Characterisation of European Marine Sites

The Severn Estuary

(possible) Special Area of Conservation
Special Protection Area

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Site Characterisation of the South West European Marine Sites

Severn Estuary pSAC, SPA

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by the Plymouth Marine Science Partnership

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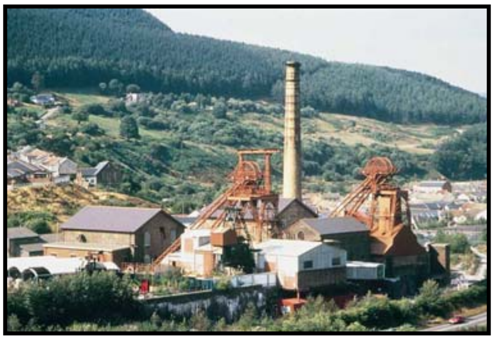
Plate 1: Some of the operations which may cause disturbance or deterioration to key interest features of the Severn Estuary pSAC, SPA

1-3: Process Industries at Severnside



5: Sewage Treatment

4: The Rhondda valley - historically the epicentre of coal mining in South Wales

6: (right) Agricultural run-off

1&5: Ian Britton, Freefoto.com **2 &3:** EA
5: Wales Tourist board photo gallery **6:** MBA

Plate 2: Some of the Interest Features and habitats of the Severn Estuary pSAC, SPA

1: Severn Mudflats

2: Saltpans, Bridgwater Bay

3: (right) Bewicks Swan *Cygnus columbianus bewickii*

4: Twaite shad *Alosa fallax*

5: Allis shad *Allosa alosa*

1: Dr Peter French, University of London **2**: English Nature (www.oursouthwest.com)
3: Tim Watts **4:** fishing-in-kite-country.co.uk **5:** Willy Van Cammeren

1. EXECUTIVE SUMMARY

The Environment Agency (EA), English Nature (EN) and Countryside Council for Wales (CCW) 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 Severn Estuary possible Special Area of Conservation (pSAC), Special Protection Area (SPA).

This project, undertaken by the Plymouth Marine Science Partnership (PMSP), (comprising Marine Biological Association (MBA), University of Plymouth (UoP) and Plymouth Marine Laboratory (PML)), has two main objectives. Firstly, to characterise the site in terms of water quality, 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). Some of the key findings are:

The unique physical features of the Estuary dominate sediment distributions and composition, which are, in turn, the main components governing distributions of organisms. Much of the sub-tidal Severn mud is impoverished in terms of biota and even some sandy areas may be impoverished because of extreme mobility of silts at spring tides. Because physical conditions dominate, for the majority of biological communities, there is little unequivocal evidence of additional impact due to contaminants across the Estuary as a whole. Individual populations may have been impacted, close to major discharges, though most evidence is correlative rather than mechanistic.

Metals have traditionally been a major concern in the Estuary, because of smelting and other metal industries. Cd, As, Cr and Hg probably originate mainly from industrial sources around the Estuary whilst Cu, Zn and Ni are predominantly riverine in origin with additional inputs from trade and sewage discharges. Concentrations in sediments are commonly above interim sediment quality guidelines over much of the Estuary, but only occasionally exceed probable effects levels. Bioaccumulation of metals occurs widely in invertebrates, though the ecological significance is still uncertain.

Hydrocarbon compounds, including PAHs, are present, locally, in elevated concentrations. Sources include a combination of fossil fuel combustion, shipping, urban run-off, STW and various point-source and diffuse discharges from industrialised areas. Local coal and oil bearing strata also contribute, though the principal component appears to be anthropogenic. Moderately high levels of PAHs are present in sediments from much of the system and for some individual PAHs concentrations occasionally exceed probable effects levels. This may have consequences for benthic fish species and invertebrates.

A number of synthetic organic compounds may be present, locally, in elevated concentrations. In the past, monitoring has indicated several pesticide and herbicide

EQS exceedences in rivers entering the Severn Estuary. It would appear that currently, exceedences in the Severn Estuary occur rarely, if at all. However, there is very little robust data with which to characterise the threats to the European marine site from the majority of these compounds. If such threats do occur, these would be expected to be largely localised issues. Only occasionally are elevated levels indicated (e.g. DDT, dieldrin and endrin in sediments; DDT, lindane and dieldrin in eels; dieldrin, triazines and endosulphan in water). Rivers probably introduce the largest loadings (particularly those from catchment areas with intensive agriculture) but there are indications that high pesticide concentrations sometimes occur in discharges. PCB contamination is related to past industrial usage and production (at Newport on the Usk). Sediments close to river mouths and docks are classified as being 'heavily contaminated' but decrease at offshore sites. PCBs in eel tissue are elevated in the lower reaches of industrialised catchments in south Wales. Although other fish from Cardigan Bay and the Severn Estuary do not appear to be seriously contaminated, high concentrations have been found in marine mammals from these locations.

Data on organotins are surprisingly scarce: however, analysis of dredge spoils in and around the pSAC suggest there may be localised reservoirs of TBT near major conurbations such as Newport and Cardiff, and presumably elsewhere. There are indications of inputs from a number of outfalls thoughout the region.

Nutrient levels and loadings in the Severn Estuary are considered significant in UK terms. However, high turbidity means that algal productivity is generally low, except in localised hotspots. Eutrophication is therefore not a major issue within the pSAC. Intermittent oxygen sags occur in low salinity regions of the Severn and in some of the principal rivers feeding the Estuary. These probably originate from the high densities of suspendable solids and associated particulate organic matter, perhaps enhanced by discharge outfalls.

Anomalously high concentrations of tritium in sediments and benthic biota in the Cardiff area have been attributed primarily to an industrial discharge. Remedial action has been initiated and an application for a revised discharge has been made. Whilst not a threat to humans, the bioavailability, assimilation pathways and effects of organically-bound tritium on marine life require further investigation.

Evidence from loadings, concentration data and EQS compliance frequencies indicates continuing improvements in water quality for the major contaminants (e.g. Cd). The balance of evidence indicates that these may coincide with biological improvements, though the extent to which these events are linked remains uncertain. The persistence and behaviour of sediment-bound contaminants, and their potential combined effects, gives rise to the greatest uncertainty about recovery. The purported decline in eel and Twaite shad populations (and several other fish species) appears to run contrary to the general perception of recovery. It has yet to be established whether the causes are related to water quality or other factors within, or even outside, the pSAC (e.g. fishing pressures, natural variability of stocks).

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

CONTENTS

2. INTRODUCTION

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

To achieve this goal, specific objectives were:

- To prepare comprehensive reference lists of previous investigations and existing datasets, including published research and unpublished reports, relevant to an assessment of the effects of water quality on the marine sites and interest features identified.
- To review the existing information 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 there is 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 and Countryside Council for Wales have provided draft advice on the Severn Estuary SPA site, given under Regulation 33(2) of the Conservation Regulations 1994 (English Nature, Countryside and Council for Wales, 2001). 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 and Countryside Council for Wales, 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 the opinion of EN & CCW may cause disturbance or deterioration to key interest features of the SPA. Toxic and non-toxic contamination are the principal threats considered in the current project. (Table adapted from draft guidance given by EN & CCW, 2001)*

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

***Note: more recently, additional interest features have been submitted to the EU as reasons for recommendation as a pSAC. These include Estuaries, Reefs, Atlantic Salt meadows, subtidal Sandbanks and intertidal Mudflats and Sandflats. Allis shad *Alosa alosa,* Twaite shad *Alosa fallax,* river lamprey *Lampetra fluviatilis,* and sea lamprey *Petromyzon marinus* have also been added as interest features of the Severn Estuary (see Annex 1) (these features are probably subject to similar threats from toxic and non-toxic contamination as SPA features, above, and perhaps additional pressures).**

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 in their *Guidance for the Review of Environment Agency Permissions: Determining Relevant Permissions and 'significant effect'* (March 1999):

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

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

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

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

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

3. REFERENCE LISTS AND SOURCES OF INFORMATION

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

- Unpublished reports and data-bases: Environment Agency, Joint Nature Conservancy Council (JNCC) Coastal Directories Reports (Sector 9), 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, which was not available.

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

Section 4 describes physical features which shape the characteristics of the site. The review of published information on contaminants is dealt with in **Section 5**, which contains sub-sections on threats from toxic contamination (5.1- e.g. metals, TBT, petrochemicals, pesticides, PCBs) and non-toxic contamination (5.2- e.g. nutrients, turbidity, dissolved oxygen). This section primarily incorporates reviews of chemical information. Studies on the ecology of the system are the focus of **Section 6**.

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

A brief description of modelling exercises of direct relevance to the environmental quality status of the pSAC is provided in **section 9**.

Concluding remarks (**section 10**) include a summary of evidence for impact in the Severn Estuary pSAC together with recommendations for future research requirements.

4. THE SITE: FEATURES AND THREATS

The Severn Estuary has been recommended as a possible Special Area of Conservation (pSAC) because it contains habitat types and/or species which are rare or threatened within a European context. Bridgwater Bay, Severn Estuary and Upper Severn Estuary are also notified SSSI's in the British context (although this does not include the sublittoral) and Ramsar sites – designated under the International Convention on wetlands of international importance, especially as waterfowl habitat (the Ramsar Convention). The Severn Estuary and the upper Severn Estuary are Special Protection Areas (SPAs) - designated under the European Commission Directive on the Conservation of Wild Birds (79/409/EEC). Bridgwater Bay and several sites along the River Severn above the Estuary are also Nature Reserves.

The Severn is the major river feeding the Estuary with a flow of some 10^{10} m^3 y^{-1}, accounting for approximately one quarter of the freshwater input, the remainder coming from several other rivers (some of which are described below) and a variety of man made inputs. Of a total freshwater flow of some 25,000,000 m^3 d^{-1}. Apte *et al* (1990) calculated that 800,000 m^3 d^{-1} came from sewage and 200,000 m^3 d^{-1} came from industrial effluents.

The River Severn is the longest river in Britain and drains over a clay plain taking tribute from several English and Welsh counties. From its traceable source at 610m on Plynlimon in the mountains of mid-Wales, the river follows a circuitous route for over 200 miles passing through the lower Midlands, through Shropshire and into Gloucestershire before reaching the Bristol Channel and the sea. With the exception of urban centres such as Shrewsbury, Bewdley and Worcester which lie along its banks, the River Severn has a mainly rural catchment area. At high Spring tides the Severn may be tidal as far as Upper Lode Lock (Tewkesbury). From the city of Gloucester, the Severn widens as it occupies the centre of the huge shallow valley known as the Severn Estuary, separating England from Wales. The upper Severn includes an extensive area of mudflats and sandflats bordered by saltmarsh which grades through to neutral pasture. It supports nationally and internationally important numbers of overwintering and passage migrant waders and wildfowl. The Estuary itself is underlain by carboniferous limestone, which forms the islands of Steep Holm and Flat Holm as well as headlands along the Estuary. The islands, together with Denny Island, SW of the Severn Bridge, the intertidal mudflats and wetlands of the Estuary provide a haven for wild birds (Severn Estuary Conservation Group, 1978).

The Severn is Britain's second largest Estuary with an area of 557 square kilometres including an intertidal area of $100km^2$. When its seaward extension, the Bristol Channel, is included the intertidal habitat of mudflats, sand banks, rocky platforms and saltmarsh is one of the largest and most important in Britain, covering 2000 km^2. The Severn Estuary experiences the second highest tide anywhere in the world, with a range which can be in excess of 14.5 metres. A high proportion of the Estuary is subtidal with a diversity of aquatic estuarine communities present that include the only extensive subtidal *Sabellaria alveolata* reef in Britain. The intertidal flats and rock platforms support a wide variety of invertebrate species (Buck 1997). There are large fringes of saltmarsh where *Spartina* is abundant, and a *Zostera* bed in the

Estuary near the maximum turbidity zone. This bed has been monitored in recent years in relation to the impacts of the Second Severn Crossing. Sediment accretion around the cofferdam for the Second Severn Crossing is reported to have caused a decrease in the area of the *Zostera* bed in the Severn Estuary pSAC.

The Severn bore, which is a tidal wave, occurs in the lower reaches of the Severn during high tides. For a bore to form, a considerable rise in tide is needed in a converging channel with a rising bed, forming a funnel shape. These conditions occur in the lower reaches of the River Severn, and under certain circumstances the bore may reach two metres in height. The average speed of the bore is approximately 16 km/h. In the late 20th century, a scheme to harness the immense tidal power was conceived – the Severn Barrage. The possibility of a barrage pre-empted a whole host of studies on the potential effects on tidal regime, sedimentation, water quality and ecology of the Severn, but the project was shelved in 1994 due to insurmountable cost problems. Recently however, the scheme has been reconsidered[1] and was included in the recent Energy Review (Cabinet Office 2002).

This immense tidal range and classic funnel shape make the Severn Estuary unique in Britain and rare world-wide. Large tidal-currents are a dominating feature providing a mechanism for transport of particles up to sand-size (moving as suspended solids or mobile bed-load). The associated variable frictional stresses result in variations in bed-types (and associated contaminant loadings), despite the ever-present turbidity of the water. In turn, the relatively sharp divisions between muddy, sandy and rocky areas, dominate the distribution of benthic macrofaunal communities (Warwick and Uncles, 1980; Boyden and Little, 1973; Dyer, 1984.)

Bridgwater Bay is on the eastern shore of the Severn Estuary near the mouth of the River Parrett. The Bay occupies the sweeping arc of coastline between the wave-cut platform of Jurassic Blue Lias at the northern tip of the Quantock Hills and the cliffs of Carboniferous Dolomites and Limestone at Brean Down which project into the Severn Estuary and provide some degree of protection from the erosive tidal currents. This has allowed the deposition of an extensive area of intertidal mud. Hinkley Point Power Station intakes are at the western end of the 18 kilometre-square Stert and Berrow intertidal flats. The point is an area of intercalated shale, slate and limestone that has been eroded into a series of ridges parallel to the shore. The sublittoral substrate is highly mobile, nearly liquid mud with some areas of sand waves and isolated reefs of agglomerated *Sabellaria* worm tubes. The intertidal area is firmer sandy mud. At Hinkley Point, between 1980 and 1996, the salinity ranged from 22 to 33%, depending on the freshwater flow from the rivers, and the sea-temperature from 2 to 23°C. The site comprises extensive areas of intertidal mudflats, saltmarsh and shingle beach that in places is backed by grazing marsh with freshwater ditches. It supports internationally and nationally important numbers of overwintering and passage migrant waders and wildfowl.

[1] http://www.bbc.co.uk/bristol/content/archive/052000/25/news2/barrage.shtml

The pSAC extends from just west of Hinckley Point on the east side to Lavernock just west of Cardiff on the west side (see figure 1). On the east side of the Estuary, there are numerous rivers and streams flowing into the Estuary, notably the upper Frome, lower Frome (Berkeley) Pill, the upper Avon, lower Avon, Kenn, Banwell, Yeo, Axe, Brue, Parrett, Washford, the Pill (E of Watchet) and Donniford Stream. On the northern banks of the Estuary (mostly in Wales) there are many rivers and streams which flow into the Estuary. These include the Rivers Leadon, Wye, Usk, Ebbw, Rhymney, Taff, Ely, Cadoxton, and the Col huw. On both sides of the Severn Estuary there are also numerous ditches, drainage channels and streams, known locally as rhynes, or reens, which flow into the Estuary and drain the Severn Estuary Levels.

The Levels are all those lands lying almost at sea level on the edge of the inner Bristol Channel as far upstream as Gloucester. They comprise some 850km^2, straddling the former counties of Gloucestershire, Avon, Gwent and Somerset. The landscape is characterised by flat meadows and scattered arable fields, defined by water-filled ditches and crossed by streams and lanes. The geological deposit - known as the Wentlooge Formation - that was formed beneath the Levels is a series of alternating layers or beds of mainly blue-green silt and brown-black peat that total l0-15 metres thick in most places. The deposit represents tidal mudflats and different types of tidal-freshwater marsh. Locally, the beds are interrupted by silted-up tidal creeks, called palaeochannels (Allen 2000). Because of coastal erosion, the layers are visible at countless places on the shores of the Levels.

The Somerset Levels (around Bridgwater Bay) are underlain by Triassic rocks. The most common of these rocks is the Mercia Mudstone (previously known as Keuper Marl). Although the Mercia Mudstone forms a thick and extensive deposit, there has also been an accumulation of Quaternary alluvium, peats and marine clays which has created the low-lying fenland landscape during the last 10,000 years. The thick blanket of Quaternary deposits is up to 35 metres deep in places. It is only broken by the more resistant rocks, such as the Lias outlier at Brent Knoll, and Mercia Mudstone around Sedgemoor.

Other Principal rivers:

The Yeovil Scarplands sweep in an arc from the Mendip Hills around the southern edge of Somerset Levels and Moors to the edge of the Blackdowns. Rivers including the Brue, Parrett and Yeo drain from the higher ground of the Scarplands cutting an intricate pattern of irregular hills and valleys which open out to the moorland basins. Their catchments have a mixed geology of lower and middle Coal Measures, Carboniferous oolitic limestones and Triassic marls and sandstones. Land use is predominantly rural with some urbanisation. Many of the rivers are used by elver, eels and salmonids as part of their migratory routes, and some of the rivers (Wye, Usk) are cSACs in their own right.

The River Parrett rises in Dorset and runs northwest before entering the Severn Estuary at Bridgwater Bay. Its catchment is principally agricultural and includes the southern half of the Somerset Levels and Moors, the largest area of wet grassland in the UK. The total catchment of the River Parrett and its tributaries cover some 1,665km^2, of which some 414km^2 is within the catchment of the River Tone.

Bridgwater, Taunton and Yeovil are the main towns in the catchment area. The river supports a commercial eel fishery which is small but the elver fishery is the second most productive in England. The area is also important for willow industry and 300 acres near Stoke St Gregory is dedicated to willow, or withy growing.

The (upper) Frome, which rises at Brimpsfield, southeast of Gloucester, passes Stroud, where it is called the Stroud River, and joins the Severn at Framilode passage.

The (lower) Frome rises in Gloucestershire at Doddington (near Tetbury, in the Cotswolds) and follows a winding southwesterly course towards Bristol where it joins the Avon. The Frome Valley between Stapleton and Frenchay is a conservation area. The river drains 68 square miles and is about 20 miles long from source to mouth, which now is part of the Floating Harbour at Bristol city centre. In Bristol, parts of the river, where it runs through built up areas, have been culverted.

The (upper) Avon, winds through Warwickshire, and part of Worcestershire, it enters Gloucestershire about three miles above Tewkesbury where it joins the Severn, the fish of this river include roach, dace, bleak, carp, bream, and eels.

The (lower) Avon drains around 857 square miles, it rises among the hills of North Wiltshire and at Clifton, cuts a deep gorge through the Failand Ridge which is composed of patches of Pennant Sandstone containing coal measures, pockets of Magnesian Limestone (Dolomite or Dolomitic Conglomerate) and igneous material. The area was mined for coal until the early 1800s. In Bristol the Avon receives the waters of the lower Frome, and, at about five miles west of Bristol joins the Severn at Avonmouth.

Avonmouth is a centre for major industries including pharmaceutical manufacture, zinc smelting, chemical recoveries, power production (from both gas, and clinical waste and tyres), chemical production and gas works. Many of these process industries discharge either directly into the Severn, or into the numerous watercourses that run into the Severn.

The River Axe rises on Mendips and flows 25m NW to join the Severn near Weston-super-Mare. There are trout and roach in upper reaches and tributaries.

The River Wye is one of the largest rivers in Britain and flows 156 miles before entering the Severn Estuary at Chepstow. Part of the Wye forms the border between England and Wales. Its major tributaries include the Monnow, Lugg, Irfon, Ithon and Llynfi. Like the Severn, the Wye also rises on the Plynlimon mountains at 741m AOD (Above Ordnance Datum) and flows over old red sandstone, which has been deeply incised by the river at many points exposing the underlying carboniferous limestone. The Wye passes through several towns including Rhayader, Builth Wells, Hay-on-Wye and Hereford before meeting the Severn Estuary at Chepstow. The total catchment area is 4136 km^2 and the population size of 226,000 is centred on the main towns. The waters of the Wye and its tributaries are reputed to be mostly unpolluted and support a salmon and trout fishery. The River Wye itself is a SAC in its own right, also an SSSI and one of the most important rivers in Britain in nature conservation terms. Much of the lower valley is an Area of Outstanding Natural Beauty (AONB).

The Wye river corridor supports a variety of plant communities, otters, water voles, several bat species, dippers, sandmartins, kingfishers and little ringed plovers. The biological quality of the river is generally good and supports several rare or scarce species including the mayfly *Potamanthus luteus*, the freshwater pearl mussel *Margaritifera margaritifera* and the native crayfish. The river also supports several rare species of non-aquatic invertebrates associated with gravel shoals.

The River Usk has its source high on the Black Mountain above the Usk Reservoir The upper and middle reaches of the Usk tumbles through the Brecon Beacons National Park. After passing Brecon and Crickhowell the river runs through the farmland of the broader valley beyond. Shortly after the attractive town of Usk the river becomes tidal and enters the Estuary at Newport, where it mixes with the waters of the River Severn in the Bristol Channel. The River Usk itself is a SAC in its own right, and is renowned as a salmon and wild brown trout fishery.

The South Wales Valley rivers rise in hilly terrain. They are relatively short, steep and drop back rapidly after spates. Many of them flow over the South Wales Coalfield, a major downfold in which some 5000m of Carboniferous coal measures are preserved. Industrial exploitation of the coalfield valleys began in earnest in the mid 1800s and with it the associated urban development. The development of the South Wales coalfield was initially aimed at supplying fuel to the iron manufacturing industry, which had developed in the late 18th century based on ore deposits in the north-east. However, the construction of canals linking the iron and coal-producing areas with the south coast, and later the coming of the railways, transformed the coal mining into a major industry. By the 1840s coal-mining had overtaken the iron industry. Mining in south Wales began to decline during the Great Depression of the 1930s, but the industry received a temporary boost during World War II. After 1945 falling demand for coal, the geological problems of the Welsh coalfields, and growing competition from cheaper producers overseas led to the closure of 115 mines. By the early 1990s, only five mines were still operating and today the mining of Welsh coal has all but ceased with mines capped and nearly all of the waste or "slag" heaps levelled or landscaped.

Slate-mining was also an important sector of the economy during the 18th and early 19th centuries, and slate is still mined in some places. Granite is also mined, and there are small gold mines in mid-Wales.

The River Taff rises on the Old Red Sandstone escarpment of the Brecon Beacons at 975ft. above sea level and flows some forty miles through Merthyr Tydfil, Abercynon, Pontypridd and Cardiff before discharging into the Severn Estuary. A barrage, built by the former Cardiff Bay Development Corporation, has been constructed across the mouth of Cardiff Bay, effectively impounding the rivers Taff and Ely and creating a largely freshwater lake. The barrage has a fish pass that is intended to allow salmon and sea trout to access the rivers.

The River Ely rises just to the north of Tonyrefail and flows south-east through Talbot Green, Pontyclun and Miskin before veering to the east past Peterson-super-Ely, through St Fagans and into Cardiff Bay at Llandough, Cardiff. Salmon and sea trout have also been making a steady return to the Ely since the late 1980s and there also roach in the lower reaches.

The River Rhymney rises two miles north of Rhymney Reservoir and flows south through New Tredegar, Bargoed, Ystrad Mynach and Caerphilly before veering east to join the Severn Estuary in East Cardiff. This river holds trout, grayling, and the occasional sea trout.

The River Ebbw rises at Llangynidr and Carno Reservoirs on Llangynidr Mountain, and flows 30 miles before flowing into the Severn Estuary between the mouths of the River Rhymney and the River Usk (below Newport Docks). En route it passes collieries and steel making sites, now mainly defunct, which in the past virtually killed off all life in the river. The Ebbw has recently experienced great improvements in water quality which has been reflected in the return of fish such as trout and sea trout.

Several canals also open into the Severn Estuary: The Gloucester and Berkeley canal was designed to form a short and safe passage for large vessels between Gloucester and the wider parts of the Severn. It is 17.5m long, and enters the Severn at Sharpness. The Stroudwater canal is approximately 7 miles long and runs from near Stroud to the Severn at Framilode. This canal used to link with the Thames and Severn canal by means of a junction, as part of a transport system for cloth manufactured in local mills. The Hereford and Gloucester canal, intended to open a communication by water between Hereford and London enters Gloucestershire in the north and joins the western channel of the Severn at Gloucester.

The boundaries of the Severn Estuary marine site are shown in figure 1. Maps of communities and features within the site can be found in greater detail elsewhere (English Nature & Countryside Council for Wales, 2001).

Figure 1: The Severn Estuary pSAC showing boundaries of marine site

Physical and chemical properties of the Severn Estuary are reviewed in a number of publications (e.g. Winters, 1973; Owens, 1994). The energetic tidal characteristics, as modified by the orientation and shape of the Bristol Channel, dominate many of the properties of the Estuary as a whole, and its community composition. In particular vertical stratification is less marked than in the Fal and Tamar, for example, whilst the very high turbidity impacts biota in a number of ways. High suspended solids loadings provide abundant surface area for microbial process whilst at the same time limiting light penetration and primary productivity. The highly energetic tidal regime strips mud from the rocky substrate and can lead to fluid mud patches extending over large areas, both factors that reduce colonisation by benthic invertebrates, restricting productive areas to more stable marginal areas. Thus, whilst discharges could result in impact, locally, it has been postulated that the aquatic system as a whole is not greatly affected by discharges, relative to the constraints exerted by natural forcing (Glover, 1984; Owens, 1984). Despite their patchy productivity, the extensive intertidal mudflats of the Severn provide feeding grounds for a great diversity and number of birds giving rise to its status as an SPA. It has been estimated that the Estuary supports up to 200,000 waders and 50,000 wildfowl (Ferns, 1984).

Discharges and activities of potential significance

The siting of a number of major industries in the area is partly based on the assumed extensive waste assimilation capacity of the Estuary. As discussed above, these industries include, or have included until recently, the smelters, incinerators, fertilizer and numerous other chemical plants in the Avonmouth area; coal and steel industry[1], paper mills, chemical and pharmaceutical manufacturers in south Wales; nuclear power plants at Hinkley, Berkeley and Oldbury. Sewage from the urban centres of Bristol, Gloucester, Newport and Cardiff add directly to the pollutant load as do domestic and agricultural sources in the large number of tributaries entering the tideway (notably the Avon, Usk and Parrett).

Not surprisingly, there are a large number of discharges into the Severn Estuary system of varying sizes. Siting of some of the more important (by volume) discharge consents impacting on the Severn Estuary are shown in figure 2. Annex 7 also includes broad estimates of some of the major loadings from selected discharges (from Bird, 2002).

The list of potential chemical pressures is substantial and includes metals, organometals (e.g.TBT), hydrocarbons, nutrients, solvents, mineral acids, biocides, fungicides, flame retardants, PCBs, pesticides and radionuclides. The decline in heavy industry and introduction of pollution control in the later part of the 20th century however, has seen a general downward trend in inputs. Improvements to treatment of wastes are an ongoing phenomenon and where recent schemes have been introduced or are in progress these are described to the best of our knowledge (see summary of water company improvements, annex 8). It should be noted however that some of the data for discharges may not be based on the very latest situation. In the Avonmouth area, for example, some of the major industries are responsible for self-monitoring under Integrated Pollution Control (IPC) and data are not available in the EA database supplied. A recent overview of pollution management in the Avonmouth complex describes some of the latest improvements by industry (Environment Agency, 2002).

