Characterisation of European Marine Sites

The Fal and Helford

(candidate) Special Area of Conservation

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Site Characterisation of the South West European Marine Sites

Fal and Helford cSAC

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

by the **Plymouth Marine Science Partnership**

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

1: (above) Aerial view of Falmouth Dockyard
2: (above right) Pressure washing a ferry in Falmouth docks
3: (right) Oil terminal at Falmouth

4: (left) Truro Newham STW in the upper estuary

5: (above) Restronguet Creek (at the Point) during the 1992 Wheal Jane minewater incident

6: (left) Agricultural land bordering Percuil

Photographs:
1: EA **2 –3**: MBA
4: Get mapping plc **5-6** MBA

Plate 2: Some of the Interest Features and important species of the Fal and Helford cSAC

1: (right) Mudflats in the Fal Estuary

2: (above) Maerl *Phymatolithon calcareum*
3: (right) Eelgrass *Zostera marina*

4: (left) Shore dock *Rumex rupestris*

Photographs:
1: MBA **2 – 3**: Keith Hiscock, MARLIN
4: Roger Mitchell, English Nature

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 Fal and Helford candidate Special Area of Conservation (cSAC).

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

Site characterisation has been accomplished by review of published literature and unpublished reports, together with interrogation of raw data sets, notably that of the EA (this does not include recent compliance data and other forms of self-monitoring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

Key findings are; that parts of the cSAC, principally Carrick Roads, are impacted by past mining activities outside the boundaries of the site (notably from Restronguet Creek) which continue to influence the area via mine drainage discharges and remobilisation of metals from sediments.

The whole system, and in particular, the Falmouth area, is affected by organotin contamination. The principal source is Falmouth Dockyard, although sediments now also contribute significantly to the overall burden.

Parts of the system, notably the upper Fal estuary, are subject to eutrophication. Although the majority of nutrient inputs in the cSAC may be due to diffuse sources such as agricultural run-off, localised enrichment from STWs, are also significant, particularly in the more enclosed reaches of the upper Fal Estuary where chronic contamination and nutrient-associated water quality problems have resulted in toxic algal blooms. The most recent incidence, in 2002, also affected the Helford Estuary, resulting in invertebrate mortalities.

The problem of hypernutrification has lead to the designation of the Truro, Tresillian, and Fal Estuaries as a Sensitive Area (Eutrophic) (under the Nitrates Directive 91/676/EC), which it is hoped will bring about improvements to the nutrient status of the region. However, nutrient enrichment in some of the more enclosed waters of the Helford Estuary is also a cause for concern and warrants further investigation.

These principal findings are discussed in detail, together with implications for key habitats and species, and the ecological significance of potential effects. Gaps in knowledge are identified and recommendations are made for future studies which may improve understanding of the system and assist Regulatory Authorities in their statutory responsibilities to ensure the favourable condition of the site and its features.

CONTENTS

2. INTRODUCTION

This review considers the characteristics of the FAL AND HELFORD cSAC and how the status of the site is influenced by existing permissions and activities, either alone or in combination. Also considered is possible impact from other factors such as unconsented activities, diffuse sources and natural processes. This information is used to prioritise in our opinion (but as objectively as possible) which permissions or activities are likely to have a significant effect on the site. This includes activities and consents outside the cSAC itself. 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 thus identified, 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 any evidence that existing water (or sediment) quality is causing impact and highlight limitations of available data.
- To identify and recommend further research which will address limitations of current information and establish cause/effect relationships.

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

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

Standard list of operations which may cause deterioration or disturbance	**INTEREST FEATURES***			
	Large shallow inlets and bays	**Sandbanks which are covered by seawater at all times**	**Mudflats and Sandflats not covered by the sea at low tide**	**Atlantic Salt Meadows**
Physical loss				
Removal (e.g. land claim, development)	✔	✔	✔	✔
Smothering (e.g. artificial structures, disposal of dredge spoil, outfalls)	✔	✔		
Physical damage				
Siltation (e.g dredging, spoils, outfalls)	✔	✔		
Abrasion (e.g. boating, anchoring, trampling)	✔	✔		
Selective extraction (e.g. aggregate dredging, entanglement, bait digging)	✔	✔	✔	
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, endocrine disruptors)	✔	✔	✔	✔
Introduction of non-synthetic compounds (e.g. heavy metals, hydrocarbons)	✔	✔	✔	✔
Introduction of radionuclides				
Non-toxic contamination				
Nutrient enrichment (e.g. agricultural run-off, outfalls)	✔	✔	✔	
Organic enrichment (e.g. mariculture, outfalls)	✔	✔	✔	
Changes in thermal regime (e.g. outfalls, power station)				
Changes in turbidity (e.g. dredging, outfalls, agricultural run-off)	✔	✔		
Changes in salinity (e.g. water abstraction, outfalls)				
Biological disturbance				
Introduction of microbial pathogens				
Introduction of non-native species and translocation	✔			
Selective extraction of species (e.g. bait digging, wildfowl, commercial and recreational fishing)				

***note: more recently, additional interest features have been submitted to the EU. These include Estuaries and Reefs (probably subjected to the same threats from toxic and non-toxic contamination as other features, above). The shore dock *Rumex rupestris* has also been added as an interest feature of the Fal and Helford (see Annex 1).**

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/CCW may need to apply during the review process as outlined 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.

[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. SOURCES OF INFORMATION AND REPORT STRUCTURE

- 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 (Bodmin), Joint Nature Conservancy Council (JNCC) Coastal Directories Reports (Region 11), Cornish Biological Records Unit, Cornwall Wildlife Trust, Cornwall County Council, Ministry of Agriculture, Fisheries and Food (MAFF); (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 IPC sites.
- The Plymouth Marine Science Partnership (PMSP) laboratories (MBA, PML, and UoP) have already undertaken a wide range of studies in the Fal Estuary system relevant to the impact of heavy metals on the biota. Published and unpublished information has been drawn upon extensively for the present assessment. These studies include: measurements of heavy metal concentrations in water, sediments, and biota (macroalgae, coelenterates, polychaetes, bivalves, gastropods, crustaceans, fish); toxicity experiments on estuarine invertebrates and meiobenthos; experiments on heavy-metal tolerance (*Fucus, Hediste, Carcinus*); measurement of uptake and effects of metals in transplanted organisms (macroalgae, *Scrobicularia, Cerastoderma, Mytilus, Littorina*); surveys of macrobenthos and meiobenthos assemblages of the intertidal flats in relation to heavy metal concentrations. MBA also has metal data for sediments and estuarine biota in a large range of UK estuaries, including other south-west marine sites (e.g. Tamar, Exe, Poole Harbour) which have been used to draw comparisons.

The review of published information is dealt with in **Section 5**, which contains sub-sections on threats from toxic contamination (5.1- e.g. metals, TBT, petrochemicals), non toxic contamination (5.2- e.g. nutrients, turbidity, dissolved oxygen) and other threats and activities (5.3). This overview incorporates reviews of both chemical and biological information.

Section 6 describes summaries statistics of previously unpublished water quality data in relation to Environmental Quality Standards and guidelines (themselves outlined in Annex2-5), using information provided by the Environment Agency (extracted from WIMS). The section again includes considerations of toxic (6.1) and non-toxic contamination (6.3). There is also a synthesis of sediment quality (6.2), using MBA metals data and mapping routines.

A brief description of modelling exercises of direct relevance to the environmental quality status of the cSAC is provided in **section 7**.

Concluding remarks (**section 8**) include a summary of evidence for impact in the Fal/Helford cSAC together with recommendations for future research requirements (specific and generic)

4. THE SITE: FEATURES AND THREATS

The catchment and sediments of the Fal reflect its underlying geology and most significantly for the current synthesis is heavily influenced by the Carnmellis granite and surrounding metamorphic aureole to the west of the estuary (see Hosking and Obial 1966, for review). The large input of china clay wastes from the St Austell area has had a major silting impact in the upper estuary and its saltmarshes. Mining of the metalliferous deposits, for Sn, Cu, Pb, Fe and lesser amounts of As, W, U and Ag, has been a major feature of the area since the Bronze Age and at the peak of activity in the 19th century ore processing in the Carnon Valley in particular remobilised millions of tons of tailings. Much of this has been deposited in Restronguet Creek making it the most metal polluted estuary in the UK (Bryan and Langston , 1992). However there is evidence that some of the contaminated sediments have been transported to other parts of the Fal, though mainly concentrated in adjacent Creeks on the western side such as Mylor, and Pill. Towards the mouth of the estuary sands of a marine origin begin to dominate. A key feature on the St Mawes Bank is the large maerl bed consisting of live and dead coralline algae. There is relatively little large industry left to affect the sediments of the Fal (the last active mine Wheal Jane was finally abandoned in 1991), though Falmouth Docks and a number of marinas are potential sources of disturbance (dredging activities, oil and release of antifouling and sewage). Residual drainage from old mines, spoil heaps and groundwater may also continue to influence sediment geochemistry.

The Helford River, like the Fal, is a drowned river valley or ria, formed when the sea level rose at the end of the last ice age. The Helford catchment consists of sedimentary, Devonian carboniferous rocks to the south and west. To the north of the river, the southern limit of a large mass of granite lies beneath Constantine and has further metamorphosed Devonian rocks associated with it which contain small amounts of mineralisation. During the 19th century, the granite was quarried to supply a worldwide market, and operations continued into the last century until the last remaining quarry closed in 1993.

Unlike the Fal, the Helford Estuary has no history of extensive mining complexes in the immediate vicinity, although the numerous adits in the area indicate that small-scale mineral exploration and extraction has occurred. There is evidence of medieval workings for metals, and ores of copper, silver and tin were mined at Wheal Vyvyan, Wheal Inow and Wheal Anna Maria, to the north of the river near Constantine and Porth Navas during the 19th century. The last of the mines to cease operations was Wheal Vyvyan, which closed in 1864. These workings will no doubt have contributed to the sediments of the main channel, as silt deposits in the upper reaches of the waterway were redistributed. This is reflected in Cu, Zn (and Sn) enrichment, relative to baseline values for the whole of the UK (table 3); compared with Restronguet however sediments in the Helford do not appear to represent an acute problem.

The boundaries of the Fal and Helford marine site are shown in figure 1. Maps of communities and features within the site can be found in greater detail elsewhere (English Nature, 2000).

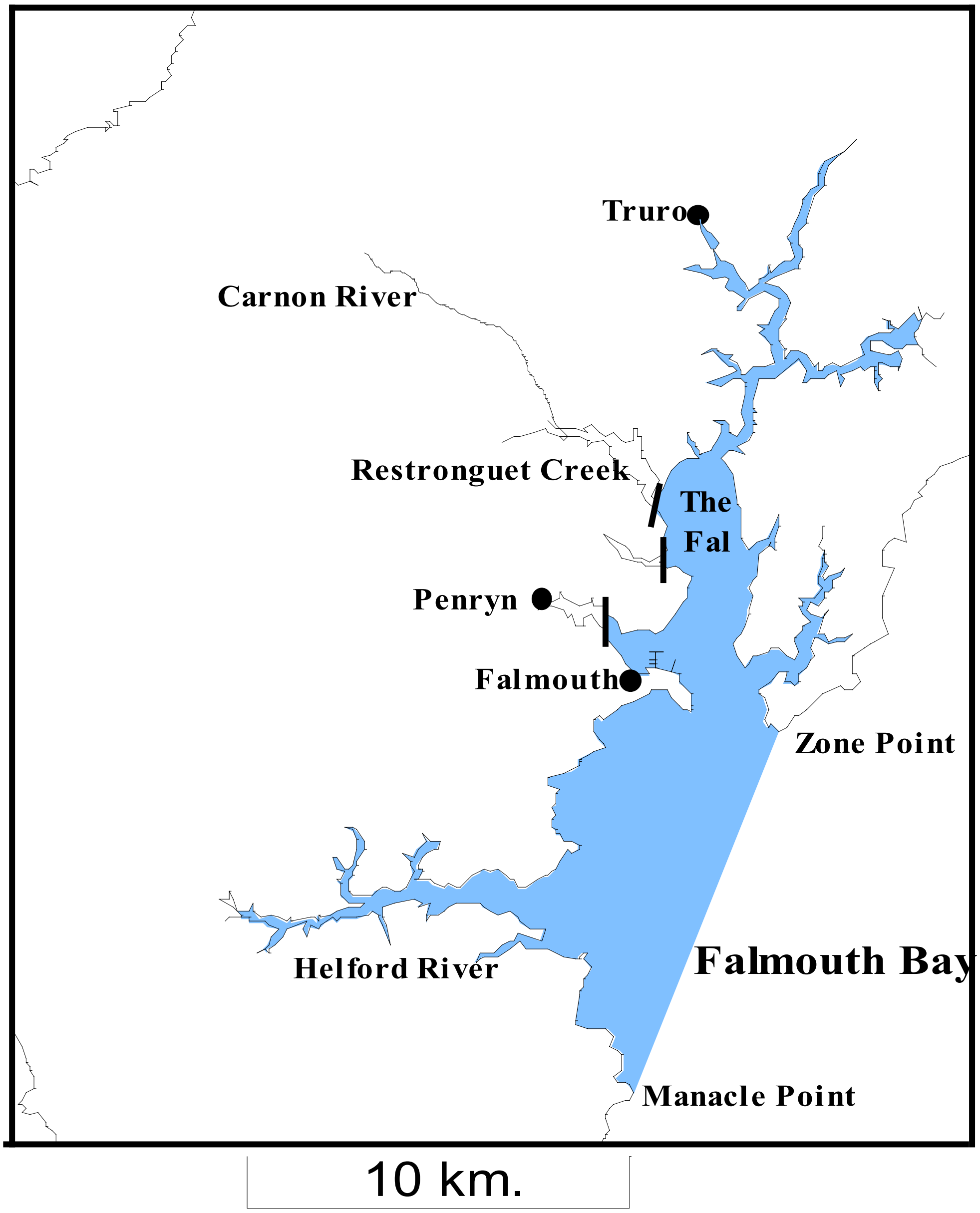


Figure 1: Fal and Helford cSAC showing boundaries of marine site

There are a large number of discharges into the Fal system of varying sizes (see full list in the accompanying electronic database). Siting of some of the more important (by volume) discharge consents impacting on the Fal Estuary system are shown in figure 2.

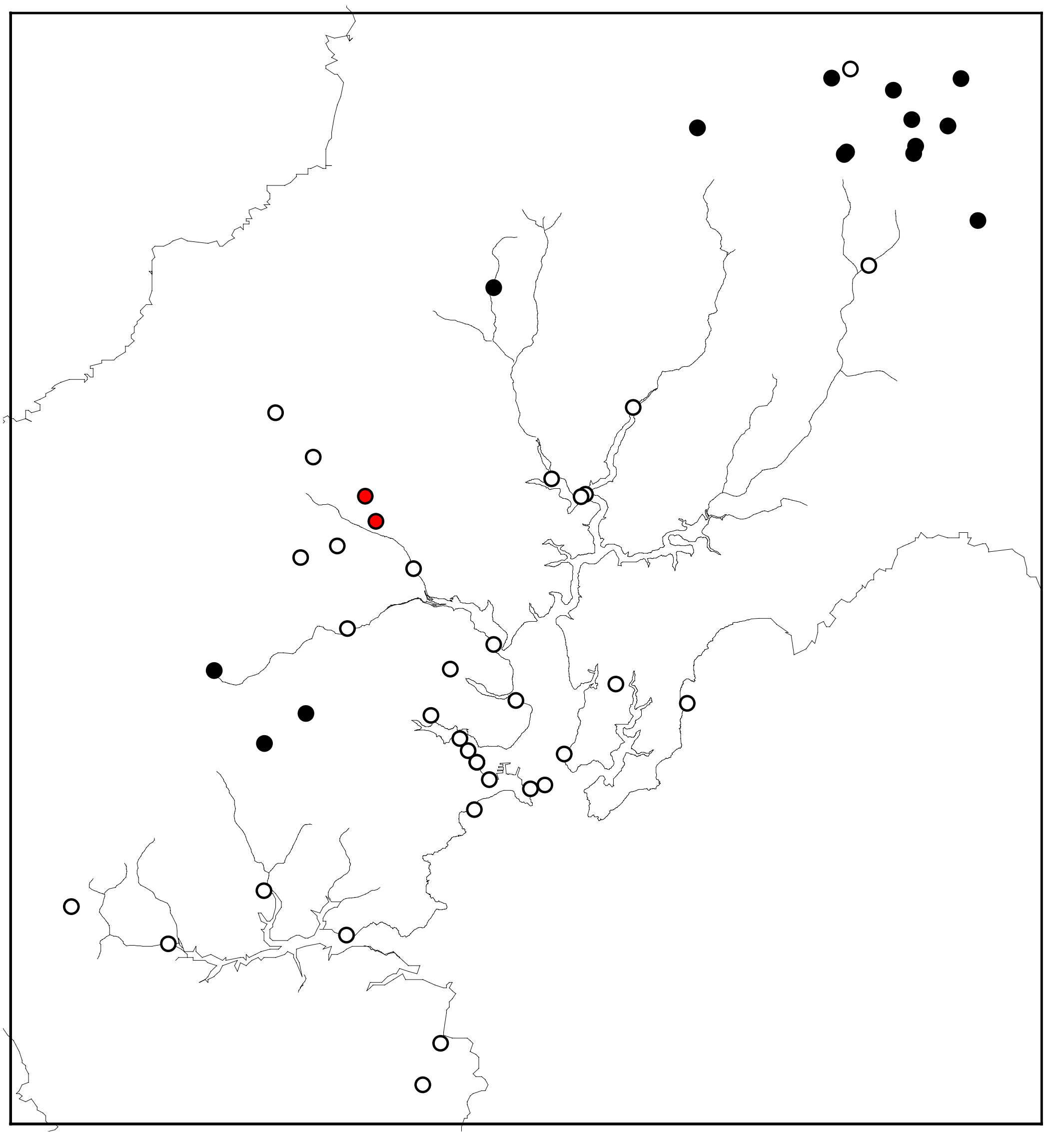


Figure 2. Locations of some of the larger discharge consents to the Fal Estuary system. Consents shown for the discharge of sewage (generally set for SWW) are those >7.3 m^3/d MAX (open symbols). Trade consents, and miscellaneous sources of effluents shown (closed symbols) are those> 120 m^3/d DWF .
NB No distinction has been made between continuous and intermittent discharges. Details of specific discharges should be clarified with the Environment Agency

5. SUMMARY OF KEY STUDIES

5.1 Toxic Contaminants

5.1.1 Metals

Restronguet Creek on the West side of the Fal Estuary complex lies outside the Fal /Helford cSAC. Its inclusion in this report stems from the possibility that water/sediment quality issues within Restronguet Creek could have a bearing on other parts of the Fal system. The character of the marine site as a whole is certainly influenced by Restronguet Creek which has dominated metal inputs for centuries due to contributions from the metal-rich water of the County Adit (draining the St Day mining area) together with the more localised inputs in the Carnon Valley. The latter includes drainage from the Mt Wellington and Wheal Jane mining complex, which was operating periodically until the latter half of the last century. The abandonment of the Wheal Jane mine, in 1991, caused an uncontrolled release of acidic metal laden water into the Carnon River and the Fal Estuary. Subsequently, the Environment Agency (Agency) implemented a series of emergency measures to control and treat the mine water discharge and, therefore, limit the impact of the release on the environment.

Emergency pumping and treatment measures have been progressively developed to form the existing treatment system, completed in October 2000, and scheduled to remain in operation for ten years. This treatment process relies on the Clemows Valley Tailings Dam for disposal of treated mine water sludge.

An initial Biological Assessment study by Plymouth scientists, part of a wider Wheal Jane mine water project, completed in 1998 (Warwick *et al*., 1998), considered a range of active and passive mine water treatment options and provided an assessment of the present biological status of the Carnon catchment and the Fal Estuary and the significance of the mine water discharges on the floral and faunal communities in the Carnon River and Fal Estuary. The main conclusion was that the flora and fauna of the Carnon River itself are greatly impoverished and of little conservation value. On the other hand, the Fal Estuary system is of high conservation, recreational and economic value. The existing relationships between the biota and metal concentrations in the water and sediments in the Fal system were established, and also some wider comparisons with other estuaries in south-west Britain were made, in order that predictions concerning the effects of various treatment options could be made by extrapolation. Literature data and more recent data collected by members of the Plymouth Partnership were used to provide these assessments.

Estuarine Chemistry (Sediment and Water)

Concentrations of metals in water and sediment in Restronguet Creek and other parts of the Fal System are described in numerous data sets held by for example the EA (and previously NRA- see summary of earlier 86-91 data by Millward 1992), MAFF and MBA. Water sampling was particularly intensive at the time of the Wheal Jane incident and subsequent Wheal Jane Project (e.g.Warwick *et al.,* 1998). There are also a number of key publications describing distributions of metals at various times over the last 30 years (Bryan and Gibbs, 1983; Langston 1983; Langston *et al.,* 1994a). These studies illustrate, however, the large temporal and spatial variations that occur, particularly in waters, making an overview difficult. An example of this is shown in table 2. which shows concentrations in the Carnon River at various stages of mining activity during the last 20 years (operational, after abandonment, during flooding, after installation of pilot treatment plant)

Table 2. Concentrations of dissolved metals (µg l^{-1}) in water entering Restronguet Creek (Devoran Bridge)

	As	**Cd**	**Cu**	**Fe**	**Mn**	**Zn**
Mine operating (1979-82)	71.7	21.3	594	6295	1910	11602
Before flooding (1991)	57.8	6.6	272	153	735	3108
During flooding (1992)	2103	232	2759	177702	4179	149768
Recent (pilot treatment)	26.3	4.6	241	129	524	2324

Concentrations of most metals have decreased significantly in the Carnon River since the treatment site. Removal of Fe is most efficient and will eliminate the threat of discoloration events, though this has implications for estuarine chemistry (see below)

Rather than snapshot-type monitoring, better evaluations of the biological significance of metals can be made by, for example, investigating their behaviour and fate in relation to salinity, and how they partition between sediments and water. The Wheal Jane Mine water project (Warwick *et al*., 1998) began to address some of these issues by:

1. Describing the concentrations of metals in the water column of Restronguet Creek (baseline review) and their behaviour throughout the estuarine system, at different salinities and tidal states.

2. Determination of the relative retention of metals in sediments of Restronguet Creek and potential for release, from water column studies. Field-based derivation of partition coefficients Kd, essentially the ratio: metal in sediment: overlying water.

3. Simple modelling of partitioning behaviour of metals between sediment and water, using data from water column studies, to show how this affects distributions and fate within the estuary (and hence the biota). Extrapolations were made to predict sediment and water quality under different treatment options.

Behaviour of dissolved metals

If metals in the Carnon River remained in dissolved form, and this water mixed completely with seawater on entering Restronguet Creek, it would be simple to predict metal concentrations at any point from the linear relation between metal concentration and salinity (conservative behaviour). Some metals have consistently conformed to this pattern e.g. Cd and Zn –two of the more soluble metals (figure 3).

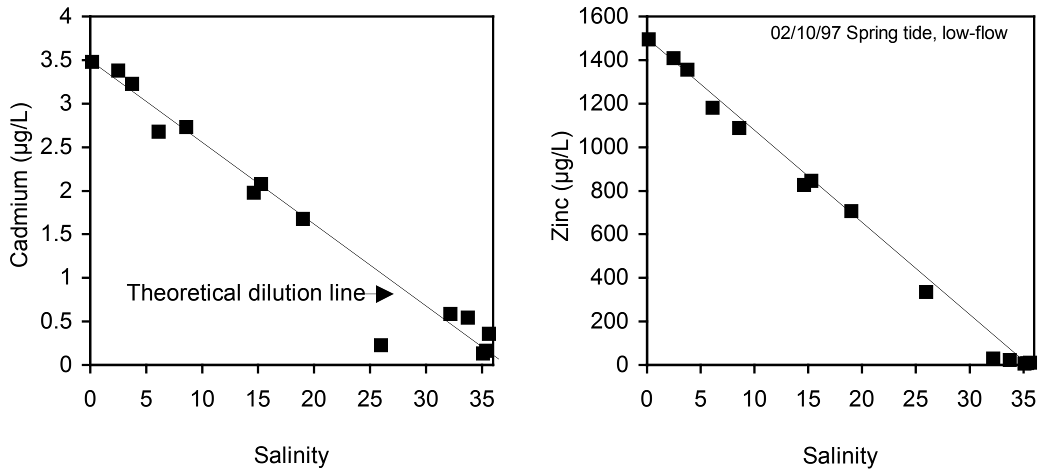


Figure 3. Behaviour of Cadmium and Zinc during estuarine mixing of metal rich water from the Carnon River and cleaner seawater from the Fal. Both elements approximate to the theoretical dilution line expected if no reactivity were displayed.

If, however, the metals become associated with particulate material (particularly Fe oxyhydroxide coatings) and there is removal from solution, their transport will not mirror that of the salt (non-conservative behaviour). Examples from surveys are shown in figure 4.

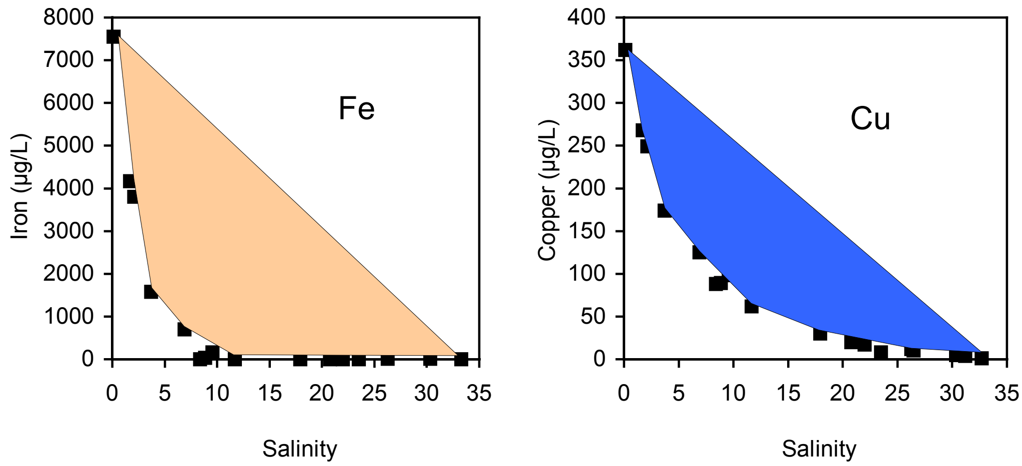


Figure 4. Removal of Fe and Cu from solution in Restronguet Creek during mixing of metal rich Carnon river water and clean seawater. (No water treatment at Wheal Jane - note high Fe concentrations in freshwater)

Treatment of mine water has recently altered the proportions of metals released into the Carnon River. By neutralising mine water at Wheal Jane, much of the Fe is removed. This has had a markedly positive effect in reducing discoloration events in the Fal. However since Fe oxyhydroxide production in the estuary is now reduced, scavenging of other metals in the estuary may also be less pronounced: apart from Fe, the behaviour of most other metals may now tend to be conservative during estuarine mixing (see example for Fe and Cu) i.e. less efficient removal.

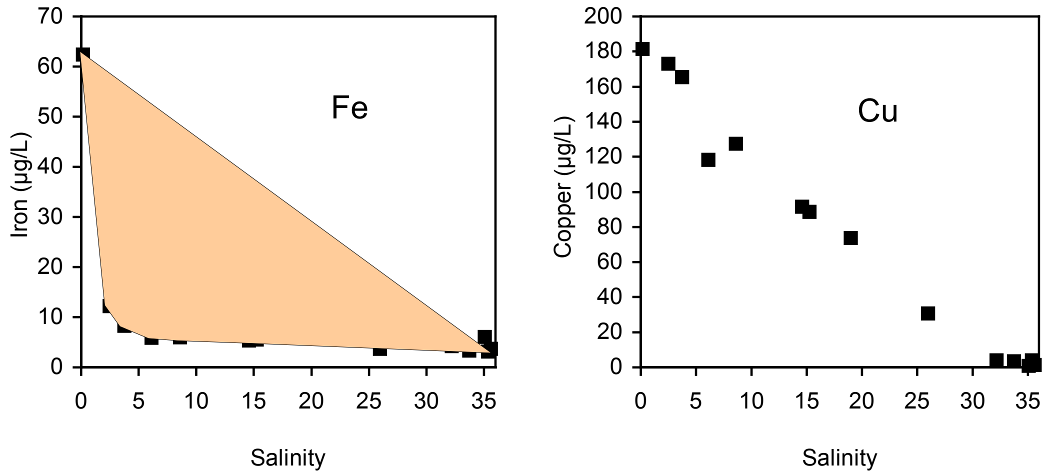


Figure 5. Low Fe (effective removal upstream by treatment plant at Wheal Jane): Removal of Fe still observed, albeit at reduced levels but Cu displays no evidence of removal from solution in Restronguet Creek during estuarine mixing. Note high Fe concentrations in freshwater

An exceptional and potentially problematic case has arisen recently for As where evidence from several profiles suggests there is now some re-release from sediments (or other sources in mid estuary) leading to the profile shown in figure 6. Since As is a class I Carcinogen it is recommended that investigation of this phenomenon be given priority to determine if it is consistent and significant, and to determine causes. The possibility of re-release of other metals from sediments should also be examined.

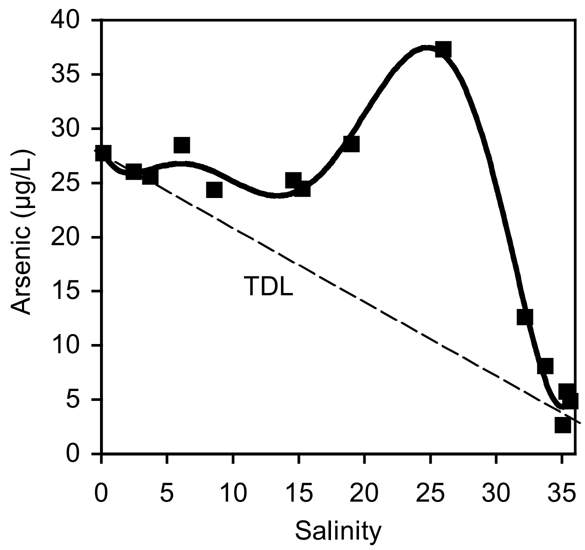


Figure 6. Plot of dissolved As against salinity, Restronguet Creek (21/10/97), showing marked deviation from the theoretical dilution line (TDL), indicating possible input of As from sediments in the estuary (Langston *et al.*, MBA unpublished data).

Non-conservative mixing might also become apparent where riverine water does not mix fully with the influent seawater. Under suitable circumstances, in a creek such as this, the less dense river water would be expected not to mix, but to flow over the denser salt water. Under estuarine conditions, several of the metals do become associated with particulate material, as indicated above, and iron and manganese themselves flocculate into particulate forms. This being so, both saline stratification and the way in which sinking material may be transported through, or trapped within, the creek, need to be assessed and taken into account. The recent observations on depth profiles confirm the stratified nature of the water column in Restronguet Creek (Warwick *et al.,* 1998). However at present all we can demonstrate is a broad picture of behaviour and provide some idea of its variability. To produce more powerful models would need a more detailed look at the nature of stratification and a better interpretation of the dynamics of the system. This is deemed necessary because metal rich freshwater may flow out from the Restronguet Creek on the surface leading to export of metals into the cSAC.

Fate of suspended particulate metals. A metal may become associated with the particulate phase in two ways; it may become sorbed onto existing particles, or it may itself form particulates. Under suitable conditions, colloidal iron dispersed in the water flocculates into particulate hydrated iron oxides. Its tendency to do this increases with increased salinity, as indicated above. This flocculated iron itself sorbs dissolved metals from the surrounding water. It is to be expected that metals will be deposited on the bed of Restronguet Creek sorbed both on mineral particles from the rivers, and on flocculated iron.

A conventional, sectionally-averaged (one-dimensional) hydrodynamic model of Restronguet creek was constructed in the ECoS simulation shell (Harris, 1992). This preliminary dynamic modelling suggested that the observed particle concentrations are consistent with a situation in which particles, which enter the creek from the rivers when the tide is in, are deposited on the bed, but are resuspended and carried out of the creek as the tide ebbs. It is concluded that the creek does not currently retain the bulk of the particulate material, which enters it from the rivers. Again this implies export of metals to the cSAC

Benthic Sediments.

The substantial temporal changes in water quality due to changes in mining activity, including abandonment and the installation of treatment plant at Wheal Jane, have not been reflected proportionately in the loadings of estuarine sediments. Here changes, if they are occurring, are relatively minimal and encompass longer timescales. Metal contamination in Restronguet sediments remain amongst the highest in the UK for a range of metals including Cu, Zn, Fe, Sn, As, and Mn, (Bryan and Langston 1992; Langston *et al.,* 1994a,b; see also review of pre-1991 NRA data in Millward, 1992). It is also likely that continuing inputs from the catchment such as the County Adit and spoil heaps will offset to some extent the beneficial effects of minewater treatment at Wheal Jane and the persistence of metals in sediments will therefore maintain elevated body burdens in some species thus delaying ‘recovery’ of estuarine ecosystems (see below). A threat of possible re-release of metals from estuarine sediments, as their chemistry is altered, has also been highlighted. Downstream effects of water improvement schemes are clearly a research topic which needs a high priority since such schemes could have a significant influence on the cSAC.

Thus, metals in Fal surface sediments generally display gradients in contamination which decrease with distance away from the major source in Restronguet Creek (table 3). Occasionally however high levels of Sn and Zn have also been reported in some of the creeks towards the tidal limit - at Truro, Calenick and the Tresillian River and are probably derived from clay workings and previous metal ore processing (Bryan *et al*., 1980; Langston *et al.,* 1994a,b). At depth, it is possible that signatures from historic inputs may also be detected in sediments. Cores at Tresillian for example have revealed horizons rich in Sn (up to 1800 $\mu g\ g^{-1}$), As (290 $\mu g\ g^{-1}$), Cu (1380 $\mu g\ g^{-1}$), Pb (508 $\mu g\ g^{-1}$)and Zn (2210 $\mu g\ g^{-1}$) at depths of 50cm, which appear to coincide with pulses of mine waste released during the peak of mining activity in the latter half of the 19th century (Pirrie *et al.,* 1997). Uncovering of such horizons through natural (weathering and erosion) or artificial mechanisms could result in the exposure of organisms to toxicologically significant concentrations.

Assessments of biological impact arising from metals are often restricted to *dissolved* forms – hence the EQS approach to predicting impact (described in a subsequent section). The situation regarding sediment metal toxicity predictions (also discussed later) is a much more complex issue. The identification of geochemical and biological processes which modify pollutant-organism interactions and exchanges is still not fully understood. There are however a number of reviews which illustrate how progress is being made in this direction, and demonstrate how bioavailability of sediment contaminants might be measured and compared (e.g. Bryan and Langston, 1992; Di Toro, 1989; Di Toro *et al*, 1991; Griscom *et al* 2000; Luoma, S.N., 1989; Swartz and Di Toro, 1997; Tessier and Campbell 1990; Langston *et al* 1998;). These approaches are largely based on direct observations in key bioindicator species, coupled with attempts at 'biomimetic' chemistry (chemical extractants, semi-permeable membrane devices) or speciation models. The question of sediment metals and their availability to biota is a particularly important issue in the Fal: existing regulatory criteria, derived from toxicity data for dissolved contaminants, may fail to protect the system where sediments are a major, and perhaps increasing, vector for contamination (Langston *et al*, 1994a,b).

To improve the overall assessment of biological impact a more detailed picture of the behaviour and fate of metals in Restronguet Creek, how they partition between sediments and water, and determination of relative assimilation from sediment is required. As will be evident from the following discussion of bioaccumulation there are few cases where the route of uptake is characterised fully.

Table 3. Fal Estuary: metal concentrations ($\mu g\ g^{-1}$ dw) in surface sediments (<100µm), August 1997 (data source: MBA)

Site	Map ref.	As	Cd	Cu	Fe	Mn	Zn
RESTRONGUET							
Perran	SW779387	1,885	2.73	2,761	64,547	564	3,267
Devoran	SW794388	3,939	3.32	4,507	95,418	635	5,809
Old Mine	SW803387	1,757	3.00	2,963	60,752	545	4,098
Old Mine River	SW803386	2,472	3.76	3,804	73,253	710	4,972
Opposite Penpol	SW808378	1,978	2.61	2,345	61,670	499	3,432
Penpol	SW813386	1,580	3.10	2,614	63,103	523	3,919
Pandora	SW813375	1,467	2.35	1,591	51,228	439	2,332
MYLOR							
Mylor Bridge	SW805361	435	2.06	1,275	41,232	397	1,540
Mylor Harbour	SW806359	415	1.32	1,101	41,981	397	1,368
PILL							
Pill	SW827385	245	1.09	661	31,336	288	891
Pill Up	SW827385	263	1.35	733	33,319	273	1,025
ST JUST							
St Just lower	SW848361	73	0.66	366	30,873	245	547
St Just upper	SW849364	65	1.18	371	31,551	254	547
PERCUIL							
Froe	SW868334	35	0.40	200	33,888	261	332
Percuil lower	SW857343	35	0.24	179	28,665	230	315
Percuil upper	SW865350	35	0.48	185	33,863	235	308
HELFORD							
Bennets.q	sw707265		1.05	272	32756	247	630
Bishops.q	sw722256		1.1	319	22196	214	662
Gweek	sw707265			255	26900	195	620
Gweek	sw707265		2	248	25600	276	963
Gweek	sw707265	23.2	1.41	252	28520	296	616
Helford.pass	sw762266		0.335	97.1	26350	376	214
Polpenwith	sw735275		0.02	328	27700	236	281
UK MINIMUM		1.68	0.003	1	2541	28	26

Bioaccumulation

The problems associated with the study of metal bioaccumulation have been outlined elsewhere (Bryan and Langston, 1992; Warwick *et al*., 1998), and it must be stressed that the assessment provided here is not comprehensive but is based on the 'best available evidence'. Some of the species are not necessarily ideal bioindicators because of their limited distribution and unusual metal handling features (e.g. oysters), nevertheless they are of importance locally. There is a need for basic information about the mechanisms controlling accumulation and toxicity, particularly the relative contributions of particulate versus dissolved metal.

Seaweed *Fucus vesiculosus*. Algae are potentially good indicators of water column metal bioavailability since there is no dietary component involved. Copper, iron and, to a lesser extent, zinc, arsenic, manganese and lead concentrations in macroalgae increase in progression from Carrick Roads (and other creeks in the Fal system) towards the mouth of Restronguet Creek, and subsequently upstream in the creek towards the upstream limit of distribution of the plants. (Data from a survey of the Fal in 1997 and an earlier survey of the Helford are shown in figure 7). This demonstrates the influence of Restronguet on the cSAC, in terms of metal bioavailability. Surprisingly, this distribution does not apply to cadmium, despite the fact that the major input of cadmium is the Carnon River. The presence of other metals, particularly zinc, is thought to suppress Cd uptake in Restronguet. Thus Cd levels in Fucus from the Helford Estuary are comparable to that in the Fal system whereas for other metals (As, Cu, Fe Zn) residues in the latter system show obvious signs of impact depending on proximity to the source of metals in Restronguet.

Estimates of concentration factors (CF = metal in *Fucus* ($\mu g\ g^{-1}$) ÷ metal in filtered water ($\mu g\ ml^{-1}$), based on field observations in the Fal, will clearly vary somewhat from site to site and from metal to metal. Examples are shown in table 4. In addition to competitive uptake, possible reasons for variation in the observed CFs include saturation of uptake sites at the most contaminated stations (n.b. Restronguet, Old Mine).

Table 4. *Fucus vesiculosus*: Concentration Factors (thousands) observed in Fal samples (data source MBA)

Site	As	Cd	Cu	Fe	Mn	Zn
Restronguet (Old Mine)	18.6	0.823	27.6	66.5	0.285	0.706
Restronguet Point	32.8	4.0	68.5	59.5	2.94	7.19
Mylor Harbour	22.9	6.83	45.2	331	9.6	7.1
Pill	19.8	7.6	72.1	115	3.9	5.67
St Just	23.4	6.4	40.4	101	4.8	3.4

Samples collected August 1997.
CF= metal in *Fucus* ($\mu g\ g^{-1}$) ÷ metal in filtered water ($\mu g\ ml^{-1}$).

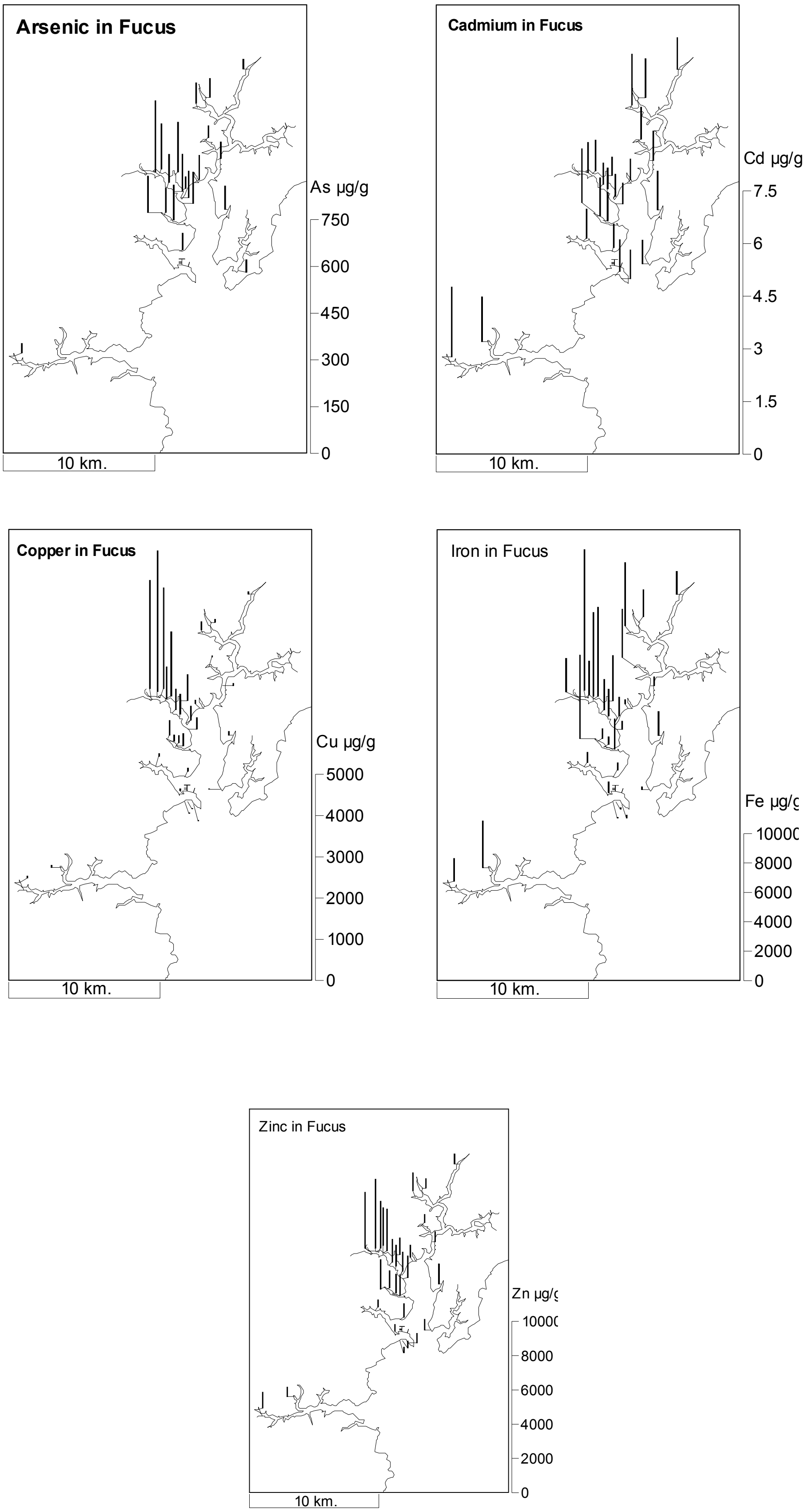
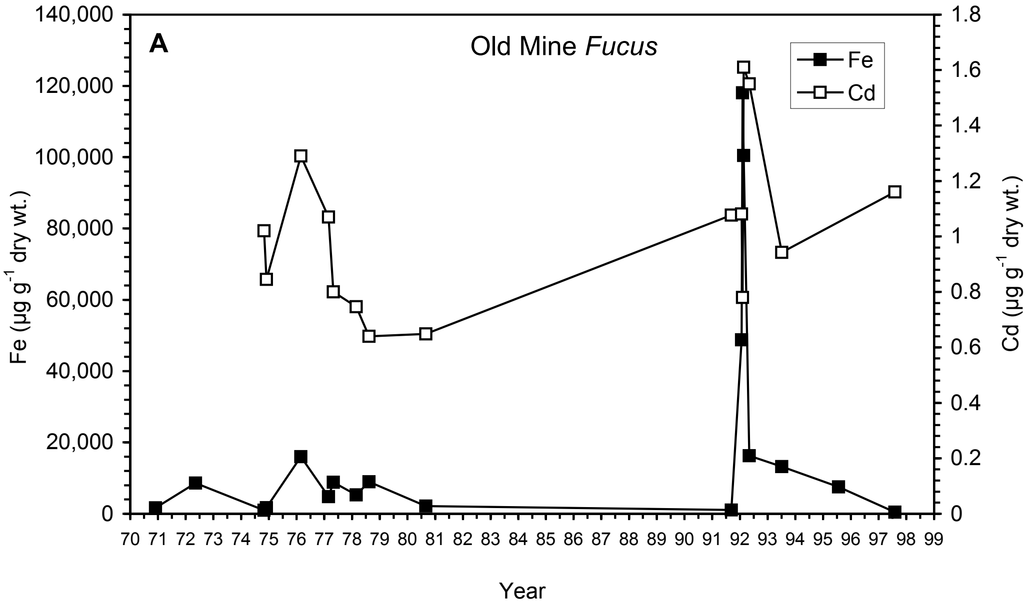


Figure 7. ***Fucus vesiculosus*****: Metal concentrations in samples from the Fal and Helford cSAC (Data source MBA; mean values after exclusion of 1991/2 data)**

MBA temporal trend data for metals in *Fucus vesiculosus,* collected in Restronguet Creek (old mine site), are depicted in Figure 8. There was no evidence of long term change prior to mine abandonment. The major influx of minewater in 1992, caused by the failure of the plug in Nangiles Adit, had a significant impact on Zn and Fe levels in *Fucus*, much less so for Cu, Cd and Mn (perhaps due to competition). Concentrations in algae from the August 1997 survey appear have returned to pre-flooding values though this return to 'steady state' has taken several years to achieve.



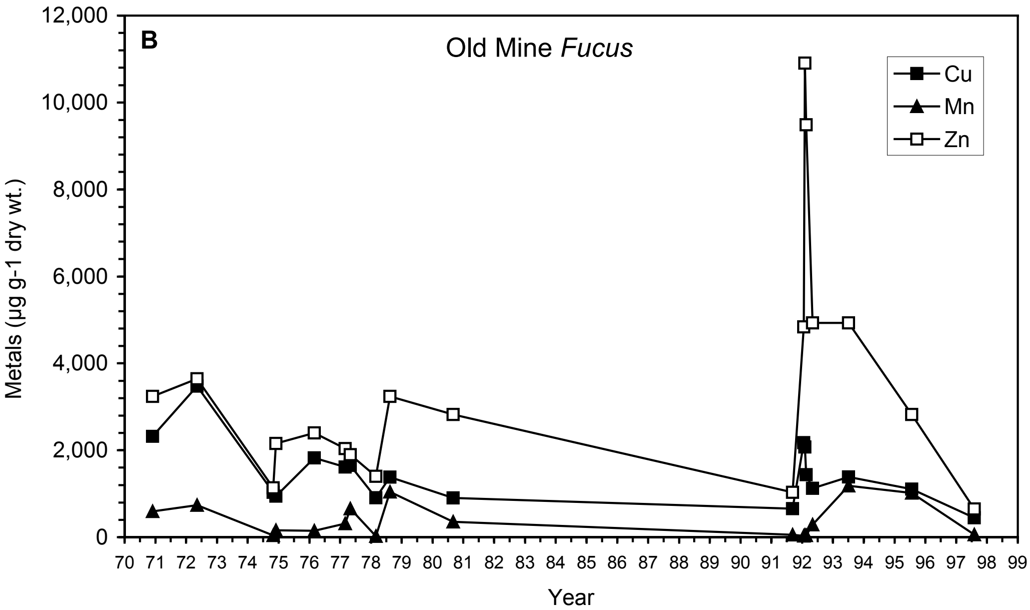


Figure 8. ***Fucus vesiculosus*: Concentrations of Fe and Cd (A) and Cu, Mn and Zn (B) in Samples From Old Mine (Grid Ref SW 8030 3862), Restronguet Creek, 1970-1997 (data source MBA)**

There are limited amounts of data available for *F.vesiculosus* at other sites, depicting the impact of the January 1992 incident. These originate mainly from NRA/EA surveys. At Carlys Rock – in Carrick Roads - a transient increase could be detected for Cd, Cu and Fe some 3 weeks after the discharge (figure. 9). Five months later concentrations of these metals had returned to pre-incident levels, however, Zn appears to have been more irreversibly bound. This illustrates that metal discharges from Restronguet Creek can impact on the biota of the cSAC.

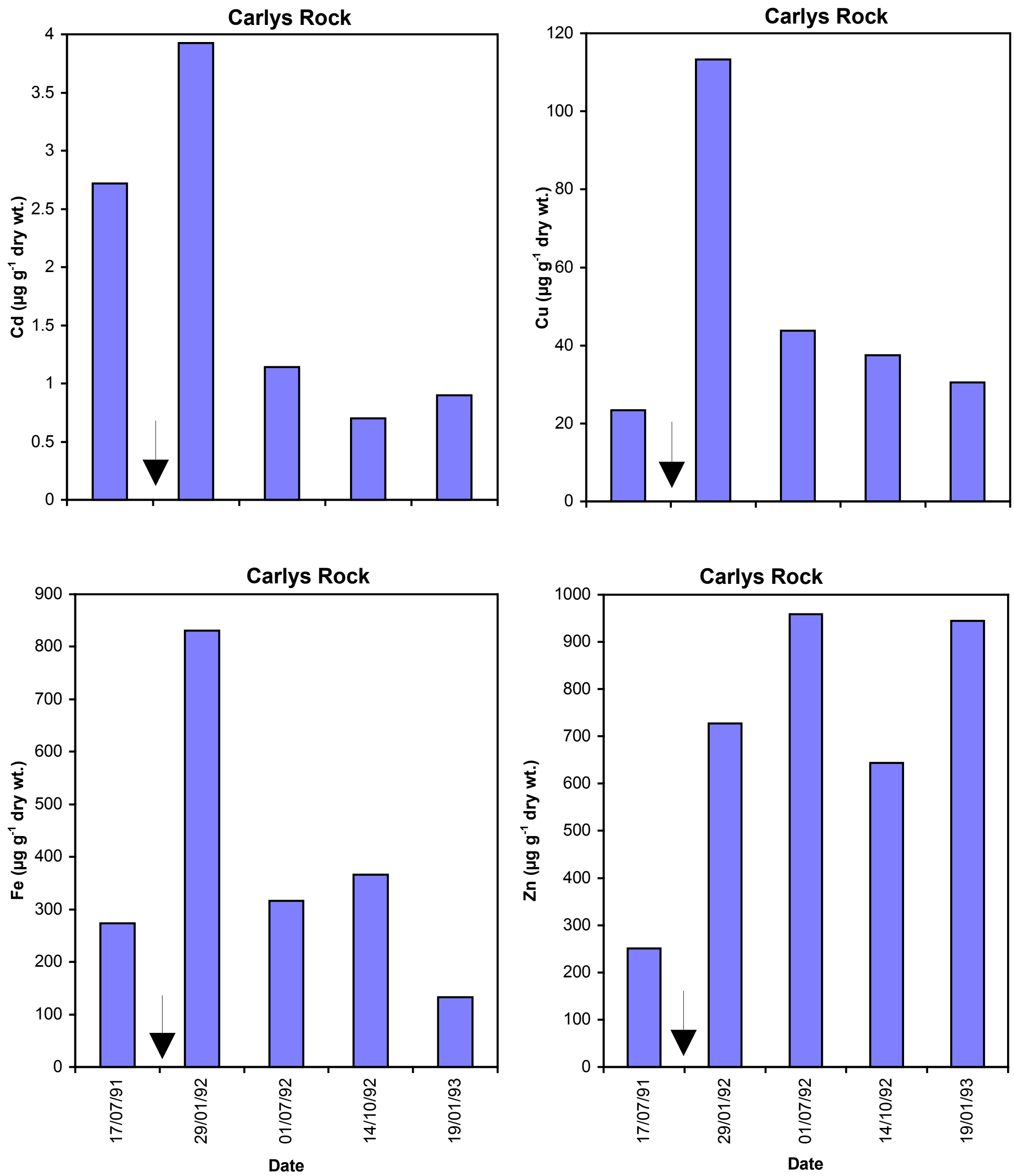


Figure 9. *Fucus vesiculosus*: Carlys Rock: Changes in Metal Concentrations Resulting from the Wheal Jane Incident, January 1992 (arrowed) data supplied by the EA/NRA.

