleet in 1995 (Annex 8). Concentrations of Cd and Hg in these oysters were below any of the upper guidelines proposed by OSPAR and ADRIS. Several metals (As, Cu, Pb, Ni, Zn) would classify as 'high' by NOAA shellfish criteria, and also by comparison to our own data for oysters from Poole (Annex 8). However oysters are exceptional accumulators of some metals, particularly Cu and Zn and it is not known whether these earlier Fleet data are an accurate reflection of the current shellfish status.

Analyses of scallops and mussels from Lyme Bay off West Bay, suggest that metal concentrations here are 'normal' for these species (Nunny and Smith, 1995). However, inter-tidal and sub-littoral surveys conducted by EA in 1996 – partly to assess the bioavailability of contaminants in consented industrial discharges[1] associated with the Weymouth/Portland long sea outfall (figure 6) - points to localised bioaccumulation of metals as a result of the discharge.

In this EA study, limpets *Patella* spp and seaweed *Fucus serratus* were collected from seven intertidal sites along the east coast of Portland as part of a wider survey of South Wessex coastal sites (Coastwatch, 1996). Results are shown in tables 7 and 8. Samples of *Fucus* from the Portland coast contained higher levels of Hg (up to 0.025 mg kg^{-1} dw), Cu (6.2 mg kg^{-1}), Cr (1.8 mg kg^{-1}) and Ni (11.5 mg kg^{-1}) - and limpets higher levels of As (24.1 mg kg^{-1}) - than other locations in the survey.

Sub-tidal samples (scallops *Pecten maximus* and *Chlamys opercularis*; whelks *Buccinum undatum*; crabs *Maja squinado* and *Cancer pagurus*) from either side of the Weymouth outfall, along a transect parallel to the coastline, revealed that highest levels of Hg, Cu, As, Zn, Mn, Sn and Co tend to occur in organisms closest the discharge. However unequivocal correlations with the outfall are difficult to establish in these off-shore samples due to the small and irregular numbers involved.

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[1] The two largest trade discharges are a photochemical plating/anodising company and an aluminium foil processor. Details of these and other trade effluents are listed in annex 9 (data for 1996).

Table 7. Metal concentrations in Lyme Bay 'shellfish' (EA data, 1996) in relation to guideline values. Numbers in parenthesis relate to crustaceans, ADRIS guidelines are for unspecified shellfish, other values are for molluscs

mg kg^{-1}(dry wt.)

	Patella spp Portland W inter-tidal	scallops (Pecten and Chlamys) Lyme Bay sub-tidal	*Buccinum* Lyme Bay sub-tidal	*Maja* Lyme Bay sub-tidal	EC[1]	JMP[2] upper	ADRIS guide	NOAA 'high'
As	19-24	12.3-15.4	28.4-163	55-77				17
Cd	2.2-3.7	3.9-19.8	9-11.2	1.8-2.6	5*	5 (<1.5*)	15	6.2
Cr	<1-2.9	<1-1.9	<1	<1				
Cu	9.7-19	10-25.3	81.5-759	109-207		<25*(<150*)		12
Pb	1.1-2.1	1.1-5.7	<1-1	<1	7.5*	<20*(<5*)	50	4.8
Hg	0.04-0.08	0.1-0.19	0.17-0.63	0.32-0.37		1(1.5*)	3	0.23
Zn	63-72	168-434	532-1126	154-218		<500*(<500*)	500	200

[1]limits adopted from recently amended EC regulations (446/2001); [2] cited in MAFF (1995); *converted to dw by assuming a wet:dry weight ratio of 5. '<' signifies expected uncontaminated values.

Concentrations in shellfish were mostly below the upper category guidelines proposed by EC, JMP (OSPAR), ADRIS or NOAA (table 7). There were exceptions notably for *Buccinum* which is an exceptional accumulator of some metals, particularly Cu (guidelines for gastropods may be higher than the generalised value shown). Cd levels were also consistently elevated in most species analysed.

The overall impression given by these bioaccumulation data are that organisms are influenced by the Weymouth outfall, as might be anticipated from early hydrographic calculations which showed that much of West Bay, including the intertidal zone, could be subject to dilute sewage (WRc, 1989). However, more rigorous sampling is needed for confirmation of the extent of impact. Comparison with other regional data suggests that enrichment as a result of the Weymouth discharge is probably only moderate for a number of metals, relative to that at more heavily contaminated UK sites. The comparison with *Fucus* in Plymouth Sound estuaries, in table 8, for example, shows that Cu, Zn, Fe and Pb are substantially higher in the Tamar, Lynher and Tavy which are impacted by the legacy of metal-mining activity.

Table 8. *Fucus spp**. Metals concentrations (µg g^{-1} dry weight) in samples from Portland (EA) and Plymouth Sound Estuaries (own, unpublished data)

	Cd	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Portland Min	1.53	0.95	<1	2.9	97	55.9	6	<1	66.8	0.016	42.2	0.58
Portland Max	2.39	5.13	1.8	6.2	164	96.9	11.5	<1	120	0.025	53	6.99
Tamar	1.57	5.23	2.52	74.8	1631	425	5.17	37.4	318	0.11	53.3	0.82
Lynher	2.38	3.06	28.4	93.0	3073	430	3.97	33.3	715	0.06	39.8	1.08
Tavy	1.77	9.69	3.33	65.7	1232	314	6.7	18.3	379	0.02	59.9	1.37
Plym	4.69	7.39	4.34	25.0	1188	253	6.98	20.4	395	ns	ns	ns
Yealm	0.92	4.25	8.2	11.4	730	306	6.63	6.35	139	0.1	37.7	0.85

* Portland samples are *Fucus serratus*, others are *Fucus vesiculosus*; both species are probably relatively good indicators of the bioavailability of dissolved metals. ns – not sampled.

Studies specifically designed to evaluate the issue of bioavailability in the Fleet, using appropriate bioindicators, have not been undertaken. Relatively low levels of bioaccumulation would be expected for the Fleet, since no obvious direct metal inputs have been established. However this needs to be confirmed by a well-designed survey of biota at the site.

The distribution of biological indicator species (e.g. *Nereis diversicolor*, *Scrobicularia plana*, *Cerastoderma edule*, *Littorina* spp. *Fucus* spp,) and the bioavailability and impact of metals in UK estuaries, 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). It would be useful to extend this database to include the Fleet in order to provide comparisons of the status of the site with regard to metals.

Studies of polychaetes and infaunal bivalves would probably be most informative. 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 likely to be relatively abundant throughout most of the Fleet.

The deposit-feeding clams *Scrobicularia plana*, *Macoma balthica and Abra tenuis* are also potentially valuable bioindicators, particularly in terms of understanding trends in sediment metal bioavailability. They are in many respects better accumulators and indicators of metals than *Nereis* (with the exception of Cu), and have the advantage of not regulating Zn. Their ability to survive in upper estuaries is presumed due to the buffering influence of burial behaviour in sediments and an ability to isolate themselves, through shell closure, from extreme low salinities. These characteristics may be valuable for monitoring extensively in the Fleet.

Both clams and worms have been useful in identifying trends in metal bioavailability in the other European marine sites of the south west (see other reports in this series on the Severn, Fal and Helford, Exe, Plymouth Sound and estuaries, and Poole Harbour). Both taxa are also important representatives of food items taken by birds and fish and would therefore provide a perspective as to the likely importance of dietary pathways for contaminants.

6.1.2   TBT and other organotins

Use of tributyltin (TBT) antifouling paint on boats less than 25m in length was prohibited in 1987, though larger vessels (essentially 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. Since there is little boating activity in the Fleet the direct impact on the lagoon would be anticipated to be relatively small, though there is the possibility of some ingress of TBT through the entrance, from Portland Harbour.

Presumed analytical difficulties make interpretation of TBT results in the EA data set problematic. Several sites in Portland Harbour have been monitored for TBT, though out of 66 water samples taken between 1997 and 2001 only nine were above limits of detection, which have varied from 2-68ng l^{-1} (tributyltin cation); the coastal water EQS is 2 ng l^{-1}, thus detection limits were sometimes above the EQS. TBT concentrations in the nine 'positives' ranged from 5-52 ng l^{-1}. Since the standard is a maximum allowable concentration, the results indicate occasional exceedence.

Summary statistics for 2001 are plotted in figure 18 by assigning half detection limits to 'less-than values' (DL=4 ng l^{-1} in 2001). Annual averages (and max) at six of these sites then becomes 2ng l^{-1}, whilst at two of the sites close to Portland Docks[1] maximum values of 5 and 7 ng l^{-1} were recorded. As these are above the EQS they may impinge to some extent on the Fleet, though probably only close to the entrance. Sensitive mollusc species would, theoretically, be the principal organisms affected. Unfortunately there are no data on TBT levels in the Fleet itself, or on associated biological effects. No reports could be found concerning TBT-related effects on shell growth in the pacific oyster stocks in the Fleet.

The source of TBT in the region is probably partly from antifouling paints on boat hulls although a small input from STW cannot be ruled out. Concentrations up to 76ng l^{-1} TBT (as cation) were measured in samples of effluent from Weymouth STW in 1996. Likewise at West Bay Pumping station one sample contained 161 ng l^{-1} (whilst two others were recorded as <24 and <105 ng l^{-1}) implying an occasional input from sewage. Triphenyl tin (TPT) in the same samples was below the limit of detection (<13 to <22 ng l^{-1}) and is probably of little concern; however; there are no data for TPT in coastal waters of the region. The EQS for TPT is 8 ng l^{-1} in sea water.
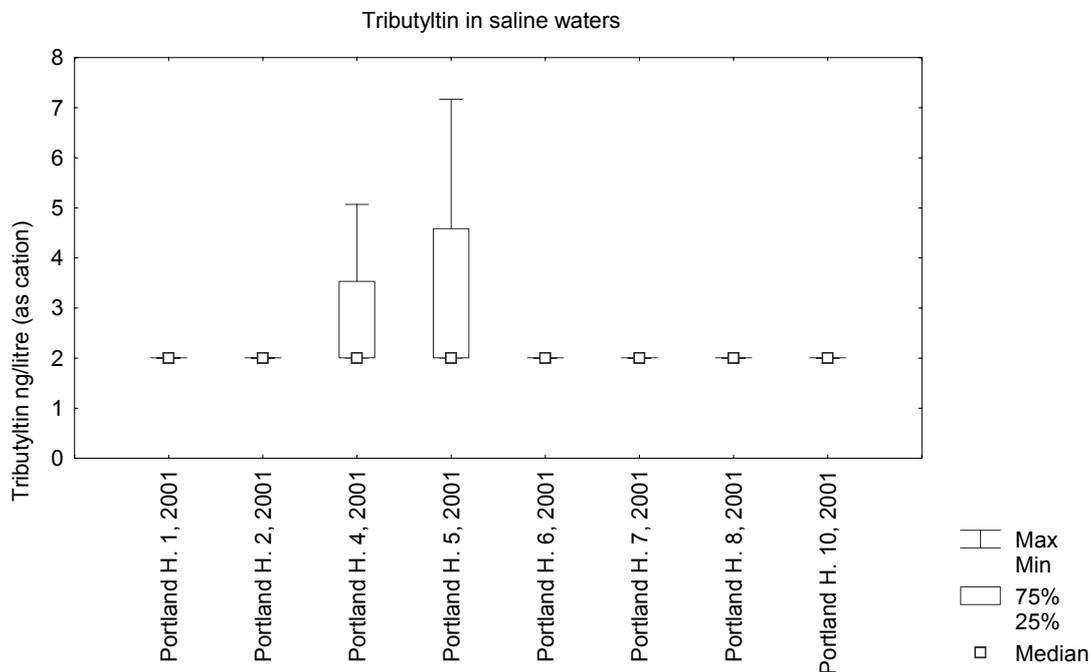

Figure 18. TBT in water (ng l^{-1} as tributyltin cation), Portland Harbour. Data souce:EA

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[1] Portland was a naval base until 1996. Since then a there may have been a shift towards increased commercial activities

TBT concentrations in sediment samples from Portland Harbour, were without exception below detection limits (8μg kg^{-1} in 2001, an order of magnitude higher in 1997). There are no statutory sediment guidelines for TBT, though OSPAR (2000) has set a provisional ecotoxicological guideline value of 0.005-0.05μg kg^{-1} – below the detection limits described here. Clearly, a more detailed survey is needed to establish current status and trends for TBT in sediments, including the Fleet.

The TBT indicator *Nucella lapillus* (dog whelk) was sampled in 1992 and 1998 at Portland Bill and West Bay (Harding *et al*., 1992; Minchin *et al.,* 1999). Imposex-related phenomena were observed on each occasion though the trends have differed between the two sites. At West Bay (200m to the west of the harbour - which is frequented by a small fleet of fishing boats and leisure craft) there were clear signs of improvement between the two surveys: average VDSI (vas deferens stage index) decreased from 3.97 to 3, RPSI (relative penis size index) decreased from 11.15% to 0.42, and the overall classification of the site improved from category C to B[1]. In contrast, at Portland Bill (outside Portland Harbour – which houses larger commercial and, until recently, naval vessels as well as the leisure fleet) the dog whelk population has been eliminated. In 1992 the animals from this site (also category C) had a VDSI of 4, an RPSI of 12.5, but, unlike the West Bay population, contained detectable levels of tin in tissues. Presumably levels of TBT measured in Portland Harbour have been high enough at times to sustain the imposex phenomenon, as indicated from EA data above, so preventing recovery.

Littorina littorea was analysed for TBT at three Portland sites in 2001(EA). Concentrations ranged from 10 μg TBT kg^{-1} (dw) just outside Portland Harbour (Nothe) to 102 μg TBT kg^{-1} near Portland Docks. Though ten-fold higher than the Nothe 'controls', body burdens in the Harbour are not thought to be high enough to initiate malformations in female reproductive structures (intersex in *Littorina* is a less sensitive index than imposex in than *Nucella*). TBT residues would be just below the 'high value' which categorises the top 15% of values seen in NOAA's Mussel Watch programme in the USA. Nevertheless, as the OSPAR ecotoxicological assessment criteria for TBT in molluscs lies in the range 1-10 μg kg^{-1} (OSPAR 2000), values in Portland gastropods indicate that further monitoring of TBT, including bioavailability, should be considered as a possible candidate for priority action in the future.

6.1.3   Hydrocarbons (Oil, Petrochemicals, PAHs)

Oil

Oil pollution is a continual threat to all inshore marine habitats, and is particularly pronounced in lagoons due to their enclosed nature (restricted flushing). Risks in the Fleet probably include small leaks, spills and discharges, although there is always the possibility of a major accident. The narrow mouth would make it relatively easy to

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[1] Imposex classification as used by Harding *et al,* 1992; Minchin *et al* 1999. Category A, VDSI = 0-1.99; category B, VDSI = 2-3.99; category C, VDSI = 4-4.99; category D, VDSI = 5+. At all sites where females have VDSI of 4 or above, sterilisation of females would be initiated, leading to extinction of the population in severe cases.

deploy booms and protect the lagoon from spillages in Portland Harbour. This would not be the case for the coastal strip along Chesil Beach.

There are a number of ways in which oil could potentially impact on the interest features of the Fleet. 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 (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 subsequently, 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 MFO (mixed function oxidases) 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 affected. 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 Fleet, 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 and the atmosphere (PAH).

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 (*see* annex 4), 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 (*see* annex 3), 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 for 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 freshwater, discharges, tidal waters, sediments or shellfish in and adjacent to the cSAC/SPA. The survey of Lyme Bay in 1994 by Nunny and Smith (1995) included one water sample from Portland Harbour which contained 6.08 µg l^{-1} total organic extract (TOE) comprising an unresolved complex mixture (UCM) of 5.3 µg l^{-1}. This feature is thought to represent residual petroleum after biodegradation, possibly from chronic oil pollution as a result of general shipping activity or terrestrial run-off. However the levels of TOE seen in these samples are typical of waters from coastal areas with low levels of petrogenic contamination and are probably of little cause for concern in relation to the ecotoxicological thresholds described above.

There was no obvious spatial distribution of the contamination in the waters of Lyme Bay. In contrast, sediments showed marked heterogeneity with an enrichment of petrogenic sources in the east of the area. Sediments in eastern Lyme Bay contain total hydrocarbons (THC) in the range 0.56 - 16.6 µg g^{-1}, with a general trend of increasing levels eastwards. Higher concentrations were found in Portland Harbour (200 µg g^{-1}) and the Fleet lagoon (two sites with 110 µg g^{-1}). The THC in Portland Harbour comprises mainly UCM, indicative of biodegraded inputs from fuel or lubricating oils, probably from shipping activity. Thus, the higher THC concentrations in this general area are due to increased UCM content, consistent with

contamination dispersing from the harbour areas or from runoff from Weymouth (Nunny and Smith, 1995).

The total hydrocarbon concentration in the CEFAS sample of pacific oysters from the East Fleet in 1995 was 5.2 µg g^{-1} wet weight (equivalent to ~130µg g^{-1} dry weight) - comparable to the level in mussels from the Shambles, off Portland Bill (19.5 µg g^{-1} wet weight, ~121µg g^{-1} dry weight). This degree of contamination appears to be unexceptional and is, perhaps, typical of chronically polluted environments.

Based on this limited data set, it is difficult to comment with certainty on the environmental quality status of the European marine site with regard to total hydrocarbons, or their impact on estuarine biota and shellfish beds. Although the perceived threat to the Fleet is probably low it would seem important to establish more extensive baseline information as a means of monitoring future improvement/deterioration at this site.

PAHs

PAHs 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). A major source is road run-off and most of this pollutant load enters waterways from storm water overflows by means of adsorption to particulates. Thus, 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 (Law *et al*., 1997). PAH concentrations in sediments may be persistent (particularly where tidal action is restricted, and degradation limited by anoxia, as in the Fleet). In addition, some PAHs have been identified as endocrine disruptors (Anderson *et al*., 1996a,b; Kocan *et al.,* 1996).

One general concern over a number of PAHs, particularly those of high molecular weight (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. Elevated levels have been linked to liver neoplasms and other abnormalities in demersal fish (Malins *et al.,* 1988). However, the risk of carcinomas is probably low at environmental concentrations of 1µg l^{-1} or less (Payne *et al*., 1988; Law, *et al.,* 1997; Cole *et al*., 1999).

The only information found for either total PAHs or individual PAHs in water, sediments or biota of the region come from the Lyme Bay study by Nunny and Smith (1995). PAH concentrations in all sea water samples, including Portland Harbour, were below the limit of detection (~0.04 ng l^{-1}) and *anticipated* to be of relatively low biological significance in the Fleet. Nevertheless, baselines in waters of the Fleet need to be established and concentrations placed in context with various guidelines such as those proposed by OSPARCOM (1994).

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 than in water. The general pattern of PAH distribution in the Lyme Bay region was consistent with the THC data – highest concentrations were associated with Portland Harbour muds (3.9 $\mu g\ g^{-1}$) and sediment of the Fleet lagoon (2.3 - 3.2 $\mu g\ g^{-1}$, see table 9). In contrast PAH concentrations were low in the area west of Portland Bill (0.14 - 0.3 $\mu g\ g^{-1}$) and slightly higher in Weymouth Bay (0.2 - 0.54 $\mu g\ g^{-1}$). Elsewhere in the UK, PAH concentrations in sediments range from undetectable at offshore sites (such as the central English Channel) to 43.5$\mu g\ g^{-1}$ dry weight on the River Tyne at Hebburn (MPMMG, 1998; Woodhead *et al*. 1999).

Table 9. Individual and ΣPAHs in Fleet sediments[†] and guideline values and probable effects levels*

PAH concentrations and thresholds in sediment ($\mu g\ kg^{-1}$ dry wet)

	Fleet (west) site 1C[†]	Fleet (east) site 4C[†]	Threshold effect level* TEL	Probable effect level* PEL
Naphthalene	50	66	34.6	391
Phenanthrene	225	220	86.7	544
Fluoranthene	710	670	113	1494
Benzaanthracene	450	320	74.8	693
Benzofluoranthenes	1100	720		
Total PAH	3200	2300	1684	16770

[†]Fleet data from Nunny and Smith (1995). Site locations in Dyrynda and Cleator (1995)
 *Threshold Effect Level and Probable Effects Level (ΣPAH) proposed by Macdonald *et al*., (1996), Thresholds for individual PAHs from CCME (1999).

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 1-3$\mu g\ g^{-1}$ dry weight, and possibly lower (Payne *et al*., 1988; Woodhead *et al*., 1999).

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 1-6 µg kg^{-1}dry weight (Alden and Butt, 1987; Ankley *et al.,* 1994). On this basis sub-lethal effects caused by PAHs in sediments of Portland and the Fleet cannot be ruled out (see data on Fleet sediments, guideline values and probable effects levels for individual and ΣPAHs in sediments in table 9. Maps showing the distribution and concentrations of PAHs in relation to ecotoxicological guideline values are also presented in section 7.2). These data from Nunny and Smith (1995) indicate that values in the Fleet are above the ISQG but below the PEL. Any effects in the European marine site, if they occur, are likely to be chronic rather than acute. However, this should be verified by further sampling of Fleet sediments, particularly with a view to identifying sources and characterising bioavailability.

There are two potential sources of PAHs in the marine environment, petrogenic (two-to-three ring aromatic compounds e.g. naphthalenes, phenanthrenes and dibenzothiophenes) and pyrogenic (four-to-six ring compounds e.g. benzfluoranthrenes, benzpyrenes associated with fallout from incomplete combustion of fossil fuels and often concentrated in urban runoff). In near-coast areas in the vicinity of Weymouth, including the Fleet, the latter dominate (table 9). Interestingly the contribution of petrogenic sources increases in the western and central areas of eastern Lyme Bay leading to speculation that there may be seabed seepages of oil or gas offshore (Nunny and Smith, 1995).

Exposure pathways to most aquatic organisms 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 likely 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 Mixed function oxidase (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 the Fleet in view of its use as shellfish water. To date measurement of PAHs has been limited to the one batch of *Crassostrea gigas* from the East Fleet sampled in 1995 by CEFAS (table 10).

Table 10. PAHs in *Crassostrea gigas* from the Fleet and *Mytilus edulis* from The Shambles, off Portland Bill (Data source CEFAS). Brownsea (Poole) mussels, and OSPAR ecotoxicological assessment criteria for mussels are shown for comparison

	PAHs in shellfish µg kg^{-1}							
	oysters East Fleet 1995		mussels shambles 1996		mussels Brownsea 1999-2001		OSPAR criteria (µg kg^{-1}dry wt)	
	wet wt	dry wt	wet wt	dry wt	wet wt	dry wt	low	high
Anthracene	0.2	5	0.6	3.7	1.8	11.3	5 - 50	
Benz-[A]-Anthracene	0.5	12.5	4.3	26.8	7.1	44.4		
Benzo-[A]-Pyrene	0.2	5			1.1	6.9	5000-50000	
Benzo-[E]-Pyrene	0.8	20	5	31.2				
Benzo-[GHI]-Perylene			1.2	7.5	3.2	20.0		
Benzo-[B]-Fluoranthene	2.1	52.5	8.7	54.3				
Chrysene	2.0	50	6.5	40.6	16.1	100		
Fluoranthene	6.2	155	19	1185	22.2	138	1000-10000	
Indeno (1,2,3-CD)-Pyrene			1.5	9.37	1.6	10.0		
Naphthalene	1.1	27.5	1.2	7.5	79.8	498	500-5000	
Phenanthrene	1.1	27.5	7.5	46.8	10.5	65.6	5000-50000	
Pyrene	5.5	137	12	75	21.3	133	1000-10000	
Total PAH	**40**	**1000**	**196**	**1225**	**164.6**	**1028**		

* data converted to dw using a wet:dry weight ratio of 25 for oysters (reported solids content 4%) and 6.25 for mussels (16% solids)

It is difficult to put these results in context as there are no UK standards for PAHs in shellfish. 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, based on human health concerns. The Fleet oyster samples would meet these recommendations and also OSPAR guidelines for individual PAHs, which are based on ecotoxicological considerations (table 10).

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. Assuming a wet:dry weight ratio of 25 (from the reported solids content of 4%) the Fleet oysters fall just below this guideline, and mussels from The Shambles, off Portland (wet:dry ratio of 6.25), marginally above. Although there appear to be no immediate concerns for the Fleet shellfish, based on this evidence, it would be useful to extend the data for biota to obtain a broader picture of PAH bioavailability in, and adjacent to, the European marine site.

There has been some monitoring of hydrocarbons and PAH in offshore shellfish (mussels and queen scallops) from Lyme Bay (off W.Bay). The THC concentrations found were in the range 2.1 to 8.2 mg kg^{-1} wet weight (expressed as Ekofisk crude oil equivalent), indicating a low level of hydrocarbon contamination offshore. The majority of PAHs were below limits of detection (Nunny and Smith, 1995).

6.1.4 Pesticides, Herbicides, PCBs, volatile organic compounds.

This section reviews data for a range of synthetic organic compounds, and describes their potential involvement in endocrine disruption.

CEFAS monitors levels of selected pesticides and other possible endocrine disrupting chemicals in bivalves from UK designated shellfish waters. A summary of analyses between 1995 and 1996 implies that pesticide body burdens in most shellfish waters in England - including, Portland Harbour and the Fleet - were close to or below detection limits. The concentrations of α,γ HCH, dieldrin, HCB, ppDDE, ppTDE and ppDDT in a sample of *Crassostrea gigas* from East Fleet, in 1995, were 0.001mg kg^{-1} wet weight (0.003mg kg^{-1}ΣDDT). The concentration of PCBs (Σ25 congeners) was 0.025 mg kg^{-1}. Concentrations of pesticides and PCBs were considered to be of little toxicological importance in relation to various guidelines (CEFAS, 2001; see Annex 8). Similar conclusions can probably be drawn from EA oyster samples from the Fleet (1991/92), tested for a comparable range of organics, though DLs were higher (0.01 mg kg^{-1} wet wt, except for PCB - 0.05 mg kg^{-1} wet wt). These appear to be the only biological samples from the within the Fleet that have been analysed.

Data for unspecified shellfish species from the Portland Harbour site (1995 and 1997), extracted from WIMS, are summarised in table 11. With the exception of lindane (γ-HCH) concentrations of most compounds are below detection limits, therefore estimates of means are largely based on ½DL values. The data are for samples outside the cSAC/SPA, though the predominance of <DL values suggests, tentatively, that sources of pesticides, herbicides and PCBs within the Harbour are unlikely to pose a threat to the European site.

Table 11. Organic contaminants (µg kg^{-1} wet wt) in unspecified shellfish (mussels?) from shellfish beds in Portland Harbour, close to the Fleet, 1995 and 1997. (EA data)

Compound	Mean µg kg^{-1} (wet wt.)*	Values <detection (%)	No. of values
ALDRIN	1.0	100%	4
DDT (PP)	2.2	100%	4
DDT (OP)	0.25	100%	4
DDE (OP)	0.55	100%	4
DDE (PP)	2.7	100%	4
DIELDRIN	0.95	100%	4
ENDOSULPHAN ALPHA	0.4	100%	4
ENDRIN	0.5	100%	4
HCH GAMMA	2.1	0%	4
HEXACHLOROBENZENE	0.5	100%	4
HEXACHLOROBUTADIENE	0.65	100%	4
ISODRIN	0.25	100%	4
TRIFLURALIN	0.8	100%	4
PCB NO.28	0.7	100%	4
PCB NO.101	1.56	100%	4
PCB NO.118	2.86	100%	4
PCB NO.138	3.16	100%	4
PCB NO.153	1.21	100%	4
PCB NO.180	6.83	100%	4
PCB Total (Σabove isomers)	16.3		

*Mean calculated using ½ DL values, where below detection

It is likely that these unspecified shellfish are *Mytilus* spp, since an earlier internal EA report refers to bioaccumulation monitoring in Portland mussels, under the Shellfish Directive (Ponting, J, Biology Unit Blandford, 1993). For samples collected between 1985 and 1992, most organic compounds analysed (similar to the list below) were below detection limits. However five of these samples contained PCB's at concentrations between 25 and 42 µg kg^{-1} wet weight - above lower Joint Monitoring Programme (JMP) guidelines.

Another survey conducted by EA in 1996, to assess the bioavailability of contaminants associated with the Weymouth/Portland long sea outfall, points to localised bioaccumulation of some organics in sub-tidal samples of scallops, whelks and crabs. In a transect either side of the Weymouth outfall, parallel to the coast, highest concentrations of PCB (congener 153) and OP' DDE tended to occur in organisms closest the discharge. However, as with metals, unequivocal correlations with the outfall are difficult to establish in these off-shore samples due to the small and irregular numbers involved. In the internal EA report on this sampling exercise, PCBs are indicated as exceeding the JMP guideline, though this may be a misinterpretation of units. From the data we have seen, none of the detectable synthetic organics exceeds upper guideline values (table 12). The majority of other synthetic compounds tested were below limits of detection (endosulphan A, aldrin, α-HCH, β-HCH, HCB, dieldrin, heptachlor, OP' DDT, PP' DDT, PP' DDE, OP' TDE, PP' TDE, endrin PCBs 28,52,101,118,138,180, isodrin, trichlorobenzene).

Table 12. Synthetic organic concentrations in Lyme Bay 'shellfish' (EA data, 1996) in relation to guideline values.

	concentrations µg kg^{-1} wet weight				
	scallops (*Pecten* and *Chlamys*)	*Buccinum*	*Maja*	FAO/WHO upper guideline	JMP[1] upper guideline
PCBs	<0.26- 4.2	<0.26- 4.4	3.6-4.2		100 (50)
OP' DDE	<1.3-3.7	<1.3	<1.3	(500*)	
γ-HCH	<0.6-1.2	0.8-1	<0.6	(50*)	

[1]cited in MAFF (1995). Guidelines in parenthesis relate to crustaceans/fish, other values are for molluscs. *fish liver

These limited bioaccumulation data for organics are in agreement with trends for metals burdens which imply that, locally, body burdens of selected compounds are influenced by the Weymouth STW outfall, though more rigorous sampling is needed for confirmation of the extent of impact.

Studies specifically designed to evaluate the issue of bioavailability of organic contaminants in the Fleet, using appropriate bioindicators, have not been undertaken. More extensive, targeted sampling of non-commercial species will be needed to confirm the absence of any significant accumulation of these synthetic compounds, and should incorporate sediments and sediment-dwelling species in the Fleet, such as clams and worms.