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[1] Many collieries and major steel making sites in south Wales have now ceased production, resulting in predictable improvement to the quality of local rivers such as the Ebbw (n.b. closure of the Corus plant at Ebbw Vale). Corus (formerly BSC) Llanwern also stopped steel and coke production in 2001 and 2002, respectively, resulting in potentially substantial reductions in metals and PAHs, and there have been reduced coking operations at the Corus plant in Port Talbot. Though the latter is seaward of the SAC, tidal flow may still direct a portion of the load towards the Severn marine site.

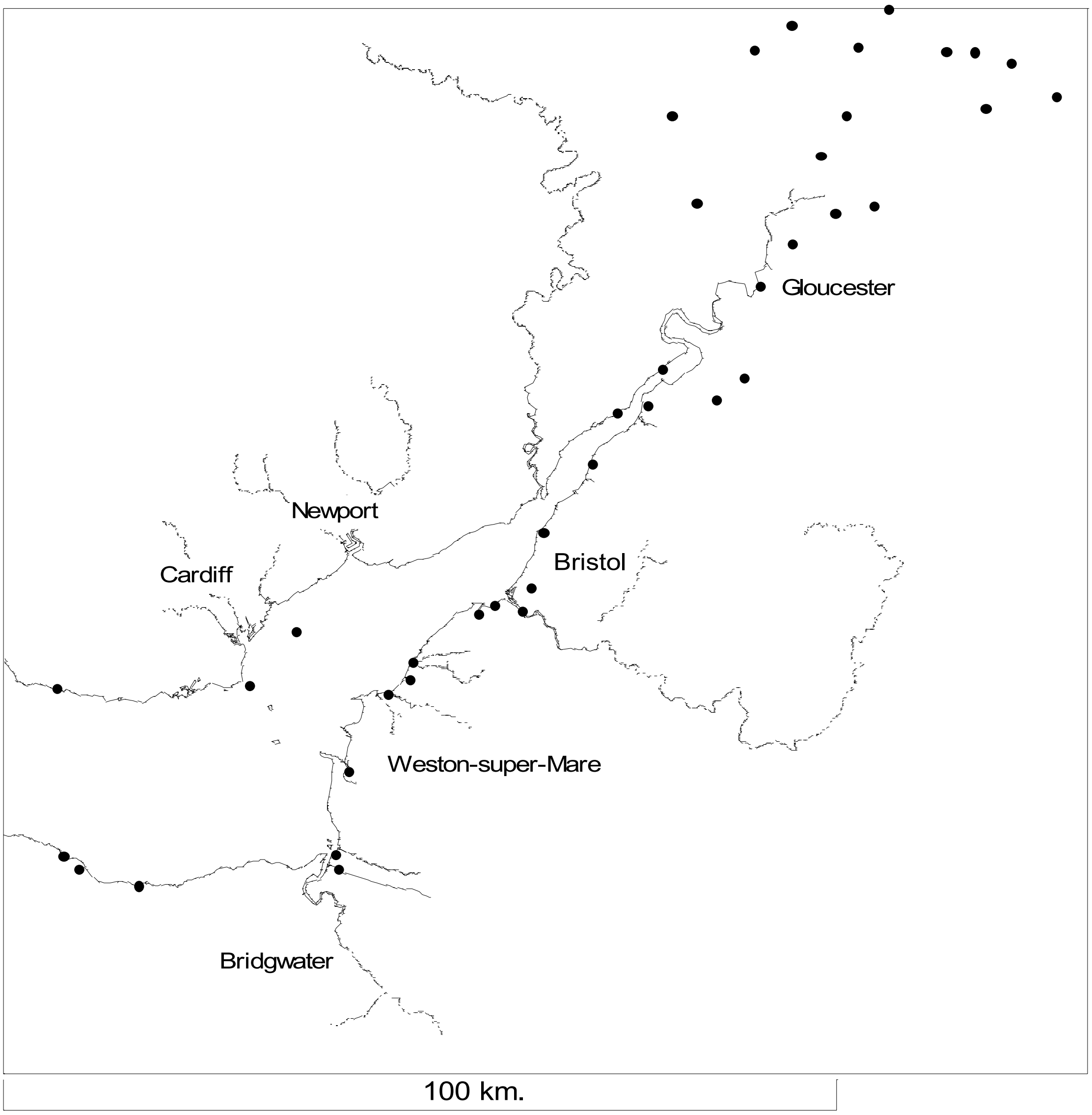

Figure 2. Locations of some of the larger discharge consents to the Severn Estuary system. Consents for the discharge of sewage, based on Dry Weather Flow (values >1000 m^3/d). From data supplied by the Environment Agency (Wales, South West and Midlands Regions).

NB No distinction has been made between continuous and intermittent discharges. Details of specific discharges should be clarified with the Environment Agency.

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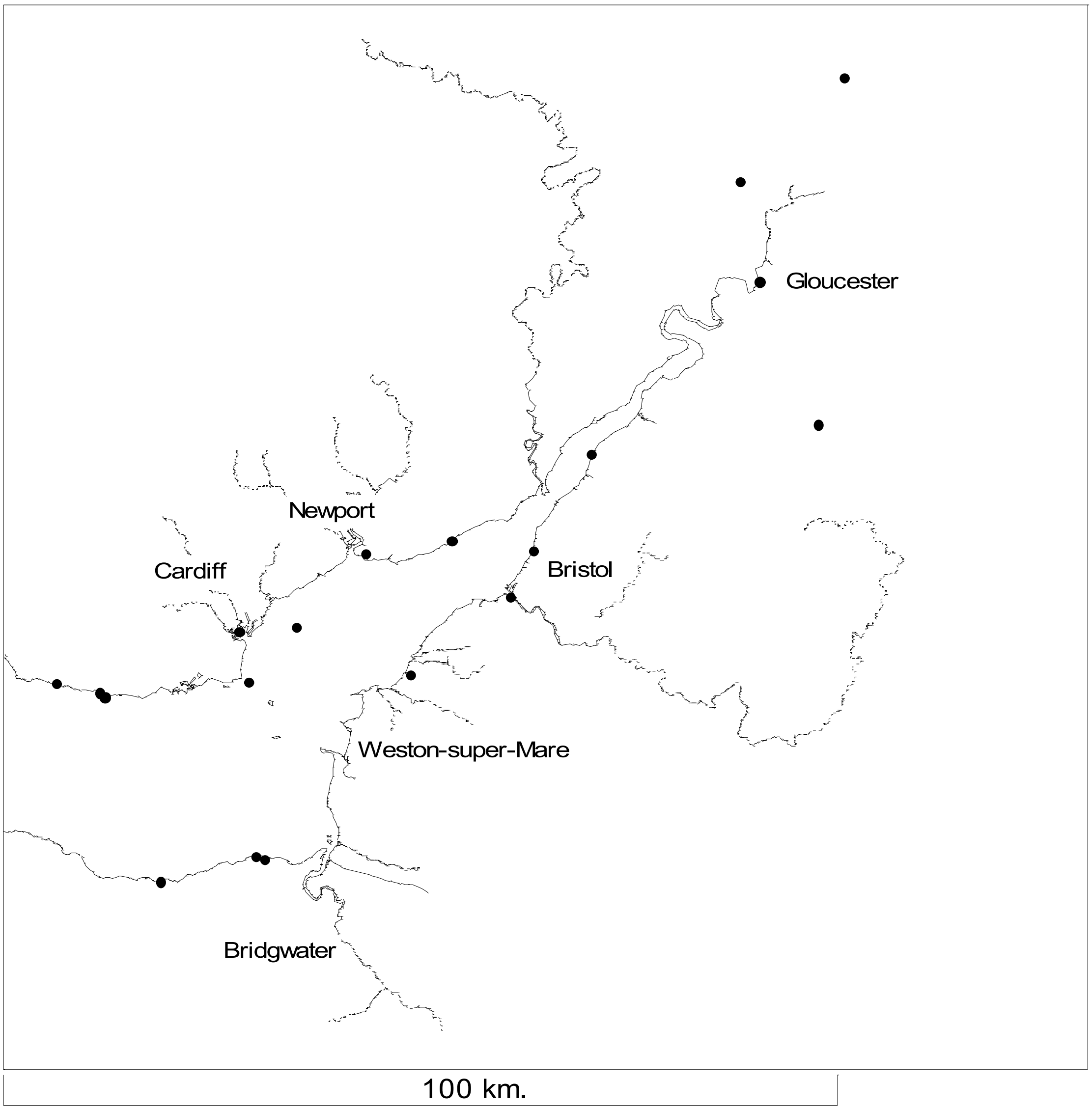


Figure 2 [……..Cont.] . Locations of some of the larger discharge consents to the Severn Estuary system. Trade consents, and miscellaneous sources of effluents, expressed as Maximum Daily Flows (values >500 m^3/d). From data supplied by the Environment Agency (Wales, South West and Midlands Regions).

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 ON CONTAMINANTS

Overview of contaminant loadings and sources

The Severn Estuary Joint Committee (SEJC) was set up in 1974 to manage the Estuary and has been served by numerous working parties and independent studies which provide appraisals of water quality. One of the earliest was an attempt to estimate inputs to the system of metals (Cd, Cu, Fe, Pb, Mn, Ni, Hg), nutrients, organic matter, suspended solids, and BOD from river waters, domestic and industrial wastes (see Severn Estuary Systems and Survey Panel 1997, 1980: first and second reports to the technical working party of the SEJC). This study covered a much larger area, seawards, than the pSAC (extending to a line between Hartland Point in Devon and Worms Head on the Gower peninsula). Nevertheless the data, summarised by Owens (1984) give some useful statistics as to relative contributions from different sources at the time (table 2).

Table 2: Sources of contaminants (as proportion of the total) entering the Severn Estuary (from Owens, 1984)

	Metals: Proportion (%) from different sources to the Severn Estuary					
metal	**Rivers & streams**	**Domestic sewage**	**Industrial effluent**	**Atmospheric input**	**Sludge dumping**	**Total input (Kg/day)**
Mercury	42.2	2.2	54.4	-	1.2	13.3
Cadmium	32.4	12.5	32.6	21.1	1.4	61.6
Chromium		20.8	55.1	9.4	14.9	224
Copper	54.2	8.6	15.8	19	2.4	621
Iron	68.9	8.1	17.2	5.8		62700
Lead	16.9	4.5	18.5	57.8	1.9	1150
Manganese	72.7	6	16.2	5.9		2780
Nickel	75.9	7.3	3.3	11.9	1.6	464
Zinc	30.3	8.8	17.8	42.2	0.9	5160

	Other water quality parameters: Proportion (%) from different sources to the Severn Estuary					
Determined	**Rivers & streams**	**Domestic sewage**	**Industrial effluent**	**Atmospheric input**	**Sludge dumping**	**Total input (10^3kg/d)**
Flow	96.4	2.7	0.9	-		
Ammoniacal N	16.5	42.6	40.9	-		35.9
Total oxidized N	91.8	1.29	7.02	-		102
Total inorganic N	74.3	8.81	16.3	-	0.67	134
Suspended solids	75.9	4.7	18.1	-	1.1	2370
Orthophosphate	37	16.3	44	-	2.6	16.1
Silicate	90.7	4.46	4.65	-		183
BOD	29.6	39.8	20.1	-	10.6	256
Diss organic matter	48.2	35.6	16.1	-		195
Part organic matter	57.1	13.8	26.2	-	2.82	602

Despite the fact that these data are now twenty years old many of the conclusions drawn are useful pointers to sources. Thus

- Rivers contributed the greatest proportions of Dissolved Organic Carbon (DOC), Particulate Organic Carbon (POC), Total Oxidised Nitrogen (TON), silicate, suspended solids, Cu, Fe, Mn and Ni and substantial contributions of phosphate, BOD, Hg, Cd, Pb and Zn

- Domestic sewage contributed the major proportion of NH_3, BOD, and substantial phosphate, DOC, POC, Cd, and Cr
- Industrial discharges contribute major proportions of Hg, phosphate, Cr, and significant proportions of NH_3, suspended solids, BOD, DOC, POM, Cd, Cu, Fe, Pb, Mn and Zn.
- Atmospheric deposition contributed the greatest proportions of Pb and Zn and substantial proportions of Cd, Cu, and Ni.
- Dumping of sludge and industrial wastes represent an insignificant source of metals
- More than 60% of all contaminants enter the system upstream of the Cardiff-R.Parrett boundary (ie inner and outer Severn Estuary) compared to much smaller proportions seawards (Bristol Channel).

Leachate from landfill sites can impact on the estuarine and coastal water quality. Such leakage led to the imposition of remedial action (synchronisation of consented discharges with tidal flow) at the Walpole Drove site near Bridgwater (Environment Agency, 1999). Leachate from landfill sites is presumed to impact at a local level, rather than as a major contributor to overall contaminant budgets in the Severn. Nevertheless, it may be useful in future to confirm their contributions to loadings and also their impact on biota.

Table 3 (also from Owens 1984) indicates the proportional mass inputs from the principal sources of contamination to the inner Estuary (defined in Figure 2), the outer Estuary (downstream to a line roughly between the R. Parrett to the west of Cardiff) and the Bristol Channel seaward of this line.

Table 3: Proportions (%) of contaminant inputs to different parts of the Severn Estuary and Bristol Channel (from Owens, 1984)

	Bristol Channel	Outer Estuary	Inner Estuary
Ammoniacal N	21	68.1	10.9
Total oxidized N	20.7	27.3	52
Total inorganic N	20.8	37.9	41.3
Suspended solids	11.3	40.9	47.8
Orthophosphate	10.6	68.3	21.1
Silicate	22.8	31.1	46.1
Dissved org matter	18.6	60	21.4
BOD	25.2	60.2	14.6
Part organic matter	10.5	50.3	39.2
Mercury: total	53.5	24.1	22.4
Cadmium: total	16.5	61.4	22.1
Copper: total	39.9	26.1	34
Iron: total	25.7	20.1	54.2
Lead: total	32.9	55.7	11.4
Manganese: total	28.6	30.3	41.4
Nickel: total	58.4	16.6	25
Zinc: total	42.2	40.9	17.9

It should be noted that there may be large uncertainties in the load estimates because of intermittent sampling, uncertain detection limits, and the inherent errors in multiplying large flows with very small concentrations. Where available, more recent data on loadings from riverine and direct inputs, as supplied for PARCOM, are described for individual contaminants in the subsequent sections, though these are subjected to the same errors.

Bird (2002) has recently provided estimates of quantities of a range of contaminants entering the Severn from key discharges, as summarised in Annex 7. These are based on 'Actual' values for 2001 from the Environment Agency's South West Region or calculated from discharge consents (maximum discharge allowed for each contaminant x the total flow). Since these assume continuous maximum allowable discharge, and in view of other difficulties in estimating loads, described above, they are only intended as a guide to possible contributions from different sources.

5.1 Toxic contaminants

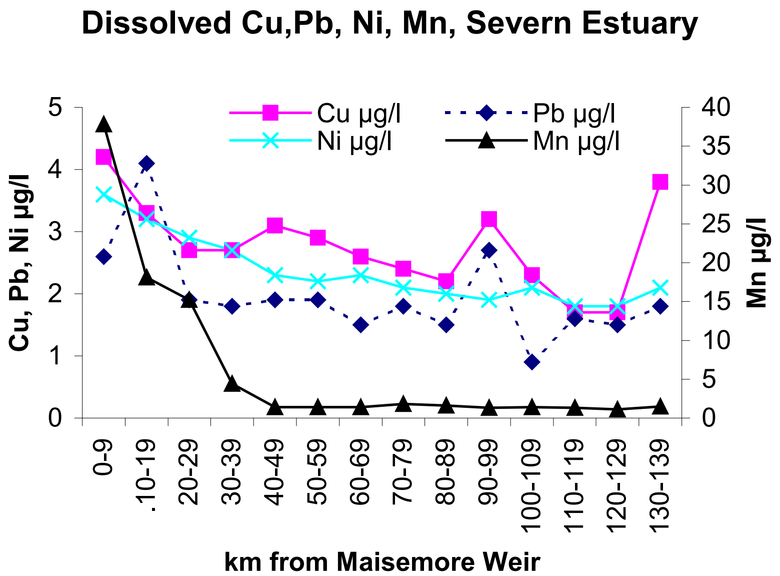
5.1.1 Metals

The history of metal refining and steel production, centered around Avonmouth and South Wales, respectively, has focused much interest on metal contamination in the SAC. Data from the 1970s suggested that metal concentrations were elevated and their distributions often showed significant deviations from expected behaviour when examined against salinity.

Mid axial estuarine profiles of water quality parameters, including dissolved metals, have been conducted since the 1970s using helicopter surveys. Owens (1984) summarises data for 1975-1979, from which figure 3 is derived. The distributions will be governed by the relative importance of riverine flux (upstream) and atmospheric inputs (largest in the outer Estuary), and modified by the various discharges along the course of the tideway.

Atmospheric sources of metals to the Severn Estuary, and mechanisms of dispersal and deposition, have been reviewed by Vale and Harrison (1994) who evaluated apportionment using budgets for rivers and discharges drawn up by Welsh Water Authority (1980) and aerial deposition data from AERE Harwell. In the late 1970s these atmospheric inputs originated mainly from local industrial and urban centers (including domestic combustion and vehicle emissions) and were estimated to account for 50% of the Pb and Zn input to the Estuary, 10-20% of Cd, Cu and Ni, but only a small proportion of Cr, Fe and Mn. There is an indication that quantities being deposited were declining in the 1980s, though spatial depositional patterns were broadly similar. Much of the atmospheric Cd, Cu and Pb was judged to be deposited in the outer Estuary between Avonmouth and Cardiff.

Metals with a major riverine component include Mn and Ni, as indicated in figure 3 Mn is also characterised by sharp reactivity and significant removal of riverine inputs at low salinity.



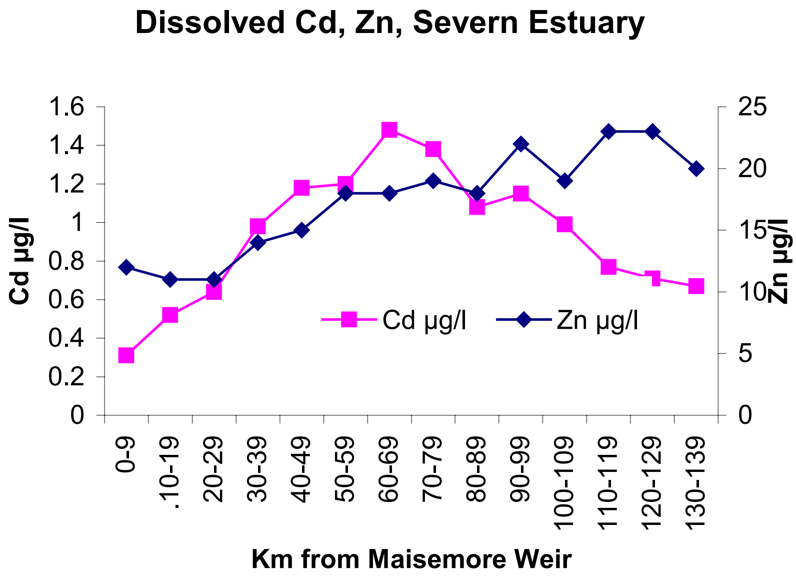


Figure 3. Profiles showing the distributions of dissolved metals along the Severn Estuary (Drawn from data in Owens, 1984). Maisemore Weir is at the head of the Estuary – see figure 1.

Metals dominated by discharges or atmospheric inputs have somewhat different characteristics. The profile for Cd (figure 3) displays a more distinctive mid-estuarine 'hump' in the Avonmouth-Cardiff area, reflecting industrial inputs (Radford, 1981), and perhaps desorption from sediments, in that region. The smelter at Avonmouth produces primarily Zn and Pb but also refined Cd. As a result the Severn receives the largest input of Cd of all UK estuaries. For much of the 1990s two of the UKs largest Cd discharges (one sewage and one industrial) were situated at Avonmouth (NRA, 1995). These discharges –as indicated in PARCOM data later in this section - are reported to be declining.

Though elevated, concentrations of Cd along the mid-estuarine transect appear to fall within EQS limits. Unusually, the data for Zn in the earlier surveys suggest concentrations increase seawards, with highest concentrations at the mouth of the Estuary (figure 3). At the time it was suggested that, as the major proportion of Zn (60%) enters upstream of the mouth (from significant industrial sources at Avonmouth and also from the River Severn), the profile was generated by gradual desorption from particulates as these move seawards. However, this may be partly artifactual due to earlier difficulties in analysing Zn in saline waters. The profile for Zn in more recent data sets do not display this feature.

Metals with a mixture of sources occupy an intermediate position: Axial profiles indicate that dissolved Cu and Pb gradually increase from the mouth of the Estuary upstream, towards the mid-Estuary, at least. Concentrations sometimes decrease again toward the tidal limit as indicated for Pb.

Examination of more recent data from axial profiles of dissolved metals taken during the 1990s (Ellis, 2002) generally confirm similar behaviour (see also section 7.1.1), though several metals display a mid-estuarine hump similar to Cd, above, suggesting inputs here (either directly from discharges or, indirectly, from sediments). The significance of mid-estuarine inputs of Cd, As, and Cr on profiles along the Estuary has also been demonstrated by Apte et al., 1990. Downstream of Avonmouth most metals display relatively conservative behaviour dependent on the degree of dilution with comparatively low-metal sea water. Concentrations of metals at the seaward end of the Estuary (33psu), calculated for 1988 data (Apte et al 1990), appear to have decreased relative to earlier levels described by Owens (1984) though it is not known whether this is reflective of genuine trends or improved detection of low concentrations.

Thus, there is still some uncertainty about the scale of reductions in recent years which may reflect the considerable day-to-day variation in metal concentrations known to occur in water samples, particularly from the inner Severn. These variations may be partly driven by the degree of sediment remobilisation and resuspension at the time of sampling. *Average* concentrations of dissolved metals across the Estuary as a whole do not appear to have changed greatly between the 1970s and 1990s with the possible exception of lead and cadmium which have generally decreased by up to a factor of two during the intervening period. Reductions in Cd were also described by Owens (1984) when comparing earlier results of Abdullah and Royal (1974).

The apportioning of sources described by Morris (1984) may therefore be out of date but nevertheless give an impression of their relative importance. Inputs of Ni via discharges are considered minor compared to riverine sources (except one or two streams in the Swansea area). Rivers also introduce some three-quarters of the Mn load to the Estuary. Atmospheric inputs are considered significant for Cd and Zn (and Pb) in the outer Estuary though this is not reflected in profiles of dissolved Pb. Copper originates substantially from rivers but may be locally modified by discharges and atmospheric sources. Two discharges were identified as significant contributors of Hg to the outer Estuary some twenty years ago (Morris, 1984) though these have presumably been reduced since then. Accurate profiles of dissolved mercury are not easy to construct, due to the rapid scavenging of the element onto abundant suspended particulate matter in the Estuary.

Very few of the metals from the mid-Estuary profiles collected in the 1970s and early 1980s, expressed as annual averages, exceeded EQS values. Individual Cu samples upto 33 μg l^{-1} were recorded in the earlier surveys, however, during 1982 none of the stations had a mean greater than 6 μg l^{-1} (EQS=5μg l^{-1}). Likewise, concentrations of Zn in individual samples ranged up to 69μg l^{-1} but in 1982 none of the site means were higher than 23μg l^{-1} (EQS=40μg l^{-1}). A discussion of more recent data, in terms of quality standards is provided in section 7.1.1. This includes data from sites closer to the shoreline.

Particulate metals form an important part of the loading discharged to estuaries. Scavenging of dissolved metals also occurs, so that the sediments of the Estuary provide an integrated record of contamination history. Because of their larger surface area and greater density of organic and oxyhydroxide binding sites, contamination loadings will be highest in fine fractions (primarily located between Avonmouth and Severn Beach; Caldicot flats; River Parrett and outer Bridgwater Bay; and between the mouths of the Usk and Taff) and lowest on sands (e.g. Middle to Welsh Grounds, Culver Sands). Because of the energetic hydrodynamic regime in the Severn, and resultant high turbidity, there is considerable mixing and redistribution of fines and their associated contaminant burden - resulting in a fairly homogenous distribution (discussed in further detail in Section 8.1). As a result, contamination arising from point source metal contamination tends to be chronic and dispersed over a large area rather than concentrated as hotspots. This is reflected in the similar metal concentrations in Severn sediments, and those of the Usk, Cardiff Bay (pre-barrage) and Bristol Avon, as shown in table 4. For comparison, data for other estuaries in the south -west, including the highly contaminated Restronguet Creek (Fal) and the Avon in Devon (relatively uncontaminated) are also shown. Clearly, for metals such as As, Cu and Zn concentrations in Severn sediments pale in comparison those derived from mining sources entering Restronguet Creek. The bulk composition of the major geogenic elements in Severn sediments were also described as similar to world averages for silts and sands by Chester and Stoner (1975) and Hamilton et al (1979).

However, despite the fact that contaminant loadings in the Severn are somewhat 'diluted' by their widespread distribution, enhancement in fine fractions is still observed for a number of metals, relative to the 'baseline' represented by the Devon Avon.

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

site	Cu	Zn	Pb	Cd	Mn	Fe	As	Hg	Ref
Severn	35	242	84	0.63	672	26805	8.4	0.44	7
Avon (Severn)	39	287	104	0.92	622	30685	8.6	0.55	7
Usk	53	288	93	0.86	639	30723	9.2	0.41	7
Cardiff	54	345	116	1.49	644	33067	11.5	0.56	7
Lynher	274	317	150	0.6	289	23120	50.7	2.1	1,7
Tamar	145-545	221-605	19-239	0.5	105-1500	21000-49000	25-236	0.2-1.5	2
Plym		256		9.3	171	12100	41	0.35	3,4,5
Restronguet	1690	1540	684	3	1030	54000	1080		6
Avon (Devon)	19	98	39	0.3	417	19400	13	0.12	7

[1]Bland et al 1982;[2] Ackroyd et al 1986 [3]Langston, 1980; [4]Millward and Herbert 1981; [5]Bryan and Hummerstone 1973; [6] Aston et al 1975; [7]own unpublished data

Contamination of the Severn with Cd and Zn has been well known for many years and is highest in the Avonmouth area, near the expected anthropogenic sources (Little and Smith, 1994). There is also published evidence for contamination of fine sediments with other metals including Pb, Cu, Ag and Hg (Nickless *et al.,* 1972; Butterworth *et al*., 1972; Peden *et al.,* 1973; Abdullah *et al.,* 1972: Hamilton *et al*., 1979; Gardner, 1978; Bryan *et al.,* 1980). Pb, Zn and Sn were also identified as being in excess (Chester and Stoner, 1975), residing predominantly in non-detrital fractions. Even some of sandier sediments from deeper waters appear to exhibit contaminant concentrations in excess of that predicted from sedimentological characteristics such as Al or organic content (Little and Smith, 1994).