Oysters *Ostrea edulis*. The Fal Estuary represents one of the few commercially exploitable fisheries for the native oyster in the UK. The principal stocks are at Messack Point (near St Just) and Parsons Bank (Turnaware Bar), though they occur in other areas and occasional specimens can be found on the shore at extreme low water (spring tides) at the mouths of many of the Creeks entering Carrick Roads, even Restronguet.

In the laboratory experiments, levels of Cd, Co, Cr, Cu, Hg, Ni, Pb and Zn (but not As) in oysters (as a taxonomic group) generally tend to reflect concentrations in surrounding water (an essential requirement for an indicator species). Under complex field conditions these relationships may be modified depending on accumulation pathways. Changes in filtration rate in *O. edulis* - perhaps induced by variable amounts of suspended solids – are known to affect assimilation of Cu from water (Martincic *et al*, 1986). A more direct route of accumulation of Zn, from particulates, has also been described (Martincic *et al*, 1987). Predictions of bioaccumulation can therefore be confounded by uptake route, and also by metal-metal interactions.

In the mid-nineteenth century, as mining activity peaked, 'greensick' oysters - discoloured by absorption of Cu from the water - were reported to be the cause of a number of poisoning events in consumers of Fal stocks. However, the concentrations of Cu (estimated at 3 000 $\mu g\ g^{-1}$ dry weight - compared to current values of 500-1 000 $\mu g\ g^{-1}$, see table 5) were not toxic to the shellfish themselves (O'Shaugnessy, 1866). This is due to the development of efficient detoxification systems in oysters, discussed later.

The long term influence on the Fal, caused by contamination from Restronguet Creek, can be gauged, in general terms, by historical observations of Cu and Zn body burdens in oysters (figure 10), though there are caveats (data prior to 1970 are probably reliable but of uncertain quality; dry weight conversions have been estimated for Ortons 1921 data using a wet:dry weight ratio of 5; samples are not from identical locations, and may be subjected to seasonal and size variability; intraspecific variability for Cu and Zn is high, some individuals appear better accumulators - or regulators - than others).

The presence of consistently high Cu concentrations in Fal oysters since records began contrasts with the more distinctive patterns of copper mining activity (peaking about 1850; reducing to zero in 1900). Though there may have been a slight decreasing trend in the Cu content of oysters since the early 1970s (see figure 10A), overall, the scale of ore production is not itself the dominating influence. A combination of contaminated sediments, continued drainage from other old mines and leaching of spoil-heaps contribute to elevated body burdens. Monitoring of oyster stocks at Parsons Bank and Messack Point by MAFF prior to and following the Wheal Jane incident also indicates that the exceptional discharge in 1992 had little additional effect on overall Cu concentrations in these populations (present in the Carrick Roads off the mouth of Restronguet Creek (figure 10 B)

Zn in oysters, likewise, has not changed systematically or significantly over the period for which data are available (see figure 10C and D). MAFF monitoring of stocks at Parsons Bank and Messack Point in the early 1990s (MAFF, 1994), and additional sampling at St Just and Mylor by MBA in 1997, also support conclusions that Zn burdens are probably not related to short-term inputs, but to the long-term historical contamination. Similar conclusions can be drawn for Cd and Fe (see table 5).

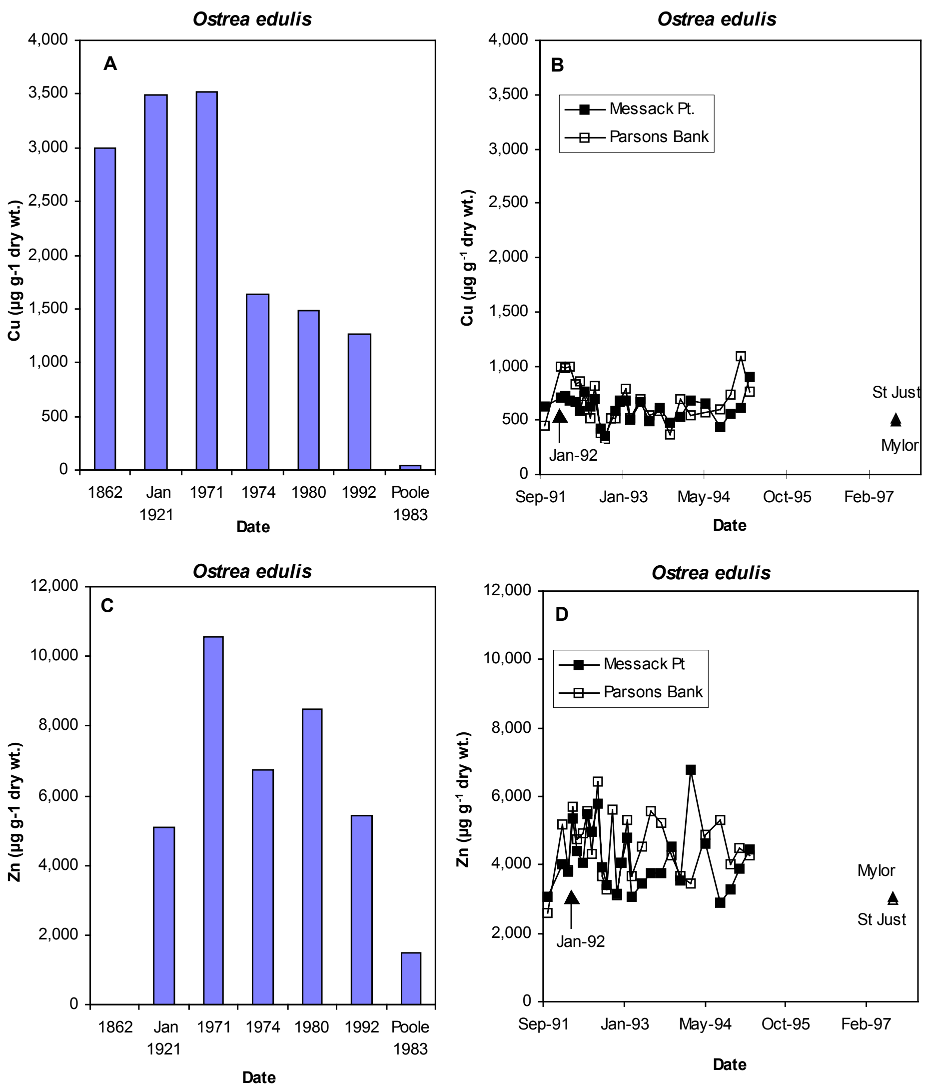


Figure 10. *Ostrea edulis*: Historic and recent data for Cu (A, B) and Zn (C,D) in oysters from Carrick Roads (Poole Harbour for comparison). Data sources: Bryan *et al*. 1987, MAFF 1994, own unpublished data. Arrows indicate flooding incident at Wheal Jane mine.

The 1992 discharge from Wheal Jane 'gave no cause for concern with regard to the consumption of oysters' (MAFF, 1994). Nevertheless, the fact that oysters from the Carrick Roads retain consistently elevated concentrations of Cu and Zn, relative to, for example, Poole Harbour oysters (see table 5) points to the extensive chronic contamination in the Fal cSAC.

Oysters from near the mouth of Restronguet Creek contain slightly higher metal levels (Cu, Zn), compared to stocks at the opposite side of Carrick Roads, and undoubtedly the Creek represents determining input for the area. However, the gradients and trends in contamination, reflected by oysters, are perhaps not as marked as expected from environmental (water and sediment) loadings. A combination of detoxification systems, described above, are presumably the explanation, though the mechanisms involved, the scale of their contribution, and their 'spare-capacity' in Fal oysters are not fully understood.

Based on comparisons between Fal and control (Poole) oysters (table 5) the elements most likely to be accumulated are Cu, Pb, Zn and Cd, though the latter element may be subjected to competition for uptake by Zn or displacement by Cu and Mn. Fe, Mn and As burdens are not significantly enhanced, suggesting that tissue concentrations are not entirely dependant on levels in the water: there may be a degree of regulation for these elements. There is no evidence currently available to suggest that any of these elements are directly harmful to existing oyster fisheries in Carrick Roads, and the populations here may be adapted (though this has yet to be tested): Cu and Zn would probably be suspected of representing potentially the most significant threat.

Table 5. : *Ostrea edulis*: Metals concentrations (μg g^{-1} dry weight) in samples from sites in the Fal Estuary compared with uncontaminated oysters (Brownsea, Poole)

	As	**Ag**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**	**Hg**
Fal Estuary												
Weir Point, 1992	12.4	0.05	3.25	0.53	2.04	1575	282	12.7	3.33	2.12	8723	
Restronguet Pt., 1992	10.9	0.01	5.23	0.57		1256	266	8.2		1.50	8408	
Mylor Creek, 1992	23.0	0.10	6.52	1.09		1489	378	14.2		2.92	7828	
Mylor Creek, 1997	6.35		1.681			483	292	10.6			3064	
Loe Beach, 1992	6.6	0.34	3.88	0.53		1052	306	10.0		1.69	8626	
Parsons Bank 1991-95 Mean (SD)			2.03 (0.51)		3.32 (1.82)	668 (204)	345 (112)		1.99 (1.66)	1.72 (1.08)	4502 (934)	0.08 (0.03)
Messack Pt. 1991-95 Mean (SD)			2.08 (0.41)		3.13 (2.07)	611 (117)	295 (122)		1.75 (1.76)	1.24 (0.45)	4163 (939)	0.10 (0.03)
St Just, 1997	5.85		1.84			518	227	12.1			2974	
Poole Harbour												
Brownsea Isl., 1983	3.80	2.85	1.56	0.32		40	140	8.0		0.35	1500	0.16
Ratio Fal (max.) / Brownsea	6.05	0.12	4.18	3.37		39.78	2.70	1.79		8.44	5.82	0.59

Data sources: data for Fal Estuary for Parsons Bank and Messack Point are means (±SD) for the period 1991–95, based on MAFFs monitoring scheme and have been converted to dry weights. Other Fal data are MBA unpublished results. The St Just and Loe Beach samples are adjacent to Messack Point and Parsons Bank, respectively.

Thus, spatial and temporal trends in contamination displayed by oysters are not as marked as expected, given the environmental (water and sediment) gradients. It seems unlikely that relatively minor changes in consents will have any impact on oyster populations in terms of their metal burdens.

Cockles *Cerastoderma edule.* These shellfish are reasonable accumulators and indicators of Ag, Ni, Cd and several other metals, but underestimate Zn and Cu, except at very high concentrations; at lower levels body burdens of these two metals appear to be regulated. *C. edule* is probably a reasonable indicator for As, but particulate contamination in the animal may lead to overestimates of Fe, Cr and Pb availability. Examples of body burden data are given in table 6.

C.edule is fairly widely distributed throughout the Fal Estuary, but is rarely found in Restronguet Creek, except perhaps when washed in by the tide, when exceptionally high Cu burdens may be encountered (table 6). Copper concentrations between 10-20 times background (signifying export of metals from Restronguet) have also been detected in the occasional samples of cockles recovered from Mylor, whilst in most other creeks copper concentrations in *C.edule* are not usually elevated to such an extent. Concentrations of Cu and Fe were elevated at the upper Percuil Creek site by approximately 5 fold, relative to controls, and Zn levels in several samples were up to 2 fold higher than baselines. Other metals (As, Cd) were not significantly enhanced. Concentrations in cockles from the Eastern side of Carrick Roads are thus significantly below toxicity thresholds and their survival is less threatened compared to those in the western creeks.

Zinc concentrations in Mylor and Restronguet cockles exceed background by up to three-fold suggesting that regulation of body burdens has broken down. Pb, Ag, Cr and Fe bioaccumulation is also evident, compared with Appledore controls (table 6). However, it is the accumulation of copper which is thought to be most damaging to Restronguet (and perhaps Mylor) cockles. Moribund cockles were found on the surface of the sand flats at Restronguet Passage immediately after the Wheal Jane discharge incident, caused by the massive increase in dissolved metals, which may have taken its toll before steady-state conditions could be achieved in the animals. No spat or juveniles have been found in Restronguet sediments which indicates that the presence of adults is a transient phenomenon (see, for example, Bryan and Gibbs, 1983).

The route of metal uptake in *C.edule*, seems to be primarily via water - rather than directly from sediments - though desorption of sediment-bound metal may be an important source of dissolved metal. Bioaccumulation during experimental exposure of control cockles to Restronguet muds was found to be negligible, if the sediments were supplied with clean flowing sea water (Bryan and Gibbs, 1983). Under static conditions, leaching of Cu and Zn to overlying sea water (30 and 270 $\mu g\ l^{-1}$, respectively) led to mortalities; body burdens in survivors (242 $\mu g\ g^{-1}$ Cu and 404 $\mu g\ g^{-1}$ Zn) were comparable to those of moribund animals from Restronguet in January 1980 (see table 6) and probably represent the critical body burden in *C.edule*. Similar data would be useful for other metals and other species, in helping to interpret the threat of biological damage in the Fal Estuary. The ability to extrapolate from environmental levels (water/sediment) to lethal burdens or sub-lethal stress indices is also an important goal, but unfortunately is hampered by the lack of information and basic research.

Table 6: *Cerastoderma edule*: Metal concentrations (µg g^{-1} dry weight) from sites in the Fal Estuary and Appledore, North Devon (control)

Site	Date	Ag	As	Cd	Co	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Zn
Feock	Jun-76	0.13		0.38	2.14	2.95	12.8	431		2.1	18.0	2.8	87
Penryn	Jul-76			0.52	2.52	4.10	4.7	328		3.3	19.4	6.5	92
Place Cove	Mar-79	0.04		0.40	3.89	0.70	5.0	572		3.2	27.6	2.5	79
Devoran	Oct-79	0.03	20.6	0.34	1.94	2.11	13.3	1904	0.13	17.2	22.7	6.0	117
Restronguet Passage	Jan-80			2.10			486.0	503		87.6		21.0	108
Restronguet Passage	Jan-92			0.12		0.84	47.0	554	<0.01		2.3	0.8	267
Mylor Mid	Mar-80	0.22	18.6	1.34	2.82		59.0	1079	0.32	7.0		4.4	106
Mylor Upper	Mar-80	0.40	28.2	1.79	4.09	3.94	97.8	2437	0.28	38.2	27.2	14.1	271
Cowlands	Apr-80	0.06	27.6	0.65	2.52	5.64	8.0	1189	0.23	8.6	40.2	3.3	99
St.Just	May-80	0.07	11.7	0.52	3.08	0.89	7.8	760	0.24	4.3	36.0	2.1	79
St Just upper	Aug-97		6.54	0.26			7.21	485		2.67			79.2
St Just lower	Aug-97			0.18			4.62	275		2.7			51.8
Percuil upper	Aug-97			0.33			26.9	2418		4.01			84.2
Froe	Aug-97		11.2	0.22			9.65	654		17.7			103
Appledore	Sep-80	0.01	9.8	0.44	2.36	0.54	4.2	431	0.14	18.9	26.9	0.36	46
Ratio Fal (max.) / Appledore		33.3	2.88	4.83	1.73	10.4	115	5.65	2.29	4.65	1.49	58.3	5.87

Data sources: MBA data, except for Restronguet Passage, Jan 92 (NRA).

Metal concentrations and distributions of other invertebrate species. MBA, MAFF and EA/NRA have extensive bioaccumulation data on several other bioindicator species found in SW estuaries including polychaetes, clams, mussels and winkles (see Bryan *et al*., 1980, 1985; Langston *et al.,* 1994a,b; NRA, 1994b). Metal concentrations in Fal mussels are often in the upper part of the UK range, but are not considered exceptional by molluscan/invertebrate standards. This may reflect, partly, their ability to regulate essential metals such as Cu and Zn, and partly account for the absence of mussels – relatively sensitive species - from the most contaminated parts of the system (mussels are not found further upstream in the Fal system than Percuil on the eastern shoreline of Carrick Roads and Flushing on the western shoreline). Nevertheless most metals are enhanced by a factor of 5-10 relative to baseline values (MBA data - see Langston *et al.,* 1994a.b, Warwick *et al.,* 1998, and summaries of MAFF surveys by NRA, 1994b)

Zinc and copper in winkles (*Littorina littorea*) are also at the upper end of the range found in the UK though these molluscs appear to be more tolerant than mussels to metals and have a wider distribution throughout the Fal. On the western shoreline *Littorina littorea* is found at Loe Beach just outside Restronguet Creek (but not inside) and also at King Harry Ferry. Flat periwinkles *Littorina obtusata* (+*mariae*) are able to survive just inside Restronguet Creek (as far as Pandora): this extended tolerance and distribution is reflected in extremely high body burdens of copper and zinc (and to some extent As). (MBA data - see Langston *et al*., 1994; Warwick *et al.,* 1998, and summaries of MAFF surveys by NRA, 1994b). Though elevated, levels of

Cd are not excessive by UK standards. This probably reflects competition for uptake sites by other metals.

The general scarcity of certain crustacean species (for example, *Corophium* and *Cyathura*) within parts of the Fal estuarine system is discussed below. The conspicuous absence of bivalve and gastropod molluscs from highly metal-contaminated sites in Restronguet Creek is a consequence of the long history of metal mining in that area (Bryan *et al.*, 1987). Because of the legacy of contaminated sediments and the persistence of associated metals it seems unlikely (and impractical) that changes in consents will bring about substantial change, at least in the short-term.

Cu and Zn are believed to act by inhibiting the settlement of juvenile bivalves, including *Cerastoderma edule* and *Mytilus edulis* and the influence of these metals may well extend beyond Restronguet into the Fal Estuary. The clam *Scrobicularia plana* is similarly affected by metals. Limited populations of *S. plana* can be found in lower reaches of Restronguet Creek, though these are patchy and restricted to the margins, at the position of mid-high water level (corresponding to the lowest metal concentrations, furthest from the river channel). Oysters are also restricted to the mouth of the estuary.

Not all species are excluded from the upper reaches of Restronguet. Populations of the ragworm *Hediste* (*Nereis*) *diversicolor* (and to a lesser extent *F. vesiculosus*) have become adapted to the high levels of metals through development of detoxification and exclusion mechanisms (see for example Bryan, 1976). In the case of *Hediste* there is a clear genetic component, and selection of populations for Cu and Zn occurs up to 1km downstream of the source (Carnon River) at sediment concentrations above 1000 and 3500 µg g^{-1} dw respectively (Hateley *et al.,* 1989, 1992; Grant *et al.,* 1989). In general however the biota of the creek are extremely impoverished. The NRA surveys of the creek prior to the Wheal Jane discharge confirm this picture, with diversity increasing towards the mouth of the creek (25 species) (NRA, 1994).

Significance of metals on individual species and Biomarker Studies

The discovery of adaptation in *Hediste* and *Fucus*, and toxicity data for whole organisms such as *S.plana* and *C.edule*, illustrates that organisms are capable of a range of specific responses to the impact of metals. Since the demonstration by Bryan and co workers that epidermal granules were able to immobilize potentially toxic metals in *Hediste,* and that a reduction in permeability in Fucus can reduce metal uptake there have only been a handful of studies which have explored the mechanisms responsible in other species. Most work has focused on the adaptations to metals in oysters which partly involves localisation of Cu (and Zn), in granular form, in amoebocytes (blood cells) (George *et al.,* 1978; Pirie *et al.*, 1984). The green discoloration of these cells is proportional to Cu content (Boyce and Herdman, 1898; Orton 1923). Localisation of metals within vesicles, in inert form, enables the oysters to tolerate body burdens of Zn and Cu, which are 10 and 100 times background, respectively (compare Cu and Zn burdens in *O. edulis* at unpolluted sites with those from Restronguet Creek, in Table 5).

Another mechanism for tolerance concerns the metal binding protein metallothionein (MT) whose induction may signal a sub-lethal response to metal pollution and which may, therefore, have potential as an early warning indicator of deleterious effects (Langston *et al.,* 1998). Oysters produce MT in response to metals such as Cd, Cu, Hg and Ag in the laboratory and there is some indication, from measurement of MT induction that oysters are attempting to adapt to metal stress in the Fal Estuary (Langston *et al.,* 1998). Thus MT levels in gills of oysters from Loe Beach were 1.25 mg/g compared with 4.02 in samples from Restronguet. Corresponding Cu levels were 1681 and 2875 $\mu g\ g^{-1}$ dry weight. The extent of MT involvement in natural oyster populations may be relatively small and requires wider validation. Nevertheless this assay could provide a useful measure of sub-lethal stress, and could be used to measure improvement or deterioration of condition. Other intra-cellular ligands, particularly very low molecular weight soluble compounds, are important in binding high levels of Cu and Zn in oysters in highly contaminated environments: their role in detoxification also needs further clarification if the significance of the unusual body burdens in oysters is to be fully understood.

The measurement of MT in winkles and in transplanted caged mussels has been shown to be a useful measure of sublethal stress and biological impact. The latter is a particularly useful alternative to the use of native animals particularly in parts of the Fal where indigenous populations are absent. UoP studies in Restronguet have demonstrated useful markers of condition which include characterisation of MT in the shore crab *Carcinus maenas*. MT levels in gills responded proportionately to Cu and Zn exposure gradients in sediments at three sites in the Fal (Restronguet, Mylor and Percuil) and were significantly elevated compared with control sites in the Yealm and Avon estuaries (Pedersen *et al.,* 1997). A further study showed that crabs from Restronguet may be less able to adapt to changes in salinity (osmoregulate) compared to individuals from control sites (Bamber and Depledge, 1997).

Further work is recommended to extend bioaccumulation data and responses of some of the above key benthic organisms in the Fal, in relation to changes resulting from water treatment. These studies are needed to evaluate the success of clean-up measures and to predicting long-term changes, whether induced by natural or anthropogenic causes. The assemblages found in Restronguet in particular are unique in terms of their adaptive status and worthy of long-term research in their own right. Studies of the adaptation and responses of individual species also provide causative links to the higher order (community level) responses discussed below.

Improved mapping (and re-mapping) of the genetic composition of tolerant populations (*Hediste* and others) would be useful in this respect and would add an interesting temporal dimension to the anticipated ‘recovery phase' following water-treatment measures at Wheal Jane.

Significance Of Metal Inputs On Biological Communities*

The Carnon River

There are very few published biological data for the Carnon River itself, but the Agency (Bodmin) have provided the raw data from two surveys of aquatic macrophytes undertaken in summer 1992 and summer 1993, and eleven surveys of macro-invertebrates taken in spring, summer and autumn 1990, 1992 and 1993, and in spring and autumn 1995. All these surveys were conducted at seven locations between Chacewater Sewage Treatment Works and Devoran Bridge. There appear to be no clear seasonal or temporal trends in these data, and the average values of various biological indices are summarised in table 7.

Table 7: Average Values of Biological Indices from Agency Surveys of the Carnon River (from Warwick *et al.,* 1998)

	Macrophytes		Invertebrates		
Location	Species	Score	Taxa	BMWP	ASPT
Chacewater STW (SW 7541 4326)	4.5	17	10.9	54	4.9
Twelveheads (SW 7615 4206)	0.5	3	9.2	46	4.9
County & Wellington adit (SW 7695 4150)	0.5	4	2	6.9	2.4
Hicks Mill Stream (SW 7670 4113)	0	0	8.7	39	4.4
Bissoe Bridge (Carnon) (SW 7748 4128)	0	0	4.1	18	4.1
Bissoe Bridge (Baldhu) (SW 7760 4144)	0	0	0.8	3.3	2.3
Devoran Bridge (SW 7909 3942)	1.5	5	4.5	18	3.8

The Carnon River is almost devoid of macroscopic plants in the stream channels, except at Chacewater, which is well upstream of the Wheal Jane mine complex. The BMWP (Biological Monitoring Working Party) scores, based on the relative sensitivities of the recorded invertebrate taxa to organic pollution, are exceptionally low. However, the average BMWP score per taxon (ASPT) is relatively high, suggesting that it is not organic pollution but toxic contaminants that are responsible for the absence of biota. Thus in the Local Environment Area Plan Consultation Report (Environment Agency, 1997) the lower reaches of the river, and the Baldhu Stream, are classified as having Biological Class, f (bad) and Biological Class, e (poor). This report also states that there is little scope for biological improvement in the near future.

The Carnon River is also devoid of fish, although other rivers entering Restronguet Creek such as the Kennall support brown trout fisheries.

The Fal Estuary Complex: historical evidence

Rostron (1985) has provided a comprehensive account of the ecology of the Fal Estuary based on surveys undertaken by the Field Studies Council in May 1985 (intertidal) and June-July 1985 (subtidal), and on a thorough review of previous studies. That report provides a picture of the distribution of most groups of

* [from the report by Warwick et al, 1998]

macroscopic organisms, but the survey did not include the most metal-contaminated creeks (Restronguet, Mylor). A more detailed study of Restronguet Creek was undertaken by Holliday and Bell (1979). It is not the intention of this report to reiterate this information in detail, but to interpret this information in terms of the significance of water and sediment quality on the floral and faunal communities. More recent research undertaken in the estuary has been used to supplement this information.

Over the centuries a marked gradient of sediment metal concentrations has built up in the Fal Estuary system. Sediments in otherwise similar creeks in different parts of the system have levels of heavy metals (including copper, zinc, cadmium), which differ by orders of magnitude. The source, and most heavily contaminated of these, is Restronguet Creek, followed by Mylor Creek, Pill Creek and St Just Creek through to Percuil Creek (lowest). (Bryan and Gibbs, 1983; Somerfield *et al.*, 1994a, 1994b). For this reason the Fal Estuary system has been the site for a number of studies on the effects of long-term heavy metal pollution on marine organisms (Bryan and Gibbs, 1983; Bryan and Langston, 1992).

Obvious deleterious effects of heavy metal discharge from the mouth of Restronguet Creek are probably restricted to a comparatively small area. However, Mylor Creek has been strongly influenced by transport of metals from Restronguet (leading to the unusual gradients in Mylor sediments, with concentrations of most elements decreasing upstream). Work undertaken as part of the Wheal Jane project was able to demonstrate that stratification of the water column in Restronguet Creek is marked under certain conditions of tide and flow and leads to significant export of metals in a plume of low salinity surface water. This illustrates that outflow from Restronguet has the potential to affect the cSAC even though it is not itself part of the cSAC. Forecasting the impact of metals on the Fal /Helford cSAC strongly depends on an understanding of the spatial and temporal behaviour of metals in Restronguet Creek. This system also provides a unique opportunity to test hypotheses on ecotoxicological effects, including community level responses, on estuarine and marine biota.

Historically, apart from a benthic survey carried out in 1979 (Bryan and Gibbs, 1983), and some casual observations, little has been published on the effects of metal discharges from Restronguet Creek on the fauna and flora of Carrick Roads. Bryan and Gibbs (1983) concluded that the effects of heavy metal discharge from the mouth of Restronguet Creek were minimal. Most of the species recorded in their survey area, in the northern part of Carrick Roads between Pill Creek and Mylor Creek, were found in the vicinity of the mouth of Restronguet Creek. There was a suggestion that the east bank was more productive than the western side, but this may have been a reflection of the sediment types. A more comprehensive survey of the sublittoral biota of the estuary, carried out by the Oil Pollution Research Unit (Rostron, 1985), found that communities were generally impoverished over most of the area, and concluded that metal contamination and the presence of china clay wastes were at least in part responsible for this impoverishment. That being said, there are communities and species of interest within the estuary, but these are generally confined to the outer part.

A further exceptional feature of the Fal arises from the fact that metal concentrations in some areas have been elevated for such a long period that events in the estuary cannot be viewed as simple pollution/recovery incidents. The induction of metal tolerance has been demonstrated for certain populations indicating that responses to metal toxicity may have been modified. (What happens to these responses, and populations, as pollution pressures are removed, remains to be investigated). It is also important to consider that a mixture of metals is involved, and that concentrations of different metals co-vary, making it difficult to ascertain which are responsible for observed effects. Chemical interactions can also confound simple predictions. Nevertheless, gradients in heavy metal concentrations (in waters and sediments) correlate most strongly with observed effects – varying from biochemical adaptations to the composition of the invertebrate communities - and are therefore the most likely causal agents. Work undertaken as part of the Wheal Jane Minewater project provided a more recent synthesis of impact

Macrobenthos. Analysis of survey data by a range of multivariate statistical techniques found in Clarke and Warwick (1994) shows that Restronguet creek has a distinct community composition compared with Mylor, Pill, St Just and Percuil creeks (Somerfield *et al.,* 1994a,b; Warwick, 2001). The ordination by PCA of environmental data showed that all the creeks were different, stations in Restronguet Creek being widely separated from the rest. Sediment copper concentrations in Restronguet Creek are in the region of 2 500 $\mu g\ g^{-1}$, whereas in all the other creeks, Cu concentrations range from 100–1 200 $\mu g\ g^{-1}$. It is unlikely, however, that the actual value of such a threshold would be of universal significance as the communities in these creeks must be adapted to survive high metal concentrations. Thus, Rygg (1985), for example suggested that a copper concentration of 200 $\mu g\ g^{-1}$ represented a threshold concentration for communities in a Norwegian fjord. The prediction of effects in an area such as the Fal is, to say the least, problematic in view of different abilities to adapt to contaminants and the long history of exposure to varying degrees. Similarly it is impossible to determine which effects to special interest features and Fal biota are directly attributable to metal contamination. Furthermore, the lower Cu ranking sites (Mylor, Pill, St Just and Percuil) were not ordered in a pattern that is entirely consistent with decreasing metal concentrations. This suggests that other factors are also influencing the community structure of macrofauna in these creeks.

Neither the sediment metal concentrations nor the macrofaunal community in Restronguet Creek altered significantly between November 1991 and March 1992 following the discharge of untreated minewater from the Wheal Jane mine in January 1992. Subsequent work indicates that neither metal levels in sediments, nor significant community changes have occurred since then (Warwick *et al.,* 1998 and unpublished observations).

Compared with other less contaminated estuarine systems in south-west Britain, the Fal has a very low abundance of the crustacean *Corophium volutator* and of *Cyathura carinata*, whilst certain small annelid worms (e.g. tubificids and spionids) are more abundant in the Fal than other estuaries (Rostron, 1985; Warwick *et al.,* 1998; Warwick, 2001). Metal pollution is clearly implicated as the cause of these differences. Crustaceans are known to be among the most sensitive of all marine taxa to pollution, including heavy metals (Rand and Petrocelli, 1985). Organic enrichment is less implicated since the capitellid polychaetes, which are regarded as indicators of organic enrichment, were absent from the Fal samples.

There is thus a tendency towards relative dominance by fewer species (most marked in Restronguet Creek) and the distinctive nature of the macro-invertebrate communities of the Fal are undoubtedly due in part to metals. These features have implications for higher trophic levels in the system (wading birds and demersal fish), which rely on the macrobenthos for food. To date however, the consequences for top predators has not been fully evaluated, though some attempt at quantifying invertebrate infauna in a survey conducted in 1990 was made by Baker (1994). For the most part observations on various components of the habitat are of a qualitative, subjective nature (see below).

Meiobenthos. Analyses of survey data (1991,1992) show a gradation in nematode community structure consistent with increasing metal concentrations, particularly Cu (Somerfield et al. 1994a and 1994b). Restronguet Creek is clearly separated from all others, which are ordered from Mylor Creek, through Pill Creek and St Just Creek to Percuil Creek, in an order consistent with gradients in metal concentrations. Nematode samples from within Restronguet Creek are also ordered in relation to proximity to the source of input at the head of the Creek. Copepods from Restronguet Creek are, likewise, separated from other Fal populations though not in a form that is so readily related to the metal gradient. Both sets of analyses suggest, as do analyses of macrofaunal communities, the possibility of threshold responses to increasing metal concentrations.

Large differences in the Cu tolerance of nematode communities from different parts of the Fal (and other estuaries) have been confirmed (in standard toxicity tests), to correlate with their previous history of exposure (i.e. level of Cu in sediment). Specimens from the more contaminated Restronguet and Mylor Creeks display a marked increase in resistance to Cu compared with individuals from Pill, Cowlands and St Just, though in turn these are more impacted than Percuil and a control site at Kingsbridge (Millward and Grant, 2000). A level of 200µg g^{-1} sediment Cu (1M HCl extractable) is suggested as threshold above which ‘pollution-induced community tolerance’ (PICT) is induced. This method of establishing tolerance may therefore be just as effective at picking out metal-affected sites than conventional ecologically-based monitoring methods, if not more so. The fact that effects of metals, manifested as enhanced tolerance, are seen in nematodes from several creeks in the Fal system (Restronguet, Mylor, Pill, Cowlands and St Just) implies these organisms are more sensitive than *Hediste* which only signifies an adverse effect in Restronguet Creek.

Nematode community composition in the Helford Estuary appears to be comparable to that of the (relatively uncontaminated) Percuil River (Millward and Grant, 2000).

Phytoplankton. There is some evidence that water quality may influence phytoplankton species composition in the Fal system. In the Carnon River, prior to recent water treatment measures at Wheal Jane, the metal- and low pH- tolerant flagellate *Euglena mutabilis* was the only representative of the phytoplankton community whereas in the less contaminated Fal and Tresillian Rivers the dominant algae were *Oocystis* and *Chlamydomonas* (Rijstenbil *et al.,* 1991). In more marine regions of Restronguet Creek, high metal levels were thought responsible for a reduction in the biomass of *Cryptomonas marina* and *Peridinium brevipes* at the expense of *Carteria marina* and *Skeletonema costatum* (thought to be tolerant to Cu).

Katodinium rotundatum (Zn tolerant) was also recorded in abundance (Rijstenbil *et al*., 1991) implying possible selection pressures due to the presence of elevated metal levels.

'Red tides', caused by the toxic strain of *Alexandrium tamarense,* occurred for the first time in the Fal in 1995 (see section 5.2.2). This species causes paralytic shellfish poisoning (PSP) and in the 1995 outbreak high levels (up to 1233mu) were found in the flesh of mussels from five of the seven sites sampled (Ruan Pontoon, Malpas and Turnaware Pontoon) (Agency, *pers comm* and MAFF, 1996). The high heavy metal concentrations in the Fal have been implicated as contributory causal agents of these blooms, but this is currently under scientific debate.

Maerl. The St Mawes maerl bed on the Eastern Bank of the Fal between Percuil and St Just creeks is the only extensive living bed in southern Britain of major ecological value (see Rostron 1985 for a historical account and e.g. Blunden *et al*., 1981, 1997; Farnham and Jephson, 1977; Farnham and Bishop, 1985; Davies and Sotheran, 1995 for detailed surveys). Maerl has also been reported from the mouth of the Helford River. Deep deposits of dead plants (described as sub-fossil) are known in other parts of Carrick Roads and in Falmouth Bay and these show that maerl formerly covered a much wider area (Birkett *et al*., 1998). Up to 30,000 tonnes of maerl were harvested commercially in the Fal, annually, from 1975 to 1991. A delicate balance between moderate water movement and moderate turbidity/sedimentation, may explain restricted distribution of maerl. Water quality is presumably an important determinand of distribution though this is largely untested. Nutrients do not appear to represent a threat, directly, though settling out of phytoplankton blooms on the maerl may lead to smothering, high BOD and anoxic conditions. Commercial dredging of live maerl deposits, or indeed any form of dredging is considered destructive since this removes the productive surface layer and dumps sediment on any plants which escape dredging, inhibiting habitat recovery (Hall-Spencer, 1994). Dead maerl has been taken from the Fal, by the Cornish Calcified Seaweed Co, since 1975, and the most serious danger to the St Mawes bed is therefore thought to be the settling out of the dredge plume. The company has attempted to minimise damage by dredging only on the ebb tide. In the Fal, the action of mooring chains of yachts has been observed to crush maerl and other organisms.

Some rare and important algae are known to occur in maerl beds whose slow growth rates (time scales of decades-centuries) make them extremely vulnerable to pollution (Birkett *et al.*, 1998). It has been suggested that many of the species typically associated with Fal Maerl, though exhibiting high variety, may be less diverse than beds elsewhere such as Galway (Birkett *et al.,* 1998). They may also lower in abundance than might be expected (Rostron 1985), possibly due to the presence of larger quantities of mud and silt in the bed. Since many of the associated fauna include molluscs - species sensitive to pollutants such as TBT and metals – toxicity issues may also be involved. Such organisms are also involved in the structural integrity of the bed.

Reports of the reduction in proportion of live maerl in the Fal should add to the concern over the conservation of this unique habitat. Perrins *et al*., (1995) suggest that in 1992, 23% of the St Mawes bed was dead – almost double that of ten years earlier. Sampling of maerl communities is difficult, however, and comparisons between sites, and over time, are likely to be partly observer-dependent and, hence, subjective. Natural events such as storm-induced turbidity and water movement can also induce reduction in maerl beds in addition to anthropogenic disturbance (presumably the occurrence of the relict dead maerl beds off the Fal estuary - some 17 km long, 2 km wide and c. 30 cm deep and representing many centuries of growth - reflect such natural changes). A survey conducted late in 2001 by English Nature will hopefully provide a clearer picture of the current status of the Fal beds (R. Covey, *pers.comm*). There appears to be a need for continued scientific study to determine impacts on this biotope.

Zostera. Though there is little specific information on the Fal, *Zostera* beds are known to be susceptible to various forms of pollution, including metals (see Williams *et al*., (1994) for general review of uptake and toxicity in saltmarsh plants, including *Z. marina*). *Z. marina* readily takes up heavy metals, mainly through the leaves, and could in future be used as bioindicator for metal levels. In the laboratory, Brackup *et al*., (1985) found that several metals (mercury, nickel and lead) along with a number of organic substances (naphthalene, pentachlorophenol, Aldicarb and Kepone) reduced nitrogen fixation in the roots, which may affect *Zostera* viability. However, as yet there is no evidence of effects to *Zostera* beds in the Fal.

Fish. A total of 110 species have been recorded for the Fal and Helford Estuaries (Potts and Swaby, 1993), and Rostron (1985) listed some 90 species of fish the Fal Estuary. Many of these are rare and found mainly in the outer and deeper parts of the estuary and in Falmouth Bay (where metal concentrations are low). It is unlikely, therefore, that any reductions in metal concentrations will have any significant effect on overall fish species diversity. The distribution ranges of some species could increase however, if water quality were to improve such that survival of prey items increased in areas where they are currently excluded. Thus, parts of the Fal (Percuil River and Fal Estuary above Turnaware Point) are designated as a bass nursery ground. There is no direct evidence that moderate levels of metal contamination are detrimental to young bass, although the Fal may have reduced levels of 0-group recruitment compared with other south-west estuaries (G.Pickett, *pers comm* to the Agency). Restronguet Creek is within the designated nursery area although there is no evidence that juvenile bass occur there, perhaps because of a paucity of food items (0-group bass feed on benthic invertebrates). Amelioration of metal concentrations may, in future, see an increase in potential prey and a consequent extension of the nursery areas (Renals, 1994).

Several experimental studies have examined physiological changes in bass in relation to heavy metal exposure. However, these tend to be rather unnatural in that the concentrations of metals used in such experiments are generally very high, metals are injected into fish or only blood cells from the fish are used. Kentouri *et al*., (1993) found that a concentration of 6.8 $\mu g.l^{-1}$ copper is toxic to bass eggs, but it should be remembered that the eggs develop offshore and the fish only enter the estuaries as juveniles. The bass farming industry recognises that 0-group fish are vulnerable to contaminants, requiring a clean medium for maximum survival, though there is no scientific literature, on which to base thresholds. Being top predators, bass are potential bioaccumulators of metals. Carpene *et al*., (1990) found fluctuations in liver concentrations of copper and zinc in bass, and attributed these to changes in food intake. Metayer *et al*., (1980), however, collected bass and other fish, along with their preferred prey, from the Loire Estuary and found no evidence of copper, zinc, lead or cadmium bioaccumulation in bass, or any of the other fish that they examined.

The Fal does not now support a salmon fishery (EA, 1997b), though anecdotal evidence suggests it may once have done so (Rostron, 1985, Holliday and Bell, 1979). Sea trout are present in several tributary rivers and can apparently tolerate the moderately polluted conditions. However, there is no run of sea trout through to the Carnon River, although sporadic records are mentioned for the Kennall (1997). The impoverished status of these species of diadromous game fish has been linked to pollution, and it is metals that are of primary concern, although the effects of high nutrient levels, such as reduced oxygen levels and high turbidity may also be contributory factors.

Birds. The Fal estuary, considered as a whole, supports no internationally important populations of birds (Cranswick *et al*, 1997) but does support a nationally important population of black-tailed godwit (a biodiversity long-list species) particularly in the Truro river. Dunlin are also found in the Truro River and at Penryn. Curlew and heron are widespread, but tend to be concentrated along the Truro and Tresillian rivers. Redshank are also widespread, with a tendency to be concentrated near the tops of inlets, including the Tresillian River and Restronguet Creek. Of wildfowl species, shelduck are found in the largest numbers in the Truro River, and to a lesser extent the Tresillian River. Widgeon are concentrated on the Ruan. Teal and mallard are widespread, although numbers are low.

Oystercatcher are quite widespread in the northern parts of the estuary, with the main concentration again on the Truro River. A few Turnstone occur on the Truro River and along Restronguet Creek, although it is likely that more of these birds occur along the rocky shoreline of Carrick Roads. Golden plover and lapwing occur along the Ruan, and lapwing along the Truro River. Other wintering species include greenshank, little egret, spotted redshank and common sandpiper. The mild climate of the estuary is considered to be the major factor attracting these species to the area. Very small numbers of ringed plover, grey plover, sanderling and bar-tailed godwit also occur.

All five of the more common gull species are found in the Fal Estuary complex, the black-headed gull being the most numerous. Great black-backed gulls are concentrated in the Ruan, where the third highest count for the species in the country was recorded in 1993-94.

The sheltered waters of Carrick Roads are important in the Cornish context, supporting the county's largest concentrations of goldeneye and red-breasted merganser, as well as good numbers of divers and rarer grebes (Conway, 1996).

The 1992 discharge incident had no appreciable effect on numbers of wildfowl and waders. The increasing mortalities and occurrences of sick mute swans in the Fal (particularly Falmouth Harbour) between the autumn of 1992 and 1995 have been attributed largely to heavy metal toxicity, but the evidence remains equivocal (McCartney, 1995). Other possible causal agents are paralytic shellfish poisoning (PSP) and tri-butyl tin (TBT) (see Simpson, 1995).

In view of the fact that redshank and other waders may overwinter in Restronguet and some of the other contaminated sites, a small pilot study to investigate the uptake of contaminants from the diet during this period might provide valuable insights into risks.

Impacts on important (rare) species. By their very nature, rare species such as Couch's goby and certain seaweeds associated with the maerl beds are difficult to survey. The fact that they are present at all means that they can tolerate the present metal concentrations in the system. However the fact that rare or scarce species (see table 8) are largely confined to subtidal areas of the main estuary and the St. Mawes maerl bed, and not in the more heavily metal-polluted creeks suggests that subtle changes to discharges are unlikely to have a substantial positive or negative effect on them. This may not apply to the more seaward discharges from STW or the dockyards (e.g Falmouth).

Table 8: Rare or Scarce Benthic Species Recorded From the Fal Estuary Complex (adapted from Warwick *et al*., 1999)

Species	Type of Organism	Habitat	Reference
Laomedia angulata	Hydroid	on eelgrass	Hayward & Ryland (1990)
Anthopleura thallia	Glaucous pimplet	St Mawes: pools, gravel	Manuel (1988)
Aiptasia mutabilis	Trumpet anemone	rock, kelp holdfasts	Hayward & Ryland (1990)
Balanophyllia regia	Star coral	gulleys and overhangs	Sanderson (1996)
Alkmaria romijni	Tentacled Lagoon Worm	Lagoons	Sanderson (1996)
*Pereionotus testudo**	Amphipod	coralline algae	Hayward & Ryland (1990)
*Microdeutopus stationis**	Amphipod	kelp holdfasts	Lincoln (1979)
Achaeus cranchii	Cranch's spider crab	subtidal rock	Ingle (1983)
*Jujubinus striatus**	Sea snail	on weeds	Graham (1988)
*Steliger bellulus**	Sea slug	shallows, among eelgrass	Sanderson (1996)
*Acanthocardia aculeata**	Spiny cockle	sediments	Hayward & Ryland (1990)
Atrina fragilis	Fan mussel	Falmouth Harbour	Sanderson (1996)
*Callista chione**	Bivalve	St. Mawes, sand	Hayward & Ryland (1990)
*Gelidiella calcicola**	Red seaweed	confined to maerl	Maggs & Guiry (1987)
Lithothamnion corallioides	Maerl	maerl beds	Irvine & Chamberlain (1994)
Schmitzia hiscockiana	Red seaweed	current-exposed cobbles	Maggs & Guiry (1985)
Parerythropodium coralloides	Soft coral	gulleys and overhangs	Sanderson (1996)
Carpomitra costata	Brown seaweed	rock	Sanderson (1996)
Cruoria cruoriaeformis	Red seaweed	confined to maerl	Maggs & Guiry (1989)

Those species marked * are nationally rare.

Minewater treatment and changing consents

Various minewater treatment scenarios were evaluated as part of the Wheal Jane Minewater project and as a prelude to the introduction of a more stringent discharge consent. For each of these scenarios, predictions of the consequences in terms of changes in biological communities, populations of key species, and bioaccumulation were made as far as was possible on the basis of the available evidence (Warwick *et al*., 1998).

The final choice was the installation of a treatment plant at the Wheal Jane site in October 2000 in line with option two. Further work is recommended to assess the efficacy of this treatment. Our own unpublished data indicate there may be some significant changes to estuarine water quality. Not all of these are beneficial, in that there may be release of certain metals from sediments. In this respect it would seem essential to review the impact of the consent and to re-evaluate priorities (i.e. the absence of discoloration vs the threat from release of toxic metals from sediments). It is recommended that there should be an attempt to take stock of the current behaviour of metals in Restronguet Creek during estuarine mixing: particularly the potential for desorption and dispersion into the cSAC. This should be done through a combination of estuarine profiling; mesocosm studies on cores (fluxes); and modelling (e.g. ECoS).

The fact that Restronguet Creek is a major influence on the character of the Fal complex is worth stressing. It is recognised by the research community as a unique site, scientifically, and there is much to be learned by continuing detailed long-term multi-disciplinary observations. This needs to be acknowledged and supported by funding bodies and regulatory authorities.

Other Toxic Contaminants

As indicated above, the most striking impacts of historic metal-mining activity are probably restricted to a comparatively small area in the vicinity of Restronguet Creek although communities are generally impoverished over most of the Carrick Roads. Metal contamination may be one explanation, though the presence of china clay wastes, and TBT contamination may be at least in part responsible for this impoverishment. These contributory causes are included in the discussions below

5.1.2 TBT

There is a significant problem in the area with tributyl tin (TBT). The discovery that populations of dogwhelks *Nucella lapillus* were declining was first made in TBT-polluted areas of the south-west by MBA staff (Bryan *et al.*, 1986) and contributed to the establishment in 1987 of restrictions on usage of TBT paints and an EQS of 2ng l^{-1} TBT. TBT levels of about 1ng l^{-1} induce imposex (the imposition of male characteristics on females) and above 10ng l^{-1} this results in sterilisation of females and eventual population decline. TBT poses risks to many aquatic organisms, especially molluscs in estuaries and coastal locations subjected to heavy boating activity, though with the possible exception of *Crassostrea gigas* (where shell thickening occurs at between 2-20 ng l^{-1}), no group is as sensitive to TBT as *Nucella* and related gastropods (see Langston, 1996; Langston *et al.*, 1997). Certainly, the native oyster *Ostrea edulis* does not appear to be exceptionally sensitive to TBT and *as far as is known* Fal stocks have not been affected by the biocide. Low harvest years were reported in the mid 1980s (coinciding with the peak in TBT usage in antifouling paints, first introduced in the 1970s), however, reductions in oyster stocks were attributed to infestations of *Bonemia ostrea*.

The 1987 TBT legislation banned the use of TBT paints on boats <25 m length, which encompasses the majority of the leisure market. Since then inputs from small vessels appear to have been curtailed successfully in most areas. Organotin (TBT) antifouling, however, is still permitted, throughout most of the world, on vessels more than 25m in length. Their non-inclusion in current legislation stems from the argument that TBT derived from commercial shipping is not a problem because of the enormous dilution factor in open seas, coupled with relatively rapid TBT degradation in water (half-life of 1-2 weeks usually). Nevertheless, large vessels do have to come into port and it is there where they may give rise to locally harmful inputs of TBT. They also need to dry-dock for maintenance purposes, and despite the issue of guidelines on containment practices during refits, the impact of TBT arising from such activities is still of concern. Samples of water collected in the vicinity of docks and maintenance slipways can exceed EQS values by a factor of between 1-2 orders of magnitude, and often signify little significant reduction in TBT in the water column, several years after legislation (Langston, 1996). Recognition of the ongoing threat has led the International Maritime Organisation's Marine Environmental Protection Committee (MEPC), to propose that prohibitions on organotins be extended to use on all vessels, irrespective of size. It is anticipated that from 2003 new applications of TBT based paints will have ceased and that some 5 years later all vessels should be free of TBT anti-fouling.

The evidence for TBT impact on dogwhelk populations in the Fal stems from long-term observations of populations by MBA and others (for example, Crothers, 1975). In 1972-73 dogwhelks were common throughout the estuary (for example, Castle Drive, Falmouth, St Mawes, the mouth of Mylor Creek, Weir Point, Restronguet Point), as far upstream as King Harry Ferry. By 1984, populations had disappeared from Weir Point, Mylor and Falmouth and only very small numbers were present at the other sites.

Being closest to the source of mining wastes, the population at the mouth of Restronguet Creek might be considered susceptible to metal pollution. Nevertheless, their presence at Restronguet was maintained constantly until 1979, when TBT paints began to increase in popularity. Thus, although metals may constitute a threat at this particular site, the disappearance of dogwhelks at more distant locations in the Fal is unlikely to be attributable to metals and points strongly to the impact of TBT (Bryan *et al.*, 1987; Langston *et al.*, 1994a). Thus, in surveys carried out by MBA staff in 1992, 1995 and 1996, for example, the Castle Drive population, outside the estuary mouth, represented the northern limit of distribution of the species in the vicinity of the Fal Estuary (compared to the 1970s when *Nucella* was common within the estuary up to King Harry ferry, 8 km upstream). Highest levels of TBT in dogwhelks from Castle Drive were reached in 1987, when the ban on the use of TBT paints on small boats was introduced.

Subsequently, tissue TBT concentrations have declined, slowly. Indices of imposex remain high in this surviving population (60% and 80% of females sterile in 1995 and 1996 samples, respectively) and since there was no evidence of breeding, this population may be doomed. Although tissue concentrations appeared to be declining at the time (five years ago) they were still sufficiently high then to sterilize the majority of females. Since this TBT-affected population lies outside the mouth of the estuary, it seems unlikely that recolonisation of the Fal, to early 1970s levels, will occur in the foreseeable future. A further survey is recommended to determine the current situation.