The discussion below describes some of the EA observations on organic contaminants in water, and potential threats from these 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 adsorb 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 the European marine site could be found. It is likely that the majority of pesticides in the environment nowadays are present at low levels in the water column, and that sediments, together with biota, form the major reservoirs for these compounds.

Summary statistics for DDT concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 19. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison. However since 99% of values were '< values', these plots are largely a reflection of detection limits and do not give a realistic idea of distributions. Nevertheless, they demonstrate compliance with the estuarine EQS (10 ng/l ppDDT, 25 ng/l for total DDT) by a substantial margin.

There are no obvious temporal trends for DDT in tidal waters.
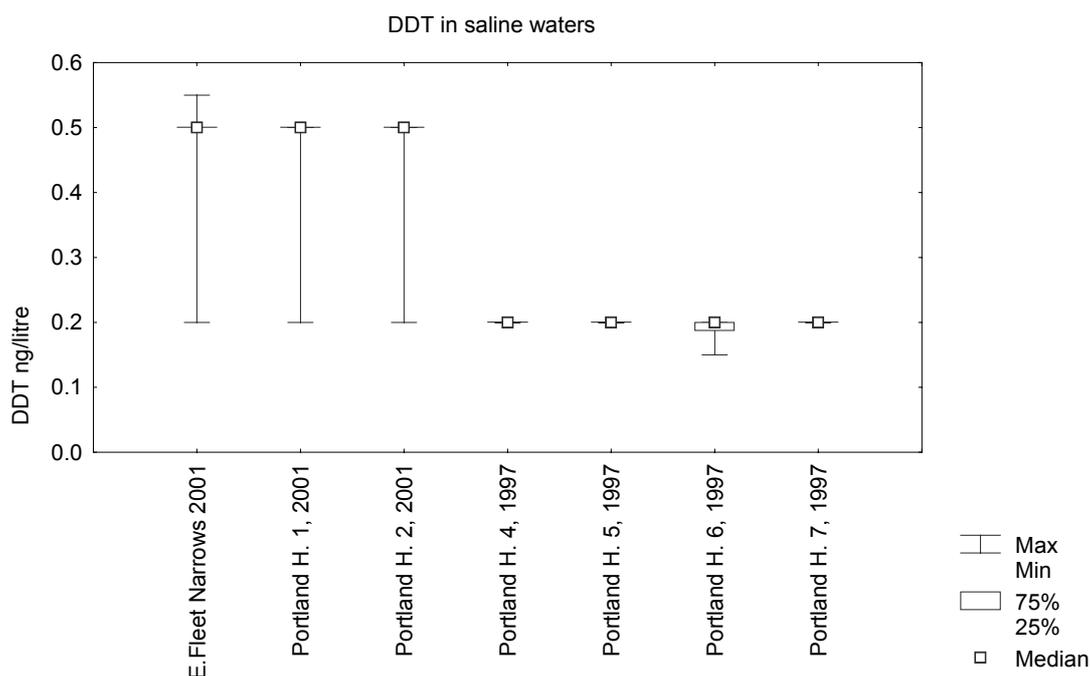
Figure 19. Concentrations of DDT (ng l^{-1}) in tidal waters, Fleet and Portland Harbour. Data source EA.

The EQS for DDT in fresh waters is 10 ng l^{-1}.

DDT concentrations have not been measured in any of the streams or tributaries entering the Fleet.

The only discharge 'data' for DDT are for Weymouth STW and West Bay STW. All records for Weymouth are lower than detection limits with the exception of one sample (8.1ng l^{-1}). The annual average concentration in 1997, based on half detection limit values, was 2.3 ± 2.0 ng l^{-1}. (This applies to both op- and pp-DDT). Similarly at West Bay the majority of values were below detection (Figure 20) and the average concentration, based on half detection limit values, was 2.83 ± 9.96 ng l^{-1}.

In view of the low concentrations observed in tidal waters of the Fleet it is unlikely that rivers or point source discharges are significant sources of DDT for the Fleet or Chesil Beach.

Pesticides in West Bay sewage pumping station
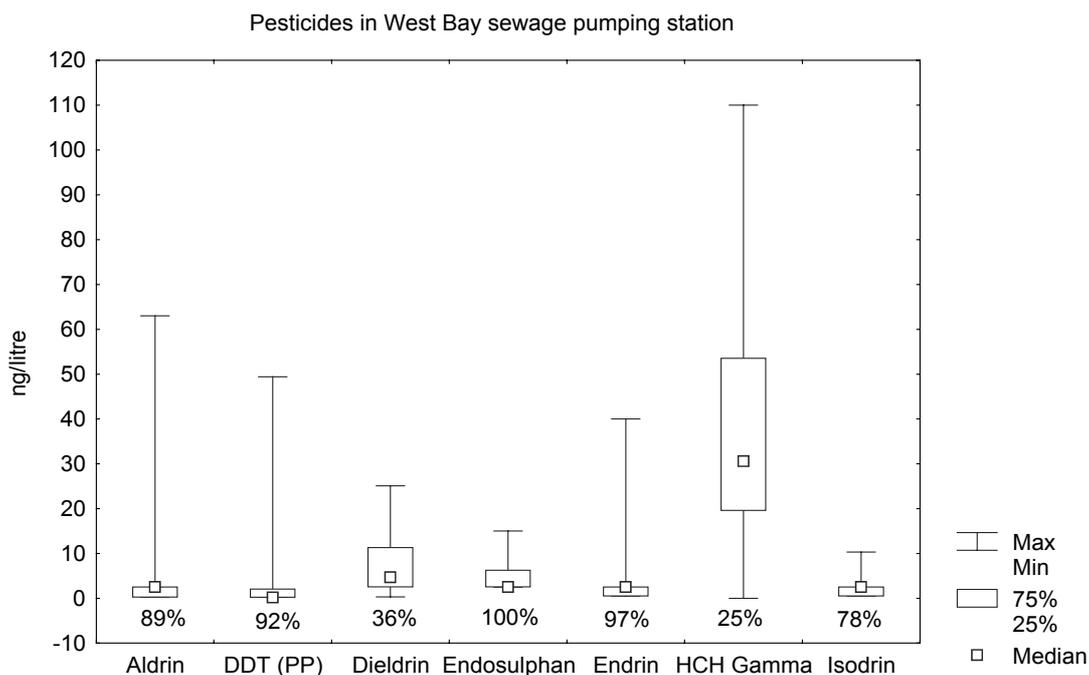
Figure 20.   Summary statistics for concentrations of organochlorine pesticides in effluent from West Bay sewage pumping station during the 1990s. Percentage of values below detection limit are show below each plot (Data Source EA).

The scarce data on DDT in sediments (from EA sources) are discussed in section 7 (see table 24). Even though the sum of DDT and DDE isomers in Portland sediments near to the entrance to the Fleet appear to be above the PEL (probable effects level), this is probably artifactual, due to the fact that the majority of values are below detection limits which are themselves often above the guideline value.

EA 'unspecified' shellfish samples contain low levels of DDT – all below the limits of detection (table 12). Even when different isomers are combined, and assuming half detection limits, and a wet:dry weight ratio of five[1], concentrations would be an order of magnitude below the recommended 'no-effects' guideline of 100 μg kg^{-1} (dw) set by ADRIS (Association of Directors and River Inspectors in Scotland) under the EU Shellfish Waters Directive, to protect shellfish and their larvae (ADRIS, 1982). They are also well below the 'high value' of 140 μg kg^{-1} which categorises the top 15% of values seen in NOAA's Mussel Watch programme in the USA.   This suggests that the sampled Portland 'shellfish' are not at risk from DDT.

Dieldrin and other 'drins'

Dieldrin, along with other 'drins' (aldrin, endrin, isodrin), is another endocrine-disrupting OC pesticide of potential concern in the cSAC/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

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[1] An approximation based on our own data for wet to dry ratios in shellfish

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).

No published papers on levels of dieldrin or other drins in sediments or water of the European marine site could be found.

Summary statistics for dieldrin concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 21. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison. However since all results were '< values', these plots are entirely a reflection of detection limits and do not give a realistic idea of distributions. Nevertheless, they demonstrate compliance with the estuarine EQS (EQS of 10 ng l^{-1} in fresh- and salt-water) by a substantial margin.

Any temporal trends for dieldrin in tidal waters are also masked by changing detection limits.
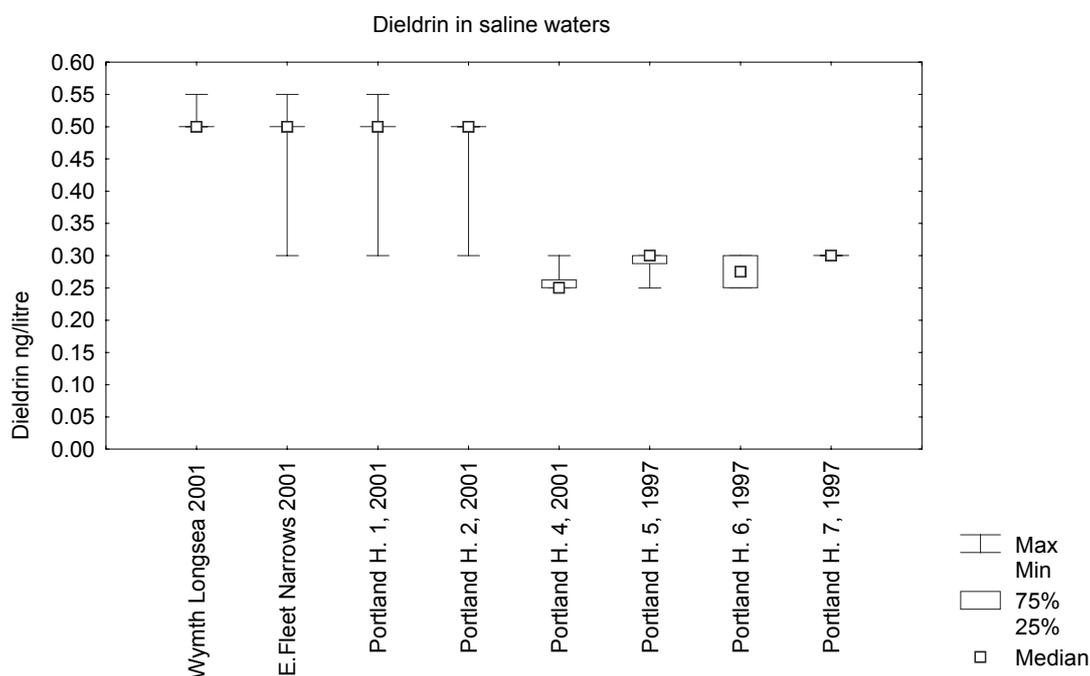

Figure 21. Concentrations of dieldrin (ng l^{-1}) in tidal waters, Fleet and Portland Harbour and near the Weymouth long-sea outfall. Data source EA.

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

Dieldrin concentrations have not been measured in any of the streams or tributaries entering the Fleet.

The only discharge 'data' for dieldrin are for Weymouth STW and West Bay STW. The annual average concentrations in effluent samples from Weymouth during the last decade are shown in figure 22 and indicate an increasing trend up to 1999. Thereafter concentrations have fallen and were below detection limits (5 ng l⁻¹) in 2001-2, presumably as a result of changes to the sewerage system. Seawater at the long sea outfall at Weymouth has been monitored since 2001 and dieldrin is consistently <1 ng/l. At West Bay STW, 36% of values in effluent samples were below detection (figure 20) and the average concentration (± SD), based on half detection limit values, was 7.4 ± 5.7 ng l⁻¹.

In view of the low concentrations observed in tidal waters of the Fleet and adjacent coastal waters it is unlikely that rivers or point source discharges are significant sources of dieldrin for the Fleet or Chesil Beach.
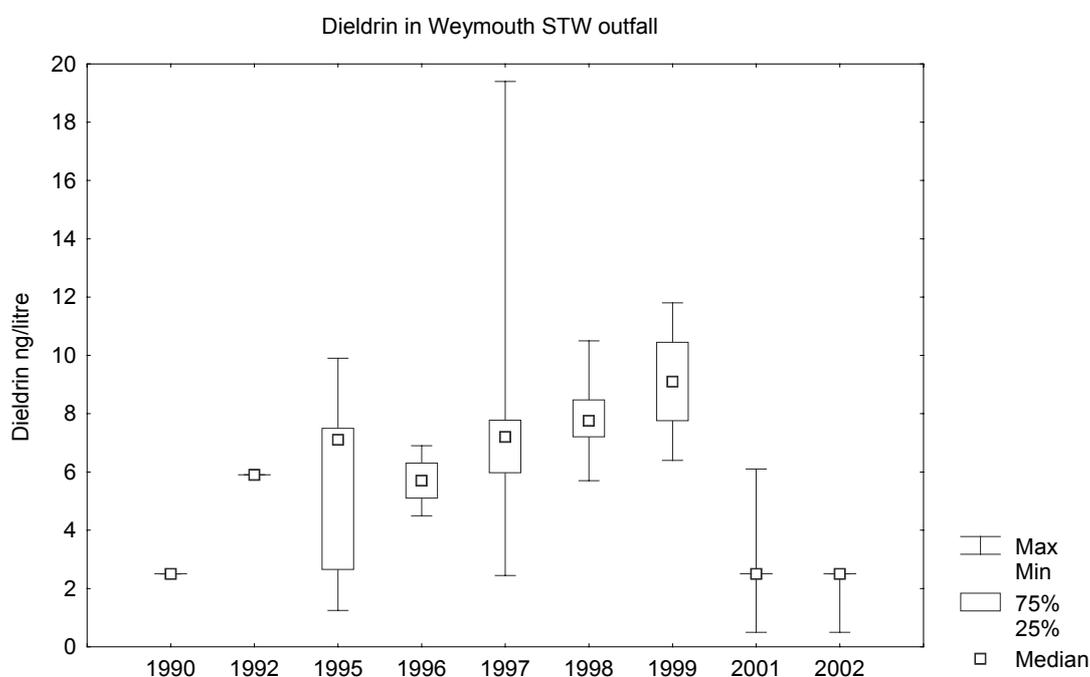
Figure 22. Concentrations of dieldrin (ng l⁻¹) in Weymouth STW discharge. Data source EA.

The scarce data on dieldrin in sediments (from EA sources) are discussed in section 7 (see table 24). Dieldrin concentrations in estuarine sediments may appear to exceed the ISQG guideline (but not the PEL), though again this appears to be largely due to detection limits being above the guideline value (section 7).

EA 'unspecified' shellfish samples from the Portland site near the Fleet contain low levels of dieldrin – all below the limits of detection (results presented on a wet weight basis, assuming half detection limits, in table 11). Adopting a wet:dry ratio of five, the average value for shellfish (~5μg kg⁻¹ dw) is more than an order of magnitude below the recommended 'no-effects' guideline of 100 μg kg⁻¹ (dw) set by ADRIS. It

is also below the 'high value' of 9.1µg kg^{-1} which categorises the top 15% of values seen in NOAA's Mussel Watch programme. This suggests that the sampled 'shellfish' are not at risk from dieldrin.

The pattern of distributions and concentrations of other 'drins' in tidal waters (East Fleet and Portland one and two) is virtually the same as for dieldrin. None of the tidal waters sampled recently had concentrations of endrin, aldrin or isodrin above detection limits (most recently <1 ng l^{-1} for each). This compares with an EQS of 5 ng l^{-1} for endrin and aldrin, respectively and 30ng l^{-1} for total drins.

Concentrations of endrin, aldrin and isodrin in the Weymouth sewage (STW) discharge were below limits of detection (< 3 ng l^{-1}). Thus impact on the marine site is likely to be minimal from this source. Recent monitoring of seawater in the vicinity of the long-sea outfall at Weymouth confirms this (all values <1ng l^{-1}). Concentrations in the West Bay sewage samples were also unlikely to give rise to problems in the receiving environment of West Bay; average values for endrin, aldrin and isodrin in effluent samples were 3.89 ± 10.5, 4.37 ± 7.97 and 2.21 ± 2.24 ng l^{-1}, respectively, with the majority of determinations being below limits of detection (figure 20).

Aldrin, endrin and isodrin concentrations in Portland sediments and shellfish were all below detection limits (and, for endrin in sediments, below the ISQG and PEL guidelines), and imply little cause for concern for the Fleet. Once, again however direct measurements in samples within the designated European marine site are needed to confirm this.

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

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 0.1µg l^{-1} in estuaries and coastal waters of the south west during 1993 (NRA 1995), but the current analysis of EA data suggests this is not applicable in the case of the Fleet.

Summary statistics for γ-HCH concentrations in tidal waters of the East Fleet (the Narrows) are plotted in figure 23. This figure also shows data for sites in Portland Harbour, including shellfish sites (=Portland 1 and 2), for comparison. Since almost 70% represent '< values', these plots are largely a reflection of detection limits and do not give a realistic idea of distributions. Thus, sites where samples were determined in 2001 in figure 23 appear to manifest higher concentrations of γ-HCH than sites monitored in 1997 because there has been a slight increase in detection limits. Nevertheless, they demonstrate compliance with the estuarine EQS. Although individual values up to 2.5 ng l^{-1} were measured occasionally, median values for all sites fall well below the standard for coastal waters (20ng l^{-1}).

Any temporal trends for γ-HCH in tidal waters are also masked by changing detection limits.

There is no indication of enhancement of γ-HCH near the Weymouth STW long out-fall, though small inputs from Weymouth STW may have occurred in the recent past. Over the last decade the average concentration of lindane in effluent samples was 41± 67 ng l^{-1} (ignoring < values). There is no substantial evidence to indicate that inputs are declining substantially : though the maximum value of 470 ng l^{-1} was recorded in 1997, values up to 100 ng l^{-1} were still present in 2001. Nevertheless, we can speculate from tidal waters data that risks to biota from γ-HCH are likely to be low.

West Bay sewage samples (figure 20) contained comparable concentrations of γ-HCH (mean 36.9 ± 28.8 ng l^{-1}), though given the dilution at sea it is unlikely that these would give rise to acute effects in the receiving environment of West Bay.
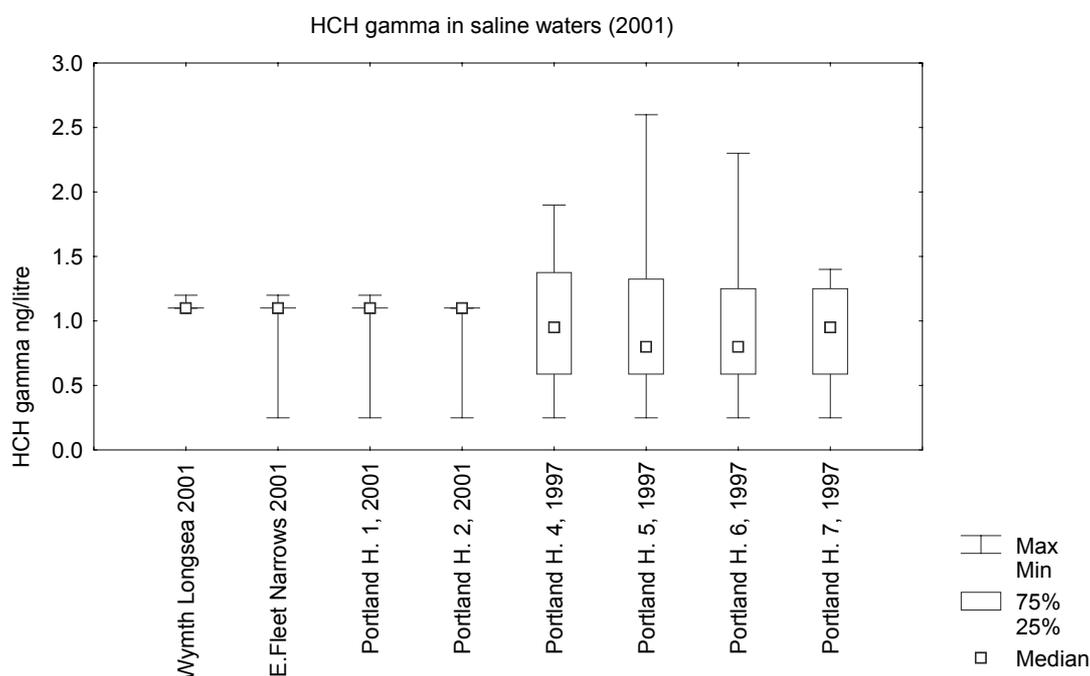
Figure 23. Concentrations of γ-HCH (ng l^{-1}) in tidal waters, Fleet and Portland Harbour and near the Weymouth long-sea outfall. Data source EA.

Calculated mean values for γ-HCH in sediments from Portland Harbour, close to the mouth of the Fleet were marginally above the probable effects level (see EA summary statistics in tables 24, section 7). Further monitoring is advisable to establish the situation for sediments within the Fleet - to ensure standstill requirements of the Dangerous Substances Directive and attainment of Favourable Condition (Habitats Directive).

EA 'unspecified' shellfish samples from the Portland Harbour shellfish site near the Fleet contain low, but measurable, levels of γ-HCH (results presented on a wet weight basis in table 12). Adopting a wet:dry ratio of five, the average value for shellfish (~10 μg kg^{-1} dw) is below the recommended 'no-effects' guideline of 30 μg kg^{-1} (dw) set by ADRIS. This suggests that the sampled 'shellfish' (and consumers) may not be

at risk from γ-HCH, though the presence of detectable levels in shellfish (and sediments) reinforces the need for more detailed information to characterise the marine site and its sources.

Endosulphan

Endosulphan is a mixture of two isomers, endosulphan 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 endosulphan has led to its ban in many countries. Endosulphan 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).

Over the last decade, there have been very few measurements of concentrations of endosulphan (A and B) and most have been below detection limits. The most recent data available are summarised in table 13 and figure 20 using ½ detection limits for '< values'.

Concentrations in discharges from Weymouth STW appear to be small, and in tidal waters at Portland were below the EQS of 3 ng l^{-1}. Similarly endosulphan was below the limit of detection in all samples from the West Bay sewage pumping station (Table 13 and figure 20). On this evidence, sources of endosulphan are probably not a significant issue, though an accurate assessment is clearly not possible.

Table 13. Endosulphan A and B in tidal waters and effluent from Weymouth STW, assuming ½ DL values (Data source EA)

Site	Year	endosulphan A median ng l^{-1}	endosulphan B median ng l^{-1}
Portland Harbour one	1992	0.75	0.75
Portland Harbour two	1992	0.75	0.75
Weymouth STW	1996	0.3	1.58
West Bay STW	1996	0.3	1.0

Greve and Wit (1971) found that 75% of the endosulphan in the River Rhine was associated with particulate matter (mud and silt), therefore sediment levels in the cSAC/SPA may be more relevant, toxicologically. All values for Portland sediments were below detection limits (see table 25, section 7). Similarly, endosulphan A levels in EA shellfish samples from the Portland shellfish site were undetectable (table 11). It is likely, therefore, that impact on the European marine site is minimal, but this conclusion requires verification by analysis of samples from within the Fleet itself.

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 $10\mu g\ l^{-1}$. Sub-lethal effects on olfactory function in Atlantic salmon were observed after exposure to the OP diazinon at concentrations as low as $1\ \mu g\ l^{-1}$, and significantly reduced levels of reproductive steroids in mature male salmon parr resulted from exposure to $0.3\mu g\ l^{-1}$ 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).

Samples analysed for OPs (azinphos-methyl, fenitrothion and malathion) were restricted to STW effluent samples. A large number of results are below detection limits (see table 14 for Weymouth, figure 24 for West Bay). As indicated for Weymouth data, these limits were sometimes at or above 'EQS values' (expressed as annual averages and for protection of marine life – not applicable to STW samples). Similar conclusions can be drawn for OPs in effluent samples from West Bay STW: median values for azinphos-methyl, fenitrothion and malathion were $\sim10\ ng\ l^{-1}$, calculated by halving 'less-than' values (statistics summarised in figure 24.). The absence of environmental data makes a valid assessment of the Fleet impossible, though if the values in table 14 and figure 24 are representative, STW inputs are unlikely to result in concentrations above the EQS outside the immediate mixing zone and it is doubtful that these sources represent a threat to the cSAC. Nevertheless, it would seem advisable in future to examine sources within the site in more detail.

Table 14. Organophosphates ($ng\ l^{-1}$) in Weymouth STW effluent. (EA Data)

OP	% values <DL	DL range 1990-2002 ($ng\ l^{-1}$)	annual median[†] (most recent year)	EQS[2] $ng\ l^{-1}$ mean	maximum
azinphos-methyl	100%	20	10 (1996)	10	40
malathion	0%		20.6(1996)	20	500
fenitrothion	100%	20 - 21	10.4 (1996)	10	250

[†]calculated by halving 'less-than' values; [2]EQS for protection of saltwater life - not strictly applicable to these samples
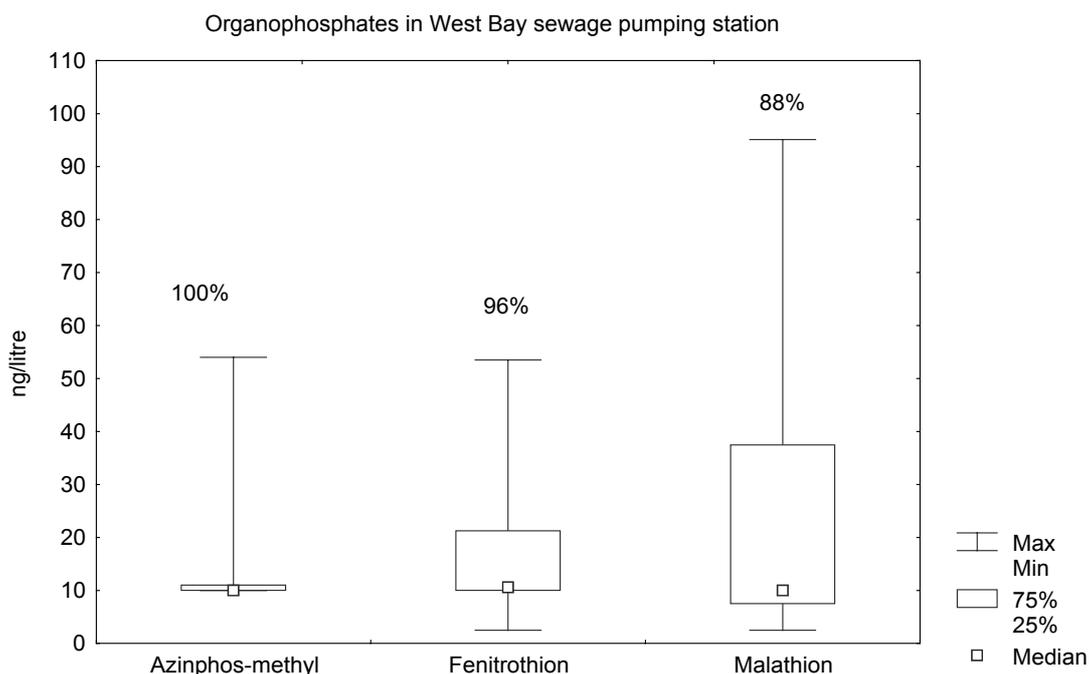
Figure 24. Summary statistics for concentrations of organophosphate pesticides in effluent from West Bay sewage pumping station during the 1990s. Percentage of values below detection limit are show above each plot (Data Source EA).

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 $2\mu g\ l^{-1}$ (annual average) or 10 $\mu g\ l^{-1}$ (maximum allowable concentration). Though toxic, they are not accumulated significantly by organisms. Nevertheless, they have been identified as potential endocrine-disrupting substances (Allen *et al*., 2000).

Because of their major usage and high water solubility, *s*-triazine herbicides 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.

Simazine and atrazine were analysed in Weymouth STW effluent in 1996 (n=2). Atrazine in both samples was below detection limits (<30 and <32 ng l^{-1}) whilst simazine was present at 44 and 56 ng l^{-1}. Summary statistics from West Bay sewage samples (up to 1996) are shown in figure 25 (together with data for trifluralin). Estimated means were 32 ± 26 ng l^{-1} for atrazine and 41 ± 55 ng l^{-1} for simazine, though as indicated in the figure a large proportion of values were derived from (half) detection limits.

Combined concentrations of these triazines would be well below EQS values (expressed as annual averages or maximum values) for both sets of samples. If these levels are representative, STW inputs are unlikely to result in concentrations above the EQS in receiving water and it is unlikely that these sources represent a threat to the European marine site. However, environmental data for the Fleet are insufficient to make a valid assessment on atrazine and simazine and it is recommended that some monitoring be carried out.
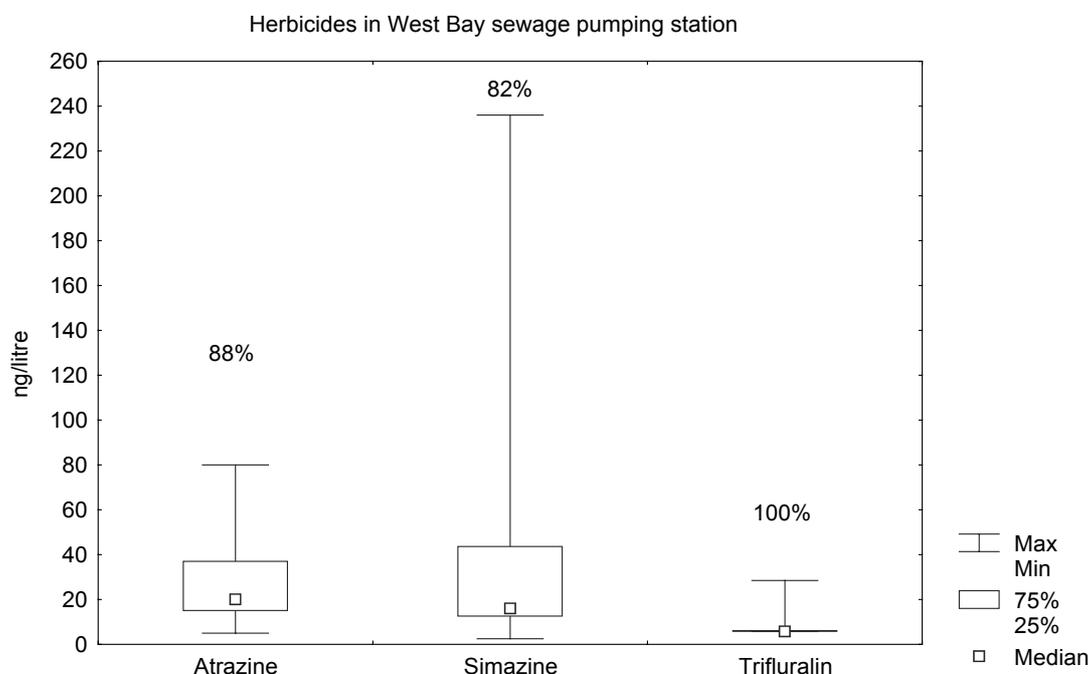

Figure 25.   Summary statistics for concentrations of herbicides in effluent from West Bay sewage pumping station during the 1990s. Percentage of values below detection limit are show above each plot (Data Source EA).