Where consolidated cores have been collected (e.g. in Swansea Bay) these confirm an increase in deposition (Fe, Pb, Zn, Cu) since the middle of the nineteenth century, which is indicative of industrialisation and other anthropogenic inputs (Clifton and Hamilton, 1979). For much of the Severn Estuary, however, it is not possible to reconstruct accurately the history of contamination due to the unstable nature of the surface deposits. There is a limited amount of data which indicate Pb, Cu, Cr, Ni and Zn concentrations in fine sediments may have decreased, by as much as 25- 50% since the 1970s, perhaps due to contraction of metal industry (Little and Smith, 1994). However, these conclusions are based on comparisons with the earlier work of Hamilton *et al*, (1979) and may involve some component due to different sampling and analytical methodologies. In the same period only Pb has decreased in sands, whilst Cr, Cu and Ni appear to have increased by 25-66% (Little and Smith, 1994).

Although sediments are a useful guide to environmental contamination, ultimately it is the impact on biota which is of most concern. A considerable amount of baseline data exists on metals in biota, much of which represents opportunistic sampling or 'food-basket' results on edible species to ensure safe levels for human consumers. Studies designed to evaluate the issue of bioavailability, using appropriate bioindicators are few. The distribution of key indicator species (e.g. *Fucus* spp, *Nereis diversicolor*, *Scrobicularia plana*, *Mytilus edulis*, *Cerastoderma edule*, *Littorina* spp.) and the bioavailability and impact of metals in UK estuaries, including the Severn Estuary pSAC have been the subject of research at the MBA over a period spanning three decades (see Bryan *et al.,* 1980, 1985; Langston *et al*., 1994). Though much of the data are comparatively old it nevertheless provides a useful background to assess characteristics of the site and acts as a valuable baseline for future changes.

A synthesis of some of the information on Cd in different bioindicators illustrates some of the issues surrounding metal bioavailability in the Severn. A transect along the Estuary from Sharpness to Minehead is shown in figure 4. Concentrations in sediment are relatively low and fairly constant along the Estuary. In contrast Cd is accumulated to high levels in *Fucus* and *Littorina*, and accumulation increases significantly upstream. Since *Fucus* accumulates Cd primarily from water this indicates that levels in this species, together with bioavailability in other bioindicators are dominated directly or indirectly by levels of dissolved Cd. (In the case of *Littorina* this may well be indirect since the winkle feeds on *Fucus*). Concentrations of Cd in *Macoma* and *Nereis* (deposit feeder/detritivore) are much lower than in *Fucus/Littorina* though trends in accumulation along the Estuary are similar, and concentrations are appreciably higher than normal (see examples in tables 5-6).

Data for Ag in the Severn (figure 5) illustrates how bioavailability, and the responses of organisms differs between metals. Again concentrations in sediments remain at a fairly consistent low level along the Estuary, and concentrations of Ag increase upstream in *Fucus*, presumably following trends in dissolved silver. However, considerably higher body burdens are accumulated in *Nereis* and, notably, in the deposit-feeding clam *Macoma balthica* (up to 100µg g^{-1}). Although these burrowing species undoubtedly absorb some Ag from solution, the influence of water seems most likely to be mediated through the ingestion of surface sediment, to which the readily available Ag is adsorbed. This adsorbed fraction may be so small that changes along the Estuary would not be detected in a 'total' analysis of sediment. There is also the possibility that the increased availability of Ag (and other metals) upstream is in some way related to salinity.

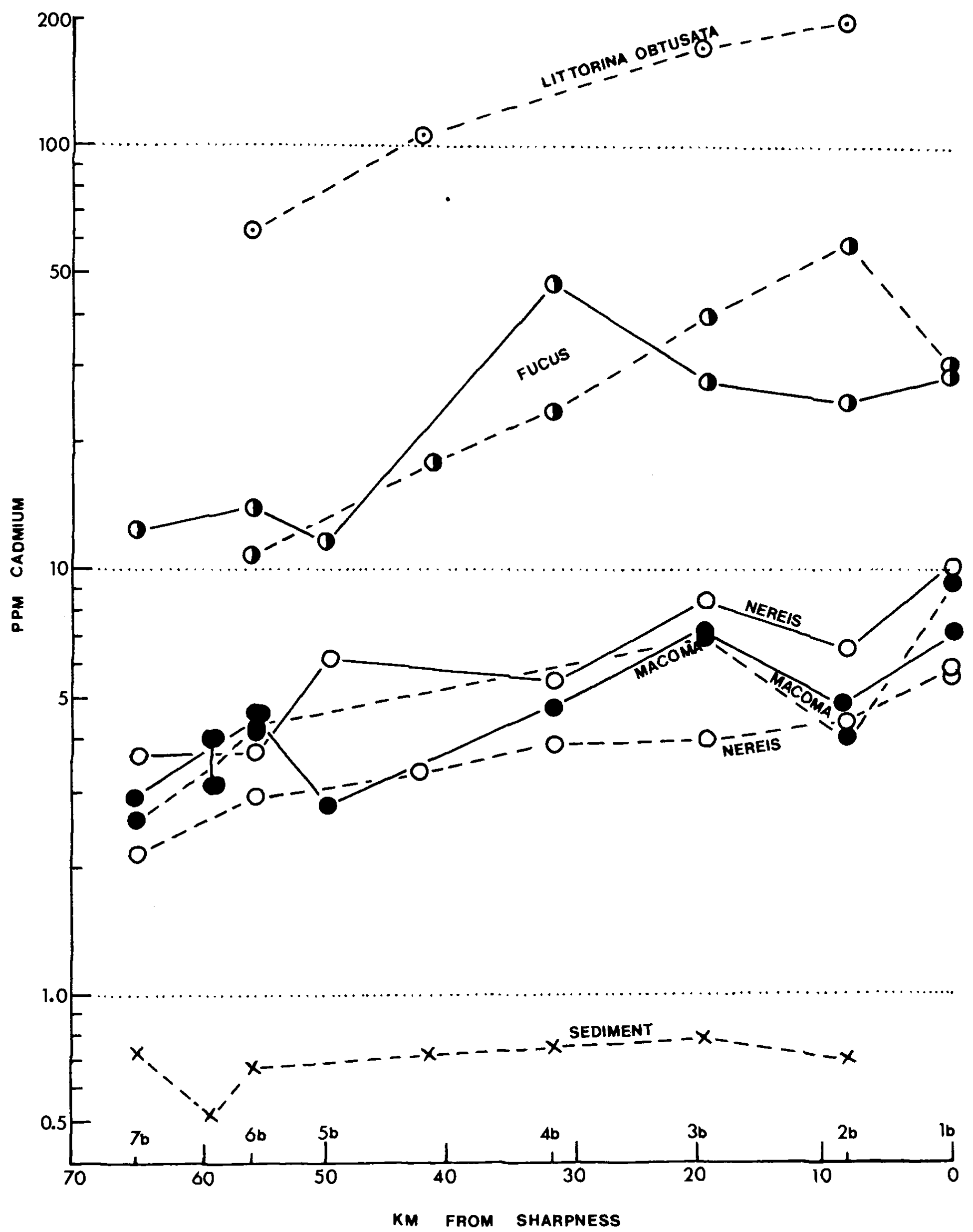


Figure 4. Concentrations of cadmium in sediment and organisms from the Severn Estuary. Broken line and solid lines are for different survey dates (from Bryan *et al* 1980).

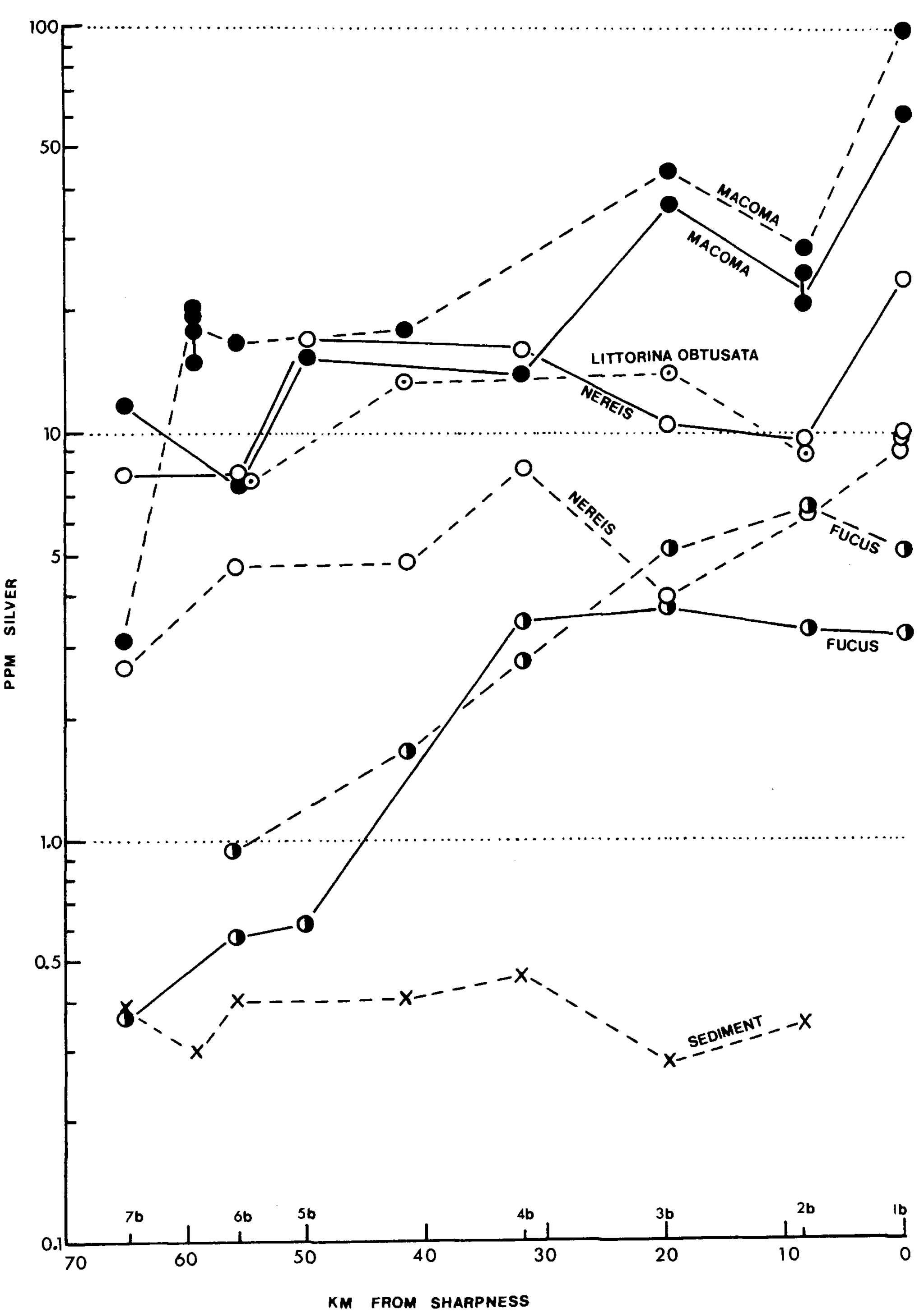


Figure 5. Concentrations of silver in sediment and organisms from the Severn Estuary. Broken line and solid lines are for different survey dates (from Bryan *et al* 1980).

Comparative data for metals in various indicator species from different sub-estuaries within the pSAC are presented in tables 5 and 6, together with UK baselines to give an indication of the relative scale of concentrations.

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

Table 5 . *Nereis diversicolor*. Metals concentrations (µg g^{-1} dry weight) in the Severn pSAC and UK baselines. (Langston, Burt and Chesman, unpublished data)

	Ag	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Severn	8.01	3.79	0.52	54.4	396	14.3	4.94	3.56	264	1.42	12.8	0.31
Avon	3.81	2.3	0.14	41.1	318	11.6	5.09	1.97	222	1.1	7.1	na
Cardiff	2.04	4.88	0.38	45.7	351	11.1	7.14	3.95	236	0.79	19.1	na
Usk	4.61	3.04	0.29	33.45	300	11.8	6.06	2.3	255	2.44	23.7	na
UK min	0.06	0.02	0.03	7.69	210	4.03	0.63	0.16	87.8	0.02	3.22	0.05
Severn ÷ UK min	133	189	17	7	1.8	3.5	7.8	22	3	71	4	6.2

na- not analysed

Table 5 shows summary statistics of our own unpublished data for metals in *Nereis diversicolor* in the Severn (mean values spanning ~25 years, therefore incorporating the recent history of contamination rather than current status). For comparison, the average of the lowest ten sites in our UK data set (encompassing the entire range of estuaries in England and Wales) is shown (UK min).

These indicate similar enrichment for all sub-estuaries within the pSAC, compared with UK baselines. The degree of enrichment for Severn worms, relative to baselines is also shown in table 5 and decreases in the order Cd>Ag>Hg>>Pb>Cr>Ni>Cu>Sn>As>Mn>Zn and Fe. The relatively low enrichment of the latter two metals is almost certainly a result of the ability of *Nereis* to regulate body burdens of these essential elements.

Figure 6 depicts a map of relative Hg bioavailability in south west England based on accumulation in *Nereis*, and highlights the enrichment present in the Severn.

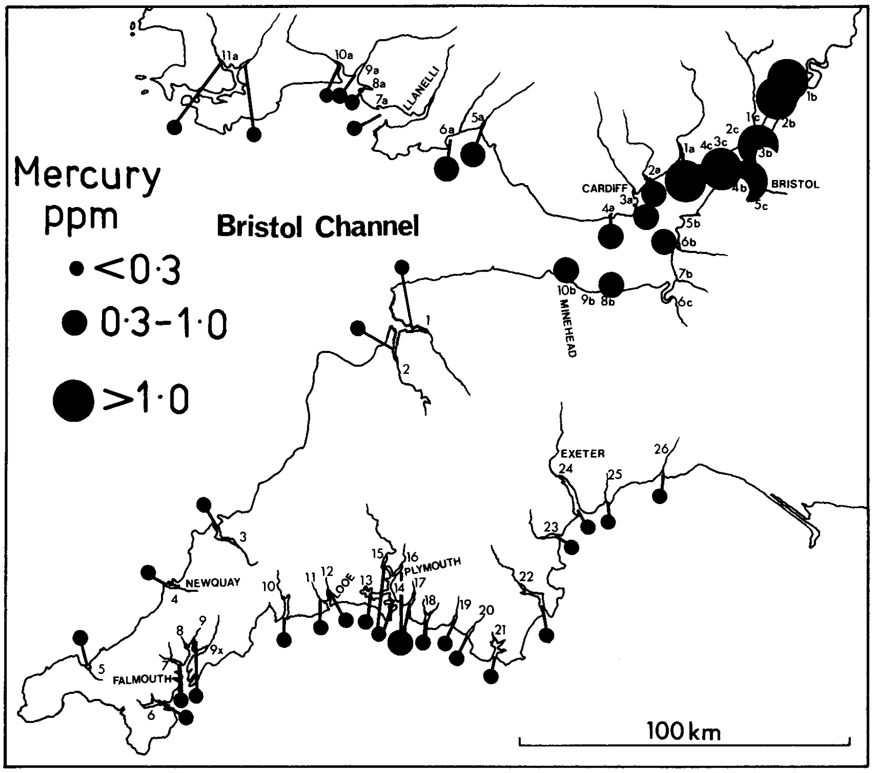


Figure 6. Mercury in worms *Nereis diversicolor.* (from Bryan *et al* 1980).

The deposit-feeding clam *Scrobicularia plana* has also proved to be a valuable indicator species, particular in terms of understanding trends in sediment metal bioavailability. Though less widespread in distribution in the Severn than *Nereis* it is in many respects a better indicator of metals (with the exception of Cu) and has the advantage of not regulating Zn. Its range in the Severn extends upstream to Shepperdine – almost as far as *Nereis* - encompassing most of the Estuary. Its ability to survive so far upstream is presumed due to the buffering influence of its burial behaviour in sediments (upto 25cm) and an ability to isolate itself, through shell closure, from extreme low salinities. Unlike *Nereis*, however, it was not found in the Avon.

Table 6 shows summary statistics for metals in *Scrobicularia plana* in the Severn, Usk and Cardiff sub-estuaries (mean values) and, for comparison, the average of the lowest ten sites in our UK data set (UK min). These indicate similar enrichment for all the SAC estuaries compared with UK baselines. The degree of enrichment was generally of similar order to that displayed by *Nereis* with some slight variation on ranking: Ag>Cd>Hg>Mn,Cr>Pb>Ni>Fe,Cu>Sn>Zn>As. Nevertheless both species highlight the significantly increased bioaccumulation of silver, cadmium and mercury above normal. As none of these metals is naturally enriched in sediments of the area, they are presumed to be of anthropogenic origin. It is stressed that these are mean, Estuary-wide, values (not extremes) spanning ~25 years, reflecting to an extent the recent history of contamination. Nevertheless the use of a validated suite of indicator species is clearly a useful way of estimating bioavailability around the system. Valuable baseline information is already in place regarding the (historic) distribution of these species and their metal burdens. The current status of the SAC needs to be defined in similar fashion, and at intervals in the future, to ensure bioavailability does not increase.

Table 6. *Scrobicularia plana*. Metals concentrations (µg g^{-1} dry weight) in the Severn pSAC and UK baselines. (Langston, Burt and Chesman, unpublished data)

	Ag	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Severn	8.37	7.18	3.68	47.4	1271	69	6.44	43.5	775	0.64	20	0.39
Cardiff	1.24	7.35	2.25	24.8	694	30.9	5.2	35.3	785	0.44	19.8	na
Usk	1.96	20.9	2.63	2.9.5	847	51	5.1	33.4	1100	0.78	32.4	na
UK min	0.05	0.13	0.26	9.5	226	4.8	0.89	3.79	193	0.04	5.7	0.08
Severn ÷ UK min	167	55	14	5	5.62	14.3	7.2	11.5	4	16	3.5	4.9

na- not analysed

Figure 7 depicts a map of relative Cd bioavailability in south west England based on accumulation in deposit-feeding clams (*Scrobicularia* and *Macoma balthica*) and highlights the enrichment present in the Severn.

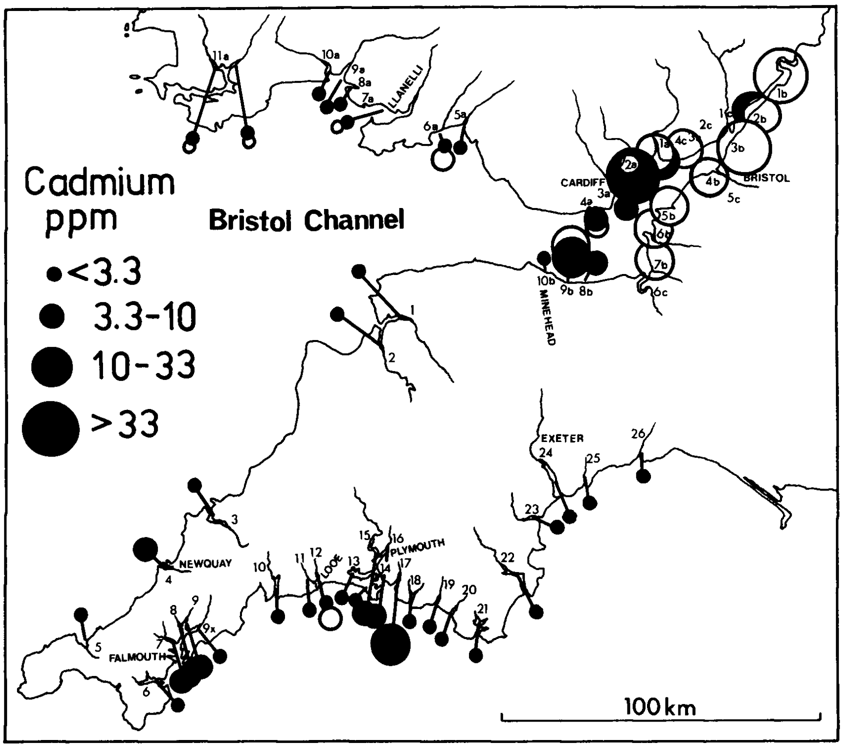


Figure 7. Cadmium in clams *Scrobicularia plana* (closed symbols)/*Macoma balthica* open symbols.

Non-commercial species, particularly in inter-tidal areas of the Estuary, clearly accumulate metals, though the ecological significance is largely unknown. The metals most likely to be of significance toxicologically appear to be Ag, Cd, Cu, and perhaps locally Zn. There is some largely anecdotal evidence suggesting a link between improvements in water quality and overall biological status though this linkage is still largely correlative and subjective, rather than established cause and effect. Ecological monitoring by Zeneca Ltd, one of the co-owners of the Severnside, Avonmouth industrial site, has been carried out since 1965 (Zeneca 1997) to assess potential impact surrounding the discharge here. The pipeline was not constructed until 1971 so earliest data represents pre-operational baselines. The effluent of 1 - 2.5 million gallons per day contains ammonia, Zn, nitric acid, and, until 1989 at least, calcium phosphate and potassium chloride. Six other sites have been monitored in the Severn Estuary to provide a baseline (Severn Beach, New Passage, Shepperdine, Goldcliff, Redwick and Blackrock). These data suggest generally small improvements in intertidal macrofauna (numbers and species) at all sites, over time, coinciding with decreases in Cd, Zn and Cu in *Fucus* (though in fact ammonia is considered by Zeneca as the most significant component in the discharge, toxicologically). Reductions in sediment metals have been less obvious and restricted largely to Cd (~50% Cd at Severnside). Since the reported biological improvements have been broadscale across the Estuary it is not certain that they relate to changes in this particular discharge, specifically. The impact of the Severnside discharge on fauna from the lower shore rock platform was assessed by Zeneca Ltd (1997) to include a reduction in species up to a distance of 250m downstream from the end of the pipeline (possibly further upstream). This coincides with modelling predictions based on effluent dilutions (see section 9). In view of recent advances in biological effects monitoring, it may be timely to consider more extensive assessment of this and other major discharges. Some of the sub-lethal indicators described in Annex 6 may provide more sensitive measures of impact than the presence or absence of a rather restricted range of species and may also give indications as to which chemicals are most responsible for impact. Clearly more work would be useful to confirm the temporal improvements (particularly in sediments and inter-tidal organisms).

Concentrations of metals in commercial fish and shellfish (usually from offshore waters in the outer Estuary) may not be affected by contamination to the same extent as some of the indicator species described above. Accumulated residues were considered 'normal' by Owens (1984) and were not thought to constitute a human-health risk (see also Jones *et al*., 1998). However, oysters collected from Porthcawl, slightly seaward of the pSAC, were cited in the 1997 CEFAS monitoring programme (CEFAS, 2000b) as one of only two samples in the UK which would not meet the proposed EC food standard for Cd (1.5 $\mu g\ g^{-1}$ wet weight): smelting operations at Avonmouth, steelworks at Newport and the power station at Aberthaw were put forward as contributory sources. Furthermore, mussels taken from No 1 Beacon in the Severn Estuary contained the highest level of Cd (9.78 $\mu g\ g^{-1}$ dw) recorded in the National Monitoring Programme (MPMMG, 1998), an order of magnitude higher than samples from the Dee. In the context of the OSPAR guidelines for Cd in mussels - split into 'lower, medium and upper' values – the Severn sample would fall into the latter category. Hg concentrations were also elevated in Severn mussels (0.61 $\mu g\ g^{-1}$ dw) and would fall under the 'medium' category in the OSPAR scheme. In addition, Zn concentrations in mollusc samples from the Bristol Channel were highlighted by CEFAS (2000b) as being among the highest in UK waters.

Harmful effects on marine biota of the outer Estuary are considered unlikely (MPMMG, 1998) but the physiological significance of such burdens, particularly in littoral/inter-tidal species needs to be understood more fully, for example by determining metallothionein induction and intra-cellular metal-binding patterns (Langston et al., 1998).

Sampling of flatfish in the NMP survey was seaward of the SAC where levels of Hg and As in muscle and Cd and Pb in liver appear to be unexceptional (MPMMG, 1998).

Metals (Cd, Pb, Zn) in fish from the inner Severn Estuary (collected from the intake screens at Oldbury power station in the 1970s), and also the Bristol Channel, are reported by Hardisty *et al.,* (1974a,b) and Badsha and Sainsbury (1977,1978a,b). These studies focus on the level of accumulation in relation to dietary habits (particularly the role of shrimp, *Crangon vulgaris* as a possible vector), and their ecological significance. Concentrations of Zn (but not Cd and Pb) in flounder *Platichthyes flesus* (youngest year groups) were found to be much higher here at certain times of year than at Hinkley lower down the Estuary. Interestingly shrimp from Oldbury contained much higher levels of Cd (125 μ g^{-1} dry weight) than in samples from the outer Bristol Channel (4.9 μ g g^{-1} dry weight) but equivalent levels of Pb and Zn. Other dietary sources – polychaetes such as *Nereis* - may be more important for young flounders (Hardisty *et al.,* 1974a). Results for older flounder were somewhat different: Cd and Pb (but not Zn) were higher in fish from Oldbury compared to those from Barnstable in the outer Bristol Channel. One possible explanation given for this apparent anomaly was the preponderance of *Macoma* (high in Zn) in the diet of the fish sample from Barnstable (Hardisty *et al*.,1974b). Clearly it is difficult to establish simple relationships for metals in fish and their food as the latter may vary. On the whole, however, concentrations in invertebrate (dietary) species tend to be higher than in predatory vertebrates. Accumulation in fish is also likely to be modified by metabolic behaviour (Zn in particular is effectively regulated) and growth rate of the fish.

Comparisons between fish species showed marked differences in metal accumulation. Zinc concentrations in flounder were notably higher than most other species sampled, whilst Cd levels were highest (by a factor of 3-4) in the sea 'snail' *Liparis liparis*. This species feeds mainly on shrimp, from which the Cd may be assimilated. The Cd (and to a lesser extent Pb) content of several other fish species was also found to correlate with the proportion of crustaceans in the diet (Hardisty *et al* 1974b). Levels of Cd, Pb and Zn were lowest in the anadromous migratory lamprey *Lampetra fluviatilis* which may reflect comparatively low levels in muscle and blood of the teleost hosts on which they feed (probably in the Bristol Channel.).

Despite some evidence of metal bioaccumulation in fish from the inner Severn, Hardisty et al (1974a) concluded that, on the basis of screen samples, there had been no evidence of deterioration in species variety since surveys taken some thirty years earlier (Lloyd, 1940). They also considered that the Severn Estuary remained a satisfactory nursery ground for a number of teleosts and a viable migratory route for anadromous species. Growth rates of flounder from the Barnstable region were higher than at Oldbury in the inner Severn but, despite higher levels of Cd and Pb in older fish from the latter site, pollution was considered an unlikely cause.