Values for TBT in water have tended to fluctuate erratically and the source is predominantly the nearby docks and shipyard at Falmouth (discharge of contaminated waste and resuspension of contaminated sediments) with perhaps smaller contributions from sewage outfalls. In 1991, seawater TBT concentrations about 100m from the docks were around 25 ng TBT/l and concentrations in the outer harbour peaked at about 500 ng Sn/l (MAFF, 1992). Recent data from the Agency's own (unpublished) surveys and opportunistic data from other sources (see review by Harris, 2001) confirm the continuing presence of TBT in the Falmouth area and clearly existing legislation is struggling to achieve the EQS in some areas. Agency data indicate TBT (and TPT) levels in Falmouth Dock are among the highest in the UK. These were scheduled to be reduced considerably in September 1998 with the advent of an Integrated Pollution Control (IPC) Authorisation for the site and the associated construction of an active-carbon treatment plant at the docks.

A synthesis of the most recent EA data (collected since between 1997 and December 2001 – see accompanying dbase) is included in a report to the Agency (Harris, 2001). Results for water indicate the highest concentrations occur at Falmouth Dockyard and in the Penryn River where levels are generally within the range 10 and 100ng l^{-1}. There is a cyclical pattern to this data with peak concentrations at the end of August and minima at the beginning of March, but no significant temporal trend overall (Harris, 2001). For the remainder of the Fal Estuary, excluding sewage outfalls, (13 sites from Malpas, upstream, to Pennance Point at the mouth) concentrations range from an upper limit of approximately 40ng l^{-1} to below to the limit of detection (2-9 ng l^{-1}) in the majority (64%) of samples. Because of the log-normal distribution in values the median concentration is close to lower limit of detection (and EQS) of 2ng l^{-1} (Harris, 2001).

There are few published studies on the toxicity of TBT in sediments though most of these indicate effects on benthic organisms occurring in the range 0.1-0.3µg g^{-1} dw sediment (Langston *et al.,* 1990; Langston and Burt, 1991; Austen and McEvoy, 1997). Reductions by an order of magnitude from this range of concentration have coincided with increases in diversity in sublittoral communities in the Crouch Estuary and recolonisation by infaunal molluscs at previously TBT-impacted sites along the south coast (Waldock *et al.,* 1999; Langston *et al.,* 1994a; 1997 and unpublished data). It should be stressed that this evidence is largely correlative, nevertheless concentrations above 0.1µg g^{-1} TBT should be considered potentially harmful. In the EA sediment survey in 1997 concentrations in the Penryn River increased from approximately 1µg g^{-1} at Penryn to a maximum of 10µg g^{-1} close to Falmouth Dockyard. Sediments from Mylor yacht club and St Mawes Harbour also display a legacy of contamination including perhaps that from earlier usage of TBT on leisure craft. The majority of other sites from the Fal (Carrick Roads and Truro River) are less contaminated though still ecotoxocologically significant (average 0.5µg g^{-1} from EA survey).

Indications from assorted limited data sets held by EA, MAFF and MBA are that TBT levels in Fal sediments have remained rather stable since measurements were initiated in the 1980s, and, close to the Dockyard, may penetrate to depths of at least one metre (Harris, 2001).

To account for the levels of TBT in the water column of the Fal Estuary it has been estimated that inputs of the order of 10Kg TBT /yr are needed (Harris, 2001) which is remarkably close to the recorded output from the Dockyard in 1997 of 10.4Kg (Waterman Environmental, 2000) and approximately half the licensed output of

20Kg. This implies that the output from the dockyard may be of the appropriate order to sustain existing concentrations in the Fal. Measured concentrations in dockyard outfalls have consistently featured in the 1-10 $\mu g\ l^{-1}$ range and have remained roughly constant over the last ten years. Levels associated with STW outfalls (e.g Middle Point and its successor at Black Rock) have occasionally reached similar levels in the past but generally appear to exhibit a continuous reduction in recent years (Harris, 2001). Other potential sources could include moored commercial ships and illegally painted leisure craft though estimates suggest this would involve 10 and 300 permanently stationed vessels respectively (Harris, 2001).

Another potential source is sediments. TBT is adsorbed fairly readily onto sediment particles though this is a reversible process and sediments have the potential to release the contaminant back into the water. At steady state the TBT partition coefficient (Kd, litres/g, - essentially the ratio between contaminant in sediment and that in water) typically found in UK estuarine sediments, including some samples from Penryn, is in the region of 4 –30 (Langston and Pope, 1995 and unpublished data). However Harris (2001) has indicated that apparent Kd's in some Fal sediments are often considerably higher (~100) which implies that the sediment fraction contains an additional component to that derived from adsorption from water. A contribution from paint particles derived from spent antifouling coatings removed during maintenance operations seems a likely explanation for these enhanced sediment levels. Release of TBT from such particles to overlying water would be anticipated to be slower than that of TBT adsorbed onto sediments (Harris, 2001). Nevertheless, it is suggested that resuspension of contaminated sediment *could* contribute substantially to the concentrations of TBT observed in the water column. Until more data becomes available on TBT partitioning and suspended solids behaviour in different parts of the Fal, however, precise analysis of the relative importance of sediment as an ongoing source to the cSAC is impossible.

Recent Agency dogwhelk surveys of the Fal confirm that the Castle Drive population, and another at Towan Beach to the east of the estuary, are the last remaining *Nucella* in the area (N. Babbedge, *pers.comm.*).

Definitive evidence for major biological impact from TBT in the Fal Estuary is so far restricted to *Nucella lapillus*. At Falmouth the European sting winkle *Ocenebra erinacea* is also affected. However, since it is slightly less sensitive to TBT than *Nucella lapillus* a small percentage of females still appear capable of breeding. The fact that TBT concentrations in water (and sediment) sometimes approach lab-derived thresholds for sensitive taxa such as molluscs (particularly larval and juvenile forms) might suggest a potential contribution towards poor recruitment of, for example, mussels, particularly in the lower sections of the Fal, which are influenced by the docks. However, the evidence is somewhat equivocal: in contrast to the TBT contamination gradient emanating from the Falmouth Dockyard (as depicted, for example, in accumulated body burdens in transplanted mussels - Agency's 1997 monitoring exercise). The absence of native mussels is most pronounced at upstream sites, particularly near Restronguet Creek, implying that the influence here is more metal-related.

Prior to the increase in TBT usage during the 1970s it is likely that metal levels prevented *Nucella* from colonising Restronguet Creek and restricted the numbers on the adjacent shorelines. Since then, the presence of TBT has dominated the distribution of this species, and perhaps others throughout the Fal. Clearly, more research is needed to determine the nature and extent of organotin contamination in the Fal and Helford cSAC, and perhaps interactions with other contaminants. Re-survey of gastropod populations is recommended, as is an evaluation of sediment TBT concentrations in the context of impact on infaunal mollusc populations. An inventory of sediment loadings should be constructed which would involve coring to evaluate vertical mixing of TBT. Experimental studies to evaluate the influence of bioturbation on mixing behaviour would also be useful in constructing budgets and predicting the fate of the TBT in sediment.

Relatively little is known concerning impact of TBT and other antifoulants on other interest features in the Fal. *Zostera marina* is known to accumulate TBT, (Francois *et al*., 1989), but no effects have been observed in the field (Williams *et al*., 1994). There is no data for maerl.

5.1.3 Oil and Petrochemicals

There is a risk of pollution by oil from the various shipping activities that are carried out in the area, including a bonded bulk oil terminal with a capacity of approximately 60,000 metric tonnes which operates in Falmouth. Potential risks include small leaks, spills and discharges as well as the possibility of a major accident. Ship to ship transfer of bulk chemicals can also result in spillages of, for example, caustic soda, carbon tetrachloride or chloroform. However, very little specific information is available of direct relevance to Fal communities.

Potentially there are a number of ways in which oil could impact on the interest features of the cSAC. Intertidal habitats are under greatest threat from the physical effects of oil pollution: the most vulnerable of these are sheltered rocky coasts, intertidal sand and mudflats and saltmarshes (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. Sub-lethal changes could be detected as increased bioaccumulation, induction of components of the MFO enzyme system, and higher order changes in productivity, fecundity and behaviour. Subtidal habitats and their associated biota may be threatened in the higher energy areas where the likelihood of oil/water emulsions forming is greater. Any marine mammals would be endangered through the consumption of contaminated food, exposure to volatile fractions (eyes and lungs) and, for seals, smothering of intertidal haul-out sites. Birds would be affected by consumption of contaminated food and damage to plumage

Sensitivity of *Zostera* beds to chronic exposure to oil (refinery effluent) may not be very high (Hiscock, 1987). The likely impact of acute exposure (oil spillage) will be influenced by the type of oil, the degree of weathering and the nature of the habitat and in general, it is the associated faunal communities that are more sensitive to oil pollution than the *Zostera* plants themselves (Jacobs, 1980, Zieman *et al.,* 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

In reality, other than anecdotal suggestions of transient oil patches in the area, no literature data for oil impact in the Fal and Helford could be found for review. However, because of the dockyard at Falmouth, commercial shipping lanes offshore and ever-expanding marina facilities the cSAC should be considered vulnerable to spillages and discharges.

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, offshore oil production (e.g. drilling, transport, refining and burning of oil, and petrochemicals) and the atmosphere (PAH's). Locally, an important source is the exhaust from outboard engines.

Chemical information on specific hydrocarbons and petrochemical products in the Fal and Helford cSAC, provided by the Agency, is briefly reviewed in the following section under 'organic compounds' which deals with their significance in terms of water quality standards.

5.2 Non-Toxic Contaminants

Sewage is discharged from treatment works at each of the main centres of population. There are also numerous waterside properties that are not connected to the sewerage system and discharge direct to the estuary. Sewage may also be discharged from commercial and pleasure craft. The main risks from sewage involve nutrient and organic enrichment (though it is difficult to separate the ecological impacts of the two) together with bacterial and viral contamination of bivalve shellfish (see below). There is local concern about the potential public health effects of sewage contamination on recreational water users[1]. There are no longer any major trade effluent discharges but the mining industry has in the past led to elevated heavy metal concentrations in marginal sediments. These latter issues, including discussion of the 1991 flooding incident from the Wheal Jane tin mine, are discussed above

5.2.1 Organic enrichment

This can result in reduced oxygen in the water and produce anoxic sediments. It may stimulate growth of benthic invertebrates. Ammonia which is often present in organic discharges e.g. sewage, can be toxic to biota. No published information specifically related to the Fal/Helford cSAC could be found (a brief review of data supplied by the EA is described in the next section)

A relationship exists between water column dissolved oxygen status, BOD and sediment oxygen demand (SOD). SOD is related to the settlement of suspended solids with a high organic content (as is the case for suspended solids discharged in STW

[1] Recently, SWW has announced via its web page plans to improve water quality by upgrading 28 estuarial sewage treatment works or fine screened estuary discharges, including the addition of UV disinfection for year round operation. In the Fal Estuary this includes Ladock, Truro, Malpas, Mylor, Falmouth and Flushing, and St Mawes. In the Helford Estuary the site at Constantine is included in this upgrade

effluent). The effluent discharges in the Truro area may therefore be linked to the poor oxygen status of the water in the upper estuary. The estuarine discharges may also contribute to the low dissolved oxygen status of the lower estuary (River Fal - mid-channel and mouth), but this is unknown at present - modelling would be required to determine this.

Suspended solids in the estuarine discharge plume could contribute to turbidity of the waters overlying the Maerl and *Zostera* habitats reducing the area colonised. Cumulative effects of organic, metals and TBT enrichment of particulates could have led to progressive changes in the associated benthic invertebrate community, however, the evidence for this is not sufficient.

A link exists between accumulation of organic nitrogen in intertidal sediments and *Enteromorpha* colonisation/higher standing crops. A substantial proportion of the organic N present in sediment close to the estuary discharge point may be derived from the outfall - tracing studies would be required to assess this. Thus, although there is a known mechanism linking *Enteromorpha* growth to the discharge, further work is required to determine whether the discharges (specifically the estuarine discharge) are the source of the problem.

5.2.2 Nutrients

Water quality with regard to nutrients is primarily assessed in terms of the trophic status, or degree of nutrient enrichment of estuaries and near shore waters. 'Nutrient enrichment' generally refers to nitrogen and phosphorus species that 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 man's 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.

Table 9 shows estimated nutrient budgets for three Welsh estuaries (based on Parr *et al.,* 1999) and although different estuaries will vary according to the geology and urbanisation of the catchment area, these figures can be considered typical for many estuaries.

Table 9. Estimated source of estuarine nutrients (based on Parr *et al.,* 1999)

Source	Nitrogen %	Phosphorus %
Agricultural sources (livestock waste, inorganic fertiliser run-off)	25 - 49	3 - 49
Sewage Treatment Works	3 - 13	26 - 62
Atmospheric deposition	2 - 6	1.5 - 1.8
Nitrogen fixation	<5	-
Background	13 - 15	10 - 19

There is a small amount of published information on nutrients and related water quality data in the Fal/Helford cSAC. Graphs depicting SWW data on nitrate levels, BOD and suspended solids at 7 sites around the Helford Estuary (Gweek Mill, Gweek Stream, Polwheveral bridge, Porth Navas Bridge, Reskilling Bridge, Manaccan Stream and Rosevear River) for the years 1981-1986 are depicted in Covey and Hocking (1987). Briefly, nitrate levels fluctuate but are generally in the range 3-7 mg l^{-1} at most sites and slightly higher (5-9 mg l^{-1}) at Gweek Mill and Porth Navas Bridge. BOD is generally in range 1-3 mg l^{-1} with occasional peaks (5-10 mg l^{-1}) occurring at Porth Navas Bridge, Manaccan Stream, and Rosevear River. Suspended solids are generally >20 mg l^{-1} for all sites, peak values (50-70 mg l^{-1}) occurring at Porth Navas bridge, Polwheveral Bridge and Reskilling in Dec 1983, and Rosevear River in Dec 1982. A report prompted by the 'red tide' event (Reid and Pratt, 1995) records nitrate and phosphate levels of up to 8, and 1.8 mg l^{-1} respectively, and suspended solids >2000 mg l^{-1} in the upper Fal Estuary (Truro area) although these measurements were taken during the peak of the algal bloom and are probably not representative of the 'normal' range.

Suggested sources of nutrients in the Helford catchment are agricultural run off and soil leaching. At the time of the 1987 survey Constantine sewage treatment works discharged into Polwheveral creek at a rate of 7 gallons min^{-1} but the receiving stream was considered 'clean' (Covey and Hocking, 1987). There were also sewage treatment works at Mawgan and Gweek with a privately treated sewage outfall at Helford Passage. It was noted that septic drainage at Helford, Gweek and Mawnan regularly overflowed allowing raw sewage to enter the river. Additionally, treated sewage from Culdrose was discharged into a tributary stream (not pinpointed). Run off from Trezise rubbish tip was thought to be causing minimum pollution of streams feeding into Mawgan Creek, though no data is available.

A more recent report (HVMCA, 2000) suggests that water quality in the Helford is relatively good, however it is noted that sewage problems can arise in the summer when visitor numbers peak and fluvial input and water mixing are at a minimum (no data). Additionally, some raw sewage inputs occur from visiting yachts (there are approx 500 deepwater and intertidal moorings along the river). South West Water's STW at Constantine has been identified for improvement in the near future for the protection of shellfish waters (UV disinfection and additional stormwater storage by 2002). The same report suggests however that a change in farming practice over recent years, from dairy to arable, has changed concerns. Run-off may now contain pesticides, fungicides and herbicides in addition to phosphates and nitrates although no data is given.

Using models to estimate nutrient inputs, Fraser *et al.,* (2000) compared the relative contributions of diffuse and point sources inputs to the Fal and Helford in 1931 and 1991 (table 10). The figures are the result of an integrated approach taking into account a wide range of physical characteristics and parameters such as the local geology and sediment type, land use, volume, dilution and flushing rate, rainfall, vertical mixing, and wave exposure, all of which influence the nutrient status of environmental waters. These variables are almost unique to individual catchments and confound attempts to make accurate predictions without taking them into consideration.

Table 10. Proportion of nitrogen and phosphorus exported from diffuse and point sources in the Fal and Helford catchment (Fraser *et al.*, 2000)

	Source	**Nitrogen %**		**Phosphorus %**	
Year		1931	1991	1931	1991
Fal	Diffuse	92.8	95.1	75.3	80.6
	Point	7.2	4.9	24.7	19.4
Helford	Diffuse	93.5	93.8	78.1	76.8
	Point	6.5	6.2	21.9	23.1

This table indicates that the relative proportion of nutrient inputs from diffuse sources has increased slightly in the Fal over the 60-year period covered, whilst inputs to the Helford have not changed significantly.

The principal effect of extreme nutrient enrichment is eutrophication, defined as 'the enrichment of natural waters by inorganic plant nutrients, which results in the stimulation of an array of symptomatic changes' (EA, 1998). These changes include an increase in phytoplankton growth that is reflected by an increase in chlorophyll α concentrations. Dissolved oxygen concentrations in the water column fluctuate during the growth phase of a 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.

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.

There is very little specific information on sensitivity of estuarine macrofauna, or on the rare species and special interest features within the cSAC, to nutrient enrichment. For example there have been no specific studies of nutrient impact on *Zostera* communities. Elsewhere, local increases in nutrient levels (e.g. from sewage, agricultural runoff or aquaculture) are reported by some to have favourable consequences for eelgrass beds, usually where *Zostera* growth is limited by available nitrate (Fonseca *et al*., 1987). Eutrophication is more often a cause of the decline, or the lack of recovery of, *Zostera* beds (Borum, 1985; Wetzel and Neckles, 1986; den Hartog and Polderman, 1975; Kikuchi, 1974). A variety of different harmful effects have been identified: Metabolic imbalance caused by high nitrate concentrations (Burkholder *et al*., 1992); Increased growth of epiphytic and blanketing algae (e.g. Deegan *et al.,* 1997; Borum, 1985; Burkholder *et al*., 1992; Wetzel and Neckles, 1986; den Hartog, 1987); phytoplankton blooms which can increase turbidity and reduce biomass production (Dennison, 1987); and increased vulnerability to wasting disease (Buchsbaum *et al.,* 1990). Indirectly therefore the secondary productivity of benthos will almost certainly linked to nutrient status through effects on sediment and epibenthic flora, including phytoplankton.

Since the mid 1990s there has been an increase in the reported incidence of algal blooms in the Fal. The first of these, in 1995, was particularly significant because it involved the 'red tide' dinoflagellate *Alexandrium tamarense* – *A. tamarense* is known to produce toxins that can be concentrated in shellfish to form PSP (paralytic shellfish poisoning). The bloom appears to have started in upper reaches of the Fal close to Newham STW, coinciding with conditions of low freshwater flow, high temperatures and nutrient levels in receiving waters (perhaps associated with the peak tourism season), and spread progressively down the Fal (Reid and Pratt, 1995).

The Upper Fal Estuary was proposed as a potentially Sensitive Area (Eutrophic) under the provisions of the Urban Waste Water Treatment Directive (91/271/EEC) in 1997. Investigations (1994-1996) highlighted the eutrophic state of the upper Fal (EA, 1997a). Findings include:

- Chlorophyll *a* levels were generally low during the winter season, with only the uppermost sites of the Truro and Tresillian Rivers occasionally exceeding the $10 \mu g\ l^{-1}$ concentraion[1].
- In summer, the $10 \mu g\ l^{-1}$ chlorophyll *a* standard was often exceeded at most sites.
- The average concentration of chlorophyll *a* increased upstream from Carrick Roads to the tidal limits at Truro.
- There was an exaggerated variation in DO concentrations at Malpas in late June 1995 which continued through July, this was considered to be primarily due to algal bloom activity and die off, combined with tidal and diurnal factors.
- Red coloured water was reported in late June, and July 1995, in the middle reaches of the Fal estuary. Analysis showed that *A. tamarense* was the dominant alga with a cell count of up to 3130[2].
- PSP was detected and the Port Health Authority issued a prohibition notice on the collection of shellfish during 1995 and 1996
- Analysis of sediment from the Fal in the late autumn of 1995 confirmed the presence of *A. tamarense* cysts (dormant wintering phase), and it was predicted that the algae would reappear within 1996.
- In the second week of June 1996, a bloom of *A. tamarense* occurred, centred on Ruan. The timing of the bloom was earlier than the previous year, and less intense (up to 800 cells ml^{-1} maximum) but still considered significant.
- The report (EA, 1997a) reiterated the results of modelling exercises: Newham STW has a localised impact on TIN concentrations within the upper Fal Estuary throughout the year. In summer, the impact of the TIN concentrations in Newham discharge extends to all sites on the Truro River, and parts of the main channel.
- It was estimated that in summer 1995, Newham STW contributed ~50% of the TIN load to the estuary, and sufficient to generate the observed algal blooms.

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[1] In the UK, the indicator (mean) value for suspected eutrophic conditions is set at $10 \mu g\ l^{-1}$ chlorophyll *a* (Dong *et al.,* 2000; also the DoE criterion, (EA, 2001).

[2] Values greater than $10 \mu g\ l^{-1}$, coupled with cell densities of 5×10^5 cells l^{-1} (500 cells ml^{-1}) are considered indicative of phytoplankton blooms (EA, 1998).

On the basis of this evidence, the Truro, Tresillian and Fal Estuaries were designated a Sensitive Area (Eutrophic)[1]. However harmful algal blooms continue to occur in the Fal and Helford system. The most recent incidence, in 2002 resulted in significant mortality of worm species *Nereis* and *Arenicola* as well as some shellfish, and was centred on Polwheveral Creek and Porth Navas, on the Helford Estuary, where high nutrient levels remain a cause for concern. The bloom was believed to be principally *Gyrodenium aureolum*, which produces toxins that can also kill fish. Also affected was Calenick Creek near Newham STW in the upper Fal. .

It is clear therefore that the Newham STW operations constitute a significant source nutrients, and were implicated in the events of 1995-6. In the more enclosed regions, STW discharges are increasingly likely to have an impact on the water quality of the cSAC. Impressions are that a more rigorous surveillance programme for nutrients is required to determine the conditions which initiate bloom formation. The results would assist environmental managers in implementing appropriate controls to ensure the SACs are brought into favourable condition within the required timescales.

In an attempt to demonstrate the significance and scale of nutrient enrichment over this period data obtained from the EA are synthesised in a later section (6.3.1).

5.2.3 Turbidity and Siltation

The effect of increased turbidity will be dependent on the background levels at the site. Some estuaries, particularly low salinity regions, are naturally turbid and the composition of resident benthic communities reflect their adaptation to ambient conditions. The level of suspended solids can be enhanced by anthropogenic activities in the river catchment as well as within the river and the estuary. Changes in river flow as a result of abstraction can influence suspended solids concentrations reaching estuaries.

The effects of excess turbidity can be direct and indirect. Siltation can inhibit filter-feeding and clog the gills of fish and under extreme conditions may destroy benthic communities or render them inaccessible to predators such as birds. The decrease in light penetration may also effect primary productivity (phytoplankton and benthic macroalgae) and hence, indirectly, reduce invertebrate standing stock. However, high turbidity levels do not necessarily preclude high phytoplankton standing crops provided there is rapid and thorough mixing (Parr *et al.,* 1998).

Parr *et al.,* (1998) identified the effects of high turbidity on macrophytes, including *Zostera, Laminaria* spp. and *Fucus vesiculosus*. Reduced growth rates, standing crop, area coverage and depth of colonisation have been reported to be related to turbidity. No published information specifically related to the Fal/Helford cSAC could be found in the literature, therefore a brief review of turbidity data, supplied by the EA are described in the next section.

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[1] If a site is designated as a Sensitive Area (Eutrophic) under the Nitrates Directive, the objective is to "reduce water pollution caused or induced by nitrates from agricultural sources" and "prevent further such pollution". Therefore, improvements to STW are not an automatic result of designation. STW improvments may be instigated if the STW is considered to be an indirect nutrient source (ie, discharges upstream of the designated site) and serves a population greater than 10000 population equivalent. However, there are no such discharges in the designated areas of the Truro, Tresillian and Fal estuaries.

5.2.4 Dissolved Oxygen

The effects of changes in dissolved oxygen concentrations are primarily related to reduced DO levels and include direct lethal and sub-lethal responses in marine organisms; enhanced release of nutrients from sediments; development of hypoxic and anoxic conditions (Stiff *et al.,*1992; Nixon *et al*., 1995). These authors identified crustacea and fish as the most sensitive organisms with the early life stages of fish and migratory salmonids as particularly sensitive. For estuarine fish, a minimum DO requirement of 3 to 5 mg l^{-1} has been suggested (Stiff *et al.,* 1992).

Exaggerated fluctuations in DO concentrations were recorded at Malpas in late June 1995 and continued through July. This coincided with a significant algal bloom (see section 5.2.2) and was considered to be primarily due to photosynthetic activity and subsequent algal die off, combined with tidal and diurnal factors (EA, 1997a). Again little published information specifically related to the Fal/Helford cSAC could be found and DO data supplied by the EA are briefly described in relation to water quality standards in the next section.

5.3 Other Threats And Activities

5.3.1 Dredging and disposal

Regular maintenance dredging is required in a number of areas, although this is relatively infrequent due to the low levels of suspended sediment in the Fal and its tributaries. There is also occasional capital dredging related to specific projects. Mobilisation of the sediments during the dredging operation can have physical and chemical impact on habitats. Dredged material that does not contain significant levels of contaminants is disposed of at the designated sea disposal site in Falmouth Bay. It is possible that, under certain conditions, fine material disposed of this site could impact on the cSAC. Sediments in some of the ports and marinas are also highly contaminated with TBT and have to be disposed of using alternative, more costly, methods

5.3.2 China clay wastes

Though there is no unequivocal evidence that normal operational activities pose a threat, occasional accidental discharges from the china clay industry probably add to the stresses already imposed on biota in the Fal. One such spill occurred in April 2000 after part of a tailings dam collapsed, just south of St Dennis, releasing waste into the River Fal. The sludge visibly impacted a 10-mile stretch of the Fal and posed two main threats to the river and estuary eco-system; a smothering effect on gills of fish and filter-feeding shellfish; reduced light levels affecting photosynthesis in algae and plants., however such effects are likely to be short lived. Less is known about the possible risks from other chemicals contained within the waste, which could include toxic metals such as Zn, Cd and Cu.

5.3.3 Fishing, shellfisheries and mariculture

The only fishing activity in the cSAC likely to cause serious water quality problems is discharge of fish wastes from factory ships carrying out onboard processing. This risks smothering the seabed and creation of anoxic conditions as well as possible localised nutrient enrichment.

Bivalve shellfish are particularly at risk from most of the human activities carried out in the area. Problems of accumulation and toxic effects from chemicals such as TBT and metals have been described above. The main risk from the latter is that it can cause the shellfish to become unfit for human consumption; the same applies to microbial pathogens (see section 5.2.2).

All bivalve mollusc production areas need to be classified according to their level of contamination, in order to determine the level of treatment that the shellfish must receive before being sold for human consumption (shellfish hygiene directive information). The classifications of shellfisheries in the Fal in 2000 are shown in table 11.

Table 11. Shellfisheries in the Fal

PRODUCTION AREA	BED NAME	SPECIES	CLASS
Truro River	Grimes Bar and Maggoty Bank	Native oyster	B
	Tregothnan	Mussels	B
	Calenick Creek Lambe Creek and Malpas	Mussels	C
Tresillian River	All beds	Mussels	B
Fal	Ruan Creek	Mussels	B
	South Wood	Mussels	B
	Flushing and Falmouth Wharves Meads	Native oyster	C
	Mylor Creek	Mussels	B
	All other beds Pandora Beach	Native oyster	B
		Pacific oyster	
Percuil	All beds	Native oyster	B
		Pacific oyster	

Data for microbiological parameters, in relation to EQS and guideline values for shellfish and bathing waters, based on EA records, are discussed in greater detail in section 6.3.5, which includes classifications for both Fal and Helford, for 2001.

5.3.4 Endocrine Disruptors

The complex endocrine system controls the biochemical and physiological functions of multicellular organisms including growth, behaviour, immune function and reproduction. Hormones transported by the blood act as 'chemical messengers' and interact with DNA sequences in the cell nucleus leading to specific functional changes. In higher organisms the brain, via the hypothalamus and pituitary, ultimately governs the system. Substances affecting the natural function of this system are therefore termed 'endocrine disruptors', defined by the European

Commission as ‘an exogenous substance that causes adverse health effects in an intact organism, or its progeny, consequent to changes in endocrine function’ (EC, 1999).

Reviews carried out by IEH (1995, 1999), CSL (2000) and the Environment Agency (EA, 1998a) have produced extensive lists of known and potential endocrine disruptors (EDs) based on existing evidence from laboratory and field studies. Also, the European Commission has drafted a more comprehensive document (EC, 1999) detailing over 500 known and potential endocrine disruptors, grouped into 38 classes. However, for many of the chemicals listed, there is no available evidence for endocrine disrupting effects.

Principal endocrine disruptors are: Pesticides, herbicides, alkylphenols, bisphenol-A, phthalates, PCB’s and other organochlorines, PAH’s, natural and synthetic hormones, phytoestrogens and TBT. Cadmium, lead and mercury have also been implicated as having endocrine disrupting effects.

Many endocrine-disrupting effects can be attributed to natural and synthetic oestrogenic hormones derived during sewage treatment (Desbrow *et al.,* 1998). Treated domestic sewage discharges have been identified as a primary source of endocrine disrupting substances, and found to be a major cause of oestrogenic effects on male freshwater fish in the UK and US (Purdom *et al.,* 1994; Folmar *et al.,* 1996). Other sources of endocrine disruptors, besides domestic sewage, include industrial effluents, leachates from solid waste disposal sites, application of agrochemicals, agricultural and urban run-off, contaminated land, incineration and shipping.

However compared with freshwater environments there is a paucity of reliable information on the nature of endocrine disrupting substances, or their effects in the marine environment. Research has generally centred on fish exposed to oestrogens and their mimics, for which there appears to be emerging evidence of some impact in species such as flounder in industrialised estuaries such as the Tees (EDMAR Project, 6th quarterly report, CEFAS, Burnham). Some preliminary and planned work on viviparous blenny and gobies is also described in this programme. Little information is available on other UK marine vertebrates such as birds and mammals. With the exception of the effects of organotins in molluscs, knowledge of endocrine disruption in invertebrates is even more sparse because their endocrine systems are poorly understood.

Apart from the specific example of TBT impact in dogwhelks (see earlier section) there is very little published evidence of endocrine disruption in Fal and Helford biota. A recent report for English Nature, collating information on endocrine disruptors in designated marine sites, concluded that the cSAC was of medium priority for further research (Allen *et al.,* 2000). However the authors acknowledge that this is a preliminary assessment based on consented discharge volume as a crude indication of point source inputs of endocrine disrupting substances. There is very little specific monitoring information for known or suspected endocrine disruptors in the EA database.

We have established that, with the exception of TBT, the majority of levels of synthetic organic substances (which includes many endocrine disruptors) are now generally very low in the cSAC (see next section). Cadmium levels in the waters of

Restronguet Creek are somewhat elevated and exceed EQS values of 2.5μg l^{-1} at times, although endocrine disrupting effects to fish are not documented at concentrations of less than 1mg l^{-1} Cd (Thomas, 1989, 1990).

Perhaps of greater concern is cadmium in sediments (which exceed interim sediment quality guidelines and probable effect levels of 0.7 and 4.2 μg g^{-1}, respectively). High levels of cadmium have recently been shown to exert endocrine disrupting effects in fiddler crabs (*Uca pugilator*), leading to inhibition of ovarian growth (Rodriguez *et al*., 2000). Mercury levels in sediments of the upper Fal are also in excess of ISQG's and PEL's (0.13 and 0.7μg g^{-1} respectively). Unfortunately, at present, there are no studies that specifically investigate the endocrine disrupting properties of mercury in fish or invertebrates, though mammalian studies (rat *in utero* exposures) have shown that mercury may act at the level of the hypothalmic pituitary unit and, directly, at the level of gonadal steroid biosynthesis.

Thus, there is a notable lack of firm data for endocrine disruptors and their effects in the cSAC and virtually nothing is known about the effects of biological processes on transport and speciation of these compounds. It would appear that implications for wildlife are linked to domestic and industrial discharges, and will depend not only on contaminant levels in such discharges, but the hydrodynamics of the area. Judging by nutrient and microbiological data for the upper Fal, there is a sewage-related problem in the Truro area, therefore the risk to biota from endocrine disrupting substances is also liable to be highest here (TBT is an exception due to the sources at Falmouth). In view of the conservation importance of both fish and invertebrates in the cSAC, more detailed investigations into endocrine disrupting substances and their effects within the cSAC are recommended.

6. WATER/SEDIMENT STATUS AND QUALITY STANDARDS

In this section we examine unpublished data, mainly supplied by EA and its predecessor the NRA, of key determinands which may influence the Fal/Helford cSAC. Summary statistics are drawn up (mainly based on the last 15 years in EA listings) in an attempt to establish further evidence as to whether or not existing water (or sediment) quality is causing impact. Where relevant we refer to site-specific models predicting pollutant concentrations and their links to impact, and highlight limitations of available data.

Millward (1992) produced a review of information on the Fal held in the NRA data banks prior to 1991 (NRA contract TWU 0003). This provides useful dimensional and topographical characteristics, flow rates and a synthesis of water quality information available at the time (1986-1991) including harmonised monitoring of riverine inputs entering Restronguet Creek at Devoran and the Fal at Tregony.

Since then, in addition to the targeted data sets collected for studies described in section ii) The Agency routinely collects data on Fal water quality for the following purposes:

- Dangerous Substances Monitoring (TBT, Dissolved Metals, Suspended Solids, salinity)
- Shellfish Waters Monitoring (D.O., pH, Dissolved Metals, Salinity, Organics, Faecal Coliforms)
- Urban Waste Water Treatment Directive Monitoring (D.O., pH, BOD, Nutrients, Suspended Solids, Salinity).

Quality Control

Nutrient, dissolved metals, TBT and organic pollutant data described below is predominantly provided by the Environment Agency whose quality control measures have recently (May 2002) been upgraded to include accreditation by UKAS to BS EN ISO/IEC 17025. Prior to this formal accreditation (when most of the analyses considered here would have been performed) internal QA/QC procedures and inter-laboratory calibrations were operated. Unpublished MBA metals data for sediments and biota in the Fal and Helford cSAC, are based on similar QA/QC protocols. The methods used have been validated in a number of intercalibration exercise (e.g. Quasimeme) and for the last 10 years has involved use of certified Reference materials as checks on quality of the data. Samples are expected to be accurate to within ± 10%.

There are some factors, related to inconsistent sampling and analysis frequencies, which can make interpretation of the EA data complex e.g. the nutrient determinands measured can differ between sites and times: nitrogen, for example may be measured as dissolved N, total nitrate or total inorganic nitrate (TIN). This example leads to particularly difficult comparisons, as the proportion of bioavailable nitrogen may differ seasonally as well as according to operational definitions. Also changes to the limits of detections makes trend determination difficult (where half-detection limits are substituted for '<' values to evaluate the data.

Screening of data

For the purposes of the current exercise we have not routinely transformed the data set, even though tests for normality indicate they are not always normally distributed. Where significance tests have been applied to specific comparisons however, data has been transformed if needed.

6.1 Toxic Contaminants: water quality

EA water-monitoring results are, in the absence of biological effects data, compared to Environmental Quality Standards (EQS) in order to gain a first-order approximation of possible impact on biota (full lists of standards for List I and List II chemicals are given in Annex 2-4). Thus, in the context of the current project, expressions 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). 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 highlight some of those sites and conditions which merit closer investigation. They do not necessarily assure Favourable Condition.

6.1.1 Metals

Table 12 indicates Environmental Quality Standard values (EQS) relevant to metals in the Carnon River/ Fal Estuary (for a full List I and List II chemicals see Annex 2-4)

Table 12. Environmental Quality Standards for metals relevant to the Fal (μg l^{-1})

WQO	Fe	Zn	Cu	As	Ni	Pb	Cd	pH
Protection of Estuarine Biology§		40 D,e,m	5 D,e,m	25 D,e,m			5 D,e,m	Range 7-9
EC DSD – Fresh Water	1000 Da	8-125 Ta[1] 30-500T*[1]	1-28 Da[1] 5-112 D*[1]	50 Da	50-200Da[1]	4-20Da[1]	1 Ta	6-9*
EC DSD – Salt Water	1000 Dsa	40 Dsa[2]	5 Dsa	25 Dsa	30Da	25Da	2.5 Dea	6-8.5*

§ parameters based on the implementation of EC Shellfish Waters Directive, NRA Water Quality Series, 16, March 1994.
EC DSD = EC Dangerous Substances Directive
T = total concentration, μg / l ; D = dissolved concentration, μg / l
* = 95 % ile a = Annual Average m = annual Maximum
e, in estuary water; s, in salt water.
[1]depending on total hardness. QS increases as total hardness of water increases
[2]in the future the EQS for Zn may be revised downwards to 10 μg / l

Zinc. Dissolved Zinc levels at EA monitoring sites (1999-2001) for upper Fal tidal waters are summarised in figure 12. Median values are below the EQS of 40 μg l^{-1} though occasional extreme values do exceed the standard. If a proposed[1] revision of the standard to 10 μg l^{-1} is accepted several sites would be on the borderline. The STW surface boil samples at Truro Newham are in excess of this indicating a potential source. However considering inputs to the Estuary as a whole the dominant

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

influence is Restronguet Creek – at the mouth concentrations are highly variable depending on the tidal state and degree of stratification. This is reflected in values ranging from 12-425µg l^{-1} (median 114 µg l^{-1}). Highest values probably reflect less dense FW from the Carnon River flowing out from the Creek on the surface, indicating the potential for export of metals to the Fal.

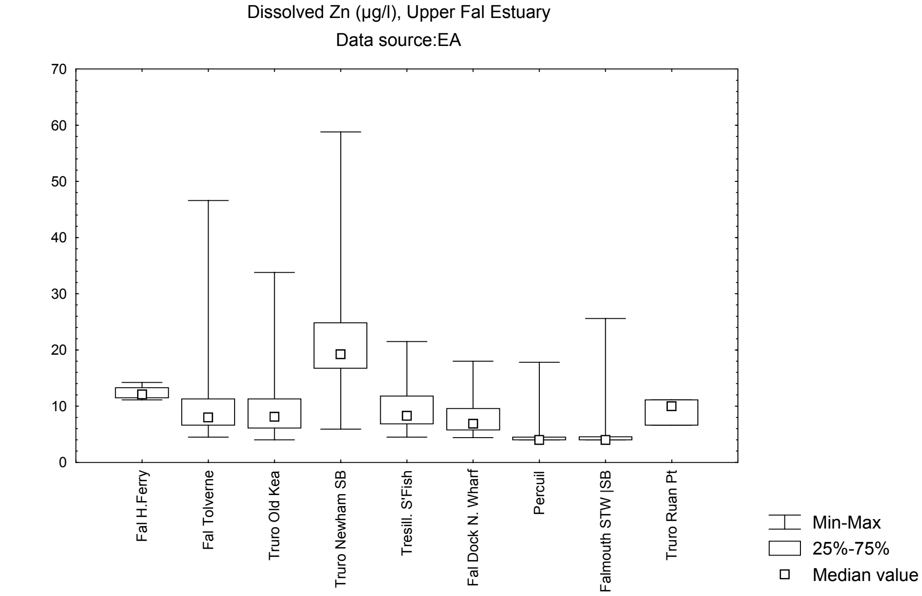

Figure 12. Dissolved Zn (µg l^{-1}) in the upper Fal

The behaviour of metals during estuarine mixing in Restronguet Creek is discussed in the previous section and is dominated by sources in the Carnon Valley, including Wheal Jane. There are additional major inputs from the County Adit and other mine drainage, together with run-off from spoil heaps. The sum of these various sources (often in the mg l^{-1} range) is demonstrated by the monitoring data for the Carnon at Devoran Bridge, just above the tidal limit in Restronguet Creek (figure 13).

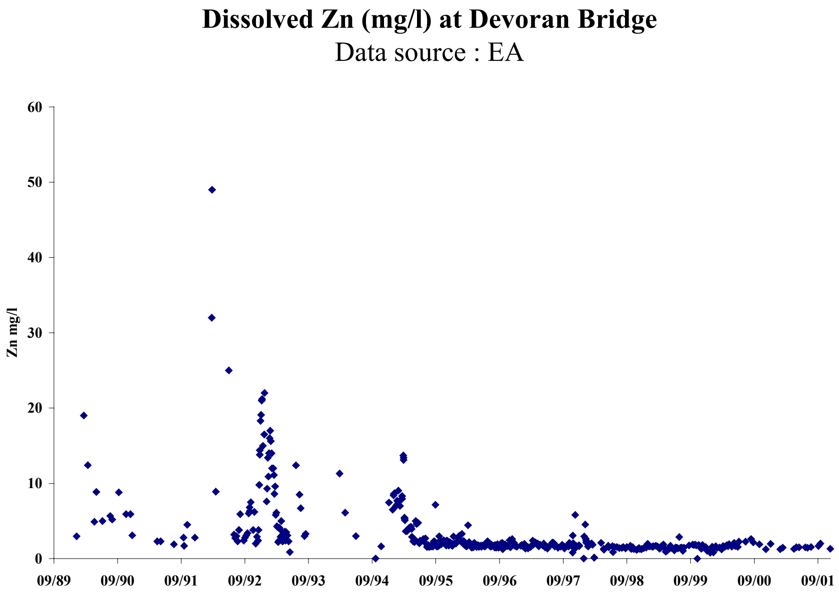

Figure 13. Dissolved Zn (mg/l) entering Restronguet Creek at Devoran Bridge.

High values in early part of 1992 represent the aftermath of the flooding of Wheal Jane. In recent years concentrations have stabilised at between 1-2mg l^{-1}.

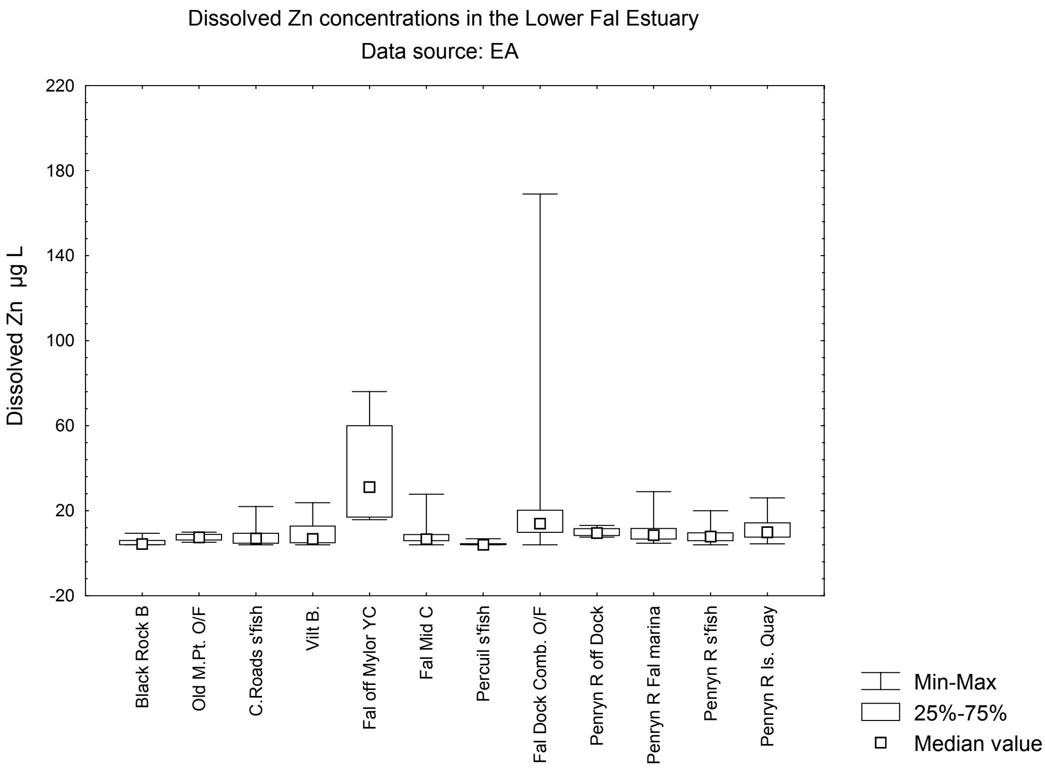

Figure 14. Dissolved Zn (µg l^{-1}) in the Lower Fal Estuary.

In the lower estuary the majority of monitoring sites would comply with the EQS for Zn. Lowest levels were those at Percuil shellfish water site (many observations below the detection limit of 4µg l^{-1}). Values off Mylor Yacht Club appear to be elevated though the number of observations here is low. Occasional high values are recorded in association with Falmouth Dockyard combined outfall though the median values are <20 µg l^{-1}. In view of the use of Zn compounds in antifouling paints these elevated levels and sources merit further monitoring.
Zn concentrations at the Helford Shellfish monitoring site are seldom much above DL and at the mouth of the estuary the median value is 5 µg l^{-1} (max 24.8 µg l^{-1}).

Arsenic. Dissolved arsenic inputs to the cSAC are also dominated by sources in the Carnon Valley, including Wheal Jane, the County Adit and other mine drainage, springs and run-off from spoil heaps. Monitoring data for As in the Carnon at Devoran Bridge resembles that of Zn with levels stabilising at a median value of 37.1 µg l^{-1} (max 115 µg l^{-1}) for data since the beginning of 1995 (below the FW EQS of 50 µg l^{-1}). On entering Restronguet there are concerns that the slightly lower saltwater EQS values (25 µg l^{-1}) may be exceeded following desorption of As from particulates (see earlier section).

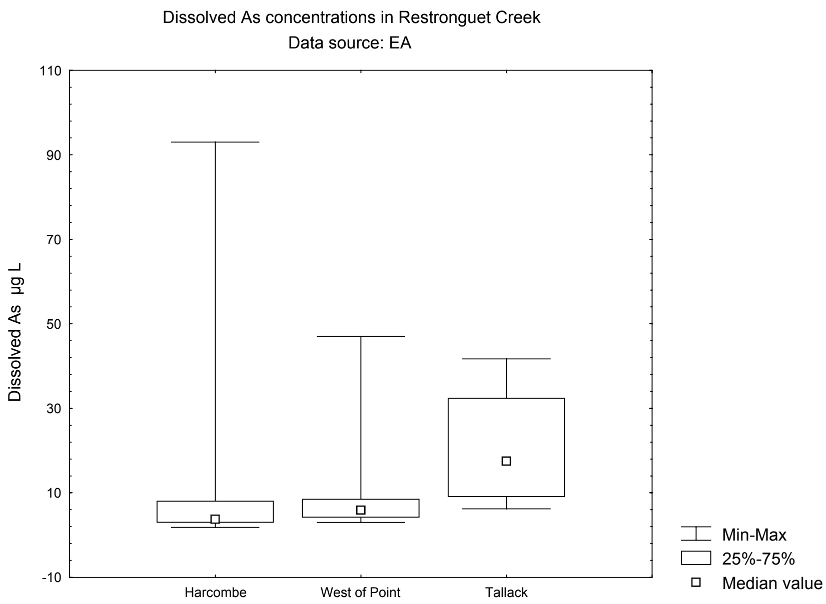

Figure 15. Dissolved As levels (µg l^{-1}) for three sites in Restronguet Creek.

EA data for Restronguet are few in number and median values appear to be below the threshold (figure 15). However maximum values can reach 90 µg l^{-1} or more and are likely to be influenced by salinity and flow rate (Langston *et al.,* unpublished data); figure 16 demonstrates this for samples from the mouth of the creek over tidal cycles on a number of occasions. Arsenic values may vary over an order of magnitude and will be highest at low water when the Carnon freshwater influence is highest. Some export to the Fal is therefore likely and investigations are needed to predict the behaviour and fate under different conditions of tide and river flow. There are also indications of a seasonal pattern for As at many sites as concentrations peak during the drier summer months.

In the upper Fal estuary (above Restronguet) a small degree of As enrichment (up to three-fold) can be seen in the data compared to typical sea water background values of ~2 µg l^{-1}. This enrichment is mostly seen in summer samples probably reflecting the inputs from Restronguet and, perhaps, benthic sediments. STW's do not appear to represent a significant source of As.

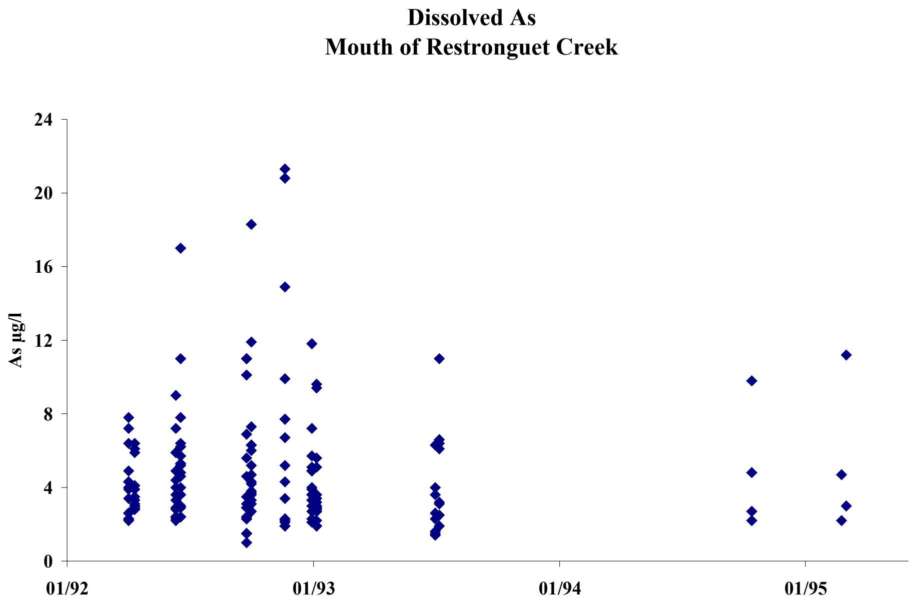

Figure 16. Dissolved As levels (µg l^{-1}) at different stages of the tidal cycle in samples from the mouth of Restronguet Creek.

In the lower Fal estuary the median As concentrations and ranges are close to background for a number of sites (figure 17) though close to Restronguet (e.g East of Weir Point, Greatwood House and to a lesser extent Mylor) maximum values may periodically, be significantly above background suggesting localised enrichment along the eastern shoreline. As demonstrated for the site at the mouth of Restronguet, this is likely to result from the influence of metal rich fresh water flowing on top of denser saline water and will be most obvious during low water.

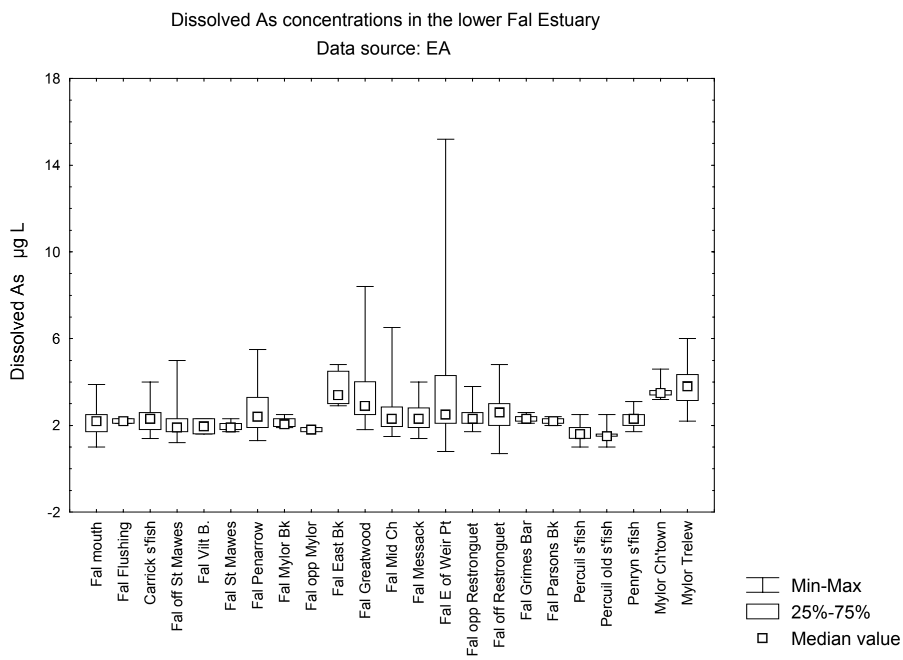

Figure 17. Dissolved (As µg l^{-1}) in the lower Fal Estuary

In the Helford Estuary, As concentrations have been close to reference values since 1993 (median 1.5 µg l^{-1}). A value of 21 µg l^{-1} As was recorded in March 1992 at the mouth of the Helford, shortly after the Wheal Jane incident, though two months later concentrations had returned to normal.