Trifluralin

Trifluralin is a dinitroanaline herbicide used for the control of broad leaved weeds. EC production is of the order of 1000-3000 tons per year of which 6 tons are used in the UK. Acute toxicity to algae has been observed at concentrations of $22 \mu g \ l^{-1}$ (Yockim, *et al* 1983), to freshwater fish such as rainbow trout at 10-40 $\mu g \ l^{-1}$ and to marine fish at 45-440 $\mu g \ l^{-1}$ (Verschueren, 1983; Manual of Acute Toxicity, 1986). The LC_{50} in *Daphnia pulex* is $240 \mu g \ l^{-1}$ (Verschueren, 1983).

Trifluralin is a list II compound (DSD), and is on the UK red list with an EQS of 0.1 $\mu g \ l^{-1}$ (annual average) and $20 \mu g \ l^{-1}$ (maximum allowable concentration).

The only analyses of trifluralin were in effluent samples from Weymouth STW and West Bay STW (summary statistics for West Bay shown in figure 25). All samples were below detection limits which ranged from <5 - <57 ng l^{-1}. Data are insufficient to make a valid assessment though if the above concentrations are representative (estimated mean 8.2 ng l^{-1} at West Bay, 5 ng l^{-1} at Weymouth), STW inputs are unlikely to result in concentrations above the EQS in receiving water and it is unlikely that these sources represent a threat to the European marine site.

Trifluralin has low water solubility and tends to be adsorbed strongly on to sediments and soils and is likely to be resistant to leaching. Sediment from the shellfish site in Portland Harbour, at the entrance to the Fleet, contained measurable amounts of trifluralin in about one third of the samples analysed. By incorporating half detection limits for samples with '< than' designations, the median concentration was estimated to be 3.7 µg kg^{-1}. There are no sediment quality guidelines for trifluralin, though it seems unlikely that the compound would cause impact at this level of contamination.

Trifluralin has been monitored by the EA in shellfish from the beds in Portland Harbour. The reported values are all below detection limits (table 11).

Polychlorinated Biphenyls - PCBs

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. Like the majority of organochlorine substances, PCBs are lipophilic, dissolving more readily in fats than in water; therefore they tend to accumulate in the tissues of living organisms. 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 UK 'red list' of dangerous substances[1].

There are no data for fresh-water, estuarine water, STW effluent or sediments to assess inputs to the European marine site. Several PCB congeners have been monitored by the EA in shellfish from the beds in Portland Harbour (13). The reported values are all below detection limits.

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 an estimated 16.3 µg kg^{-1} wet wt in Portland shellfish. For a group of seven PCB isomers (ΣPCBs 28, 52, 101, 118, 138, 153, 180), similar to the ones analysed by EA, OSPAR ecotoxicological guidelines for mussels are in the range 5-58µg kg^{-1} dw (estimated 1-11.6 µg kg^{-1} ww) – marginally below levels in Portland shellfish. It should be stressed however that these are merely estimates based on half detection limits, and real values may be lower. The inference is that the level of PCB contamination and bioaccumulation is relatively low in waters adjacent to the European marine site, though it would be useful to verify this on a broader scale.

Volatile organics (solvents, freons).

A number of volatile organic compounds (VOCs) are, potentially, endocrine disruptors, as well as being toxic directly and may be discharged into the European marine site in small quantities. These include:

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[1] The UK has a Red List of 23 of the most dangerous substances which have been selected for priority control including through the system of integrated pollution control. The Red List includes the substances on the EC List I under the Dangerous Substances Directive and there are statutory Environmental Quality Standards (EQSs) in place for their discharge into surface waters.

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).

Chloroform has an EQS of 12µg l^{-1} (annual average) in all waters. The only data for tidal waters relate to the shellfish sites in Portland Harbour (one and two), sampled in 1991. The majority of values were at or below detection limits (75%). These DLs ranged from 1-2 µg l^{-1}. Annual average concentrations, based on ½DL, are invariably below the EQS (table 15) implying that chloroform is not a significant concern in tidal waters.

Some small discharges of chloroform occur from STW, though concentrations in effluent from Weymouth STW appear be relatively low for the four years monitored up to 1996 (data for 1992 given in table 15). The annual median concentration (<1µg l^{-1}) does not appear to be a direct toxicological concern.

Table 15. Annual median concentrations for chlorinated solvents in sea water and effluent samples, Portland and Weymouth (Data source, EA).

site	year	trichloromethane (chloroform) median µg l^{-1}	tetrachloromethane median µg l^{-1}	trichloroethene median µg l^{-1}	1,1,1 trichloroethane median µg l^{-1}
sea water					
Portland Harbour one	1991	0.5	0.5		
Portland Harbour two	1991	0.5	0.5		
Off Weymouth long sea outfall	2001			0.025	
effluent					
Weymouth STW	1992	0.79	0.025	0.77	0.48
EQS		12	12	10	100

Carbon tetrachloride (Tetrachloromethane)

Carbon tetrachloride is mostly produced for use in the manufacture of chlorofluorocarbons (CFCs) and 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}.

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

The only data for tidal waters relate to the shellfish sites in Portland Harbour (one and two), sampled in 1991. The majority of values (88%) were below detection limits (1 µg l^{-1}). Annual average concentrations, based on ½DL, were invariably below the EQS (table 15) implying that carbon tetrachloride (tetrachloromethane) is not a significant concern in tidal waters.

Some small discharges of tetrachloromethane occur from STW, though concentrations in effluent from Weymouth STW appear be relatively low for all monitored years (data for 1992 given in table 15). The annual median concentration throughout the 1990s was <1µg l^{-1} and does not appear to be a direct toxicological concern.

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 are unlikely to exceed relevant EQS (10µg l^{-1} annual average) derived for the protection of saltwater life.

Data for trichloroethylene (trichloroethene) are examined here as being representative of this group of compounds. The only data for tidal waters relate to sea water sampled near the Weymouth long sea outfall, since 2001. All values were below detection limits (0.05 µg l^{-1}). Annual average concentrations, based on ½DL, were thus invariably below the EQS (table 15) implying that trichloroethene is not a significant concern in tidal waters.

Discharges of trichloroethene via STW appear to be relatively minor. The estimated annual average concentrations in effluent from Weymouth STW throughout the 1990s was <1 µg l^{-1} and appears to have been decreasing since 1992 (figure 26).

Trichloroethene in Weymouth STW outfall
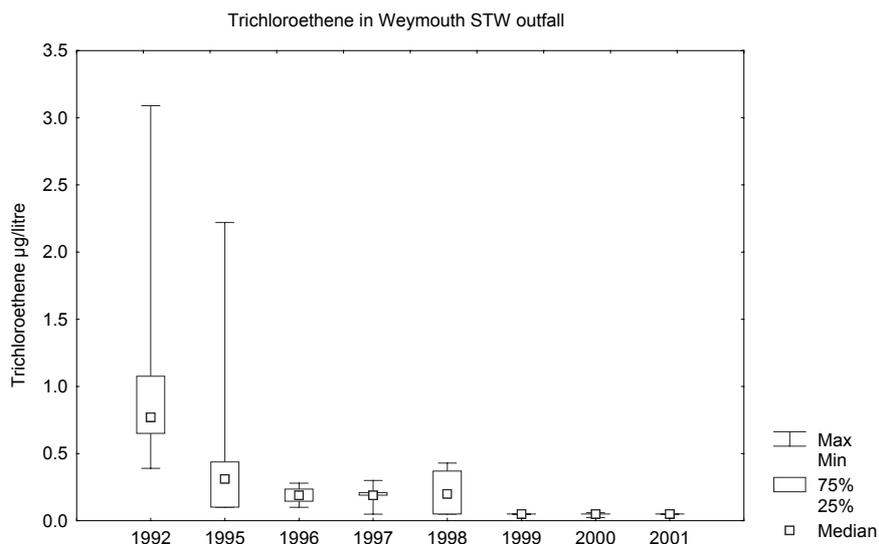
Figure 26. Trichloroethene in effluent samples from Weymouth STW (data source EA).

Trichloroethane

Trichloroethane, a list II substance, is used as an industrial solvent. Potential sources of contamination include direct discharge of wastewaters, accidental spillages and deposition from the atmosphere.

1,1,1 Trichloroethane has an EQS of 100μg l^{-1} (annual average) in all waters. There are no data for tidal waters although occasional pulses of trichloroethane are encountered periodically in Weymouth STW effluent (up to 37 μg l^{-1}, figure 27). Overall, however, the estimated annual average concentrations in effluent from this source are unlikely to raise concentrations in receiving waters to toxic levels.

Trichloroethane in Weymouth STW outfall
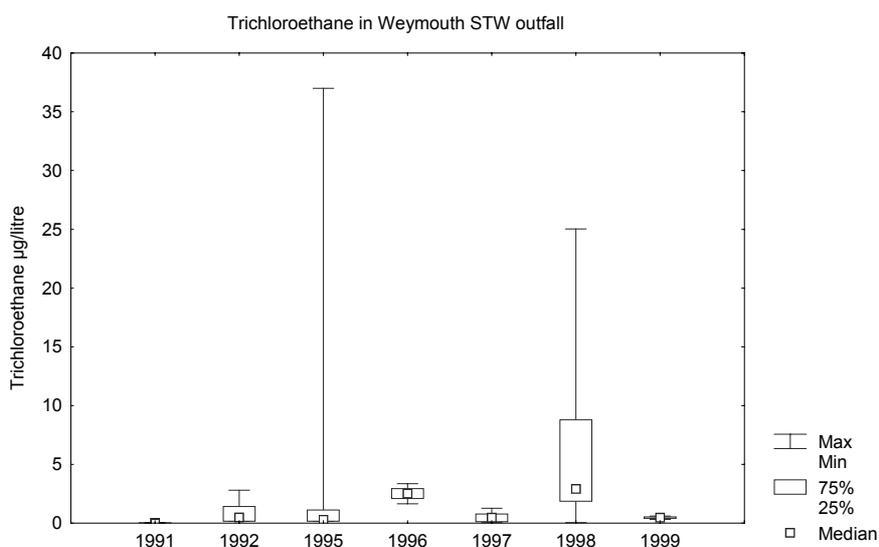
Figure 27. Trichloroethane in effluent samples from Weymouth STW (data source EA).

It seems unlikely, therefore, that volatile organics represent a significant threat to the European marine site, though confirmation should be sought by analysis of waters from the Fleet.

6.1.5   Alkylphenols And Other Endocrine Disruptors

Alkylphenolpolyethoxylates (APEs) are a major component in surfactants in detergents and their presence in sewage effluent and adjacent freshwaters has been associated with 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.,* 1999). There is now a voluntary ban on their use in the UK domestic market.

In fresh water, concentrations up to $180\mu g\ l^{-1}$ of the degradation product nonyl phenol (NP) have been measured near to STW on the River Aire, Yorkshire (Blackburn and Waldock, 1995; Blackburn *et al.,* 1999). Concentrations of alkylphenols in tidal waters are generally much lower than in fresh water, except in industrial estuaries such as the Mersey and Tees ($5\ \mu g\ l^{-1}$ nonyl phenol). In freshwater sediments close to known sources (eg in the River Aire), concentrations up to $15\mu g\ g^{-1}$ have been found, and concentrations in the sediments of the Mersey and Tees Estuaries may exceed $1\mu g\ g^{-1}$ (Blackburn *et al.,* 1999). Unfortunately data on these compounds, or other endocrine disruptors such as steroid hormones could not be found for the Fleet. The possibility of endocrine disruption does not appear to have been investigated in this system and should perhaps be addressed in future.

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 $25\mu g\ l^{-1}$, 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 the Fleet.

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). Levels of metal bioaccumulation in shellfish near the mouth of the Fleet (Portland) do not appear to give cause for concern. However, experimental studies on freshwater crayfish have indicated that Cd and Hg at the relatively low concentration of $0.5\mu g\ g^{-1}$ body weight can arrest ovarian maturation due to inhibition of gonad stimulating hormone and serotonin, respectively (Reddy *et al.*, 1997). Until a systematic survey of endocrine disruption is carried out for the Fleet, similar reactions in marine crustaceans and other organisms from the European site cannot be ruled out, though as yet, methods to establish field effects of endocrine disruptors in invertebrates are few.

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. Although there are currently no grounds to suspect such problems in the Fleet, it would be informative to test these general stress responses in organisms (along with other 'biomarkers' described in annex 6), if only to establish baseline information on chronic effects.

Many bird species are extremely sensitive to endocrine disrupting substances such as pesticides, PCBs, PAHs and metals (reviewed by Fry, 1995). The threat of food chain biomagnification through consumption of prey species such as worms and molluscs (which are excellent bioaccumulators of contaminants), or accidental ingestion of sediment by waders feeding on mud-flats, might appear to represent relatively small risks within the Fleet, based on the rather limited evidence available. However, more detailed study is needed to evaluate, thoroughly, current risks to both invertebrates and higher organisms.

6.2 Non-Toxic Contaminants

This section deals with non-toxic contamination in the European marine site. 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). The following provides an overview of the nutrient status of the Chesil and the Fleet European Marine Site. For further information, Johnston and Gilliland (2000) have produced a comprehensive report, which includes an in-depth case study of nutrients in the Fleet, as well as reviews of studies and modelling exercises relating to the water quality of the lagoon, and implications for management of the Fleet.

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 tidal lagoons, such as the Fleet, point source inputs may be important due to the enclosed nature of the waterbody. The potential for nutrient enrichment and localised effects will be determined by physico-chemical and biological characteristics of the lagoon such as flow, seasonal variability, flushing, tidal regime, primary production and rates of remineralisation. These variables are unique to individual catchments and confound attempts to make accurate predictions without taking them into consideration.

Using modelled estimates of nutrient inputs, Mainstone and Parr, (1999) constructed annual nutrient budgets for both nitrogen and phosphorus in the Fleet (table 16). Point sources consisted of the two public sewage treatment works, Abbotsbury and Langton Herring, and two private sewage treatment works at Moonfleet Manor Hotel and Royal Engineers Training Camp at Chickerell. Diffuse source loads estimated were from atmospheric deposition, agriculture, bird populations and groundwater. Additionally, the swannery at Abbotsbury represented a potential source of nutrients. This was estimated by calculating the nutrient load delivered to the swans in feed (whole wheat, crumbs and pellets), and applying a conversion efficiency to estimate the loads generated in excreta.

Table 16. Summary of estimated annual nutrient loads from different sources to the Fleet (from Mainstone and Parr, 1999)[1].

Source		Nitrogen %	Phosphorus %
Point sources	STWs	1-2.5	12-39
Agricultural sources	Livestock	34	37-47
	Fertiliser application	49-50	18-23
Diffuse sources	Abbotsbury swannery	0.3	2
	Other bird species	0.2	2
	Atmospheric deposition	14	10-13

Despite concerns over the reliability of some aspects of the data on which the estimates were based, the authors considered agricultural sources to be highly important for both nitrogen and phosphorus. Up to 84% of the annual load of nitrogen, and 70%, of the annual total phosphorus load to the Fleet was estimated to come from agricultural sources. Contributions from pigs and poultry were not included in estimations of inputs from livestock; their inclusion may have increased the significance of the agricultural load. The two sewage works at Abbotsbury and Langton Herring could contribute as much as 40% of the annual phosphorus load to the Fleet, although this figure may be as low as 12%. Bird populations (including the mute swans at Abbotsbury and their feed) did not appear to be providing a major

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[1] The consented DWF of 140 m^3 day^{-1} at Abbotsbury STW is likely to have been exceeded for many years. Wessex Water has calculated a theoretical DWF of 243 m^3 day^{-1} based on the connected population, and this is predicted to increase to 290 m^3 day^{-1} by 2020 (EA pers comm.). This obviously casts doubt on modelled contributions of P and N.

contribution to nutrient loads entering the lagoon as a whole, although they may be locally important.

In terms of actual loading, Murdoch (1999) estimated annual total inorganic nitrogen (TIN) and phosphorus (P) loads to provide an assessment of relative contributions from various sources (figure 28).
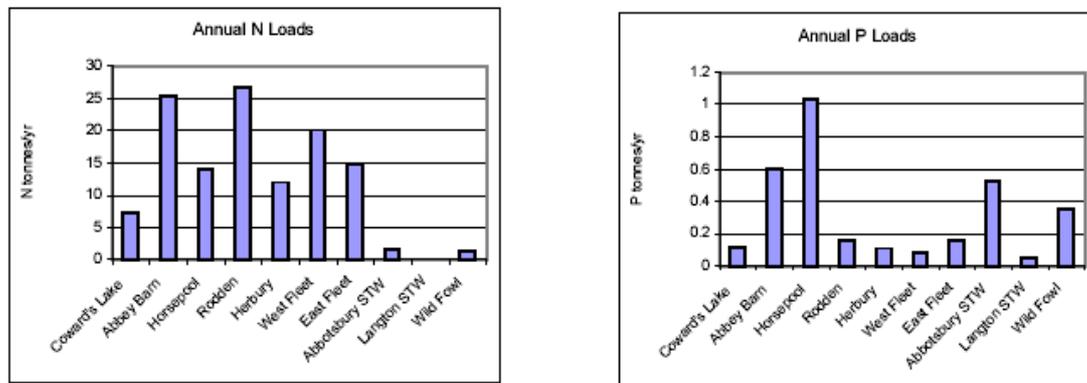

Figure 28. Estimated annual loads of TIN and P to the Fleet (tonnes/year) from different sources (from Murdoch, 1999). NB The loads for Mill stream (Abbey Barn) and for Rodden stream include the inputs from Abbotsbury and Langton STWs respectively, although both are also shown individually. NB See footnote on previous page.

The estimates showed that the dominant TIN load was from freshwater streams, and the chief source of nitrogen again considered to be from agricultural fertiliser use. Total inorganic nitrogen (TIN) loads were greatest at the Abbotsbury (western) end of the Fleet. There was a strong annual cycle for the stream inputs, which peaked in winter, this was thought to be a result of winter runoff and flushing of nitrates. Murdoch (1999) considered the principal area for arable land cultivation in the Fleet catchment to be around Langton Herring, between Roddon and West Fleet streams. In agreement with Mainstone and Parr (1999), contributions from point sources and wild fowl appeared relatively small.

In contrast to TIN, the relative contribution of P from STWs and wildfowl were thought to be more significant, on an annual basis, in comparison with the loads from streams. The principal source of phosphorus inputs into the freshwater streams is also estimated to be from agricultural sources, primarily livestock farming (concentrated in the western Fleet), but also from mixed fertiliser application and land runoff. The model showed winter and summer peaks in phosphorus loads. The winter highs appeared to be related to stream inputs (particularly Mill stream (Horsepool) and Mill stream (Abbey Barn) (which both receive STW effluents). Summer peaks appeared to be related to wildfowl inputs.

It should be noted that the estimated input for wildfowl in this model assumes a worst case of all wildfowl faeces entering the lagoon directly rather than only a third, with two thirds being deposited on land (as estimated by Mainstone and Parr 1999). If the latter were the case, then the stream inputs (largely from run-off and sewage) would be likely to account for a greater proportion of the nitrogen and phosphorous budgets.

In most estuarine and coastal lagoon environments, 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 *a* 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 $0.07 - 1.85$mg l^{-1} TIN and $0.007 - 0.165$mg l^{-1} TRP are found in coastal waters, whilst the upper reaches of estuaries have nitrogen concentrations similar to those in river water, $0.1 - 15$mg l^{-1} TIN. TRP in upper estuaries, as in rivers can also be variable, $0 - 11.4$mg l^{-1}. Nutrient levels in semi-enclosed lagoons such as the Fleet have been little reported, and will be determined by a number of factors, such as individual physico-chemical parameters.

There is surprisingly little information available in the literature regarding nutrients in the Fleet. Johnston and Gilliland (2000) reviewed reports, journal articles, letters, and other items of relevance to the Fleet (1933 to 2000) held in the Fleet Study Group

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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).

archive and noted that information concerned primarily the biological and historical interest of the Fleet and none contained reliable data on nutrients. Despite concerns raised in some of the reports reviewed about possible impacts (including eutrophication) on the Fleet (e.g. Elton, 1991, Holmes, 1983), there appear to have been few comprehensive surveys of water quality in the lagoon prior to the Environment Agency investigations from 1996 onwards (see below). Thus, it has not been possible to effectively review past nutrient status of the Fleet, or to identify long-term trends in nutrient concentration.

A study in the mid-nineties (John, 1995) measured the nutrient content of the water column at eight sites along the length of the Fleet. Generally, concentrations of nitrate, phosphate and ammonium along the Fleet were considered average for brackish waters, however, occasional peaks in nitrate concentration were recorded at Abbotsbury, Langton Herring and Clouds Hill, up to 0.35, 0.13 and 0.08 mg l^{-1} N, respectively. Some of the more elevated measurements coincided with the occurrence of dinoflagellate blooms, when large numbers of *Glenodinium foliaceum* and *Oxyrrhis* sp. were observed. Up to 3.72mg l^{-1} N was detected in Mill Stream (Abbey Barn) during a *Glenodinium* bloom during August, 1995, although a short distance from the stream in the Abbotsbury Embayment itself, the concentration was an order of magnitude lower.

Regression analysis indicated that the elevated total inorganic nitrogen levels observed at Abbotsbury were significantly correlated with the increased population density of *Glenodinium foliaceum* during the August bloom, and an attempt was made to determine if phytoplankton populations were nutrient limited, by comparison of the ratios of different plant pigments (carotenoids and chlorophyll *a*). However, the ratio was found to vary considerably, and did not correspond to variations in nutrient content of the water, probably because the phytoplankton populations were of mixed species. Thus, no clear conclusions could be drawn from the comparison.

A more useful indicator of nutrient limitation can be the ratio between nitrates and phosphates. Table 17 gives estimates of the N:P ratio based on EA survey data using dissolved available nutrients. At all sites, N:P is greater than 10, therefore P could theoretically limit plant growth.   Johnston and Gilliland (2000) suggest that both nutrients could theoretically limit plant growth to varying degrees at different times of the year, although in practice, nutrients were unlikely to be limiting because concentrations were excessively high, particularly in the Abbotsbury embayment, throughout the year and it could only be postulated as to whether this would be the case under natural conditions (Johnston and Gilliland, 2000).

Table 17.   Estimated N:P ratios at sites within the Fleet using dissolved available nutrients. N is derived from TON plus ammonia. (Based on means from EA nutrient data for April 1996-August 1997) From Johnston and Gilliland (2000)

Site	N:P Ratio
Abbotsbury Swannery	14.6
Langton Hive	15.9
Chickerell Hive	15.7
The Narrows	12.1
Small Mouth	10.3

These concerns that the Fleet Lagoon is exhibiting signs of eutrophication have lead to its recent designation as Polluted Waters (Eutrophic) (PWE) under the EC nitrates Directive (91/271/EEC). The catchment area for the Lagoon was designated as a Nitrate Vulnerable Zone (NVZ) in order that restrictions on agricultural activities, in the form of Good Agricultural Practice, could be enforced and thus address the problem of diffuse source pollution. Referring to the reports of Mainstone and Parr, 1999, Murdoch, 1999 and Johnston and Gilliland (2000), combined with additional monitoring information, a recent EA report has summarised the eutrophication problem in the Lagoon EA (2001). Key findings include:

- Chemical and biological data collected over the period 1996-1998 indicate that the Fleet Lagoon is hypernutrified.
- Winter (Dec-Feb) mean DAIN concentrations exceeded the CCST (Comprehensive Studies Task Team) standard defining hypernutrification (12mmol m^3 in the presence of 0.2mol m^3 DAIP) at all monitoring sites within the Fleet. All samples collected at Abbotsbury exceeded this standard.
- Nutrient load analysis indicates that agricultural sources are overwhelmingly important for nitrogen, contributing over 80% of the annual load.
- Agricultural sources are also important for phosphorus, contributing an estimated 56-69% of the annual load.
- With the exception of Abbotsbury, concentrations of DAIN in summer (June-August) were markedly lower than in winter and close to DLs.
- In contrast, summer DAIP concentrations in West Fleet were higher than in winter, possibly due to release from sediments.
- Chlorophyll a concentrations were above the DETR criterion for eutrophic saline waters (10μg l^{-1}) at all sites other than Smallmouth. In West Fleet, exceedences were recorded in both spring and summer, the highest number of which occurred at Abbotsbury.
- In 1997, a spectacular 4-week long bloom of the dinoflagellate *Procentrum micans* occurred in West Fleet, causing a highly visible 'red tide' and elevated chlorophyll a and DO levels.
- A red tide event caused by blooms of *P. micans* was recorded at Chickerell and Moonfleet in November, 2000[1].
- Extensive growths of green macroalgae *Ulva sp.* have occurred in West Fleet leading to complaints from members of the public.

Algal monitoring at Ferrybridge, conducted by CEFAS over the period 1996-2000 was also summarised:

- Blooms of the dinoflagellate *Alexandrium tamarense*, a potential PSP producing species were recorded during 1996, 1999 and 2000. In 1999 the bloom was sustained for at least 3 weeks with counts of up to 40000 cells l^{-1}.
- *Procentrum lima*, a potential DSP producing species was recorded at low levels (<100 cells l^{-1}) in all years.
- The diatom *Pseudo-nitzchia*, a potential ASP producing species, was recorded at levels of up to 8720 cells l^{-1} in 1996.
- DSP was detected in mussels collected from Ferrybridge, and Pacific oysters collected from the Fleet in 2000. This led to a temporary Prohibition Order on shellfish harvesting in the lagoon between July and September 2000.

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[1] Winter algal blooms are unusual and indicative of potentially eutrophic conditions

The report recommended the Fleet Lagoon's designation as a PWE site, and designation of the catchment as an NVZ.

Nutrient concentrations vary with salinity, therefore measurements collected simultaneously from different regions within the lagoon, 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 saline lagoons, 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 the Fleet Lagoon, 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 sites, a number of 'guideline values' have been established which could be of relevance for assessment of the status of nutrients in the catchment of the Fleet Lagoon 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 overenrichment 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 a 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 (Seager *et al.,*, 1988), 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.

- The proposed EQS of **0.021mg l^{-1} un-ionised ammonia** (NH$_3$ N) also applies to EC designated salmonid and cyprinid freshwaters. In addition there is an EQS of **0.78mg l^{-1} total ammonia** for these waters (Seager *et al.,*, 1988).

In the absence of site-specific guidelines these values 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 available data and trends in the Fleet Lagoon and comparisons with other areas
- validity of guideline values and classification schemes

We have used measurements of orthophosphate, elemental P, nitrate, nitrate (as N), and ammonia (as N) as markers of nutrient status in the European marine site. (Nitrate typically makes up the largest proportion of TIN inputs to estuaries and coastal waters, with nitrite and ammonia usually accounting for < 10%).

Rather than relying on a classification scheme for the estuary as a whole it may be more beneficial to investigate the recent distribution of key determinands in finer detail:

6.2.1 Phosphate

Recent values for concentrations of orthophosphate in freshwaters entering the Fleet Lagoon are summarised in figure 29. *It is stressed that the data are for concentration only and do not take flow rates into account[1].* Mill Stream at Horsepool stands out as having highest concentrations (up to 1.7mg l^{-1}), and there is a long term (1990-2001) trend toward increasing annual averages. This stream receives effluent from Abbotsbury STW, although there have been reductions in the orthophosphate content of the effluent over the same period (see below), which might imply that other sources may be responsible (probably agricultural). Orthophosphate levels are moderate to high in West Fleet Stream at Moonfleet (median 0.19mg l^{-1}), also in Rodden, and Mill (Abbeybarn) Streams where although median values were 0.09 and 0.096mg l^{-1} peaks of up to 0.6 and 0.49mg l^{-1} respectively, were recorded. These streams also receive discharges from STWs (Langton and the Swannery respectively). There have been reductions in annual average orthophosphate

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[1] Also for more accurate nutrient budgets, consideration of more relative sources of particulates and dissolved nutrients is needed in order to evaluate sources

concentrations (1990 - 2001) in East Fleet Stream (Old Church) and the Brit at Bridport, whilst at other sites concentrations remained relatively constant.
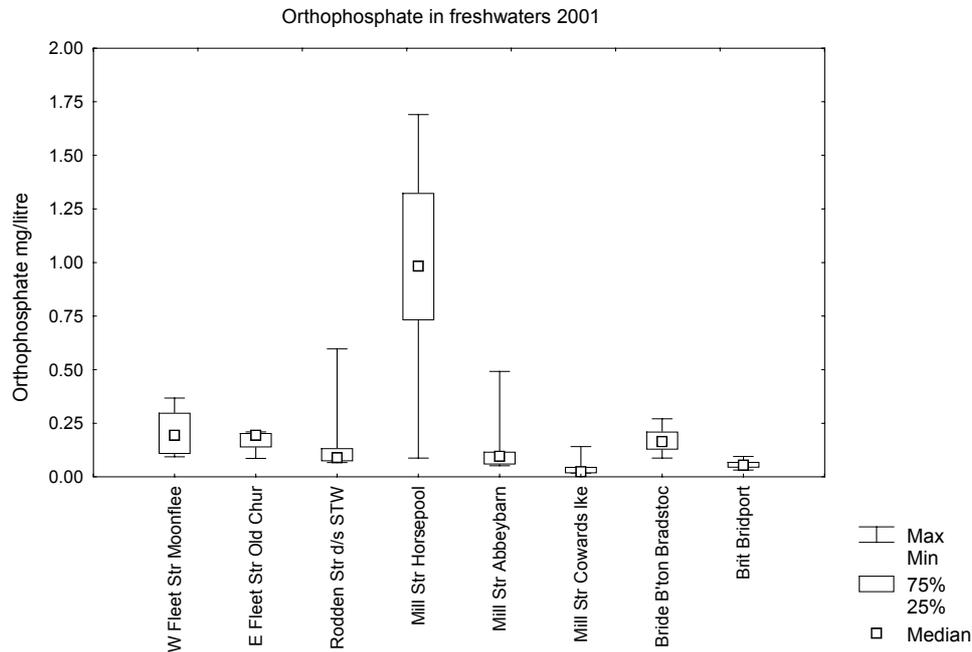

Figure 29. Orthophosphate in freshwaters entering the Fleet Lagoon, and Lyme Bay (west Chesil Bank) 2001. Data source EA.
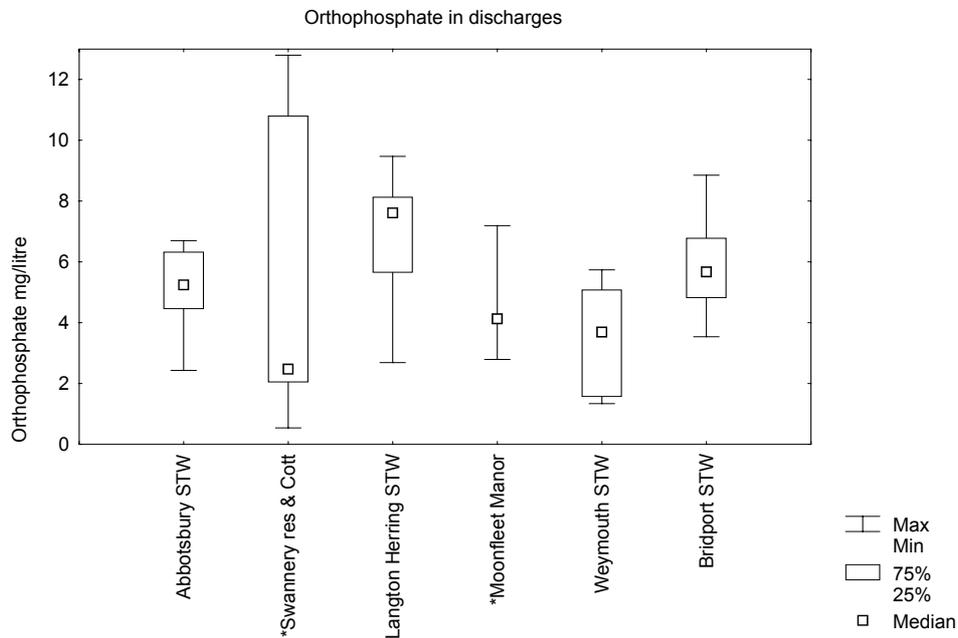
denotes non-water company discharge, combined sewage and trade.