Concerns have been expressed that the environmental quality of the Severn Estuary could affect the passage and survival of migratory fish, notably shad. Twaite shad which spawns in the Wye, Usk and Severn are reported to have been lost from several rivers (Environment Agency, 1999). Intuitively, larval and juvenile stages are likely to be most contaminant-sensitive phases. Shad in the Severn Estuary may also be vulnerable to sediment contamination either directly or indirectly. The diet of adults from the Severn and Wye estuaries consists of a wide range of prey items that includes other fish, particularly herring, sprat and gobies (Wheeler 1969; Aprahamian 1988). However, the 0+ age group of Twaite Shad (*Alosa fallax*) feed mainly on mysid shrimp from sediments and harpacticoid and calanoid copepods in the Severn Estuary (Aprahamian 1988), and in spite of their adaptation to a pelagic life, adult Twaite shad (in the Tagus Estuary, Portugal) have been reported to feed on some benthic organisms, including the brown shrimp, *Crangon crangon* (Assis *et al*. 1992). Crustaceans are excellent accumulators of Cd and other metals and might be a critical exposure route for these fish. An alternative concern is that the sensitive olfactory responses of anadromous fish could be disturbed by metals and other contaminants. Until further evidence is produced, however the causes of declining migratory fish remains speculative.

Owens (1984) could find no evidence to suggest zooplankton communities were any different to those expected, or that fish diversity had changed between 1940 and 1970, and cited extensive salmon runs, and general compliance with EQS, as indicative of satisfactory water quality.

There are indications in the above synthesis of metals data that concentrations of some elements in water, sediments and biota may have decreased in the 1990s, most consistently for Cd. This broadly agrees with data for loadings for the Severn supplied by the UK as part of its obligations to Parcom (Environment Agency, 1999). Figure 8 illustrates trends in low load[1] estimates to the Severn for Cd, Cu, Pb and Zn for the period 1991-1997. Whilst Cd has decreased significantly, no consistent trend was evident for the other metals, including Hg (data not shown). During the same period the volume of discharge from the major industrial source of Cd at Avonmouth was reduced to one third which may account for some of the observed load reductions. Between 1999-2000, however the treatment plant at the smelter experienced problems, and levels of Cd, Zn and Pb were higher than anticipated. An improved scheme is underway which is expected to return aqueous discharges to below consented levels (Environment Agency, 2002).

Previous estimates (Owens , 1984) suggest Cd inputs were split between rivers (30%), industrial discharges (30%), atmospheric deposition (20%) and domestic sewage (10%). More recent (PARCOM 1994) information has broadly comparable ratios for the aqueous sources, though quantities discharged - based on low load estimates - appear to be an order of magnitude lower (Environment Agency, 1996). For Zn the relative importance of riverine (30%), industrial (17%) and domestic sewage sources (9%) has also remained approximately constant (except perhaps for a small decrease in the proportion from industrial discharges), as has the total Zn loading.

[1] The are two principal means of estimating inputs each of which is based on the product of flow rate (of the river or discharge) and the conccentration of the determinand. The low-load estimate assumes that where the determinand cannot be detected its concentration is zero. The high load estimate assumes the determinand is present at a concentration equal to the limit of detection.

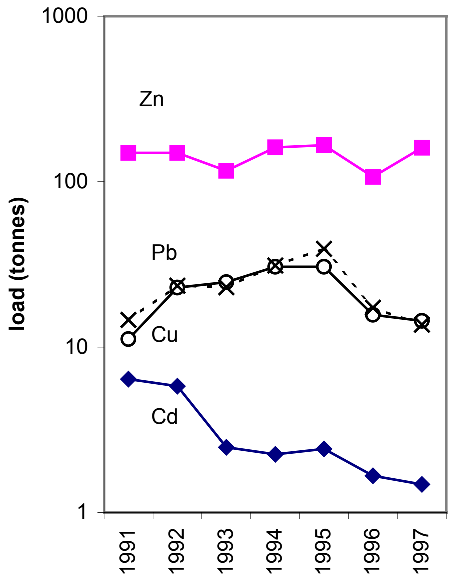


Figure 8. Temporal trends in PARCOM metal loads to the Severn Estuary 1991-1997 (plotted from low metal load estimates, Environment Agency, 1999).

It has to be stressed however that some of these figures for apparent apportionment between industrial, sewage and riverine inputs are based on broad estimates rather than real data and, in the case of Cd at least, reflect the uncertain status of the major inputs at Severnside/Avonmouth (NRA, 1995).

5.1.2 TBT and other organotins

Use of tributyltin (TBT) antifouling paint on boats less than 25m in length was prohibited in 1987, though larger vessels (essentially the commercial fleet and Navy) are still entitled to use them, at least until 2003 when recommendations from IMO for a total ban should be implemented. Remarkably, there are scarcely any published records of organotin in water, sediment or biota from the SAC. The classic indicator *Nucella lapillus* is not a widespread resident of the Severn Estuary (unsuitable substrates and other physical and chemical constraints) and therefore its use in monitoring has been restricted to rocky shores further west. Harding *et al* (1998) for example have recorded imposex levels at Milford Haven close to current sources of TBT on large vessels. However, since organotin compounds can arise from various sources (including PVC manufacture, agriculture, fungicides and wood preservatives and sewage) in addition to shipping, and in view of their well-documented endocrine disruptive effect, the distribution and impact of butyl and phenyltins may require further assessment.

Analysis of some dredge spoils in and around the SAC have been undertaken by CEFAS as part of the procedure for licensing for disposal at sea. These appear to be highly variable and in 1997 ranged from 0.01µg g^{-1} wet weight in the Severn Estuary to 0.32,0.56 and 2.37 µg g^{-1} wet weight in sediments from Cardiff, Newport, and Swansea, respectively (CEFAS, 2001b). These latter three samples indicate there are substantial reservoirs of TBT, albeit localised. Processes including physical resuspension and bioturbation could remobilise these sinks. Furthermore TBT in such contaminated sediments is likely to be available and potential harmful to deposit-feeders and infauna (Langston and Burt, 1991).

5.1.3 Hydrocarbons (Oil, Petrochemicals, PAHs)

Oil

Oil pollution is a continual threat to all inshore marine habitats, and particularly pronounced in the pSAC due to the intensity of shipping traffic. Risks include small leaks, spills and discharges as well as the possibility of a major accident. The threat to the Severn was highlighted in 1996 when a major oil spill from the 'Sea Empress' impacted upon more than 100km stretch of environmentally important south west Wales coastline (outside the pSAC) at Milford Haven.

There are a number of ways in which oil could potentially impact on the interest features of the pSAC. Intertidal habitats are under greatest threat from the physical effects of oil pollution: the most vulnerable of these are the sheltered rocky coasts, intertidal sand and mudflats and saltmarshes of the enclosed inlets and bays (see reviews of vulnerability of shores to oil damage by Gundlach and Hayes, 1978; Elliott and Griffiths, 1987). In extreme events lethal effects would induce community changes.

The direct effects of oil on inshore shellfish beds, and fish with small stock size and restricted spawning area (or in aquaculture cages) are potentially serious. These may be unable to move to unpolluted waters in the event of moderate spillages and significant mortalities, of , for example, bivalves, would be expected.

Subtidal habitats (e.g. reefs and sandbanks) 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.

Oil pollution may result in hydrocarbons becoming incorporated into sediments and buried. Sediment re-suspension and movement can result in long-term exposure of marine organisms to potentially damaging hydrocarbons. Heavily contaminated sediments are likely to have acute effects on populations of bottom-dwelling fish such as plaice and sole (in extreme cases, such as the Amoco Cadiz spill, whole year classes were wiped out over large areas of northern Brittany). Sub-lethal changes could be manifested as increased bioaccumulation and tainting (see section 7.1.5), induction of components of the MFO enzyme system (e.g. ethoxyresorufin-O-deethylase – EROD - activity), and higher order changes in productivity, fecundity and behaviour. Olfactory responses in crustacea can affect their searching, feeding and grooming responses. Fish behaviour, including migration, is also known to be affected by oil spills with most fish avoiding areas of heavy contamination.

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

Eggs and planktonic larval stages of fish molluscs and crustacea are also vulnerable to contact with oil in surface waters. Fortunately, because of the widespread dispersal of many species, in the event of a major spill their distributions and recruitment would only be threatened, locally and over short time scales. This may not be the case in species whose grounds were restricted to physically contained habitats (e.g. bays).

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).

Thompson and Eglinton (1978) found 105µg g^{-1} (dw) aliphatic hydrocarbons in sediments from Aust Warth (1 mile downstream of the Severn Bridge). A large unresolved envelope (UCM) on the GLC trace of the fraction was a sign of degraded or chronic oil contamination, and was particularly high suggesting that almost all the hydrocarbons were of petroleum origin, indicating pollution from crude oil. Similarly, Killops and Howell (1988) reported the presence of UCMs in chromatograms of hydrocarbons from the sediments of Bridgwater Bay. Sediments from three sites in the Bay; Sterte Flats, Kilve Beach and Kilve Cliff were analysed in this study and all displayed a bimodal UCM interpreted as evidence for at least two sources of crude oil input. The relative quantities of pristine and phytane in the mixture reflected chronic petrogenic contamination.

This study also noted a variety of source-related parameters which indicated that the Sterte Flats sediment had received a greater petrogenic input (oil) relative to pyrogenic (PAH – combustion of fossil fuel), up to 50% greater than Kilve Beach. This could imply that Sterte Flats had been subject to localized oil pollution, or may reflect a preferential association of alkanes with fine-grained sediment, which has been observed in other estuarine sediments (Readman *et al.,*1984). Hydrodynamic sorting of sedimentary material according to density (Dyer 1984, and references therin) could further increase differences in the relative distribution. Sterte Flats is an intertidal area in the mouth of the River Parrett with finer-grained sediment than Kilve, which is approximately 50km west.

PAH's

PAH's are ubiquitous environmental contaminants. Although they can be formed naturally (e.g. forest fires) their predominant source is anthropogenic emissions, and the highest concentrations are generally found around urban centres (Cole *et al*., 1999). In the aquatic environment PAHs are generally highest in sediments, intermediate in biota and lowest in the water column (CCME 1992). They are of particular concern in the marine environment as the lower molecular weight PAHs are toxic to marine organisms and metabolites of higher molecular weight PAH's are carcinogenic (Law *et al*., 1997). PAH concentrations in the sediments have been linked to liver neoplasms and other abnormalities in fish (Malins *et al.,* 1988). In addition, some PAHs have been identified as endocrine disruptors (Anderson *et al*., 1996a,b; Kocan *et al.,* 1996).

There are few reports available relating to PAHs in waters of the Severn Estuary. A report by Law *et al*., (1997) indicates that PAH concentrations may be concentrated around Port Talbot, which is just outside the pSAC: in a survey of coastal waters around England and Wales undertaken as part of the UK National Monitoring Programme (NMP), the total concentration of 15 PAHs measured in unfiltered water from Nash Point (west of Llantwit Major) in 1993 and 1994 were 104 and 164 ng l^{-1} respectively (table 7) (Law *et al*. 1997). EQS exist only for naphthalene.

Table 7. Concentrations of PAHs in water, Nash Point, Severn Estuary (adapted from Law *et al.,* 1997)

	Concentrations of PAHs in unfiltered water (ng l^{-1})	
	1993	1994
Naphthalene*	<15	6
Acenaphthalene	<3	<1
Fluorene	4	<2
Phenanthrene	17	<3
Anthracene	3	3
Fluoranthene	27	30
Pyrene	25	18
Benzo(a)anthracene	13	5
Chrysene	15	9
Benzo(e)pyrene,	7	6
Benzo(b)fluoranthrene	15	11
Benzo(k)fluoranthrene	5	5
Benzo(a)pyrene	15	11
Dibenzo(a,h)anthracene	2	<12
Benzo(ghi)perylene	14	<17
Total PAHs	164	104

*EQS 5µg l^{-1} annual average; 80µg l^{-1} maximum allowable concentration

Much higher concentrations (1150ng l^{-1}) were detected at Aberavon Beach near the Port Talbot steelworks, which is further west along the Bristol Channel, and PAHs in seawater in the Celtic Deep, the nearest National Monitoring Point, Celtic Deep (NMP 605) outside the Bristol Channel, were undetectable (Law *et al*. 1997).

In comparison to concentrations in other industrialised estuaries (e.g River Tees, up to 10724ng l^{-1}), PAHs in waters of the Severn appear to be relatively low (Law *et al*. 1997). There is no published information on PAH levels in the water upstream.

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

Since many PAHs have such an affinity for particulates, concentrations in estuarine sediments tend to be much higher. Thus, in the Severn, the sum of 15 PAHs in sediments from Severn Beach (adjacent to the M4 road bridge) was 5425 µg kg^{-1} (dry weight), and 5472 µg kg^{-1} in sediments from the English Grounds in mid-Estuary (between Cardiff and Weston Super-Mare). Further west at Port Talbot, PAH concentrations in sediments were higher (7124 µg kg^{-1}) but decreased to below threshold effects levels[1] outside the Bristol Channel (464 to 1014µg kg^{-1}, Celtic Deep) (Woodhead *et al*. 1999). The highest sediment concentrations at any of the 80 coastal sites in the UK survey of PAHs was obtained from Neyland Spit in Milford Haven where levels of 102,471µg kg^{-1} were recorded (Woodhead *et al*. 1999). Eglinton *et al* (1975) reported concentrations of up to 150, 000µg kg^{-1} PAHs in sediments of the Usk Estuary.

Sources of PAHs in Severn Estuary sediments have been investigated in several studies: John *et al* (1979) considered the environmental contribution, noting that the Taff and its tributaries drain a large area of the south Wales coalfield, and oil bearing shales are exposed on the southern coast of the Estuary (Bridgwater Bay). Sediments from the Taff, one of its tributaries (Rhondda Fach) and the south bank of the Severn (around Severn Beach) were analysed for 9 PAHs in this study. In general, levels were higher in the tributary than the Taff itself and PAHs in the Estuary samples were lower than either river (e.g anthracene concentrations were up to 6400, 1800 and 300µg kg^{-1} dry weight in the Rhondda Fach, Taff and Severn Estuary, respectively)[2]. Findings indicated that the coal bearing strata and (recent) mining activity make a significant contribution to PAH contamination in the Severn system. Cooke *et al* (1979) looked at PAHs in sediments from the south east banks of the Severn (Sharpness, Aust and Arlingham) and found high levels of 10 PAHs which were attributed to the high coal dust content of the sediments.
However, Thompson and Eglinton (1978) found 9000µg kg^{-1} (dw) PAHs in sediment from 1 mile downstream of the Severn Bridge. The PAH distribution was consistent with an anthropogenic origin (combustion of fossil fuels - notably petroleum) and

[1] Threshold effects level 1884µg kg^{-1}, probable effects level 16770µg kg^{-1} (Macdonald *et al*., 1996)
[2] these are higher than both the interim sediment quality guideline (ISQG) of 46.9µg kg^{-1} and the probable effects level (PEL) of 245µg kg^{-1} (from CCME, 1999)

included many PAHs which may be expected to harm the estuarine environment including B(a)P (470µg kg^{-1})[1], phenanthrene (1100µg kg^{-1})[2] and methylphenanthrene (550µg kg^{-1}). The source of these PAHs was considered to be land run-off and the precipitation of airborne particulates derived from combustion products.

Examining hydrocarbon sources and distribution in Bridgwater Bay, Killops and Howell (1988) found an extended range of PAHs in sediment samples from intertidal mud at Sterte Flats and Kilve Beach, with molecular weights of up to 326. PAH distributions did not indicate a significant contribution from coal as had been suggested in earlier studies (John *et al.,*1979; Cooke *et al.,* 1979). Two major hydrocarbon sources were identified; pyrogenic PAHs, probably from fossil fuel combustion and a biodegraded petrogenic input (see section above - oil). Other probable sources were algal, higher-plant and DDT-related pesticides. The nearby Liassic (oil-bearing) shale did not appear to contribute to the hydrocarbon content of the sediment therefore it was concluded that sources were principally anthropogenic.

Thus, PAH concentrations in the Severn Estuary system are reported to be relatively high in sediments and of principally anthropogenic origin. Although concentrations in Severn sediments are not as high as those recorded in the NMP from highly industrialised estuaries of north-east England, notably the Rivers Tyne and Wear (MPMMG 1998), the reported levels in Severn sediments are a potential hazard to sediment and bottom dwelling organisms and effects to some of the identified interest features of the marine site would seem to be likely[3].

Implications for biota were demonstrated in a 1996 study of bioaccumulation of benzo(a)pyrene in mussels along the Welsh coast. A steep increase from Milford Haven to Cardiff Flats, was tentatively related to the trend in urban development along the coastline (increasing eastwards) and to the delivery, from upstream, of PAHs from other parts of the Severn catchment (CEFAS, 2000b). Possible biochemical consequences for fish are discussed in section 10.2.

5.1.4. Pesticides, Herbicides, PCBs and other Endocrine Disruptors

This section deals with pesticides in the Severn Estuary pSAC and SPA, together with any evidence of their involvement in endocrine disruption. Other endocrine disruptors are described at the end of this section.

In the past, monitoring has indicated several pesticide EQS exceedences in rivers entering the Severn Estuary. In 1994, for example, these included diazinon, dichlorvos, PCSD and dieldrin (Environment Agency, 1996). Pesticide concentrations measured in waters and sediments of the pSAC (1995-1999) are listed in table 8 compiled from a review by Allen *et al.,* 2000. Compounds which may be present in elevated concentrations are discussed in the text below, under the relevant sections.

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[1] above the interim sediment quality guideline (ISQG) of 88µg kg^{-1} but below probable effects level (PEL) of 763µg kg^{-1} (from CCME, 1999)
[2] above the ISQG of 87µg kg^{-1} and PEL of 544µg kg^{-1}(from CCME, 1999)
[3] Other exceedences of PEL have been recorded for phenanthrene (English Grounds and Port Talbot) and naphthalene (Port Talbot) (Woodhead et al, 1999).

Table 8. Median concentrations of pesticides measured in water and sediments of the Severn Estuary pSAC (1995-1999). (data from Allen *et al.,* 2000)

	Seawater ng l^{-1} (median)	Sediment mg kg^{-1} dw (75^{th} percentile)
Organophosphates (total)	5	
Atrazine	10	
Simazine	11	
Dieldrin	5	0.0019 *
Aldrin	5	0.0019
Endrin	10*	0.0038 *
Isodrin	5*	
DDE	0.005	
DDT	5	0.0019*
Endosulfan	10*	
HCB	5	0.0019
Alpha-HCH	5	
Beta-HCH	5	
Gamma-HCH	5	
Trifluralin	5	

* Likely to exceed EQS or interim marine sediment quality guidelines (ISQGs)
N.B. EQS values are expressed in terms of annual average, maximum allowable concentrations or other (see Annexes) therefore it is only possible to annotate where exceedences appear likely.

Organochlorine pesticides (OCs - chlorinated hydrocarbons)

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

Many OCs are toxic List I contaminants (see annex 2), and undesirable effects on environmental quality and animal health led to a ban and/or severe restriction on the production and use of many OCs in most developed countries during the 1970's and 1980's.

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

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

DDT

DDT and its residues interfere with calcium metabolism and were responsible for the well-documented phenomenon of eggshell thinning in sea and land birds during the 1960's when many eggs did not survive incubation, and a number of species were threatened with extinction. In general, environmental concentrations of the parent compound DDT are now lower than its metabolites and, like other organic substances, preferentially absorb onto sediments, particularly where these are fine-grained and/or contain a high proportion of organic carbon (Cole *et al*., 1999). Thus, Allen *et al*., (2000) report low DDT concentrations in waters of the pSAC (table 8), which appear to be well below EQS levels of 0.025µg l^{-1} (annual average). However, maximum concentrations of DDT in sediments of the pSAC are reported as 1.9µg kg^{-1}, just above the interim marine sediment quality guidelines of 1.19µg kg^{-1}.

MPMMG (1998) reported low levels of DDT (<0.3µg kg^{-1}) in sediments of the Severn Estuary, and median concentrations of approximately 4µg kg^{-1} DDE, although 50% of samples in the survey were below detection limits. Higher levels of DDE in relation to DDT indicate that there have been no recent inputs of the parent compound.

In a study of organochlorine pesticides and PCBs in the muscle of eels from Welsh rivers, Weatherley *et al* (1997) found high concentrations of DDT (total of all isomers) in eels sampled, particularly those from catchment areas with intensive agriculture and other predominantly industrialised rivers which drain into the Severn Estuary. MPMMG (1998) report *pp*-DDE in more than 50% of the livers of fish sampled from the Severn Estuary (dab or flounder). Concentrations were approximately 10µg kg^{-1} wet weight, whilst *pp*-DDT was detected in <50% of fish sampled and concentrations were approximately 3µg kg^{-1} wet weight.

Dieldrin

Dieldrin is another endocrine-disrupting OC pesticide which may be of concern in the pSAC: concentrations in sediments (reported by Allen *et al.,* 2000) of 1.9µg kg^{-1} (table 8) exceed the interim marine sediment quality guidelines of 0.71µg kg^{-1}. MPMMG (1998) indicate that levels of dieldrin found in sediments of the Severn at Nash point (west of Llantwit Major) were <0.3µg kg^{-1}. Dieldrin is highly toxic to fish and other aquatic animals and is said to be largely responsible for the dramatic decline of the otter population in the UK during the '50s and '60s. Dieldrin used in sheep dips and seed dressings leached into water systems and became concentrated in the fatty tissues of fish such as eels, which are a major component of the otter diet. The result was a dramatic decline, which reached its nadir nationally in the early '70s, when otters were restricted to a handful of upland tributaries on the cleanest rivers[1].

Weatherley *et al* (1997) found concentrations of dieldrin to be between 10 – 50µg kg^{-1} (wet weight) in the muscle of eels from several industrialised Welsh rivers, notably the Taff (at its tidal limit), the Afon Llwyd which joins the Usk above Newport, and the Sirhowy which joins the Ebbw. Elevated concentrations were also found in eels from higher up in the freshwater, and more rural areas of the Usk, Wye and Frome (50 - 100µg kg^{-1} ww). MPMMG (1998) also reported dieldrin in fish: measurable levels of dieldrin were found in more than 50% of livers from bottom dwelling fish (dab or flounder) sampled from the Severn Estuary. Concentrations were approximately 10µg kg^{-1} wet weight. Dieldrin is the most common compound 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).

Endrin

Endrin may exceed both water and sediment guidance values in the Severn Estuary: Quality standards are 0.005µg l^{-1} (annual average) for both estuarine and marine waters, and 0.00267µg kg^{-1} for sediments (see annexes). Allen *et al.* (2000) report concentrations of 0.01µg l^{-1} in water and 0.038mg kg^{-1} in sediments (table 8). Weatherley *et al* (1997) found up to 10µg endrin kg^{-1} (wet weight) in the muscle of eels from several industrialised Welsh rivers, notably the Taff (at its tidal limit). Endrin is a persistent, acutely toxic organochlorine insecticide which was used mainly on field crops but is now banned in the UK and many other countries. Endrin exhibits very high acute toxicity among crustaceans, fish, and other aquatic organisms and is a known endocrine disruptor.

Isodrin

Concentrations of isodrin in waters of the Severn are reported to be 0.005µg l^{-1} (table 8) equal to the EQS for both estuarine and marine waters. Isodrin is one of the three organochlorine pesticides that remains in use, and is a list I substance although not on the red list as are the other 'drins'. There no reports available to review isodrin in the environment or the effects on marine life exposed to the pesticide.

γ-HCH (lindane)

Due to its toxicity and endocrine-disrupting effects, the use of lindane is currently being phased out in Europe following an EU decision in 2000 to ban it. However its use on food crops (especially cocoa) imported from other counties results in lindane residues in sewage effluent. Lindane, and dieldrin were amongst the five pesticides listed as most frequently exceeding 0.1µg l^{-1} in estuaries and coastal waters of the south west during 1993 (NRA 1995), but was below limits of detection in 50% of water samples taken from the Severn Estuary in an MPMMG study (1998). The latter found median concentrations of lindane to be below the EQS of 20ng l^{-1} in the Severn. Allen *et al* (2000) report median concentrations of lindane to be 5ng l^{-1} in the seawater of the pSAC between 1995-1999. Weatherley *et al* (1997) found high concentrations of lindane (10 – 50µg kg^{-1} ww) in the muscle of eels from the Taff (at

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

its tidal limit), and up to 10μg α- β- and γ-HCH kg^{-1} (ww) in eels from several other Welsh rivers which drain into the Severn.

Trends in lindane concentrations in the Severn are difficult to ascertain, though EA data suggest some sources appear to be declining (see section 7). The Parcom data on lindane loadings for the Severn for the period 1991-1997 are also fairly ambiguous in terms of temporal trends (figure 9) but presumably will begin to decline as EU directives filter through. Sources are mainly from rivers, together with a small component from domestic sewage.

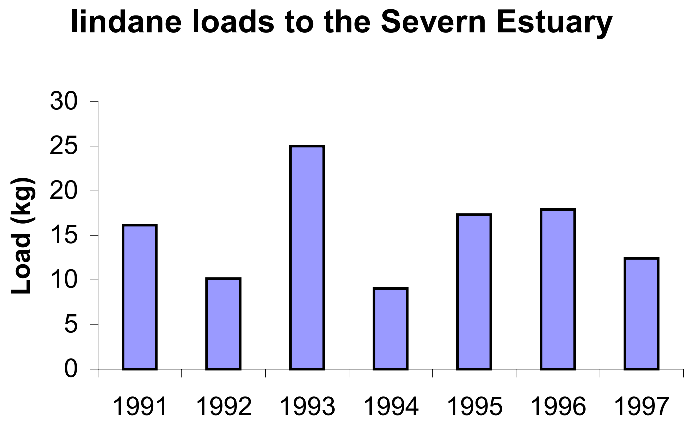


Figure 9 Temporal trends in PARCOM lindane loads to the Severn Estuary 1991-1997. (plotted from low load estimates, Environment Agency, 1999).

Endosulfan

Endosulfan is a mixture of two isomers, endosulfan a, and b, and is one of the few organochlorine pesticides which is still in use in the UK. It is a 'red list', and list II compound linked to fatal poisoning incidents in West Africa (Ton *et al.,* 2000). The high toxicity of endosulfan has led to its ban in many countries. Endosulfan has been identified as an endocrine disruptor, and is toxic to algae and invertebrates (particularly crustaceans) at concentrations above the EQS of 0.003μg l^{-1} (Cole *et al*., 1999). Allen *et al* (2000) report high levels in seawater of the pSAC (median 0.01μg l^{-1}, table 8), in excess of the water quality standard for the protection of saltwater life. No further information is available regarding endosulfan levels in the pSAC. Greve and Wit (1971) found that 75% of the endosulfan in the River Rhine was associated with particulate matter (mud and silt) therefore sediments levels in the pSAC may be important. However the ultimate fate in the marine environment, its metabolites and degradation products are not known, and further research into sources and impact of this toxic compound is recommended.

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 mean 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 affects on olfactory function in Atlantic salmon were observed after exposure to the OP diazinon at concentrations as low as 1 µg l^{-1}, and significantly reduced levels of reproductive steroids in mature male salmon parr resulted from exposure to 0.3µg l^{-1} diazinon (Moore and Waring, 1996).