Copper. Sources of Cu in the Carnon valley are similar to those described above and are dominated by mining impacts, including the County adit and Wheal Jane. Figure 18 shows concentrations entering Restronguet Creek at Devoran Bridge over the last decade. Apart from the time of flooding at Wheal Jane, Cu concentrations have remained at a similar overall level during that period. Since the beginning of 1995 the median concentration has been 0.256 mg l^{-1} (i.e. considerably above the EQS for Cu in FW, see table 12).

Dissolved Cu (mg/l) at Devoran Bridge

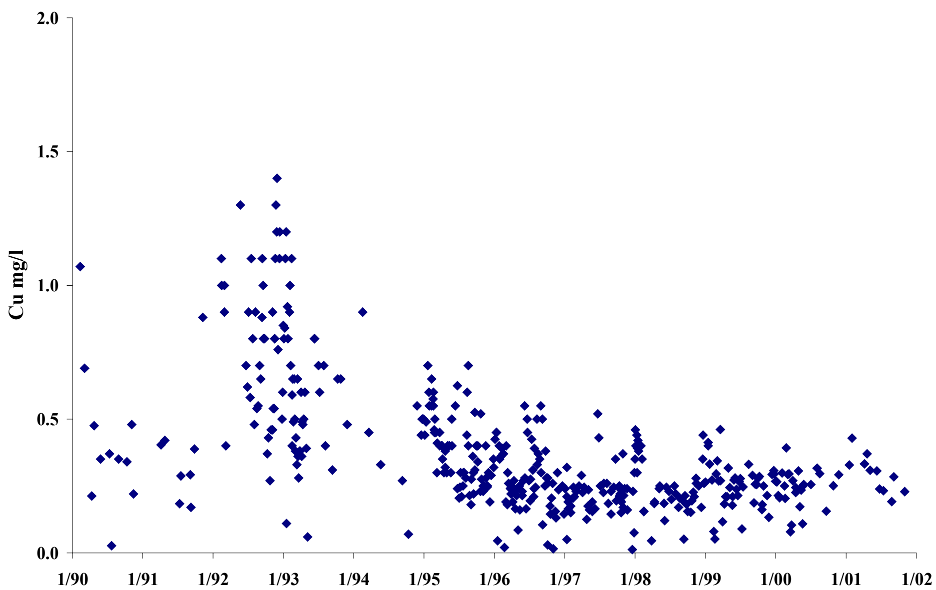

Figure 18. Dissolved Cu (mg l^{-1}) at Devoran Bridge (Data source :EA)

In Restronguet Creek, mixing with seawater (and some removal to sediment) dilutes the threat of Cu considerably and in the upper part of the Fal Estuary proper the majority of dissolved Cu measurements fall below EQS values. Close to the mouth of Restronguet Creek however, concentrations up to 25 µg l^{-1} may be encountered in less saline surface samples and even the median value appears to be higher than estuarine and salt water standards of 5 µg l^{-1}.

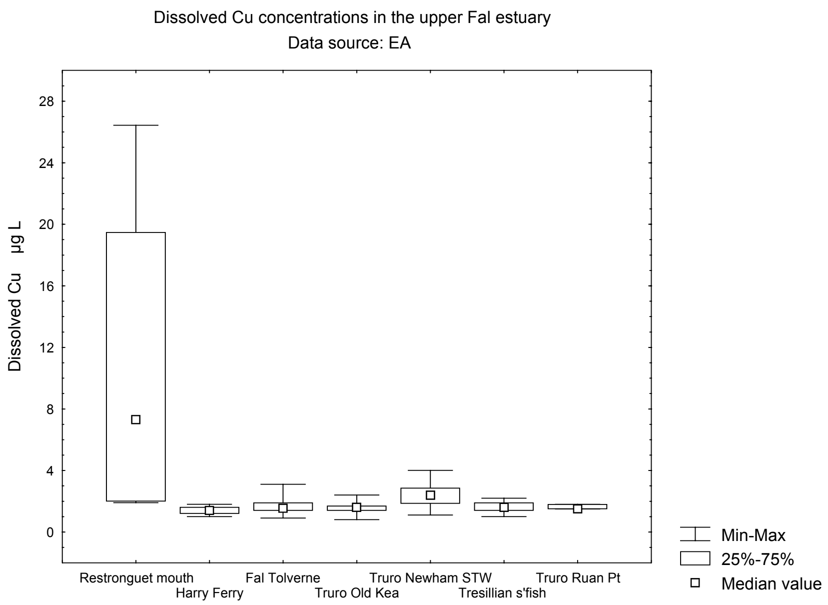

Figure 19. Dissolved Cu (µg l^{-1}) in the upper Fal Estuary.

Median Cu concentrations at most sites in the lower Fal are relatively low. As might be expected, close to the Falmouth Dockyard combined outfall, higher values are encountered. The median here is 7.1 µg l^{-1} (slightly above the EQS), and maximum 105 µg l^{-1}. In the Helford estuary median dissolved Cu concentrations are close to 1 µg l^{-1} (maximum 2.6 µg l^{-1}) and would not be expected to be problematic.

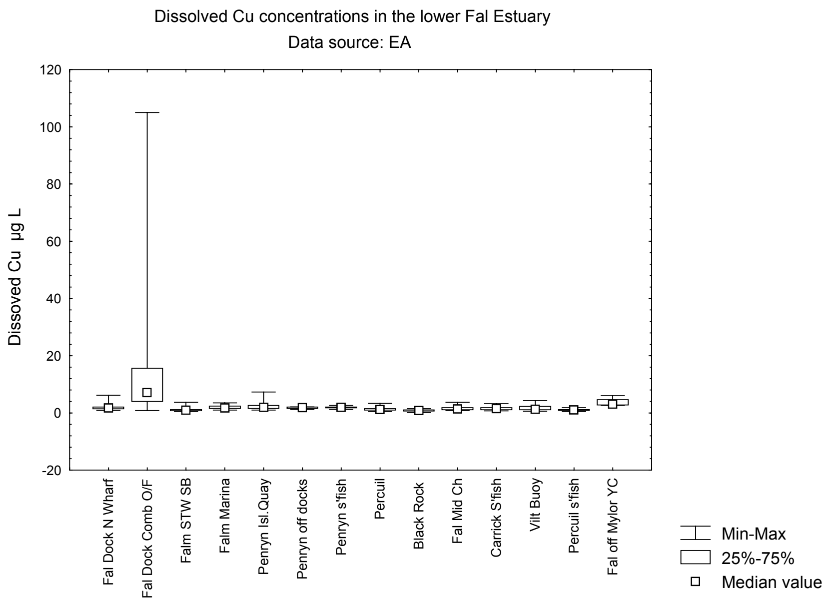

Figure 20. Dissolved Cu (µg l^{-1}) in the Lower Fal Estuary

Cadmium. Cadmium is also derived primarily from various mining sources in the Carnon Valley, notably Wheal Jane. This was particularly evident at the time of flooding in early 1992 (see figure 21). Data for Devoran Bridge for the years since 1995 indicate a return to pre-flooding levels with a median value of 3µg l^{-1} (three times higher than the EQS in FW, as shown in table 12. Note the EQS is for total Cd)

Dissolved Cd (µg/l) at Devoran Bridge

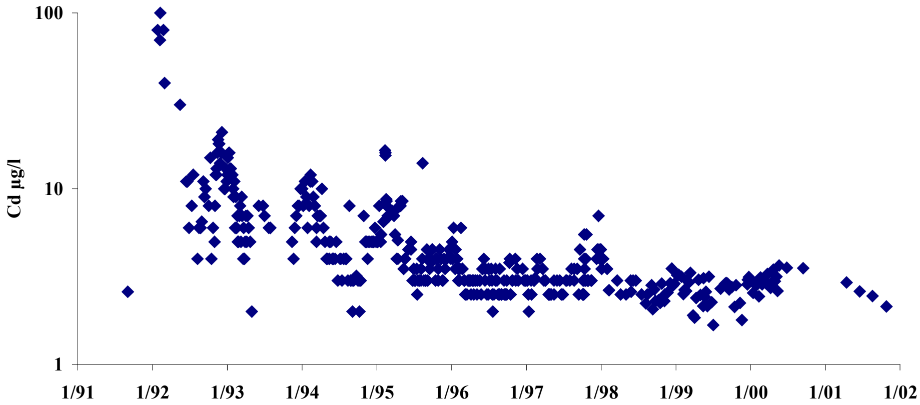

Figure 21. Dissolved Cd (µg l^{-1}) in the Carnon at Devoran Bridge (Data source EA). Note log scale for Cd.

Towards the mouth of Restronguet Creek dilution of Carnon water reduces average concentrations below EQ Standards (Harcombe and Point, figure 22). However, higher up, at Tallacks Creek, both estuarine and saltwater EQS values (5 and 2.5µgl^{-1}) may be exceeded at times (figure 22).

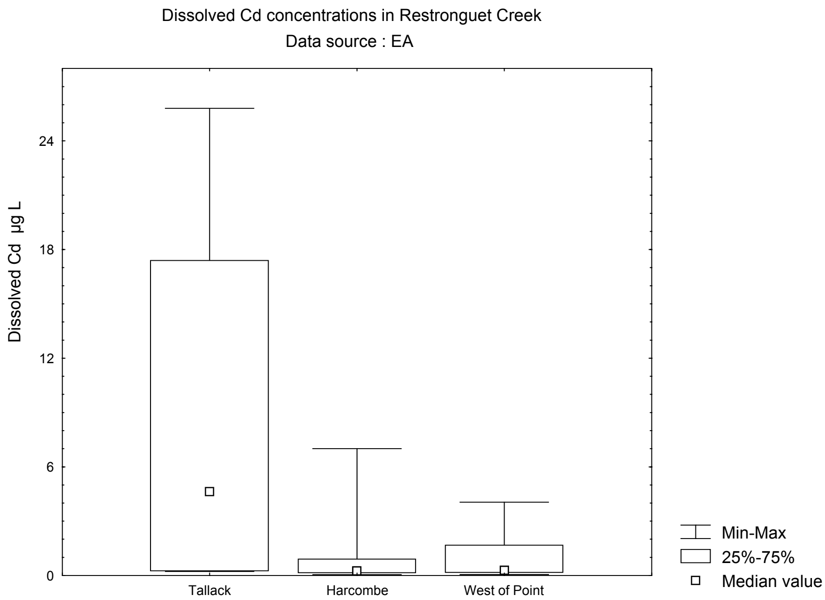

Figure 22. Dissolved Cd (µg l^{-1}) in Restronguet Creek

For the majority of sites in the upper estuary Cd values are significantly below EQS levels, though at less saline surface samples taken during the low water period may reach levels above the standard (up to 9 µg l^{-1} for data between 1992-5; see Figure 23). In the lower Fal estuary, median values across all sites are below the EQS though occasional elevated concentrations have been observed (up to 7 µg l^{-1} Cd) opposite Restronguet Creek. Cadmium concentrations are consistently low in the Helford Estuary and pose no threat to biota

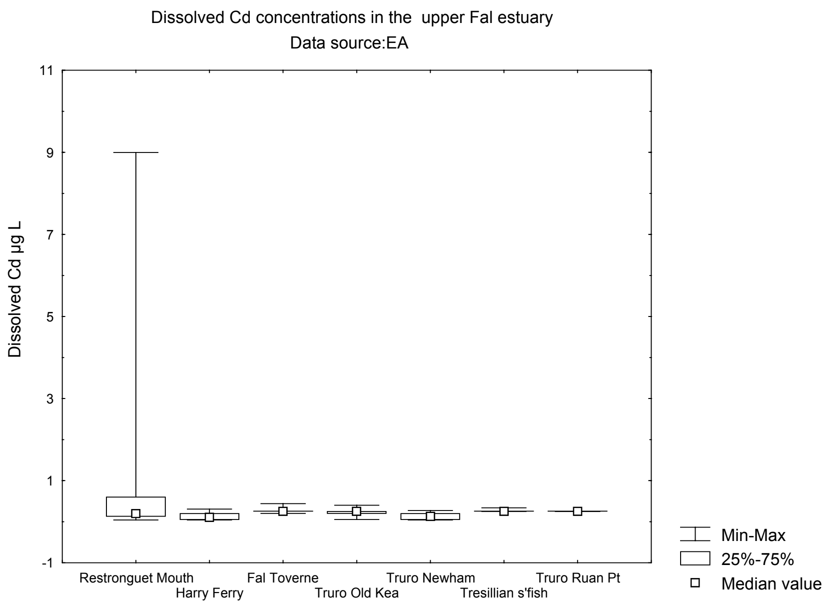

Figure 23. Dissolved Cd (µg l^{-1}) in the upper Fal Estuary.

Iron. Concentrations of dissolved Fe in the Carnon could not be found in the EA database but sources will be comparable to those described for other metals. In the estuary of Restronguet Creek concentrations will be strongly influenced by salinity. As described earlier dissolved Fe in FW rapidly oxidises and precipitates on mixing with sea-water. This is reflected in variability at the two Restronguet sites shown in figure 24. Thus although mean values are well below the salt water EQS (DSD) for Fe of 1000 µg l^{-1} values occasionally exceed this, depending on time of sampling.

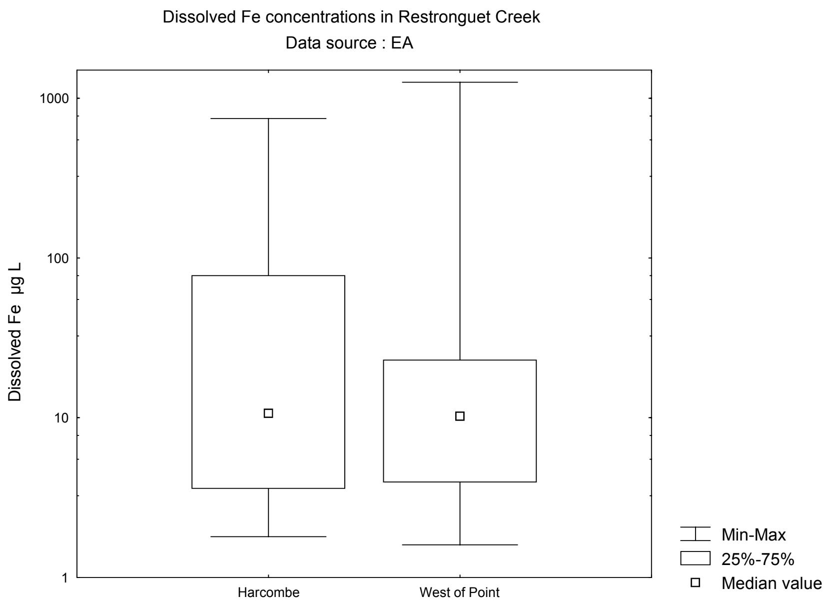

Figure 24. Dissolved Fe (µg l^{-1}) in Restronguet Creek. (note log scales; data are for 1993-5).

The same is true for samples at Restronguet mouth where dissolved Fe concentrations may vary from baseline seawater levels of 1-3 µg l^{-1} (at high water) to more than 450 µg l^{-1} (surface sample, low water) over the course of a tidal cycle. Further afield in the upper Fal, Fe concentrations are highest in the region of the Newham STW at Truro (median 63 max 191 µg l^{-1}). Apart from the occasional elevated value, levels of dissolved Fe at other site in the upper Fal are close to background (figure 25).

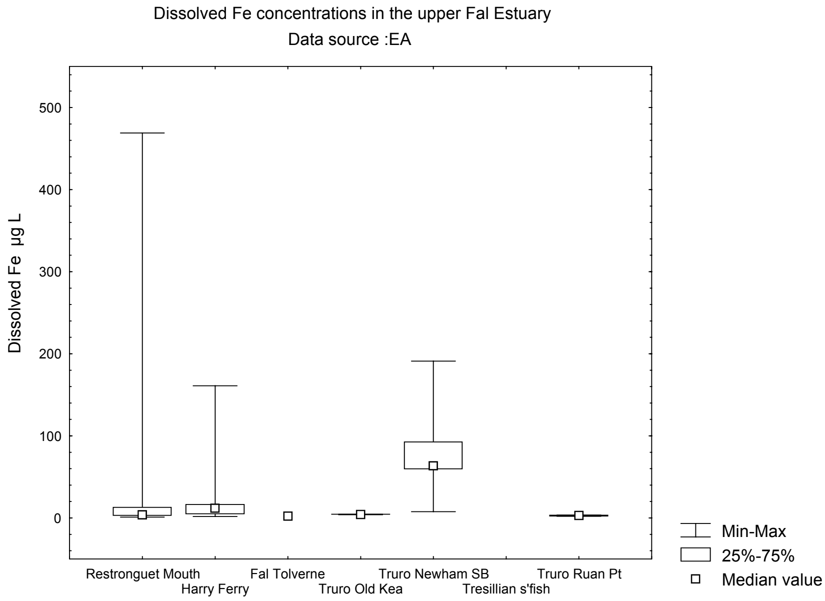

Figure 25. Dissolved Fe (µg l^{-1}) in Upper Fal Estuary .

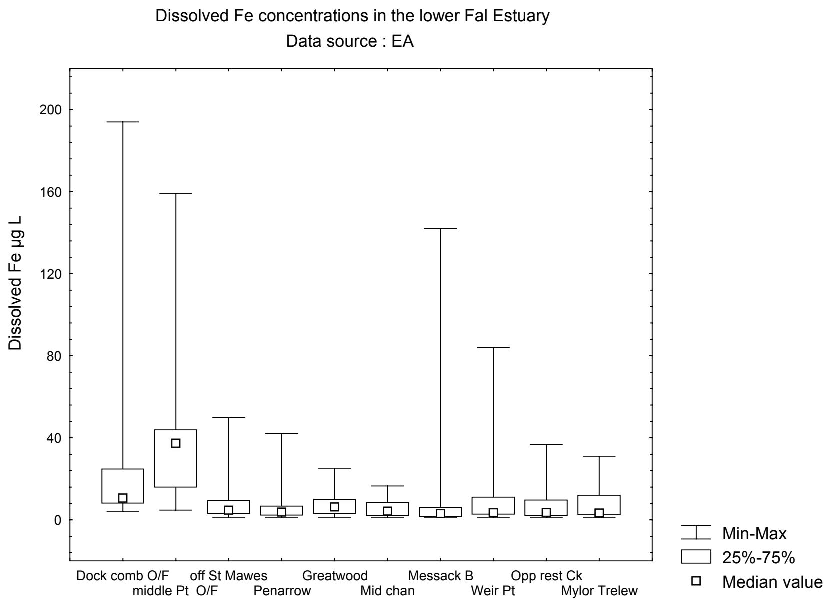

Figure 26. Dissolved Fe (µg l^{-1}) in the lower Fal Estuary. Data mainly from mid 1990's

In the lower Fal estuary median dissolved Fe concentrations lie between 3 – 6 µg l^{-1} but occasionally much higher values may be encountered perhaps reflecting the influence of metal rich low salinity water on the surface (figure 26). Median values near the combined outfall at Falmouth Dockyard and at Middle Point are somewhat higher (10 and 37.4 µg l^{-1}) but occasionally reach much higher concentrations (150-200 µg l^{-1}). Nevertheless, these still lie below the EQS for Fe.

A number of other *List II* metals occur in the database including chromium, boron and vanadium, though in tidal waters they were seldom found at concentrations above limits of detection. Detection limits used were generally well below the EQS concentrations indicating low background concentrations in the Fal and Helford cSAC. Other List II metals which were detected were:

Lead; For the data available (1985-2001), 1479 out of 1992 results were below limits of detection. Mean and median values for all sites were below the EQS of 25µg l^{-1}. However individual values above the EQS have been recorded in the lower Fal Estuary, and near Restronguet Creek. Temporal trends indicate reducing Pb concentrations at many sites.

Nickel: Of the 1216 results available for dissolved Ni, 698 were below detection limits. Between 1990-2001, the EQS of 30µg l^{-1} was regularly exceeded at Devoran Bridge, where the River Carnon issues into Restronguet Creek. Mean and median values for this site were 62 and 55 µg l^{-1} respectively (all data), whilst for both 2000 and 2001, mean and median values were 46 and 45µg l^{-1}. Ni concentrations were also elevated (up to 47 µg l^{-1}) at the mouth of Restronguet Creek although mean and median values for this site were lower than the EQS (11.4 and 7.5 µg l^{-1}). Positive results for all other sites were generally low and less than one tenth of the EQS.

6.1.2 Organotins

Water quality standards for organotins (µg l^{-1}) are shown below (table 13).

Table 13. Water Quality Standards for organotins

Tributyltin TBT	0.002µg l^{-1} MT
Triphenyltin TPT (and its derivatives)	0.008 µg l^{-1} MT

M= Maximum T= Total concentration (ie without filtration)

The EQS for TBT of 2 ng l^{-1}, as discussed in the previous section, is often exceeded in the Fal by a considerable margin. Acute effects in a range of species such as algae, molluscs, crustaceans and fish are reported to occur at dissolved concentrations of the order of 1 µg l^{-1} which is above the range usually encountered. However chronic effects are known to occur in several species down to 1ng l^{-1} and are clearly of significance in the Fal cSAC.

TBT. Of the 644 results for TBT (1996-2001), 244 were below the detection limits, which for tidal waters were generally set at 20ng l^{-1} or below. However in the vicinity of outfalls and discharges, detection limits were variable and set at up to <388ng l^{-1}, and exceptionally >15000ng l^{-1} TBT (Falmouth Docks outfall). For the purposes of this brief summary, only positive results are included, and tidal waters, and STW outfall/discharge waters are considered separately. (A more thorough review of EA TBT data is provided in Harris, 2001)

Tidal waters: Considering all positive results, mean and median values were 23 and 14ng l^{-1} respectively (n=177) with maximum values of 247 and 200ng l^{-1} occurring in the Penryn River at Islington Quay and the river Fal (mid-channel). Figure 27 summarises the data for the tidal waters of the Fal. Values are generally very high with maximum values exceeding the EQS at all but two sites. There are no significant temporal trends in the data to suggest a reduction of TBT levels in the water column.

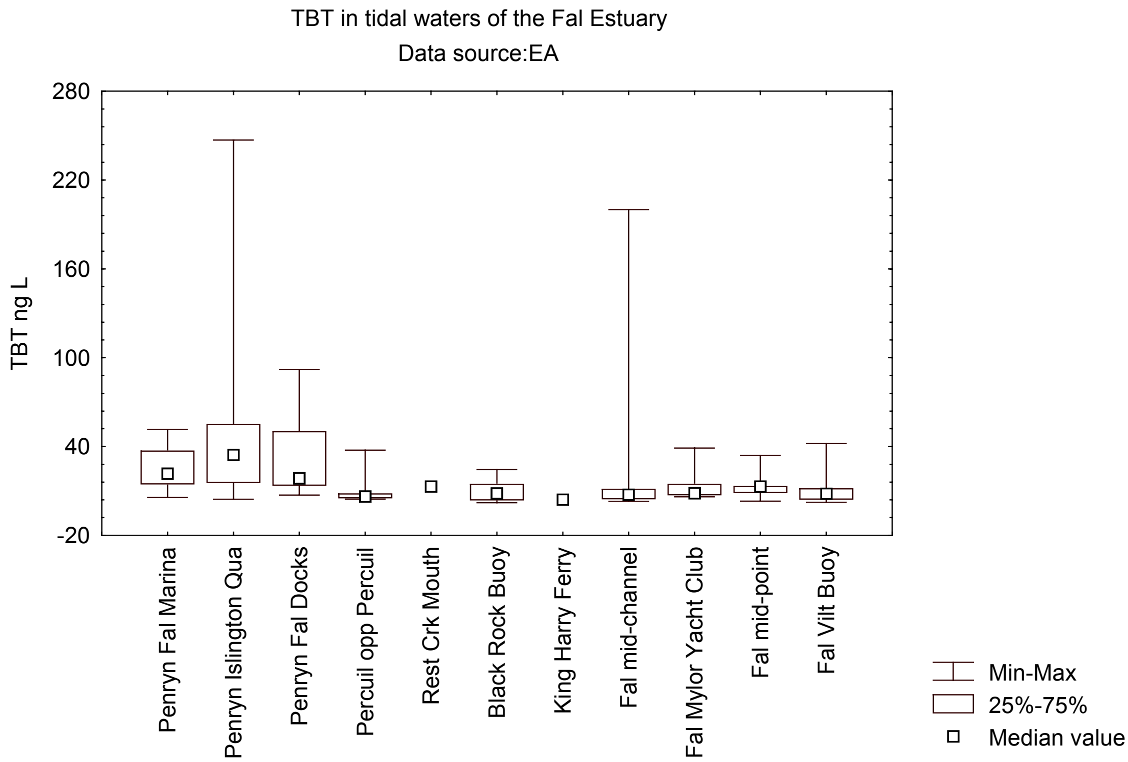

Figure 27. TBT (ng l^{-1}) in the tidal waters of the Fal Estuary

Outfall/discharge waters: Positive results available for TBT in discharges (1996-2001) are summarised in figure 28. Mean and median values for all discharge/outfall data are 2462 and 283ng l^{-1} respectively, with maximum values for TBT of 27.7 and 9.4µg l^{-1} occurring in Falmouth Docks combined outfall and surface waters. The outfall at Falmouth Docks stands out as the principal source of TBT in the area, with mean and median concentrations of 7240 and 6007ng l^{-1} respectively. Nevertheless, maximum TBT concentrations at all sites are far in excess of EQS values and no doubt contribute significantly to the TBT problem in the area. Again, there are no temporal trends in the data to date, to indicate a reduction of TBT concentrations in discharges and sewage treatment effluent entering the Fal Estuary. However recent washwater inputs of TBT from the Docks are now anticipated to be lower following the commissioning of an active-carbon treatment plant (EA personal communication).

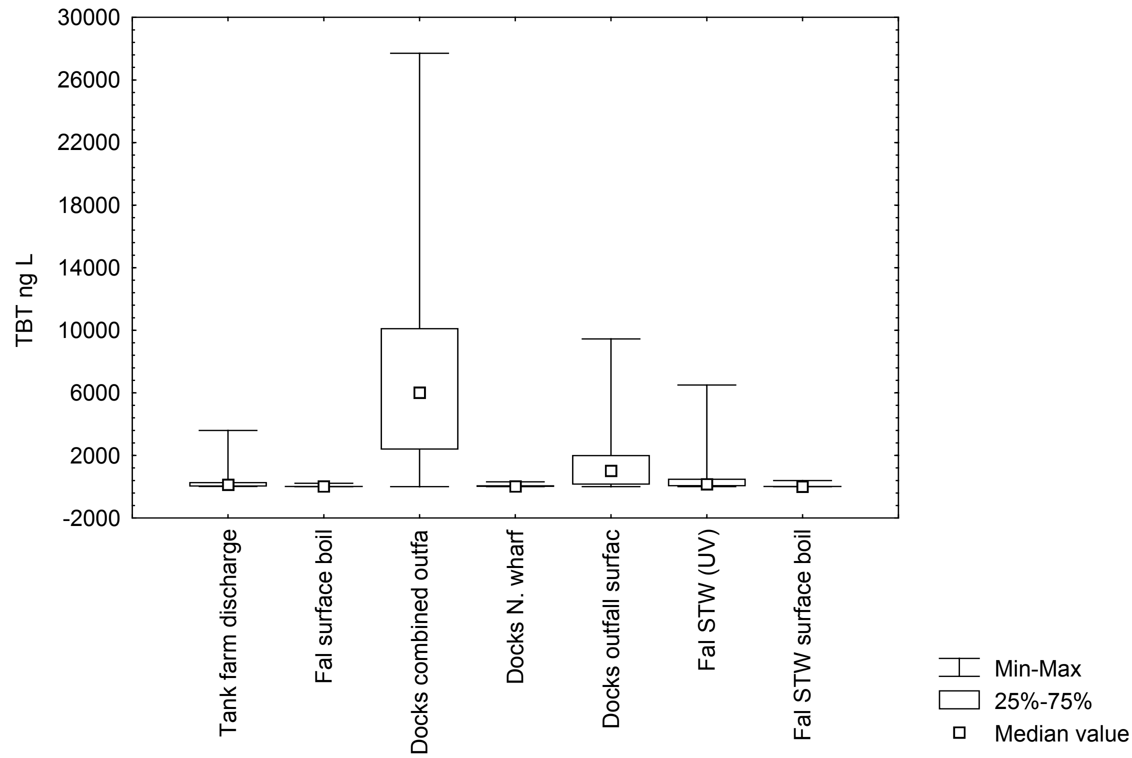

Figure 28. TBT (ng l^{-1}) in discharges entering the Fal Estuary (1996-2001)

TPT. Triphenyltin is also used in antifouling paints, though to a lesser degree than TBT. Data available for TPT monitoring in the Fal are for the period 1993-1997 and is mostly restricted to sites in the immediate vicinity of Falmouth STW and dockyard discharges. Of the 167 results available, 136 are denoted as below detection limits, however those 'less than' values range from 11-411ng l^{-1} and all exceed the EQS of 8ng l^{-1}. Positive results are available for two sites primarily (Falmouth Docks outfall and surface waters at Falmouth Dockyard outfall) and are in the range 6-216ng l^{-1}. With one exception, all positive results exceed the EQS value (mean and median values 45 and 30ng l^{-1}). Concentrations in this range would be expected to contribute significantly to the overall TPT burden in the estuary. In the absence of more recent data we cannot comment on current levels of TPT, however there was no indication in the available data to suggest that levels in discharges were reducing therefore we may conclude that TPT, like TBT is an ongoing water quality problem in the area.

Tetraphenyltin: Although tetraphenyltin is not listed as a 'list II' substance, up-to-date monitoring information is available in the database for non-discharge sites in the Fal Estuary, therefore a brief summary is warranted: Of the 559 results for tetraphenyltin (1996-2001) 515 are denoted as above or below detection limits. Positive results for tetraphenyltin in STW effluent and discharges in the Falmouth area are again very high (mean and median 398 and 38ng l^{-1}). However, results for tetraphenyltin in waters away from discharges and outfalls are largely reported as less than 7ng l^{-1} or less than 1ng l^{-1}.

No data are available for TBT, or TPT concentrations in the Helford river therefore we cannot comment on Helford waters with regard to organotins. This represents a substantial gap in our knowledge of the water quality in the cSAC. The estuary has approximately 500 deepwater and intertidal moorings, and there is a boatyard at the head of the estuary at Gweek. Around the time of the 1987 ban on use of TBT-based antifouling paints, Covey and Hocking (1987) noted a 'dramatic decline in numbers of dogwhelk *Nucella lapillus* due to TBT' and recorded that Nucella populations were present only at outer headlands Nare Point and Rosemullion Head. As noted in the previous section, TBT is readily adsorbed onto particles and may persist in the sediment where there is potential for re-release into the water column.

6.1.3 Hydrocarbons (oils, petrochemicals, PAHs)

EQS values for hydrocarbons are listed under two directives. The Bathing Waters Directive, under the heading organic substances recommends 300µg l^{-1} as the 90th percentile (non-routine sampling prompted by visual or olfactory evidence of hydrocarbon presence). The Shellfish Waters Directive, under organic substances, 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 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) has occurred in commercial species contaminated with crude and refined oils. GESAMP (1993) report studies detecting taints in fish and macro-crustaceans resulting from exposure during acute incidents, chronic discharges and in experimental studies. Experimental studies indicate that taints can be detected when fish are exposed to concentrations of oil in water in the range 0.01 to 1 mg 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 be still tainted days or weeks after a spill of fuel oil (GESAMP 1993).

As discussed above no literature concerning oil impact in the Fal and Helford could be found for review. However, the EA database provided limited information on unspecified 'hydrocarbon oils' which we include for reference purposes. Of the 253 values for hydrocarbon oil in water, 158 were below the limits of detection (generally 0.1 – 2 mg l^{-1}). Data were generally for the period 1992-1994, although values for the 'tank farm discharge site' were more recent (1996-2001). The data for those sampling sites with positive results are summarised in figure 29 and table 14.

We have looked at the data in the light of the EQS standards for the Directives mentioned above. These EQS may not necessarily apply to the sampling sites, nonetheless it serves as an indication of water quality with regard to hydrocarbon oil in the cSAC.

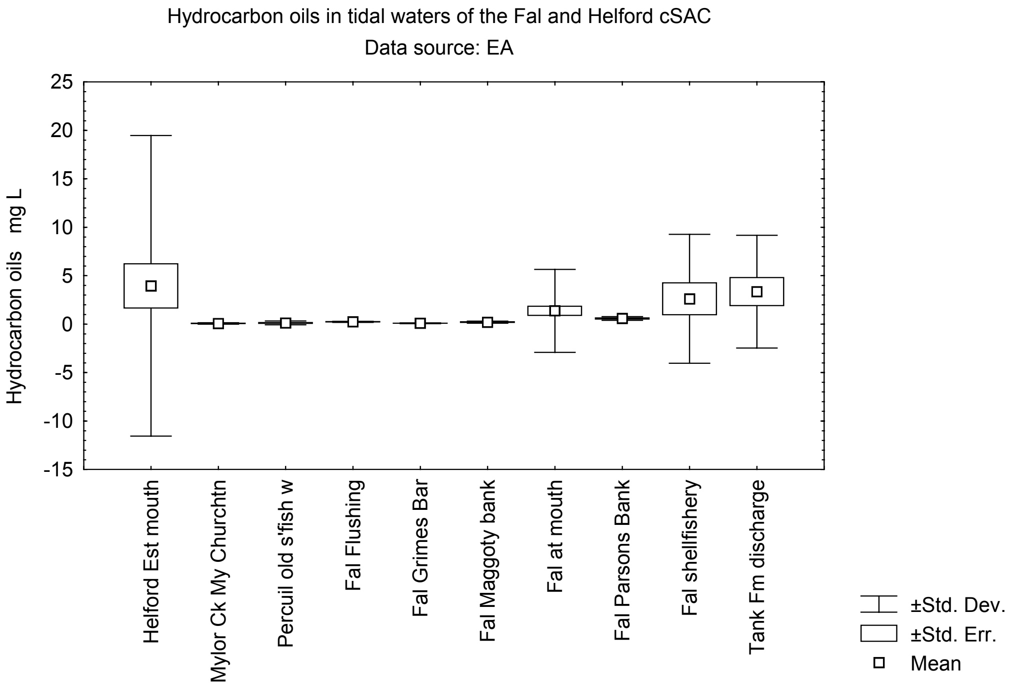

Figure 29. Hydrocarbon oils (mg l^{-1}) in the waters of the Fal and Helford cSAC
NB For statistical purposes, values below the limit of detection have been assigned a nominal value of 0.05 mg l^{-1} (half the mean and median 'less than' values).

Hydrocarbon oil concentrations at the Helford Estuary mouth were sometimes relatively high (mean, median and 90th percentile 4, 0.1 and 7.4 mg l^{-1} respectively) clearly exceeding both EQS's. Hewett (1995) noted that boatyards, boating activities and the few industrial units on the shores of the Helford posed a possible threat of oil /chemical pollution via spills, run-off and exhaust emissions. More recently, HVMCA (2000) report that oil products spilled from boats caused some damage, and also expressed concerns about deposition of airborne fuel particulates originating from operations at the nearby RAF Culdrose.

All sites listed can be considered liable to exceed the shellfish directive EQS as regards tainting concentrations, notably: the Helford River at the estuary mouth, the mouth of the River Fal and the River Fal (non-designated) shellfishery (between Restronguet Point and Loe Beach). Exceedences of the bathing waters EQS 90th percentile value 300µg l^{-1} occurred at 50% of the sampling sites (though in fact none are designated bathing waters).

The three EC designated bathing waters within the cSAC (Gyllyngvase, Swanpool and Maenporth have never recorded an EQS failure due to the presence of hydrocarbons (David Marshall EA - *pers comm*. 2002)

In the absence of comprehensive data for concentrations, trends and characterisation in water and discharges, we cannot comment accurately on the current water quality status with regard to hydrocarbons. It is recommended that this situation be addressed, particularly in the vicinity of shellfish beds. The presence of the dockyard at Falmouth, commercial shipping lanes offshore and ever-expanding marina facilities, indicates that the cSAC should be considered vulnerable to spillages and discharges.

Table 14. Hydrocarbon oils (mg l^{-1}) in Fal and Helford cSAC

Site	Period	n	Mean	median	90th percentile
Helford River Estuary mouth	1992-1994	46	4.0	0.1	7.5
Mylor Creek at Mylor Churchtown	1994	8	0.1	0.1	0.26
Percuil River (old Shellfish water)	1994	16	0.1	0.1	0.3
River Fal – Flushing	1994	5	0.2	0.2	0.3
River Fal – Grimes Bar	1993	8	0.1	0.1	0.15
River Fal – Maggoty Bank	1993	8	0.2	0.2	0.34
River Fal - mouth	1992 – 1994	77	1.4	0.6	3.2
River Fal – Parsons Bank	1993	8	0.6	0.1	0.9
River Fal – non-designated Shellfishery	1992 – 1993	16	2.6	0.1	15.5
Tank farm discharge – Pendennis Point	1996 - 2001	16	3.4	1.7	5.7
All sites		**208**	**1.9**	**0.05**	**2.82**

PAHs

Highest concentrations of PAHs occur in major estuaries and generally reflect inputs from a wide range of combustion processes involving industrial sources. There is a paucity of information for the Fal and Helford estuaries regarding PAHs, nevertheless we have included it for reference purposes.

Results for PAHs in tidal waters of the Fal and Helford cSAC are few and only available for the River Carnon at Devoran Bridge. 7 PAHs were listed on the database: naphthalene, chrysene, benzo(a)pyrene, benzo(b)fluoranthrene, benzo(k)fluoranthrene, ideno(123-cd)pyrene and benzo(ghi)perylene.

Highest concentrations were for naphthalene, for which 4 results were positive (710-820ng l^{-1} in 1999) whilst the remaining 35 were <500ng l^{-1}. The remaining 6 PAHs were present in concentrations of less than 30ng l^{-1}. Concentrations greater than 1µg l^{-1} (total PAHs) in estuaries are considered as significant (Cole *et al.,* 1999). The few results available suggest that levels such as this could occur in cSAC, therefore it is recommended that steps be taken to address the notable lack of information on PAH levels in the Fal and Helford cSAC.

6.1.4 Pesticides, Herbicides, PCBs and volatile organics

The report 'Pesticides in the Aquatic Environment' (1995) (prepared for the National Rivers Authority - TAPS 1995) presents information on the concentration of various toxic organic substances in estuarine and coastal waters of England and Wales and suggests very few areas in the UK where EQS values are exceeded (MPMMG 1998). None of these were in the Fal. Therefore only a brief review of sources and distributions of representatives of major groups of these synthetic organic determinands are given here.

We have interrogated the EA database for the following List 1 substances: HCH, DDT, pp-DDT, PCP, aldrin, dieldrin, endrin, isodrin, HCB, HCBD, chloroform, carbon tetrachloride, 1,2-dichloroethane, perchloroethylene, trichlorobenzene and trichloroethylene.

List 1 substances which were detected on a number of occasions included several organochlorine (OC) pesticides. OCs of relevance include agricultural pesticides such as 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).

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).

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

There are few results in the EA database for total hexachlorocyclohexanes (HCH). α-, β-, γ- and δ-HCH are listed separately but the majority of results are for γ-HCH (lindane) used in insecticides.

HCH isomers are recognised endocrine disruptors. HCH is toxic to invertebrates and fish at concentrations in excess of the EQS of 20ng l^{-1} (all isomers). Accumulation and persistence in the sediments can pose a hazard to infauna at concentrations greater than 0.32µg kg^{-1} (CCME 1999). 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 has resulted in lindane residues in sewage effluent. Hence, individual results in the EA database, above the EQS, were restricted to the vicinity of known sewage discharges and generally declined away from these areas, possibly due to transport or sedimentation on particulate material.

In the early 1990s, mean annual concentrations of lindane exceeded the EQS concentrations of 20ng l^{-1} (total HCH, annual average) at Falmouth (mid-point) outfall (now ceased operations) although more recently, annual concentrations were below the EQS at all sampling sites. In 2001, with the exception of one value of 11 ng l^{-1} (Falmouth STW surface boil), all tidal water samples from the Fal, and the shellfish site in the Helford, contained γ-HCH concentrations below the limit of detection (<2 ng l^{-1}).

Limited sampling of outfalls and freshwater, mainly in the early 1990s, failed to highlight any significant sources of γ-HCH other than Falmouth and Truro STW. The average γ-HCH concentration in Falmouth discharges was 146 ng l^{-1} in 1992 though this has been declining. Most samples in 2001 were <12 ng l^{-1} (calculated mean 6.6 ng l^{-1}). Surface boil samples at Newham, Truro STW contained 10.95 ng l^{-1} in 1993.

Dieldrin

Along with other 'drins' (aldrin, endrin, isodrin), dieldrin is another endocrine-disrupting organochlorine pesticide of potential concern (Allen *et al.,* 2000). Dieldrin is highly toxic to fish and other aquatic animals at concentrations of ~0.1µg $l^{-1}$and is said to be largely responsible for the dramatic decline of the otter population in the UK during the 1950s and 1960s. Dieldrin, used in sheep dips and seed dressings, leached into water systems and became bio-concentrated in the fatty tissues of fish such as eels, which are a major component of the otter diet. The result was a dramatic decline, which reached its nadir, nationally, in the early 1970s, when otters were restricted to a handful of upland tributaries on the cleanest rivers[1]. Dieldrin also accumulates in sediments posing a potential hazard to infauna.

With the exception of Falmouth STW outfall site in the early 1990's, dieldrin concentrations were below the EQS of 10ng l^{-1} at all sites. Concentrations in environmental waters have been reducing since the use of dieldrin in insecticides was banned in 1989 (NRA 1995). In the Fal and Helford, 92% of all measurements (n=837) between 1991-2001 were below detection limits, which were generally 10ng l^{-1} or less (<1ng l^{-1} in 2001).

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). Isodrin, endrin and aldrin were seldom found in measurable quantities in tidal waters from the European Marine Site: in 2001 all samples were <1ng l^{-1}, compared with EQS values of 5 ng l^{-1} (endrin and isodrin) and 10 ng l^{-1} (aldrin).

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

Limited sampling of outfalls and freshwater, mainly in the early 1990s, failed to highlight any significant sources of 'drins' other than occasional elevated levels of dieldrin (upto 55 ng l^{-1}) in effluent samples from Falmouth STW. Most samples in 2001 were <5 ng l^{-1}.

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).

Summary statistics for ppDDT concentrations in tidal waters of the Fal and Helford are predominantly (98%) '< values' with detection limits ranging from 0.4 -5 ng/l (1 ng/l in 2001). The maximum measured concentration was 1ng/l. Calculated annual averages thus demonstrate compliance with the estuarine EQS (10 ng/l ppDDT) by a substantial margin.

Analysis of outfalls and freshwater, mainly in the early 1990s, did not reveal significant sources of DDT other than occasional elevated levels (upto 27 ng l^{-1}) in effluent samples from Falmouth STW.

Volatile organic compounds (VOCs)

A number of List I 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 chloroform (trichloromethane), carbon tetrachloride (tetrachloromethane), and trichloroethylene (trichloroethene).

Chloroform 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 (Davies and Middaugh, 1978).

Chloroform has an EQS of 12μg l^{-1} (annual average) in all waters. Some small discharges of chloroform (and other organic solvents) occur from STW: mean annual concentrations have varied between 1 (2001) and 4.75μg l^{-1} (1991) in Falmouth discharges. However, following dilution, concentrations do not appear to be a direct toxicological concern for the cSAC. The majority of data for tidal waters were at, or below detection limits (83%). These DLs ranged from 0.1-1 μg l^{-1}. The highest concentration measured (3μg l^{-1}) was associated with Falmouth STW outfall. Annual average concentrations across all sites, based on ½DL, were invariably below the EQS by an order of magnitude, implying that chloroform is not a threat to the biota. Apart from Falmouth STW surface boil samples, most sites have not been monitored since 1993-4.

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 also has an EQS of 12μg l^{-1} (annual average) and, as with chloroform, concentrations do not appear to be a direct toxicological concern for the cSAC. Only two out of almost 300 measurements for tidal waters were marginally above the detection limits of 0.05-0.1μg l^{-1}: these were at the mouth of the Fal. Consequently, annual average concentrations across all sites, based on ½DL, were of the order of 0.025μg l^{-1} when last measured in 1994 -well below the EQS. Concentrations in STW samples (Falmouth) are < 0.2 μg l^{-1}, and do not appear to represent a significant source.

Statistics for another chlorinated solvent, trichloroethene, reveal comparable low levels in effluents (mean < 0.1 μg l^{-1} in 2001). Also concentrations in tidal waters are mostly below DL (< 1μg l^{-1}) - substantially below environmental standards (10μg l^{-1}) - and therefore are unlikely to impact on the interest features of the Fal and Helford cSAC.

Thus, with the exception of γHCH and dieldrin in crude sewage, sewage effluent and 'surface boil' waters, most list I pesticides, herbicides and chlorinated solvents occur largely at concentrations below analytical limits of detection. Detection limits are generally well below the EQS concentrations (annex 2) indicating low background concentrations in the Fal and Helford cSAC.

List II organic substances which featured in the database included PCSDs, cyfluthrin, sulcofluron, flucofluron, permethrin, atrazine and simazine, azinphos-methyl, dichlorvos, endosulphan, fenitrothion, Malathion, trifluralin, 1,1,1-trichloroethane, dimethoate and linuron. List II substances which were detected regularly included:

PCSD's, Cyfluthrin, Sulcofluron, Flucofuron and Permethrin.

These pesticides are used as mothproofing agents (also associated with the textile industry) for which EQS are reported as tentative values based on few, if any freshwater studies. Sources include factories where they are produced and also sites where they are used in the treatment of textiles and carpets. Hence, they may enter the aquatic environment, either in direct discharges or in sewage effluents.

PCSDs , flucofuron and sulcofuron inhibit the synthesis of certain enzymes whilst pyrethroids, such as cyfluthrin and permethrin, are neurotoxins. PCSDs readily accumulate in the tissues of fish, birds and marine mammals and acute toxicity to fish and invertebrates is reported at concentrations of 1mg l^{-1} Results from the NMP survey in 1994 indicated that some UK samples contain PCSDs in excess of the EQS of 0.05 mg l^{-1}. Data for flucofuron and sulcofuron are scarce but indications are that they may be less toxic (EQS 1 mg l^{-1} and 25 mg l^{-1}, respectively) and less likely to accumulate than PCSDs. EQS values for cyfluthrin and permethrin are 0.001 mg l^{-1} and 0.01mg l^{-1} in salt water.

EA monitoring data for these substances in the Fal and Helford are limited and largely restricted to outfalls and sites such as the Carnon River (Devoran Bridge) and the Percuil River (Tretham Mill) at the uppermost tidal limits. Recent results (post-1998) are consistently below detection limits and EQS values, indicating low background values which are unlikely to affect the interest features of the Fal and Helford cSAC.

Atrazine and Simazine

The *s*-triazine family of herbicides to which atrazine and simazine belong have been used in large quantities (several hundred tons annually) in the UK to control weeds on croplands, roads and railways. Both atrazine and simazine are on the UK red list of toxic compounds with a combined EQS of 2µg l^{-1} (statutory annual average) or 10 µg l^{-1} (maximum allowable concentration, non-statutory). 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. Only occasionally are elevated levels are seen in STW samples (0.1µg l^{-1} atrazine, 0.8µg l^{-1} simazine at Falmouth) though even these are unlikely to impair water quality beyond the mixing zone.

EA water sample data are generally for the period 1992-1999 although results for the Carnon River at Devoran Bridge are up until 2001. Values are predominantly '<', making statistical analysis impractical, however all results (including the sum of results for atrazine and simazine, where sampling times coincided) were significantly lower than the EQS.

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 22µg l^{-1} (Yockim, *et al* 1983), to freshwater fish such as rainbow trout at 10-40 µg l^{-1} and to marine fish at 45-440 µg l^{-1} (Verschueren, 1983; Manual of Acute Toxicity, 1986). The LC_{50} in *Daphnia pulex* is 240µg l^{-1} (Verschueren, 1983).

Trifluralin is on the UK red list with a statutory EQS of 0.1 µg l^{-1} (annual average) and 20µg l^{-1} (maximum allowable concentration, non-statutory). The herbicide has been analysed in tidal water samples from off the mouth of the Fal and Helford between 1990-1994. Since concentrations were always below detection limits (0.001-0.01µg l^{-1}), it is not possible to comment on distribution trends. The most recent data available (1994), summarised using ½ detection limits for '< values', places annual averages at 0.005 µg l^{-1} at these seaward sites, well below the EQS. Concentrations in freshwater and STW effluent samples were also below detection limits. Only a small number of samples have been analysed but on this evidence, trifluralin does not appear to be a significant threat to interest features of the cSAC.

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 3ng l^{-1} (Cole *et al*., 1999).

Concentrations of endosulphan in the tidal waters of the Fal and Helford, between 1990-1994, were invariably below detection limits (0.6-7 ng l^{-1} endosulphan A, 1.5 –7 ng l^{-1} endosulphan B). Evaluation of spatial and temporal trends is therefore not possible. The most recent data available (1994), summarised using ½ detection limits for '< values', places annual averages at 0.3 and 1ng/l for A and B isomers throughout most of the system. A very small number of fresh water and discharge samples have been analysed (all < DL, though this may be as high as 60 ng l^{-1}). On this evidence, sources of endosulphan are probably not a significant issue, but broader more up-to date analyses would be useful for confirmation.

Organophosphate pesticides (OPs)

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

Organophosphates enter the marine environment via spillage, industrial effluents, spray-drift and run-off from agricultural land. 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).

Results available for azinphos-methyl were primarily for one site, the River Carnon at Devoran Bridge. All of the results were 'less than' values, although it is worth noting that prior to 2001 these were set above the statutory EQS of 10ng l^{-1} (annual average). Azinphos-methyl could not be detected in effluent samples (DLs ranging from 20-565 ng l^{-1}).

Samples analysed for malathion at the mouth of the Fal and Helford in 1990/1991 were consistently below the limit of detection 5 ng l^{-1} - i.e. below the statutory EQS of 20ng l^{-1} (annual average). Malathion could not be detected in effluent samples (DLs ~15 ng l^{-1}).

The small amount of environmental data for OPs makes a valid assessment of the cSAC impossible, though if the values above are representative, it is doubtful that any of these compounds represent a threat to the interest features. Nevertheless it would seem advisable, in future, to examine sources within the site in more detail.

Most other list II substances were found at concentrations below analytical limits of detection. Insufficient positive results were available for analysis therefore these data were not considered further. Detection limits used are generally well below the EQS concentrations (unless discussed below) indicating low background concentrations in the Fal and Helford cSAC.

PCBs

PCBs (polychlorinated biphenyls) were widely used in industry during the mid-1900's, due to their chemical stability and electrical insulating properties. Evidence that PCBs persist in the environment and cause harmful effects led to a ban and/or severe restriction on their production and use in most developed countries in the 1970's and 1980's. Existing PCBs however, continue in use and enter rivers, estuaries and the sea via run-off from the land and industrial discharges.

There is a little monitoring data for PCBs in the EA database, principally ‘less than’ values for estuarine waters at shellfish sites within the cSAC (these range from <1 to <8ng l^{-1}), and one or two positive values (~2ng l^{-1}).