Figure 30. Orthophosphate in discharges to the Fleet Lagoon, tributaries and seaward of Chesil Bank 2001. Data source EA. NB. 25th and 75th percentiles shown where sufficient data are available.

Recent data for orthophosphate concentrations in discharges are summarised in figure 30. *Again it is stressed that the data are for concentrations only and do not take into the account the volumes discharged, details of DWF given in table 18.* Data for Bridport and Weymouth STWs are included as they discharge via long sea outfalls and may therefore impact upon the seaward side of the Chesil Bank. Highest values (up to 12.8mg l^{-1}) occur in the discharge from Swannery Restaurant and Cottages (Abbotsbury), which is described as a combined sewage and trade discharge (unspecified). It discharges into Abbotsbury Mill Stream (Mill stream - Abbeybarn), and may account for the peaks in concentration recorded in the stream during 2001 (see above). The highest median concentration during 2001 (7.61mg l^{-1}) is for Langton Herring STW, which contributes to the overall loading for Rodden stream. There are only three sample values for Moonfleet Manor Hotel, which has a private STW with a DWF of 27 m^{3}day^{-1}. It discharges directly into the Fleet, and has a history of poor consent compliance (Johnston and Gilliland 2000).

Table 18. DWF for Sewage Treatment Works

Discharge	DWF (m^3 day^{-1})	Receiving waters
Abbotsbury STW	140[1]	Mill Stream (Horsepool)
Swannery Restaurant & Cottages	10.3	Mill Stream (Abbey Barn)
Langton Herring STW	<100 *	Rodden Stream
Moonfleet Manor Hotel STW	27	Fleet Lagoon

*estimated (Johnston and Gilliland, 2000)

Additionally, an outlet from Abbotsbury Oysters purifying unit discharges of 2m^3 day^{-1}. There have been two other discharges to the Fleet: RETC Chickerell STW had a DWF of 64m^3 day^{-1} discharging direct to the Fleet. Ammoniacal nitrogen concentrations of the effluent were very variable (probably due to short term variations in population at the camp). This discharge was incorporated into the mains sewer system by April 1999. RETC Wyke Regis was a small septic tank with a DWF of 9m^3 day^{-1} and entered the Fleet at the Narrows where there are strong tidal currents and dilution is large. No elevation of nutrients at this point was detected during surveys. This flow has also been transferred to mains sewerage (Johnston and Gilliland (2000).

Data for orthophosphate in discharges from Abbotsbury STW show distinct seasonality with summer peaks in concentration. There have been gradual reductions (1990-2001) in annual average values of orthophosphate for all discharges except Weymouth which remained relatively constant (3-6mg l^{-1}) apart from a peak (up to

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[1] The consented DWF of 140 m^3 day^{-1} at Abbotsbury STW is likely to have been exceeded for many years. Wessex Water has calculated a theoretical DWF of 243 m^3 day^{-1} based on the connected population, and this is predicted to increase to 290 m^3 day^{-1} by 2020 (EA pers comm.).

10mg l^{-1}) in 1999-2000, and Bridport STW (no long-term data). Data for the Swannery restaurant are for 1998 to date, but also shows a downward trend.

Orthophosphate (filtered) in saline waters

1998

Prefix W or E = West or East Fleet
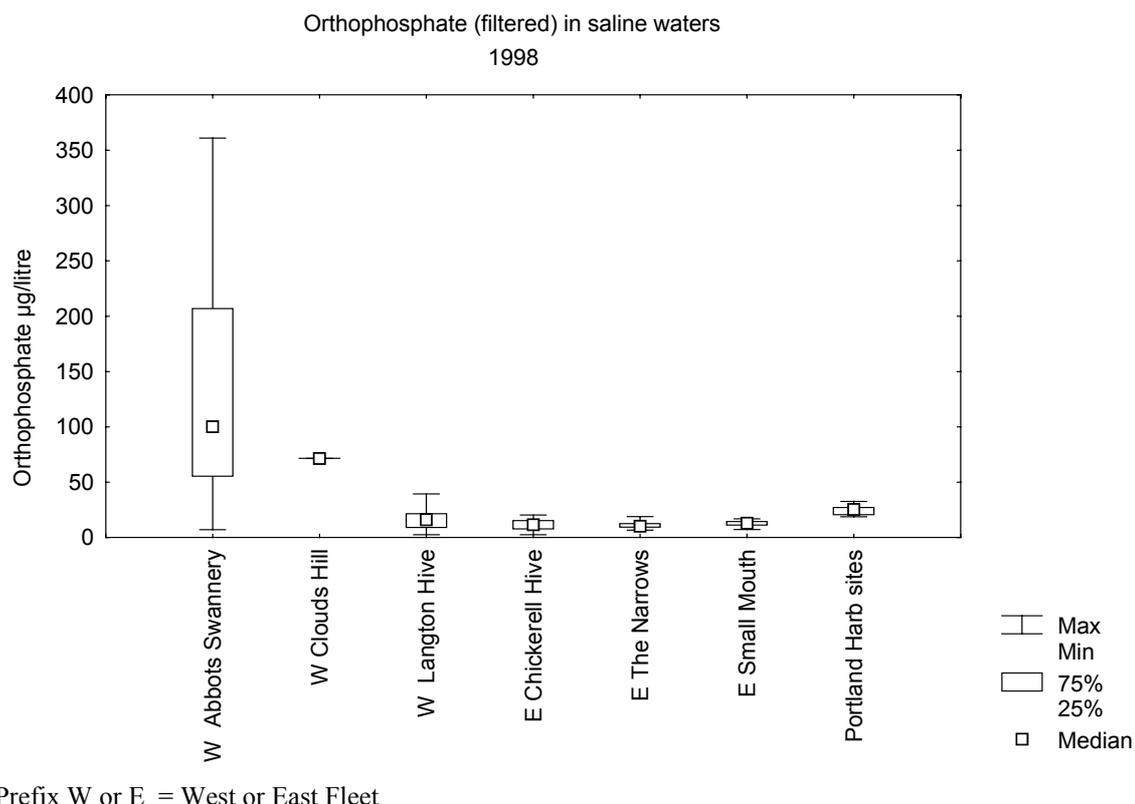
Figure 31. Orthophosphate (µg l^{-1}) in saline waters of the Fleet Lagoon (1998) 'Portland Harbour sites' summarises data for 10 sites in the Harbour. Data source EA.

The most recent available (complete year) data for orthophosphate in tidal waters are summarised in figure 31. West Fleet stands out as having the highest concentrations by far, with up to 361µg l^{-1} at Abbotsbury Swannery. The Swannery site was also the most variable. Concentrations reduce eastwards, where lowest median concentrations occurred at Chickerell Hive, the Narrows and Small Mouth (11.6, 10.2 and 13µg l^{-1} respectively). The median value for all monitoring in Portland Harbour during 1998 was 25.4µg l^{-1}, therefore the Harbour may represent a small source of orthophosphate to East Fleet during the flood tide (tracer studies indicate that the eastern Fleet has good tidal exchange with Portland Harbour - Johnston and Gilliland, 2000). However it is not clear whether there is a net gain or loss of orthophosphate from this source.

Calculated as elemental P, the approximate background for the tidal waters (25th percentile) is in the range 2.5 – 23.3µg l^{-1} (table 19), 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}). Calculated in this way, the more elevated levels of P are still in West Fleet, Clouds Hill manifesting the highest background concentrations.

Table 19.   Elemental P: 25th percentile for the Fleet tidal water sites 1998. Data source:EA

Site	Elemental P 25th percentile $\mu g\ l^{-1}$
West Fleet At Abbotsbury Swannery	18.04
West Fleet At Clouds Hill	23.33
West Fleet At Langton Hive Point	2.97
East Fleet At Chickerell Hive Point	2.55
East Fleet At The Narrows	3.00
East Fleet At Small Mouth	3.59
Portland Harbour 10 Sites	6.72

As there are no comparable saline lagoon sites in the Southwest, The range of background values for elemental P in tidal waters of other European Marine sites in the Southwest are shown in table 20. Waters of the East Fleet appear to be amongst the lowest, whilst background levels in West Fleet are moderate to high.

Table 20. Elemental P: 25th percentile for tidal waters of European marine sites in the Southwest . Data source:EA

European Marine Site	Elemental P 25th percentile $\mu g\ l^{-1}$
Exe	7.5 – 38.6
Poole Harbour	2.01 – 12
Plymouth Sound & Estuaries	1.6 – 27.9
Severn	17 – 125
Fal & Helford	3.2 – 5.8
Fleet Lagoon	2.55 – 23.3

Note that although the sites contains similar salinity ranges, nutrient concentrations are likely to vary with this and other ecological factors, therefore these comparisons are only intended as a broad indication of concentration ranges.

6.2.2   Nitrate

Recent values for nitrate in freshwaters entering the Fleet Lagoon are shown in figure 32. West Fleet Stream is dry for 3-4 months of the year during summer (Johnston and Gilliland, 2000) although most nitrate values appear to be higher than those in other freshwater sources (max value 13 mg l^{-1}). Concentrations are elevated in Rodden Stream, where maximum nitrate levels (15.4 mg l^{-1}) have also exceeded the lower threshold of 10mg l^{-1} during 2001. Rodden stream receives discharges from Langton STW.

There is a trend toward increasing annual mean nitrate concentrations (1990 – 2001) in Mill Stream (Abbey Barn), also Mill Stream (Horsepool) and in the River Brit at

Bridport. The trend is significant for the latter two (p<0.05). In West Fleet Stream (Moonfleet) nitrate levels have fallen overall since 1990, although there was a peak in annual average concentrations between 1997-9 when values of up to 32.9mg l^{-1} were recorded. Concentrations in East Fleet (Old Church) have also fallen over the same period, the highest mean was 17.3 mg l^{-1} in 1995 .
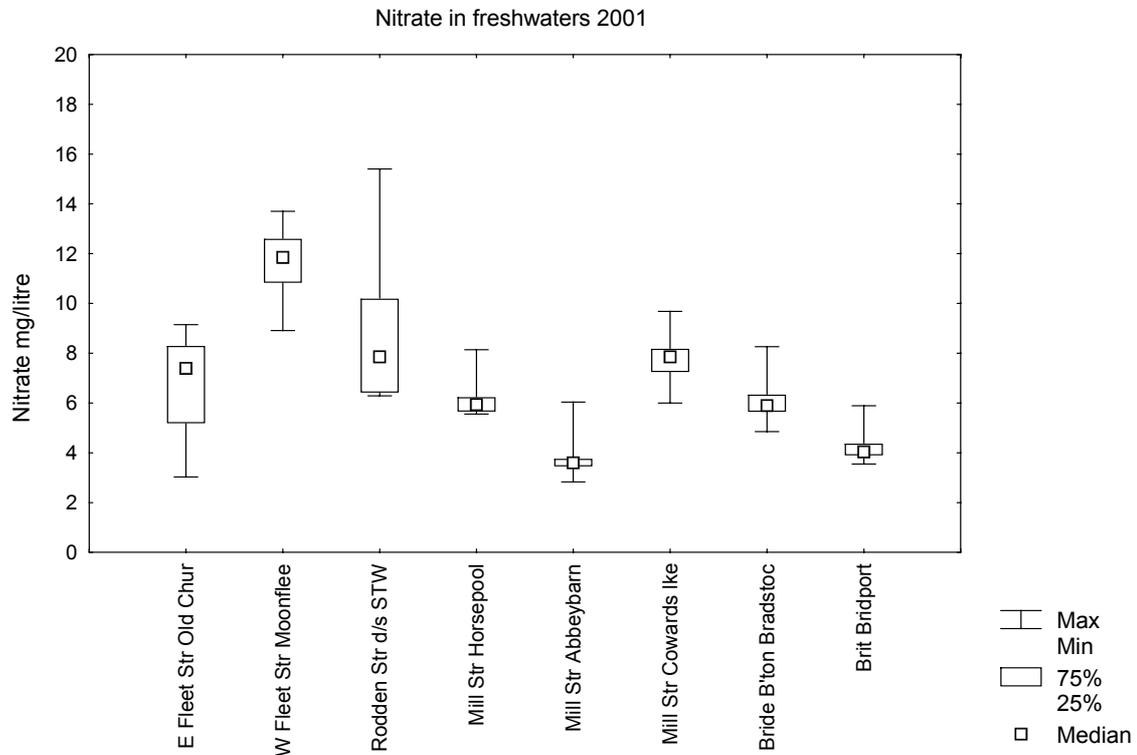
Figure 32. Nitrate in freshwaters entering the Fleet Lagoon, and Lyme Bay (west Chesil Bank) 2001. Data source EA.

Figure 33 summarises data for nitrate in sewage discharges to the Fleet lagoon and tributaries. *Again, it is stressed that the data are for concentration only and does not take into the account volume discharged.* Highest concentrations occurred in discharges from Abbotsbury STW, which is also is the largest input and discharges into Mill Stream (Horsepool). Nitrate concentrations were also high in the combined discharge from Abbotsbury Swannery restaurant and cottages; this discharges into Mill Stream (Abbey Barn). Most of the discharges did not appear to result in extreme concentrations in receiving waters during 2001 (see figure 32), which might, as for phosphate, imply other sources. Langton Herring STW may be the exception, where high concentrations (up to 17mg l^{-1}) in the discharge probably contribute to the elevated nitrate levels in Rodden stream.

Nitrate in discharges
2001
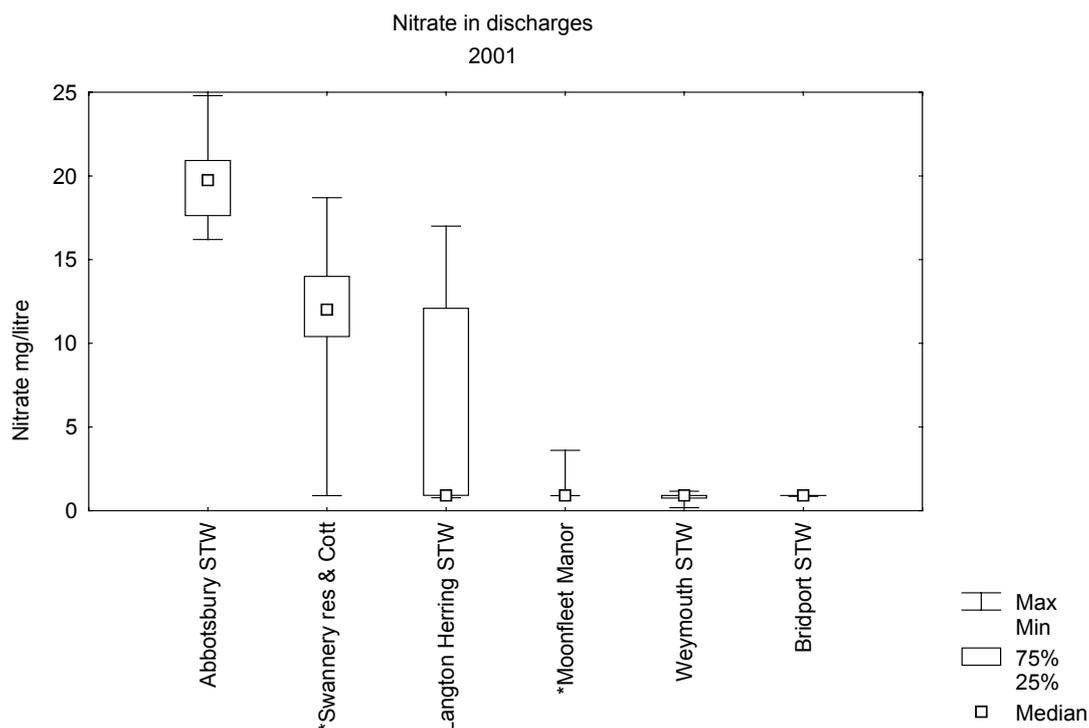
* denotes non-water company discharge.

Figure 33.   Nitrate in discharges to the Fleet Lagoon, tributaries and seaward of Chesil bank 2001. Data source EA.

From 1990 to 1996 a general decrease in concentrations of nitrate was observed in the final effluent from Abbotsbury STW, although this trend has since reversed. Annual average nitrate concentrations in the Langton Herrring discharge were high (up to 30.6mg l^{-1}) in the early 1990's but dramatically reduced (down to 1.98mg l^{-1}) between 1995 and 1998, and have subsequently fluctuated between 6-13mg l^{-1}. Nitrate levels in effluent from Moonfleet Manor Hotel have been steadily decreasing since 1994 (3.6mg l^{-1} annual average in 2001) and there have also been reductions in the nitrate content of the discharge from the Swannery restaurant, although this is a relatively new source (data since 1998). Weymouth STW has remained consistently low (0.2 to 2.5mg l^{-1}) during the period 1990 – 2001. There is no temporal data available for Bridport STW.

The most recent values for nitrate in tidal waters of the Fleet Lagoon are summarised in figure 34. Identical median values for five of the sites (0.22mg l^{-1}) are a reflection of detection limits (0.44mg l^{-1}). As is the case for phosphate, nitrate concentrations are generally highest in West Fleet and reduce toward the eastern end of the Lagoon. This may be as much a result of the poor flushing characteristics of the West Fleet as of point or diffuse source inputs. Waters of Abbotsbury Swannery stand out as having the highest nitrate concentrations, with values up to 17.58mg l^{-1} recorded in 1998. There were also occasional high levels recorded in waters of Langton Hive (up to 9.83mg l^{-1}) which could reflect the transient influence of the relatively high nitrate concentrations in Rodden Stream.
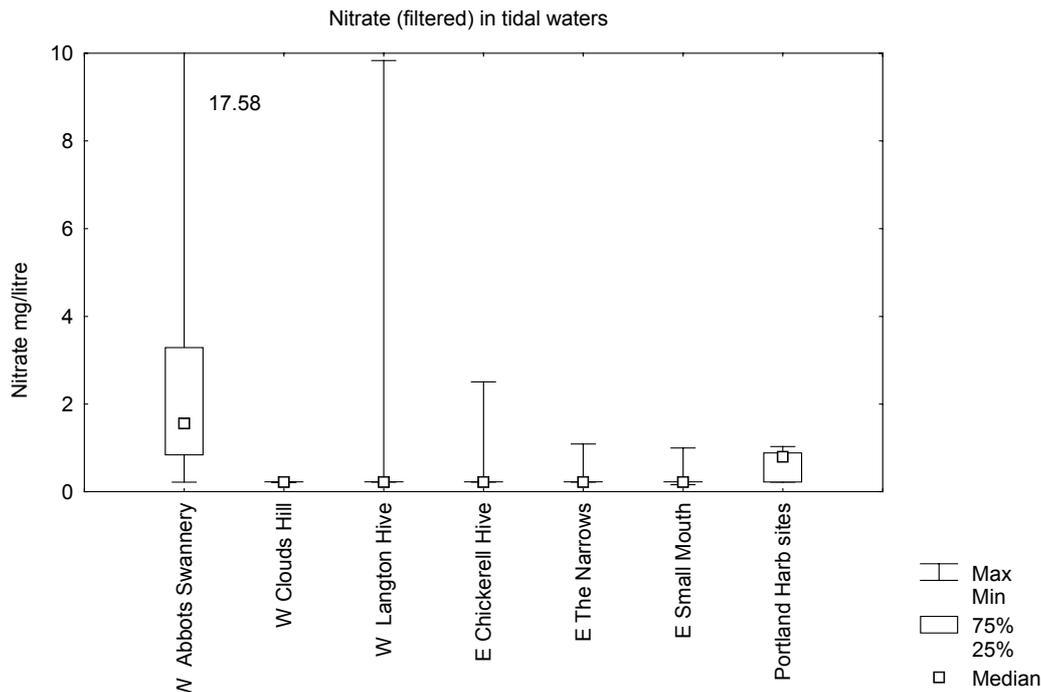
Prefix W or E = West or East Fleet

Figure 34.   Nitrate (mg l^{-1}) in saline waters of the Fleet Lagoon 1998. 'Portland Harbour sites' summarises data for 10 sites. **Data source EA.** NB Where values are below detection limits, half DLs are used

Annual average values for nitrate (expressed as N) in tidal waters are shown in table 21. Almost all mean and max values are above the 0.144mg l^{-1} N, which indicates hypernutrification (dependant on P levels) as suggested in the North Sea Status Report (see above); the exceptions are West Fleet at Clouds Hill and the Narrows and Smallmouth in East Fleet. The maximum value (3.97mg l^{-1} N) recorded in waters at Abbotsbury swannery is well in excess of this figure, as is the maximum at Langton Hive Point (2.21mg l^{-1}). Both mean and maximum values for the swannery site exceed the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations. As is the case for phosphate, the median value for Portland Harbour is relatively high (0.8mg l^{-1}) therefore the Harbour may represent a source of nitrate to East Fleet during the flood tide

Table 21. Nitrate as N: Mean and max values for tidal waters 1998.

Site	Mean annual nitrate (as N) mg l^{-1}	Max value for nitrate (as N) mg l^{-1}
West Fleet At Abbotsbury Swannery	1.103	3.97
West Fleet At Clouds Hill	0.050*	0.05*
West Fleet At Langton Hive Point	0.503	2.22
East Fleet At Chickerell Hive Point	0.127	0.566
East Fleet At The Narrows	0.089	0.246
East Fleet At Small Mouth	0.082	0.226
Portland Harbour 10 Sites	0.178	0.233

* One sample only for Clouds Hill

6.2.3 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 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 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, ammonia toxicity is greater, the higher the temperature and pH and the lower the levels of dissolved oxygen and salinity. Of these three factors, salinity is the least important

Table 22. 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 22), 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 Chesil and the Fleet cSAC/SPA.

Johnston and Gilliland (2000) briefly noted the results of stream water analysis by Avon & Dorset River Authority in 1970. The stream water (Mill Stream below Portesham) was said to be clean at that time, with good oxygen levels, low BOD and no detectable free ammonia, which indicated a lack of animal or plant waste in the water. The stream was therefore considered suitable for rearing trout (the subject of the proposal).   Biological analysis supported this conclusion, though the abundance of invertebrates was not considered sufficient to support a large population of fish.

Saunders-Davies (1995) described nitrogen levels in the tidal waters of the Fleet, both as nitrate and ammonia, to be generally low, but variable.

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 (as N) l^{-1} - ($\equiv$ 26.3 and 37.4mg l^{-1} total ammonia (as 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} ($\equiv$24.6mg total ammonia (N) l^{-1}), and tests on several life stages showed a No Observed Effect Concentration (NOEC) of 0.106mg NH_3 (N) l^{-1} ($\equiv$3.34mg total ammonia (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.

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 at 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 appears to be lowest at intermediate salinities (~10psu), but below this may increase as salinity reduces towards freshwater (Seager et $al.$, 1998, Miller et $al.$, 1990). This may be of relevance, especially in estuaries where DO sags occur at low salinities. In the Mersey, diverse invertebrate populations can survive, and flounder and salmonids can pass through the estuary at a mean unionised ammonia concentration of 0.008 mg NH_3 (N) l^{-1} (Cole et $al,$ 1999).

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 the Fleet Lagoon, 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).

Recent data for ammonia (N) indicates that concentrations in freshwater entering the Fleet (riverine sources) are relatively low (figure 35) in comparison to concentrations in discharges (figure 36). $However,$ $this$ $does$ not $necessarily$ $reflect$ $ammonia$ $loadings.$ Highest values (up to 3.16mg l^{-1}) were recorded in Rodden Stream, where the sampling point is downstream of Langton Herring STW. All other values recorded in 2001 (mean and max) were below 0.7mg l^{-1}, implying that the majority of freshwater sources comply with EQS of 0.78mg l^{-1} total ammonia (N) for freshwater.

Ammonia (N) in freshwaters 2001
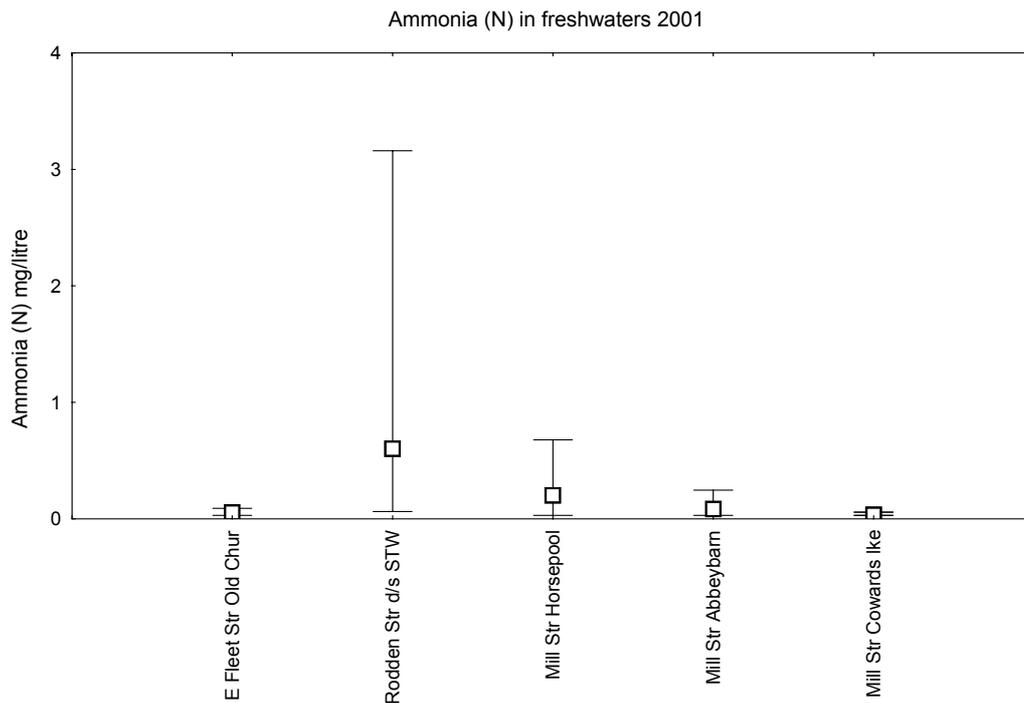
Figure 35. Mean ammonia (N) in freshwater entering the Fleet Lagoon 2001. Data source EA. Data are for 2001. Error bars show min and max concentrations.

Ammonia (N) in discharges
2001
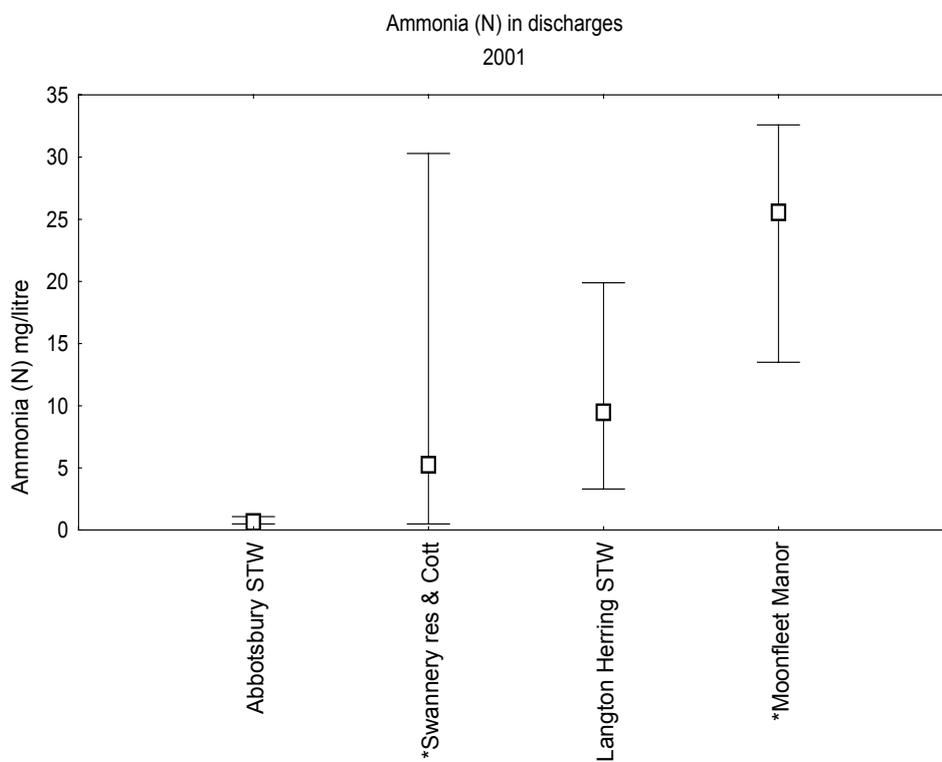
Figure 36 Mean ammonia (N) in discharges to the Fleet Lagoon and tributaries. Data source EA.   Data are for 2001.   Error bars show min and max concentrations.