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

Little information is available from the literature for levels of individual OPs in the pSAC. However, Allen *et al* (2000) report the median total organophosphate concentration in seawater of the pSAC (1995-1999) to be 5ng l^{-1} (see table 8). This is similar to levels found in other UK estuaries and does not appear to be elevated in the light of EQS values: 10ng l^{-1} for azinphos-methyl and fenitrothion, 20ng l^{-1} for malathion, and 1000ng l^{-1} for dimethoate (annual averages). Maximum allowable concentrations are 40, 500 and 250ng l^{-1} (azinphos-methyl, malathion and fenitrothion respectively). A survey by CEFAS in 1997 detected dimethoate (and similar concentrations of three carbamate pesticides) at 24 ng l^{-1} at the Nash point NMP site (CEFAS 2001b). Further upstream in the Severn Estuary reported concentrations of chlorpyrifos (OP) ranged from 34-46 ng l^{-1}, and the carbamate pirimicarb in the range 28-35 ng l^{-1}.

Simazine and Atrazine

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

Because of their major usage and high water solubility they are widespread in aquatic systems. In 1992 and 1993, elevated levels of atrazine and simazine were found in groundwater, freshwater and estuarine water of the southwest region. Principal sources were the Rivers Severn and Taff (NRA 1995). 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. Elevated concentrations were reported by MPMMG (1998) for the Thames, Tamar, Severn and Mersey, all of which have predominantly riverine sources (Evans *et al*., 1993; NRA, 1995; Environment Agency, 1996).

A recent review of endocrine disruptors in marine sites (Allen *et al*., 2000) reports median concentrations of 10ng l^{-1} for atrazine, and 11ng l^{-1} for simazine (see table 8) in seawater of the pSAC (between 1995-1999). The 1999 EA State of the Environment report suggests there have been reductions over the last decade and that there are 'no significant measured inputs of atrazine or simazine from direct or riverine sources in Wales or the south west of England'. However, in view of the analytical difficulties and low levels, load concentrations lack precision and care is needed interpreting results. A summary of the EA data made available to us is provided in section 7.

Polychlorinated Biphenyls - PCBs

PCBs have low water solubility and a high affinity for suspended solids, especially those with high organic carbon content, therefore in the aquatic environment they usually found in much higher concentration in sediments, where they are amongst the most persistent of environmental contaminants. The sediment quality guideline value for (total) PCBs in sediments is 21.5µg kg^{-1} dw, and the probable effect level (PEL) is 189µg kg^{-1} dw (CCME 1999 see annex 5).

MPMMG (1998) found elevated concentrations (up to 25µg kg^{-1} median value) of the individual congener, PCB 153, in sediments of the upper Severn Estuary in the Newport area. PCB 153 is relatively abundant therefore may be used to give an overall impression of PCB contamination.

In a study of PCB contamination of sediments from the inner Severn, Cooke *et al* (1979) reported high levels of PCBs and indications of preferential adsorption to coal dust. The highest PCB concentrations were found where sediments contained a high proportion of coal dust such as Arlingham (table 9).

Table 9. Levels of PCBs in sediments from the upper Severn Estuary (data from Cooke *et al.*, 1979)

Site	Total PCB	Arochlor 1260*
	(µg kg^{-1} dw)	
Aust (nr first Severn road bridge)	791 ± 89.1	1689
Sharpness	589 ± 101	1120
**Arlingham[1]	46 ± 4.21	85
Arlingham[2]	38.8 ± 7.18	71
Arlingham[3]	1120 ± 171	2322

* calculated   ** Sediments from Arlingham contained high coal dust fraction and were analysed before and after separation: 1 = the mixed sediment (average). 2 = coarse sand fraction with 90% coal dust removed. 3 = separated coal dust

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 very high concentrations in upper trophic levels such as fish species, birds and marine mammals. PCB burdens in muscle have been determined for eels (*Anguilla anguilla*) in Welsh rivers (NRA 1995; Weatherley *et al*. 1997). Concentrations of the PCB, Arochlor 1260, were greater than 100µg kg^{-1} (wet weight) at 46% of the sites surveyed and exceeded 1000µg kg^{-1} (ww) in eels from river stretches in industrialised areas draining into the Severn Estuary such as the Taff, the Afon Llwyd which joins the Usk above Newport, and the Sirhowy which joins the Ebbw (Weatherley *et al*. 1997). In another study, the concentrations of PCB congeners in eels, collected from the freshwater Severn at Stourport-on-Severn, Worcestershire, were much lower and ranged between 1.8 and 30µg kg^{-1} ww, with bioaccumulation factors (BFs) for different PCBs of between 4.28 and 6.27 (Harrad and Smith 1997).

MPMMG (1998) report concentrations of approximately 20µg kg^{-1} ww PCB 153 in the liver of fish (dab or flounder) from the Severn.

With the exception of isolated cases of exposure to concentrated compounds, the effects of PCBs on marine life tend to be chronic rather than acute. PCBs are implicated in endocrine disruption and linked to eggshell thinning and deformities in seabirds (Allen and Thompson 1996). Biomagnification of PCBs may result in impaired reproductive success in fish and seals (von Westernhagen *et al.,* 1981, Reijnders 1986), 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.

Thus, in higher organisms such as marine mammals some organic EDs are linked with immune system suppression and also population decline though it is difficult to establish cause-effect relationships (Allen *et al.,* 2000). In local terms, further study would be needed to evaluate the presence and possible current risk to higher organisms from substances such as, PCBs, DDE and TBT; in general terms however UK seal populations are regarded at potentially at risk based on measured residues of PCBs and DDE in their fish diet (CSL, 2000). Food items of various sorts will also be the most likely route of exposure to sea birds and waders, many of which are extremely sensitive to ED substances such as DDE, lindane, PCBs and dioxins, as has long been known. Studies of effects of these and other chemicals such as PAHs and metals (e.g. on egg shell thinning; abnormal reproductive behaviour and development) are reviewed by Fry (1995). Consumption of prey species such as molluscs (which are excellent bioaccumulators of contaminants), or accidental ingestion of sediment by waders feeding on mud-flats, might represent additional, but as yet unquantified, risks (Allen *et al.,* 2000, CSL, 2000).

Pentachlorophenol (PCP)

PCP is a list I and red list substance widely used as a wood preservative. MPMMG (1998) found PCP in <50% of samples from the Severn Estuary, and report concentrations of ~100ng l^{-1}, again well below the EQS – 2000ng l^{-1} annual average).

Volatile organic solvents (VOCs)*:*

Chloroform

Chloroform is a list I substance with a EQS of 12µg l^{-1} (annual average). MPMMG (1998) found chloroform in >50% of water samples from the Severn Estuary (upper- and mid-estuarine sites). Concentrations were approximately 0.15 – 0.75µg l^{-1} which were well below the EQS. Chloroform is an industrial solvent used in the UK in the production of fumigants and anaesthetic manufacture. It is also a principal transformation product of chlorine-based biocide products used principally 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), and it has been known for some time that reproductive tissues, especially sperm, and the immature stages of the organisms are sensitive to very low concentrations of organohalogens (Davis and Middaugh, 1978).

Carbon tetrachloride

Carbon tetrachloride is a list I substance mostly produced for use in the manufacture of chlorofluorocarbons (CFCs). Carbon tetrachloride is also used as a chemical intermediate in the manufacture of pharmaceutical and pesticide products. Carbon tetrachloride production in the United Kingdom has recently ceased and the major use for carbon tetrachloride (production of CFC-11 and CFC-12) is now in decline. MPMMG found carbon tetrachloride at only 20 out of 44 sites sampled and median concentrations were almost one tenth of the EQS of 12µg l^{-1}. Measurable levels were found in <50% of water samples from the Severn Estuary although concentrations were low (~0.1µg l^{-1}). For UK marine waters, Willis *et al* (1994) reported levels to be between <0.1 – 44µg l^{-1}. Again, higher levels were found in source dominated areas. Levels measured in the open ocean were generally much lower, at around 0.5 ng l^{-1}.

Chlorinated Ethylenes (trichloroethylene, tetrachloroethylene [perchloroethylene])

These are list I substances produced in large quantities and widely used in industry in the production of food packaging, synthetic fibres and industrial solvents. MPMMG (1998) indicate that concentrations of chloroethylenes in UK coastal and estuarine waters appear unlikely to exceed relevant EQS (10µg l^{-1} annual average) derived for the protection of saltwater life. However positive results (above detection limits) for both solvents were recorded in the Severn Estuary which receives inputs from point source discharges (NRA 1995).

Trichloroethane

MPMMG found trichloroethane (a list II substance and used as an industrial solvent) in water samples from the Severn Estuary. Concentrations were very low in this instance (~0.1µg l^{-1}), well below the EQS of 100µg l^{-1} (annual average). Potential sources of contamination include direct discharge of wastewaters, accidental spillages and deposition from the atmosphere.

Metals and organometals

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

Likewise for Pb and Hg, although it is known from mammals that ED action can occur at the level of the hypothalmic pituitary unit or on gonadal steroid biosynthesis, evidence of comparable activity on estuarine and marine biota exposed chronically in the field (see section 5.1.1) is not available (Allen *et al.,* 2000). Experimental studies on freshwater crayfish have nevertheless suggested that Cd and Hg at a concentration of 0.5µg g^{-1} body weight can arrest ovarian maturation due to inhibition of gonad stimulating hormone and serotonin, respectively (Reddy *et al*., 1997). In view of enhanced bioaccumulation of these and other metals at a number of sites (tables 5 and 6) similar reactions in marine crustaceans and other organisms from the SAC cannot be ruled out.

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

Among organometallic compounds the androgenic impact of TBT on neogastropods is most widely documented. However there is no information on this with regard to the Severn pSAC (see section 5.1.2).

In summary, information on the distributions of potential endocrine disrupting compounds in the SAC appears to be relatively limited, though there are some indications from the above data that several may be present in sufficiently high concentrations to warrant concern. A recent report by CEFAS on Endocrine Disrupters in European Marine Sites in England, commissioned by EN, has identified the Severn Estuary pSAC as a high priority for future research in this area, based on a combination of discharge volume and the conservation importance of the Estuary (Allen *et al.,* 2000). Some point source discharges are known to be associated with particular chemicals, but in most cases a cocktail of material enters rivers in the effluent from STWs (Bird, 2002; see annex 7.). The combined pressure caused by this mixture could contribute to the exhaustion of the general cortisol stress response (see above) in some species.

5.1.5 Radionuclides

There are three nuclear power plants (Berkley and Oldbury, Gloucestershire; Hinkley Point in Somerset), a manufacturer of radiopharmaceuticals in Cardiff, and a collection of smaller discharges (hospitals and research establishments) which could potentially impinge on water quality in the pSAC. A recent assessment of environmental radionuclides and the major discharges[1], for the year 2000, is provided in the RIFE-6 report published in 2001, from which the following synthesis is drawn.

Berkley Power Station ceased electricity generation in March 1989, but radioactive wastes have been and are still generated by decommissioning operations. Oldbury Power Station has continued operation. Berkley and Oldbury are considered together for the purposes of environmental monitoring since liquid radioactive wastes from both are discharged to the inner Severn Estuary. Radionuclides are analysed in fish and shellfish to assess risks from food consumption and gamma dose rates are monitored over intertidal muds to assess the risk from external exposure. Measurements of tritium in seafood were introduced in 1998 as part of the assessment of the local effects of discharges, including the radiopharmaceutical plant in Cardiff.

Radionuclide concentrations in 2000 in fish and sediments, presented in table 10, were similar to those in previous years, for similar samples. Unfortunately shrimp samples (thought to be good bioindicators) were not available in 2000, since fishing in the Estuary was poor. Most of the artificial radioactivity detected was due to radiocaesium (thought to be the combined effect of all discharges into the Bristol Channel, weapons testing, and possibly a small Sellafield-derived component). Comparisons with other areas are difficult as different bioindicators have been used, but to help put results in context, the ^{137}Cs concentration in salmon from the Severn was lower by an order of magnitude than the concentration of ^{137}Cs in sea trout from the Duddon Estuary, Cumbria (11 Bq kg^{-1}). Small concentrations of other radionuclides were detected but, taken together, were of low radiological significance.

The total gamma dose rate to the most exposed group of fish- and shellfish-consumers, including external radiation, was estimated to be less than 0.005 mSv which was less than 0.5% of the principle dose limit for members of the public of 1 mSv. This dose includes an estimate of the concentrations of radionuclides that would have been present in samples of shrimps.

Table 10. Radioactivity in food and the environment near Berkley and Oldbury nuclear power stations. 2000 (from RIFE, 2001)

Material	Location	No. of samples	Mean radioactivity concentration (wet) Bq kg^{-1}					
			^{3}H	^{14}C	^{134}Cs	^{137}Cs	^{155}Eu	^{241}Am
Salmon	Beachley	1	< 25	46	<0.07	0.27	<0.18	<0.17
Elvers	Littleton Warth	1	<25		<0.09	0.16	<0.15	<0.08
Mud	Hill Flats	2			0.75	25	<2.4	<1.5
Mud	1km south of Oldbury	2			<0.51	25	<1.7	<1.3

[1] Only the major establishments are included in the RIFE reports cited here. It is recognised however that although discharges from hospitals and research establishments may be small individually, their combined inventory (n~20) can exceed the larger establishments.

At Hinkley Point, Somerset, one station comprises Magnox reactors and the other station AGRs. The former stopped generating electricity in May 2000 and the station is now undergoing de-fuelling. A description of radiation exposure pathways from liquid effluents at Hinkley Point power station, based on survey of local habits and diet is given by Doddington *et al* (1988).

Analysis of seafood and marine bioindicators, measurements of external radiation over intertidal areas, and measurements of tritium and carbon-14 have been carried out. Environmental results for 2000 are presented in table 11.

Table 11. Radioactivity in food and the environment near Hinkley Point nuclear power stations, 2000 (from RIFE, 2001)

Material	Location	n	mean radioactivity concentration (wet) Bq kg^{-1}						
			^{3}H	^{14}C	^{54}Mn	^{60}Co	^{134}Cs	^{137}Cs	^{155}Eu
Flounder	Stolford	2		99	<0.06	<0.07	<0.07	0.74	<0.11
Shrimps	Stolford	2	1300	92	<0.12	<0.11	<0.26	1.0	<0.28
Fucus vesiculosus	Pipeline	2			<0.57	<0.25	0.55	2.6	<0.16
mud	1.6 km east of pipeline	2			<0.58	<0.52	<2.3	27	<1.5
mud and sand.	River Parrett	2			<0.49	<0.44	1.7	32	<1.8
sea water	Pipeline	2	8.2						

Material	Location	n	Mean radioactivity concentration (wet) Bq kg^{-1}					
			^{238}Pu	^{239}Pu+ ^{240}Pu	^{241}Am	^{242}Cm	^{243}Cm+ ^{244}Cm	Total Beta
Flounder	Stolford	2			<0.08			
Shrimps	Stolford	2	0.00051	0.0024	0.0016	0.00014	0.000058	
Fucus vesiculosus	Pipeline	2			<0.13			210
mud	1.6 km east of pipeline	2			<2.0			
mud and sand	River Parrett	2			1.4			

The concentrations observed in seafood and other materials from the Bristol Channel were generally similar to those measured in previous years. ^{137}Cs values in the biota of Bristol Channel (table 11) again appear to be substantially lower than equivalent samples from the NE Irish Sea (3.7 Bq kg^{-1} in Cumbrian shrimps and 18 Bq kg^{-1} in flounder from the Solway, for example). However, tritium (^{3}H) concentrations in shrimps near Hinkley (1300 Bq kg^{-1}) were much higher than in the Solway (8.1 Bq kg^{-1}). Sea water surveys also indicate the combined influence of Berkley, Oldbury, Hinkley Point and the Cardiff radiopharmaceutical plant (together with other smaller sources) on tritium distributions, which increase from about 5 Bq l^{-1} off the Gower to between 10-40 Bq l^{-1} in the Bristol Channel and Severn Estuary (RIFE 2000, 2001). Considering the different possible inputs from these and other sources (including atmospheric fallout, Chernobyl and Sellafield) apportionment is generally difficult at the low levels detected. However, a substantial component of tritium and carbon-14 in seafood was considered likely to be due to disposals from Nycomed Amersham (now Amersham plc), Cardiff. The concentrations of transuranic nuclides in seafoods and gamma radiation dose rates over intertidal sediment were of negligible radiological significance. The most exposed group of local fishermen were estimated to receive a dose of 0.012 mSv which was around 1% of the principal dose limit for members of the public of 1 mSv.

The Amersham laboratory produces radiolabelled compounds used in research. Liquid wastes are discharged into the Severn Estuary via the sewer system. Routine monitoring, includes consideration of consumption of locally produced food and external exposure over muddy, intertidal areas. Measurements of external exposure are supported by analysis of intertidal sediment. Indicator materials including sea-water and *Fucus* seaweed provide additional information. Supplementary monitoring and research was undertaken in 2000, which mainly targeted organic tritium in foodstuffs. (Swift, 2001; Leonard *et al*., 2001a).

The results are presented in table 12. The main effect of liquid discharges appears to be enhanced tritium and carbon-14 activities. Concentrations in surface sediment, seaweed (*Fucus vesiculosus*) and mussels (*Mytilus edulis*) were in the order of 6 x 10^2, 2 x 10^3, and 10^5 Bq kg^{-1} (dry weight), respectively. The levels of total tritium in fish and shellfish increased in 2000 compared with 1999 (mean concentrations in flounder, the main indicator used, increased from 23,000 Bq kg^{-1} to 54,000 Bq kg^{-1}). This change runs counter to the trend in the overall reported discharge of tritium from the site (1998 - 277 TBq; 1999 - 105 TBq; 2000 - 80 TBq) and may be due to short-term biotic/sampling variables, rather than overall increase in bioavailability. Also, the concentrations observed in biota appear to be strongly dependent on the relatively small amounts of organic tritium-labelled compounds in the wastes, which may not have declined in line with overall reductions in tritium discharges (Leonard *et al*., 2001a). It is possible that sediment could act as a store for organically bound tritium and it may take some time for activity levels to decline (McCubbin, 2001). The results of sample analyses show that over 90% of the total tritium was organically bound. It is suggested that bioaccumulation of ^3H by benthic organisms and demersal fish occurs primarily via a pathway of physico-chemical partitioning into particulate organic matter, and subsequent transfer up a web of sediment dwelling microbes and meiofauna. Variations in ^3H accumulation between individual organisms have been interpreted in terms of their different feeding behaviour. Relatively low concentrations were observed in the herbivorous winkle (*Littorina littorea*) and the pelagic Sprat (*Sprattus sprattus*) compared with higher levels in other benthic organisms and demersal fish. An overview of what is known about tritium behaviour in the Severn Estuary has recently been published by Williams et al., (2001). Clearly, however, this is an issue for further research since relatively little is known of the bioavailability, assimilation pathways and effects (e.g. genotoxicity) of organically-bound tritium on marine life.

Freshwater fish from the River Taff may have also accumulated tritium, though there are no authorised discharges made directly to the river. The Cardiff sewerage system has, in the past, discharged into the Taff via the combined sewer overflow, following storm surges (Swift, 2001). Authorised discharges into the sewerage system could thus be transferred to freshwater biota after heavy rain (and hence presumably to the sea). It is also possible that some tritium may have been transported upstream, from the Estuary, prior to the Cardiff barrage closure.

The presence of relatively high levels of tritium in sediments and biota in the area has resulted in a requirement for tritium discharges to be reduced significantly. It is believed that the company has subsequently withheld from discharging liquid containing very high concentrations of tritium and is developing means to trap and recycle much of the radioactive carbon and tritium, instead of discharging it to the environment. In addition, the introduction of a new STW in 2002 should retain particle-entrained tritium along with the sludge, so that any discharge to the Estuary is as tritiated water. An application for a revised discharge authorisation has been made.

Concentrations of other radionuclides in aquatic samples from the Cardiff area were low (table 12) and can be explained by other sources. Gamma and Beta dose rates over sediment were also low. The dose to the most exposed group of fish and shellfish consumers including external radiation was 0.064 mSv which was about 6% of the principle dose limit for members of the pubic of 1 mSv.

Table 12. Radioactivity in food and the environment near Cardiff, 2000 (from RIFE, 2001)

Material	Location	n	Mean radioactivity concentration (wet) Bq kg^{-1}					
			Organic ^{3}H	^{3}H	^{14}C	^{134}Cs	^{137}Cs	^{155}Eu
Flounder	East of new pipeline	14	51000	54000	730	<0.05	0.27	<0.09
Sole	East of new pipeline	3	28000	43000	460	<0.14	0.19	<0.20
Cod	East of new pipeline	1		4500	86	<0.13	0.71	<0.20
Mullet	East of new pipeline	1	420	450	92	<0.14	0.34	<0.32
Brown Trout	River Taff	1	6200	7800	150			
Chub	River Taff	1	8100	10000	150			
Barbel	River Taff	1	30000	30000	490			
Grayling	River Taff	1	14000	17000	230			
Roach	River Taff	1	17000	15000	130			
Eel	River Taff	1	23000	30000	380			
Green Crabs	East of new pipeline	1	57000	59000	620			
Shrimps	East of new pipeline	1	39000	40000	470			
Mussels	Orchard Ledges	14	24000	27000	440	<0.14	0.43	<0.28
Whelks	East of new pipeline	1	65000	74000	1100			
Fucus vesic.	Orchard Ledges	12	310	310	21	<0.05	0.32	<0.11
Mud	Orchard Ledges East	12	710	460	17	<0.55	16	<1.9
Sea water	Orchard Ledges	12		7.1				
Sea water	Orchard Ledges East	2		8.1				

A survey was undertaken in 1997 to determine the concentration of technetium-99 in selected animals (e.g. winkles) and plants around the Welsh coast. This showed lowest amounts present at sites in the Bristol Channel, increasing northwards, dominated by the major source at Sellafield. Health risks to humans were considered negligible (Leonard *et al.,* 2001b).

5.2 Non-Toxic Contaminants

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

Table 13. Contributions (%) from rivers, discharges and sludge to total inputs of various contaminants to the Severn Estuary (adapted from Owen 1984)

Determinand	Rivers and streams	Domestic sewage	Industrial effluent	Sludge dumping	Total input (10^3 kg day^{-1})
Flow	96.4	2.7	0.9	-	-
Ammoniacal N	16.5	42.6	40.9	-	35.9
TON	91.8	1.29	7.02		102
TIN	74.3	8.81	16.3	0.67	134
Orthophosphate	37	16.3	44	2.6*	16.1
Silicate	90.7	4.46	4.65	-	183
Suspended solids (105°C)	75.9	4.7	18.1	1.1	2370
DOC	48.2	35.6	16.1	-	195
POC	57.1	13.8	26.2	2.82†	602
BOD	29.6	39.8	20.1	10.6	256

* Estimated phosphorus content
† Assumes 65% POM

5.2.1 Organic Enrichment

Natural sources of organic carbon, or organic matter, include river-borne phytoplankton and organic detritus, run-off from the land and marginal vegetation. These inputs are supplemented considerably in urbanised estuaries by anthropogenic point sources such as sewage effluent, some industrial effluents and fish and shellfish installations. The addition of large amounts of organic matter from anthropogenic sources can exceed the capacity of parts of the marine environment to process it, resulting in accumulation, usually in the sediments.

Organic matter occurs in natural waters in dissolved (dissolved organic matter DOM) and particulate (particulate organic matter POM) forms. Much of the biological DOM is metabolised by heterotrophic bacteria, while most of the DOM with geological origins (e.g. humic substances) is resistant to microbial breakdown. Some other animals can use biological DOM but are largely out-competed by bacteria.

Concentrations of POM depend significantly upon discharge and on concentrations of suspended particulate matter. During summer months, the 'living organic carbon' of algae accounts for much of the POM in all but the most polluted rivers (Tipping *et al*., 1997).

Organic carbon is readily assimilated into the tissues of marine organisms, and can be transformed from particulate to dissolved and lost to the atmosphere as CO_2 by respiration. Organic matter may exert a controlling influence on the oxygen mass balance of aquatic systems by stimulating the biological productivity both in the water column (increasing biological oxygen demand - BOD), and in the sediment (increasing sediment oxygen demand - SOD) (Tipping *et al.,* 1997). SOD is related to the settlement of suspended solids with a high organic content (as is the case for suspended solids discharged in STW effluent). Thus, a relationship exists between water column dissolved oxygen status and organic enrichment, which might contribute to the oxygen sag which occurs in the upper reaches of the Severn Estuary (see section 5.2.5).

In addition to its relationship with oxygen status of rivers and estuaries, DOC is intimately involved in a range of important biogeochemical processes. Furthermore the potential toxicity of industrial and urban effluents may be ameliorated through complexation of trace metals (Suffet and McCarthy 1989) and partitioning of organic pollutants such as PAHs and PCBs (Chiou *et al.,* 1986). DOC adsorbed onto particles causes the alteration of their physico-chemical properties and can modify bioavailability in various ways (Bryan and Langston 1992: Keil *et al*., 1994).

Organic carbon enters the Severn Estuary from several sources: Randerson (1986) modelled carbon flow in the *Spartina* marshes of the Severn and concluded that *Spartina* marsh and mudflat systems contributed in excess of 90% of autochthonous carbon inputs to the Estuary. The model took into account extensive grazing on the upper marsh system, which is covered only occasionally by high spring tides (Smith 1979) and was thought to remove much of the organic carbon produced, and phytoplankton production which was reported to be relatively low because of the high suspended sediment levels in the Estuary (Kirby and Parker, 1977).

Allochthonous sources of organic carbon were considered by Randerson (1986) to arise principally form sewage discharges. However, Owens (1984) reported results of an intensive investigation programme to establish the quantities of contaminants discharged to, or entering the Severn Estuary, which pointed to rivers and streams as the principal source for both dissolved and particulate carbon (table 13). This study also apportioned % inputs to geographical areas of the Severn (and Bristol Channel) and suggested that the major inputs of DOC and POC originated in the mid-Estuary (roughly between Aust and Burnham-on-Sea). These waters include the estuaries of many of the principal rivers which discharge into the Severn including the Wye, Usk, Taff, Ely, Avon, Axe, Brue, Yeo). The area also receives discharges from the larger urban conurbations of Bristol, Newport, Cardiff, and Weston Super-Mare.

Sources, chemistry and fate of DOC in the Severn Estuary were investigated by Mantoura and Woodward (1983). Concurring with Owens (1984), findings in this study also indicated that rivers were the principal source of DOC in the Estuary. 70% of the total flux (up to 2.67×10^{10} g C year) to the Bristol Channel occurred during the winter quarter and dissolved carbon concentrations ranged between 3.1 – 7.8 mg l^{-1}.

A seasonal pattern of autochthonous DOC production by phytoplankton was superimposed on the inputs from rivers but amounted to only a negligible contribution (<10%) of the total flux and occurred mainly in the less turbid, euphotic marine waters (salinity > 32 ‰).

Mantoura and Woodward (1983) also reported a linear relationship between river flow and DOC concentration in the Severn. This was interpreted as characteristic of soil-derived carbon inputs, as concentrations of DOC in rivers originating from densely populated catchment areas usually vary inversely with river flow (Stumm and Morgan, 1981). Riverine DOC was diluted conservatively in the Estuary and removal by chemical or microbial degradation was not indicated; adsorption onto the surfaces of estuarine particulates was estimated at <0.2% of the DOC. This finding has implications for the toxicity and bioavailability of certain contaminants as such adsorption processes can profoundly affect the surface chemistry of particles (see above).