PCBs have low water solubility and a high affinity for suspended solids, therefore in the aquatic environment they are usually found in higher concentration in sediments, where they are amongst the most persistent of environmental contaminants. However, there are no data available for PCBs in sediments of the cSAC.

Like the majority of organochlorine substances, PCBs are lipophilic, dissolving more readily in fats than in water, therefore tend to accumulate in the fatty tissues of living organisms. Sediment dwelling organisms are obviously the most vulnerable of estuarine biota, and PCB’s accumulated in the tissues of invertebrates can be transferred and magnified along the food chain resulting in high concentrations in upper trophic levels such as fish, birds and marine mammals.

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

No information could be found on levels of PCBs in sediments, or infauna in the Fal and Helford cSAC, and data from the EA database for levels in unspecified shellfish (probably oysters judging by Cu and Zn levels) seem remarkably low, indicating little or no PCB accumulation (table 15). However, an unpublished report (Harris, 1993) records measurable levels of 5 PCB isomers in oysters from the Helford Estuary (up to 138µg kg^{-1} ww - above the OSPAR upper guideline for mussels). These were from a re-lay bed in at Lower Calmansack and had been collected from a number of areas in Carrick Roads before their transplantation to the Helford Estuary.

At the time, no PCBs were found in water or sediments from the re-lay site and it was concluded that the oysters were from elsewhere in the Fal or Helford Estuary (possibly Mylor Creek, Parsons Bank, East Bank, Malpas or Maggoty Bank). No further information is available, and although we cannot comment on current PCB contamination, this inconsistency highlights the need for more rigorous sampling regimes in the cSAC, especially for organochlorines and other organic substances that accumulate in sediments and affect estuarine biota through food-chain magnification.

Table 15. Organic contaminants (mean and maximum concentrations, expressed on a wet weight and approximate dry weight basis[1]) in unspecified shellfish (probably oysters)[2] from the Fal and Helford cSAC, 1990-1994 (EA Data)

compound	n	% <DL	concentration in 'shellfish' µg kg^{-1} mean[3] (wet wt)	mean (dry wt)	max (wet wt)	max (dry wt)	Site of max conc
ALDRIN	39	97	0.98	24.5	7.1	177.5	Helford at estuary mouth
DDE (OP')	25	96	1.70	42.6	0.8	20	Fal non designated shellfishery
DDE (PP')	31	65	2.61	65.4	14	350	river Fal at mouth
DDT (OP')	35	97	2.11	52.8	0.8	20	river Fal at mouth
DDT (PP')	34	68	4.46	111.5	40.6	1015	river Fal at mouth
DIELDRIN	39	79	1.59	39.9	6.3	157.5	river Fal at mouth
ENDOSULPHAN A	26	100	1.16	29.0	ns[4]	ns	
ENDOSULPHAN B	9	100	3.31	82.9	ns	ns	
ENDRIN	39	100	1.03	25.8	ns	ns	
γHCH-GAMMA	39	74	3.42	85.5	39	975	river Fal at mouth
HEPTACHLOR	32	97	1.22	30.6	0.5	12.5	river Fal at mouth
HCB	33	100	0.95	23.8	ns	ns	
HCBD	32	97	1.83	45.8	30.4	760	Helford river at estuary mouth
ISODRIN	33	100	1.17	29.4	ns	ns	
TRIFLURALIN	26	96	2.12	53.1	4.2	105	Truro river at Old Kea
PCB NO.101[5]	31	87	0.000013	0.00033	0.00012	0.003	Helford river at estuary mouth
PCB NO.118	33	88	0.000012	0.0003	0.00012	0.003	Helford river at estuary mouth
PCB NO.138	33	91	0.000006	0.00015	0.00004	0.001	Helford river at estuary mouth
PCB NO.153	33	91	0.000006	0.00015	0.00005	0.00125	Helford river at estuary mouth
PCB NO.180	33	100	0.000005	0.00013	ns	ns	
PCB NO.28	33	100	0.000005	0.00013	ns	ns	

[1]Values are expressed as wet weight in EA data base and converted to dry weight assuming a wet:dry wt ratio of 25 (from CEFAS oyster data)
[2]Most are likely to be oysters because of exceptionally high Cu and Zn levels, typical of these shellfish
[3]Mean calculated using ½DL where <value in data base.
[4]ns: Where all samples are below detection limits, no maximum sample given
[5]Values for PCBs seem exceptionally low and are being queried with EA, regarding units

Residue data for other organic contaminants in EA oyster samples (taken from various sites around the Fal and Helford between 1990 and 1994) are included in table 15, showing the majority of results to be below detection limits. This corroborates the conclusions of CEFAS who monitor levels of selected pesticides and other possible endocrine disrupting chemicals in bivalves from UK designated shellfish waters. A summary of CEFAS analyses between 1995 and 1996 implies that pesticide body burdens in most shellfish waters in England - including, presumably, the Fal and Helford - were close to or below detection limits. Typically, concentrations on a wet weight basis were; α,γ HCH - 3 μg kg^{-1}; dieldrin 1 - 7 μg kg^{-1}; DDT - 1- 3 μg kg^{-1} (CEFAS, 2001). These were similar to mean values in EA samples, though occasionally, individual EA results were an order of magnitude higher for pp DDT and γ HCH (max values, table 15). At these extremes, both compounds may exceed guideline values, though this is difficult to establish with certainty as advisory limits are usually expressed on a dry weight basis. Assuming a wet:dry weight ratio of 25 for oysters, concentrations in a small number of Fal and Helford samples would be above the recommended 'no-effects' guidelines of 100 μg DDT kg^{-1} (dw), 100 μg dieldrin kg^{-1} (dw) and 30 μg γ HCH kg^{-1} (dw) set by ADRIS[1] under the EU Shellfish Growing Waters Directive, to protect shellfish and their larvae (ADRIS, 1982). They are also above the 'high value' which categorises the top 15% of values seen in NOAA's[2] Mussel Watch programme in the USA. This would apply even if wet:dry ratios were considerably lower. It is stressed however, that these data are some 10 years old and 'exceedences' are rare.

The predominance of <DL values suggests that, overall, pesticides, herbicides and PCBs pose little toxicological threat to the European marine site as a whole. Unfortunately, bioaccumulation evidence appears to be based on a single species (oysters) and more extensive, targeted sampling should be considered to evaluate bioavailability more thoroughly. This should incorporate sediment-dwelling species such as clams and worms, and also algae and eelgrass.

6.2 Toxic contaminants: Sediment Quality

By definition, reliance on an EQS approach assumes that organisms principally derive (or are effected by) contaminants from overlying waters. In practice this may not always be the case. There is increasing recognition that as water quality is improving in most estuaries, sediments are likely to become a predominant and persistent (diffuse) source of contaminants. Action taken to improve water quality might be expected to confer similar benefits in sediments, though in reality there is insufficient information on the toxicity of contaminants in sediments to confirm this.

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[1] Association of Directors and River Inspectors in Scotland

[2] NOAA (National Oceanic & Atmospheric Administration): 'high' concentration' = 140 μg DDT kg^{-1}, 9μg dieldrin kg^{-1} (all dw),

At present there are no statutory quality standards for sediments in the UK. However, several guidelines on sediment quality are emerging, and CEFAS has recently cautiously recommended the Canadian/US approach (see Long *et al*., 1995; CCME 1999). This involves the derivation of Threshold Effects Levels (TELs - affecting the most sensitive species) and Probable Effect Levels (PELs - likely to affect a range of organisms) from published toxicity data to a variety of aquatic organisms (laboratory and field exposures). TELs are proposed as an Interim Sediment Quality Guideline (ISQG) value.

The recent advocation of sediment quality guidelines reinforces the importance of adequate sediment monitoring already encompassed within the Dangerous Substances Directive, where a 'standstill' provision applies whereby the concentration of the substance in sediments (and organisms) must not increase with time. Sediment quality is also crucial 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 caveats to the application of sediment guidelines as discussed by Grimwood and Dixon (1997) in the context of List II metals, including possible fundamental differences in sediment geochemistry and the use of test species which are not indigenous to the UK, in deriving thresholds. Also, with regard to the current site evaluation, metal concentrations in the Fal estuary system have been elevated for so long that organisms may have adapted to these conditions and 'effects' values derived from 'normal' populations may not be applicable. 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 and identify instances where efforts should be made to minimise further inputs of these substances to the cSAC.

Guideline values for metals are summarised below (table 16). Metals are the only determinands for which we have substantial reliable sediment data in the Fal and Helford cSAC. These are based on MBA's own published and unpublished records[1].

Table 16. Interim marine sediment quality guidelines (ISQGs) and probable effect levels (PELs; dry weight) for metals (from CCME, 1999)

Metal	ISQG ($mgkg^{-1}$)	PEL ($mgkg^{-1}$)
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

[1] [Note: these data have been collected at intervals over the last 30 years in connection with research projects and were not intended as a monitoring programme. Nevertheless the quality of the data is considered to be good. Methodologies have been successfully validated in numerous intercalibration exercises, including Quasimeme. The sediment measurements described here are for the <100µm fraction and are concentrated nitric acid digests(Langston *et al.,* 1994 a,b)].

With the possibility of biological effects in mind, we have summarised data for metals in Fal and Helford inter-tidal sediments by partitioning sites into 'zones' according to the interim sediment guideline criteria for each metal (Figures 30-36). Green zones denote areas where no harm to the environment is predicted (below ISQG's), grey zones are where effects cannot be excluded (between ISQG's and PEL's) and red zones are where harmful effects are expected (above PEL's).

In sediments of the cSAC all of the seven metals in table 16 are, to some extent, in concentration ranges where adverse effects to biota cannot be excluded. There are 'red zones' for many of the metals in all major rivers and tributaries of the cSAC. Arsenic is notable in that sediment concentrations at all sites (excluding Helford, for which there is little information) are within the range where harmful effects to biota are expected. Arsenic concentrations are particularly high in sediments in and around the area of Restronguet and Mylor Creeks (up to 5515 mg kg^{-1}). Such exceptional levels are a result of the area's mining history (discussed above), a legacy which continues to influence water quality in the cSAC: as indicated in the previous section evidence suggests there is now some re-release of As from sediments (or other sources in mid estuary), which is potentially problematic since As is a class I carcinogen.

Elevated levels also place sediment concentrations of Zn, Cu and Pb in the Fal estuary primarily in the 'red' category. Again, the highest levels are in and around Restronguet Creek, although biota in all areas of the cSAC (including Helford) are vulnerable and may be affected by these metals.

Hg levels in sediments generally fall into the 'grey zone' range, and are highest in Mylor Creek and off Weir Point at the mouth of Restronguet Creek, 1.9 and 1.0 mg kg^{-1}, respectively (both 'red zone' areas), although concentrations in sediments of the Helford River at Gweek (0.68 mg kg^{-1}) are also approaching the PEL. Mercury is the element usually singled out as being the most important in terms of biomagnification, and advisory limits exist for Hg in fish and shellfish (upper guideline of 1 mg kg^{-1} (dry weight) suggested for molluscs by the Oslo and Paris Commissions). Hg levels in Fal and Helford shellfish, up to 0.32 mg kg^{-1} dw were measured in Jan 1992 (MBA data) and were therefore within public health guidelines at that time. No recent data for Hg in shellfish are available. However, considering that top predators such as fish, birds and mammals are capable of further biomagnification of Hg residues, an up-to–date evaluation of Hg in sediments and biota of the cSAC is recommended.

Cadmium levels in sediments also fall primarily into the 'grey zone' range where effects cannot be ruled out, and are again greatest in Restronguet Creek (up to 12.63 mg kg^{-1}). Information for Cd levels in Helford sediments is sparse, although the few sites for which we have measurements suggest that Cd in the Helford may be relatively elevated (up to 2 mg kg^{-1} at Gweek).

Cr levels are fairly uniform in the estuarine sediments, albeit slightly elevated at Mylor ('grey zone' up to 66.3 mg kg^{-1}), and are by and large in the 'green' zone with no adverse effects predicted.

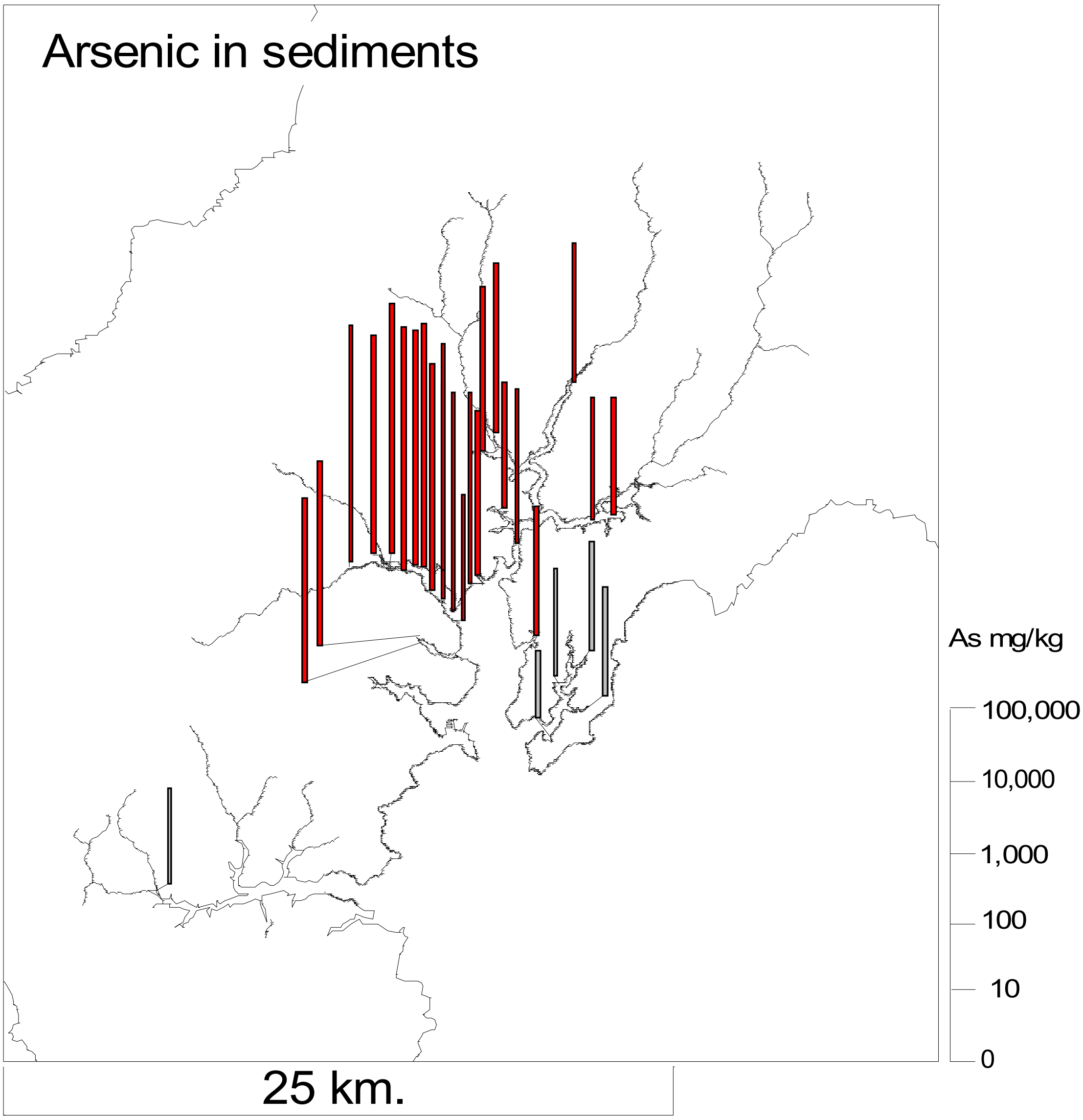

Figure 30. Arsenic in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. Note: logarithmic scale. (Data source: MBA)

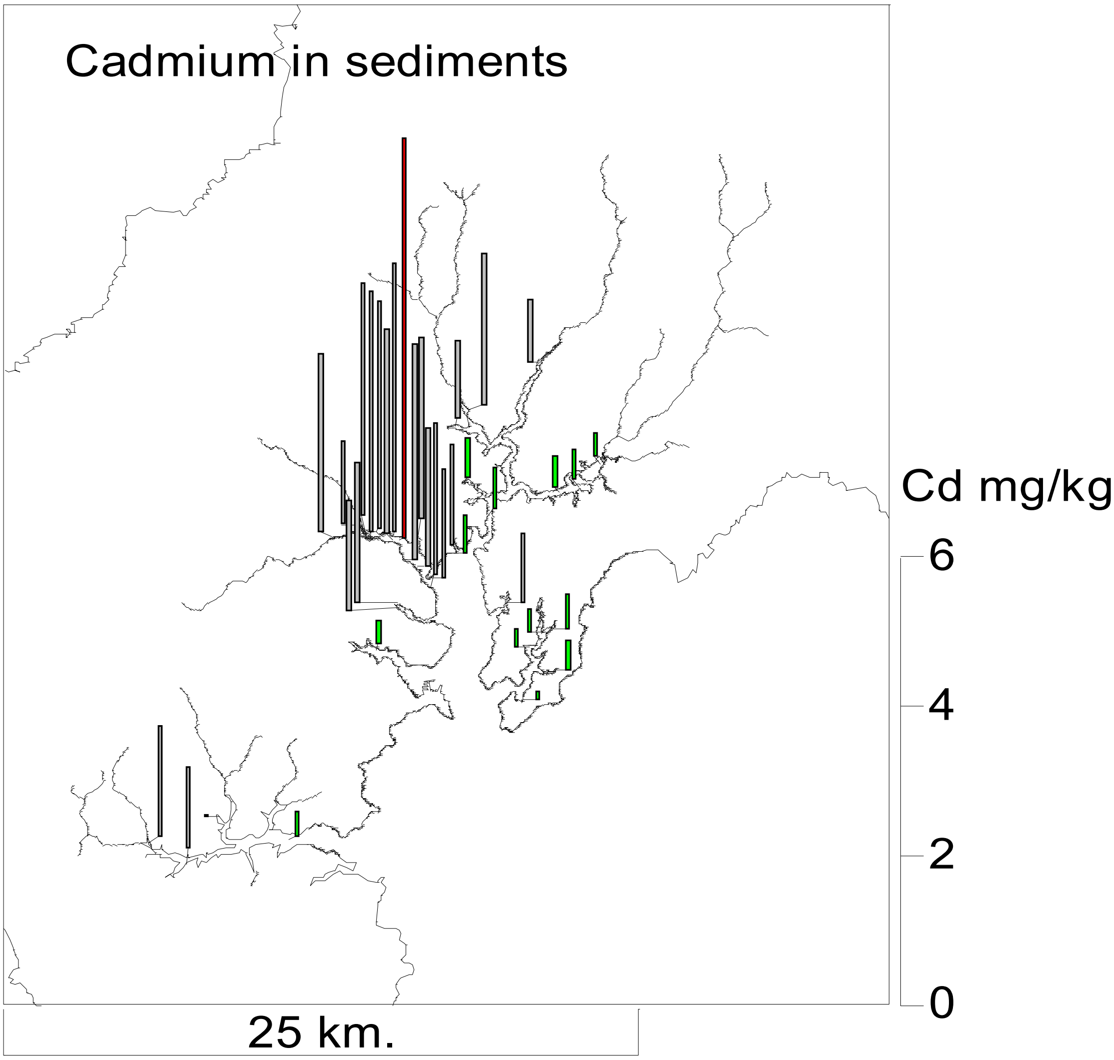

Figure 31. Cadmium in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

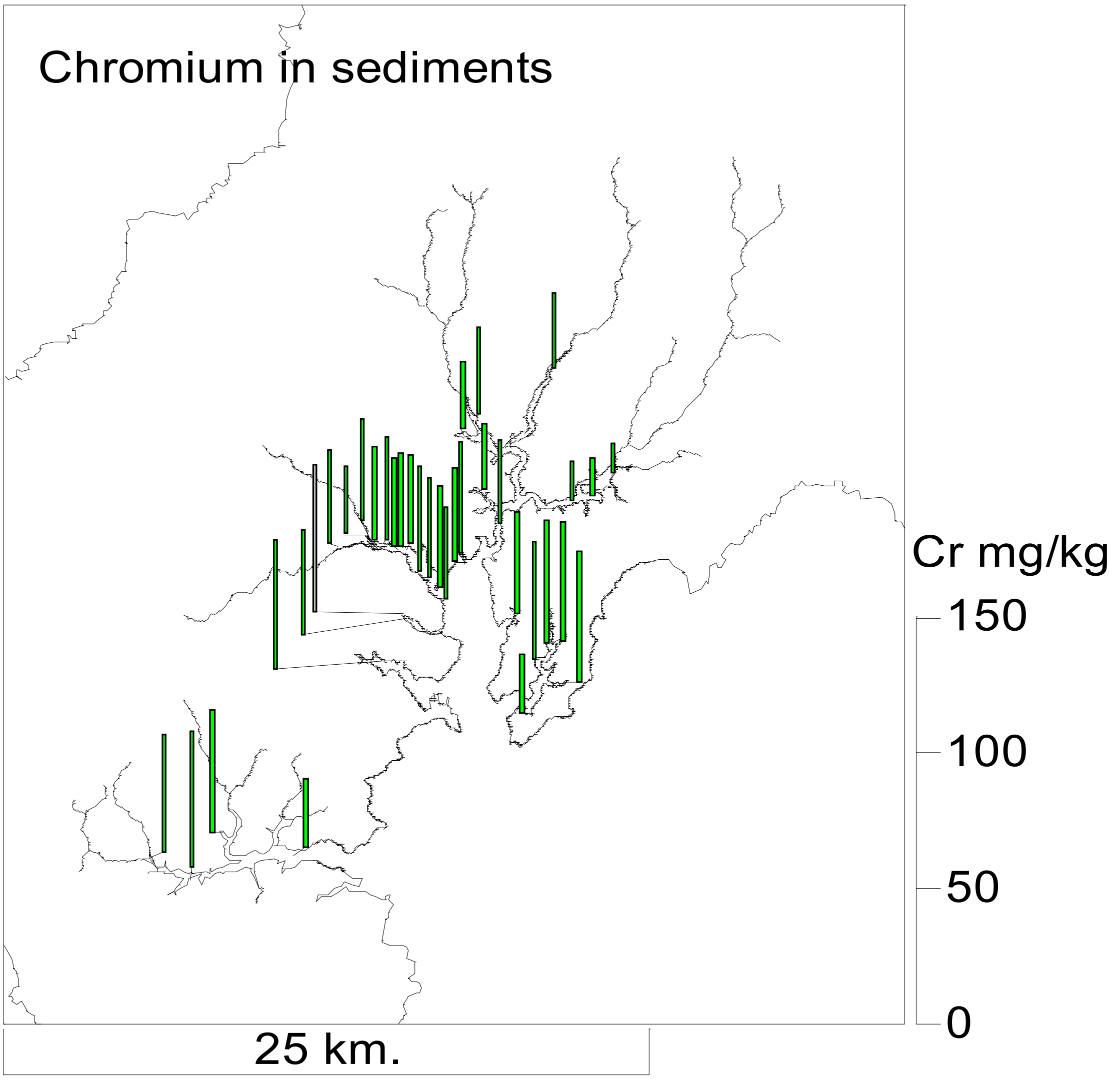

Figure 32. Chromium in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

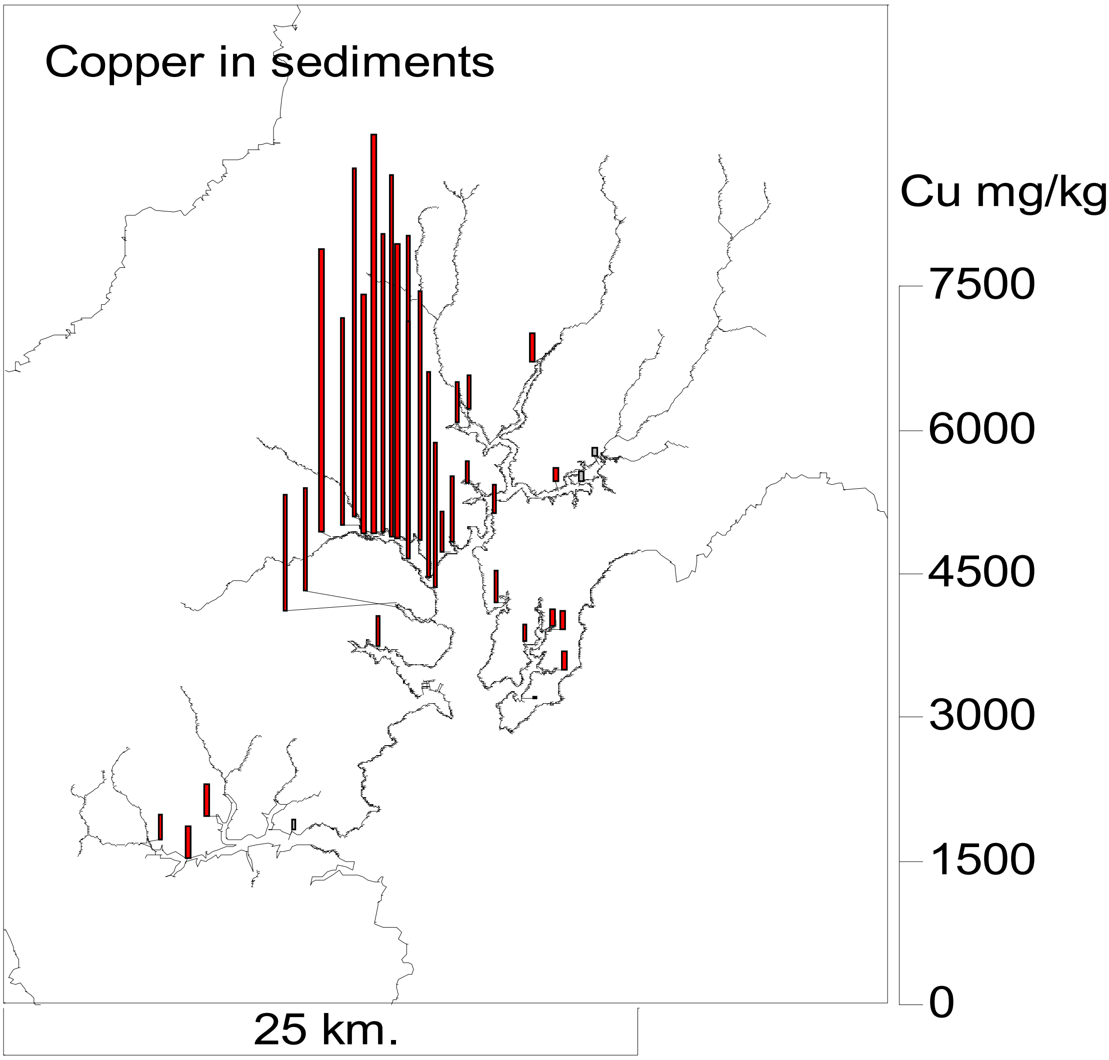

Figure 33. Copper in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

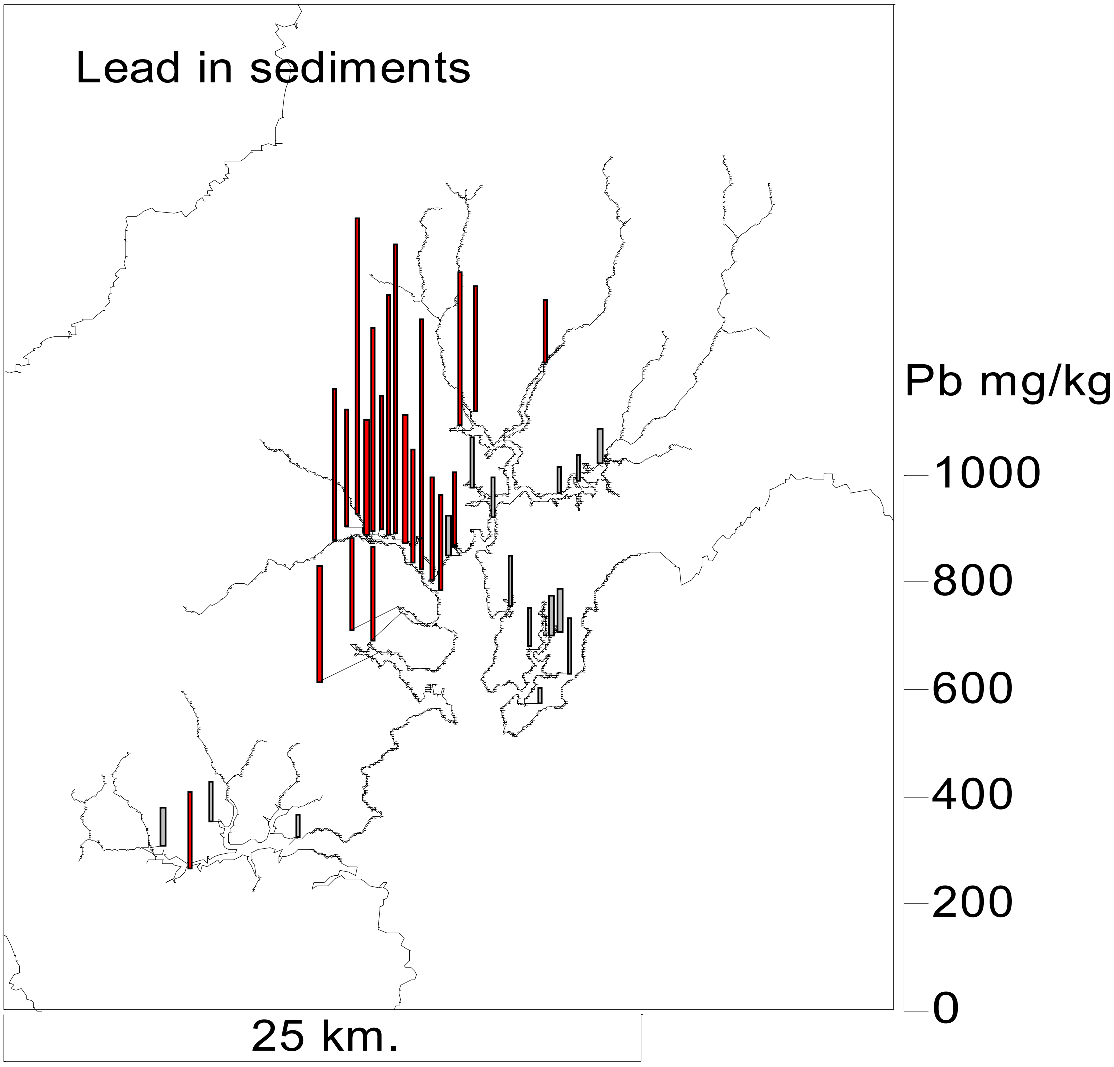

Figure 34. Lead in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

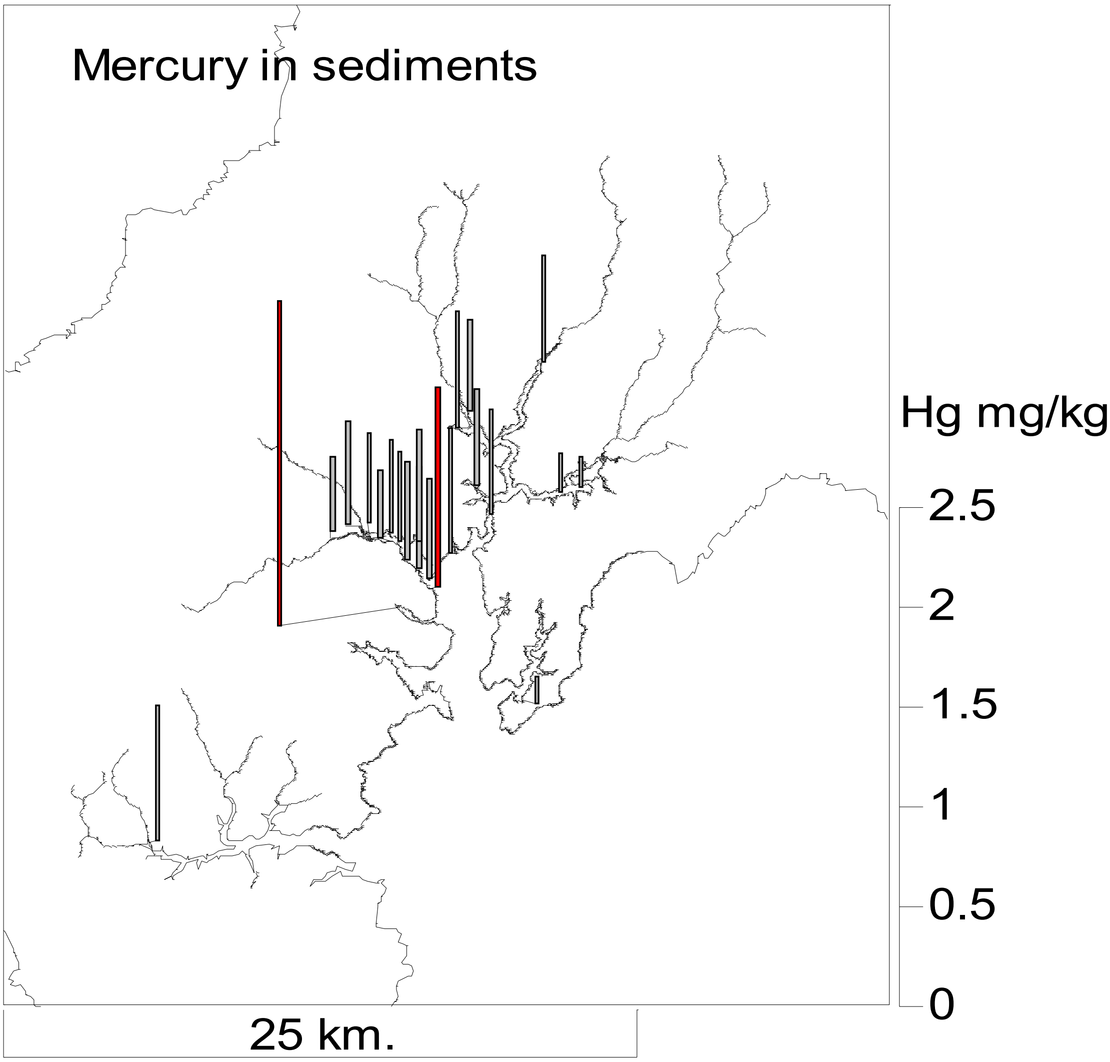

Figure 35. Mercury in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

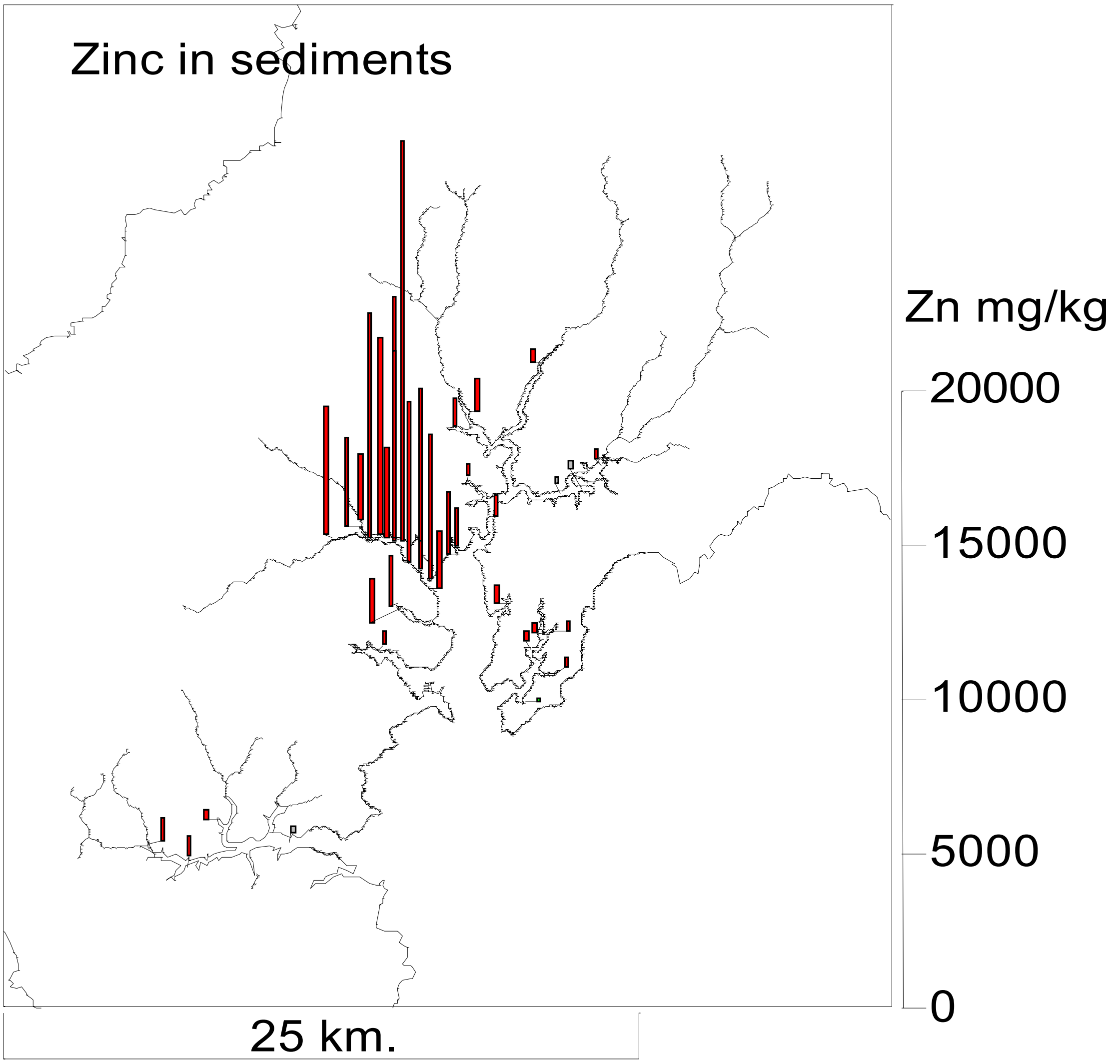

Figure 36. Zinc in sediment. Classification of the Fal and Helford cSAC into zones based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data source: MBA)

Thus, metal levels in Fal and Helford sediments are dominated by high concentrations in and around Restronguet Creek where, for most metals, adverse effects on biota are predicted. However, elevated levels also occur in some of the creeks and tributaries, especially towards the tidal limit - at Truro, Calenick and the Tresillian River, many of which also fall into the 'red' category according to the interim guideline criteria. 'red zones' in the Helford may be significant as the site is of particular conservation importance. Unfortunately, much of the existing data focuses on the Fal and Carrick Roads area of the cSAC, and little recent information is available upon which to evaluate the current status of the Helford. This notable lack of information should be addressed.

Environment Canada has not produced guideline values for TBT in sediments presumably because of insufficient information on toxicity levels. However, the Oslo-Paris Commission (1994) has drawn up a set of ecological assessment criteria for the identification of possible problem areas relating to TBT in the aquatic environment (table 17). These are derived from a diverse body of information, including national limit values (e.g. EQS), toxicity thresholds, sediment quality and body burden data. It should be stressed that these are 'guidelines' arising from evaluation of the best available evidence. Based on OSPARCOM guideline ecotoxicological assessment criteria (Oslo and Paris Commissions, 1994), much of the Fal would be classified as a 'problem area' though arguably these guidelines may be overcautious.

Table 17. OSPAR provisional ecotoxicological assessment criteria guidelines for TBT (Oslo and Paris Commissions, 1994)

Sample type	TBT concentration range (as TBT)	TBT concentration range (as Sn)
Water	0.1 –1 ng l^{-1}	0.04-0.4 ng l^{-1}
Sediment (1% organic C)	0.0001-0.001 µg g^{-1}	0.00004-0.0004 µg g^{-1}
Mussel (dw)	0.05-0.5 µg g^{-1}	0.02-0.2 µg g^{-1}

Perhaps a more realistic evaluation, based on limited publications and research into sediment toxicity of TBT, would be that effects on benthic organisms might be expected to occur in the range 0.1- 0.3µg g^{-1} dw sediment (Langston *et al.,* 1990; Langston and Burt, 1991; Austen and McEvoy, 1997). Even then however there are large parts of the Fal system (particularly near the mouth) where TBT would be in the range expected to cause effects to organisms. This is clearly one case where at the very least efforts should be made to demonstrate that 'stand still' requirements are being met and where possible to minimise further inputs of these substances to the cSAC. Under the remit of the Habitats Directive (attainment of Favourable Conservation Status - FCS) it may be appropriate to seek improvements.

The majority of data for other organic contaminants in sediments are restricted to one site off Falmouth (Middle Point) sampled on up to three occasions between 1990 and 1994 (entries are labelled as any solid/sediment in WIMS and assumed to be sediment). 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.

Concentrations of most organic contaminants were, in fact, predominantly below detection limits. 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 18).

The 'apparent' average concentration for γ-HCH was above the ISQG guideline and just below the PEL this is probably artifactual, since two out of three values were below detection limits. Similarly, even though the sum of DDT and DDE isomers appear to be above the respective ISQG guidelines this is due to the fact that the values are below detection limits which are themselves sometimes above the guideline value. None of the other compounds exceeded interim sediment guidelines.

Table 18. Organic contaminants in sediments off Falmouth, Middle Point (1990-1994), 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	n	Mean[a] µg kg^{-1}	Values <DL (%)	ISQG[b] µg kg^{-1}	PEL[c] µg kg^{-1}
ALDRIN	3	0.37	100%		
DDT (PP)[d]	3	0.96	92%	1.19	4.77
DDT (OP)[d]	3	2.66	100%	1.19	4.77
DDE (OP)[d]	3	1.16	100%	2.07	3.74
DDE (PP)[d]	3	0.96	100%	2.07	3.74
DIELDRIN	3	0.67	100%	0.71	4.3
ENDOSULPHAN ALPHA	2	0.63	100%		
ENDOSULPHAN BETA	2	0.63	100%		
ENDRIN	2	0.25	100%	2.67	62.4
γ-HCH	3	0.81	66%	0.32	0.99
HEXACHLOROBENZENE	2	1.75	50%		
HEXACHLOROBUTADIENE	2	0.83	100%		
ISODRIN	3	0.63	100%		
TRIFLURALIN	1	3.1	0%		

[a]results with '<' sign are halved for calculation
[b] interim marine sediment quality guidelines (ISQG) and
[c]probable effects levels (PEL) (CCME, 1999)
[d] guideline = sum of pp' and op' isomers;

It is therefore unlikely that sediment pesticide concentrations would cause harm, if this level of contamination were typical throughout the Fal and Helford cSAC. However this should be verified by further sampling, particularly with a view to identifying sources and characterising bioavailability.

There do not appear to be any data for PAHs or PCBs in sediments from the European marine site. It is recommended that future sampling programmes incorporate more high quality information on sediment contaminants over a broader range of sites.

6.3 Non-Toxic Contaminants

6.3.1 Nutrient quality criteria.

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. The ratio of these principal nutrients in river water is generally 10:1.

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

Although no statutory standards exist for N and P in estuarine and marine SACs, 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 Fal and Helford cSAC, 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 **50 mg 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 criteria was reported some years ago in the EPA 'Red Book' as **0.1 µg l^{-1} (as P)** to protect estuarine and marine organisms against the consequences of bioaccumulation (EPA, 1976). However, this was not established as threshold for eutrophication and is currently under review.

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

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

- There is a proposed EQS of **0.021mg l^{-1} un-ionised ammonia** (NH_3-N) for the protection of saltwater fish and shellfish, although due to the technical difficulties in measuring the unionised form, total ammonium is usually monitored and NH_3 calculated. However, even calculations can be difficult as the relative proportion of ionised and un-ionised ammonia depends on salinity, temperature and pH.

- 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).

Recognising the dilemma in arriving at standards, there have been attempts in recent years to develop and test General Quality Assessment (GQA) schemes for nutrients in estuaries and coastal waters which may be adopted nationally and internationally. One such scheme is proposed for the EA by the WRc as part of their General Quality Assessment (GQA) scheme (Gunby *et al.,* 1995). For nitrogen, this method uses the combined concentrations of nitrate, nitrite and ammonium concentrations in tidal waters (total inorganic nitrogen, TIN), as an approximation of bioavailable nitrogen. Assuming conservative behaviour for TIN and a standard concentration in marine waters, allows the TIN concentration in the freshwater input to be calculated, provided salinity data are available. For phosphorus, Total Reactive Phosphate (TRP - phosphate in unfiltered samples) is measured and as for nitrogen, the concentration in freshwater calculated. Estuaries are then be grouped according to the following class boundaries (table 19):

Table 19. TIN and TRP classification criteria for estuaries (based on Gunby *et al.*, 1995)

Class	Median projected TIN (mg l^{-1})	Class	Median projected TRP (mg l^{-1})
A/B	5.3	A/B	0.087
B/C	8.1	B/C	0.35
C/D	11.1	C/D	1.00

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

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

We have used measurements of total inorganic nitrogen (TIN), nitrate, total reactive phosphate and ortho-phosphate as markers of nutrient status in different regions of the cSAC. (Nitrate typically makes up the largest proportion of TIN inputs to estuaries, with nitrite and ammonia usually accounting for < 10%)[1].

GQA scheme: TIN and TRP

In their efforts to establish nutrient budgets for the Fal and Helford, shown earlier, Fraser *et al*., (2000) report that low turbidity over most of the region imply that nitrate, nitrite and ammonium distributions within the estuaries will show essentially conservative mixing. Thus, assuming insignificant mid-estuarine source inputs or losses, this method *should* allow an estimation of water quality in the cSAC in relation to other estuarine systems using the GQA scheme described above. Cole *et al*., (1999) made such a comparison, having extrapolated freshwater values (from seawater values) on the basis of conservative mixing. Using these criteria, the projected classification for TIN is average for the Fal and below average for the Helford. Projected TRP for both sites is high, more than twice the average for the Fal (table 20).

The use of actual EA data for TIN and TRP in freshwaters entering the cSAC, for subsequent years (1999-2001), allows us to classify cSAC waters according to the same scheme (table 21). The resultant TIN classification is similar to the earlier projection of Cole *et al.,* (1999), but for TRP, 1999-2001 values are significantly lower than the projected FW concentrations described by Cole *et al.,* (1999), and the Helford and Fal would now be classed as grade A, (compared to D, previously). It is not known for certain whether this discrepancy is artifactual (a result of using real measurements as opposed to modelled values) or the result of genuine improvement in water quality. However the latter explanation seems unlikely since there is no evidence of widespread temporal change in Agency data. Nevertheless, this example serves to illustrate the problems of assessing nutrient status.

[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 FW) P is thought to be limiting and at N:P ratios < 10:1 (mainly in SW) N is thought to be limiting (though there are 3 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). In many estuaries however enrichment may be such that nutrients are more likely to promote algal growth; turbidity may be the limiting factor instead (Parr *et al.,* 1999).

Table 20. Classification nutrient status of selected estuaries in England according to GQA TIN/TRP projection methodology (Cole *et al.,* 1999)

Estuary	Projected median TIN concentration (mg l^{-1}) in freshwater	GQA TIN class	Projected median TRP concentration (mg l^{-1}) in freshwater	GQA TRP class
Blackwater	14.3	D	6.8	D
Camel	5.9	B	0.4	C
Carrick	**5.4**	**B**	**4.6**	**D**
Colne	12.7	D	4.2	D
Crouch	11.3	D	5.3	D
Dart	4.3	A	0.2	B
Deben	11.5	D	6.2	D
Exe	5.4	B	0.3	B
Fal	**9.4**	**C**	**5.1**	**D**
Fowey	4	A	0.1	A
Hamford Water	10	C	6.8	D
Helford	**7.3**	**B**	**3.2**	**D**
Humber	8.8	C	0.1	B
Itchen	5.6	B	0.3	B
Lynher	5.5	B	0.1	A
Medway	5.1	A	0.4	C
Mersey	7.1	B	0.4	C
Nene	15.1	D	0.9	C
Ore/Alde	9.5	C	-1.0	A
Orwell	14	D	3.2	D
Ouse	12.2	D	0.8	C
Roach	11.9	D	11.4	D
Severn	7.6	B	0.5	C
Stour	13.3	D	2.5	D
Tamar	4.6	A	0.2	B
Test	6.3	B	0.3	B
Thames	12	D	2.4	D
Wash	13	D	1.5	D
Welland	13.1	D	0.4	C
Witham	21.9	D	0.5	C
Wyre	9	C	7.9	D
Yare	9.7	C	0.6	C
Yealm	5.9	B	4.2	D
MEAN	**9.5**	**C**	**2.5**	**D**

Table 21. TIN and TRP in waters entering Fal and Helford cSAC based on observations 1999-2001. (Data source: EA)

Estuary	Year	Median TIN conc. (mg l^{-1}) in freshwater	GQA TIN class	Median TRP conc. (mg l^{-1}) in freshwater	GQA TRP class
Carrick	1999	5.40	B	0.302	B
	2000	6.80	B	0.2065	B
	2001	8.61	C	1.815	D
Fal	1999	7.58	B	0.023	A
	2000	6.48	B	0.029	A
	2001	7.2	B	0.021	A
Helford	1999	5.56	B	0.026	A
	2000	5.66	B	0.033	A
	2001	5.85	B	0.036	A

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

The issue of whether or not to focus on nutrient concentrations in the tidal waters or loading criteria has been a contentious one among both scientists and managers. As noted above, the characteristics of estuaries differ significantly, and therefore nutrient sources, their fate and effects in the estuarine environment are not easily predicted. Rather than relying on a classification scheme for the estuary as a whole it may be more beneficial to investigate the distribution of key determinands in finer detail: Data for different parts of the cSAC and its catchment are treated separately in an attempt to further apportion sources of nutrients.

6.3.2 Phosphate

The freshwater catchment of the Fal lies in an intensive farming area (both arable and dairy). Farming practice in the region has changed over the past 60 years, evolving from unfertilised marginal grazing to fertilised intensive agricultural grazing land with an accompanying increase in livestock. This represents an increase in the nature and intensity of potential nutrient sources to the estuary. The underlying bedrock is impermeable and annual rainfall relatively high, resulting in an estimated annual runoff of 719mm (Fraser *et al.,* 2000). There are no major urban centres in the freshwater catchment and the population is scattered throughout the area in small villages and farmsteads. Around the lower reaches of the estuary (Carrick Roads), centres of population are Truro, Falmouth and Penryn, and again, small villages and isolated farms are scattered throughout the area. The Carnon valley catchment to the northwest of Carrick Roads drains an area of historic mining activity.

Carnon Valley sources: Figure 37 shows values for phosphate concentrations at some principal fresh water sources in the Carnon Valley (including the adits, tailings dam and effluent stream at Wheal Jane) between 1985-2001. At certain sites, relatively high limits of detection (0.5 mg l^{-1}) have been applied which have the effect of inflating values somewhat. Mean and median values for ortho-phosphate in rivers and streams throughout the Carnon (excluding STW sites) are 0.23 and 0.08mg l^{-1} respectively (n=1965). Mine drainage sites are implicated as an important phosphate source, with values at Devoran Bridge, Clemows Stream and the Wheal Jane adit reaching 0.7 and 0.8, and an isolated extreme value of 3.3 at Nangiles adit.

On a temporal scale (1985 – 2001), ortho-phosphate values in Carnon rivers and streams show peaks between 1989 – 1990, then drop before steadily increasing from 1992 – 2001. Values have risen significantly ($r = 0.4$, $p<0.001$) in the Carnon river at Bissoe Bridge, and in water entering Restronguet Creek at Devoran Bridge ($r=0.35$, $p<0.001$) over the 15 year period.