Mean, minimum and maximum concentrations for ammonia (N) in discharges to the Fleet Lagoon are shown in figure 36 *Again it is stressed that the data are for concentrations only and do not take into the account the volumes discharged, details of DWF are given in table 18.* Ammonia levels in effluent from Moonfleet Manor Hotel STW are extremely high (mean 25.5mg l^{-1}), although the consent is for 40mg l^{-1} ammonia (Johnston and Gilliland, 2000) therefore no compliance failures are reported for this source during 2001. The only compliance failure recorded in 2001 is for the Swannery Restaurant and Cottages discharge at Abbotsbury, where the annual mean was 5.2mg l^{-1} and values peaked at 30.3mg l^{-1}, well above the consent limit of 10 mg l^{-1} ammonia. Ammonia levels in the Langton Herring discharge were moderate to high 9.5 and 19.9 mg l^{-1}, mean and max, respectively.

Information on temporal trends for ammonia in discharges (figure 37) shows that mean annual levels in effluent from Moonfleet Manor and Langton Herring STWs were generally decreasing from 1990 to 1996, although the trend has since reversed. There is also an increase in ammonia concentration in the relatively new Swannery Restaurant and Cottages discharge during the period that it has been operational (1998-2001). Levels in the Abbotsbury STW discharge have remained consistently low.
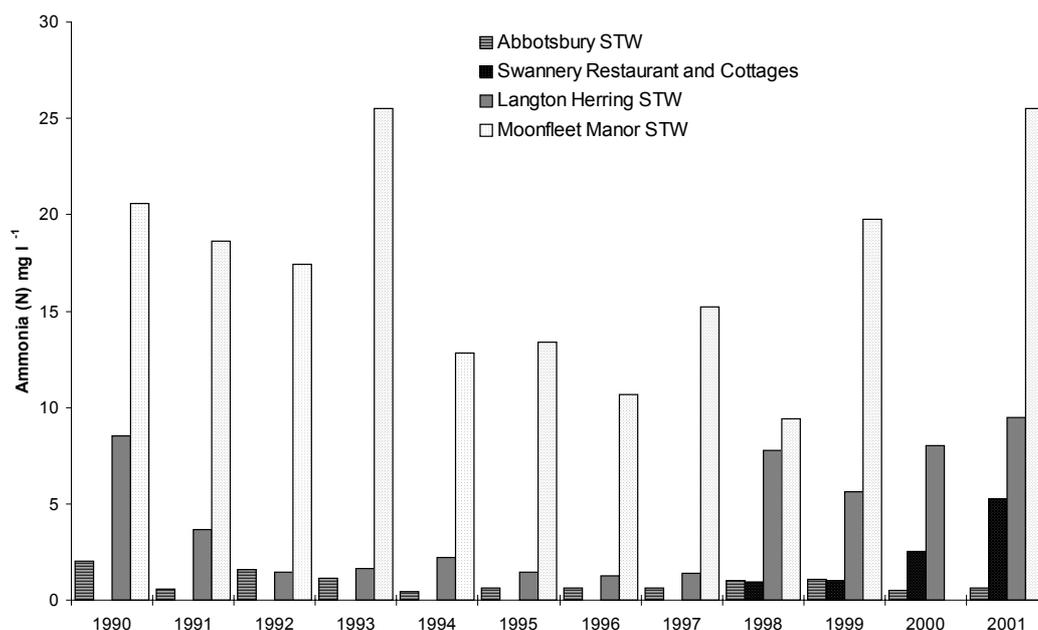
Figure 37. **Temporal trends for mean annual concentrations of ammonia (N) in discharges. Data source:EA**

Ammonia standards for discharges are set on a case-by-case basis although standards are generally higher where discharges are to sensitive areas, therefore, should the Fleet Lagoon be designated, it is possible that ammonia concentrations in point sources will be reduced. The aim of discharge standards is to minimise 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.
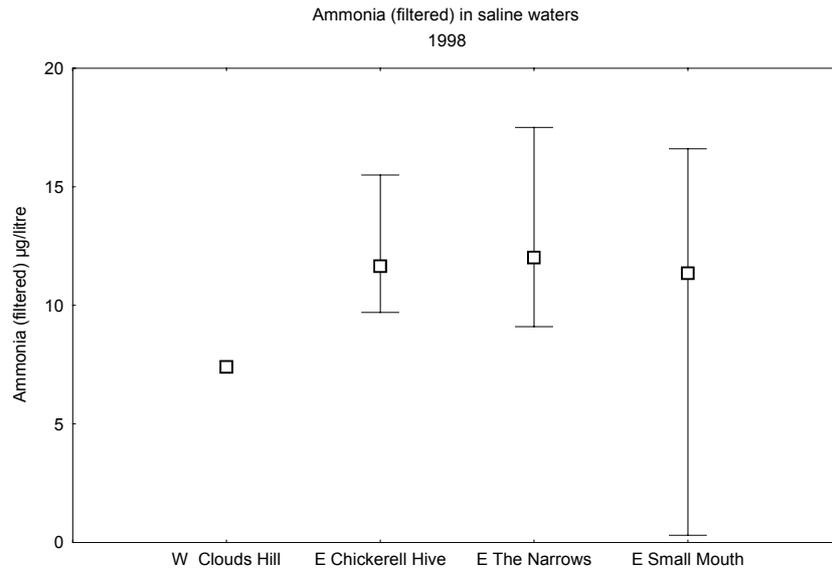
Figure 38. Ammonia (filtered - µg l⁻¹) in tidal waters of the Fleet Lagoon (annual mean). Data are for 1998. Data source EA. Error bars show min and max concentrations where n>1.

The most recent (1998) available values for ammonia in tidal waters of the Fleet Lagoon are shown in figure 38. Mean values were roughly similar for East Fleet sites, and the highest concentration was recorded in the Narrows (17.5µg l⁻¹).

Unfortunately there is little recent information available regarding ammonia concentrations in waters of West Fleet, therefore EA data for 1996-7, summarised in Johnston and Gilliland (2000) are shown below (figure 39).
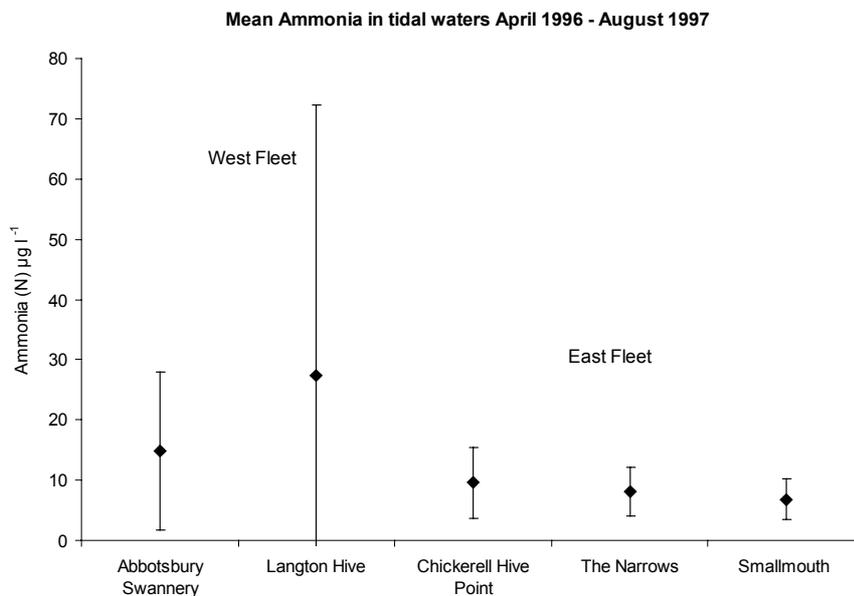
Figure 39. Ammonia (µg l⁻¹) in tidal waters of the Fleet Lagoon (mean 1996-1997). Error bars show standard deviation. Data source: Johnston and Gilliland (2000).

In common with nitrate and orthophosphate, mean ammonia levels were highest in West Fleet at Langton Hive, and decline eastwards toward Small Mouth. The high standard deviation at Langton Hive shows there may be a great deal of variation in ammonia concentrations, and indicates occasional extreme levels at this point. This probably reflects inputs from STWs to the more enclosed areas of the Lagoon, and may also be related to releases from sediment during macroalgal decline (see above).

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, 0.44mg l^{-1} total ammonia (N) relates to about 0.021mg $l^{-1}$ NH_3 (N), which is the proposed EQS. There is not enough detailed information available to establish unequivocally whether this EQS is likely to be exceeded in the Lagoon though estimations from data in figures 38 and 39 suggest exceedences would not be expected.

Several authors have reported an abundance of green macroalgae in the Fleet: e.g. Whittaker (1978) noted thick mats of the green algae *Ulva lactuca* on the mudflats of both East and West Fleet, which was said to die back after releasing its spores around May, and Holmes (1983) described the co-dominance of *Ulva* and *Chaetomorpha* sp. in the Abbotsbury embayment, whereas elsewhere, *Enteromorpha* and *Cladophera* spp. were more common. More recently, Saunders-Davies (1995) reported thick mats of chlorophytes (including *Ullothrix* and *Enteromorpha* spp.) growing below the high tide level along most of the length of the Fleet on the coast side. Thus, there may be great potential for ammonia release from sediments as described above) although more work is needed to establish the contribution from this source.

6.2.4 Chlorophyll *a* and micro-algae

Elevated spring and summer levels of chlorophyll *a* are one of the primary symptoms of increased nutrient inputs to estuarine and coastal 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, notably in restricted exchange environments such as estuaries. 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, 2001c)). 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).

John (1995) recorded two phytoplankton blooms in the Fleet Lagoon during July-August 1995. Samples were taken from eight sites along the length of the Fleet for

analysis of plant pigments and enumeration of plankton species. At the end of July, the first bloom was principally composed of the dinoflagellate *Oxyrrhis sp.* Following the first bloom, which was principally in the West Fleet, elevated numbers of *Oxyrrhis sp.* were observed in samples from further eastwards to Moonfleet and Chickerell.

The second occurred towards the end of August and involved the dinoflagellate *Glenodinium foliaceum*, which caused red colouration of the water. High numbers of *G. foliaceum* were detected in the Clouds Hill sample, and lower numbers recorded from samples taken eastwards in the Fleet as far as Chickerell Hive Point. Neither of these two dinoflagellates were detected in samples from the Narrows or further eastwards on any sampling occasion. Measurements of plant pigments (including chlorophyll *a*) in the samples reflected the concentrations of phytoplankton found in the samples, with the exception of the bloom of *Oxyrrhis sp.*, as this organism is not pigmented. During the second bloom, Mill Stream (Abbey Barn), which enters the Fleet at Abbotsbury Swannery, was also sampled and found to contain high numbers of single-celled blue green algae

Parr *et al* (1998) reported on chlorophyll-a in English and Welsh waters and although data were not available for the Fleet itself, results for sites at some distance to seawards of the Fleet (in Lyme Bay) indicated that marine waters off Chesil Beach were in the lowest category for presence of chlorophyll-a (0-1µg l^{-1}) in both the summer and winter of 1995.

EA data for chlorophyll-a in the Fleet lagoon (1996-7), summarised in Johnston and Gilliland (2000), are shown below (figure 40).
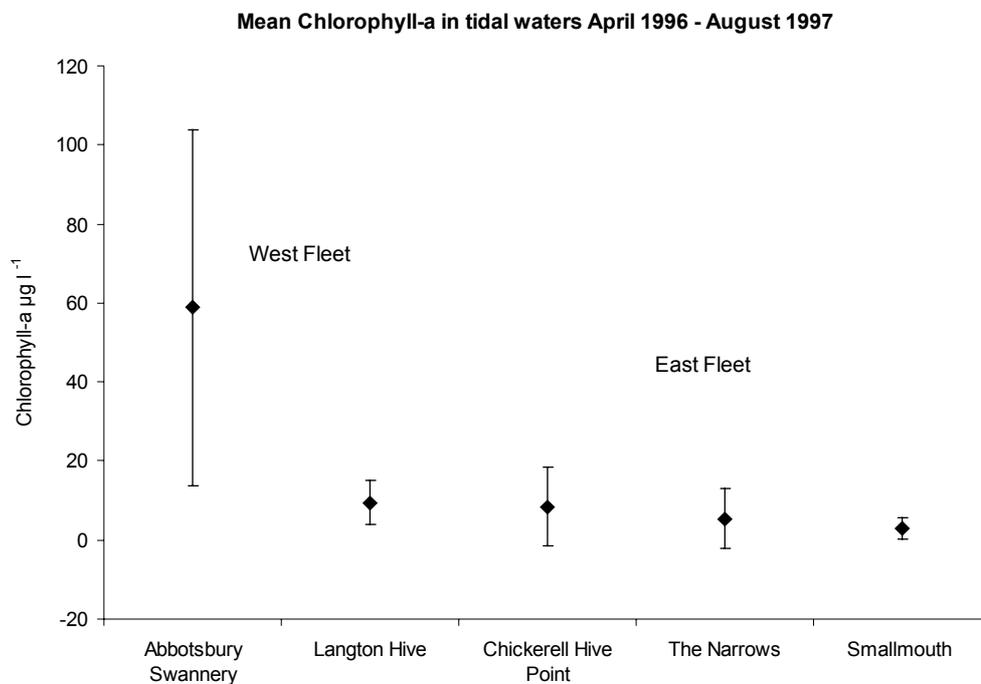

Figure 40. Chlorophyll-a (µg l^{-1}) in tidal waters of the Fleet Lagoon (mean 1996-1997). Error bars show standard deviation. Data source: Johnston and Gilliland (2000).

Generally, concentrations were highest in West Fleet and decreased toward Small Mouth. There was considerable variation in the data for Abbotsbury Swannery (where the most elevated mean level occurred), which suggests that occasionally, extreme levels – which could indicate bloom conditions – were present during this period.

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 'nuisance' blooms which cause concern.

Elevated chlorophyll-a levels at Abbotsbury in August 1997 corresponded with high levels of planktonic algae, identified as the dinoflagellate *Prorocentrum micans*, which caused red discolouration of water in the Abbotsbury area. Lower numbers of *Prorocentrum lima* were detected in samples from Langton Hive and, to a lesser degree, Chickerell Hive. The bloom duration lasted some time - at least four weeks (the interval between sampling occasions). Both the occurrence and duration of the bloom is of possible concern given the associated potential increase in light attenuation and the fact that above a certain level (>100 cells l^{-1}), *P. lima* can cause toxicity (including Diarrhetic Shellfish Poisoning) through the food chain (Johnston and Gilliland, 2000).

Johnston and Gilliland (2000) also reported results of algal sampling in 1998: In July EA samples from Abbotsbury Swannery showed the presence of bloom numbers of the dinoflagellate *Alexandrium Tamarense*, a toxin-producing species which can cause paralytic shellfish poisoning (PSP). The duration of this bloom was unknown, but a sample taken a month later was clear. Very low numbers of *Prorocentrum* sp. (a potential diarrhetic shellfish poison (DSP) producer were observed in a sample taken at Ferrybridge in July.

CEFAS results showed the presence of ASP (amnesic shellfish poison) cells in April and August 1998, and a sample taken by CEFAS in July showed the presence of PSP cells, with numbers lower at Ferrybridge than at Abbotsbury. No toxins were detected in shellfish flesh during 1998, although in 2000, DSP was detected in mussels and oysters from the Lagoon and a Temporary Prohibition Order (July – September, 2000) was placed on shellfish harvesting in the Fleet (EA, 2001) (see above)

Chlorophyll-a in saline waters
1998
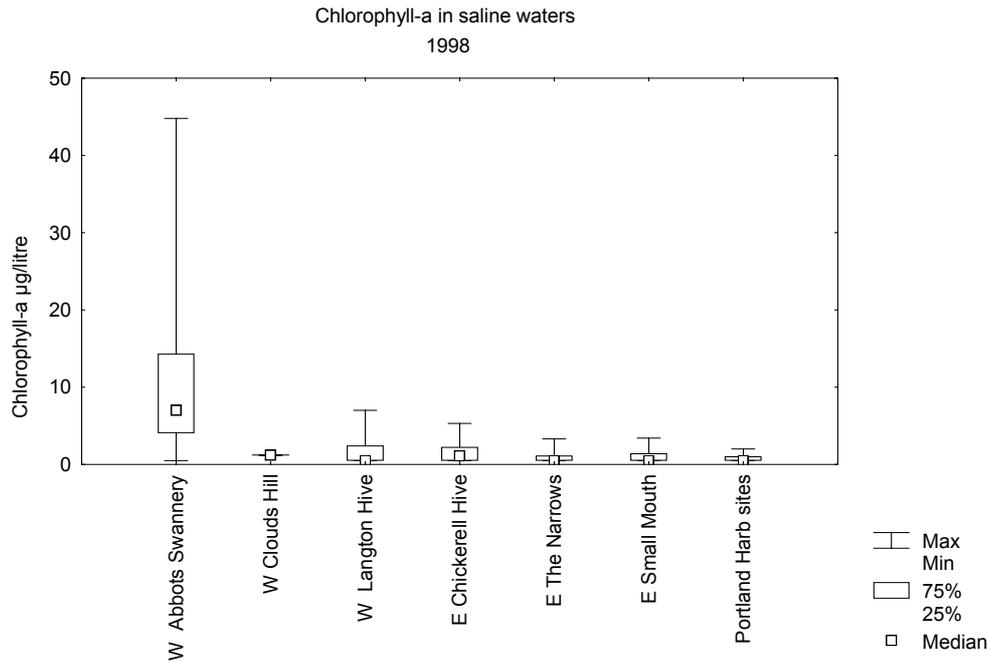
Figure 41. Chlorophyll-a in tidal waters of the Fleet Lagoon 1998. Data source EA. 'Portland Harbour sites' summarises data for 10 sites. NB Where values are below detection limits, half DLs are used

The most recent (complete year) data for chlorophyll *a* concentrations in tidal waters of the Fleet Lagoon (1998) are summarised in figure 41. Generally, data are for January-October, although measurements taken in December are included for some sites. In the UK, an indicator (mean) value for suspected eutrophic conditions, and the DOE criterion is set at 10µg l^{-1} chlorophyll *a* (EA, 2000c; Dong *et al.,* 2000). This value was exceeded at Abbotsbury Swannery (mean and max 12.1 and 44.8µg l^{-1}, respectively) with the maximum concentrations (bloom numbers) occurring in March and July.

Long-term trend analysis is not possible due to insufficient data, however, results for early 2002 (figure 42) again show relatively high levels of chlorophyll-a, indicating a pronounced spring bloom, with the majority of median, mean and maximum values over 10µg l^{-1}. This would seem to reinforce continuing concerns over the persistent nature and frequency of algal blooms in the Fleet.
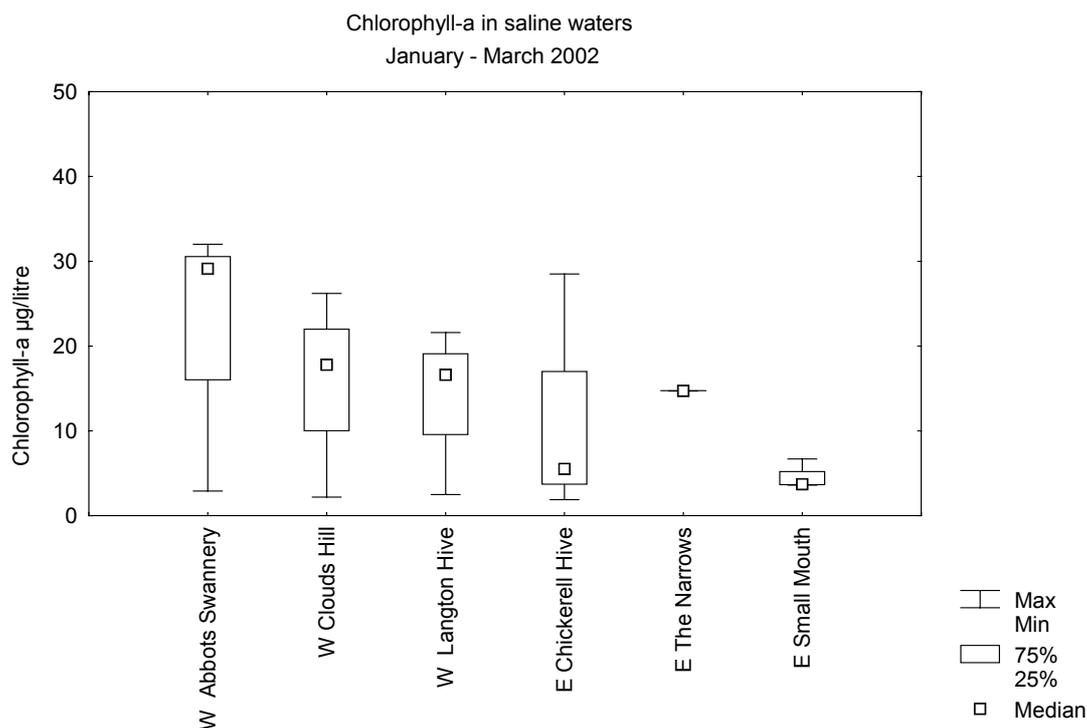
Figure 42. Chlorophyll-a in tidal waters of the Fleet Lagoon, January-March 2002. Data source EA.

Continuous (half-hourly) monitoring of chlorophyll-a in West Fleet, carried out by the EA in 2002, shows distinct diurnal fluctuations (figure 43a). Maximum concentrations occurred at approximately midnight and minima at midday due to variations in photosynthetic activity of the plankton. During plankton blooms, this successional activity can have a 'knock-on' effect resulting in severe depletion and/or supersaturation of dissolved oxygen at various points in the diurnal cycle (see section 6.2.5).

Continuous chlorophyll monitoring in West Fleet is also providing information on longer time-scale variations in phytoplankton numbers and activity, as shown in figure 43b[1]. The mean concentration of chlorophyll between June and October 2002 was $14.3\mu g$ l^{-1} – again, above the UK indicator value for suspected eutrophic conditions. Peaks in chlorophyll, indicating phytoplankton blooms, can be clearly seen (notably during September).

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[1] The chlorophyll readings should be regarded as indicative rather than absolute due to uncertainties introduced by variations in cell structure, particle size and organism type, and the effects of temperature and photosynthetic activity on fluorescence. The probe is equipped with a mechanical wiper that periodically cleans the optical face, thereby making it suitable for long-term deployment in heavily fouling environments such as the Fleet, although occasional spurious results occur (e.g. $663\mu g$ l^{-1} recorded in early August) which may be due to the presence of filamentous algae (Richard Acornley, EA Wessex Region, pers. comm.).
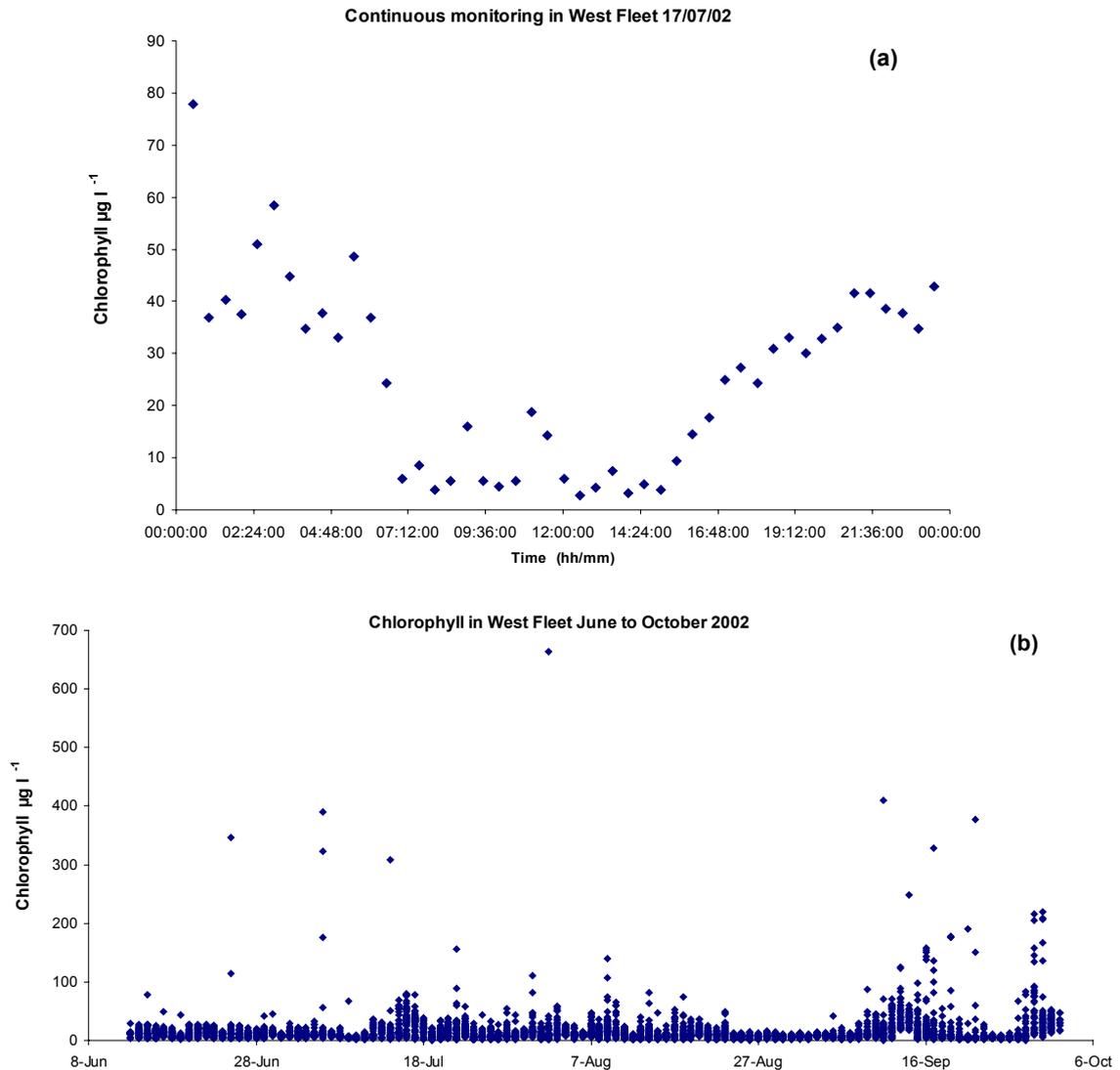
Figure 43 Continuous chlorophyll monitoring data from the West Fleet. (a) 17/07/02. (b) June to October, 2002. Data source EA.

Recent (2001) values for chlorophyll *a* in freshwaters entering the Lagoon are shown in figure 44. Median concentrations were relatively low (2.5 – 5.9µg l^{-1}) although very high levels, suggesting algal blooms, were recorded in Mill Stream (Horsepool) during August (up to 104µg l^{-1}) and in Rodden Stream in October (up to 88µg l^{-1}). Both streams receive effluent from STWs (Abbotsbury and Langton Herring, respectively).

There are few consistent temporal trends discernable from the chlorophyll-a data for fresh waters (for the period from 1997-2002). Median and mean annual concentrations at Abbey Barn in the Mill Stream have fallen, as has the annual median in East Fleet Stream, whilst mean, median and maximum values have risen steadily at Horsepool in the Mill Stream. Elevated chlorophyll-a concentrations in freshwater sources arise as a result of nutrient enrichment and are indicative of the presence of large numbers of freshwater phytoplankton species.
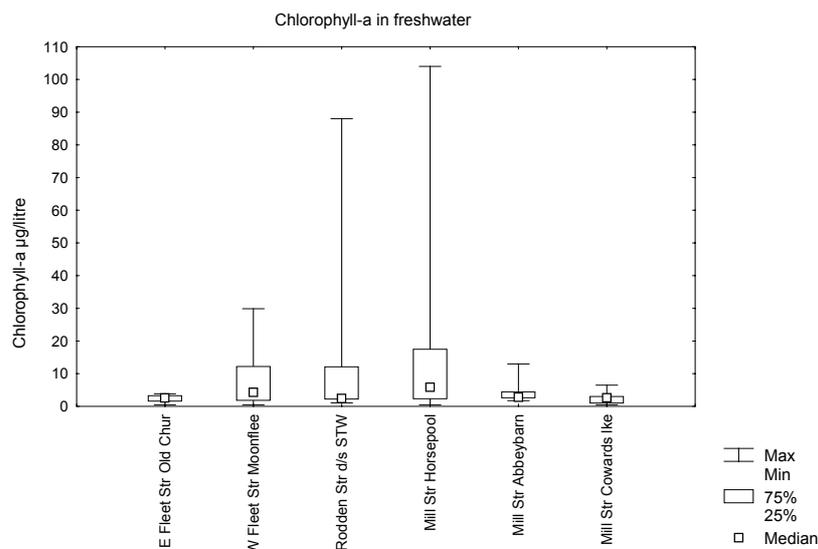
Figure 44. Chlorophyll *a* in freshwaters entering the Fleet Lagoon. Data source EA. Data are for 2001.

Potential consequences of high freshwater phytoplankton densities, for the Fleet, would be due to die-off and release of DOC if these species entered saline waters in any great numbers.

Overall, data for chlorophyll *a* indicate regular phytoplankton blooms in the Fleet lagoon which may be increasing in intensity, and are almost certainly related to elevated nitrate and phosphate concentrations.

6.2.5 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 23 shows recommended EQS values for saline waters derived from the review of Nixon *et al.,* (1995).

Table 23. Recommended EQSs for dissolved oxygen in saline waters (from Nixon *et al.*, 1995)

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

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 24) 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 24. 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, particularly during bloom die-offs, when microbial decomposition of algal cells can lead to a further increase in oxygen demand, thus compounding the problems of DO depletion. The resultant low DO concentrations in the early morning can lead to the death of invertebrates and fish (EA, 2001b).