5.2.2 Nutrients

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

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

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

The principal effect of extreme nutrient enrichment is eutrophication, defined as 'the enrichment of natural waters by inorganic plant nutrients, which results in the stimulation of an array of symptomatic changes' (EA, 1998). These changes include an increase in phytoplankton growth that is reflected by an increase in chlorophyll α concentrations. Dissolved oxygen concentrations in the water column fluctuate during the growth phase of a 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. Nitrogen levels can be monitored as nitrate, nitrite and ammonium concentrations in tidal waters which, when added together, produce total inorganic nitrogen (TIN), an approximation of bioavailable nitrogen. Phosphorus is present in the aquatic environment in both inorganic and organic forms, although the principal inorganic form is orthophosphate and is measured as dissolved orthophosphate (soluble reactive phosphate SRP), or as total reactive phosphate (TRP) by measuring phosphate in unfiltered samples.

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

{Note: It is generally assumed that an N:P ratio of 10:1 is ideal for plant growth. At N:P ratios >10:1 (mainly in freshwater, P is thought to be limiting, and at N:P ratios < 10:1 (mainly in seawater) N is thought to be limiting. (However there are 3 large coastal areas in the UK where P may be limiting – from the Solent to Dartmouth; around the Severn from Padstow to Oxwich and from the Humber to Essex) (Parr *et al.,* 1999). In the Severn, there is an additional factor; when enrichment is such that nutrients are sufficient to promote algal growth, light limitation may govern primary production (see section 5.2.3)}.

Owens (1984) has estimated the contribution of nutrient inputs from various sources to the Severn Estuary (table 13). The greatest proportion of inorganic nitrogen (TIN) came from rivers and streams (mainly diffuse sources), and ammoniacal nitrogen form sewage discharges. Whilst these freshwater sources contributed substantial inputs of orthophosphate to the Estuary, industrial effluent and domestic sewage comprised the greater proportion. It must also be remembered that freshwater sources may introduce sewage effluents from the catchment to the Estuary.

The lower Bristol Avon was designated as a Sensitive Area (Eutrophic) in 1998 under the provisions of the UWWTD, and nine qualifying STWs were identified and required to install nutrient stripping treatments by 2004. Subsequent monitoring continued to provide evidence of eutrophication in the Lower Avon and identified four additional qualifying STWs which comprised the ecology of the River. It was recommended that similar treatment be installed at the additional works (EA, 2001).

Owens (1984) apportioned inputs of nutrients to geographical areas of the Severn (and Bristol Channel) and found that the major proportion of TIN inputs were into the inner Estuary and orthophosphate to the outer-Estuary (table 14). Ammoniacal N inputs were greatest in the outer Estuary, although the highest concentration of ammonia in the water column (1.6mg l^{-1}) was recorded below Maisemore Weir (at the head of the Estuary, figure 1) probably reflecting the lower volume of receiving waters.

Table 14. Proportionate mass (%) inputs to geographical areas of the Severn (adapted from Owens, 1984)

Determinand	Outer Bristol Channel	Inner Bristol Channel (Minehead to Burnham-on-Sea)	Outer Severn Estuary (Burnham-on-Sea to Aust)	Inner Severn Estuary (Aust to Maisemore Weir)
TIN	14.9	5.9	37.9	41.3
TON	13.3	7.4	27.3	52
Ammoniacal N	19.5	1.5	68.1	10.9
Orthophosphate	8.2	2.4	68.3	21.1
Silicate	19.4	3.4	31.1	46.1

Owen (1984) reported that TIN, orthophosphate and silicate concentrations generally decreased with increasing salinity although some discontinuities were observed which reflected the effect of seasonality and localised inputs.

These discontinuities were enlarged upon by Morris (1984) who noted that the waters of the Severn Estuary and Bristol Channel do not constitute a simple two-component mixing gradient between the primary river input and offshore waters, as the River Severn supplies only about a quarter of the total freshwater input to the system. The remainder enters through a multiplicity of additional natural and man-made discharges located throughout the system (shown in figures 1 and 2) but concentrated on the outer Estuary. Due to the complexity of the inputs, some dissolved constituents do not show linear relationships with salinity in this area: Silicate and nitrate in freshwater inputs are higher in winter than summer and both show a linear correlation with salinity in winter indicating conservative mixing. Concentrations of nitrate in winter ranged from roughly 6mg l^{-1} in freshwater to 1.0mg l^{-1} at 32‰, and for silicate roughly 3.5 – 1.0mg l^{-1} (fresh- to sea-water). However, in summer, there is depletion of nutrients by primary production in the lower Estuary, toward the sea, where lower turbidity levels allow more light penetration. Dissolved silicate along the Estuary ranges from roughly 0.2mg l^{-1} in freshwater, to 1.5mg l^{-1} at 20-25‰ and down to 0.5mg l^{-1} in fully saline waters.

Phosphate in freshwater inputs to the Estuary is also seasonally affected, with higher levels in summer - dissolved concentrations in freshwater are approx 0.6mg l^{-1}, and reduce to ~0.05mg l^{-1} at 32‰. During winter, phosphate in river inputs is lower (~0.3mg l^{-1}), but ‘appreciable’ phosphate concentrations in industrial and domestic discharges to the outer Severn result in a more even distribution along the Estuary with roughly 0.1mg l^{-1} at 32‰ (Morris 1984).

Information gathered between 1990-1993 for the Paris Convention (NRA 1995) reported that the River Severn discharged the second largest amount of nitrogen of any English or Welsh estuary consistent with its size, and the Wye and the Bristol

Avon were amongst the top 14 for total N discharges. NRA (1995) also apportioned loading to the Bristol Channel and reported that the greatest source of nitrogen was rivers, which supplied more than twice the amount of N to the Channel than sewage or industrial discharges. MPMMG (1998) also found (winter) nitrate concentrations in the Severn to be amongst the highest around the UK, partially reflecting the high loadings reported in the Paris Convention Survey (NRA 1995).

For orthophosphate, the River Severn ranked fourth for discharges. The principal source of orthophosphate was reported to be rivers, but in 1990, sewage discharges almost equalled the riverine inputs the Bristol Channel. The relative proportion supplied by sewage discharges was reduced by 1993 however, when rivers supplied the vast majority. A survey of the quality of coastal waters by MPMMG (1998) reported that orthophosphate concentrations were relatively low in the Severn compared to other UK estuaries.

On balance, the available literature does not appear to indicate major nutrient enrichment problem in the Severn Estuary. Though the proportion of phosphate and ammonia from sewage and industrial discharges is reported to be greater to the outer Estuary (Burnham-on-Sea to Aust on the English side: Lavernock Point to Caldicot on the Welsh side), it is considered that the characteristics of the Estuary - dilution, turbulence and the energetic tidal regime –combine to prevent any widespread eutrophication.

5.2.3 Chlorophyll *a*

Chlorophyll *a* concentrations in excess of 10µg l^{-1} are considered indicative of phytoplankton blooms, though not necessarily eutrophication (Anon, 1993). Phytoplankton production is reported to be relatively low in the Severn because of the high suspended sediment levels in the Estuary (Kirby and Parker, 1977). Results for the Severn Estuary from the NMP programme conducted between 1992 and 1995 also indicate relatively low concentrations of chlorophyll-*a* for both winter and summer (~5µg l^{-1}) with the maximum values occurring in the upper Estuary (MPMMG, 1998). Owens, (1984) noted that concentrations of chlorophyll-*a* were consistently greater in the freshwater entering the Estuary than in the outer Estuary itself, and concentrations declined seaward from an average of 80µg l^{-1} below Maisemore Weir, to less than 5µg l^{-1} at the Severn Bridge (about 50km downstream of Maisemore) and remain at approximately that level throughout the lower Estuary.

Distributions of chlorophyll-*a* are likely to be influenced in estuaries by a complex array of factors. In the Severn Estuary, concentrations of nutrients may be sufficient for phytoplankton proliferation but high turbidity limits sunlight which is essential for algal growth (MPMMG, 1998). The high levels of suspended solids (see section 5.2.4) result in a euphotic zone which amounts to only ~3% of the water column in some stretches (Joint and Pomroy, 1981).

Vertical mixing processes in the Estuary are important to phytoplankton ecology: Uncles and Joint (1983) found that phytoplankton, as measured by chlorophyll-*a* concentration, is homogeneously distributed with depth in the water column and the vertical mixing is too great to allow development of high concentrations of phytoplankton cells in the shallow surface, euphotic zone. Joint (1984) reports concentrations which range from ~0.8µg l^{-1} in summer to 1.6µg l^{-1} in summer. These factors result in less seasonal variation in primary productivity than in other regions.

However, occasional blooms of the colonial Haptophyte *Phaeocystis* have been reported to occur in the lower Estuary. In June 1974, higher concentrations of chlorophyll-*a* (~12µg l^{-1}) were recorded which were due almost entirely to *Phaeocystis*. The colonial *Phaeocystis* appeared better able to exploit the low light levels than a population in which the species was absent, but Joint (1984) considered this physiological adaptation unlikely to provide a complete explanation for the development of *Phaeocystis* blooms.

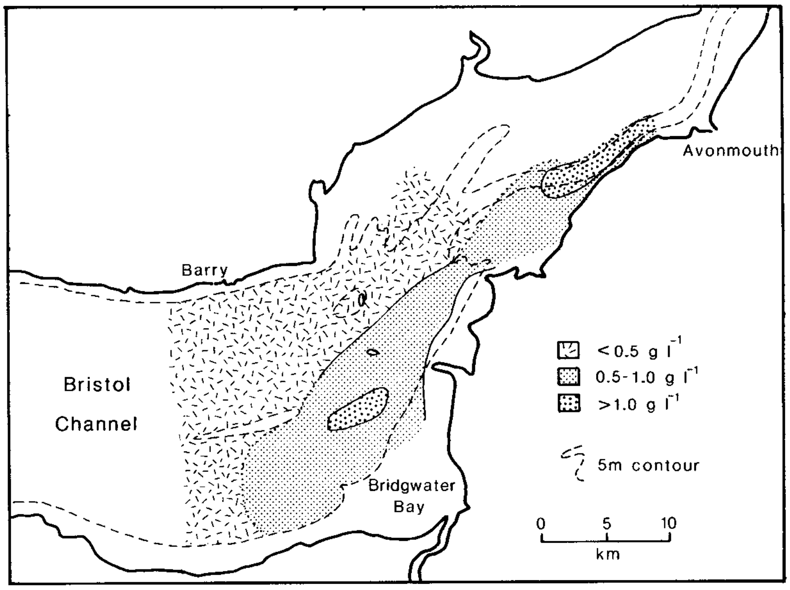
There are no recent studies of phytoplankton (or chlorophyll-*a*) in the Severn available, although Dong *et al.,* (2000) looked at environmental limitations of phytoplankton in estuaries and noted that the significant correlations between nutrient loads and chlorophyll-*a* values which link nutrient enrichment to algal biomass in most estuaries do not exist in the Severn. This was attributed to turbidity restricting algal growth.

5.2.4 Turbidity and Siltation.

The Severn is one of the most turbid estuaries in the UK. The wide mouth of the Bristol Channel and the converging coastlines of the Estuary create an extremely energetic system with very large tides and turbulent conditions. The pronounced turbulence prevents vertical stratification and maintains massive amounts of particulate matter in suspension (Glover, 1984). In common with other estuaries, the Severn contains more sediment in suspension than the visible annual inputs, and there is a continual exchange of material from areas of erosion to areas of deposition through the turbidity maximum which occupies the whole of the Estuary, east of Bridgwater Bay (Dyer, 1984). Parker and Kirby (1982) used a combination of vertical profiling and horizontal traverses using optical turbidity meters to show that the highest concentrations occur on the SE (English) side of the Estuary, and there is a relatively sharp front roughly down the centre of the Estuary separating the most turbid water from clearer water on the Welsh (NW) side (figure 10). This is a persistent feature which was said to occur on both flood and ebb, as well as spring and neap tides.

Vertical profiling of turbidity has shown that the suspended sediment concentration varies throughout the tidal and spring/neap cycle; at maximum (spring tide) current the concentration is relatively uniform from surface to bed – typically 5000mg l^{-1}. As the current diminishes the turbulence is insufficient to maintain the suspension and the particles settle causing ‘steps’ in the concentration profile, (Kirby and Parker, 1982).

A



B

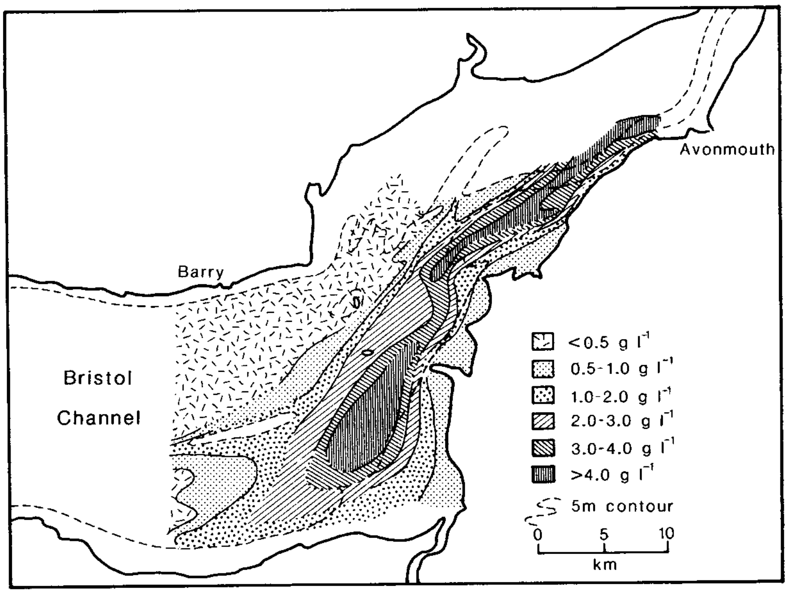


Figure 10. Distribution of surface (A) and bed (B) suspended solids – mean concentrations during spring tides (from Parker and Kirby 1982 with permission)

Suspended solids from inputs contribute to the turbidity caused by re-suspension of estuarine sediments. Table 15 (a) shows the relative contribution of suspended solids from various sources in the Estuary. Rivers and streams supply the greatest proportion, which is not surprising as, for example, the turbidity maximum of the Usk is reported to almost exit the Estuary during spring tide low water events (see section 5.2.5). Table 15(b) serves to confirm this as it shows that the geographical areas receiving the highest proportion of suspended solids from these inputs are those with the principal rivers (Usk, Avon, Wye, Taff).

Table 15. (a) Contributions (%) of suspended solids (105ºC) from rivers, discharges and sludge to the Severn Estuary. (b) Proportionate mass (%) inputs of suspended solids (105º) to geographical areas of the Severn (adapted from Owen 1984).

(a)

Rivers and streams	Domestic sewage	Industrial effluent	Sludge dumping	Total input (10^3 kg day^{-1})
75.9	4.7	18.1	1.1	2370

(b)

Outer Bristol Channel	Inner Bristol Channel (Minehead to Burnham-on-Sea)	Outer Severn Estuary (Burnham-on-Sea to Aust)	Inner Severn Estuary (Aust to Maisemore Weir)
10.3	1	40.9	47.8

There are few indications from the literature, as to the amount of re-suspended particulates relative to those attributable to inputs. Kirby, (1994) considered riverine inputs to be small (1.0M tonnes/yr) compared to the instantaneous mobile, fine sediment loads on spring tides, which may exceed 30M tonnes. This presumably will vary with flow rates, tidal cycles and tidal amplitude, but may be important in determining contaminant (especially metal) distributions in the Estuary (e.g. see section 5.1.1. 5.1.2, 5.1.3). The reactivity of these contaminants is a function of the highly dynamic and cyclical nature of the suspendable sediment transport system (Ackroyd 1983).

The behaviour of suspended sediment over the spring-neap tidal cycle and also seasonally is also probably a major factor in relation to the biology. Based on a PhD thesis 1976/77 by Arwen Griffiths, (University of Bristol), work on the spring-neap cycling of sediments in the upper reaches showed deposition of fine sediment during neaps and erosion of fines and deposition of coarser sediment during springs. There can therefore be short-term variations of substrate in the upper reaches which must affect the benthos. It is expected that there will be a similar seasonal signal which will affect the substrate in various parts of the Estuary.

5.2.5 Dissolved Oxygen (DO)

MPMMG (1998) indicated that dissolved oxygen conditions in the Severn Estuary are similar in winter and summer (~12mg l^{-1}). However, oxygen sags have been reported intermittently, both in the Severn itself and in the estuaries of several of the principal rivers opening into the Severn: Historically, spring and neap tide DO profiles of the Severn both showed a sag (down to 70-80% saturation) centred above Avonmouth (Winters, 1973). During a spring tidal cycle the oxygen depletion occurred 35 – 60km downstream from Maisemore Weir (between Sharpness and Aust), whilst the neap tide depletion zone was between 0-30km. These were high-water profiles indicating that the region of maximum oxygen demand would have been downstream of the sag. Maskell (1985) observed that the greater depletion of oxygen during

spring tides compared to neaps is a striking phenomenon in turbid estuaries of the UK, especially tributary estuaries of the Severn. This is the reverse of the situation in estuaries with low turbidity.
Owens (1984) indicated that surface water throughout most of the Severn is frequently supersaturated with oxygen in respect to air but also reported an oxygen sag in the upper 20km reach of the Severn, below Maisemore Weir. Owens suggested that the depletion could be attributable to the resuspension of deposited organic material during flood and spring tides. The oxygen sag was reported here to occur primarily under summer conditions of low flow and high temperature. Morris (1984) noted that the sources and mechanisms controlling the extent and location of the oxygen demand had not been fully characterised but concurred with Owens that it was exerted predominantly by particulate organic materials, of natural and anthropogenic origins, which were hydrodynamically localised within the low salinity region, either in suspension or oscillating between suspended and deposited states.

Table 16. Proportionate mass (%) BOD in inputs to geographical areas of the Severn (adapted from Owens, 1984)

	Outer Bristol Channel	Inner Bristol Channel (Minehead to Burnham-on-Sea)	Outer Severn Estuary (Burnham-on-Sea to Aust)	Inner Severn Estuary (Aust to Maisemore Weir)
BOD	21.6	3.6	60.2	14.6

The input of effluent to the Estuary has a high Biological Oxygen Demand (BOD), particularly in the outer Estuary (see table 16), although the larger volume of receiving waters in this region probably contains more than sufficient DO to fulfil the demand. However in the more restricted upper 20km of the Estuary, oxygen levels may be more readily depleted.

Dissolved oxygen sags have also been reported in Usk Estuary, at the confluence of the Ebbw and Usk with the Severn (Henderson, 1972: Maskell, 1985; Larsen *et al.,* 1992; Parker *et al.,* 1994). In periods of mid- to high-flow DO can be up to 12.8mg l^{-1} (Parker *et al.,* 1994) but minimum DO concentrations of 1.5mg l^{-1} have been recorded during low water spring tides when much of the flow is effluent (Larsen *et al.,* 1992). The oxygen sags also coincide with high levels of suspended solids, and during spring tide low water events, the turbidity maximum of the Usk almost exits the Estuary (Parker *et al.,* 1994).

In the Wye and Parrett Estuaries, DO levels as low as 60 and 40% saturation (respectively) have been shown to occur during spring tides (Welsh Water Authority, 1981). Maskell (1985) modelled particulate BOD and oxygen balance in the Parrett Estuary during a period of steady low fluvial flows taking account tidal propagation, saline intrusion, sediment transport and particulate BOD. Simulations included various tidal conditions and effluent flows. Principal conclusions were that the BOD of naturally occurring marine detritus in the Estuary, which is periodically resuspended, was almost certainly more than the BOD due to dissolved and particulate effluent discharges to the Estuary, and DO levels would naturally fall below 60% saturation during spring tides. The detritus is derived from the Severn

Estuary and a major source was thought to be dead phytoplankton from the more turbid parts of the Estuary.

Also there is a draft report on the historical data for DO in the Parrett Estuary which does not come to any major conclusions on the trends in DO in the Estuary. However, in the 1960's there were persistent anoxic conditions found which do not appear now to prevail. There were also major salmon kills in the Estuary in the 1950's and 60's which were probably related to the anoxic conditions.

6. STUDIES ON BIOLOGICAL COMMUNITIES

There have been numerous studies on biological communities in the Severn Estuary. Proposals for tidal power prompted a host of investigations, therefore many focus on pre-barrage surveys and (anticipated) post-barrage changes. Regular sampling at Hinckley Point power station has been the basis for a long-term (21-year) time series study of animal abundance, which, together with physico-chemical and meteorological data, form the Severn Estuary Data Set (SEDS) - this unique data set, is probably one of the largest time series for a community in existence. The data set holds time series for about 80 species of fish, 20 species of macro-crustacean and about 40 species of mysid and other small planktonic organisms (see reference list at the end of this report). More information can be obtained from the web site www.irchouse.demon.co.uk.

An early series of studies on the biology of the Severn and Bristol Channel was introduced in the Proceedings of the Bristol Naturalists Society (Yonge, 1937) and provides a comprehensive species-by-species guide to the distribution of organisms. The first was an account of ecological comparisons made between two sites, Royal Beach and the dock at Portishead (Purchon, 1937), the work included full fauna lists (as far as possible), and an indication of the range of chemical and physical factors at the sites. This early faunistic work was later extended to include more sites, with the aim of a full description of the estuarine fauna linking the studies with work on morphological and physiological adaptations to the wide range of environmental variation in the region. The studies were over a 20-year period, although the Bristol Naturalists Society continues to publish reports on the natural history of the area. Further information on studies relating to key organisms, or groups of organisms are reviewed below.

Plankton

The shallowness of the euphotic layer has a considerable influence on phytoplankton ecology in the Severn Estuary (see section 5.2.4). However, Joint and Pomroy (1981) estimated the annual primary productivity to be 6.8g C m^{-2} y^{-1} in the outer Estuary, and noted that even in the most turbid regions phytoplankton were present and capable of photosynthesis in spite of severe light limitation.

The standing stock of phytoplankton is quite low and does not vary much between regions of the Estuary. In the outer Estuary, there is less seasonal variation in primary production than further out into the Bristol Channel, probably as a result of lower light levels (Joint, 1984). There are no accounts available in the literature detailing the phytoplankton species in the Estuary, and the only named planktonic species is *Phaeocystis* in relation to an algal bloom (see section 5.2.3).

However, Underwood (1994) has described epipelic (sediment surface) diatom assemblages. Intertidal epipelic diatoms are important primary producers in estuarine systems (McLusky, 1989) serving as a resource for many groups of deposit feeders (Morrisey, 1988) and as agents of biogenic stabilization (Paterson and Underwood, 1992: Underwood and Paterson, 1993a,b). Underwood (1994) investigated species composition at three sites, lower- (Sand Bay), middle- (Portishead) and upper- (Aust) - Estuary between March 1990 and February 1991. Salinity (mean) varied between sites, 31.6, 29.9 and 25.6‰, respectively) but as a single factor, was not found to be a significant variable in determining species assemblage.

Diatoms were the dominant algal group comprising >95% of living cells in almost all samples, but occasionally, non-flagellated euglenoid *Euglena deses* was abundant. *E. deses* was present in greatest numbers on the mid-shore at Portishead with a peak in abundance (50% of intact cells) during late July. Over 60 diatom taxa were identified although only 15-20 species occurred regularly and 12 species were dominant.

Nitzschia epithemoides dominated upper and mid-shore assemblages during the early summer months and *Navicula pargemina* during the spring and autumn. Throughout the year there were high relative abundances of *Rhaphoneis minutissima* on lower shores, although in winter, this species was more abundant on upper- and mid-shores. Seasonal changes in assemblages were more pronounced on the upper shores of intertidal mudflats. The upper reaches of the Severn mudflats are physically more stable than low shore but subject to a greater degree of environmental variables (e.g. temperature and desiccation) allowing greater temporal variation in the dominant diatom species, each showing different growth optima across the wide spectrum of ambient conditions. Generally these upper- and mid–shores were dominated by single taxa, (e.g. *N. epithemoides* and *N. pargemina*), whilst diversity was greater on the lower shore. This finding confirms the findings of Sullivan, (1978) who suggested that the dominance of a single species is prevented in constantly disturbed habitats, resulting in high species diversity (the 'intermediate disturbance' hypothesis). Lower shore assemblages also contained higher proportions of epipsammic (attached to sand grains) species (*Coscinodiscus* sp., *Cymatosira beligica*). The relative abundance of these species, along with *R. minutissima* correlated with periods of low biomass and temperature, high empty frustule counts and high diversity values. The author suggested that these cells are carried around the Estuary in suspended sediments and deposited on the mudflats after periods of rough weather. Motility is considered a necessary survival strategy for epipelic diatoms, to avoid burial (Hay *et al.,* 1993), but repeated re-suspension in the water column may prevent burial of non-motile epipsammic taxa, and is therefore especially important for primary production in the Severn Estuary because of its strong tidal and storm-driven currents.

One other study focussing on microalgae is that of Cox (1977), which investigated the distribution of tube-dwelling diatoms in the sediments of the Estuary between Framilode in the upper Estuary and Beachly (below Aust). As would be expected, the results showed a decrease in freshwater species with distance downstream from Framilode with those least tolerant to salinity disappearing first, e.g. *Nitzschia angularis* and *Navicula incerta*. Other freshwater species in evidence at the upper estuarine sites were *Navicula viridula, Nitzschia filiformis, Cymbella prostrata* and *Frustulia vulgaris.* Brackish water species *Navicula ramosissima* and *Berkeleya rutilans* were very widespread in the Estuary as a whole and occurred as far down as Brean. However*, B. rutilans* is not truly a euryhaline species and was confined to the supralittoral zone on those shores without a large freshwater inflow.

Zooplankton
The most comprehensive account of zooplankton in the Bristol Channel and the Severn Estuary was carried out by the Institute for Marine Environmental Research (IMER) over the 10 years between 1971 – 1981 as part of a multi-disciplinary study.

Several publications resulted from this work, many in relation to the proposed barrage, but perhaps the best account is given by Williams and Collins (1985) which is a zooplankton atlas of the Estuary with a full list of organisms present. It also includes a full account of environmental variables (temperature, salinity, extinction depth [light], chlorophyll *a*, tidal residuals, bed stress, mixing times and sediment type) and a bibliography of published studies resulting from the IMER work. It would not be useful to attempt to reiterate all the results of this work here, but to summarise:

The communities or associations of zooplankton species described are normal for estuaries in northern latitudes, both in abundance and in species composition. Salinity was found to be the most important environmental variable affecting the distribution of zooplankton in the Estuary although other factors may exert some influence including seasonal changes, hydrodynamics and succession of species. Four groups, or assemblages were described; these were numerically dominated by the calanoid copepods *Eurytemura affinis, Acartia bilfilosa* var. *imermis, Centropages hamatus* and *Calanus helgolandicus* and represented true estuarine, estuarine and marine, euryhaline marine and stenohaline marine, respectively. The distribution and abundance varied seasonally with the salinity regime and succession of species (Collins and Williams, 1981), but the groupings in table 17 are fairly representative.