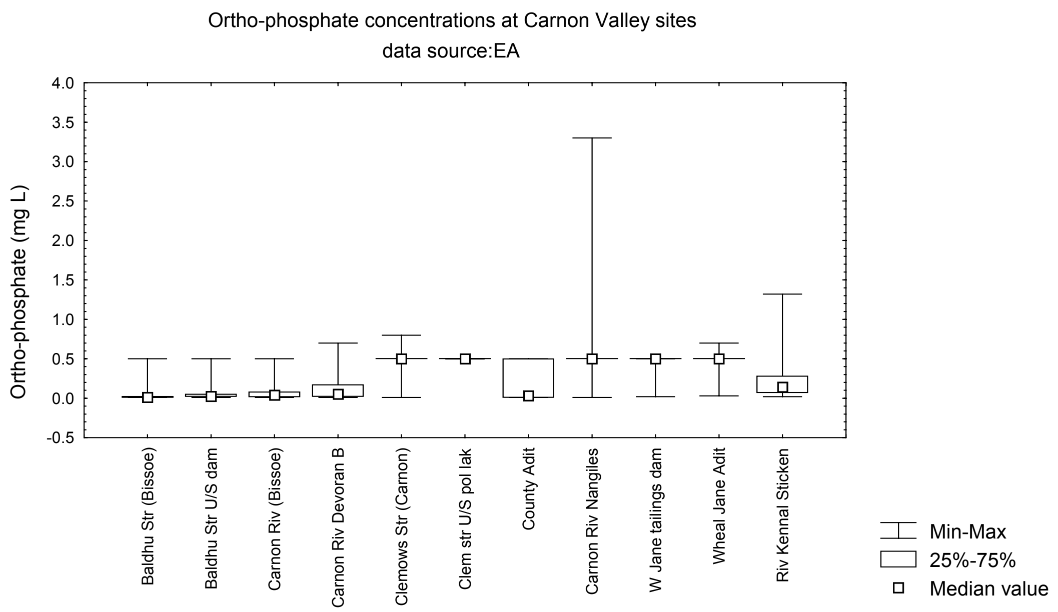

Figure 37. Phosphate concentrations (mg l^{-1}) in waters at Carnon Valley sites. (1985-2001) Data source: EA NB Where concentrations are classified as below detection limits (0.01 – 0.5mg l^{-1} dependant on site/date), detection limits have been applied.

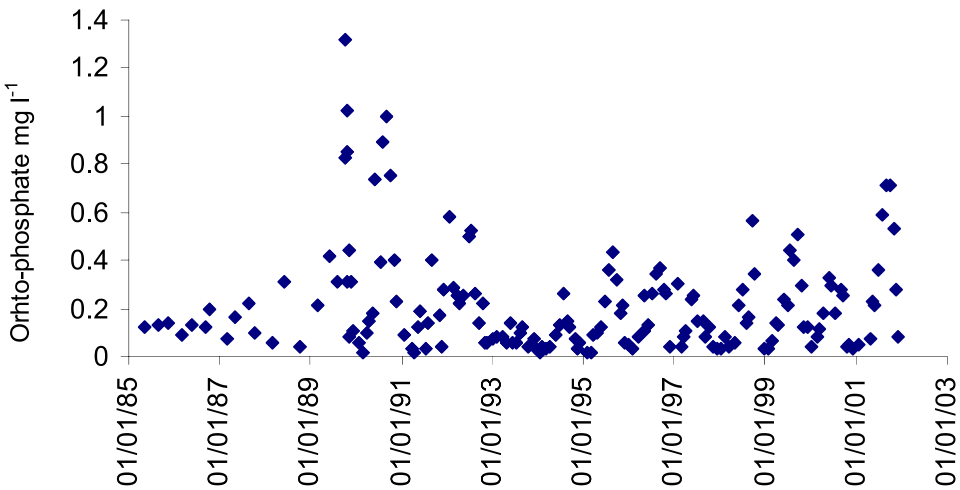

Figure 38. Orthophosphate (mg l^{-1}) in River Kennal at Sticken Bridge (Data source:EA)

Sewage treatment works are also an important source of phosphate; concentrations in waters of the River Kennal at Sticken Bridge (up to 1.32mg l^{-1}) may be influenced by inputs further upstream. Temporal trends for this site 1985 - 2001 (figure 38) show that phosphate levels peak during summer months, coinciding with low rainfall conditions when STW discharges comprise a larger percentage of the river flow. Mean and median values for ortho-phosphate at Carnon STW discharge sites for this period are 0.37 and 0.14mg l^{-1} respectively (max 5.8). Sediments can also be a considerable source of phosphate during summer months, if the sediment becomes oxygen-depleted under low-flow conditions phosphate may become desorbed and diffuse into the water column.

FW sources in Lower Fal (Carrick Roads): Values for ortho-phosphate in freshwater entering Carrick Roads (including STW) are summarised in figure 39. In streams and rivers, phosphate levels are generally low, mean and median (excluding STW) are 0.35 and 0.05 mg l^{-1} respectively, with the highest values occurring up- and down-stream of STW. STW values are considerably higher as expected confirming their potential importance as sources of phosphate to the lower estuary. There are no significant temporal trends in the data.

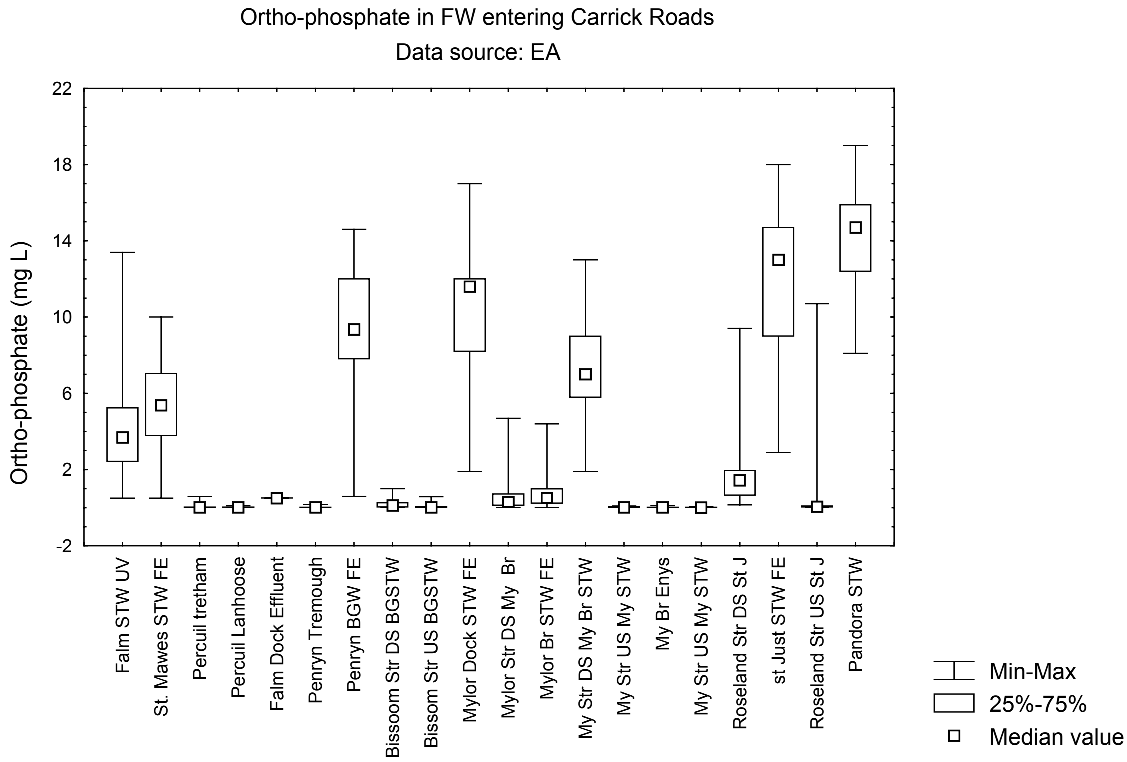

Figure 39. Ortho-phosphate concentrations in waters (FW) entering Carrick Roads (1985-2001) Data source:EA NB Where concentrations are classified as below detection limits (0.01 – 0.5 mg l^{-1} dependant on site/date), detection limits have been applied

Fal Estuary/ Truro River tidal waters: The most recent values (complete year) for phosphate in tidal waters of the upper Fal estuary are summarised in Figure 40. The higher levels in the Truro River area suggest significant phosphate enrichment in these waters. Freshwater inputs are low and discharges from sewage treatment works or fine-screened estuary discharges at Malpas and Truro no doubt contribute to the relatively high load, but unfortunately we have no data for phosphate in STW discharges in this area. In estuaries, phosphate can be adsorbed onto particulates, therefore in turbid waters such as the fresh/seawater interface of the Fal, there tends to be significant removal to sediments, resulting in variable values for phosphate in water over the tidal cycle. Phosphorus thus removed to sediments may be recycled slowly or released more rapidly when these sediments are disturbed, for example during a storm or flood, during dredging, or under low-flow conditions as noted above. Pollution from phosphorus in this area may therefore represent a long-term problem.

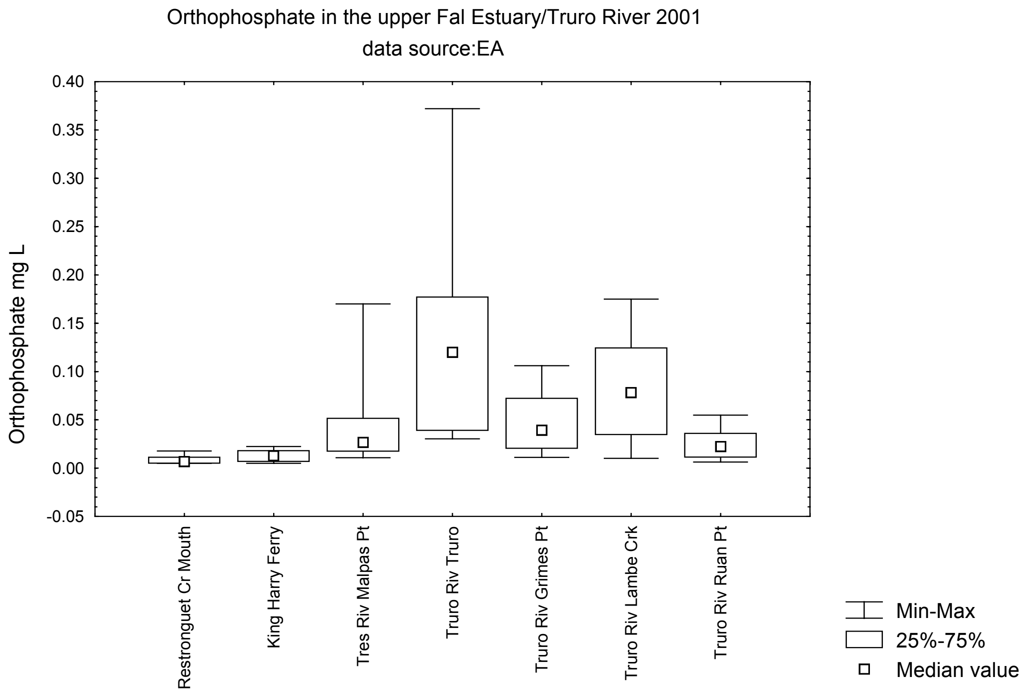

Figure 40. Phosphate concentrations (mg l^{-1}) in Fal tidal waters. upper estuary 2001 . Data source: EA NB Where concentrations are classified as below detection limits (0.005 – 0.025 mg l^{-1} dependant on site/date), detection limits have been applied.

As discussed, there are no statutory UK standards or guidelines for phosphorus species in estuarine and marine SAC's. Mean total reactive phosphate (TRP) (unfiltered orthophosphate) values in English and Welsh coastal waters are reported to range from 7-165 µg P l^{-1} (Parr *et al* 1999). Values in the upper Fal Estuary (re-calculated as elemental phosphorus µg l^{-1}) are generally within this range but invariably above the apparently overcautious 0.1µg l^{-1} criteria set by the EPA (US) to protect estuarine and marine organisms. Mean and median values for the upper Fal are 15.7 and 9.1 µg l^{-1} (as P) respectively (n=919). The 25th percentile (which the USEPA tentatively suggests approximates to a background for the area) is calculated to be 5.8µg l^{-1} and the 75th percentile, 16.3µg l^{-1}. The maximum value of 214.7µg l^{-1} (as P) is for Truro River (at Truro) where P is consistently higher than other monitoring sites (Figure 40). The minimum (0.1µg l^{-1}) is for the mouth of Restronguet Creek. There are no significant long-term temporal trends in this upper section of the Fal, though seasonal (summer) peaks occur as illustrated for the Truro site in figure 41.

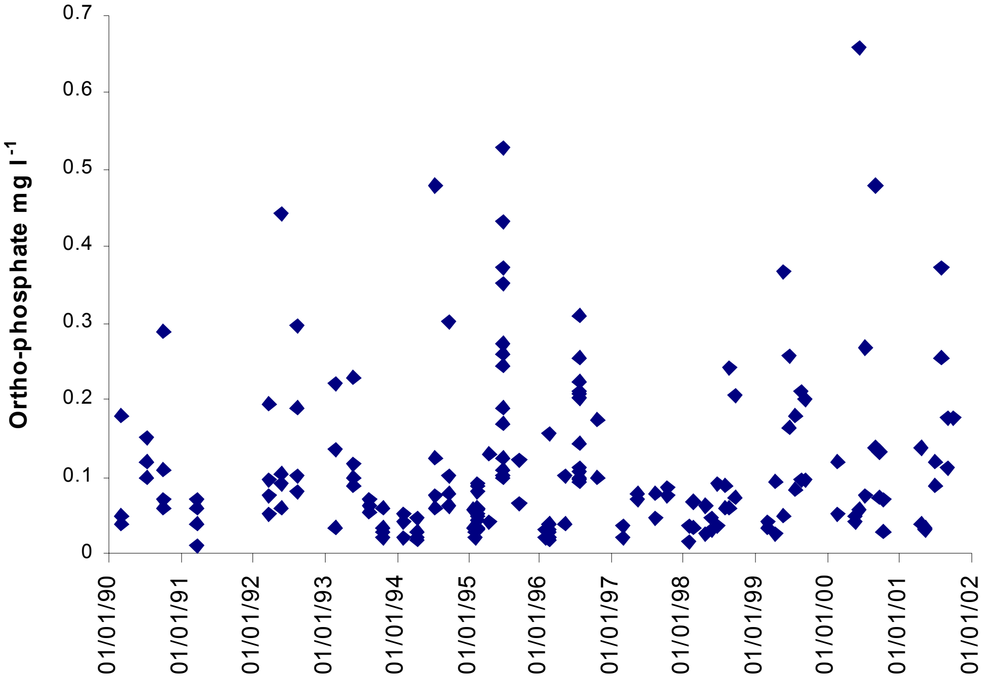

Figure 41. Orthophosphate (mg l^{-1}) in River Truro at Truro (Data source:EA)

The lower Fal Estuary (Carrick Roads) is a more open area with the relatively small freshwater inputs from Penryn, Percuil and Mylor diluted extensively. Although discharges from Falmouth, Penryn, Mylor and St Just STWs may impact locally, phosphate levels in lower estuary waters are generally lower (min 0.002, max 0.06mg l^{-1}) than for the upper reaches of the estuary (Truro area).

Calculated as elemental P, mean and median values are 4.81 and 4.24 μg l^{-1} respectively (n=743). The maximum value of 19.6μg l^{-1} is for the lower Percuil River and the minimum (0.6μg l^{-1}) is for the mouth of Estuary. An approximate background for the area (the 25th percentile) is calculated to be 3.26 μg l^{-1} and the 75th percentile, 6.2. These are invariably above the 0.1 μg l^{-1} criteria set by the EPA (US) to protect estuarine and marine organisms but are generally at the lower end of the range reported by Parr *et al.,*(1999) for coastal waters.

With the exception of one location at the mouth of the Fal there is a significant temporal trend toward reducing phosphate levels in the lower estuary (highly significant for some sites). This trend is exemplified in the Percuil River 1990-2001 (figure 42). Table 22 summarises the regression statistics for these temporal trends, which may be due in part to improvements in sewage discharge treatment.

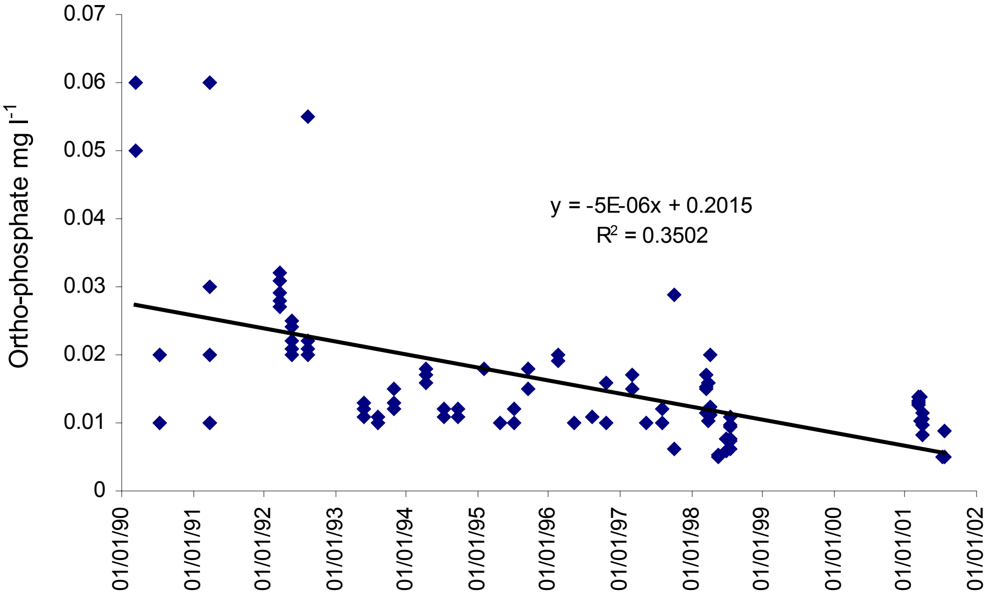

Figure 42. Trends in phosphate concentrations (mg l^{-1}), Lower Percuil River 1990-2001

Table 22. Regression statistics for reductions in phosphate concentrations over time at sites in the lower Fal Estuary.

Site	Period	r	p
Penryn River (at Falmouth Road)	1990-2001	0.47	<0.0001
Percuil River at lower estuary	1990-2001	0.59	<0.0001
River Fal at Black Rock Buoy	1998-2001	0.33	<0.01
River Fal mid-channel	1990-2001	0.39	<0.0001
River Fal at St Mawes	1990-1998	0.5	<0.0001
River Fal at Vilt Buoy	1998-2001	0.23	<0.05
River Fal at Mouth	1987-2001	0.09	0.4

The freshwater catchment for the Helford also lies wholly within a region of intensive mixed arable and dairy farming, with livestock and fertilisers representing a potentially large source of phosphates. Mean annual rainfall is higher than for the Fal catchment and the area is also underlain by impermeable bedrock. Annual run-off is estimated to average 779mm (Fraser *et al.,* 2000). The largest centres of population are Helston, to the east, and Gweek, at the head of the estuary, with small villages and isolated farms scattered throughout the catchment. Freshwater flows are small, especially in the summer months, so that the Helford is largely a saltwater estuary.

Helford FW sources: Values for ortho-phosphate in waters entering the Helford (including STW) are summarised in figure. 43 (n=1224). Values for STW waters are relatively high (max value 16.7 mg l^{-1}) and these discharges comprise a significant source of phosphate. Excluding STW waters, the mean phosphate value at these sites is 0.08 mg l^{-1} (median 0.05 mg l^{-1}). The 25th percentile, a tentative approximation of the background reference value for the area, is 0.03 mg l^{-1}, and the 75th percentile is 0.087 mg l^{-1}. Minimum and maximum values are 0.006 and 3.08 mg l^{-1} respectively. No temporal trends were seen in the data, which covered a period of 16 years.

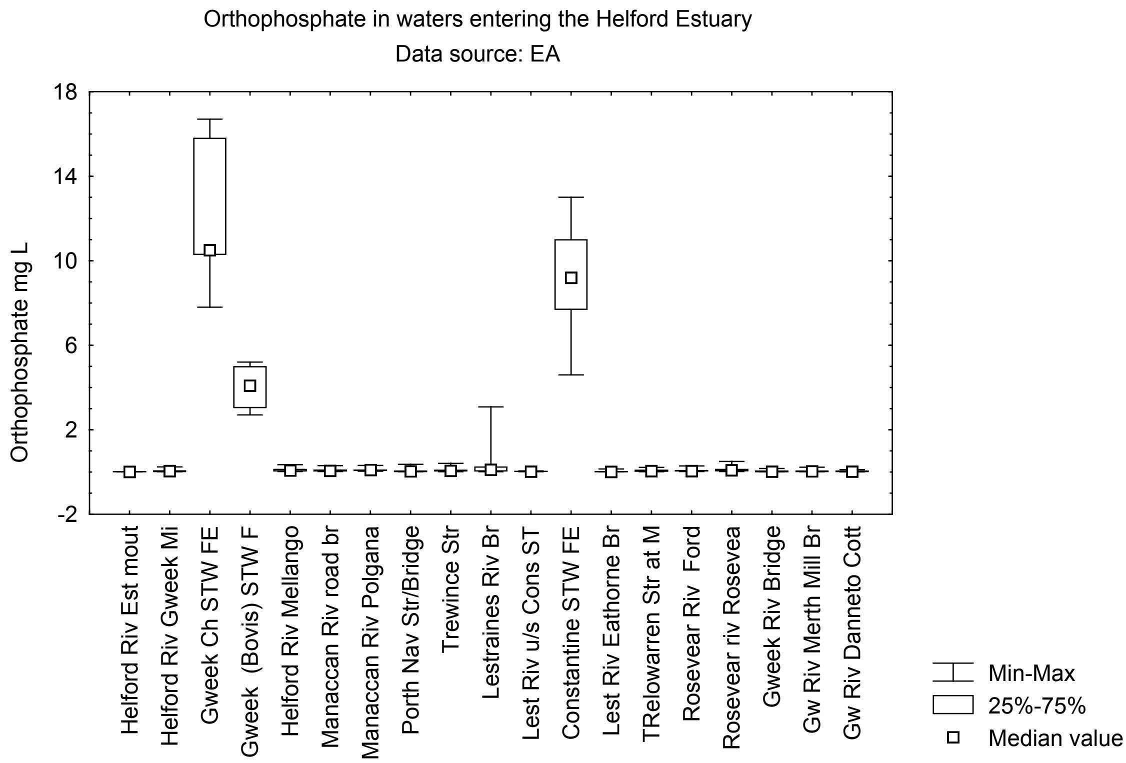

Figure 43. Orthophosphate (mg l^{-1}) in freshwater entering the Helford (1990-1997). Data source: EA NB Where concentrations are classified as below detection limits (0.01 – 0.05 dependant on site/date), detection limits have been applied

Helford Estuary tidal waters: The saltwater reaches of the Helford are highly stratified in both temperature and salinity, and the generally low amounts of suspended particulate matter are concentrated primarily in the fresh/saltwater interface near the head of the estuary and tributaries (Fraser *et al* 2000). Data for phosphate values in saline waters are summarised in figure 44. Mean and median values are 22 and 18 µg PO_4 l^{-1} respectively (n = 611). The highest values for the estuary, at Polwheveral Creek, are probably influenced by discharges from Constantine STW situated to the north of the creek.

Mean and median values in the estuary, re-calculated as elemental phosphorus, (7.13 and 5.87 µg l^{-1}, respectively: n=611) are consistently above the questionable 0.1 µg l^{-1} criteria set by the EPA (US) to protect estuarine and marine organisms. As for the lower Fal, however, they tend to be at the lower end of the range for UK coastal waters (Parr *et al* 1999). The 25th percentile (approximate background for the Helford area) is 3.26 µg l^{-1}, and the 75th percentile 81.6 µg l^{-1}. The maximum value of 95.3 µg l^{-1} is for Polwheveral Creek; the minimum (0.33µg l^{-1}) is for the estuary mouth.

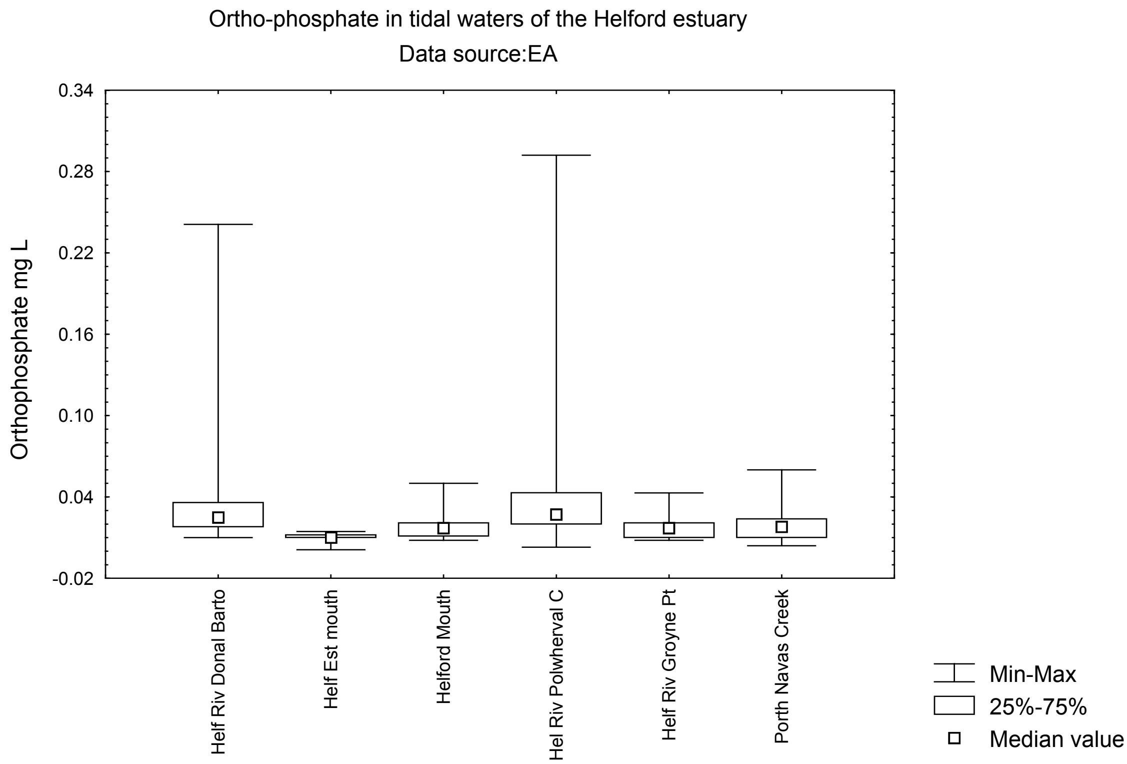

Figure 44. Ortho-phosphate concentrations (mg l^{-1}) in the tidal Helford Estuary. Data source:EA NB Where concentrations are classified as below detection limits (0.001 – 0.01 dependant on site/date), detection limits have been applied

Phosphate levels in the Helford Estuary (Polwheverval) appear to have risen significantly ($r=0.29$, $p<0.05$) between 1990-1998 (figure 45); more recent data are not available. No other significant temporal trends were apparent for tidal Helford waters.

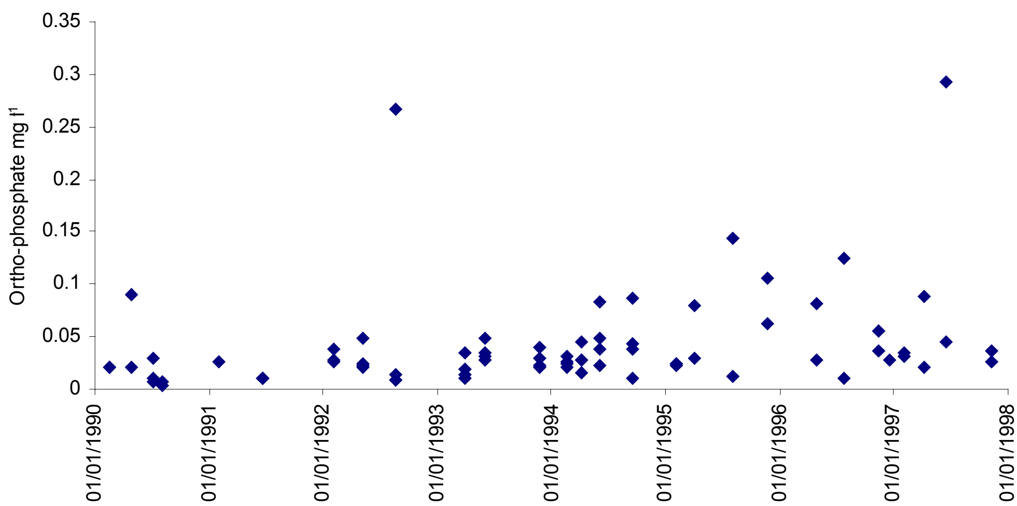

Figure 45. Ortho-phosphate (mg l^{-1}) in Helford Estuary waters (Polwheverval Creek). (Data source:EA)

The Fal and Helford are both rias and as such do not have the large freshwater flows and mixing which are a feature of many large river estuaries. Phosphate levels are high in the tidal waters of both estuaries, compared the criteria set by the EPA (US) to protect estuarine and marine organisms. However it has yet to be demonstrated that this guideline is appropriate.

6.3.3 Nitrate

Carnon Valley sources: The values for nitrate levels at a selection of sites in the Carnon Valley (including the adits, tailings dam and effluent stream at Wheal Jane) are summarized in figure 46. The latter suggest mine drainage is not an important source of nitrate - values are usually lower than in the Carnon River. In streams and rivers throughout the Carnon area mean values (and 25th and 75th percentiles) are invariably below the 50 mg l^{-1} threshold guidelines and usually below the lower threshold of 10mg l^{-1} (EPA). Mean and median values are 2.27 and 2.0 mg l^{-1} respectively (n=1959). The 25th percentile value is calculated to be 1.0 mg l^{-1}, which arguably approximates to a background reference for the area. The 75th percentile was 3.07 and the maximum value 13.1 mg l^{-1}. Few obvious temporal patterns were seen in the data screened, data for Devoran Bridge show an increase 1991-9 although recent years have seen reductions (figure 47).

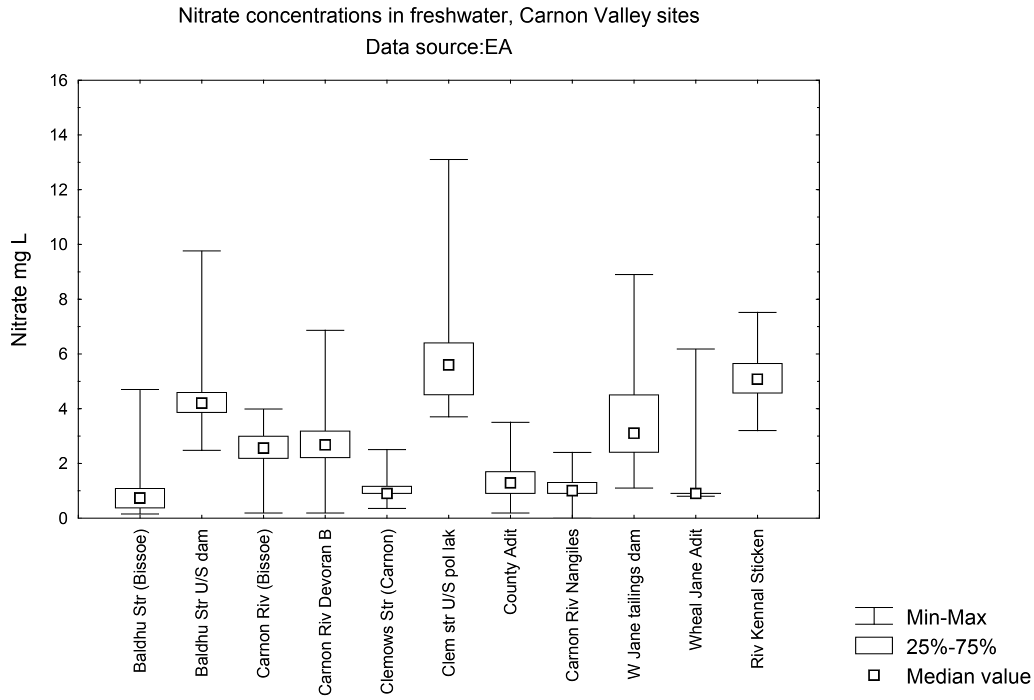

Figure 46. Nitrate concentrations (mg l^{-1}) at sites in the Carnon Valley. (1985-2001). Data source: EA NB Where concentrations are classified as below detection limits, detection limits have been applied

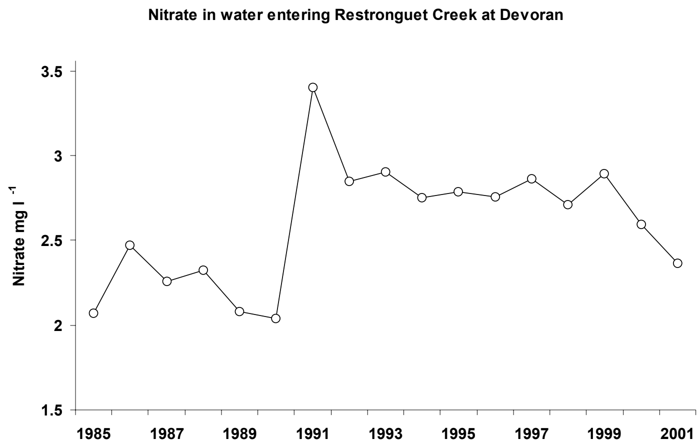

Figure 47. Temporal trends in mean annual nitrate concentrations at Devoran Bridge, Carnon River.

FW sources in Lower Fal (Carrick Roads): The values for inputs into the lower Fal estuary – Carrick Roads (including the Final Effluent streams from several sewage treatment work) are shown in figure 48. Not surprisingly FE values are highest implying they are the most important sources. Even so maximum values are below the 50 mg l^{-1} threshold guidelines for nitrates. In streams and rivers (Mylor, Penryn, St Just, Percuil) mean values (and 25th and 75th percentiles) are generally below the lower threshold of 10mg l^{-1}, only occasionally do values exceed this. Excluding the STW values from the data set the mean and median values are 5.7 and 5.1 mg l^{-1} respectively (n=554). The 25th percentile value is calculated to be 3.89 mg l^{-1}, which arguably approximates to a background reference for the area. The 75th percentile was 7.5 and the maximum value 25.5mg l^{-1}. No obvious temporal patterns were seen in the data screened.

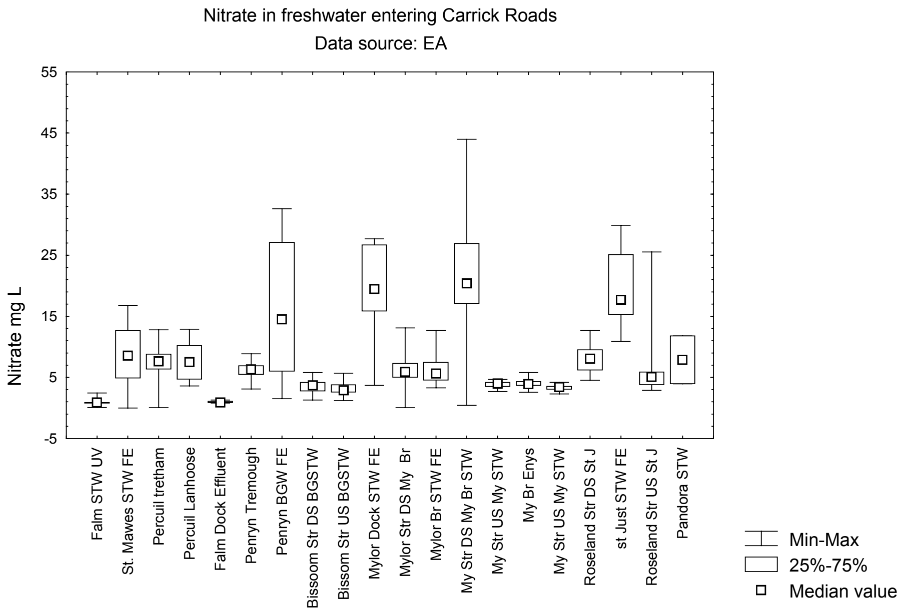

Figure 48. Nitrate concentrations in freshwater entering Carrick Roads.
Data source: EA NB Where concentrations are classified as below detection limits, detection limits have been applied

Fal Estuary/Truro River tidal waters: These sites are represented by dissolved (filtered) N in estuarine and sea water in the EA data base and have therefore been re-calculated here as nitrate values. The most recent data (2000) for the Upper Fal sties are summarised in figure 49. Maximum values (23mg l^{-1}) occur upstream in the Truro River, at Truro, and presumably reflect nearby urban wastewater inputs such as the Newham STW, although nitrate concentrations at all Truro and Tresillian River sites are relatively high (median values 2.11 – 4.9mg l^{-1}).

In the lower Fal Estuary (St Mawes, Penryn, Percuil, Fal Mouth, Black Rock Buoy, Vilt Buoy, Mid Channel) nitrate concentrations appear to have been considerably diluted. Median values for 2000 are in the range 0.43-0.76mg l^{-1}. The maximum concentration recorded during the year was 5.89mg l^{-1} at the point where Carrick Roads joins Falmouth Bay.

When expressed as N, all mean and max values for the Upper Fal in 2000 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), and almost all mean and maximum values exceed the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations. At the Truro site, mean, median, and 25th percentile values (3.9, 4.01 and 3.6mg l^{-1}, respectively) are particularly elevated giving some cause for concern

Of the Lower Fal sites, mean values (as N) for several sites (Carrick Roads/Falmouth Bay, mid channel, St Mawes northern buoy, Penryn, Percuil, Black Rock buoy and Vilt buoy) exceeded the 0.144mg l^{-1} indicator value, but were less than half of the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997). With the exception of Carrick Roads (at Falmouth Bay), all maximum values recorded in 2000 were also less than 1mg l^{-1}.

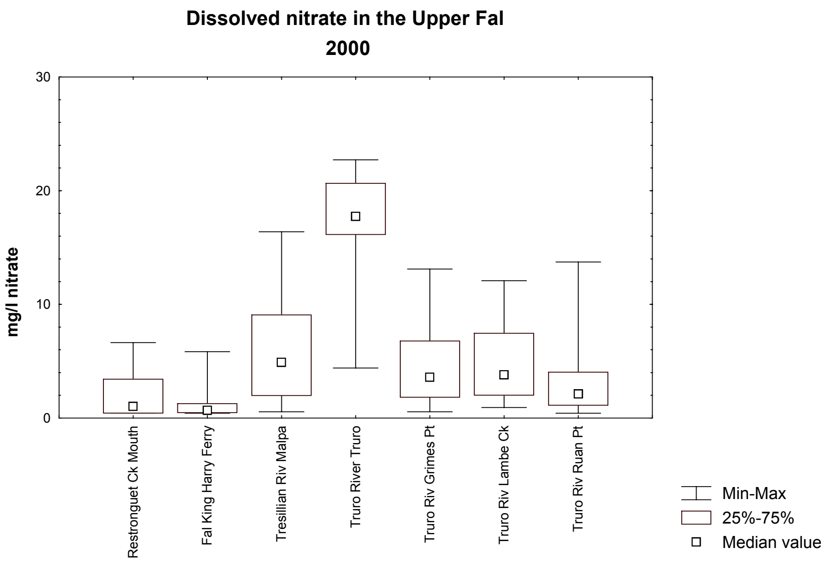

Figure 49. Dissolved N (calculated as mg l^{-1} nitrate) in tidal waters of the Upper Fal Estuary 2000. Data source:EA

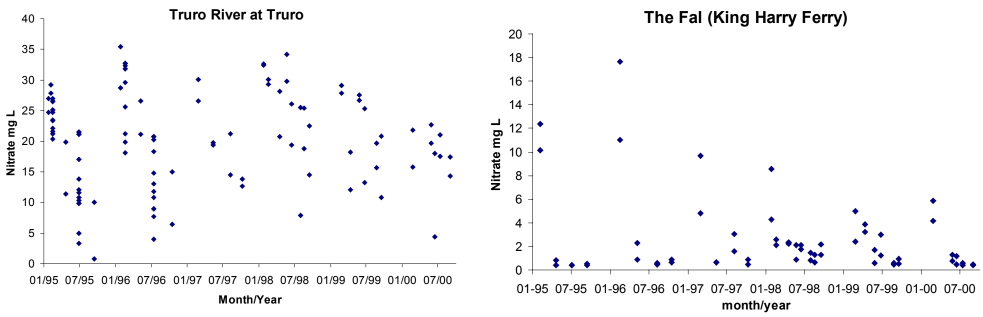

Figure 50. Temporal trends in Dissolved N (calculated as mg l^{-1} nitrate) in the Truro River (at Truro) and Fal Estuary (King Harry Ferry). Data source:EA

At many sites there is evidence of seasonal trends in the nitrate values, as indicated for the King Harry Ferry and Truro sites in Figure 50. These indicate that maximum values occur in mid winter, corresponding to periods of high run-off and riverine flow. There is also a slight indication that peak seasonal values have been declining since the mid 1990's (as has reported incidence of algal blooms). Nevertheless, the highest nitrate levels in the Fal/Helford cSAC occur in the upper Fal, and appear to be linked, spatially and temporally, with the origins and intensity of algal blooms in recent years.

Helford FW sources: The values for nitrate levels at a selection of FW monitoring sites around the Helford (Helford, Manaccan, Rosevear, Gweek, Lestraines rivers and Porth Navas, Trewince and Trewarren Streams) were reasonably consistent and generally below 10mg l^{-1}. Mean and median values are 5.52 and 5.5 mg l^{-1} respectively (n=1195). The 25th percentile value is calculated to be 4.4 mg l^{-1}, which arguably approximates to a background reference for the area. The 75th percentile was 6.7 and the maximum value 11 mg l^{-1}. No obvious temporal patterns were seen in the data. Median FE values at Constantine and Gweek (Bovis) STW were 23 mg l^{-1} and 13mg l^{-1} (EA data over the last 15 years), however there are plans to upgrade the Constantine works in the near future.

Helford Estuary tidal waters: A smaller number of EA data are available for dissolved nitrate concentrations in estuarine waters in the Helford. Data for the most recent year (1997) are summarised in figure 51. Concentrations at Polwheveral Creek and Bonallack Barton are somewhat elevated relative to other sites (medians 25.4 and 19.3mg l^{-1}) and probably reflect their proximity to Constantine and Gweek STWs, respectively.

Expressed as N, mean values for Helford Estuary sites in the (1997) are 0.19 – 5.08 mg l^{-1}, invariably above the TIN value (0.144 mg l^{-1}) considered to represent the threshold for hypernutrification in coastal waters (North Sea Quality Status Report), and at three sites, Bonallack Barton, Polwheveral and Porth Navas, mean values also exceed the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations, (due in part to cloaking effects of macroalgal mats on *Zostera* beds).

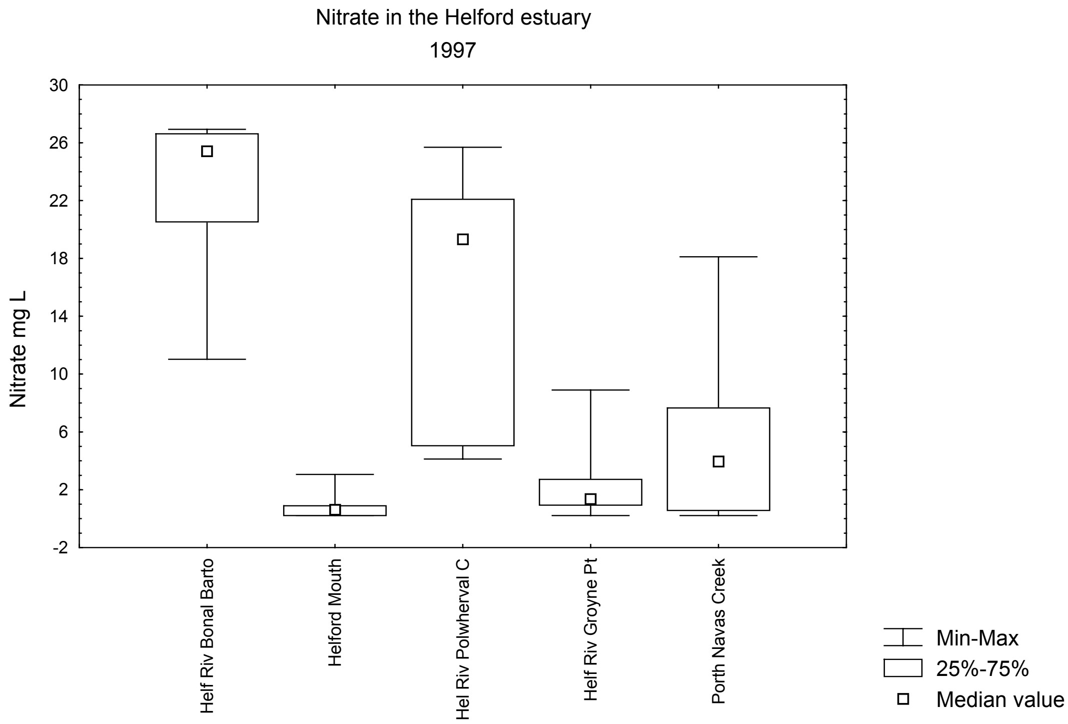

Figure 51. Dissolved N (calculated as mg l^{-1} nitrate) in tidal waters of the Helford Estuary (1997). Data source:EA

6.3.4 Ammonia

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 potential toxicity of ammonia can therefore be a cause for concern in estuarine European marine sites and close to sewage outfalls in coastal waters.

Whereas the effects of nutrient enrichment tend to be indirect, some forms of ammonia can be toxic to marine life. A review of the effects of ammonium on estuarine and marine benthic organisms is given in Nixon *et al* (1995). Toxicity data are presented for shrimps, mysids and lobsters (in which ammonia appears to interfere with the ability of lobsters to adjust to different salinities). Estimated 96-hour LC50s for juvenile school prawns *Metapenaeus macleayi* and leader prawns *Penaeus monodon* are 1.39 and 1.69 mg un-ionised ammonia NH_3 (N) l^{-1} (26.3 and 37.4mg l^{-1} total ammonia (N)) respectively (Allan *et al*., 1990). For the nauplius of the marine copepod *Tisbe battagliai*, Williams and Brown (1992) estimated a 96-hour LC50 of

0.787 mg NH_3 (N) l^{-1} (24.6mg NH_4 (H) l^{-1}), and tests on several life stages showed a No Observed Effect Concentration (NOEC) of 0.106mg NH_3 (N) l^{-1} (3.34mg NH_4 (N) l^{-1}). For invertebrates, toxicity appears to increase as salinity decreases (Miller *et al.,* 1990, Chen and Lin 1991), although more work is needed to establish whether this pattern is typical for all, or most, invertebrates (Nixon *et al.,* 1995). Several studies indicate that ammonia toxicity is greatest to early life stages of invertebrates.

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

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

Ammonia does not accumulate in the sediments, although ammonifying microbial activity in sediments can result in ammonia release. This activity is greatest when large quantities of macroalgal biomass decline (Owens and Stewart, 1983) and is 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).

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

Freshwater (riverine) sources

Data for ammonia (N) in freshwater entering the cSAC during 2001 are summarised in figure 52. Values below detection limits (0.01-0.25mg l^{-1}) have been halved. Annual median concentrations are generally low (below 0.1mg l^{-1}), the exception is for the Carnon River at Devoran Bridge, where values were all relatively elevated (mean, median and max, 0.39,0.38 and 0.47mg l^{-1} respectively). Upstream from Devoran Bridge (Bissoe Bridge), the mean value for ammonia was much lower (0.09mg l^{-1}) indicating a source of ammonia between the two bridges, possibly the (disused) mine workings at Wheal Jane: extremely high levels (up to 19mg l^{-1}) of ammonia have been recorded in waters associated with the mine in the mid 1990s (mine shaft, tailings dam and Clemows Stream), and up to 27mg l^{-1} in the Carnon at Grenna Bridge (between Bissoe and Devoran) although there is no recent data available for ammonia in these waters to establish current levels.

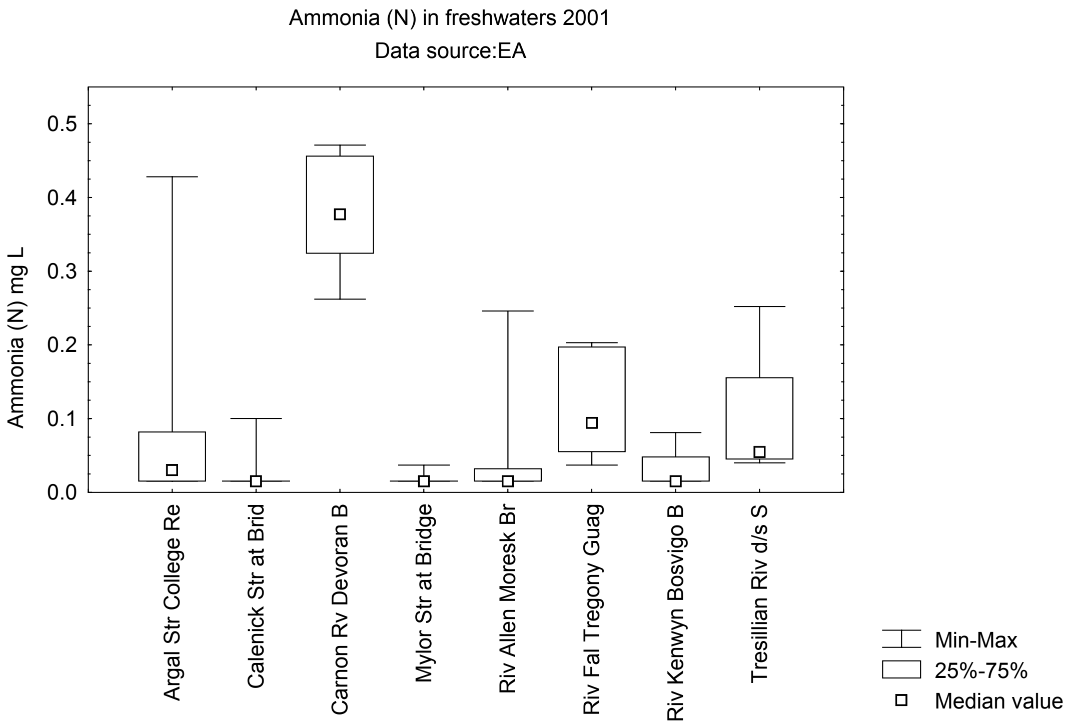

Figure 52. Ammonia (N) in freshwater entering the Fal and Helford cSAC, 2001. Data source EA.

On a temporal scale, there have been reductions in ammonia concentrations over the last decade (1992-2002) in waters of Calenick Stream, the Carnon at Devoran Bridge, Mylor Bridge, and the Rivers Allen and Kenwyn. Similar reductions in Argal Stream, the Fal at Tregony and the Tresillian River (downstream of Ladock STW), have been reversed over recent years and concentrations appear to be increasing in these waters.

For discharges, the highest median ammonia concentrations for 2001 were for effluents from Falmouth, Ladock Valley and Constantine STWs (figure 53). *Note that figures 52 and 53 represent concentrations only and do not necessarily reflect ammonia loadings.* The highest concentration (73.9mg l^{-1}) was recorded in effluent from Mylor Dockyard STW. Ammonia levels in discharges from Falmouth STW were consistently high throughout the year (mean, median and max >/=20mg l^{-1}, 25th percentile >15mg l^{-1}).

There have been reductions in ammonia in discharges from several of the STWs 1992-2002, notably, Mylor Bridge and St Mawes STWs, where dramatic reductions are apparent in the early 1990s, possibly due to sewage treatment improvements. There have been significant reductions in ammonia levels in the discharge from Truro (Newham) STW since 1998, due to the introduction of denitrification processes. A more gradual decrease in ammonia concentration has occurred in effluent from Falmouth since it's commissioning in 1998.

Ammonia levels in effluent from Constantine STW, which discharges to Polwheveral Creek in the inner Helford Estuary, have remained relatively unchanged, whilst there have been recent increases in discharges from Tregony, which discharges to the upper Fal. Figure 54 shows contrasting temporal trends for ammonia in discharges from Tregony and Truro (Newham) STWs.