Johnston and Gilliland (2000) have reviewed DO monitoring work carried out in the Fleet during 1998. Continuous monitors sited in the Abbotsbury embayment and in the channel off Chickerell Hive Point gathered data on late summer diurnal variations in DO for 15 days in October. Overall, oxygen saturation varied between <80% and well over 100% (supersaturation). For three of the days, diurnal variations were barely discernible, with DO consistently around 80% saturation, presumably corresponding to tidal influx of higher salinity water. Dissolved oxygen at Chickerell did not vary demonstrably with the tidal cycle, but was dominated by diurnal variation, with higher levels (>100% saturation) occurring during the middle of the day (photosynthesis), and lower levels of 70-80% during the night (respiration). Whittaker (1978) has reported DO values of up to 270% saturation in the Fleet in spring and summer.   The extremely high levels were thought to result from photosynthetic activity of *Zostera,* although supersaturation (>150%) was also recorded during winter when *Zostera* had died back. Whittaker suggested that this could be due to the greater oxygen-carrying capacity of colder water.

There is little information in the literature regarding DO in freshwaters feeding the Fleet Lagoon. Avon & Dorset River Authority (1970) reported DO in Mill Stream (Horsepool), below Portesham, to be 11mg l^{-1} and commented that the water was

clean, with good oxygen levels. Recent EA data for DO in freshwater sources are summarised in figure 45. Median values are relatively high at all sites, although minimum concentrations of 5.4 and 5.98mg l^{-1} indicate occasional depletion in Rodden Stream and Mill Stream (Horsepool). The data results from daytime spot sampling, therefore does not encompass diurnal variation.

Dissolved Oxygen in freshwaters
2001
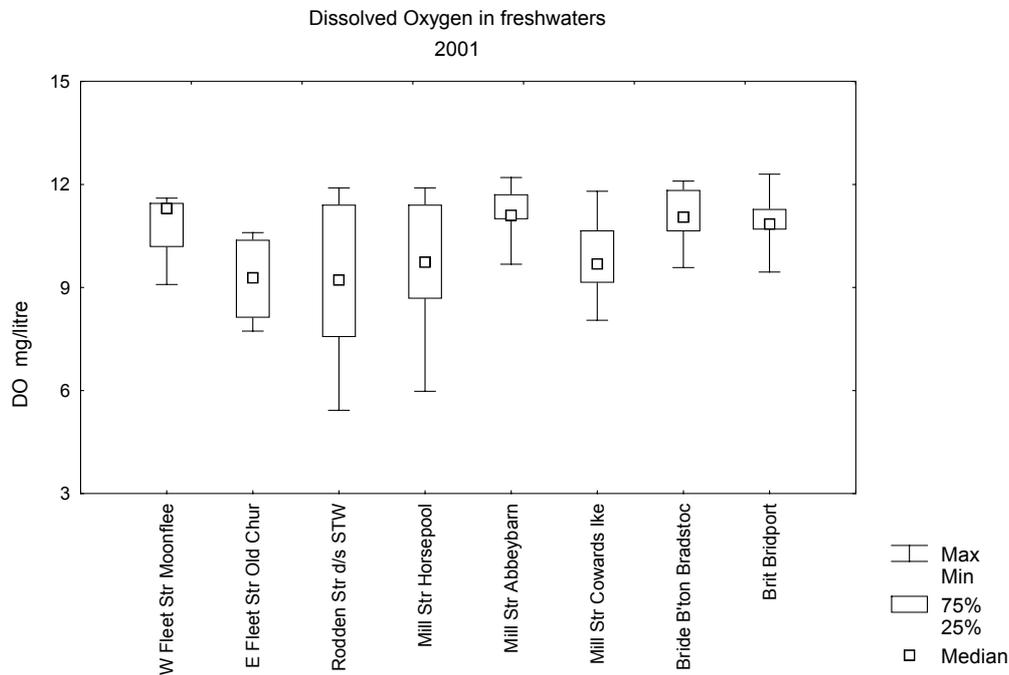
**Figure 45. Dissolved oxygen in freshwaters entering the Fleet Lagoon and Lyme Bay (West Chesil Bank) 2001. Data source: EA**


Dissolved Oxygen in saline waters
1998
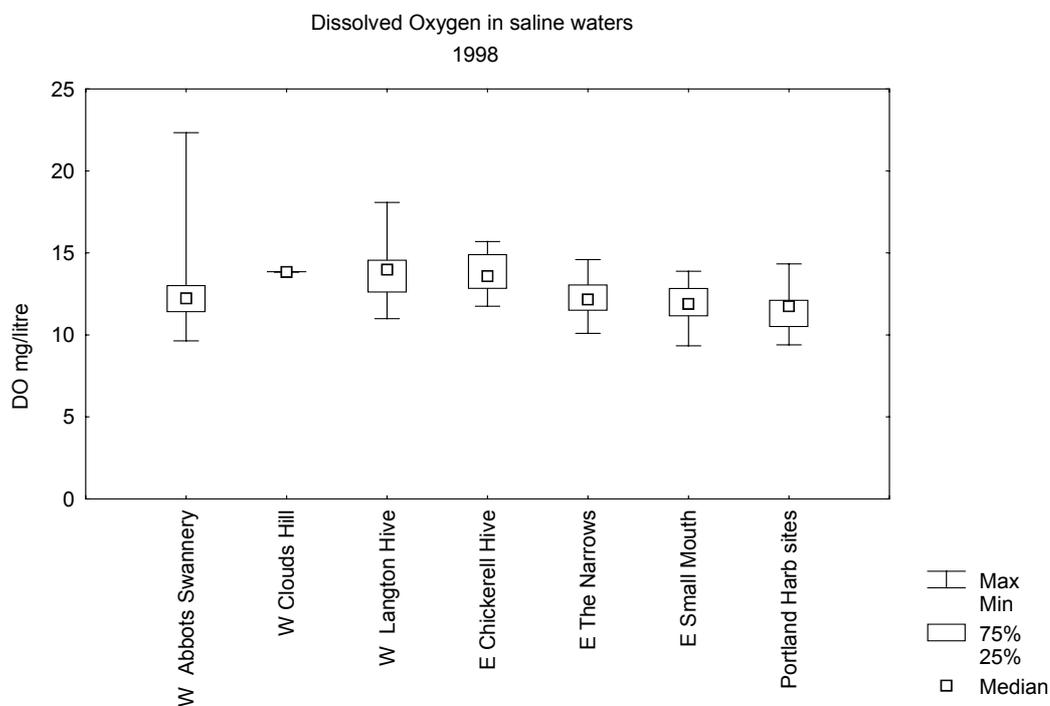
Figure 46. Dissolved oxygen in tidal waters of the Fleet Lagoon. Data source EA. Data are for 1998. W or E prefix signifies West or East Fleet

The most recent (complete year) data for DO in tidal waters of the Fleet (1998), collected by the Agency, are shown in figure 46. Median values are high at all sites (11.7 to 13.9mg l^{-1}), and above the 5mg l^{-1} (median) recommended EQS value for saltwater life and for migratory fish (see table 23). Minimum values during the year do not fall below this figure, although these are a result of spot sampling during the hours of daylight.

High values, indicating oxygen supersaturation, occurred in West Fleet notably at Abbotsbury where values of up to 22.3mg l^{-1} (220% saturation) were recorded during October 1998. There is a general temporal trend toward decreasing concentrations of DO at West Fleet sites although this data are for a limited period only (1996-2002) and does not reflect minimum concentrations which occur during the hours of darkness.

However, continuous monitoring carried out in West Fleet during late summer 2001 by the EA, for the purpose of determining the most suitable location for a permanent monitoring station, confirmed that severe oxygen depletion in the Fleet does occur in the night/early morning. Daytime DO exceeded 200% saturation on several occasions, generally in the late afternoon (1530 – 1800) and minimum saturation (down to 26.8%) was recorded in early daytime hours (0300 – 0630).

During this time, pH exceeded 9.0. Levels of planktonic algae were low during the monitoring period (chlorophyll-a values <5 μg l^{-1}) so the high DO and pH was attributed to photosynthetic activity by the extensive growth of *Ruppia* and *Chaetomorpha*. DO fluctuations were greatest during neap tides.

6.2.6 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 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 value 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'.

Parr *et al* (1998) described turbidity in English and Welsh waters. Data were not available for the Fleet itself but results for sites at some distance to seawards of the Fleet indicated that marine waters off Chesil Beach were in the highest category for light transmission (75-100%) in summer 1995, and the second highest (50-75%) in winter 1995.

Johnston and Gilliland (2000) reported on monitoring work carried out in the Fleet at Abbotsbury and Chickerell in October 1998. Continuous monitoring in the Abbotsbury embayment and in the channel off Chickerell Hive Point showed that turbidity was generally fairly constant and at a low level. A few groups of peaks of up to 600NTU (Nephelometric Turbidity Units) were recorded which corresponded with disruptions to the tidal and diurnal variations in salinity and DO. At the same time as some of the peaks in turbidity, salinity was reduced, and the diurnal variation in DO levels was less pronounced. It was suggested that these peaks might have been associated with rainfall events, although unfortunately no rainfall data was recorded.
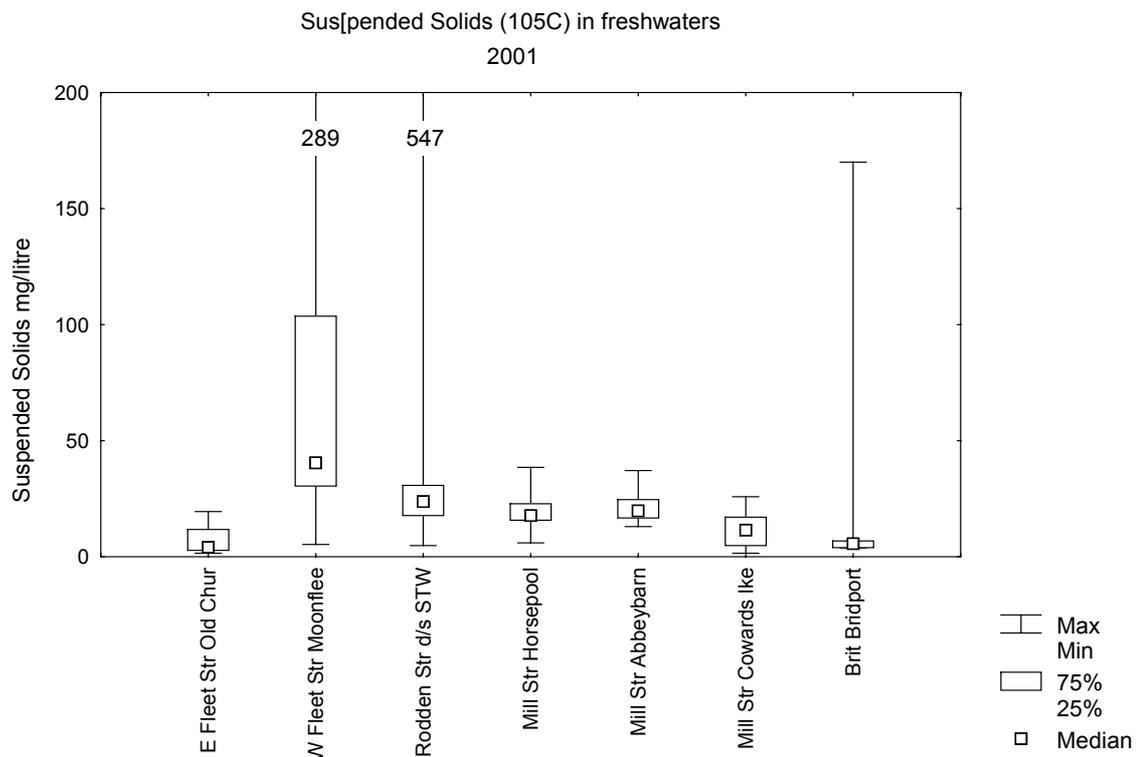
Figure 47.   Mean levels of suspended solids @ 105°C in freshwaters entering the Fleet Lagoon and Lyme Bay (West Chesil Bank).   Data source EA. Data are for 2001.

The principal method used by the EA for quantifying turbidity in the Fleet lagoon is suspended solids (at 105°C) (units: mg l^{-1}). Figure 47 shows suspended solid levels in freshwater sources to the Fleet in 2001. Median values for the year are relatively high (up to 40 mg l^{-1} in West Fleet Stream) and therefore may constitute a significant source of suspended matter. Maximum values occurred in West Fleet Stream and Rodden Stream in February and October, respectively, and may, in the Rodden Stream, be partly attributable to STW inputs (see below).
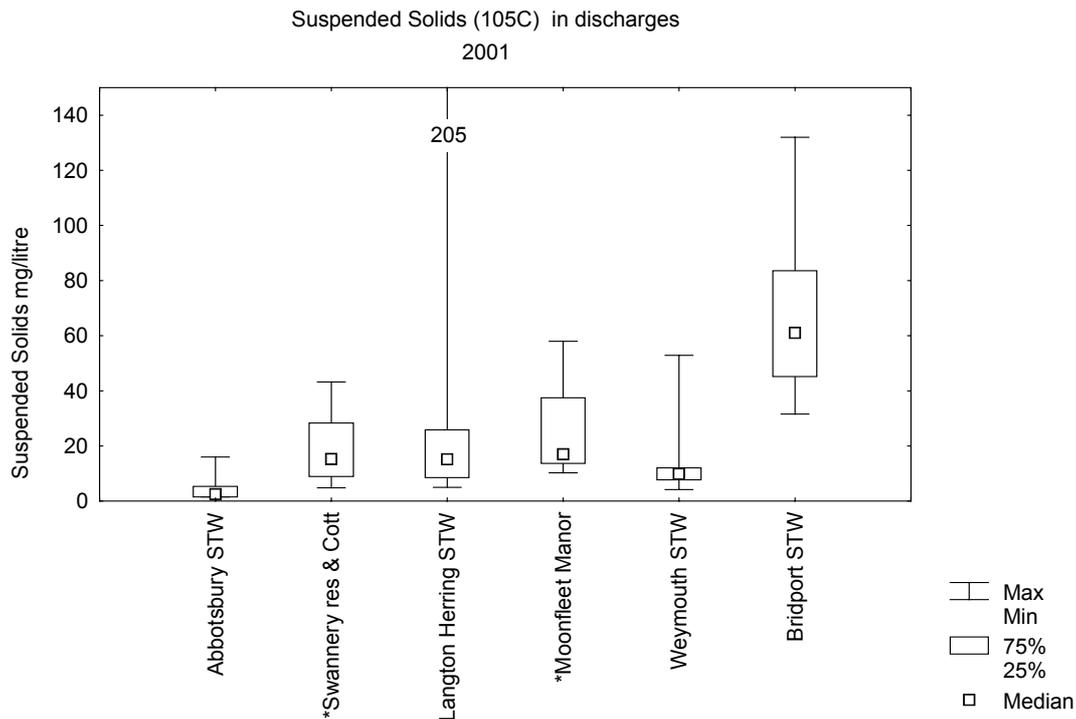
* denotes non-water company discharges

Figure 48. Mean levels of suspended solids @ 105°C in discharges to the Fleet, tributaries and seaward of Chesil Bank, 2001. Data source EA. Data are for 2001.

For discharges (figure 48), recent data show that the highest annual median value occurred in effluent from Bridport STW, which discharges via a long sea outfall to Lyme Bay. Median values for discharges to the Fleet and tributaries are lower, and in the range 2.5 to 17mg l^{-1} (Abbotsbury and Moonfleet Manor Hotel STWs, respectively). However, Moonfleet Manor Hotel did not exceed its consent of 60mg l^{-1} suspended solids during the year. The highest individual values occurred in the Langton Herring STW discharge (up to 205mg l^{-1}) which, with an estimated DWF of <100 m^3 day^{-1} could therefore represent a major, if transient, source of solids, median values are generally an order of magnitude lower. Annual averages for suspended solids in this discharge have increased over recent years (since 1999), as have those for the Swannery Restaurant (since records began in 1998). There were significant reductions (1990-1999) in levels of solids in the Moonfleet Manor discharge, although there have been small increases annually over the last three years, whilst data for Abbotsbury STW show that suspended solid concentrations have remained low since 1992.

Suspended Solids (105C) in saline waters
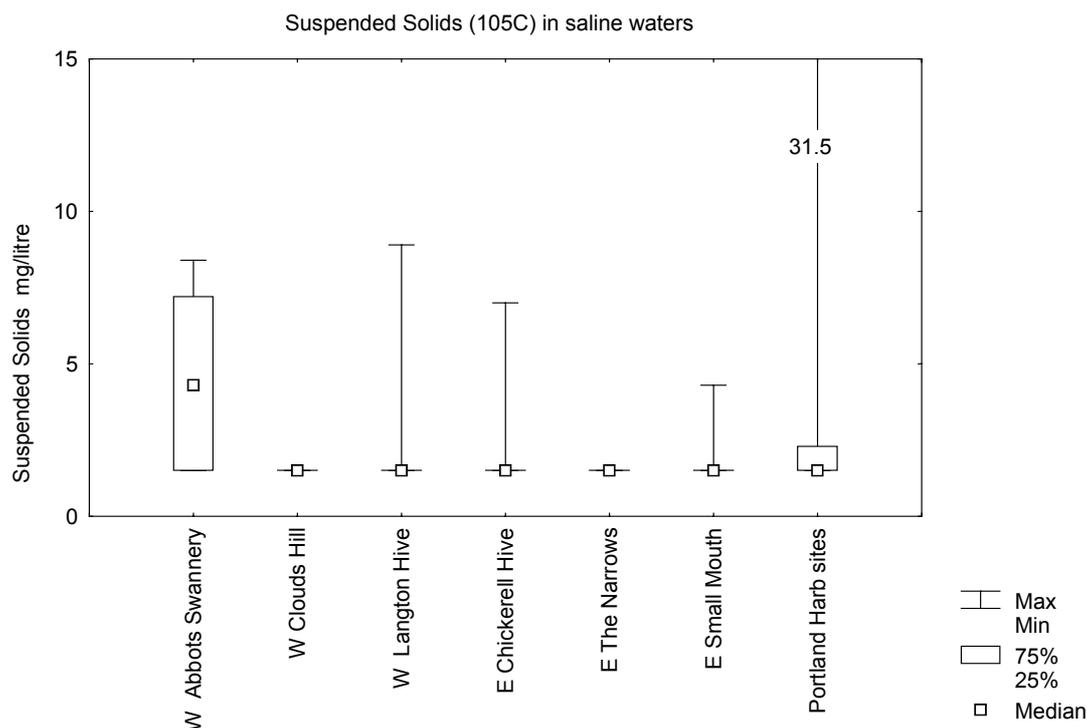
W or E prefix signifies West or East Fleet

Figure 49. Suspended solids @ 105°C in tidal waters of the Fleet Lagoon. Data source EA. Data are for 1998 except Small Mouth (1997). NB Where values are below detection limits, half DLs are used

The most recent (complete year) available data for suspended solids in tidal waters are summarised in figure 49. Values are generally low, and median concentrations of 1.5mg l^{-1} are a reflection of detection limits (3mg l^{-1}); the exception is Abbotsbury Swannery, West Fleet, where the annual median was 4.3mg l^{-1}, and the data show a summer (June/July) maxima. The peaks in suspended solid levels at Langton Hive and Chickerell Hive occurred in March and October (respectively). Waters of Portland Harbour appear to be relatively clear, with low levels of suspended solids. The maximum concentration recorded during the year (31.5mg l^{-1}) occurred at a site close to Small Mouth, presumably during tidal exchange.

The more elevated values recorded in the lagoon may be associated with land run-off and increased stream inputs during high rainfall events. There might also be some sediment resuspension at East Fleet sites during tidal flushing, although in West Fleet, there is little water exchange, and consequently, suspended solids would be expected to settle out. Suspended solids in this area may therefore contain a large algal component. Generally, turbidity does not appear to be a problem in the lagoon, and adverse effects to biota would not be anticipated.

Compared to concentrations of suspended solids in UK estuaries, values for the Fleet are very low. To put these levels into some perspective, Cole *et al* (1998) cited typical mean annual values for suspended solids (105°C) around the English and Welsh coast as 1-110mg l^{-1}, and suggested that anything >100 mg l^{-1} could be considered high.

7. SEDIMENT STATUS AND QUALITY GUIDELINES

At present there are no environmental quality standards for sediments applicable in the UK. However, 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 (discussed in the footnote below)[1] and the use of non-indigenous test species in 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 Chesil and Fleet cSAC/SPA.

7.1 Metals

In general, levels of metal contamination in estuarine sediments decrease significantly towards offshore sites, partly due to distance from riverine and land-based 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 advisable to compensate for such granulometric and geochemical effects, to allow meaningful comparison of contamination levels (Langston *et al.*, 1999; MPMMG, 1998).

The only available regional information for sediments in the EA data set is a site in Portland Harbour, close to the mouth of the Fleet (Portland Harbour One in figure 5), sampled in May 1997. For some metals (Cr, Cu, Hg), solids have been sieved at

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[1]The need to standardise/normalise sediment measurements 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 oxyhydroxide 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.

63μm (digest not specified). For As, Pb and Zn (hydrogen fluoride digest) there is no indication of grain size but as they appear to be the same samples as the above suite of metals, they too are assumed to represent the <63μm fraction.

In order to demonstrate the likelihood of biological effects at the EA Portland Harbour site, sediment metal concentrations are summarised in figures 50 and 51 in relation to the interim sediment guideline criteria for each metal.

As indicated for Hg in figure 50, the lower horizontal line for each element represents the lower threshold or interim marine sediment quality guideline - ISQG (no harm to biota is predicted). The upper, dashed line represents the probable effects level - PEL (sediment concentrations where harmful effects might be expected). In the zone between these thresholds (between ISQG's and PEL's) - where the median and majority of values for Hg sit at the Portland site - sediment concentrations may not cause harm though effects cannot be excluded.
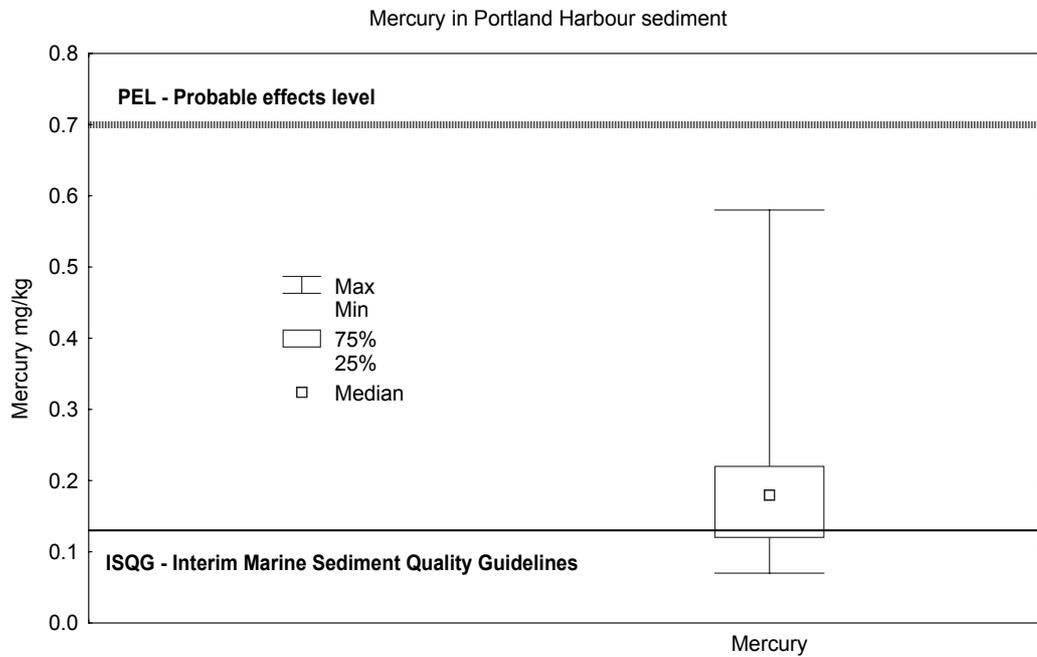
Figure 50. Mercury in sediment, Portland Harbour near the entrance to the Fleet at Small Mouth (May, 1997). Classification based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Above PEL = effects expected; Below ISQG = no harm to the environment expected. Between ISQG and PEL = possible effects cannot be excluded. (Data source EA)

Metals in Portland Harbour sediment
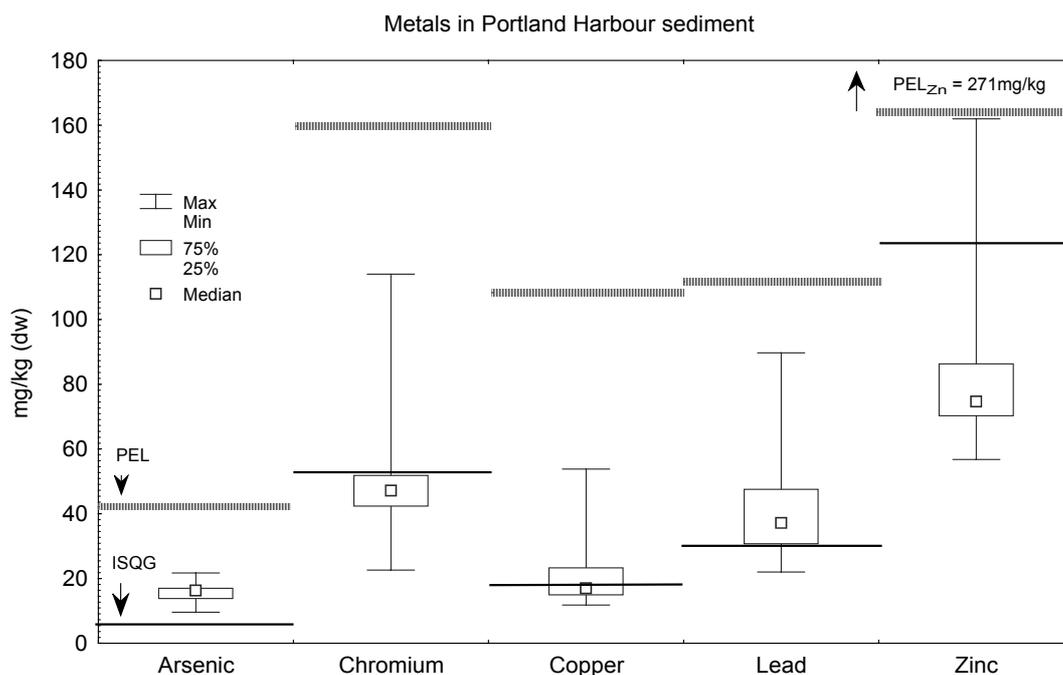
Figure 51. Arsenic, chromium, copper, lead and zinc in sediment, Portland Harbour near the entrance to the Fleet at Small Mouth (May 1997). Classification based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Above PEL = effects expected; Below ISQG = no harm to the environment expected. Between ISQG and PEL = possible effects cannot be excluded. (Data source EA)

The trend in concentrations is very similar for arsenic, copper and lead (figure 51). For chromium and zinc the majority of values are below the ISQG. There are no cases where concentrations exceed the PEL and therefore toxicological significance for most metals at the Portland site, near Small Mouth, appears quite low.

Using slightly more extensive data from Nunny and Smith (1995) it is possible to demonstrate, in map form, the likelihood of biological effects in relation to ISQGs and PELs. Sediments from six sites in the Fleet and one each from Portland and West Bay Harbours have been used to classify sites according to interim sediment guideline criteria for each metal (figures 52, 53 and 54). Green bars denote sites where no harm to biota is predicted (below ISQG's) and grey bars denote sites where effects cannot be excluded (between ISQG's and PEL's). Red bars would be used to represent sediment concentrations where harmful effects might be expected (above PEL's), though for the metals assessed here there are no such examples.

Chromium levels in sediments fall below the ISQG value at all sites (green bars in figure 52). Concentrations are broadly similar at all sites, but with some slight enrichment at inner Fleet sites. None of the sediment Cr values is anticipated to result in biological effects.