Table 17. Mean abundance of each species in each group of sampling sites as a percentage of the abundance of the species in all groups of sites. Bold type identifies the abundant species associated with each group of sampling sites – April 1974. (adapted from Collins and Williams 1982).

Species	True estuarine	Estuarine and Marine	Euryhaline marine	Stenohaline marine
Eurytemura affinis	**98**	2	-	-
Gastrosaccus spinifer	**97**	2	-	1
Pleurobranchia pileus	**83**	14	3	1
Polychaete larvae	**56**	28	15	-
Pleurobranchia pileus juveniles	**69**	10	21	-
Cshistomysis spp. juveniles	**100**	-		-
Acartia bilfilosa	8	**91**	1	-
Schistomysis spititus	34	**47**	16	3
Mesopodopsis slabberi	11	**56**	27	6
Sagitta elegans juveniles	1	**47**	44	8
Evadne nordmanni	-	**100**	-	-
Centropages hamatus	-	25	**72**	3
Temora longicornis	-	8	**82**	10
Acartia clausi	-	2	**48**	50
Tomopteris helgolandica	-	11	**49**	40
Calanus halgolandicus	-	15	12	**73**
Paracalanus parvus	-	5	5	**90**
Psuedocalanus elongatus	2	24	19	**55**
Metridia lucens	-	-	-	**100**
Sagitta elegans	6	27	39	**28**
Meganyctiphanes norvegica	-	20	18	**62**
Nyctiphanes couchi	-	-	32	**68**
Centropages typicus	-	-	-	**100**
Podon intermedius	-	-	-	**100**

Like other estuaries in the British Isles, the permanent planktonic animals (holoplankton) in the Severn Estuary are mostly copepods. The temporary plankton (meroplankton) is represented by phyla such as decapods, molluscs, echinoderms, annelids and fish. Thus samples in the IMER study were dominated by calanoid copepods, but at certain times of the year mysids, especially *Schistomysis spiritus* constituted the major part of zooplankton biomass from the inner channel, reaching >80% of the total in summer.

There was a gradient of high biomass (measured as µg C) at the seaward end (Bristol Channel) to low values for the inner Estuary, which was more marked in the spring for the omnivores and summer for the carnivores. The peak for omnivores occurred in July, one month after the maximum standing crop of phytoplankton (measured as chlorophyll *a*). The peak for carnivore biomass occurred in August/September when numbers of the carnivorous chaetognatha (*Sagitta*) dominated in the plankton suggesting that for six months of the year, the structure of this planktonic ecosystem is probably regulated by this predator.

Macroalgae

In a study of the ecology of the upper Severn, Little *et al* (1985) noted that there were no comprehensive floral lists compiled at that time, a situation which does not appear to have been remedied since. However, there are a few reports describing algal species and their local distributions; notably Smith, 1978, Smith and Little, 1978:

Eleven intertidal sites between Kilve (west of Bridgwater Bay) and Sharpness (upper Estuary) on the SE (English) side of the Estuary were visited between 1975-1978 as part of a study on estuarine ecology (Smith, 1978). A striking feature of all the sites was that they lacked a subtidal zone of vegetation, the intertidal algal zone did not extend below the limit of Mean Low Water Springs (MLWS) therefore at low water there was a 'conspicuous bare zone above low water mark which is devoid of any algae'. Apart from this anomaly, the zonation of the intertidal algae follows the pattern generally found, with *Pelvetia canaliculata* and *Fucus spiralis* at the top of the shore, and *Fucus serratus* and *Ascophyllum nodosum* extending from mid- to lower-shore.

A distinct zonation of algae into two communities was apparent at most sites: in the region of Mean High Water Neaps (MHWN) to MHWS, the dominant species were the green algae *Enteromorpha intestinalis, E. prolifera, Blidingia minima* and *B. marginata* and the fucoids *F. vesiculosus* and *P. canaliculata*, although at any one site, all of these species were not necessarily present. Between MHWN and Mean Low Water Neaps (MLWN) the dominant species were *A. nodosum, F. vesiculosus* and *F. serratus*. *F. serratus* sometimes extended beyond MLWN although never below MLWS (Smith and Little, 1978). The authors suggest that the lack of macroalgae below MLWS is due to the effect of the high turbidity which reduces available photosynthetic light, coupled with the scouring effect of the silt which interferes with the settlement of algal spores.

The red algae were the most variable in their occurrence along the Estuary, whilst green and brown were more widely distributed. Ladye Point (between Weston Super-Mare and Avonmouth) was the upstream limit of *Callithamnion hookeri*, *Polysiphonia lanosa, P.denudata* and *Corallina officinalis*, whilst Aust is the upstream limit of *Ceramium rubram* and *Porphya umbilicalis*. Apart from the green algae, none of the other macroalgae was found upstream of Sharpness.

Generally, the reports suggest that the Severn as a whole supports an impoverished algal flora, especially with regard to the more delicate red algae. However, there may be isolated, locally important red algal communities such as the *Corallina* run-offs reported in Bridgwater Bay (Bamber and Irving, 1993). Bridgwater Bay is on the eastern shore of the Estuary near the mouth of the River Parrett. Hinkley Point Power Station intakes are at the western end of the 18 kilometre-square Stert and Berrow intertidal flats. *Corallina officinalis* is a small red calcareous alga and a common denizen of lower littoral rock pools around the coast of the UK. On the littoral limestone and mudstone rock platforms at Hinckley Point, erosion has created a topography of scarp cliffs facing the land with gentle slopes to the Estuary. The resulting gullies above the scarps retain water as the tide drops, either as pools or streams running off along the gullies and through breaches in the scarp. It is where the water runs through breaches and over the limestone slopes toward the Estuary, that water is present across the rock surface at all states of the tide, thus forming an environment analogous to that of a rock pool but of negligible depth. In these run-offs, dense mats of *C. officinalis* have developed. Twelve were identified along the shore from west of the power station to Benhole Point. At certain times of the year, the green alga *Ulva lactuca* grows within the *Corallina* and there is often *F. serratus* growing around the periphery of the mats. These *Corallina* run-offs are interesting in that *Corallina* is more often found in a constantly submerged environment, also that the mats support a diversity of animals representing the densest faunal community of the foreshores in the area (see section on macrofauna, below). In conservation terms, this is one of the most important habitats in the region.

Saltmarsh communities

A 1985 report (Little *et al.,*1985) reviews literature appertaining to Severn Estuary salt marsh areas and lists the extent, and species composition of the salt marsh. *Spartina*, which was introduced into the Estuary at Clevedon in 1913, was reported to have spread dramatically and covered 55% of the salt marsh area (Teverson, 1981: Mitchell *et al.,* 1981). The predominant species was *Spartina anglica* and its spread was reported to have caused an increase in establishing new marshes in the Estuary. A general historical background of the Severn salt mashes is referred to: Teverson (1981) and Moss (1907).

Generally, *Spartina* invades the low salt marsh zone, *S. anglica* is more tolerant to submersion than other *Spartina* species and is frequently found extending beyond the *Salicornia* zone. At some sites (e.g. Portbury Wharf) it dominates the whole marsh. The mature zones, higher up the marshes are more likely to be dominated by *Pucinella maritima* and *Festuca rubra* especially where the marsh is subject to grazing. Ungrazed marshes supported a more mixed community including *Limonium vulgare, Juncus gerardii* and *Aster tripolium.*

A comprehensive field survey using National Vegetation Classification (NVC) methods was carried out for CCW and EN (Dargie, 1998) which recorded and mapped the vegetation of saltmarsh habitats in the Severn Estuary.

The total area reported was 1521 ha, the majority of which, 1150 ha, was on the SE (English) side of the Estuary. Arlingham is the upper limit of Saltmarsh and it extends seawards as far as Catsford Common in Bridgwater Bay on the east side, and Lamby (S. Glamorgan) in Wales, extending up the Rhymney River. The report gives a detailed account of the distribution and associations of several nationally and locally scarce species, and although not useful to reiterate long lists of species here, a brief description of the whereabouts of locally and nationally scarce plants may assist in identifying vulnerable areas:

The bulbous foxtail (*Alopecurus bulbosus)* was recorded primarily in the upper Severn on the NW (Welsh) shores, and at various sites on the SE shores. It was often recorded in Bridgwater Bay and the River Parrett during July, but noted less frequently thereafter. *Althea officinalis*, or marsh mallow, was present in great numbers at Goldcliff Pill (SE of Newport), with one specimen only recorded in Bridgwater Bay. The slender hare's-ear *Bupleurum tenuissimum* was frequently found in Bridgwater Bay and the Rivers Parrett and Brue but was uncommon elsewhere. This species appeared to be declining, since a previous survey found it in most of the survey areas (Leach, 1994). *Hordeum marinum,* the sea barley was present in several saltmarsh areas, with a particularly dense population at Bridgwater Bay, and the meadow barley *Hordeum secalinum*, although deemed to be a locally rare species, had the most widespread distribution by far, leading the author to suggest that its status as rare be reconsidered.

Dittander *Lepidium latifolium* was recorded at a single location, the mouth of the Rhymney River, whereas it had previously been noted at three locations south of Gloucester (Rumsey, 1994). *Pucinnella rupestris*, the stiff saltmarsh grass was found primarily in the Welsh saltmarsh at the Wye and Lydney Sand, with the largest individuals and overall population north of W. Usk lighthouse. On the English side of the Estuary, *P. rupestris* was found only on the Bristol Avon, whereas, the sea clover, *Trifolium squamosum* was exclusive to the English side and recorded at Berkeley Pill, the Brue River, Sand Bay and near Clevedon.

Evidence was found of a substantial reduction in the area of *Spartina* in the outer Estuary since the late 1970's, together with an increase in the centre and upper Estuary. Results suggest that estuarine vegetation is currently involved in a transgression up the Severn Vale in response to rising sea level.

Zostera

Zostera spp. or eelgrass is reported to occur in only one area of the Severn Estuary, between Magor and Caldicott on the NE (Welsh) side, upstream of Newport and close to the second Severn crossing. There is a paucity of information as to the extent, density and associated fauna of the *Zostera* bed and most of the available information is contained in a series of short reports produced for CCW 1993 - 1995 to monitor the effect of the second Severn crossing (EAN 1993; SGS Environment, 1995a, b). Two *Zostera* species were definitely identified, *Z. noltii* and *Z. marina*. *Z. angustifolia* was also thought of be present but difficulties in identification (differing relative lengths of stigmas and styles - only possible at certain times of the year) prevented positive recording of this species. Over the monitoring period (1992-5), a decrease in the density of the eelgrass was noted and also a reduction in the size of the *Zostera* bed. Changes in the relative abundance of each species were apparent; density of the lower shore species, *Z. marina* declined, whilst after an initial decline, density of *Z. noltii*, an upper shore species, stabilised within the eelgrass bed (SGS Environment, 1995b).

The reduction in area of the *Zostera* bed was attributed to deposition of sediment due to the construction of the second Severn crossing, and had been predicted in modelling exercises (EAN, 1991). Invertebrate populations associated with the *Zostera* may also have been affected by extreme episodes of erosion and deposition of sediment, although the natural levels of these events were said to have been exceeded in only one transect of the *Zostera* bed and anticipated to have long-term effects on invertebrate density or biomass (SGS Environment, 1995b). However, there are no available records of the fauna associated with the eelgrass bed in the Estuary, therefore faunal changes cannot be demonstrated.

Sediment accretion is only one of a number of factors which may contribute to a decline in *Zostera* beds, although in the Severn Estuary, it is probably the single most important. For a full account of other possible factors see Davison and Hughes (1998).

It was recommended that the work on *Zostera* in the Estuary be continued (SGS Environment, 1995b) but no further information is available as to the current status of the *Zostera* or the diversity of associated fauna.

Macrofauna

There are several early reports published in Proceedings of the Bristol Naturalists Society relating distributions, and providing lists of macrofaunal organisms in the Estuary (see above). A study by Boyden (1977) which documents the status and distribution of almost 600 species, and summarises older records dating back to pre-1950s was used to review faunal changes in the Severn Estuary over several decades (Mettam, 1979) and provides a good starting point for a general description.

Apparent losses of species pre-1979: Mettam (1979) found the Crustacea to show the greatest losses as a proportion of the total, this included 15 copepod species (including meiofaunal elements) and of the larger Malacostraca, Bassindale (1942) had recorded 15 amphipods which were not listed in the 1977 studies. *Anthura gracilis, Aplheus macrochelis, Haemoniscus balani* and *Macropipus holsatus* were also absent from the later species lists. Species of polychaetes were reduced by nine, and gastropod molluscs by seven (five of these, opistobranchs). Bryozoan species were down by 11 although the author acknowledged that as nine were previously recorded at the seaward edge of the Estuary, some erratic fluctuation in distribution might be expected to occur. Other losses included less well-represented animal groups such as

Chondrophora and Siphonophora previously recorded as rare strandings of pelagic drifters *Velella* and *Physalia.*

Apparent gains of species: 'New' records included a substantial number in almost all groups which may have been due to increased and more widespread searching intensity. It was also postulated that environmental change described by Southward *et al.,*(1975) which had widespread effects on the biology of the English Channel had affected the biota of the Severn Estuary. Notable new species included *Crepidula fornicata*, the non-native slipper limpet (although distribution and numbers of this invasive species are not reported) bivalve mollusc *Lasaea rubra* and ascidian *Dendrodoa grossularia*. The native oyster *Ostrea edulis*, absent for more than 30 years was also noted although there were no further details (sites, size, abundance). There are no other references to this species in the available literature.

Other new species to the Estuary included *Cliona celata, Cordylophora caspia. Laomedea flexuosa, Emplectonema neesi, Eunereis longissima, Nereis irrorata, Cirriformia tenteculata, Idotea pelagica, Dymamene bidentata, Athanus nitescens, Hippolyte varians, Anapagurus hyndmanni, Portumnus latipes, Pinnotheres pisum* and *Macrpipus puber,* the latter species had not been recorded since the 1920's.

Mettam (1979) also reported briefly on changes in distribution and abundance which had occurred in the Estuary: A particularly cold winter, 1962/3 had caused habitat destruction and severely reduced numbers of many species including the honeycomb worm *Sabellaria alveolata,* gastropod mollusc *Monodonta lineata* and deposit-feeding bivalve *Scrobicularia plana.* Many *Sabellaria* reefs were eroded and as recruitment to colonies was poor in most years, there was a question mark over the status of the reefs. However, where the reefs were damaged, thick crusts of *Pomatoceros triquiter* were evident; this species had not been reported in earlier studies. *M. lineata* was absent from the western shores of the Estuary, and populations of *S. plana* were recovering patchily since their decimation.

The low-shore crab *Pirimela denticulata* and gastropod mollusc *Nassarius reticulatus* were lost from the Estuary during the 1962/3 winter and had not recolonised their former habitats. Numbers of polychaete *Platynereis dumerili* and Risso's crab *Xantho pilipes* were significantly reduced.

No other significant changes had occurred between the 1940's and late 1970's but comparison with earlier records (early 1900's) showed that some common shore animals (sea anemone *Actina equina* and gastropod mollusc *Littorina littorea*) had penetrated further up into the Estuary, whilst others (including gastropod mollusc *Rissoa parva,* cowrie shell *Trivia monacha, oyster drill Ocenebra erinacea*) had retreated seaward.

Other notable changes included increases in numbers of the hairy crab *Pilumnus hirtellus* and two bryozoans. Numbers of the non-native barnacle *Elminius modestus* had also increased. *E. modestus,* first reported at Barry Island in 1947 (Purchon, 1948), later became the dominant barnacle species in the Estuary (Mettam, 1994).

Two *Gibbula* species had increased their range toward the inner Estuary as had barnacle *Balanus perforatus* and limpet *Patella vulgata,* the latter two in response to two successive years of low rainfall (1975, 1976). Boyden and Little (1973) had reported a paucity of macrofauna around industrialised Avonmouth, and found little seasonal change (Little and Boyden (1974). Mettam (1979) concluded that changes in faunal distribution over the decades may have been due to a number of factors and could not confidently be attributed to long-term environmental change.

For a more recent account of the fauna, Little *et al* (1985) reported on the ecology of the Severn Estuary and included lists of species present and their distribution, and the biological Journal of the Linnean Society published proceedings from a workshop held in 1989 to discuss current understanding of the Severn Estuary ecosystem, again, in relation to a proposed tidal power scheme (Vol 51[1-2]). The latter publication includes several reports relating to macrofauna (benthic and intertidal) that can be referred to for in-depth detail, and we briefly discuss the more relevant ones below:

The vertical distribution of common species of macroalgae and fauna on rocky shores of the Estuary is described by Mettam (1994). Key findings include the relative abundance of *Sabellaria alveolata*, which featured at several sites, principally in the outer Estuary around the mean low water mark and subtidally. *Sabellaria* species are adapted to survive under the hostile conditions of the area; *S. alveolata* is especially notable as it is only found in sublittoral habitats in the Severn Estuary. However parts of the *Sabellaria* reef in Limpert Bay, between Llantwit Major and Barry Island, were uninhabited and a large expanse of completely uninhabited *Sabellaria* reef, the eastward extent of *Sabellaria* species (living or dead reefs) was recorded in Sully Bay East of Barry Island. Frost has been suggested as the factor limiting the upstream distribution of reefs in the Severn Estuary (Bamber and Irving, 1997), and probably explains the reduction in abundance of the species during 1962/3 (see above). The shore and seabed, especially east of Porlock and Swansea Bays, is naturally impoverished because of the high turbidity, variable or low salinity conditions and strong tidal currents. Crusts and sometimes extensive reefs on the seabed formed by the tube worms *S. spinulosa* and *S. alveolata* can provide a solid substratum for a variety of epifauna species. Isolated reefs of agglomerated *Sabellaria* worm tubes are also found on sublittoral substrate of Bridgwater Bay, which is highly mobile, nearly liquid mud with some areas of sand waves and an intertidal area of firmer sandy mud.

The most recent account of the *Sabellaria* reefs in the Severn Estuary (Warwick *et al.,* 2001) indicates that the eastward extent of the reefs has not changed significantly, but that compared with *Sabellaria* reefs situated further to seaward in the Bristol Channel and in other parts of the British Isles, the fauna of these reefs is extremely impoverished. For example, littoral reef at Duckpool, North Cornwall, was found to hold 45 species in comparison with a maximum of 16 species supported on Severn reefs.

This finding was part of a survey of the benthic fauna of the deep-water channel of the Severn Estuary between Flatholm Island and King Road (Warwick *et al.,* 2001). Other principle findings were that the deep channel of the Severn Estuary has an extraordinarily impoverished fauna, 22 of the 62 grab samples taken held no animals. The sandy substrata were particularly impoverished. The total number of infaunal species at 13 stations with a sand substrate was only 5, which between them only comprised 18 individuals. At any one station with a sand substrate, the maximum number of infaunal species (2) was represented by 4 individuals. *S. alveolata* was the most abundant benthic animal in the survey area, found in 17 grab samples taken from hard bottom areas in and immediately adjacent to the deep channel between the Bristol Deep and the Holm Islands.

Mettam (1994) also recorded the distribution of dog-whelk *Nucella lapillus*. This species was only present in the outer Estuary/inner Bristol Channel as far up the Estuary as Porthkerry (west of Barry Island) on the Welsh shore. There are also records of a population at Porlock Weir (English shores), where *N. lapillus* are reported to have distinct differences in size, shell shape and growth patterns to populations from further west, although no explanation (environmental or otherwise) are offered for these differences (Crothers, 1980).

The presence of the non-native species, the slipper limpet *Crepidula fornicata,* was noted by Mettam (1979), It has subsequently become common as far east as Sully Bay (Mettam, 1994). *C. fornicata* is known to have been introduced to Essex between 1887 and 1890 from North America, in association with the American oyster *Crassostrea gigas* (Crouch 1894, 1895; Orton 1912; Fretter and Graham 1981) and spread fairly rapidly (Franklin and Pickett 1974). Its success in this country is probably due to a lack of predators and the unusual method of reproduction (which relies upon individuals settling upon each other to form breeding 'stacks' as they develop from males to females). A pelagic larval stage aids the spread of the species, once introduced.

High densities of *C. fornicata* may modify the nature and texture of sediments in some bays (Ehrhold *et al.,* 1998) and where *Crepidula* stacks are abundant, few other bivalves or other filter-feeding invertebrates can live amongst them. This is due to spatial competition, trophic competition and alteration of the substratum (the pseudofaeces of *C. fornicata* smother other bivalves and render the substratum unsuitable for larval settlement) (Fretter & Graham, 1981; Blanchard, 1997). In this way, *C. fornicata* has become a serious pest on oyster beds and has caused many traditional oyster fisheries to be abandoned (e.g. in the Norman Gulf, France) (Blanchard, 1997). However, De Montaudouin *et al*. (1999) showed that *Crepidula* had no major influence on the local density or diversity of smaller coexisting macroinvertebrates and did not affect the growth of 18 month old oysters. At present, there are no indications that the slipper limpet is a problem in the pSAC, but with the trend of rising temperatures it could become more widespread; Minchin *et al.,* (1995) indicate that temperatures may be an important limiting factor in its ability to develop extensive populations.

Locally important for macrofauna are parts of the shore in Bridgwater Bay, in the locality of Hinckley Point. This area normally supports a low density and diversity of plant and animal species, as is characteristic for the Severn Estuary littoral zone where extreme conditions (sediment instability, turbidity and scouring) limit the range of 'usual' estuarine organisms. However, the *Corallina* run-offs of Bridgwater Bay (see section on macroalgae, above) support a total of 38 species including several which are not recorded elsewhere in the region including the isopod *Jaera praehirsuta* and the pycnogonid *Anoplodactylus pygmaeus.* A full account of the associated fauna can be found in Bamber and Irving (1993).

Warwick *et al* (1989) conducted a comparative study of the intertidal benthic invertebrate communities at 40 estuaries (including the Severn) in SW England. Multivariate analysis of species abundance and biomass data showed that the fauna of the Severn Estuary sites was significantly distinct from all other estuaries and had a number of features which set it apart from other more ‘normal’ estuaries. The differences were thought to result from features of the large hypertidal Estuary: high turbidity and sediment instability. There were indications that the majority of sites studied in the Severn were unaffected by pollution or other anthropogenic disturbances.

In comparison with other estuaries, populations of several invertebrate species, notably *Nephtys, Macoma* and *Hydrobia*, and to a lesser extent, *Nereis*, were dominated by small individuals suggesting a shorter lifespan. Apart from this observation, the Severn fauna did not appear to be significantly different to other estuaries in terms of condition factors.

The virtual absence of suspension-feeding bivalves (and all suspension-feeding invertebrates), including *Cerastoderma edule* and *Mya arenaria*, using conventional benthic sampling, was attributed to the very high levels of turbidity. The deposit-feeding bivalve *Scrobicularia plana* is also uncommon (probably because of sediment instability). This is a relatively robust species which is used as an indicator organism for the bioavailability and impact of metals in estuaries (see section 5.1.2).

It is also worth noting key points regarding crustacea from recent reports arising from monitoring of fish and crustacea entrained on the intake screens of Hinkley Point ‘B’ Power Station (see above):

- During 2000/2002 the normal patterns of seasonal abundance were observed for all species
- There was one new crustacean record in 2001, a single specimen of *Axius stirhynchus*.
- There are clear indications that numbers of crabs are increasing. This is particularly apparent in the time series records of the edible crab, *Cancer pagurus*.
- *Crangon crangon* has remained the most abundant species (macro-crustacean or fish) caught at Hinkley Point and the population remains remarkably stable, showing no clear trend in abundance over a 21-year period.
- The second most abundant macro-crustacean, the pelagic prawn *Pasiphaea sivado,* for which 1999 was exceptional in terms of numbers caught, has been less abundant in 2001
- The pink shrimp, *Pandalus montagui*, has maintained an almost constant annual mean abundance since 1981.
- There is a clear trend of increasing abundance of the large edible prawn *Palaemon serratus* within the Estuary. In the 1998 report it was noted that the highest number of *P. serratus* in a single sample was 403 in October 1997, in September 1998 the highest number was 700, rising to 1195 in June 2000. In April 2002 some berried females were observed suggesting that the mild conditions are allowing an extended breeding season and possibly enhanced recruitment.
- The sardine crab, *Polybius henslowi* is only rarely recorded, however in 2001, 7 individuals were captured.

- Other swimming crabs were also relatively abundant, particularly *Liocarcinus holsatus*. The velvet swimming crab, *Necora puber*, has also been captured more frequently

Examining the time series, the authors found that since 1997 the seasonal dynamics for species abundance had become remarkably cyclic and the pattern is markedly different from earlier years. Two possibilities were under investigation. First, the abundance of predators such as the whiting is also showing enhanced seasonality – perhaps there are intensifying predator-prey interactions as the abundance of fish and crustaceans increases. Second it may be linked to intra and inter-specific competition, there is evidence of strong seasonality in prawns such as *Palaemon serratus* which have increased greatly in abundance in recent years.

The fauna of the principal rivers in the Estuary (Avon, Usk, Wye) is reported to be similar to that for the soft sediments of the Severn (Morrisey *et al*., 1994). Communities in these river estuaries were dominated by *Nereis diversicolor, Corophium volutator, Macoma balthica* and oligochaetes. Densities of *Nereis* were similar to the main Estuary, while that of *Corophium and Macoma* were higher than in the main Estuary, indicating that these estuaries constitute an important reserve of estuarine invertebrates.

Fish

Monitoring of fish and crustacea captured on the intake screens of Hinkley Point 'B' Power Station has provided (possibly) one of the best accounts of fish populations in the Severn in terms of numbers and species. Sampling gave estimated total annual captures ranging from 6.0- to 17 x 10^5 fish (Henderson)[1].

The 2001 annual report of Hinckley fish captures (Henderson and Seaby, 2001) noted two new species recorded in 2000/2001: Firstly, Raitt's sandeel *Ammodytes marinus*, an abundant offshore species that is known to occasionally come into shallow water and even enter the mouths of estuaries. It has been recorded from most marine power stations where long-term studies have been undertaken. Secondly, the rock goby *Gobius paganellus* a common species of rocky shores, (the habitat in Bridgwater Bay is probably too muddy to allow the establishment of a resident population). Three infrequent visitors captured were the three bearded rockling, *Gaidropsaurus vulgaris*; greater sand eel, *Hyperoplus lanceolatus* and blue whiting, *Micromesistius poutassou*. The three bearded rockling was recorded in March 2000 and April 2000, suggesting that a small migratory influx of individuals probably occurred during spring 2000. The greater sand eel may be increasing in numbers in some regions near to Bridgwater Bay, as this was the second summer in succession that the species was recorded. All of these are fish that can be commonly found in other habitats and should be viewed as occasional migrants. It is however, notable that these should be recorded in a single year and is a reflection of the increasing diversity of species in the Estuary.

[1] http://www.irchouse.demon.co.uk/indexlatestreports.html

Another particular feature of note in the 2001 report (Henderson and Seaby, 2001) was the relatively high abundance of cod *Gadus morhua*, herring *Clupea harengus*, sole *Solea solea* and prawn, *Palaemon serratus*, within Bridgwater Bay. Fish abundance in the Estuary was an estimated 3 times higher than that recorded in the early 1980s and there was also a clear trend for increased species richness. The large increases in abundance within the system as a whole offered hope that the populations of these important commercial species were improving in this region (the Severn Estuary also supports a prawn and shrimp fishery).