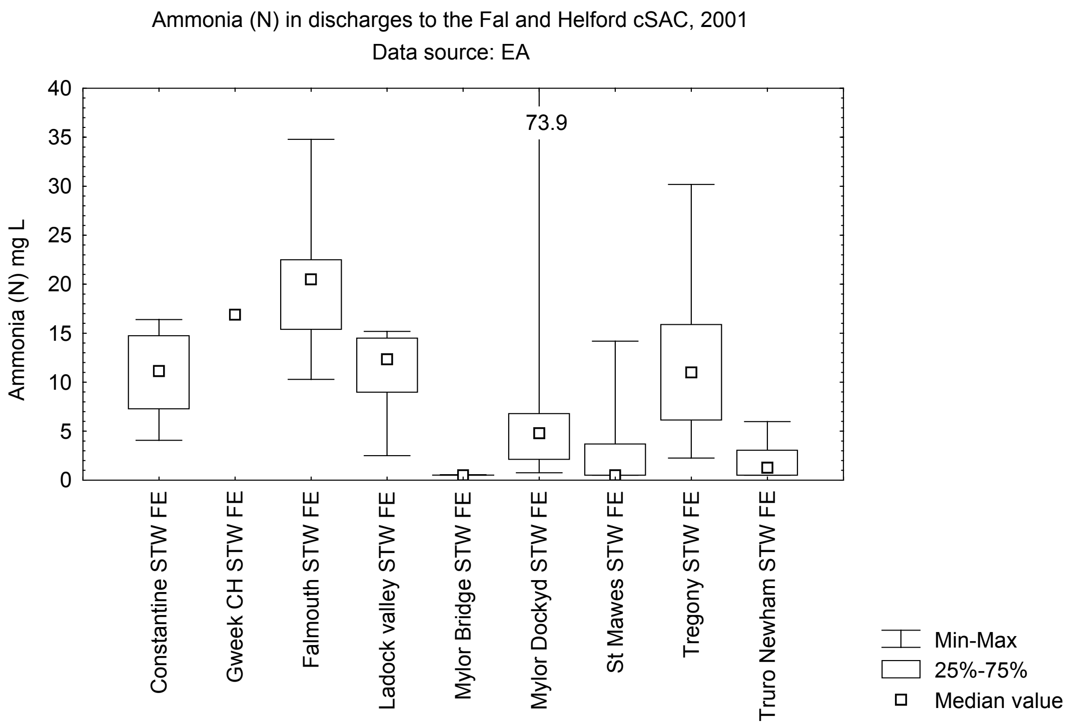

Figure 53. Ammonia (N) in discharges to the Fal and Helford cSAC 2001. Data source EA. NB One sample value only for Gweek CH STW (non water company).

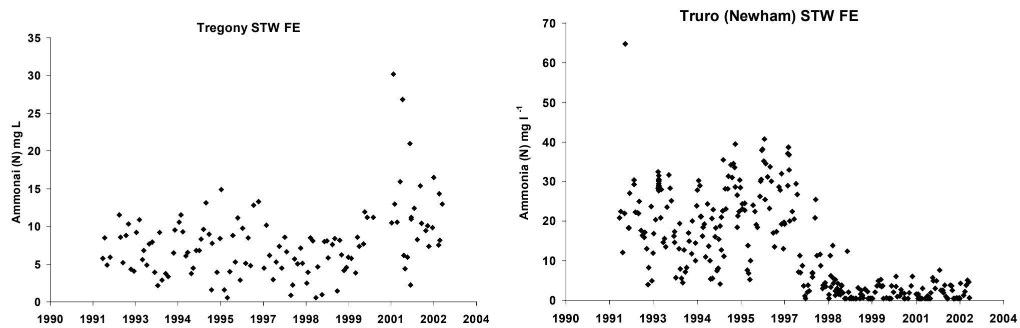

Figure 54. Temporal trends for ammonia (N) concentration in effluent discharges from Tregony and Truro (Newham) STWs. Data source EA. .

Ammonia standards for discharges appear to be set on a case-by-case basis, and are generally more stringent for discharges to sensitive waters (e.g 10mg l^{-1} as 95th percentile).

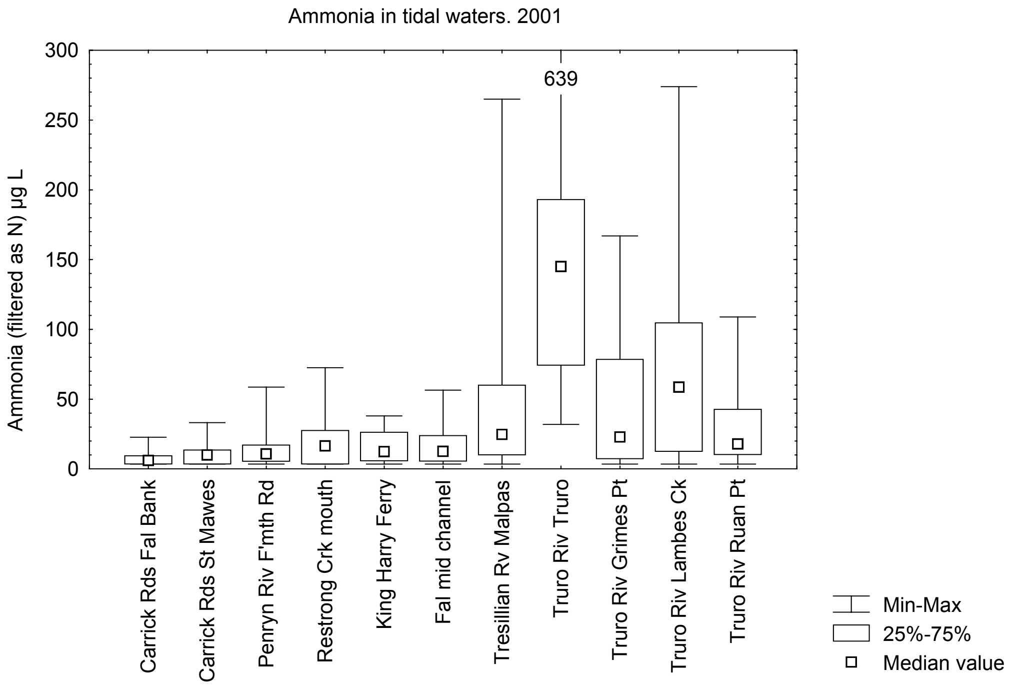

Figure 55. Ammonia (filtered as N) in tidal waters of the Fal and Carrick Roads. Data source EA

Tidal waters. Data for ammonia in tidal waters of the Fal and Carrick Roads (2001) are summarised in figure 55. Median values are in the range 6 to 145µg l^{-1}. The waters of the Truro River stand out as having the highest concentrations overall. In the vicinity of the STW at Truro, mean, median and maximum concentrations were 179, 145 and 639µg l^{-1}, respectively, and levels were elevated around Malpas (Grimes Point and Lambe Creek on the Truro River, and on the Tresillian River. In the lower estuary, highest concentrations occur at the mouth of Restronguet Creek, the Penryn River at Falmouth Roads and mid channel.

Although concentrations are still relatively high, there has been a significant reduction in ammonia levels in the Truro River at Truro (figure 56), probably directly attributable to the introduction of denitrification processes introduced at Truro (Newham) STW in the mid 1990s. There has also been a gradual reduction over the last decade at Lambe Creek in the upper Fal. However, this effect appears to be fairly localised as there have been general increases in ammonia concentrations in the upper Fal/Tresillian River at Malpas and Ruan Point.

In the lower Fal, the trend is also toward increasing concentration, notably at Falmouth Bank and St Mawes Bank. For other sites (Restronguet Creek mouth, Penryn River, King Harry Ferry and mid-channel, ammonia levels were decreasing until the late 1990s, but since when then increases are apparent. This trend is exemplified in figure 56 (Penryn River). These increases may be due to an increase in ammonia concentrations in STW effluent discharges (e.g Tregony) although analysis of ammonia loadings would need to be carried out to confirm this.

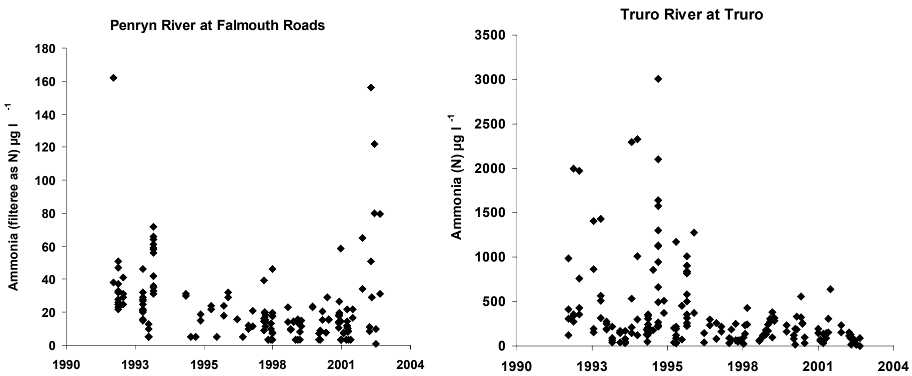

Figure 56. Temporal trends for ammonia (filtered as N) in tidal waters of the Fal and Carrick Roads. Data source EA**.** .

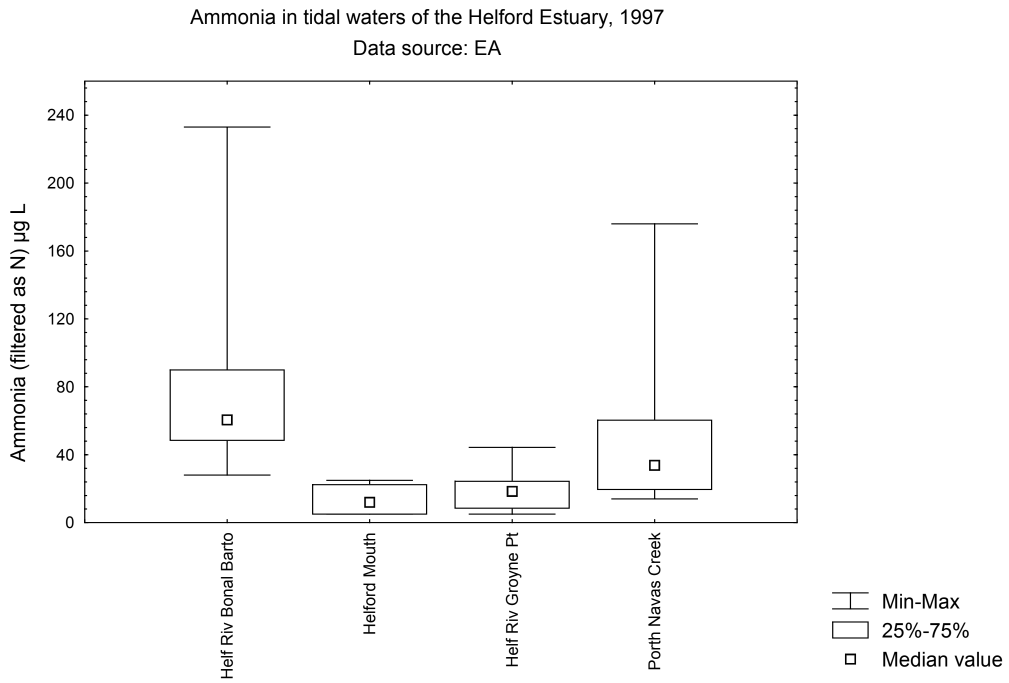

Figure 57. Ammonia (filtered as N) in tidal waters of the Helford Estuary (1997). Data source:EA

Data for ammonia in waters of the Helford Estuary are for the period 1992-1997. The most recent (complete year) data (1997) are summarised in figure 57. Median values are in the range 12 to 60µg l^{-1}, with highest concentrations occurring in the more enclosed regions of the Estuary (Bonallack Barton and Porth Navas Creek), which possibly reflects the close proximity of STWs (Constantine and Gweek).

On a temporal scale, there have been gradual reductions in ammonia concentrations in tidal waters of the Helford Estuary (1992-7); the exception is at Bonallack Barton, where levels remained relatively unchanged.

Note that the ammonia data are for ammonia as N, and values for the more toxic unionised ammonia, NH_3 (N), would need to be calculated from the total data, taking account of pH, temperature, and salinity. As a rough guide; for a pH of 8.2, a temperature of 20˚C, and a salinity of about 30, 0.44 mg l^{-1} total ammonia (N) relates to about 0.021mg l^{-1} NH_3 (N), which is the proposed EQS.

Thus, ammonia levels are elevated in the more enclosed inner estuarine areas of the cSAC, notably, in the Truro River, where DO sags are likely to coincide with high summer temperatures, conditions which increase the likelihood of ammonia toxicity. However, it is difficult to establish whether the 21µg l^{-1} EQS for NH_3 (N) is exceeded, and as monitoring of the Helford Estuary tidal waters ceased in 1997, further increases in ammonia concentrations from discharges should be avoided, as a precautionary requirement.

Conclusion on Nutrient status

Temporal trends for nitrogen and phosphorus in the cSAC indicate alternate seasonality; waters are high in phosphate during the summer months and lower in winter, whilst the opposite is true for nitrate. Considering that uptake of nutrients by phytoplankton populations peaks during the summer, and coincides largely with periods of low rainfall when inputs of nitrogen and phosphorus in runoff from the land are at a minimum, it could be expected that levels of both nutrients should be lowest in summer. The seasonal pattern for phosphate therefore indicates major anthropogenic enrichment, particularly in the summer. The relative importance of inputs, diffuse versus point sources, appears to be site-dependant in the cSAC, with enclosed areas such as the upper estuary more vulnerable to the effects of waste discharges. An appraisal of current nutrient source modelling in the Fal can be found in Fraser *et al.,* (2000), although there are limitations to these type of models (discussed above).

Whilst no statutory standards for nutrients exist, the weight of evidence suggests nutrients are an issue of importance in the upper part of the Fal system and seem to point to urban waste at Truro as an important source of additional load (see also Reid and Pratt, 1995). The issue already appears to have been taken on board by SWW as there are plans in the near future to upgrade the STW at Newham. In view of potential threats from algal blooms however continued surveillance is important.

In the Helford Estuary, monitoring of nutrients in tidal waters appears to have ceased in the late 1990's, when nutrient values were high enough to be of concern, particularly nitrate, which even exceeded levels in waters of the upper Fal. It is recommended that regular monitoring of the situation be resumed.

In estuarine and marine ecosystems there is a complex interaction of nutrients between sediment and overlying water, which in the case of N, for example involves a range of processes including nitrification, denitrification, mineralisation, assimilation and fixation which may all vary spatially and temporally. Phosphorous transformations in tidal waters are usually dominated by the behaviour, amounts and fluxes of orthophosphate, which may for example be released from sediments under conditions of low O_2. Thus as SOD and BOD levels vary across the seasons, perhaps exacerbated by stratification in the Fal, sediments may oscillate between being sources and sinks for PO_4. Suspended solids loadings and adsorption/desorption characteristics (e.g. Kp values) will also determine P (and N) concentrations and the ratio of dissolved to particulate forms.

The complexity of the nitrogen and phosphorous 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 any attempt at evaluating the significance of sediment as sources or sinks of N and P is difficult. In order to construct more meaningful budgets the needs are to determine N and P removal rates to sediment, estuarine mixing behaviour, and to look at export rates from the estuary on suspended particles, at different salinities, tidal states, flow rates and seasons.

In terms of ecological impacts it is difficult to establish, precisely, the effects of nutrients. Many of the gaps in our knowledge are discussed at length by Parr *et al* (1999). There is no published evidence to indicate that estuarine benthos are directly affected by nutrient enrichment. However since primary productivity is linked to nutrient status (and appears to be elevated in parts of the Fal, periodically), secondary productivity of the benthos will also probably be linked, indirectly. The consequences could be beneficial (in terms of knock-on effects such as increased food items for tertiary consumers, e.g. birds and fish) as well as negative (e.g increased algal smothering of seagrass beds and intertidal flats.) Where the increase in primary productivity involves harmful microalgal blooms the consequences have certainly been highly detrimental in the Fal in the recent past (e.g. concentration of PSP transmitting toxins in shellfish).

Effects on many of the rarer species in the cSAC are largely unresearched, but in view of their conservation importance, it would seem that an increase in nutrients should be strongly avoided, as a precautionary requirement. Changes to consents (quantities and location) should therefore be considered very carefully to avoid the risk of hypernutrification.

In view of the sensitivity/vulnerability of the Fal & Helford cSAC to nutrients it would seem that surveillance for evidence of impact should be made a reasonably high priority. Potential diagnostic features of this could include:

- Direct toxicity of ammonia to invertebrates and fish
- Stimulation of phytoplankton growth and fluctuations in dissolved oxygen (potential for toxicity and reduced biodiversity in invertebrates and fish). Likewise from high BOD as a result of algal decomposition after blooms
- Stimulation of macroalgal growth particularly *Enteromorpha* and *Ulva* spp. on subtidal and intertidal substrata (reduced feeding areas for some fish and birds: Reduction in oxygen availability in sediments under intertidal algal mats)
- Contribution to increased turbidity in the water column (reduced photosynthesis for macroalgae and seagrasses in the photic zone).
- Toxicity of dinoflagellate blooms ('red tides'). Closure of shellfisheries through the impacts of anoxia or toxic algae.

- Potential damage to maerl beds due to heavy overgrowth of epiphytic algae, reduction in light, increased turbidity.
- Aesthetic impacts (increased turbidity, discolouration, odours, slimes and foam)

For the current project we have briefly considered available EA data on DO, turbidity and primary productivity as potential indicators of impact in different sections of the cSAC.

6.3.5 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 60% saturation 80% saturation	50%ile, mandatory standard Minimum, mandatory standard 95%ile, guideline value	EC Shellfish Water Directive
Saltwater life	5 mg l^{-1} 2 mg l^{-1}	50%ile 95%ile	
Sensitive saltwater life (e.g. fish nursery grounds)	9 mg l^{-1} 5 mg l^{-1}	50%ile 95%ile	
Migratory fish	5 mg l^{-1}3 mg l^{-1}	50%ile95%ile	Higher values may be required where fish have to traverse distances >10 km, or where high quality migratory fisheries are to be maintained

Various class thresholds for estuaries in England and Wales, based on DO over a continuous period of >1 hour were proposed by Nixon *et a.,l* (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 cause effects if organisms are exposed for a continuous period of greater than one hour.

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. During bloom die-offs, microbial decomposition of algal cells leads to an increase in oxygen demand and acute DO depletion, which again can result in lethal and sub-lethal effects to fish and invertebrate communities. Figure 58(a) illustrates the variation in DO, both seasonal and diurnal, in waters of the Truro River at Truro.

The Fal and Helford cSAC is particularly vulnerable to such problems, as these changes in water quality are likely to be greatest in semi-enclosed bodies of water with long retention times, and where stratification of the water column occurs (Cole *et al.,* 1999). Reduced DO levels could be anticipated in some areas of the cSAC such as the upper estuary, where nutrient levels are shown to be increasing.

Data for DO (expressed as mg l^{-1}) in the tidal regions of Fal and Helford encompasses a period spanning 15 years for some sites, and taken as a whole, values do not fall below recommended EQS's. However, values for individual sites do indicate that there may be a problem; therefore it is appropriate to examine the data more closely.

Data for DO in the Fal and Helford estuaries are summarised in table 25. During the period 1985-1998, 11% of all DO measurements in the Helford Estuary fell below 9 mg l^{-1} (n=1001). With the exception of waters at the Estuary mouth, where 51% of measurements were low (n=131), DO in Helford waters does not appear to be particularly problematic.

The value of 9 mg l^{-1} DO is the recommended 50th percentile EQS for sensitive marine life, and in Fal Estuary as a whole, 5 individual sites fell below this criteria in 2000. Between 1990-2001, 25% of all measurements for DO in the upper Fal Estuary (Truro area) fall below 9 mg l^{-1}, and more recently (1999-2001) 35%. Indications are that the DO status of the upper estuary is deteriorating and that levels in parts of the lower Fal (Carrick Roads) may be a cause for concern. This situation appears to support findings reported above, namely that there is a nutrient enrichment problem in the upper Fal.

Table 25. DO in Fal and Helford cSAC estuaries. Data source EA.

Site	Period	No. of samples	DO <9mg l^{-1} %	No. of samples 1999-2001	DO <9mg l^{-1} %	50^{th} percentile in 2000 (mg l^{-1})
HELFORD ESTUARY						
Bonallack Barton	1990-1998	79	13	-	-	
Estuary mouth	1985-1998	131	51	-	-	
Helford River Mouth	1990-1997	140	2	-	-	
Polwheveral Creek	1990-1997	74	5	-	-	
Groyne Point	1990-1997	292	3	-	-	
Porth Navas Creek	1990-1997	85	12	-	-	
All Helford sites	**1985-1998**	**1001**	**11**	-	-	
UPPER FAL ESTUARY						
Restronguet Creek Mouth	1993-2001	87	15	35	20	
King Harry Ferry	1992-2001	448	14	35	40	
Tresillian River Malpas Pt	1998-2001	51	27	35	40	8.8
Truro River Old Kea	1990-2001	366	33	26	38	
Truro River at Truro	1990-2001	192	32	38	32	
Truro River Grimes Pt	1998-2001	61	31	35	37	8.7
Truro River Lambe Creek	1998-2001	51	29	35	43	8.6
Truro River Ruan Pt	1998-2001	58	31	35	34	
All upper Fal sites	**1990-2001**	**1314**	**25**	**274**	**35**	
Lower Fal Estuary						
River Fal at Mouth	1985-2001	196	36	35	31	8.7
Penryn River: Falmouth Road	1990-2001	226	16	36	-	
Percuil River lower estuary	1990-1997	165	4	-	-	
Fal: Black Rock Buoy	1999-2001	35	37	35	-	
Fal: mid-channel	1990-2001	240	25	33	36	8.8
Fal: St Mawes	1990-1997	585	10	66	-	
Fal: Vilt Buoy	1999-2001	37	32	37	-	
All sites	**1990-2001**	**1484**	**17**	**33**	**10**	

With regard to the proposed GQA, 4% of DO measurements for the Helford, and 10% for the whole of the Fal Estuary (n=1314) fell below the threshold GQA class A/B (8mg l^{-1}), a level at which effects are considered likely if organisms are exposed for continuous periods of greater than 1 hour. Indications are that this situation is more common in the upper Fal Estuary where levels of less than 8mg l^{-1} may persist for up to 6 hours (Figure 58b).

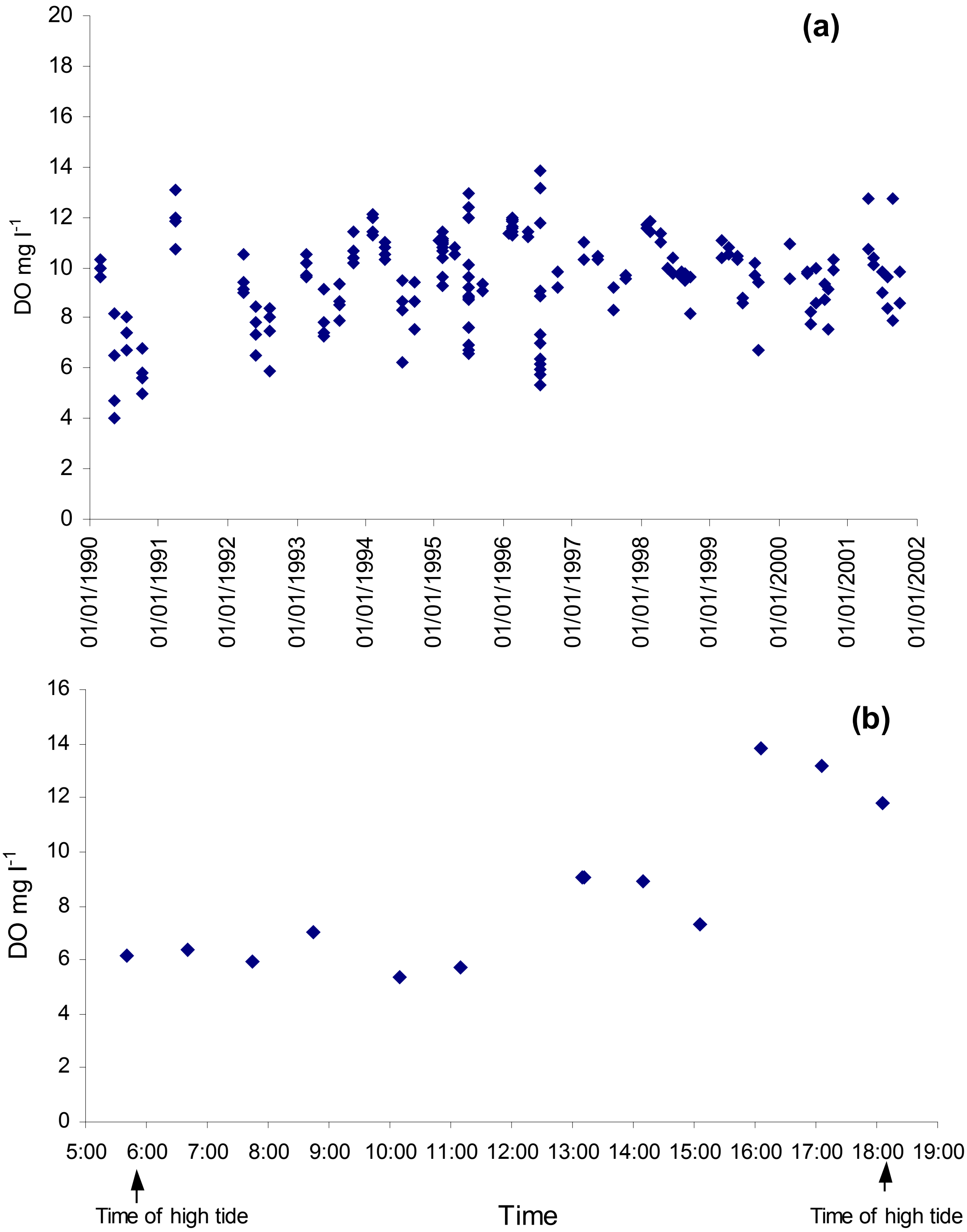

Figure 58. (a) Dissolved oxygen (mg l^{-1}) in the waters of the Truro River at Truro 1990-2001. (b) DO over tidal cycle 15/07/1996

6.3.6 Chlorophyll *a*

Elevated spring and summer levels of chlorophyll *a* are one of the primary symptoms of increased nutrient inputs to estuarine waters and as such are another response variable measurement. Chlorophyll *a* is the molecule mediating photosynthesis in almost all green plants including phytoplankton. Rapid proliferation or blooms of phytoplankton, as reflected in elevated chlorophyll *a* levels, can occur throughout the ocean but are typically associated with temperate coastal and estuarine waters such as the Fal and Helford cSAC. During winter months growth of phytoplankton populations are at a minimum because of reduced temperature, light availability, and water column stability, and chlorophyll-*a* levels generally remain low. Monitoring of chlorophyll *a* is more often restricted to spring and summer months when estuarine concentrations in optimum growing conditions may exceed 50-80μg l^{-1} (Monbet 1992).

In the UK, an indicator mean value for suspected eutrophic conditions is set at 10μg l^{-1} chlorophyll *a* (Dong *et al.,* 2000). Data for mean annual chlorophyll *a* concentrations in the Fal and Helford cSAC are summarised in figure 59. Values for most sites are for the period April-September, although measurements taken February-October are included at some sites.

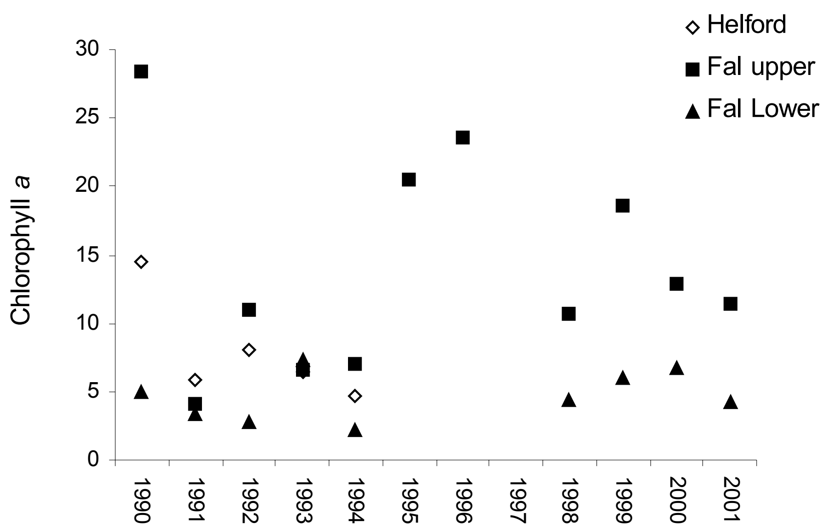

Figure 59. Mean annual chlorophyll *a* (μg l^{-1}) in the Fal and Helford cSAC. Data source EA. NB Where values are below the limit of detection (1μg l^{-1}) detection limits have been applied.

Mean annual chlorophyll *a* values for the Helford (1990-1994) are generally low and at no time are they indicative of significant plankton blooms. Similarly, values for the lower Fal Estuary (1990-2001) are consistently below the UK indicator value of 10μg l^{-1}. However in the upper Fal Estuary, with the exception of years 1991,1992 and 1994 (no data for 1997) annual mean values are all above this figure.

Considering data for the upper Fal only (1990-1996, 1998-2001), highest levels of chlorophyll *a* occur in the uppermost reaches of the estuary (Truro River at Truro, Malpas Point, Grimes Point and Lambe Creek) where mean values are up to 20μg l^{-1} (figure 60) and individual values of 219 and 240μg l^{-1} have been measured during summer months of 1999. Such elevated values indicate significant algal blooms, and even exceed levels recorded during the 1995 'red tide' event in the area. There is no information available at present to indicate which phytoplankton species are contributing to these blooms, but such levels are probably a direct result of the nutrient enrichment observed in this area and are likely to have 'knock on' consequences for estuarine biota whether or not they are toxic species (discussed elsewhere). These include fluctuations in dissolved oxygen (with the potential for sublethal and lethal effects on invertebrates and fish), increased turbidity and reduction in light levels (see relevant sections in this report).

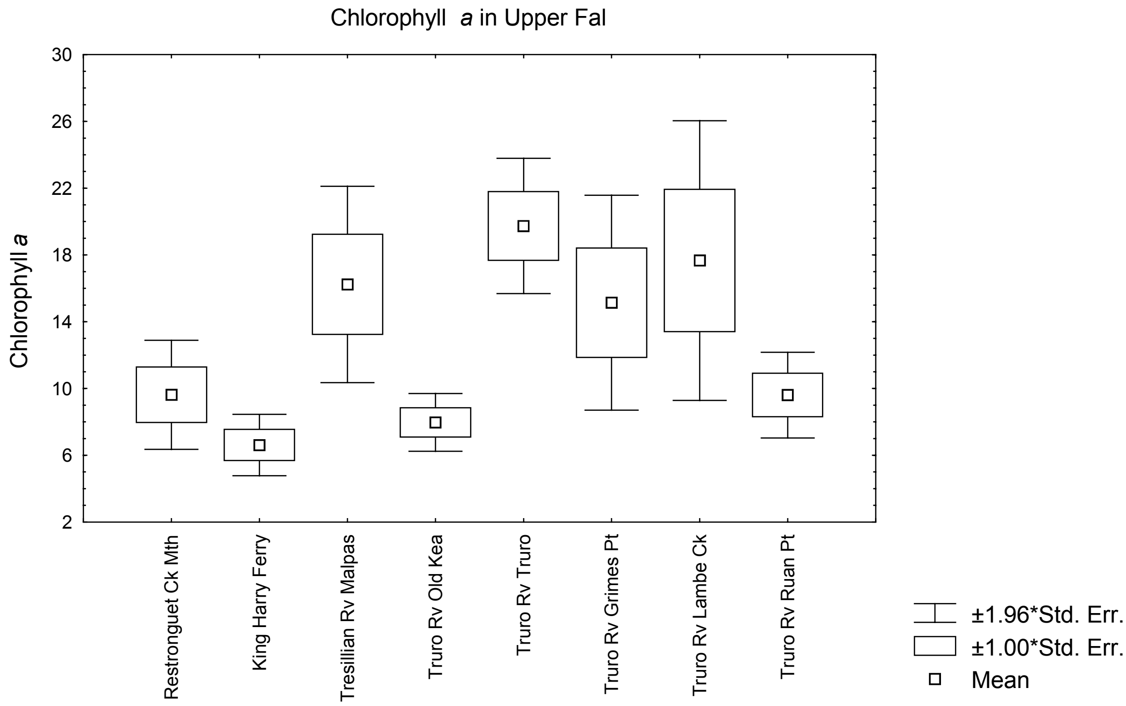

Figure 60. Mean chlorophyll *a* levels (µg l^{-1}) in waters of the upper Fal Estuary 1990-1996, 1998-2001. Data source: EA NB Where values are below the limit of detection (1µg l^{-1}) detection limits have been applied.

6.3.7 Turbidity

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

The composition of particulate matter varies but is derived from: directly eroded material, sediments that have settled to the substratum and become resuspended during periods of high flow, dredging, suspended solids in discharges, chemical flocculation (at the salt/freshwater interface) and plankton.

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

These values are only applicable during the bathing season and may be waived in the event of 'exceptional weather or geographical conditions'.

Two principal methods are used by the EA for quantifying turbidity in the Fal and Helford cSAC, suspended solids (at 105^{o}C) (units: mg l^{-1}), and light scattering, measured using a turbidimeter calibrated with Formazin (units: Formazin Turbidity Units, FTU). Turbidity data (for the tidal waters) derived using both methods are highly comparable and presents a similar overall pattern; therefore FTU units have been used for the purposes of this report.

For the Fal estuary, levels are consistently higher in the upper reaches, reaching a peak in the Truro River at Truro (Figure 61), where the mean value for all measurements between 1990-2001 is 81 FTU (n=194). It should be stressed, however, that this sampling location is in the STW outfall 'boil' and would not be typical of undisturbed waters. Mean values for other sampling points in the upper estuary range from 4 - 12 FTU, whilst for the lower estuary, mean values are in the range 1 – 3 FTU. Statistical analyses reveal no indication of change over time at the uppermost sampling points (Truro area). However there have been significant reductions in the turbidity of waters near King Harry Ferry ($p<0.01$) and Restronguet Creek (mouth) ($p<0.001$), which presumably contribute to a highly significant reduction in turbidity, over time, found for the lower estuary as a whole ($p<0.0001$).

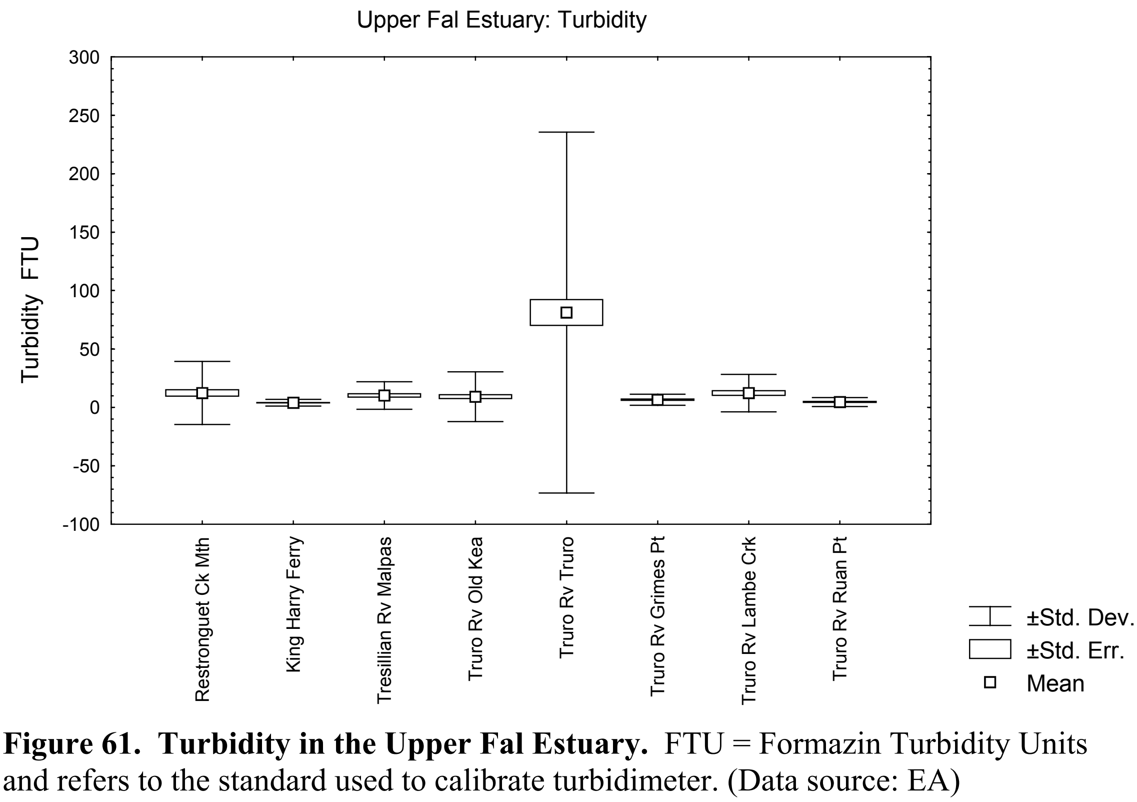

Figure 61. Turbidity in the Upper Fal Estuary. FTU = Formazin Turbidity Units and refers to the standard used to calibrate turbidimeter. (Data source: EA)

In the Helford Estuary (1990-1997), sampling point mean values are in the range 4 – 20 FTU, with turbidity highest at Bonallack Barton close to the tidal limit. Results of regression analyses showed significant reductions in turbidity over the period, for waters at Groyne Point ($p<0.001$), Polwheveral Creek ($p<0.05$) and the estuary mouth ($p<0.01$).

To put turbidity levels into some perspective, Cole *et al* (1998) quoted typical annual mean suspended solids (105^{o}C) around the English and Welsh coast of 1-110mg l^{-1}, and suggested that anything >100 mg l^{-1} could be considered high. Annual mean suspended solids (105^{o}C) in the water column at Truro range from 70-1584 mg l^{-1}.

The primary source of turbidity is often quoted as being sediment resuspension (Parr *et al.,* 1998). Peak levels are confined to a discrete area generally in the mid-upper reaches of the system, which moves up and down with the tide (Cole *et al*., 1999). For both the Fal and Helford estuaries, peak turbidity levels (the turbidity maximum) are in the upper reaches where there are significant silt deposits. However, the levels of turbidity in the upper Fal suggest additional sources: suspended solids in discharges, chemical flocculation and plankton may contribute to this load.

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 to higher trophic levels.

6.3.8 Microbiological Parameters

EQS for microbiological parameters are defined by standard values set by three EC Directives;

- Quality required for Shellfish Waters 91/923/EC – sets classification standards for harvesting areas in addition to guidelines for faecal coliforms in shellfish waters and flesh (Annex 3).

- Quality required for Shellfish Hygiene 91/492/EC – set conditions for the production and placing on the market of live bivalve molluscs (for classification criteria see table 26).

- Quality required for Bathing Waters 76/160/EC (currently under revision) – sets guideline and imperative standards for coliforms, faecal streptococci, salmonella and enterovirus in bathing waters (microbiological parameters summarized in table 28, for full list of standards see Annex 4).

The standards in the Shellfish Waters Directive (Annex 3) are applicable only in designated shellfish waters and are designed to protect shellfish populations which are harvested for human consumption. EQS for waters under this directive is 300 faecal coliforms per 100ml as 75th percentile value (in intervalvular fluid and waters from which shellfish are taken for direct human consumption).

Table 26. Classification of shellfish harvesting areas under the requirements of Shellfish Hygiene Directive 91/492/EC

Category A	Less than 230 *E. coli*/100 g flesh, or Less than 300 faecal coliforms/ 100 g flesh	May go direct for human consumption if end product standard met
Category B	Less than 4,600 *E. coli*/100 g flesh (in 90% of samples), or Less than 6000 faecal coliforms/ 100 g flesh (in 90% samples)	Must be depurated, heat treated or relaid to meet Category A requirements
Category C	Less than 46,000 faecal coliforms/100 g flesh	Must be relaid for long period (at least two months) whether or not combined with purification, or after intensive purification to meet Category A or B.
	Above 60,000 faecal coliforms	Unsuitable for production

Classification of shellfish harvesting areas in England and Wales is compiled by the Centre for Environment, Fisheries and Aquaculture Science, Weymouth (CEFAS) and has recently been updated, the following applies from 12th September 2001: The EEC Directive 91/492/EC is now implemented by means of the 1998 Regulations which themselves were amended by the Food Safety (Fishery Products and Live Shellfish) (Hygiene) (Amendment) Regulations 1999. Classifications that apply to designated shellfish beds in the Fal and Helford cSAC are shown in table 27.

Table 27 Classification of shellfish beds in the Fal and Helford cSAC (Sept 2001)

Production area	Bed name	**Species**	**Category**	**Notes**[1]
Truro River	Grimes Bar and Maggoty Bank	*O. edulis*	B	
	Tregothnan	Mussels	B	Provisional 1
	Calenick Creek, Lambe Creek and Malpas	Mussels	C	
Tresillian River	All beds	Mussels	B	
Fal	Ruan Creek	Mussels	B	Provisional 1
	South Wood	Mussels	B	Provisional 1
	Flushing and Falmouth Wharves	*O. edulis*	C	
	Meads	*O. edulis*	C	Provisional 2
	Mylor Creek	Mussels	B	
	All other beds	*O. edulis*	B	
Percuil	All beds	*O. edulis* & *C. gigas*	B	
Helford	Port Navas and Calamansack Bar	*O. edulis* & mussels	B	
	All other beds	*O. edulis* & mussels	C	
	Rosehill	mussels	B	3

[1]The comment Provisional is given for information only and indicates that the classification may be subject to review before the periodic update. The following criteria apply;

1. area classified at higher level, although shows marginal compliance;
2. area classified at lower level due to enforcement issues.
3. Rosehill is a designated relaying area

The production areas have been classified according to the extent to which shellfish sampled from the area are contaminated with *E. coli*. The classification of a production area determines the treatment required before molluscs may be marketed and designated production areas in England and Wales are now revised annually. The revision includes a list of areas designated as prohibited for bivalve mollusc production under the regulations, and also areas designated by food authorities as relaying areas. These designations have been made by the Food Standards Agency under the Food Standards Act 1999 (Transitional and Consequential Provisions and Savings) (England and Wales) Regulations 2000.

Values for principal microbiological parameters in water were high in the upper Fal Estuary (Truro/Tresillian area) and in the vicinity of known sewage discharges/outfalls. Of the 319 results for faecal coliforms (confirmed) in the period 1996-2001, 124 were below detection limits of 10 per 100ml. Results are summarised in Figure 65. With the exception of Tresillian, the EQS guideline value for shellfish waters of 300 faecal coliforms per 100ml (as 75th percentile) was not exceeded, although mean values for several sites were relatively high. In Tresillian shellfish waters the 75th percentile for the period was 512, with no indication of significant change over time.

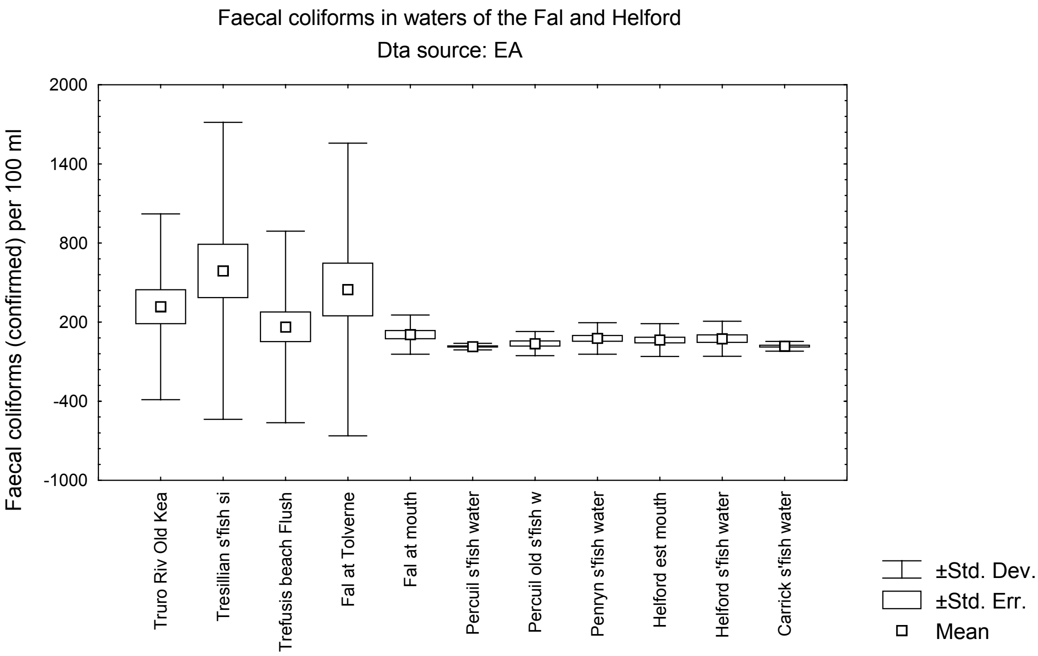

Figure 62. Faecal coliforms in waters of the Fal and Helford cSAC.
NB For statistical purposes, values below the limit of detection have been assigned a nominal value of half the detection limit (generally 10 per 100ml).

EU bathing water standards apply only to designated bathing waters during the bathing season, and are designed to protect human health. Member States are obliged to meet the imperative (mandatory) standards in designated waters, however, Cole *et al* (1999) note that scientific reasoning for their derivation is unknown, and suggest that where such waters have been designated within European marine sites, the conservation agencies should argue that guideline values, at least, should be met. There may be a requirement for higher standards to secure the favourable condition of cSAC features of interest. Additionally, there are non-designated waters in both the Fal and Helford used by the public for swimming and water sports, although the Agency has no statutory powers to take actions to reduce point source inputs to such areas. EQS for microbiological parameters in UK bathing waters are shown in table 28 (full list of standards in Annex 4).

Table 28. EQS under the Bathing Waters Directive

	Unit	Guideline value	Imperative value
Faecal coliforms	per 100 ml	100 80%ile	2000 95%ile
Total coliforms	per 100 ml	500 80%ile	10000 95%ile
Faecal streptococci	per 100 ml	100 90%ile	
Salmonella	per l^{-1}		0 95%ile
Enterovirus	PFU 10 l^{-1}		0 95%ile

The three EC designated bathing waters within the cSAC are Gyllyngvase, Swanpool and Maenporth, all south of Falmouth. Table 29 shows the record of compliance for the designated bathing waters 1990 – 2001.

Table 29. The Fal EC Designated Bathing Water Results (EA 2002)

	Gyllyngvase		Swanpool		Maenporth	
	Imperative	Guideline	Imperative	Guideline	Imperative	Guideline
1990	Pass	Fail	Pass	Fail	Pass	Fail
1991	Pass	Fail	Pass	Fail	Pass	Fail
1992	Pass	Pass	Fail	Fail	Pass	Fail
1993	Pass	Pass	Pass	Fail	Fail	Fail
1994	Pass	Pass	Pass	Pass	Pass	Fail
1995	Pass	Pass	Pass	Pass	Pass	Fail
1996	Pass	Pass	Pass	Pass	Pass	Fail
1997	Pass	Pass	Pass	Fail	Fail	Fail
1998	Pass	Pass	Pass	Pass	Pass	Fail
1999	Pass	Pass	Pass	Fail	Pass	Fail
2000	Pass	Pass	Pass	Pass	Pass	Pass
2001	Pass	Pass	Pass	Pass	Pass	Pass

The bathing season runs from 15 May to 30 September and sampling begins two weeks before the start of the season. At least 20 samples were taken for coliform and faecal streptococci analysis at each designated bathing water.

Imperative standards given in the Directive require there to be no more than 10,000 total coliforms or 2,000 faecal coliforms per 100ml. For a bathing water to comply, 95% of samples taken must meet these standards. Guideline standards for coliforms are 20 times more stringent. 80% of samples must not contain more than 500 total coliforms or 100 faecal coliforms per 100ml. Also 90% of samples must not contain more than 100 faecal streptococci per 100ml (table 27). Using these standards as criteria, bathing waters of the cSAC recently complied with the imperative and guideline values for total coliforms.

Values for faecal coliforms (and faecal streptococci) are sometimes elevated at other sites within the cSAC as indicated in figure 62. (It should be stressed however that the only designated Bathing Waters are Gyllyngvase, Swanpool and Maenporth; other sites are included for comparison only and compliance is not mandatory).

Microbiological parameters such as faecal coliforms are generally monitored in the estuarine environment to indicate the presence of other human derived microbial pathogens, and are principally associated with sewage discharges. In rural areas, animal-derived slurry in run-off from waterside fields and farms may provide an additional source of coliforms. The marine environment is hostile to most microbial pathogens and they rapidly die off, especially in the presence of sunlight. However they may become associated with suspended particles and can accumulate to some extent in sediments, surviving for days or weeks. This is therefore a consideration when permitting dredging operations close to bathing waters or shellfish waters during the bathing or harvesting seasons (respectively). Microbial pathogens also accumulate in filter feeding organisms to levels that can be harmful to themselves (microbial toxins), to humans and perhaps other consumers (e.g. birds and marine mammals). In addition, the aesthetic appearance of the site could also be damaged.

Values for faecal coliforms in the cSAC indicate a sewage-related problem in the upper Fal (Truro and Tresillian) and lower Carrick Roads (Flushing, Tolverne) areas. The Shellfish Hygiene Directive has been a significant driver for recent improvements to STWs and sewerage facilities in the cSAC. SWW have also scheduled plans to further improve water quality in these areas, this will include the addition of UV disinfection to the current processes (see Appendix 7 and footnote on p39), which may alleviate the problem somewhat, however careful monitoring of the situation is strongly recommended .

7. MODELS

Several modelling exercises of direct relevance to environmental quality status of the Fal/Helford cSAC have been undertaken. These are summarised below. Of particular importance are modelling studies described in a report produced to assess the impacts of Falmouth Sewage Treatment Scheme (Pell Frischmann Water Ltd 1999).

7.1 Assessment of Impacts on Local Hydrodynamics and Marine Conservation Interests

A comprehensive assessment of the overall impact that the proposed Falmouth sewage treatment scheme might have on the Fal and Helford system was carried out in 1999 (Pell Frischmann Water Ltd 1999). The SWW scheme included the placement of two aeration tanks and a protective rock revetment on the foreshore of the proposed development site, a former landfill site at the eastern end of Falmouth Docks. Physical flume modelling enabled optimisation of the design for the revetment profile. Numerical, hydrodynamic and wave modelling was used to determine the local processes affecting the site and to make recommendations to minimise the potential impact on the important subtidal habitats of the cSAC.

Results from wave flume modelling and measured wave data, coupled with MIKE21 modelling (see section 7.3), an extreme wave climate analysis and a literature review demonstrated that the scheme would not have adverse effects on the interest features of the Fal and Helford cSAC. The protection afforded to the east cliff face by the rock revetment was forecast to result in a net reduction in potential erosion, and therefore reduce sediment supply and contamination to the marine environment compared with that occurring under the existing (pre-scheme) conditions.

7.2 Catchment modelling and Wheal Jane

Under the Wheal Jane Minewater Project, development of hydrological and hydrogeochemical models of the catchment was progressed to allow the prediction of the extent of treatment required to achieve various WQOs set by the Agency for the Carnon River and Fal Estuary (i.e. prediction of the effect of the treatment options on water quality and discolouration in the Carnon River, Restronguet Creek and Carrick Roads (W S Atkins, 1999)). Prediction of water quality from the catchment modelling studies was used to assess the effect of treatment on sediment concentrations and the biological communities and populations in the Fal Estuary (Warwick *et al.,* 1998).