104
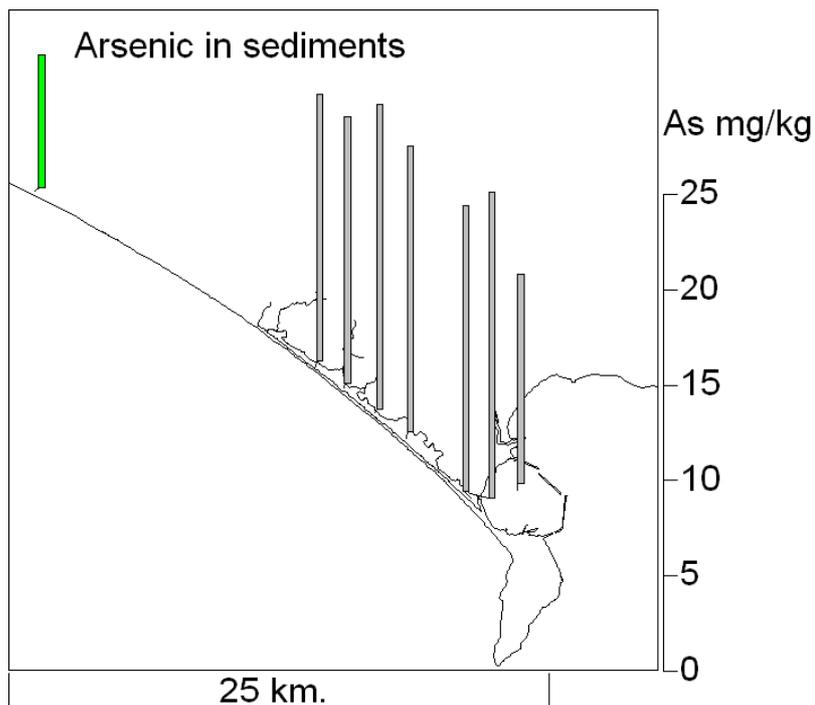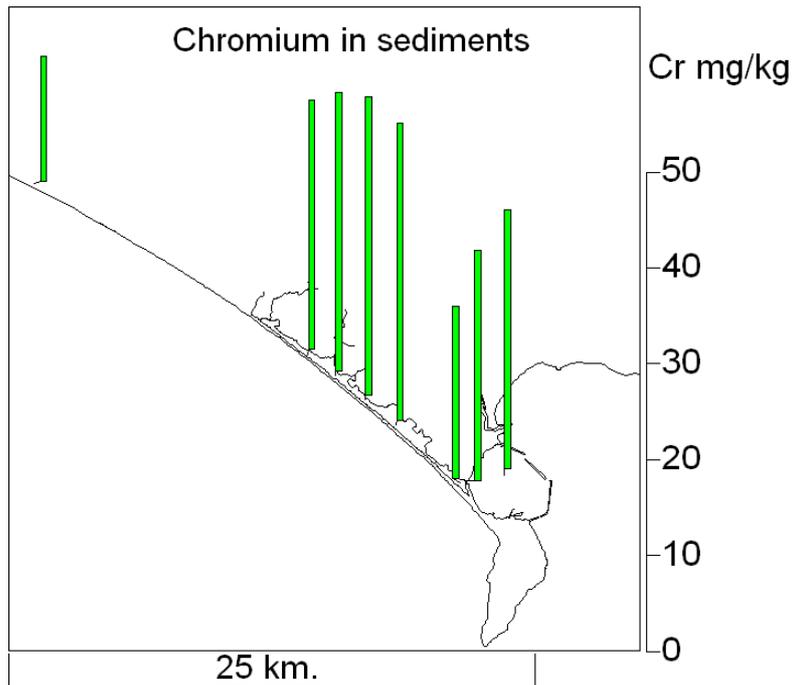
Figure 52. Chromium and arsenic in sediment. Classification of Fleet, Portland and West Bay sediments based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source: Nunny and Smith, 1995).
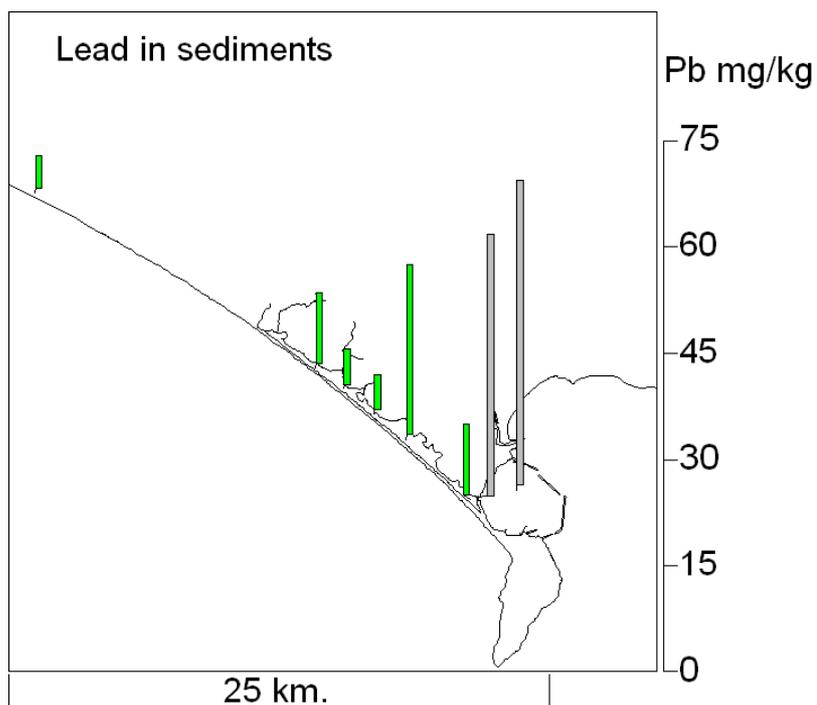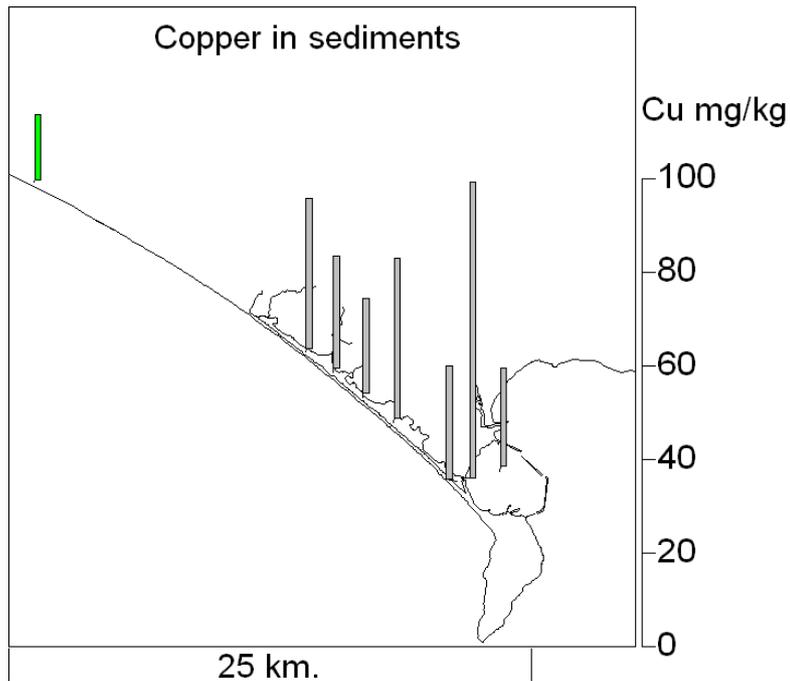
Figure 53. Copper and lead in sediment. Classification of Fleet, Portland and West Bay sediments based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source: Nunny and Smith, 1995)
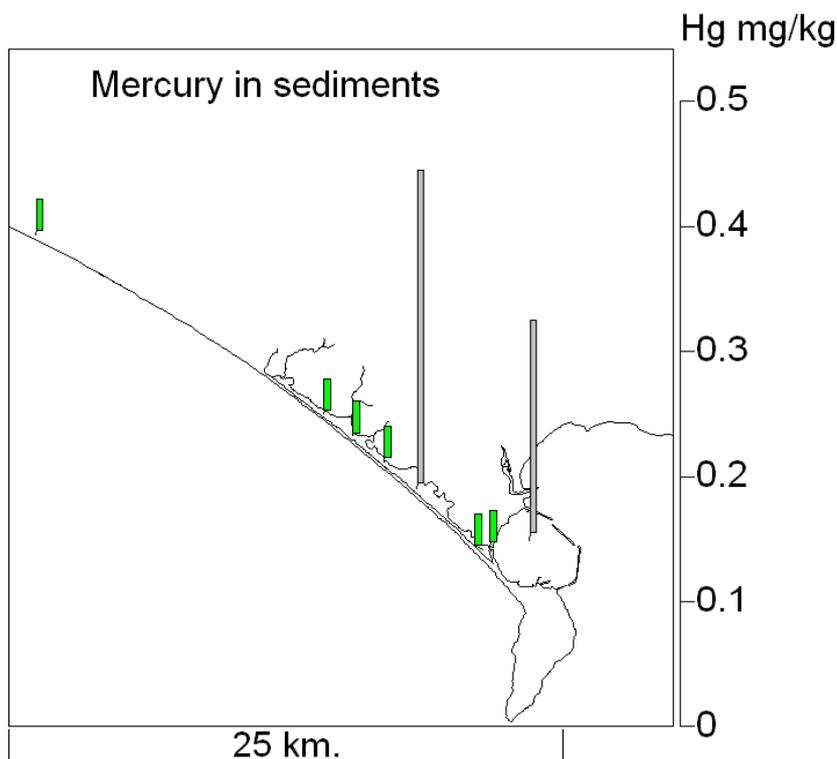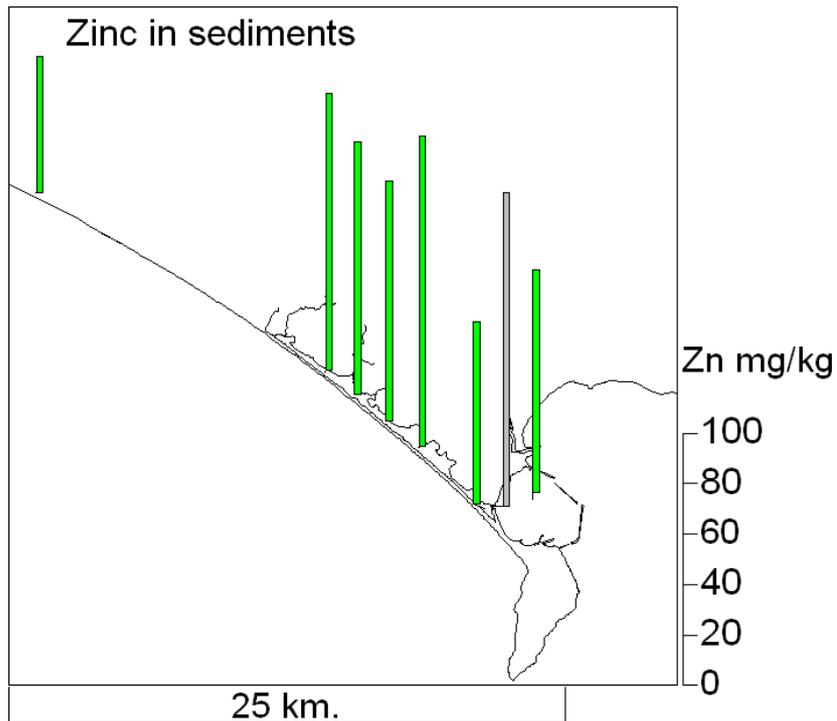
Figure 54. Zinc and mercury in sediment. Classification of Fleet, Portland and West Bay sediments based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source: Nunny and Smith, 1995)

The distribution gradient for As is virtually identical to that for Cr, though at sites in the Fleet and in Portland Harbour concentrations marginally exceed the ISQG (grey bars in figure 52 – biological effects are not expected but cannot be excluded). Precisely the same classification of sites is applicable to Cu, though it should be noted that Cu levels are highest at Ferrybridge (Figure 53).

Enrichment above the ISQG is also seen for Zn and Pb near the mouth of the Fleet (and, for Pb, in Portland Harbour), whilst at sites in the inner Fleet and West Bay sediment concentrations of both metals are below guideline values (green bars figures 53 and 54).

Mercury concentrations are generally very low throughout most of the Fleet with the exception of one site near east Fleet and also Portland Harbour, where concentrations exceed ISQG values (grey bars in figure 54).

It is emphasized that these are interim guideline values only, set according to the CCME criteria and, for some contaminants, may tend to be overcautious. Where sediments have exceeded the ISQG this is usually by a relatively small margin, rather than by orders of magnitude. Effects, if they occur, would be chronic rather than acute.

Although these sediment classifications do not themselves constitute direct evidence for effects (or the absence of effects), the results nevertheless indicate that diffuse source contamination from sediment is unlikely to be of significance throughout much of the European Marine site.

However, it would be useful to extend the data on sediments - in particular sediments within the Fleet – both to establish the current status, and as baselines to evaluate temporal trends in the future. This is seen as a particularly important issue in terms of meeting standstill requirements 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. Future studies should attempt to identify the causes of localised anomalies for Cu, Pb Zn and Hg in sediments and to characterize their bioavailability.

7.2 Organic contaminants in sediments – PAHs, PCBs, pesticides, herbicides

There are data for individual and total PAHs for just two sediment sites in the Fleet (Nunny and Smith, 1995). These are at the subtidal survey sites 1C and 4C described in Dyrynda and Cleator (1995). Concentrations are plotted in relation to guideline values and probable effects levels in figure 55. All PAHs exhibit similar trends, with slightly higher values at the inner site. Consistently these plots indicate values in the Fleet are above the ISQG but below the PEL (denoted as grey bars) i.e. sediment concentrations may not cause harm though effects cannot be excluded. This should be verified by further sampling of Fleet sediments, particularly with a view to identifying sources and characterising bioavailability.
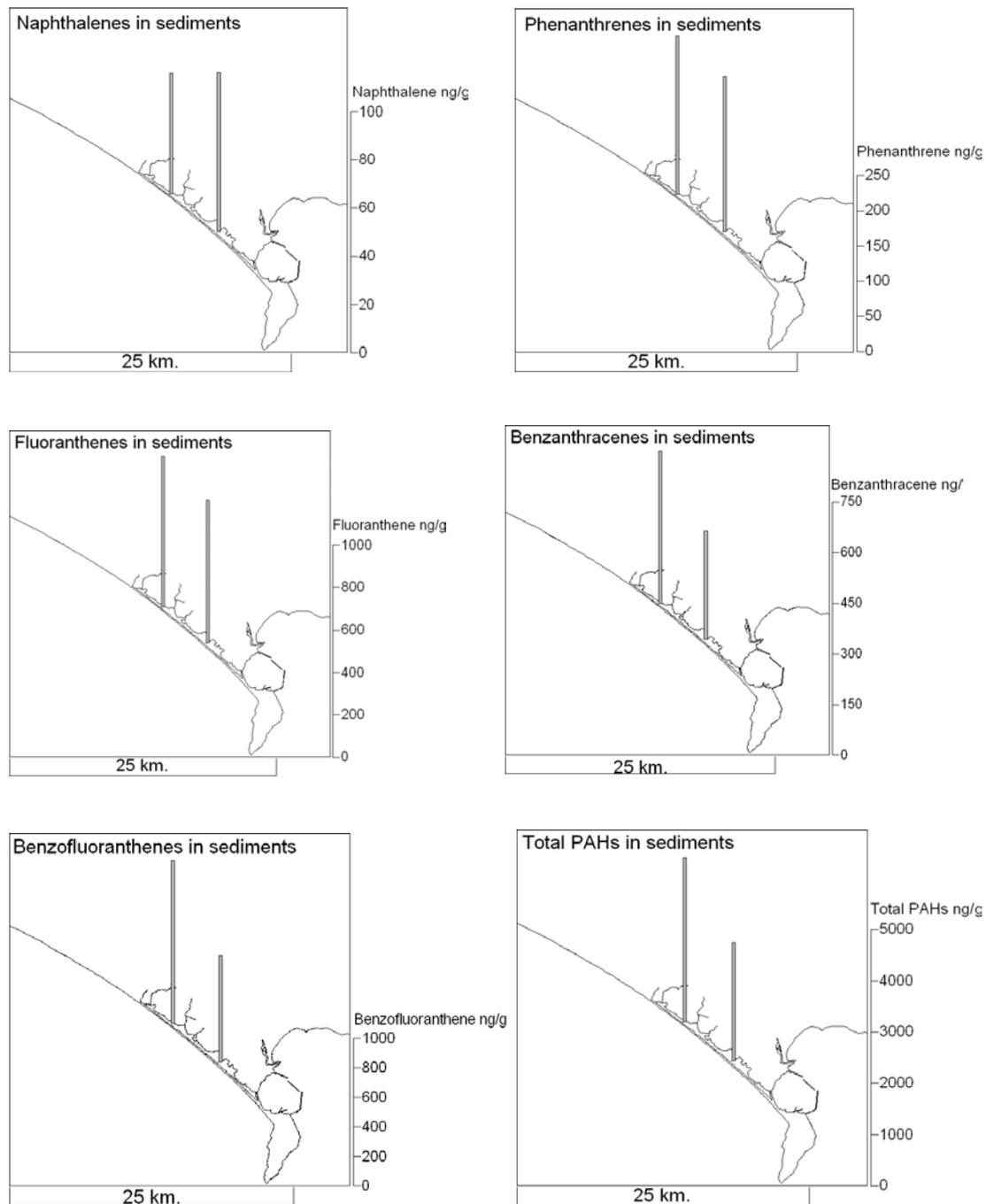
Figure 55.   Individual PAHs and total PAHs.   Classification of the Fleet sediments based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source: Nunny and Smith, 1995)

The majority of data for other organic contaminants in sediments are restricted to the one site in Portland Harbour, close to the entrance to the Fleet at Small Mouth, sampled by the EA in 1997. Thus, there is not enough information to map trends in the distributions of these compounds, or to give more than a rudimentary evaluation of the impact of these diffuse sources. Concentrations of most organic contaminants were, in fact, predominantly below detection limits with the exception of γ-HCH, trifluralin and, very occasionally, DDT and pentachlorophenol. Results in the data base accompanied with '<' sign have been halved in the present evaluation to provide at least some indication of environmental levels (table 25).

The average concentration for γ-HCH, the one compound regularly above detection limits, was above the probable effects level for sediments (table 24).

Even though the sum of DDT isomers appears to be above the ISQG guideline and just below the PEL this is probably artifactual, due to the fact that the majority of values are below detection limits which are themselves sometimes above the guideline value. None of the other compounds exceeded PEL values. Dieldrin exceeds the ISQG though, again, this is due to detection limits being higher than guidelines.

Table 25. Organic contaminants in sediments of Portland Harbour, near the entrance to the Fleet at Small Mouth (May 1997), compared with interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) from CCME, (1999). All units μg kg^{-1} expressed on a dry weight basis. (Data source EA)

Compound	Mean μg kg^{-1}	Values <DL (%)	ISQG[a] μg kg^{-1}	PEL[b] μg kg^{-1}
ALDRIN	1.1	100%		
DDT (PP)[c]	0.88	92%	1.19	4.77
DDT (OP)[c]	3.32	100%	1.19	4.77
DDE (OP)[c]	0.97	100%	2.07	3.74
DDE (PP)[c]	0.69	100%	2.07	3.74
DIELDRIN	0.97	100%	0.71	4.3
ENDOSULPHAN ALPHA	2.63	100%		
ENDOSULPHAN BETA	2.07	100%		
ENDRIN	0.83	100%	2.67	62.4
HCH GAMMA	2.15	24%	0.32	0.99
HEXACHLOROBENZENE	0.8	100%		
HEXACHLOROBUTADIENE	3.5	100%		
ISODRIN	2.48	100%		
TRIFLURALIN	5.87	68%		

[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 high quality information on sediment contaminants over a broader range of sites.

8. MODELS

Robinson *et al* (1983) constructed a mathematical model to simulate the observed salt distribution in the lagoon (considered to be the result of a balance between weak tidal flushing and a small freshwater input). A one-dimensional numerical model was deemed appropriate due to the shallow, linear nature of the Fleet, and its lack of observable stratification (which indicated negligible vertical circulation). The model represented tidal propagation dynamics and was also used to predict further properties of water movements and transport across different sections. Hydrodynamic modelling predicted that should reduction of the channel cross-section near Moonfleet occur, perhaps due to sediment deposition, the spring tidal range in East Fleet could be reduced by two-thirds. A reduction in tidal energy could accelerate deposition and would be important, as even a relatively small amount of siltation in this channel might seriously reduce the tidal propagation to the western lagoon. These findings were presumably taken into account when changes to the Smallmouth entrance were undertaken.

To determine the residence time of water in different parts of the Fleet lagoon and the relative importance of river/stream discharge, Robinson (1983) subsequently developed a one-dimensional tidal exchange box model. The model was based on data obtained by analysing time-series of tidal records taken in 1967/1968 during a hydrographic survey for CEGB (Robinson *et al.,* 1983) and the bathymetry reported by Whittaker (1978). The model also incorporated physical mechanisms which control water level and fluctuations, such as non-linear propagation of long waves in very shallow water, and unequal progression speeds at high and low water.

Initial results had shown that the model was able to predict the observed principal features of salinity distribution therefore it was applied to simulate flushing characteristics of the Fleet under different run-off conditions, and thus predict contamination within the lagoon arising from the release of agricultural or domestic waste into tributary streams. Findings indicated that discharges through Abbotsbury Brook would be diluted by only ~25% in the lagoon at Abbotsbury. Discharge through Rodden Brook is diluted by 10% of stream concentration on entering the Fleet main channel, whilst Langton Brook is diluted by ~5%. Modelling clearly showed that discharges entering the Fleet west of Herbury point had the potential to significantly contaminate the waters of West Fleet. Conversely, discharges to the Fleet, east of Herbury are rapidly diluted and flushed out by tidal action.

More recent modelling of tidal currents and solute distributions (Westwater *et al.,* 1999) produced an operational two dimensional model of the Fleet lagoon, calibrated against field data. Bathymetric data originally used (from Robinson 1983) were found to be insufficiently detailed; therefore data obtained in 1998 for English Nature were used in the model. However data on bathymetry of the western part of the Fleet were sparse and this was thought to compromise the accuracy of the model predictions to a certain extent. The study again established the flushing characteristics of the lagoon, but used simulations of the transport of a conservative tracer. The model and tracer simulations were then used to predict distribution of

water quality indicators such as salinity and nitrogen distribution in the Fleet and from stream inflows.

Modelling confirmed the earlier findings of Robinson *et al* (1983), that the eastern Fleet up to Chickerell Hive has good tidal exchange, and that tidal exchange with the west Fleet was extremely weak. Tracer studies simulating release of a tracer at Abbotsbury showed that over ten tidal cycles the tracer only travelled as far as the narrow section adjacent to Abbotsbury, and did not even reach Rodden Hive Point. Tracer studies simulating release of contaminants in western Fleet streams also confirmed that Abbotsbury is a potential problem area. Again, contaminants did not travel further than the narrow section adjacent to Abbotsbury over ten tidal cycles. The other streams appeared to have little effect on the gross water quality characteristics as their flows are lower, and they enter the Fleet further eastwards where tidal exchange is better. Some localised effects were seen in the Littlesea area.

Murdoch (1999) identified nutrient sources and estimated input loads on a daily, seasonal and annual basis (discussed in section 6.2). This data was used to quantify the impact of the loads on nutrient concentrations in the lagoon using simple modelling. EcoS, a one-dimensional numerical estuarine simulating shell was used as it can perform annual simulations in a reasonable time and had been applied successfully in previous studies. Nitrogen (TIN), and orthophosphate (as P) inputs from the following sources were considered:

- Stream inputs (Cowards Lake, Mill Stream – Abbey Barn, Horsepool, Rodden, Herbury Stream, West Fleet and East Fleet)
- Wild fowl
- The sea (via the tidal boundary at Smallmouth).

STW inputs were included in the stream inputs.

Model profiles for nitrogen showed a strong annual cycle, with the concentration high in winter and lower in summer. The inference appeared to be that winter runoff and flushing of nitrates was responsible for the winter peaks observed. Phosphate profiles showed a double annual cycle of concentration, with highs in both winter and summer. The winter peaks were thought to be related to the stream inputs (Cowards Lake and Mill Stream have higher winter concentrations). In summer, the highs appear to be related to wild fowl loads.

The impact of point source and wild fowl loads was also investigated: Load analysis suggested that these would have little impact on the lagoon nitrate (TIN) concentrations, but that for phosphate, wild fowl related loads appear to be significant in the summer. Modelling scenarios confirmed the load analysis.

Murdoch (199) also examined salinity profiles as a check on the dilution and dispersion mechanisms in the model. Figure 56 shows examples of salinity profiles including samples and model data.

a)

b)

Salinity Profile

Salinity - Samples & Model

c)

d)

Salinity - Samples & Model

Salinity - Samples & Model
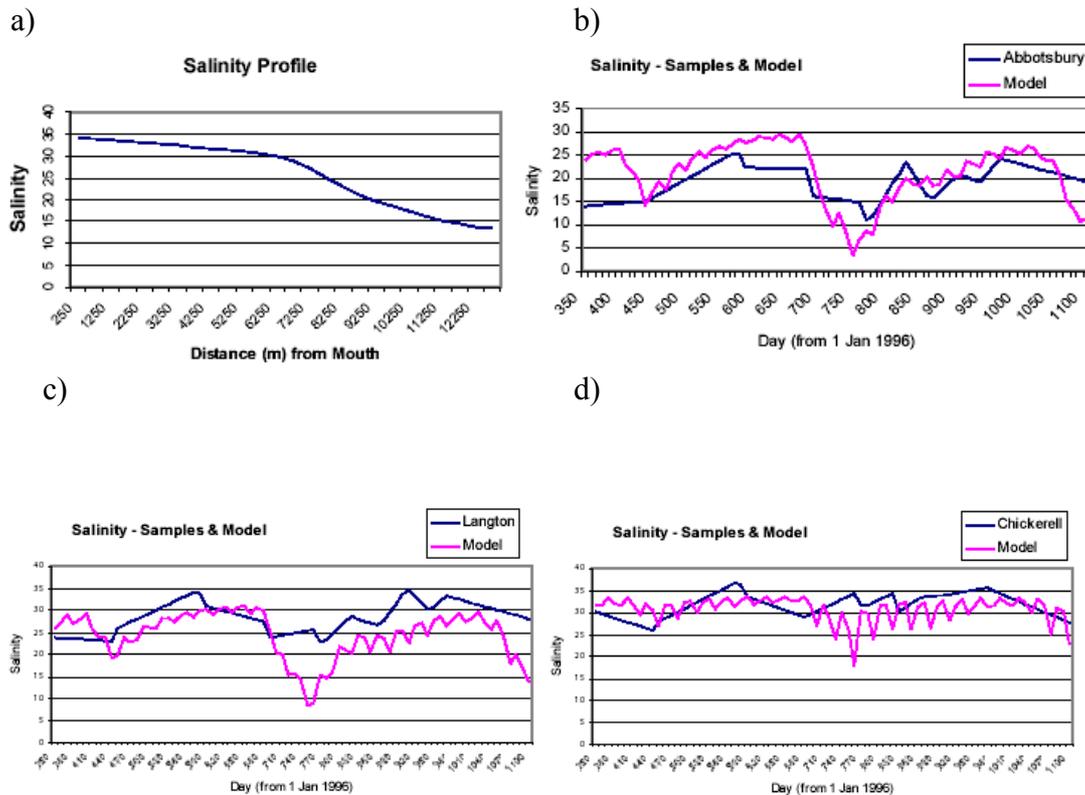
Figure 56. Salinity in the Fleet Lagoon. a) Longitudinal salinity profile. b) Abbotsbury model. c) Langton model. d) Chickerell model. From Murdoch (1999)

Dilution was considered to be controlled by the freshwater inputs, and dispersion by 2 parameter; the longitudinal dispersion and the advection velocity, which is normally taken as some fraction of the flow velocity. Reduced dispersion was modelled at the Abbotsbury end of the Lagoon, which was considered plausible because of reduced tidal mixing. However, both model and observed profiles show salinity to vary temporally at individual sites (figure 56b,c,d).

This could presumably be a result of rainfall (increasing freshwater inputs) or abstractions of freshwater (decreasing freshwater inputs).

An export coefficient model was used to estimate nutrient loadings to the Fleet from diffuse sources (Mainstone and Parr, 1999). The modelling was based on detailed information of land use from 1997 MAFF agricultural census data (crop areas, livestock numbers etc.), background levels, atmospheric deposition, in-stream loads, point source load estimates, and estimates of the contributions from the swannery (and other bird and wildfowl species). The finalised nutrient budget for the Fleet Lagoon is given in table 16 (nutrients section).

Results showed that nitrogen loads are derived primarily from agriculture, with arable cultivation likely to be particularly important. Nitrogen loads to the East Fleet can be expected to be somewhat higher than those to the West Fleet, which may have

113

important implications for eutrophication processes. Phosphorus loads from agriculture also appeared to be highly important, particularly livestock (predominantly dairy) farming, although there may be significant contributions from the two main sewage treatment works (particularly Abbotsbury). Since the Abbotsbury STW load ultimately discharges into the West Fleet, and dairy farming appears to be focused on the western half of the catchment, phosphorus loads to the West Fleet can be expected to be somewhat higher than those to the East Fleet, which may also have important implications for eutrophication processes.

Mainstone and Parr (1999) expressed concerns in that export coefficient modelling can produce only crude estimates, however, the nutrient budget was considered to provide strong indications as to the relative importance of different nutrient sources within the catchment which could be used as the basis for further study.

Note: We have recently been informed that the consented DWF of 140 m^3 day^{-1} at Abbotsbury STW is likely to have been exceeded for many years. Wessex Water has calculated a theoretical DWF of 243 m^3 day^{-1} based on the connected population, and this is predicted to increase to 290 m^3 day^{-1} by 2020 (EA pers comm.). This obviously casts doubt on modelled contributions of P and N.

9. CONCLUDING REMARKS AND RECOMMENDATIONS

9.1   Biological Status

In common with other saline lagoons, there is a limited range of species present in the Fleet compared with other marine habitats. The fauna is characterised by brackish water species and there are a number of specialist lagoonal species with restricted distributions in the UK. A number of nationally rare species, generally 'southern' or warmer water forms, have been identified within the lagoon. Species diversity and abundance both peak in the Langton Hive area where the salinity is generally 20 – 35‰, and the subtidal lagoon bed is dominated by soft mud and mixed stands of *Zostera* and *Ruppia spp.* There is an exceptionally high abundance of some of the more common marine species in the link channel due to the highly specialised conditions which exist in the region. Changes in the distribution of foxtail stonewort *Lamprothamnium papulosum* within the Fleet appear to be related to nutrient enrichment and warrant further investigation. There have also been increases in green macro- and planktonic algae which can impact on several conservation interests including *Zostera* and its associated community (Johnston and Gilliland, 2000).

Quantitative diversity indices are frequently used to assess the effects of environmental degradation on the biodiversity of natural assemblages of benthic organisms. Several indices, which help to put biological status into perspective, have been applied to other cSAC/SPAs in the region, but not to the Fleet. To help in comparisons between sites, and in long-term assessment of condition, it may be useful to apply a selection of these quantitative techniques as part of a future monitoring strategy for the Fleet (see Annex 6). The difficulty may be that lagoons are a relatively exceptional coastal feature and there are few lagoons of the same type and size (the most similar in physiographic terms is in northern Scotland) for valid comparisons. However, there are likely to be numerous lagoon SACs in other EU Member States, which could provide useful information for comparative study in these priority habitats[1].

Regarding bird populations, a recent review of statistics for Chesil and the Fleet cSAC, SPA by BTO has shown that numbers of the internationally important Dark-bellied Brent Goose increased during the 1970s and 1980s but have decreased in recent winters. This mirrors regional and national trends and therefore no alerts have been triggered; adverse factors such as pollution were not considered sufficient to warrant additional investigation (Armitage *et al.*, 2002). However, in view of the suspected sensitivity of the site to nutrients, and the largely unknown status of many toxic contaminants, the acquisition of any further information which addresses possible links between environmental quality and biological consequences seems desirable.

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[1] Priority habitats: those considered in danger of disappearance and for the conservation of which the European Community has particular responsibility. Sites submitted for priority habitats to the European Commission are automatically classed as Sites of Community Importance (one step on from being identified as a candidate SAC in the SAC selection and designation process).

9.2   Chemical Status

At present there is little unequivocal evidence from chemical data indicating that modifications to biota of the European Marine Site have occurred, or would be expected to occur, due to toxic contaminants. The threat from nutrients is probably of most concern. However, the available evidence on biological effects from both toxic and non-toxic contaminants is very patchy. 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 9.3). Until more appropriate integrated chemical and biological effects monitoring is put in place to answer this challenge it is only possible to assess, subjectively, the possible impact (or lack of impacts) of individual contaminants, or groups of contaminants, based on available evidence. A brief overview of the major water and sediment quality parameters that could, in theory, impinge on benthic communities in the Chesil and the Fleet cSAC, SPA is given below, and summarised in table 26.

1) Organotins

There have been no measurements of TBT (or other organotins) in water, sediment or biota from the Fleet, making it impossible to provide an up-to-date and accurate assessment.

Tidal waters in Portland Harbour, sampled by EA since 1997, were often below the detection limit, though this has varied between 2-68ng l^{-1} compared with the EQS for coastal waters of 2 ng l^{-1} (maximum). At two of the sites, maximum values of 5 and 7 ng l^{-1} were recorded in 2001. As these are above the EQS they may impinge to some extent on the Fleet, though probably only close to the entrance.

The source of this TBT may be partly from antifouling paints on boat hulls although a small input from STW cannot be ruled out. Concentrations up to 76ng l^{-1} and 161 ng l^{-1} TBT (as cation) have been measured in effluent samples from Weymouth STW and West Bay Pumping station implicating sewage as an occasional source.

It is likely that Fleet sediments are low in TBT. Sediment samples taken from one site in Portland Harbour, in 1997 and 2001, were without exception below detection limits (8μg kg^{-1} in 2001, an order of magnitude higher in 1997). However, as with water, these limits were above provisional ecotoxicological guidelines (0.005-0.05μg kg^{-1}) set by OSPAR. Clearly, a more detailed survey is needed to establish current status and trends for TBT in sediments.

In 1992 imposex was established in *Nucella lapillus* from Portland Bill (commercial and naval vessels, leisure craft) and West Bay (small boats predominantly), at similar levels of intensity (Harding *et al*., 1992). Six years later (1998) there were clear signs of improvement at West Bay: average VDSI (vas deferens stage index) decreased from 3.97 to 3, RPSI (relative penis size index) decreased from 11.15% to 0.42. In contrast, the dogwhelk population at Portland Bill had been eliminated (Minchin *et al*., 1999). Presumably, levels of TBT in Portland Harbour have been high enough at times to sustain the phenomenon (confirmed by EA water data).

Table 26. A Summary of Water and Sediment Quality issues in Chesil and the Fleet cSAC, 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?)	Largely undefined but likely to be of most significance in the East Fleet near the entrance to Portland Harbour	Mostly historic but possibly some continuing input from shipping (docks), STW and sediments.	Molluscs (No data for the Fleet)
2) Metals (Hg, Cu, Pb, As, Cu, Zn, Cd)	No acute problems identified but sediment Cu and As exceed ISQG guidelines throughout the Fleet; also Hg at East Fleet; Zn and Pb at Ferrybridge.	Possibly STWs, urban run-off, boats and ships. Sediments principle sink for metals. Weymouth outfall increases metal bioaccumulation, locally, but outside cSAC/SPA	Invertebrates (primarily molluscs and crustaceans), species composition, larval fish and birds (Very few bioaccumulation data for the Fleet)
3) Nutrients (macro- and micro-algal blooms, algal toxins)	Especially West Fleet	Agricultural sources (livestock, fertiliser application) STW discharges also important in W. Fleet Possibly sediments – (phosphate, ammonia)	*L. papulosum* and *Zostera (Ruppia* less sensitive). Invertebrates, fish (esp. early life stages), seabirds, General diversity
4) Dissolved oxygen fluctuations	Especially West Fleet	Micro- and macro-algal blooms (linked to nutrient enrichment from agricultural sources and STWs)	Fish, invertebrate communities
5) Hydrocarbons, PAHs	Poorly defined, due to lack of sampling but some evidence of enrichment in the Fleet and Portland Harbour sediments	Incomplete combustion of fossil fuels, run-off, discharges, boats and ships: sediments main reservoir (PAHs probably of pyrogenic origin)	Benthic invertebrates and fish (NB those in contact with sediment). Very few bioaccumulation data. ΣPAHs in oysters marginally below guidelines
6) Pesticides, herbicides and other synthetic organics	Sediments probably main reservoir but no information for the Fleet. Direct toxicity improbable; extent of endocrine disruption not tested.	Not quantified due to insufficient data but probably includes a component from agricultural run-off, and sewage discharges. Sediments probably main reservoir. Weymouth outfall increases PCB, DDE bioaccumulation, locally, but outside cSAC/SPA	Invertebrates (esp. crustacea), fish Very few bioaccumulation data for the Fleet

Littorina littorea from Portland Harbour, near the docks contain residues of 102 µg TBT kg^{-1} (ten times higher than controls outside the harbour). Though unlikely to impair reproduction in winkles, these levels are above the OSPAR ecotoxicological assessment criteria for TBT in mussels (1-10 µg kg^{-1}), confirming that further monitoring of TBT, including bioavailability, should be considered as a possible candidate for priority action in the future, if only to establish or eliminate the possibility of ingress of TBT to the Fleet from Portland Harbour.

2) Metals

There are no data for metal concentrations in rivers or consented discharges entering the Fleet, nevertheless the balance of evidence suggests these would be unlikely to cause serious problems. Although the current monitoring of tidal waters of the Fleet lagoon by the EA is limited to a single site in the East Fleet (the Narrows), there have been no cases of EQS exceedences in recent years. (However, in 1996 the standards for copper and zinc were exceeded at the adjacent shellfish sites in Portland Harbour). Metal concentrations in sediments of the Fleet are considered fairly typical for estuaries in the region. The data of Nunny and Smith (1995) suggests that none of the sediment metals in the Fleet was above sediment 'probable effects levels'. Chromium concentrations were consistently below the lower interim sediment quality guideline (ISQG). Lead and zinc were generally below the ISQG except at Ferrybridge. Mercury concentrations were also extremely low throughout the area except at one site in the Fleet (east Fleet) and in Portland Harbour. Copper and arsenic concentrations exceeded the ISQG throughout the Fleet, though by a relatively small margin. On this basis, according to the guideline criteria, no harm to biota would be predicted, though where concentrations exceed the ISQG chronic effects cannot be excluded.

Nunny and Smith (1995) have suggested that the enrichment seen in these coastal sediments, relative to off-shore samples in Lyme Bay, may be due in part to urban sources (run-off and sewage). However, it is also likely that the relative abundance of organic and oxyhydroxide coatings – characteristics which govern the ability of sediments to sequester metals and act as sinks – are a factor: estuarine sediments tend to be enriched naturally in these coatings compared to those of marine origin.

It would be useful to extend the data on metals to include characterisation of 'speciation' in sediments, to help establish whether or not deterioration of the site due to metals is likely. It is stressed that the data on sediments is almost ten years old and may not be representative of conditions now. Re-survey is also seen as a particularly important issue in terms of meeting standstill requirements for sediments under the Dangerous Substances Directive and attainment of Favourable Condition under the Habitats Directive.

Bioindicator studies with infaunal species such as worms and clams are also recommended, as there is little information on metal bioavailability or bioaccumulation in the Fleet other than in occasional commercial oyster samples. Several metals (As, Cu, Pb, Ni, Zn) would register as 'high' in 'mussel-watch' classification schemes. However oysters are exceptional natural accumulators of some metals, particularly Cu and Zn. Shellfish monitoring (mussels) at Portland, between 1985 and 1992, indicates metals as being generally below expected/guideline values and that sources in the Harbour may not be important for the Fleet, in terms of

bioaccumulation. EA survey data suggests that the Weymouth long-sea outfall increases metal bioaccumulation near to the discharge, though this is outside the cSAC/SPA.

Although ecological impact due to metal contamination in the Fleet lagoon is not anticipated, conditions in sediments, particularly anoxia, could increase the bioavailability of certain metals (Cu, Cd, Ag and Hg). There is scope for more research to establish the physiological and ecological significance of metal body burdens. Sub-lethal biomarkers, 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

Nutrient-associated water quality problems in the Fleet have been apparent for several decades. Monitoring data show that waters of West Fleet are subject to a high degree of nutrient enrichment, and modelling exercises imply that the principal source is agricultural (fertiliser application and livestock), although localised enrichment from STWs and the swannery also appears to be significant. The poor flushing characteristics of the lagoon, coupled with its shallow depths and extremes of temperature compound the enrichment problems and result in eutrophic conditions and macro- and micro-algal blooms. DSP has been detected in mussels and oysters which led to the issue of a Temporary Prohibition Order on shellfish harvesting from the lagoon between July and September 2000. The algal blooms appear to be increasing in regularity and intensity.

The evidence suggests that Abbotsbury STW is likely to be the most significant point source input to the Fleet, especially for P. It discharges enter the most sensitive area and provide bio-available nutrients, the concentrations of which increase, during summer months, possibly due to a local influx of holidaymakers as Dorset is a popular holiday area. At this time algal and plant growth are at a maximum. Data for seasonal variation in nutrient concentrations and effluent flows from Abbotsbury STW, coupled with further tidal water monitoring is recommended to establish the extent of this point source influence on nutrient levels, and to determine whether P (or N) reduction from point sources would be of benefit.

By their nature, coastal lagoons are probably the most sensitive of marine/brackish water ecosystems to changes in water quality. 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). It is likely that this occurs in the Fleet, as reports indicate the presence of significant biomass of green macroalgal species (*Enteromorpha Ulva* and *Chaetomorpha spp.*) which die back annually. Conditions in the lagoon, particularly West Fleet, are liable to increase the toxicity of ammonia, posing a threat to sensitive lagoonal communities. 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, the significance of related microbial activity, and attempts at evaluating the sediment as source or sink for nitrogen, ammonia and phosphorus is difficult.

The issue of nutrient enrichment has been acknowledged by the EA and the Fleet lagoon designated Polluted Water (Eutrophic) (under the Nitrates Directive 91/676/EC). The catchment area for the Fleet has been designated as an NVZ. This could provide the framework to reduce nutrient inputs from all sources.

However regular nutrient (N, P and ammonia) analysis in the lagoon (which appears to have ceased in the late 1990's) will need to be reinstated to enable the effectiveness of any remediatory measures to be accurately monitored.

Preliminary research into the effects of the high levels of nutrients on individual species in the Fleet has provided some evidence for impact on *L. papulosum*, which is sensitive to high levels of phosphate and has become locally extinct in the Abbotsbury embayment. There have also been increases in green macro- and planktonic algae which can an impact on several conservation interests including *Zostera* and its associated community (Johnston and Gilliland, 2000). In view of the conservation importance of the Fleet Lagoon 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.

4) Dissolved oxygen

The limited information available indicates that exaggerated DO fluctuations occur as a result of the high levels of photosynthetic activity in the lagoon. Whilst daytime levels are elevated to the point of supersaturation, DO at night may drop below mandatory levels required by the shellfish directive and possibly those required for sensitive saltwater life. DO levels are also an important factor in ammonia toxicity; low levels can increase the threat to aquatic organisms, particularly in the Abbotsbury embayment where high temperatures and pH occur.

5) Hydrocarbons, PAHs

The EA database contains no information on 'hydrocarbon oils' in freshwaters, discharges, tidal waters, sediments or shellfish in, or adjacent to, the SPA/cSAC. The survey of Lyme Bay in 1994 by Nunny and Smith (1995) included one water sample from Portland Harbour which contained 6.08 μg l^{-1} total organic extract (TOE). This was made up predominantly of an unresolved complex mixture (UCM), indicative of biodegraded products of petroleum, possibly from chronic oil pollution as a result of general shipping activity or terrestrial run-off. However, the level of TOE seen in this sample is typical of waters from coastal areas with low levels of petrogenic contamination and, if representative, is probably of little cause for concern in the Fleet in relation to ecotoxicological thresholds.

Total hydrocarbons (THC) in sediments in the Lyme Bay area displayed substantial heterogeneity. Highest concentrations were found in Portland Harbour (200 μg g^{-1}) and the Fleet lagoon (110 μg g^{-1}). Again, the higher THC concentrations in this

general area are due to increased UCM content, consistent with contamination dispersing from shipping or from runoff from Weymouth (Nunny and Smith, 1995).

The total hydrocarbon concentration in the CEFAS sample of pacific oysters from the East Fleet in 1995 – the only bioaccumulation data available - was 5.2 μg g^{-1} wet weight (equivalent to ~130μg g^{-1} dry weight). This degree of contamination appears to be unexceptional and is, perhaps, typical of chronically polluted environments.

Based on this limited data set, it is difficult to comment with certainty on the environmental quality status of the European marine site with regard to THCs, or their impact on estuarine biota and shellfish beds. Although the perceived threat to the Fleet is probably low it would seem important to establish more extensive baseline information as a means of monitoring future improvement/deterioration at this site.

The only published information found for either total PAHs or individual PAHs in water, sediments or biota of the region also come from the Lyme Bay study by Nunny and Smith (1995). PAH concentrations in Portland Harbour waters were below the limit of detection and anticipated to be of relatively low biological significance for the Fleet. The general pattern of PAH distribution in sediments of the Lyme Bay region was consistent with the THC data – highest concentrations were associated with Portland Harbour muds (3.9 μg g^{-1}) and sediment of the Fleet lagoon (2.3 - 3.2 μg g^{-1}). Pyrogenic components were dominant, indicative of fallout from incomplete combustion of fossil fuels (perhaps concentrated in urban runoff).

Comparison of PAH concentrations in sediment, in relation to ecotoxicological guideline values, indicates that values in the Fleet are above the ISQG but below the PEL and, on this basis, sub-lethal effects caused by PAHs would not be predicted but cannot be ruled out. In view of the small number of samples, however, this tentative conclusion should be verified by further sampling of Fleet sediments, particularly with a view to identifying sources and characterising bioavailability.

Measurement of PAHs in biota of the Fleet is limited to the one batch of *Crassostrea gigas* from the East Fleet sampled in 1995 by CEFAS (discussed above for THC).
These would meet OSPAR guidelines for individual PAHs and fall just below NOAA's 'high' classification category for 'Mussel watch'. Although there appear to be no immediate concerns for the Fleet shellfish, based on this evidence, it would be useful to broaden the data on PAHs in molluscs (and perhaps other infaunal bioindicator species) to obtain a more extensive picture of PAH bioavailability in, and adjacent to, the European marine site.

6) Pesticides, Herbicides, PCBs and other organic contaminants.

Most synthetic organic compounds analysed by the Agency in tidal waters (East Fleet Narrows and Portland Harbour) are below detection limits and appear to comply with EQS standards.

For the majority of pesticides, herbicides, PCBs, and chlorinated solvents reviewed in the current report there are no data to evaluate riverine sources and only limited information for STW discharges. Effluent from Weymouth and West Bay STW have

contributed small loadings of γ-HCH (and probably other pesticides and herbicides) to coastal waters in the recent past. EA survey data suggests that the Weymouth long-sea outfall increases bioaccumulation of some synthetic organics (PCBs, OP' DDE) near to the discharge, but this is outside the cSAC/SPA. We can speculate, that risks to biota within the European Marine Site, from aqueous sources, are likely to be low.

Sediments tend to act as a reservoir for many of these compounds, though there are no records of particulate concentrations within the Fleet – only for one adjacent site in Portland Harbour near Ferrybridge, where most organic determinands were below detection. The sum of DDT and DDE isomers in these sediments *appears* to be above the probable effects level (PEL), though this is artifactual, due to the fact that the majority of values are below detection limits - which are themselves often above the guideline value. The same is true of dieldrin, which is calculated to be above the interim sediment quality guideline (ISQG) if half detection limits are used in place of 'less than' values. However, concentrations of γ-HCH in Portland sediments, measured in 1997, were above the PEL. Further monitoring is advisable to ensure stand-still requirements of the Dangerous Substances Directive and to characterise possible sources, which may include diffuse agricultural inputs from the small streams entering the Fleet.

Thee CEFAS bivalve monitoring programme in shellfish waters in England (1995-6) included one oyster sample from the Fleet shellfishery. Body burdens of α,γ HCH, dieldrin, HCB, DDT and PCBs were close to detection limits, and were mainly within ecotoxicological guidelines and food standards. Despite higher detection limits, similar conclusions can be drawn from earlier EA sampling of Fleet oysters (1991-2). Pesticide/PCB levels in the Fleet would therefore be anticipated to be of little toxicological importance. However these appear to be the only biological samples from within the European Marine site that have been analysed for synthetic organics.

With the exception of γ-HCH, the predominance of <DL values in Portland Harbour shellfish supports the notion that most pesticides and herbicides pose little bioaccumulative threat in the region. For samples collected between 1985 and 1992, a number of mussel samples contained PCB's at concentrations between 25 and 42 μg kg^{-1} wet weight - above lower JMP guidelines.

More extensive, targeted sampling in the Fleet is needed to assess bioaccumulation of organics contaminants thoroughly, and should incorporate sediments and sediment-dwelling species such as clams and worms. The possibility of endocrine disruption from synthetic organics has not been investigated in the European marine site and should, perhaps, also be addressed in future.

9.3   Future Research Requirements

Better, more integrated information on environmental chemistry, 'health' and biodiversity are obvious, generalised top-level needs to address the 'quality' of the Chesil and the Fleet 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] as encompassed in the Habitats Directive. Priority should be given to monitoring the condition and extent of macroalgae, *Lamprothamnion papulosum* (foxtail stonewort), *Zostera* spp (eelgrass) and *Ruppia spp* (wigeon grass), particularly with respect to response to nutrient inputs (n.b. in the west Fleet). 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 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 (Davison and Hughes, 1998). The selection of techniques will of course have to take in to account questions of scientific objectives, practicality and cost. They should also take into account, where possible, the sampling strategies and sites used in previous studies in order to capitalize on historic data. It is anticipated that aspects of these schemes are likely to be adopted by the Fleet study group under the monitoring programme outlined for the site by English Nature (1999).

In view of the sensitivity of the site to the possible effects of eutrophication it is recommended that monitoring of both phytoplankton and benthic macroalgae is undertaken, alongside nutrient determination. Palaeoecological evidence from dated cores may help reconstruct the history of blooms and ascertain if this is a recent phenomenon. More work is needed to construct spatial and seasonal models of nutrient sources and fluxes, particularly from Abbotsbury STW, and should also include the role of sediments. Further hydrodynamic modelling and simulation of nutrient distributions would be a useful supplement.

The requirement to fulfil FCS and other drivers relating to toxic contaminants and water and sediment quality (such as the 'standstill' provision under the Dangerous Substances Directive) may be difficult to monitor. The fragmented nature of much of the available environmental quality data prevents all but a first approximation of the status of the site for many contaminants. This needs to be addressed if we are to progress our understanding of how environmental quality, and in particular anthropogenic inputs, are affecting the status of the European marine site.

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[1] Favourable Conservation Status (FCS) for a given habitat/species requires that it's 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 Chesil and the Fleet).

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 (including consented discharges) 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 Fleet surveys have been discussed 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. In common with other SACs, routine monitoring of nutrients (N, P and ammonia) in tidal waters was discontinued in the late 1990's. It is strongly recommended that targeted monitoring be reinstated in order to maintain an up-to-date assessment of water quality in the Fleet and other European marine sites, as results could highlight problem areas and expedite remediatory action when necessary[1]. Additionally, modelling exercises should be repeated using the most recent estimates of DWF from Abbotsbury STW, as the consented DWF is likely to have been exceeded for many years (EA pers comm.), casting doubt on modelled contributions of P and N.

Thus, in addition to nutrients (the main priority), accurate, up-to-date chemical data on e.g. metals, 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 Fleet. It is particularly important that future sampling programmes incorporate more information on sediment contaminants and their role as diffuse sources. Concentrations in sediments presumably consist of both anthropogenic and geogenic components which have yet to be distinguished in the Fleet. Methods are becoming available which address this problem and the consequences for bioavailability and re-release. It is recommended that these be applied. Biota should be collected, concurrently, to try to link sediment loadings and 'speciation' of contaminants with their biological consequences.

There is virtually no information on contaminant body burdens in the Fleet. An accurate picture of patterns of bioaccumulation of inorganic and organic contaminants needs to be constructed in order to establish the current status of the European marine site. The use of a validated suite of indicator species, at a selection of reference sites, is a useful way of estimating bioavailability around the system. This should incorporate infaunal organisms, particularly bivalves and polychaetes, which are capable of reflecting sediment-bound contamination in the lagoon. Surveys should be repeated at intervals in the future, to ensure that bioavailability does not increase. Furthermore, many sediment-dwelling organisms are essential food items for the important bird species and assemblages for which the site is designated. The threat posed by bioaccumulation and food chain transfer of, for example, PAHs, PCBs and metals, needs to be quantified in these species, requiring a much more extensive data set.

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[1] Since the time of writing, monthly nutrient monitoring has been reinstated, and monitoring is required to review the status as a PW(E)

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 Fleet, including possible problem discharges. Also, an important priority is how to assess (and then minimise) the risk from diffuse sources. 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 (if any) in the Fleet with greater certainty. This would ideally include; conventional quantitative ecological survey (for identifying changes in the abundance and diversity of species – see section 9.1 and annex 6), chemical and biomonitoring procedures as outlined above (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 examples and 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, sediment quality guidelines, 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 the Fleet) it does not necessarily mean the environment is healthy. Biological effects may occur if several contaminants act together. 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 and also the local environment. Fleet sediments, for example, may become highly anoxic as a result of overlying macroalgal mats - a feature that can modify the impact of a variety of contaminants. 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.

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11.   ANNEXES

Annex 1. Chesil and The Fleet cSAC, SPA: Summary Of The Interest Features and conservation features, (adapted from English Nature, 1999)

<u>Under the Birds directive</u>

Internationally important (breeding) populations of regularly occurring Annex 1 bird species: Little Tern (*Sterna albifrons*),

- up to 100 pairs (~5% British Breeding Population)

Conservation objectives focus on feeding habitats i.e.

- Lagoon waters- maintain frequency and abundance of crustaceans, annelids, fish and molluscs: should not deviate significantly from baseline

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Internationally important populations of migratory species:
Citation is for Widgeon (*Anas penelope*) – ≥ 1% nw European population.

The Fleet also supports important populations of Ringed Plover, Gadwall, Pochard, Red-Breasted Merganser, Coot, Dark-Bellied Brent Goose, Little Grebe, Shoveler, Goldeneye. The resident mute swan population is the largest in Britain. Teal Tufted Duck and pintail are also present

Conservation objectives focus on condition of habitats:

- Seagrass bed communities
    - (brent geese and wigeon require e.g. *Zostera*, waders prefer invertebrates – see above)
    - maintain absence of obstructions to view lines
- Intertidal sediment communities
    - maintain extent in terms of habitat and food avilability
    - ensure no increase in disturbance in feeding and roosting areas

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Annex 1 cont.

Chesil And The Fleet cSAC, SPA: summary of the interest features and conservation Features, (adapted from English Nature, 1999)

<u>Under the Habitats directive</u>

Lagoons

For which the area is considered to be the largest and supporting the greatest diversity of habitats and species in the UK.

Conservation objectives focus on maintaining attributes such as extent, water clarity, salinity regime, nutrient status (no increase in algal mats), algal grazers (*Rissoa membranacea*), indicators of nutrient status -foxtail stonewort (*Lamprothamnium papulosum*), fish species assemblages.

Maintaing the extent and species composition of the following sub-featuresare an integral objective:
- seagrass bed communities (area and density of *Zostera* and *Ruppia*)
- Tide swept communities (nb species composition at the mouth)
- Sub-tidal coarse sediment communities; gravel, cobbles pebbles(nb *Anemonia viridis*).
- Inter-tidal sediment communities
- shingle springline communities)

Annual vegetation of drift lines

Selected of one of two representatives on the south coastis considered to be subject to little anthropogenic influence.

Conservation objectives focus on maintaining attributes such as extent and absence of landward constraints.

Maintaing the extent and species composition of the following sub-featuresare an integral objective:
- *Beta vulgaris maritima* (sea beet) - *Atriplex* (orache) community
- *Honkenya peploides* (sea sandwort) - *Cakile maritima* (sea rocket) community

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Mediterranean and thermo-atlantic halophilious scrub

For which the area contains a major concentration and is the western limit in the United Kingdom

Conservation objectives focus on maintaining attributes such as extent and absence of landward constraints

Maintaing the extent of of the following sub-feature is the principle objective:

- *Suaeda vera* (sea blight) saltmarsh community. *S. vera,* together with *Atriplex portulacoides* (sea-purslane) lines much of the lagoon above the tidal limit and is in dynamic equilibrium with the sea beet dominated frift-line vegitation described above. The latter tends to replace the scrub community when disturbed by waves and erosion and is in turn displaced by scrub after disturbance ceases.

Annex 1 cont.

Chesil and the Fleet cSAC, SPA: summary of the interest features and conservation features, (adapted from English Nature, 2000)

Perennial vegetation of stony banks

For which this is considered to be one of the best areas in the United Kingdom

Coastal shingle vegetation outside the reach of waves. This encompasses a widw range of vegetation found on coastal shingle outside the reach of waves. It includes the open pioneer stages cloe to the limit of the tide, in which there are a number of specialised flowering plants , such as yellow horned-poppy *Glaucium flavum*. It also includes the grasslands, heath, scrub and moss- and lichen dominated vegetation of very old, stable shingle further inland.

Additional proposed interest

European interest:

Atlantic salt meadows (*Glauca-Puccinellietalia maritimae*)

- **For which the area is considered to support a significant presence**

This habitat encompasses saltmarsh vegetation containing perennial flowering plants that are regularly inundated by the sea. The species found in these salt marshes vary according to duration and frequency of flooding with seawater, geographical location and grazing intensity. Salt-tolerant species, such as common salt marsh grass *Puccinellia maritima*, sea aster *Aster tripolium* and sea arrowgrass *Triglochin maritima*, are particularly characteristic of the habitat.

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 CCl4/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 CHCl3/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$_6$H$_5$OH	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; 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 EROD and metallothionein) 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 Fleet biota with those elsewhere to look for signs of adaptation.

Biodiversity indices: quantitative techniques to assess community-level response to environmental degradation - *Species richness* - an indication of the number of taxa per unit area; -*Shannon-Weiner diversity index* (H^1) - expresses the relationships between the occurrence of species and the apportioning of individuals among those species (relative dominance); -*Pielou's evenness index (J)* compares the diversity of the data with its theoretical maximum, where all species would be equally abundant. Lower values would be associated with samples from sites numerically dominated by only one or two species, which is generally indicative of stressed communities; -*The Simpson index* relates the contribution made by each species to the total population. Although observing similar aspects of the dataset, this index is unrelated to the Shannon-Weiner diversity index. Higher values in the Simpson index equate to the presence of a few dominant species in the assemblage; -*Taxonomic distinctness* This index, which captures phylogenetic diversity rather than simple species richness, is more linked to functional diversity. Clarke and Warwick (2001) have refined and developed this index to pick out degraded locations; -*Abundance Biomass Comparison* - can be used 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.

Various forms of multivariate statistical analysis of benthic communities and associated environmental variables are useful in examining 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 quantities of contaminants in Weymouth STW discharges to controlled waters in 2001

material released	Kg released
arsenic	8.1
cadmium	<1
chloroform	<4
chromium	<20
copper	120
lead	24
mercoprop	<1
mercury	1.1
nickel	68
pentochlorophenol and compounds	0.62
permethrin	0.041
zinc	170

Annex 8. Contaminants in oysters *Crassostrea gigas* and guideline values

Metals in oysters from East Fleet (CEFAS data) and Poole Harbour (own data), in relation to guideline values.

mg kg^{-1} (dry wt.)*

	'Abbotsbury oysters' East Fleet 1995	Brownsea Island, Poole, 1983	JMP upper	ADRIS guide	NOAA 'high'
Arsenic	37.5	12.9			17
Cadmium	2.75	4.8	5	15	6.2
Chromium	37.5				
Copper	500	61			370
Iron	1750	341			
Lead	3.5	2.5		50	0.84
Mercury	0.5	0.26	1	3	0.23
Silver	10.3	1.6			
Nickel	22.3				3.3
Selenium	12.5	3.74			
Zinc	6350	1925		500	5100

* CEFAS data converted to dw using a wet:dry weight ratio of 25 (reported solids content 4%).

Organic contaminants in oysters from East Fleet (CEFAS data), in relation to guideline values.

	oysters East Fleet 1995 (mg kg^{-1})		ADRIS shellfish guidelines (mg kg^{-1})	NOAA 'High' (molluscs) (mg kg^{-1})	OSPAR ecotoxicological criteria (mg kg^{-1}dry wt)	
	wet wt	dry wt	dry wt	dry wt	low	high
ppDDE	0.001	0.025	0.1		0.005-0.05	
DDT	0.001	0.025	0.1	0.14		
dieldrin	0.001	0.025	0.1	0.009	0.005-0.05	
lindane (γ-HCH)	0.001	0.025	0.03			
ΣPCBs CBs 28,52,101,118,138,153,180)	0.007	0.175			0.005-0.058	

* CEFAS data converted to dw using a wet:dry weight ratio of 25 (reported solids content 4%).

Annex 9. Trade effluent consent details, Weymouth STW, 1996

	Concentration in effluent (mg l⁻¹)								Daily volume (m³)
	Cu	**Cr**	**Pb**	**Ag**	**Zn**	**Ni**	**Cd**	**Sn**	
Al foil processing									170
Photochemical plating/anodising	5	5	5	2	5	4			124
printed circuits manufacture	5		5						10
screen stencils		5	5	2	10				3.5
screen cleaning	3		3	2				2	3
printing	5	5	5	2	10	4	2		7.8
Photographic				2					12.6
Vehicle wash									106.7
Launderettes									46.7
other									29.25
Estimated daily metal load (g)	**718**	**667**	**725**	**302**	**713**	**527**	**16**	**6**	

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Back cover: Smallmouth, East Fleet
Photograph: Getmapping plc © Dorset County Council

"The lake is good for nothing except sea fowl, herons, and oysters...".

Reference to the Fleet Lagoon
in "Moonfleet" by J. Meade Falkner, (1896)

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