7.3 Dynamic models of the Fal

These have been reviewed recently in relation to TBT impacts in the Fal (Harris 2001) but are also relevant to simulations of the distribution of metals. Sherwin (1993) and Sherwin and Jones summarise the dynamics of the water in Carrick Roads, in particular with reference to the sewage discharges around the mouth of the Estuary at Falmouth. Other major studies include those of Metocean 1991; Millward 1992; South West Water Services Ltd 1992 and, for the Penryn River, Remmer (1998). W S Atkins (1999) describe a high resolution MIKE 21 model for the Estuary in connection with the Wheal Jane minewater project which predicts metal distributions under different water treatment scenarios. The relative merits of these hydrodynamic models are discussed by Harris (2001) who suggests they may not be sufficient to accurately predict long term fluxes across the mouth: for this purpose recommendations are made for a 1D, tidally averaged numerical model to quantify the long-term dynamics of TBT in the estuary.

Observational data indicate that there is some export of metals from Restronguet and accumulation of metals in sediments on the western bank of Carrick Roads, including Mylor Creek (Bryan and Gibbs, 1983; Warwick *et al.,* 1998), which is consistent with the anticlockwise water circulation pattern (Sherwin and Jones 1999). Harris (2001) proposes that further modelling of sediment transport be developed to address this issue (and that of TBT- contaminated spoils), perhaps by adaptation of existing 2D models (Sherwin and Jones 1999; Metocean 1990).

7.4 Nutrient modelling

Using models to estimate nutrient status is inherently problematic. Parameters such as the local geology and sediment type, land use, volume, dilution and flushing rate, rainfall, vertical mixing, and wave exposure influence the nutrient status of environmental waters and are unique to each estuary.

In 1996, nutrient load impacts were quantified using an estuarine modelling system, ECoS (in EA, 1997a). The estuary was modelled as a branched system, with the main branch from the tidal limit at Truro following a line along the lowest depth to a point 15000metres down estuary, off Restronguet Creek in Carrick Roads. The River Tresillian, from its confluence at Malpas to a freshwater point 5000 metres upstream was the second branch. TIN was used for river and STW nitrogen inputs. Two simulations for 1995 were carried out, one with all freshwater and STW loadings, and one without Newham STW. The impact of Newham was clearly shown, with substantial enrichment at sites in the upper estuary. It was estimated that in summer, Newham STW contributed up to 50% of the TIN loading in the estuary.

A project taking into account a wide range of physical characteristics in order to determine the relative contribution of diffuse and point source inputs to the Fal and Helford was recently undertaken (Fraser *et al.,* 2000). Models used were P-EXPERT, EXPORT COEFFICIENT MODELLING and INCA-N (Integrated Nitrogen in Catchments). It was also intended that the information gained be used to develop a catchment-wide list of consent licences likely to have an effect under the Habitats Directive. The initial aims of the work appear to have been achieved and estimates of nutrient budgets to the system as a whole have been made (discussed in relevant section of the current report). Efforts to complete the secondary objective appear to have been thwarted by lack of good quality, high resolution data, however, results suggest that the china clay works in the upper part of the Fal catchment may have a significant effect on the total nitrogen load.

In the same study, these models been used to simulate the catchment dynamics of the Tamar Estuaries Complex (for which more comprehensive data were available), apparently with some success. Clearly, the modelling approach has potential benefits but needs more accurate data to be applied successfully.
There are also limitations in that models appear to predict nutrient loading to the estuary as a whole and do not take into account site-specific variability, such as that which occurs in the more enclosed and low-flow areas within the system.

8. CONCLUDING REMARKS: FAL AND HELFORD cSAC

This report has set out the status of the Fal and Helford cSAC as regards ***available evidence*** on water and sediment quality and environmental impact. Wherever possible, we have used UK EQS values to identify problem areas. However, for many substances, the UK has not set benchmark values and we have drawn on proposed standards, guidelines adopted in other countries, and published literature on dose/effect levels, to assess environmental quality and links to impact. It is also important to recognise that environmental concentration data for many chemicals is non-existent for the Fal and Helford (as elsewhere). This supports the argument that biological effects-type monitoring must accompany the suite of chemical measurements made, to ensure a more acceptable degree environmental protection.

Table 30 is a much simplified, quick-reference summary of contaminants and their influence on water quality within the cSAC, based on available information. This highlights (i) areas of the system where problems are perceived (if any), (ii) sources of contaminants and (iii) threatened features/biota. There is also a brief synopsis of these issues with findings for each of the numbered 'contaminant categories' explained in more detail.

1) TBT. Although originally emanating from Falmouth docks and marinas (where highest concentrations are nowadays observed) waters and sediments throughout the entire Fal system are chronically contaminated with TBT. The relative contributions of sediments as a source is still to be resolved adequately but in view of the fact that concentrations are higher than expected from partitioning calculations, its significance is likely to be high and may include a considerable proportion from paint particles. Concentrations throughout much of the Fal are high by national and international standards and do not appear to have been reduced by initial TBT legislation on small vessels. There is clear evidence of impact on dogwhelk populations, which have been eliminated from most of the Fal by TBT. There is also circumstantial evidence based on comparisons with EQS values and sediment toxicity guidelines that other infaunal organisms, particularly molluscs, will be affected. Most faunal surveys indicate that the Fal is impoverished compared to similar systems, and though this probably relates in part to the long history of mining impact, a significant component of the damage and a continuing impediment to recovery, may be attributable to TBT

The principal source of TBT is thought to be Falmouth dockyard discharges, which could well maintain TBT levels above 2ng l^{-1} throughout significant parts of the system. However there will also be a component arising from sewage inputs, sediment-water exchange and large shipping. The relative contributions of these components have yet to be accurately assessed. The potential for short-term sediment releases during artificial remobilisation - dredging, for example - is likely to be considerable and is likely to contribute substantially to ecological damage in the long term.

There are limitations for a definitive assessment of TBT status caused by a lack of detailed information, notably - inputs to the system, environmental inventories (particularly for sediments), and more detailed information on the variables controlling partitioning. This information would be vital to clarify the processes

Table 30. 'Contaminants' and their role in influencing water quality in the Fal and Helford cSAC. (Findings for each of the numbered 'contaminant categories' are explained in more detail in the accompanying text).

Principal problems	Area	Sources	Most vulnerable features/biota
1) Organotins (TBT, TPT)	Entire system esp. Falmouth Docks area	Falmouth Docks, shipping, sediments, sewage discharges	Molluscs (primarily gastropods, also some bivalves
2) Metals (esp. As, Cu, Cd, Fe, Zn,)	Restronguet and Mylor area, upper Fal	Past mining activity, sediments, discharges	Invertebrates (primarily molluscs, crustaceans), species composition, fish and birds.
3) Nutrients	Upper Fal (Truro area) Helford (Bonallack Barton, Polwheveral)	Sewage discharges, run-off from land, mine drainage.	Invertebrates, fish (estuarine and migratory, esp. early life stages), seabirds, mammals, Zostera and Maerl beds (and associated diverse fauna)
4) Turbidity	Upper Fal (Truro, Tresillian area)	Sewage discharges, aggregate extraction (dead Maerl)	Live Maerl, Zostera, benthic communities, fish
5) Microbiological parameters	Upper Fal (Tresillian) Penryn River (Flushing)	Sewage discharges, farm animals (run-off from land)	Bivalves, birds, marine organisms
6) Hydrocarbon oils, PAHs	Lower Fal and Helford (oils). PAH distributions unknown	Discharges, Shipping, urban run-off, Atmospheric deposition	Bivalves, species composition, fish, seagrasses and shoreline communities
7) Pesticides, herbicides and other synthetic organics	Poorly defined but silty sediments probably main reservoir	Not quantified but probably includes a significant component from agricultural run-off (herbicides and pesticides). Inputs of eg lindane and dieldrin from STW probably declining	Invertebrates (esp. crustacea), fish and food chain bioaccumulation Endocrine disrupting effects not studied

determining organotin distributions in the estuary and to develop models that simulate the fate of TBT in the cSAC.

Harris (2001) has already identified many of the shortfalls for modelling requirements. In addition to more detailed partitioning data, and TBT profiles in sediment cores, these include improved parameterisation of sediment movement within the cSAC, and hydrodynamic modelling, with particular relevance to the potentially damaging effects on maerl beds.

2) Metals Sources of metals in the cSAC are dominated by the Carnon Valley and Restronguet Creek, and levels in waters and sediments of Restronguet Creek remain high. Treatment of water draining from the Wheal Jane has reduced inputs from this source. However, removal of dissolved metals to sediments in Restronguet Creek may now be less efficient.

Arsenic. Export of As from Restronguet has resulted in some arsenic enrichment in the upper Fal (above Restronguet) and the western shore of Carrick Roads. Arsenic concentrations in sediments of Restronguet and the upper Fal are in the 'probable effect level' range. Releases of As to the water column from contaminated sediments may be particularly problematic. Early life stages of crustacea are vulnerable to As toxicity, and relatively low concentrations of As in water can inhibit growth of some important primary producers.

Copper. Generally, Cu concentrations in water outside the influence of Restronguet Creek are below EQS and not significantly elevated to cause concern. However, a second major source of Cu in the Fal is the outfall at Falmouth Dockyard and this is reflected by elevated Cu concentrations in waters and sediments of the area. In addition, based on PEL's, Cu in sediments of all major Rivers and Creeks of the system (including Helford) is present at levels predicted to cause effects to biota. Some organisms, e.g. cockles, accumulate Cu to levels which can have lethal or sublethal effects, and desorption from sediments may be an important source for such infauna. The impact of increasing use of copper-based antifoulants on leisure craft has still to be evaluated. Despite the acknowledged toxicity of Cu relatively little is known of the implications for many potentially sensitive species in the Fal and Helford cSAC.

Cadmium. With the exception of elevated levels near Devoran Bridge, cadmium is below EQS and not considered as a water quality problem. However, Cd in sediments could be a cause for concern, and in view of the notable lack of information for the Helford estuary and the potential risk to sediment dwelling organisms and their predators, Cd distribution should be investigated further. Cd is not generally associated with sewage discharges in to the cSAC and any elevated levels in the area are likely to have arisen as a result of past mining and ore processing activities.

Iron. Apart from Restronguet Creek, Newham STW at Truro, and the occasional elevated value around the dockyard, levels of dissolved Fe at other sites in the cSAC are close to background and below EQS, therefore direct effects of Fe are not a water quality issue. The distribution of Fe in the cSAC is strongly influenced by salinity as dissolved Fe in freshwaters entering the estuary oxidises and rapidly precipitates on mixing with seawater. This precipitation process has implications for other dissolved metals, which are scavenged by Fe oxyhydroxides produced during estuarine mixing. Since the introduction of minewater treatment at Wheal Jane Fe inputs and oxyhydroxide production is now reduced, and scavenging of other metals in the estuary may also be less pronounced. Thus, Fe has become an important factor in the water/sediment distribution of other potentially more damaging metals, and clearly implications for the biota of the cSAC need to be examined.

Zinc. In addition to Restronguet Creek, occasional high values for dissolved Zn are recorded in waters of the upper Fal (Tolverne, Truro, Old Kea and Tresillian), which probably originate in sewage discharges. High concentrations of Zn are also associated with Falmouth Dockyard combined outfall. These warrant further investigation in view of the use of Zn compounds in antifouling paints. Zn levels in sediments are also of concern, and based on 'probable effect levels' pose a threat to benthic and sediment dwelling organisms in most of the creeks and tributaries of the system. In addition, the tolerance developed to elevated Zn by some organisms can result in bioaccumulation and they themselves may present a hazard to predatory fish and birds of the cSAC.

3) Nutrients. It is evident that nutrient enrichment is a problem in the Fal/Helford system. Elevated nitrogen and phosphorus species are an important factor in poor water quality, and in the Fal Estuary have lead to symptomatic changes such as algal blooms, fluctuations in dissolved oxygen, and turbidity which come under the broad heading of eutrophication. The most potent indication of this was the 1995 and 1996 toxic algal blooms originating in the upper Fal Estuary/Truro River, and a more recent (2002) toxic bloom which resulted in invertebrate mortalities in the Helford Estuary. The nutrient status in parts of the estuary, notably the upper Fal and parts of the Helford, therefore comprises a significant threat to many important conservation features. Though there is no confirmation of a clear link between nutrient enrichment and ecological status, evidence suggests indirect links to the decline of *Zostera* and Maerl beds, an increase in epiphytic, blanketing algae, algal blooms, (including toxin-producing algae), fish kills and reduced biodiversity.

The principal sources of excess nutrients in the problem areas are considered to be sewage discharges, although diffuse inputs from agricultural sources may also be a contributing factor, especially for nitrogen. The problem may be exacerbated by re-release of phosphorus from sediments, and seasonal fluctuations in N:P ratio of waters entering the tidal estuaries.

With the exception of the lower Fal estuary, there are no apparent temporal reductions in nutrient levels; in fact for many areas there is evidence of significant increases over the past 15 years. The designation of the Fal, Truro and Tresillian Rivers as Sensitive Waters (Eutrophic) in 1998 may herald improvements to treatment at Newham STW, the principal source of enrichment in the upper Fal Estuary during the summer. However, harmful algal blooms continue to occur in the area. The most recent incidence, in 2002, also affected Polwheveral Creek and Porth Navas, on the Helford Estuary, causing mortalities of rag- and lug-worms and some shellfish. Thus, high nutrient levels in the more enclosed areas of both estuaries remain a cause for concern.

Unfortunately the lack of water and sediment quality standards for nutrients make it impossible for consents to be judged in terms of statutory limits in receiving waters. Also, monitoring of nutrients in tidal waters appears to be non-routine. This is identified as a significant gap in the ability to manage the system optimally.

4) Turbidity In the Helford estuary, and the lower Fal as far up as King Harry Ferry, significant reductions over time in the level of turbidity have no doubt benefited sensitive benthic communities such as those associated with Maerl and *Zostera* beds. However, the upper Fal (Truro area) has seen little or no change and turbidity levels remain consistently high with the potential for causing adverse effects to zooplankton, benthic communities and fish populations. Suspended solids in sewage discharges are considered to be the principal source of turbidity, coupled with re-suspension of sediments during periods of high flow and tidal movements. During spring and summer blooms, the problem is exacerbated by significant amounts of algae (reflected in increased Chlorophyll α concentrations). Extraction of dead Maerl may contribute to turbidity in the lower estuary and threaten live beds, although the impact is said to be minimised by restricting dredging operations to coincide with tides which carry the plume away from the live Maerl.

5) Microbiological parameters Values for coliforms and faecal streptococci exceed EQS values in the upper Fal (Truro and Tresillian) and lower Carrick Roads (Flushing, Tolverne) areas and highlight the sewage-related problem in the Fal Estuary. High levels of these organisms can indicate the presence of microbial pathogens and toxins, and although principally a threat to shellfisheries in the area can, in theory, constitute a potential threat to almost any plant or animal. In marine waters, bacterial pathogens have been linked to fish kills and infections in sea birds and mammals. Plans to improve sewage treatment may alleviate the problem somewhat, although the hydrodynamics of the upper estuary does not appear to favour the removal of discharges and the problem may call for other measures to be taken.

6) Hydrocarbon oils. The little information we have on hydrocarbon oils indicates that a problem may exist in the cSAC. Concentrations of (unspecified) hydrocarbons at the mouth of the Fal, and the mouth of the Helford suggest some general contamination in the area which possibly originates from the dockyard and shipping, but may be related to run-off and/or aerial deposition. In enclosed systems such as the Fal, the potential for dispersion of hydrocarbons is minimised, and much of the contamination can be washed ashore. Perhaps the most vulnerable areas are intertidal habitats where sensitive shoreline seagrass and saltmarsh communities could be at risk. Although EQS's for hydrocarbons are difficult to define, levels are, or have been sufficiently elevated to cause concern and result in the tainting of shellfish and fish. Further investigation into the sources, levels and composition of hydrocarbons (including PAHs) in the cSAC would seem to be necessary.

7) Synthetic organic compounds have not been widely monitored, though concentrations of the majority of these substances (which includes many endocrine disruptors) appear, generally, to be very low in the cSAC. Results for pesticides in sediments are largely restricted to one site, off Falmouth, sampled on up to three occasions in the early 1990s. If the level of contamination found here were typical throughout the Fal and Helford cSAC it is unlikely that pesticide concentrations would cause harm. Bioaccumulation data appears to be based on a single species (oysters) and the predominance of <DL values suggests that, overall, pesticides and herbicides pose little toxicological threat, though there are isolated examples of elevated concentrations of DDT, dieldrin and γ-HCH. Current trends in bioaccumulation should be established by further sampling, particularly with a view to identifying sources and characterising bioavailability.

EA records of PCBs in water and oysters are largely below limits of detection at shellfish sites within the cSAC, though an unpublished report from the early 1990s indicates measurable levels of 5 PCB isomers in oysters from the Helford Estuary (above the OSPAR upper guideline for mussels). These oysters were from a re-lay bed at Lower Calmansack and had been collected from a number of areas in Carrick Roads before their transplantation to the Helford Estuary. Since PCBs were not found in water or sediments from the re-lay site it was concluded that the oysters were from elsewhere in the Fal or Helford Estuary. This inconsistency highlights the need for more rigorous sampling in the cSAC, especially for organochlorines and other organic substances that accumulate in sediments and affect estuarine biota through food-chain magnification. This should incorporate sediment-dwelling species such as clams and worms, and also algae and eelgrass.

8.1 Summary of Evidence for Impact: Fal Estuary

The Fal Estuary has long suffered the impacts of mining and ore extraction which have been a feature of the area for centuries. For this reason the Fal Estuary system, and in particular Restronguet Creek, has been the site for a number of studies on the effects of long-term heavy metal pollution on marine organisms (Bryan and Gibbs, 1983; Bryan and Langston, 1992). However, there are features, communities and species of interest within the estuary which have resulted in it's candidacy as a European Marine Site, not the least of which are the extensive Maerl, and *Zostera* beds and the diversity of their associated flora and fauna. Some of these features are under threat. Rostron (1985) reported a decline in many species and suggested that this was attributable to organotin compounds and dinoflagellate blooms. The decline has continued unchecked and in some instances almost resulted in their complete elimination form the system.

Notable changes/findings include:

- Dramatic fall in the numbers of dogwhelks, limpets and topshells. Dogwhelks *Nucella lapillus* have now all but been eliminated from the Fal and Helford cSAC by TBT. Populations of European sting winkle *Ocenebra erinacea* are also at risk.
- Evidence of a long term decrease in proportion of live maerl. The flora and fauna of the maerl bed are also lower in abundance than might be expected by comparison with other such beds in Ireland, Scotland and France.
- Low diversity of species associated with *Zostera marina* beds compared to *Zostera* communities in other areas such as the Isles of Scilly.
- Absence of two marine crustaceans (amongst the most sensitive of all marine taxa to pollution) *Corophium volutator* and *Cyathura carinata*.
- Evidence for effects of pollutants (metals, organotins, microbial toxins) to organisms at all levels of the food chain, e.g. meiofauna to mute swans.
- Recruitment of 0-group bass (*Dicentrarchus labrax*) in the Fal nursery grounds is considered to be impoverished in relation to comparable estuaries in the south-west, although this has not been attributed to an identifiable cause.
- Low diversity of bivalves, with *Kellia suborbicularis*, and cockles, *Cerastoderma* and *Laevicardium* disappearing from some sand and gravel banks.
- Reduction in the richness of algal populations in many parts of the Fal and Carrick Roads.
- Relative impoverishment of invertebrate communities in areas of the Fal compared to otherwise similar systems.
- Evidence of sub-lethal effects to some organisms at the most polluted sites in the system.

8.2 Summary of Evidence for Impact: Helford Estuary

The waters of the Helford River are *reputed* to be amongst the cleanest in Western Europe and the area is notable for the absence of industry and associated discharges. Some water and sediment quality and body burden data has been presented in previous sections which generally support the impression of an unpolluted area. However it will be obvious that the number of studies and amount of information addressing contaminants and impacts in the Helford is rather small in comparison to the Fal. This should be rectified, particularly in view of the recent invertebrate mortalities and observations made in the late 1980s which are described below.

As an area of outstanding marine biological significance the Helford River has been the focus of several surveys monitoring the habitats, flora and fauna of the area. In 1987 Rostron produced a report of a subtidal survey of the Helford containing information on the historical perspective, physical conditions (geology and hydrography), human influences, intertidal and subtidal habitats and abundance and distribution of species. This report describes thriving maerl beds off Bosahan Point and several nearshore *Zostera* beds. *Zostera* is also listed in low-level tidal pools on extensive rock platforms close to the mouth of the Helford River. These sites were considered important as other *Zostera* beds within the Helford River (including Flushing Cove, Penarvon Cove) are reported to have been largely destroyed. The author suggests that the disappearance of intertidal *Zostera spp.*, clam *Chlamys varia,* razor shell *Solen marginatus* and peacock worm *Sabella pavonia* was due to increasing numbers of bait diggers and winkle pickers although this is based on anecdotal evidence. During the same year information regarding the geology of the area, land use, river use and 'water quality', and descriptions of intertidal and subtidal habitats, together with species lists was published by Covey and Hocking (1987). Importantly, this report compares survey results with past records and anecdotal evidence and concludes that there is evidence of a 'deterioration of intertidal life for the mid 1970's onward'.

Notable changes reported include:

- The erosion of eel grass (*Zostera*) beds and silting over of adjacent areas of gravel and clitter. *Zostera spp.* are reported to have totally disappeared from many areas; Gweek, Scots Quay, and Helford Creek (*Zostera noltii*), Penarvon Cove (*Z.marina* and *Z. noltii).*
- Reductions in numbers of peacock worm *Sabella penicillus*, edible cockles *Cerastoderma edule*, mussel *Modiolus adriaticus*.
- Dramatic decline in oyster numbers since 1982, (due to *Bonamia ostreae*)
- Dramatic decline in numbers of dogwhelk *Nucella lapillus* due to TBT (not substantiated and no data given). Populations only present at outer headlands Nare Point and Rosemullion Head.

The latter observation, in particular, flags the need for more information on TBT in the Helford (and Falmouth Bay) in view of the known sources in the Fal.

A later report on behalf of the HMVC (Tompsett, 1994) records major changes in biota for nine sites over a six-year period (1986-93) during which 4 surveys were carried out. These changes include the disappearance of *Zostera marina* from Helford

Passage and Treath. An increase in the population of Australian barnacle *Elminius modestus* and other barnacle species is attributed to the TBT ban, although no TBT measurements were made. In a recent update (HVMCA, 2000), which includes accounts of physical and biological features (including rare species) and human activities, *Zostera* distribution is described as being limited to a large bed off Durgan, with small areas at Bosahan, Helford Passage and Men-aver reef (west of Nare Point). Maerl beds are reported at Bosahan and North of Dennis Head. A decline in numbers of birds using the river since 1970's is noted although no explanations are put forward for this. Pollution in the Helford, overall, is depicted as being slight, except for occasional evidence of sewage in the tourist season (Constantine STW has been identified for improvement by SWW), although, again, water quality measurements are not included in the HVMCA report.

8.3 Future Research Requirements

This overview of site characteristics has indicated, throughout, a need for better assessment of the impact of consents on the Fal and Helford cSAC.

In summary requirements are:

- Further targeted investigations in the Fal and Helford to determine the concentrations of pollutants of concern (Arsenic, metals, TBT, nutrients, hydrocarbons) within the water column and sediment: improved understanding of their behaviour and distribution.
- Continuation of specific long-term data sets (Fal; metals and TBT) to examine temporal trends.
- Regular re-assessment of EA monitoring data in terms of compliance with statutory standards (and guideline values for sediments): to describe and map more accurately threatened areas in order to target biological surveillance and to re-focus chemical monitoring.
- The process of predicting risks posed by discharges to aquatic environments, based on comparisons with EQS values, is only a first order approximation, not least because each stressor is considered individually without consideration of issues such as synergistic and additive effects. Predicting total exposure risks in receiving waters subjected to 'complex mixtures', should be addressed.
- Improved hydrodynamic models and validation. Modelling dispersion of metals, nutrients and TBT in relation to the major sources and comparing with water and sediment quality standards and guidelines.
- Determine spatial and temporal trends in benthic communities and their tolerance in relation to major toxic and non-toxic contaminants. Combination of traditional survey, continuation of major time series, and comparisons with similar habitats in other SACs.
- Apply novel biomarker-type techniques to assess biological impacts and their causality (including sediment bioassays and laboratory studies to determine how metals, TBT and nutrients react with each other to affect their potential toxicity in the environment).
- Research to address the notable lack of firm data on endocrine disruptors and their effects in the cSAC, including effects of biological processes on transport and speciation of these compounds.

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

Targeted chemical measurements are a prime consideration. Some suggestions have already been provided but in summary these should include profiles of metals in waters of the Fal, paying particular attention to the distribution and dispersion of metals from Restronguet Creek as a function of river flow and tidal state (to include modelling of profiles and inputs under different tidal and hydrometerological conditions). Some initial intensive work on Arsenic remobilisation from sediments (and perhaps other metals is warranted). Annual surveys of sediments metal loadings, bioaccumulation in key bioindicators and attempts to characterise sediment bioavailability is also recommended to dovetail with previous surveys and thus provide clear evidence of long-term trends. Comparable work on TBT is also recommended and would be made more cost effective if sampling was integrated with some of the metals programme.

There should be a more comprehensive analysis of nutrient information and probably generation of more targeted data for distributions, sources, and seasonality, including sediments, to establish budgets and improve models.

There are requirements in ENs advisory documents to ensure that key communities and habitats are not degraded. Some of the multivariate – type approaches discussed in an earlier section have proved to be extremely useful in plotting spatial and temporal trends in benthic communities in relation to metal levels. This would seem an ideal platform on which to build for future indications of biological change. These might be conducted at intervals of perhaps five years or to coincide with the regulatory authorities review cycles

Techniques are now becoming available to assess biological impact in individuals, which would allow screening of the entire cSAC, including possible problem discharges. By selection of an appropriate suite of indicators/biomarkers, a sampling strategy could be tailor-made to establish with greater certainty the causes and extent of damage. These include ecological survey procedures for identifying changes in the abundance and diversity of species, chemical and biomonitoring procedures for determining the concentrations and bioavailability of anthropogenic contaminants, and biochemical, physiological and behavioural biomarkers which signal exposure to, and in some cases, adverse effects of pollution (see Annex 6 for further details).

When these procedures are used in combination in well-designed survey programmes, they can provide insight into which pollutants are responsible for environmental degradation. They are equally amenable for measuring long-term trends e.g in the assessment of recovery.

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10 ANNEXES

Annex 1. Fal And Helford cSAC: Summary Of The Interest Features

Fal and Helford cSAC: Summary of the interest (or qualifying) features, and conservation objectives, (adapted from English Nature, 2000)

Large shallow inlets and bays

Overall objective – Maintain/ensure favourable condition re: extent, water clarity, nutrient status

Subfeatures

- Rocky shore communities
  - maintain extent and distribution
  - maintain species composition of low shore boulder community
  - maintain species composition of rock pools
- Subtidal rock and boulder community
  - maintain distribution
  - maintain composition
- Kelp forest communities
  - maintain algal species composition
  - characteristic species population size
    - *Laminaria spp.*
    - *Distomus* (tunicate)
- Subtidal mud communities
  - Maintain species composition of biotope

Subtidal sandbanks

General objectives – Maintain/ensure favourable condition re: extent; sediment character (granulometry); topography; water density

Subfeatures

- Eelgrass bed communities
  - maintain extent and density of beds
  - maintain epiphytic community
  - maintain nutrient status (extent of competing green algal bed)
- Maerl bed communities
  - maintain extent of maerl beds (live and dead)
  - maintain distribution of biotope
  - maintain species composition of communities
  - maintain nutrient status (extent of competing green algal mat)
- Gravel and sand communities
  - maintain species composition
  - maintain characteristic biotopes
- Mixed sediment communities
  - maintain species composition
  - maintain characteristic biotopes

Intertidal mudflats and sandflats

General objectives – Maintain/ensure favourable condition re: extent (area), sediment character (granulometry and compaction), topography, nutrient status (algal mats).

Subfeatures

Mud communities
- maintain extent and distribution of characteristic biotopes

Sand and gravel communities
- maintain extent and distribution of characteristic biotopes

Muddy sand communities
- maintain extent and distribution of characteristic biotopes

Atlantic "Salt Meadows" (Saltmarsh)

General objectives – Maintain/ensure favourable condition re: extent; creek patterns, range and distribution of characteristic saltmarsh communities, vegetation structure (height, impact of grazing).

Subfeatures based on zonation

- Low marsh/low mid-marsh communities
  - maintain frequency/abundance of characteristic species
- Mid/mid-upper marsh
  - maintain frequency/abundance of characteristic species

……Cont

Annex 1.(cont.)

Fal and Helford cSAC: Summary of additional interest (or qualifying) features, and conservation objectives, proposed since those described in English Nature, 2000

Estuaries

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

Defined in the submission as semi-enclosed bodies of water which have a free connection with the open sea and within which the sea water is measurably diluted by freshwater from the surrounding land. They are usually large features containing a complex range of habitats that reflect the variations in tidal influence and substrate type.

Reefs

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

Defined as areas of rock or biological concretions formed by various invertebrate species. Reefs occur in the subtidal zone, but may extend onto the shore.They form the habitat for a variety of biological communities such as those characterised by encrusting animals and attached sea weeds

Rumex rupestris (shore dock)

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

Shore dock grows on rocky and sandy beaches, at the foot of cliffs and infrequently in dune slacks where there is a supply of fresh water. It is thought to be the world's rarest dock and is one of the rarest plants in Europe. In the UK it is found only on a small number of sites in south west England and Wales.

Annex 2. Water Quality Standards

List I (EC Dangerous Substances Directive) and list II substances (from Cole *et al.,* 1999, derived by WRc according to the methodology described in Grimwood and Dixon 1997)

List I substances

Parameter	Unit	Water quality standard		Standstill Provision [a]
		Estuary[b]	Marine	
Mercury	µg Hg l-1	0.5 DAA	0.3 DAA	yes[c]
Cadmium	µg Cd/l	5 DAA	2.5 DAA	yes
Hexachlorocyclohexane[d]	µg HCH/l	0.02 TAA	0.02 TAA	yes
Carbon tetrachloride	µg CCl_4/l	12 TAA	12 TAA	no
Dichlorodiphenyltrichloroethane				
(all 4 isomers, total DDT)	µg DDT/l	0.025 TAA	0.025 TAA	yes
(para, para-DDT)	µg ppDDT/l	0.01 TAA	0.01 TAA	yes
Pentachlorophenol	µg PCP/l	2 TAA	2 TAA	yes
Total "drins"	µg l-1	0.03 TAA	0.03 TAA	yes
Aldrin	µg l-1	0.01 TAA	0.01 TAA	yes
Dieldrin	µg l-1	0.01 TAA	0.01 TAA	yes
Endrin	µg l-1	0.005 TAA	0.005 TAA	yes
Isodrin	µg l-1	0.005TAA	0.005 TAA	yes
Hexachlorobenzene	µg HCB/l	0.03 TAA	0.03 TAA	yes
Hexachlorobutadiene	µg HCBD/l	0.1 TAA	0.1 TAA	yes
Chloroform	µg $CHCl_3$/l	12 TAA	12 TAA	no
1,2-Dichloroethane (ethylenedichloride)	µg EDC/l	10 TAA	10 TAA	no
Perchloroethylene (tetrachloroethylene)	µg PER/l	10 TAA	10 TAA	no
Trichlorobenzene (all isomers)	µg TCB/l	0.4 TAA	0.4 TAA	yes
Trichloroethylene	µg TRI/l	10 TAA	10 TAA	no

Notes: Substances are listed in order of publication of Directives.

D Dissolved concentration, ie usually involving filtration through a 0.45-µm membrane filter before analysis
T Total concentration (ie without filtration).
AA standard defined as annual average
[a] Most directives include, in addition to the standards for inland, estuary and marine waters, a provision that the total concentration of the substance in question in sediments and/or shellfish and/or fish must not increase significantly with time (the "standstill" provision).
[b] In the UK the standards for estuaries are the same as for marine waters - The Surface Waters (Dangerous Substances) (Classification) Regulations 1989
[c] In addition to a standstill provision applying to sediments or shellfish there is a further environmental quality standard of 0.3 mg Hg/kg wet flesh "in a representative sample of fish flesh chosen as an indicator".
[d] All isomers, including lindane

Annex 2 (cont.) Water quality standards for the protection of saltwater life.

List II substances

Parameter	Unit	WQS (see footnotes)	Uncertainties in the derivation : Details obtained from the relevant EQS derivation reports
Lead	µg Pb/l	25 AD[1,5]	The preliminary EQS was multiplied by a factor of 2 to account for overestimation of Pb toxicity in laboratory studies compared to the field environment. The EQS was considered tentative as a result of the paucity of reliable data, in particular for sub-lethal chronic studies with invertebrates and fish, and for field studies.
Chromium	µg Cr/l	15 AD[1,5]	There were limited data on the sub-lethal effect of Cr and long-term exposure to freshwater and saltwater life. Separate standards for different Chronium valences (Cr(VI) and Cr(III)) were not recommended as a consequence of the lack of data for Cr(III). In addition, a comparison of the toxicities of each oxidation state was not possible. Some data were available that indicated higher sensitivity of some saltwater organisms to low salinities. The EQS was based on data generated at salinities typical of normal seawater. Therefore, further research on the effect of Cr at lower salinities was recommended.
Zinc	µg Zn/l	40 AD[1,5]	The dataset available for the toxicity of Zn to saltwater life illustrated that at the EQS, adverse effects on algal growth had been reported. However, it was considered that there was currently insufficient evidence to suggest that the EQS would not adequately protect saltwater communities.
Copper	µg Cu/l	5 AD[1]	Further data were considered necessary on the sensitivity of early life stages and life-cycle tests to confirm the sensitivity of saltwater life.
Nickel	µg Ni/l	30AD[1]	Marine algae were reported to be adversely affected by Ni at concentrations as low as 0.6 _g l^{-1} which is below the EQS to protect saltwater life However, it was considered that there was insufficient evidence to justify a lower EQS based solely on results with algae and that further research into this area was desirable. There was also limited evidence to suggest that invertebrates in estuarine systems may be more susceptible to the effects of Ni than invertebrates in marine systems. Thus, an EQS to protect estuarine life may be needed in future when further data become available.
Arsenic	µg As/l	25AD[2]	Based on crab 96 hour LC50, and an extrapolation factor of 10 applied. Standards may need to be more stringent where sensitive algal species are important features of the ecosystem
Boron	µg B/l	7000 AT[1]	Few data available. However the standard was based on Dab 96 hour LC50, with an extrapolation factor of 10 applied
Iron	µgFe/l	1000AD[1,5]	The EQS for the protection of saltwater life was based on observed concentrations and general assessments of water quality. It was recommended, therefore, that the standard should be reviewed as soon as direct observations of water concentrations and biological status become available. Limited data did not allow an assessment of the importance of Fe species.
Vanadium	µgV/l	100 AT[1]	Data on the toxicity of vanadium on saltwater life were limited. As there were limited data for vanadium, it was not possible to recommend standards based on dissolved concentrations or separate standards for migratory fish. With regard to the latter, it may be necessary to base judgement of any risk in applying the EQS on knowledge of local risks and circumstances.

Tributyltin	µg l-1	0.002 MT[2]	The standards for TBT weres tentative to reflect a combination of the lack of environmental data, toxicity data or data relating to the behaviour of organotins in the environment.
Triphenyltin (and its derivatives)	µg l-1	0.008 MT[2]	The standards for TPT were tentative to reflect a combination of the lack of environmental and toxicity data or data relating to the behaviour of organotins in the environment.
PCSDs	µg l-1	0.05 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Cyfluthrin	µg /l	0.001 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values
Sulcofuron	µg /l	25 PT[1]	As a consequence of the general paucity of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values. The data for sulcofuron suggested that embryonic stages for saltwater invertebrates could be more sensitive than freshwater species and, therefore, the EQS for the protection of marine life, derived from the freshwater value, may need to be lower.
Flucofuron	µg /l	1.0 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were based on freshwater values.
Permethrin	µg /l	0.01 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Atrazine and Simazine	µg /l	2 AA[2] 10 MAC[4]	The EQSs for the protection of saltwater life were proposed as combined atrazine/simazine to take account of the likely additive effects when present together in the environment.
Azinphos-methyl	µg /l	0.01AA[2] 0.04 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Dichlorvos	µg /l	0.04 AA 0.6 MAC[2]	Based on data for sensitive crustaceans
Endosulphan	µg /l	0.003 AA[2]	There is little evidence on the ultimate fate of endosulfan and its metabolites or degradation products in sediments and on any effects on freshwater benthic organisms. Consequently, it is possible that some sediment-dwelling organisms, such as crustaceans, may be at risk.
Fenitrothion	µg /l	0.01 AA[2] 0.25 MAC[4]	As there were limited data with which to derive EQSs to protect saltwater life, the freshwater values were adopted. However, the annual average for the protection of freshwater life may be unnecessarily stringent in view of the uncertainties associated with the acute toxicity data used in its derivation. The uncertainties exist because the original sources were unavailable for certain studies. Lack of confirmatory data existed in the published literature and data for warm water species were considered in the derivation.
Malathion	µg /l	0.02AA[2] ;0.5MAC[4]	It was recommended that further investigation for both field and laboratory conditions into the effects of malathion on crustaceans and insects and on UK *Gammarus* species, in particular, should be carried out.

Trifluralin	µg /l	0.1AA[2] 20 MAC[4]	None mentioned with regard to the annual mean.
4-chloro-3-methyl phenol	µg /l	40 AA[3] 200 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2-chlorophenol	µg /l	50 AA[3] 250 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4-dichlorophenol	µg /l	20 AA[3] 140 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4D (ester)	µg /l	1 AA[3] 10 MAC[4]	For the EQS proposed for 2,4-D esters, comparison of the data and derivation of standards were complicated by the number of esters and organisms for which studies were available. In addition, the toxicity of the esters may have been underestimated in some of the studies due to their hydrolysis. There were limited data on the toxicity of 2,4-D ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
2,4D	µg /l	40 AA[3] 200 MAC[4]	There were limited data on the toxicity of 2,4-D non-ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
1,1,1-trichloroethane	µg /l	100 AA[3] 1000 MAC[4]	The 1,1,1-TCA dataset available for freshwater species contained comparatively few studies where test concentrations were measured and, consequently, comparison of studies using measured concentrations vs. those using nominal values indicated that data from the latter type of study could be misleading.
1,1,2-trichloroethane	µg /l	300 AA[3] 3000 MAC[4]	For 1,1,2-TCA, few data were available on chronic toxicity to freshwater fish. There were limited data on the toxicity of 1,1,2-TCA to saltwater life and, consequently, the EQS to protect freshwater life was adopted.
Bentazone	µg /l	500 AA[3] 5000 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Benzene	µg /l	30 AA[3] 300 MAC[4]	Limited and uncertain chronic data available.
Biphenyl	µg /l	25 AA[3]	The data available for marine organisms were considered inadequate to derive an EQS for the protection of marine life. However, the reported studies for saltwater organisms indicate that the EQS for freshwater life will provide adequate protection.
Chloronitrotoluenes (CNTs)	µg /l	10 AA[3] 100 MAC[4]	The dataset used to derive the EQS to protect freshwater life was limited. Toxicity data were available for comparatively few species and there was limited information on the bioaccumulation potential of the isomers. There were few chronic studies available to allow the assessment of the long term impact of CNTs. There were no reliable data for the toxicity to or bioaccumulation of CNTs by saltwater species and, therefore, the EQSs proposed for freshwater life were adopted.
Demeton	µg /l	0.5 AA[3] 5 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.

Dimethoate	µg /l	1 AA[3]	The available data for marine organisms were considered inadequate to derive an EQS for the protection of marine life. Crustaceans were considered to be the most sensitive organisms, but more data are required to confirm this. In view of the uncertainties associated with the marine toxicity dataset, the freshwater EQS was adopted. This was based on the toxicity of dimethoate to insects. Although there are no marine insects, there is some evidence that marine organisms are more sensitive than their freshwater counterparts.
Linuron	µg /l	2 AA[3]	In view of the lack of data for saltwater life, the EQS proposed for the protection of freshwater life was adopted until further data become available.
Mecoprop	µg /l	20 AA[3] 200 MAC[4]	There were limited data relating to the toxicity of mecoprop to aquatic life. The dataset for saltwater life comprised data for one marine alga, a brackish invertebrate and a brackish fish. Consequently, the freshwater values were adopted until further data become available.
Naphthalene	µg /l	5 AA[3] 80 MAC[4]	Limited and uncertain chronic data available.
Toluene	µg /l	40 AA[3] 400 MAC[4]	The dataset used to derive the EQS to protect saltwater life relied on static tests without analysis of exposure concentrations. Consequently, the derived values are considered tentative until further data from flow-though tests with analysed concentrations become available.
Triazophos	µg /l	0.005 AA[3] 0.5 MAC[4]	The dataset available for freshwater life was limited to a few studies on algae, crustaceans and fish. No information was available for the target organisms (insects), on different life-stages or on its bioaccumulation in aquatic organisms. There were no data on the toxicity or bioaccumulation of triazophos in saltwater organisms. Consequently, the EQSs to protect freshwater life were adopted until further data become available.
Xylene	µg /l	30 AA[3] 300 MAC[4]	Limited information available. Freshwater data used to § back up§ the standards.

Notes

Substances are listed in the order of publication of Directives.

A annual mean
D dissolved concentration, ie usually involving filtration through a 0.45-µm membrane filter before analysis
T total concentration (ie without filtration)
µg/ l micrograms per litre
AA standard defined as annual average
MAC maximum concentration
[1] DoE Circular in 1989 (Statutory standard)
[2] Statutory Instrument 1997 (Statutory standard)
[3] Statutory Instrument 1998 (Statutory standard)
[4] Non- statutory standard
[5] revised standards have been proposed but are not statutory

Annex 3. Quality Standards Stipulated In The Shellfish Waters Directive

(from Cole *et al.,* 1999)

Parameter	Unit	G	I
A. GENERAL PHYSIO-CHEMICAL PARAMETERS			
Colour			(a)
Dissolved oxygen	% sat	>80 T95	>70 TAA(b)
pH			7-9 T75
Salinity	g/kg	12-38 T95	40 T95(c)
Suspended solids			(d)
Tainting substances			(e)
Temperature		(f)	
B. METALS AND INORGANIC ANIONS			
Arsenic		(g)	(h)
Cadmium		(g)	(h)
Chromium		(g)	(h)
Copper		(g)	(h)
Lead		(g)	(h)
Mercury		(g)	(h)
Nickel		(g)	(h)
Silver		(g)	(h)
Zinc		(g)	(h)
C. ORGANIC SUBSTANCES			
Hydrocarbons			(i)
Organohalogens		(g)	(h)
D. MICROBIOLOGICAL PARAMETER			
Faecal coliforms	per 100 ml	300 T75(j)	

Notes:

G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)
D dissolved concentration ie usually involving filtration through a 0.45-µm membrane filter before analysis
AA standard defined as annual average
75 standard defined as 75-percentile
95 standard defined as 95-percentile
MA maximum allowable concentration
Pt/l concentration of platinum (Pt) determined photometrically on the Platinum/Cobalt scale as a measure of colour in water

[a]A discharge affecting shellfish waters must not cause an increase in colouration of more than 10 mg Pt/l compared to the waters not so affected (waters filtered in both cases). This standard is expressed as a 75-percentile.
[b]If an individual result indicates a value lower than 70% of saturation, the measurement must be repeated. Concentrations below 60% of saturation are not allowed, unless there are no harmful consequences for the development of shellfish colonies.
[c]A discharge affecting shellfish waters must not cause an increase in salinity of more than 10% compared to the water not so affected. This standard is expressed as a 75-percentile.
[d]A discharge affecting shellfish waters must not cause an increase in the concentration of suspended solids by more than 30% compared to the water not so affected. This standard is expressed as a 75-percentile.
[e]The concentration of substances affecting the taste of shellfish must be lower than that liable to impair the taste of the shellfish.
[f]A discharge affecting shellfish waters must not cause an increase in temperature of more than 2 °C compared to the water not so affected. This standard is expressed as a 75-percentile.
[g]The concentration of this substance or group of substances in shellfish flesh must be so limited that it contributes to the high quality of shellfish products.
[h]The concentration of this substance or group of substances in water or in shellfish flesh must not exceed a level which gives rise to harmful effects in the shellfish or their larvae. Synergistic effects must also be taken into account in the case of metal ions.
[i]Hydrocarbons must not be present in water in such quantities as to produce a visible film on the surface of the water and/or a deposit on the shellfish, or to have harmful effects on the shellfish.
[j]In shellfish flesh and intervalvular fluid. However, pending the adoption of a directive on the protection of consumers of shellfish products, it is essential that this value be observed in waters from which shellfish are taken for direct human consumption.

Annex 4. Bathing Waters Quality Standards

Quality standards for fresh and saline waters stipulated in the Bathing Waters Directive (from Cole *et al.,* 1999)

Parameter	Unit	G	I
A. INORGANIC SUBSTANCES AND GENERAL PHYSICO-CHEMICAL PARAMETERS			
Colour			(a, b)
Copper	mgCu/l		
Dissolved oxygen	% saturation	80-120 T90	
pH			6-9 T95(b)
Turbidity	Secchi depth m	>2 T90	>1 T95(b)
B. ORGANIC SUBSTANCES			
Floating waste(c)		(d)	
Hydrocarbons	µg l-1	300 T90(e)	(f)
Phenols	µgC_6H_5OH	5 T90(e)	50 T95(e)
Surfactants(g)	µg l-1 as lauryl sulphate	300 T90(e)	(k)
Tarry residues		(d)	
C. MICROBIOLOGICAL PARAMETERS			
Faecal coliforms	per 100 ml	100 T80	2 000 T95
Total coliforms	per 100 ml	500 T80	10 000 T95
Faecal streptococi	per 100 ml	100 T90	
Salmonella	per 1 l		0 T95
Entero viruses	PFU/10 l		0 T95

Notes

G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)80 standard defined as 80-percentile*
90 standard defined as 90-percentile*
95 standard defined as 95-percentile*
It is further stipulated that of the 20, 10 or 5% of samples from designated waters which exceed the standard, none should do so by more than 50% (except for microbiological parameters, pH and dissolved oxygen) and that "consecutive water samples taken at statistically suitable intervals do not deviate from the relevant parametric values" (Article 5 of CEC 1976).
[a]No abnormal change in colour
[b]May be waived in the event of exceptional weather or geographical conditions
[c]Defined as wood, plastic articles, bottles, containers of glass, plastic, rubber or any other substance
[d]Should be absent.
[e]Applies to non-routine sampling prompted by visual or olfactory evidence of the presence of he substance
[f]There should be no film visible on the surface and no odour
[g]Reacting with methylene blue
[k]There should be no lasting foam

Annex 5. Sediment Quality Guidelines

Interim marine sediment quality guidelines (ISQGs) and probable effect levels (PELs; dry weight) [1]: metals and organics (from Cole *et al.,* 1999)

Substance	ISQG	PEL
Inorganic ($mgkg^{-1}$)		
Arsenic	7.24	41.6
Cadmium	0.7	4.2
Chromium	52.3	160
Copper	18.7	108
Lead	30.2	112
Mercury	0.13	0.70
Zinc	124	271
Organic (μgkg^{-1})		
Acenaphthene	6.71	88.9
Acenaphthylene	5.87	128
Anthracene	46.9	245
Aroclor 1254	63.3	709
Benz(a)anthracene	74.8	693
Benzo(a)pyrene	88.8	763
Chlordane	2.26	4.79
Chrysene	108	846
DDD[2]	1.22	7.81
DDE[2]	2.07	374
DDT[2]	1.19	4.77
Dibenz(a,h)anthracene	6.22	135
Dieldrin	0.71	4.30
Endrin	2.673	62.4[4]
Fluoranthene	113	1 494
Fluorene	21.2	144
Heptachlor epoxide	0.60[3]	2.74[4]
Lindane	0.32	0.99
2-Methylnaphthalene	20.2	201
Naphthalene	34.6	391
PCBs, Total	21.5	189
Phenanthrene	86.7	544
Pyrene	153	1 398
Toxaphene	1.5[3]	nd[5]

[1] from CCME, (1999)
[2] Sum of *p,p´* and *o,p´* isomers.
[3] Provisional; adoption of freshwater ISQG.
[4] Provisional; adoption of freshwater PEL.
[5] No PEL derived.

Annex 6. Examples of Recommended Biological Monitoring Techniques

Immunotoxicity Assay – this 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 (Raftos and Hutchinson, 1995).
Metallothionein induction and associated changes in metal metabolism are an obvious choice in the case of metal discharges.
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 metals and TBT in Fal biota to establish their adaptation to contamination levels. Mapping (and re-mapping) of the genetic composition of tolerant populations (*Hediste* and others) should be considered in this respect and would add an interesting temporal dimension to the anticipated 'recovery phase' following water-treatment measures at Wheal Jane. This may be relevant to the achievement of Favourable Condition Status (under the Habitats Directive) and to standstill requirements of the Dangerous Substances Directive).
Toxicity Studies - 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 (e.g. those associated with maerl) and perhaps to compare responses in Fal biota with those elsewhere to look for signs of adaptation.
Multivariate Statistical Analysis of biota and environmental variables in order to examine spatial and temporal trends in communities in relation to contaminants.

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. A Summary of Water Company Improvements in the Fal and Helford cSAC

(Information source - EA)

Asset Management Plans (AMPs) timescale

	Timescale
AMP1	1990 – 1995
AMP2	1995 – 2000
AMP3	2000 - 2005

The provision of secondary treatment was identified for Falmouth in AMP1. This was consequently deferred by the government. The sewage treatment plant now provides secondary treatment and UV disinfection and was commissioned in 2001.

The Falmouth treatment works treat flows from the sub-catchments of Middlepoint, Swanvale, Pennance Mill and Penryn. The continuous foul discharges at Middlepoint, Pennance Point and Maenporth have therefore been discontinued and now act as CSOs (combined sewer overflows).

A storage tunnel was provided upstream of the works and this is partly used to regulate discharges from the works to occur mainly on the ebb tide. In addition to this a new storage tank of 6000 m^3 at the sewage treatment works was commissioned in 2002.

All the CSOs in the Middlepoint catchment were provided with screening in accordance with the AMP2 guidelines.

The flows from Flushing which previously discharged via an old septic tank direct to the estuary are now transferred across the estuary into the Middlepoint catchment for treatment at the Falmouth works.

Storage has been provided at Devoran pumping station.

23 overflows within the Truro catchment have been targeted for improvements under AMP3. Six of the overflows initially targeted have proved not to spill significantly on extreme rainfall events and these have been removed from the upgrade list and replaced with overflows not previously on the list but understood to spill frequently.

Titles in the current series of Site Characterisations

Characterisation of the South West European Marine Sites: **The Fal and Helford cSAC**. Marine Biological Association of the United Kingdom occasional publication No. 8. pp 160. (2003)

Characterisation of the South West European Marine Sites: **Plymouth Sound and Estuaries cSAC, SPA**. Marine Biological Association of the United Kingdom occasional publication No. 9. pp 202. (2003)

Characterisation of the South West European Marine Sites: **The Exe Estuary SPA** Marine Biological Association of the United Kingdom occasional publication No. 10. pp 151. (2003)

Characterisation of the South West European Marine Sites: **Chesil and the Fleet cSAC, SPA**. Marine Biological Association of the United Kingdom occasional publication No. 11. pp 154. (2003)

Characterisation of the South West European Marine Sites: **Poole Harbour SPA**. Marine Biological Association of the United Kingdom occasional publication No. 12. pp 164 (2003)

Characterisation of the South West European Marine Sites: **The Severn Estuary pSAC, SPA.** Marine Biological Association of the United Kingdom occasional publication No.13. pp 206. (2003)

Characterisation of the South West European Marine Sites: **Summary Report**. Marine Biological Association of the United Kingdom occasional publication No.14. pp 112 (2003)

The Carnon River
Photograph Martin Rule

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