The 2001 report indicated that recent closures of direct-cooled power stations in the region are coincident with the increased abundance of common fish and crustaceans at Hinkley Point. Although a number of climatic and anthropogenic changes may be contributing to the observed increase in species richness and abundance, the recent closure of a number of direct-cooled power stations since sampling commenced in October 1980 is considered important (Berkeley closed in 1989, Uskmouth in 1995, Pembroke in mid-1997 and Hinkley 'A' in May 2000). All of these stations would have been killing fish and crustaceans and reducing the population subject to capture at Hinkley 'B'. It was considered unlikely that entrainment and impingement in power station cooling water systems would have changed species richness in the region because the Estuary presents an open system that would receive a flow of recruits from other waters. Nevertheless, if mortality rates were sufficiently high it is possible that direct cooled power stations could reduce abundance by a detectable amount.

However, the authors point out that these observations do not prove that power stations have, in the past, reduced animal abundance. Future observations may further clarify the impact of direct-cooled power stations.

Key observations from the most recent report (Henderson *et al.,* 2002) are as follows:

- Overall fish abundance in 2001/02 was the lowest since 1995. This was related to the clear reduction in over-winter adult sprat.
- There were no 'new or unusual' observations during 2001/2002
- There is a general trend of increasing water temperature observed within the Bristol Channel which may be related to large-scale climatic trends
- This is producing a gradual change in fish (and crustacean) abundance in Bridgwater Bay.
- There is clear long-term trend of increasing species richness and more frequent capture of warmer water species (bass *Dicentrarchus labrax*, mullet *Liza aurata, L. ramada* and *Crenimugil labrosus*, gurnard *Eutrigla gurnardus*, trigger fish *Balistes capriscus* etc).
- Species that are close to the southern limit of their range in the Bristol Channel such as dab *Limanda limanda*, northern rockling *Ciliata septentrionalis* and sea snail *Liparis liparis* have declined in abundance though they are still common.
- There has been a gradual increase in cod (*Gadus morhua*) abundance, surprising as cod is a colder water species.
- Conditions within the Estuary may be becoming more favourable for fish and crustaceans.
- The abundance of some species, most notably the common eel (*Anguilla anguilla*), are declining and may possibly decline further without vigorous conservation effort. At present this species is still subject to a fishery.

Table 18. Fish species recorded at Hinckley Point together with brief observations on abundance (adapted from Henderson *et al* (2002).

Species	Notes
Cod *Gadus morhua*	More abundant within Bridgwater Bay since 1986; generally increased in abundance. Captures for the year 2001-2002 were lower than last year but still higher than during the 1980s
Whiting *Merlangius merlangus*	Less abundant than in recent years, possibly linked to the reduced abundance of sprat during the winter
Poor cod *Trisopterus minutus*	Abundance has remained far below levels recorded in 1986
Pout *Trisopterus luscus*	Abundance was low in 2001 - presence is positively correlated with abundance of poor cod.
Hake *Merluccius merluccus*	No hake have been caught over the last year although once quite common can now be considered rare in Estuary
Pollock *Pollachius pollachius*	This species is never abundant but continues to be caught in low numbers.
Norway pout *Trisopterus esmarkii*	This species has continued to be caught in low numbers.
Five-bearded rockling *Ciliata mustela*	Abundance has increased since 1997 and appreciably so since 1980s.
Northern rockling *Ciliata septentrionalis*	Once considered rare in Southern British waters, uncommon at Hinkley Point
Twaite shad *Alosa fallax*	Abundance declined since 1988 – 1991. A number of O-group fish were caught in 2001/2002 suggesting improved recruitment.
Bass *Dicentrarchus labrax*	Strong recruitment in autumn 1997 – numbers remain higher than those observed in the 1980s
Common eel *Anguilla anguilla*	Long-term decline continues – conservation measures may be needed.
Conger *Conger conger*	Low numbers - some quite large animals (> 1 m long) caught 2001/2
Lumpsucker *Cyclopterus lumpus*	Only captured infrequently when large numbers of juveniles enter the region during the winter. One individual was captured in both 2000 and2001. Increasing water temperatures may be having an adverse effect on this species,
Sea snail *Liparis liparis*	Exceptionally low abundance in 2001 – possibly due to water temperature increase and the loss of shallow habitat where favoured food, shrimp are found
Thornback ray *Raja clavata*	Last caught in April 2001. Reduction in abundance possibly due to increased water temperatures
Dab *Limanda limanda*	Dab were not abundant in the Estuary in 2001/2002; this probably reflects high water temperatures - one of the most abundant flatfish within Bridgwater Bay
Flounder *Platichthys flesus*	The total catch in 2001 was one of the highest recorded - more abundant since 1986
Dover sole *Solea solea*	Abundant within the Estuary -most of the sole captured were ‘O’ group juvenile. Recruitment has been above the long-term average for 9th successive year – possibly due to increase in water temperature
Plaice *Pleuronectes platessa*	Least abundant of the common British flatfish within Bridgwater Bay. Numbers reduced since 1997
Dragonet *Callionymus lyra*	Occasional visitor to Estuary.
Grey gurnard *Eutrigla gurnardus*	Few caught each year
Hooknose *Agonus cataphractus*	Low abundance during 2001/2002 - occasional visitor in Estuary
Nilsson's pipefish *Sygnathus rostellatus*	Occasional visitor - more abundant during 2001/2002.
Sand goby *Pomatoschistus minutus*	Gradually increasing in abundance within Bridgwater Bay
Transparent goby *Aphia minuta*	Only two caught during 2001/2002 – generally more southern species
Herring *Clupea harengus*	Dramatically increased in abundance during 2000/2001 -considerable improvement in recruitment
Sprat *Sprattus sprattus*	Not abundant in Bridgwater Bay during the 2001/2002 winter. normally most common pelagic fish at Hinkley Point - captures in December 1988 were the highest since recording began in 1980.
Grey mullets *Liza aurata, Liza ramada and Crenimugil labrosus*	Both thin-lipped and golden-grey mullet have continued to be caught in small numbers

A similar long term, (but not continuous) study of fish collected from the intake screens at Oldbury-on-Severn nuclear power station (inner Severn Estuary) also found an increase in abundance of many species (Potter *et al.,* 2001). This study examined fish caught between 1972-7 and 1996-9 and report that annual catches demonstrate that the abundance of fish at Oldbury was far greater in the 1990s than 1970s, mainly due to marked increases in the numbers of certain marine species, such as sand goby, whiting, bass, thin-lipped grey mullet, herring, sprat and Norway pout. The authors suggest that observed increases may reflect changes in levels of pollutants in the Severn Estuary; the increased regulation on the discharge of heavy metals and organic waste is said to have resulted in a significant increase in estuarine quality over recent decades (Little and Smith, 1994; Martin *et al.*1997; Vale and Harrison, 1994) – see sections 5 and 7 for further information on temporal trends in contaminants.

Potter *et al* (2001) also report that the only species to have declined markedly in abundance was poor cod. Modest declines in flounder and River lamprey *Lampetra fluvialatilis* paralleled those occurring elsewhere in the UK. The species composition in the two decades also differed, reflecting changes not only in the relative abundances of the various marine estuarine-opportunistic species, which dominated the ichthyofauna, but also in those of the suite of less abundant species in the Estuary (Norway pout, Dover Sole, Bib *Trisopterus luscus*).

The most abundant of thirty eight post-larval fish species recorded during the IMER studies in the Estuary (described in the section on zooplankton, above) was the sprat *Sprattus sprattus* which spawns in spring (Russell, 1980).

A recent study described the biology and status of the four migratory fish included in the list of interest features for which the Severn Estuary has been recommended as a pSAC: the sea and river lampreys (*Petromyzon marinus and Lampetra fluviatilis*)[1] and two species of shad, the Twaite (*Alosa fallax*) and the Allis shad (*Alosa alosa*) (Bird, 2002). This report contains accounts of the life cycle and ecology of the species, in addition to sources, effects and implications of contamination for the fish in the Severn Estuary. Information on the seasonal numbers of migratory fish is based largely on data obtained from power station water-intake screens for fish entering the Severn Estuary and from a hydroacoustic fish counter for fish entering the Wye.

The Severn is one of few estuaries in Europe where juvenile Twaite shad have been recorded in any numbers (Potter *et al*. 1986; Potts and Swaby 1993) and they have remained about the 10th most abundant species at Oldbury over the last 30 years (Potter *et al.* 2001).

Apart from the rivers Usk and Wye, the Allis shad is rare in British waters and there are no confirmed spawning sites in the UK (Maitland and Lyle 1990; Maitland and Lyle 1991). In most respects, the life cycle of the Allis shad is very similar to that of its more common relative, except that adult Allis shad tend to be larger and migrate further upstream during their spawning migration. Between July 1974 and April 1977, only two adult individuals of this species were recorded from samples collected at Oldbury Power Station and no juvenile specimens were observed over the same period (Claridge and Gardner 1978).

[1] The term 'fish' is used loosely here since it is incorrect to describe lampreys as fish. They are quite distinct from the Gnathostoma or 'jawed fishes' that contains the cartilaginous (Chondrichthyes) and bony (Osteichthyes) fishes (Bird, 2002).

Lampreys, together with the marine hagfishes, are the sole extant representatives of the most primitive vertebrates, the Agnatha or 'jawless fishes'. Adult anadromous parasitic lampreys feed on the blood and tissues of teleost fish and marine mammals using a suctorial disc to attach to their prey and a tooth-bearing piston-like tongue to rasp away host tissue. Both species enter rivers in south-west UK and are stimulated to begin their spawning migration by the increase in freshwater discharge from rivers which coincides with high tides in the spring and autumn. Fish samples collected from the water-intake screens at Oldbury-upon-Severn Power Station in the inner Severn Estuary contain adult sea lamprey in March and river lamprey April - July (Abou-Seedo and Potter 1979), although numbers of river lamprey are recently reported to be declining (Potter *et al.,* 2001). Principal threats to Lamprey and Shad as perceived by Bird (2002) are listed in table 19.

Table 19. Summary of perceived threats to Lamprey and Shad species in the Severn Estuary (from Bird, 2002)

Stage in life-cycle (LAMPREY)	Environment	Risk from metals	Risk from organics	Comments
Larvae	Freshwater	+++	+++	Contact with contaminated sediments Filter feeding & diet important? Sensitive to oxygen depletion Endocrine disruption?
Young adults & downstream migrants	Freshwater Estuarine Marine	+ + –	++ + –	Lipid mobilisation may release high concentrations of lipophilic toxicants? Olfactory disturbance? Endocrine disruption?
Feeding adult	Estuarine Marine	+ –	+ –	Exposure via gills & skin possible? Dietary exposure unlikely?
Adult upstream migrants	Estuarine Freshwater	++ ++	+++ ++	Lipid mobilisation may release high concentrations of lipophilic toxicants? Degenerative changes may limit capacity for xenobiotic metabolism? Olfactory disturbance? Endocrine disruption?

Stage in life-cycle (SHAD)	Environment	Risk from metals	Risk from organics	Comments
Eggs, embryos & larvae	Freshwater	+++	+++	Sensitive stages in life cycle Endocrine disruption? Sensitive to oxygen depletion
Young 0+ & 1+ fish	Freshwater Estuarine	+ +	++ +	Capacity for xenobiotic metabolism not fully developed? Endocrine disruption?
Adults	Marine	+ –	+ –	Exposure via gills & skin? Dietary exposure possible but unlikely?
Adult upstream migrants	Estuarine Freshwater	++ ++	+++ ++	Lipid mobilisation may release high concentrations of lipophilic toxicants? Olfactory disturbance? Endocrine disruption?

Overall, the general trend of increasing water temperature observed within the Severn Estuary and Bristol Channel, which may be related to large-scale climatic trends, appears to be producing a gradual change in fish and crustacean abundance and species richness. It may also be possible that the environment of the Estuary is improving for fish and crustaceans, which is the general conclusion from monitoring at Hinckley Point and Oldbury. Given the large amount of estuarine habitat available within the Bristol Channel and Severn Estuary there can be little doubt that this region is an exceptionally important nursery area for juvenile fish and that it is increasing in importance. There are, however, several species for which abundance is reportedly declining, such as the Twaite Shad, eel, pout, hake, poor cod, thornback ray, plaice and sea snail.

Of these, the decline in numbers of the Twaite shad is notable. There is presently no information on effects and levels of contamination in young shad in the Estuary (Bird (2002). The decline of the common eel is also a cause for concern. It is possible that the decline may continue without vigorous conservation effort. White and Knights (1994) also note the reduced abundance of juvenile eels and elver in the Estuary and comment on possible causes (poor water quality due to sewage effluent, turbidity, increased siltation and oceanic events), however there is no firm evidence that these factors can explain reduced catches and recruitment into freshwaters.

Birds

Recently the British Trust for Ornithology (BTO) has carried out a review of species trends in SPAs over the last 5, 10 and 25 year time periods (upto 2000) using data collected as part of the Wetland Bird Survey (WeBS). SPAs where species are declining at a rate of greater than 25% over a specified time period when the larger-scale regional or national trends indicate stable or increasing population sizes are targeted as being of concern. Population declines of between 25% and 50% are flagged as 'Medium Alerts' and declines of greater than 50% as 'High Alerts'. Alerts are intended as advisory measures triggering further investigation. The report, produced for the Environment Agency, English Nature and the Countryside Council for Wales (Armitage *et al*., 2002) summarises statistics for 16 Evaluated Species in the Severn Estuary SPA: Bewick's Swan, European White-fronted Goose, Shelduck, Wigeon, Gadwall, Teal, Mallard, Pintail, Shoveler, Pochard, Tufted Duck, Ringed Plover, Grey Plover, Dunlin, Curlew and Redshank.

Alerts were triggered for all five species for which the Severn Estuary SPA is internationally important – Shelduck, Pintail, Dunlin, Curlew and Redshank. For Shelduck and Pintail a medium alert for the 25 year period applies because numbers remain below peak values in the mid-1970s. Dunlin numbers have also fallen over the last 25 years despite a brief recovery in the late 1980s (high alert for 25 and 10 year periods, medium alert for 5 year period). Curlew numbers rose through the 1970s and 1980s and peaked in the mid 1990s but since then have declined sharply. Redshank number in the Severn peaked in 1987/88 but have declined subsequently (giving rise to medium alerts for both the 10 and 25-year periods).

Alerts were also triggered for seven further species for which the site is important. Notably, Ringed Plover and Mallard numbers have declined steadily since the early 1970s. The nationally important population of Bewick Swans has also shown a general downward trend over the last two decades, despite a peak in numbers in the mid 1990s.

The fact that populations of 12 of the 16 species for which the SPA is important (and for which data were evaluated) have shown declines is considered cause for concern at the site. Detailed investigation into causes of these declines, particularly for the five species of international importance, has been recommended by BTO. Factors such as habitat loss due to vegetation succession, dredging and erosion, industrial pollution, changes in water quality resultant from improvements to waste water discharges and changes in recreational disturbance are suggested as priorities in any future ‘Level 2’ assessment (Armitage *et al*., 2002).

7. WATER STATUS AND QUALITY STANDARDS

In this section we examine unpublished data, mainly supplied by EA, of key determinands which may influence the Severn Estuary pSAC. Summary statistics have been drawn up by the EA (mainly based on monitoring since 1990), and the raw data analysed in an attempt to establish further evidence as to whether or not existing water (or sediment) quality is causing impact. Where relevant, temporal trends are discussed - otherwise only the most recent data are shown.

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

1) Discharges – to gauge the importance of specific industrial and trade effluent point sources. Here another major caveat has to be introduced, since information for a number of discharges has not been available. Furthermore, calculation of reliable loadings for most determinands is beyond the scope of the current project; only available concentration data are discussed (with regard to relative potential threat to the SAC).
2) Harmonised monitoring points (HMP); freshwater site immediately above the tidal limit (to characterise riverine input). Equivalent sites have been selected for other catchments where there is no designated HMP. Figure 11 shows the location of these sites. Again data are not always available at each site.
3) Tidal waters – a review of data within saline waters of the pSAC itself. To help describe the data the Estuary have been divided regionally according to the scheme shown in figure 12. These regions include, Severn East, Severn West, Usk, Cardiff and Parrett; most samples in these regions are close to the shoreline. 'Mid Severn' refers to an axial profile of offshore (mainly mid-channel) samples along the whole length of the Estuary and includes some of the sites of the helicopter surveys used by WRc to produce the recent synopsis of water quality trends (late 1970s-1997) for the Severn Estuary Monitoring Group (Ellis, 2002). Because the EA data set do not contain widespread information on contemporary values, entries recorded over the last ten years have been summarised to provide a more integrated picture of water quality issues, and to make comparisons with Environmental Quality Standards. Where high values are indicated in the regional summaries, data for individual sites have been analysed further to establish their significance and temporal variation.

   Where possible we have attempted to draw comparisons between our own examination of the data set and the synthesis provided by WRc (Ellis, 2002). One distinction is that the WRC summarises data over longer timescales and highlights temporal changes since 1980, whereas our treatment of data only examines evidence of trends since 1990. Both studies highlight the problem of apparent outliers, which though often present in older data cannot be excluded.

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

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

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

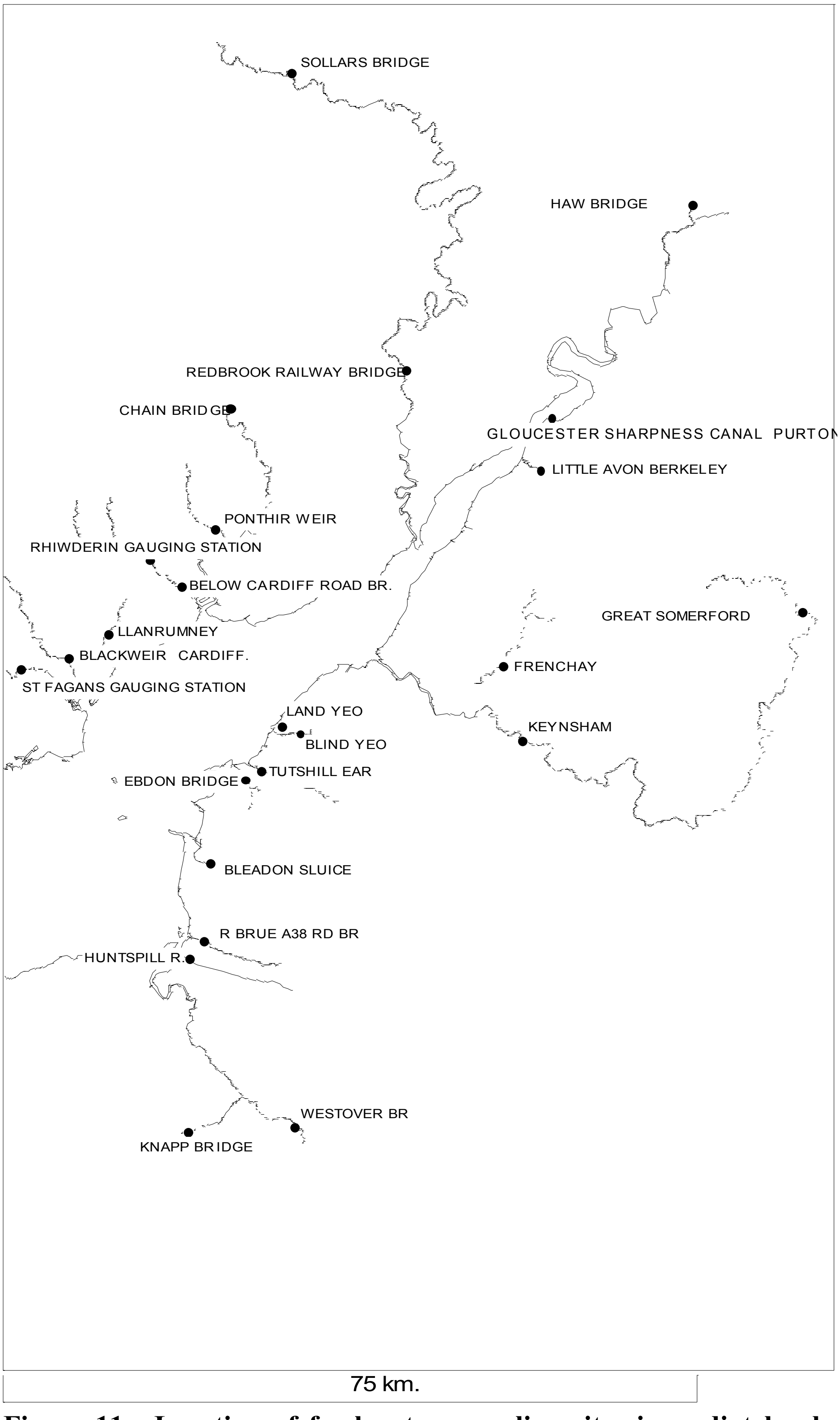


Figure 11. Location of freshwater sampling sites immediately above the tidal limit to characterise riverine input (harmonised monitoring points - HMP - or equivalent).

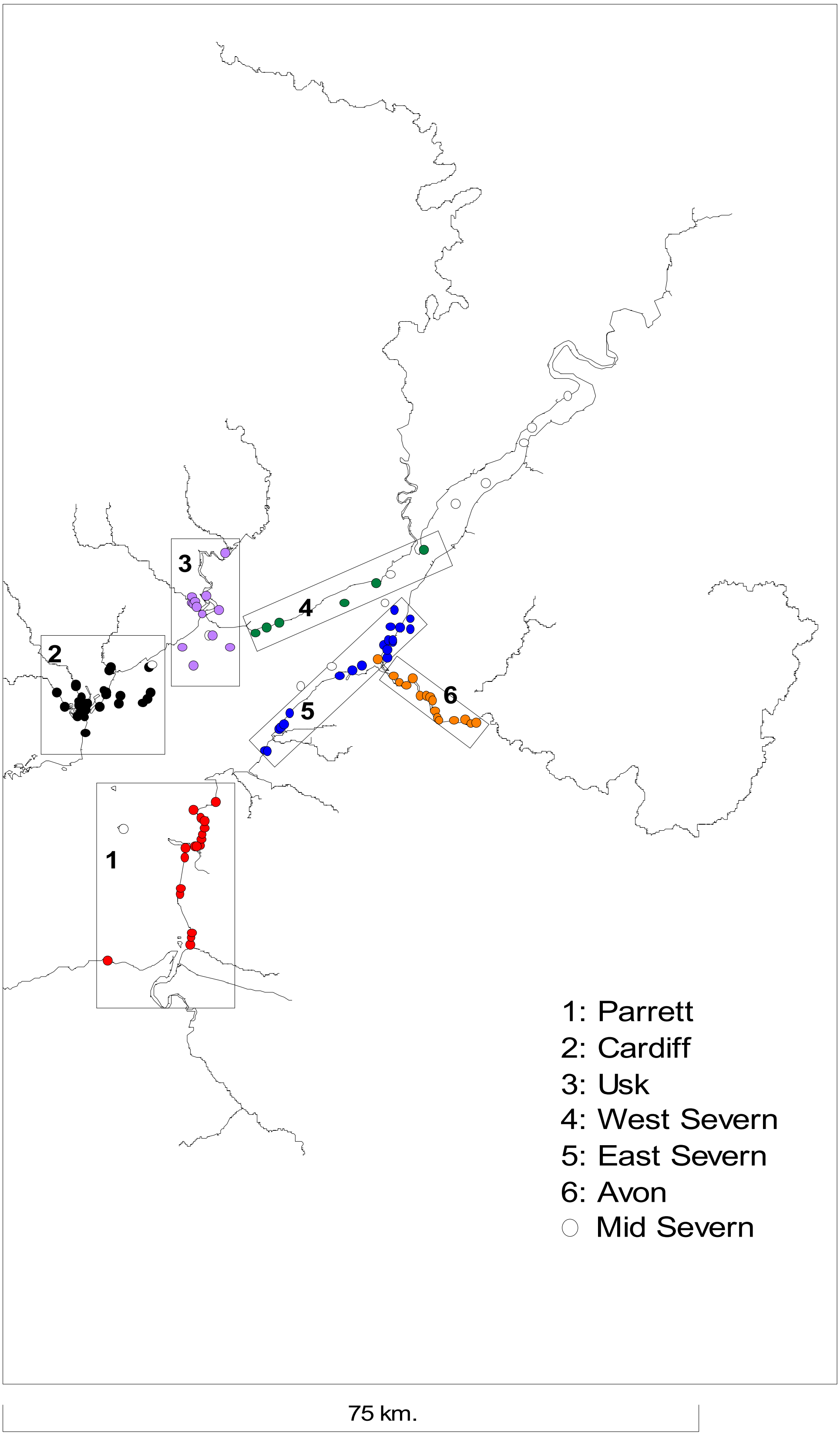


Figure 12. Location of estuarine water sampling sites in the EA data-base, and zonation used to derive summary statistics.

7.1 Toxic Contaminants

7.1.1 Metals

Results from the National Monitoring Programme (NMP) surveys of marine water chemistry (metals, nutrients, organics, including PAH's) conducted between 1992 and 1995 revealed no evidence that EQS standards for Cd, Cu, Ni, Pb and Zn were exceeded in the Severn Estuary pSAC though concentrations did increase from offshore, through intermediate, to inshore sites and included some of the highest Cd values in the UK.

As a general rule EA axial profiles confirm previous observations that dissolved metal concentrations gradually increase from the mouth of the Estuary upstream, towards the mid-Estuary, at least for Cd, Cu, Mn, Hg, Ni and Zn. Concentrations sometimes decrease again toward the tidal limit (see also profiles in section 5.1.2 and Ellis, 2002).

As indicated in the previous sections this pattern is most probably due to the major inputs sited in the middle reaches of the Estuary, coupled with natural processes of accumulation that can occur in low salinity regions.

Results for dissolved metals from the helicopter surveys reveal almost universal compliance for mid-channel samples, between 1992-7 (Ellis, 2002). At nearshore stations close to outfalls and rivers however, elevated concentrations could give rise to potential water quality issues. These are discussed here on a metal by metal basis, based on EA statistics for freshwater, estuarine water and outfall data, collected over the last ten years.

Arsenic

Riverine sources to the pSAC have been assessed by comparison of monitoring data for harmonised monitoring points (HMP[1]), or their equivalent. Only limited quality assured data exist for As in the various catchments. The EQS for dissolved As in freshwater is 50µg l^{-1} and summary statistics for the Avon, Parrett, Tone and Severn from the EA database for 2001 indicate compliance. All values were at least ten times lower than the EQS. Highest values, up to ~ 5 µg l^{-1}, were reported in the Tone, see figure 13. Data for other HMP sites appear to represent (½) detection limits (of the order of 0.5 µg l^{-1}). No obvious temporal trends were observed in annual mean data.

[1] Harmonised monitoring point is usually the freshwater site immediately upstream of the tidal limit used to guage riverine inputs to the Estuary. If no HMP data available nearest equivalent site has been taken.

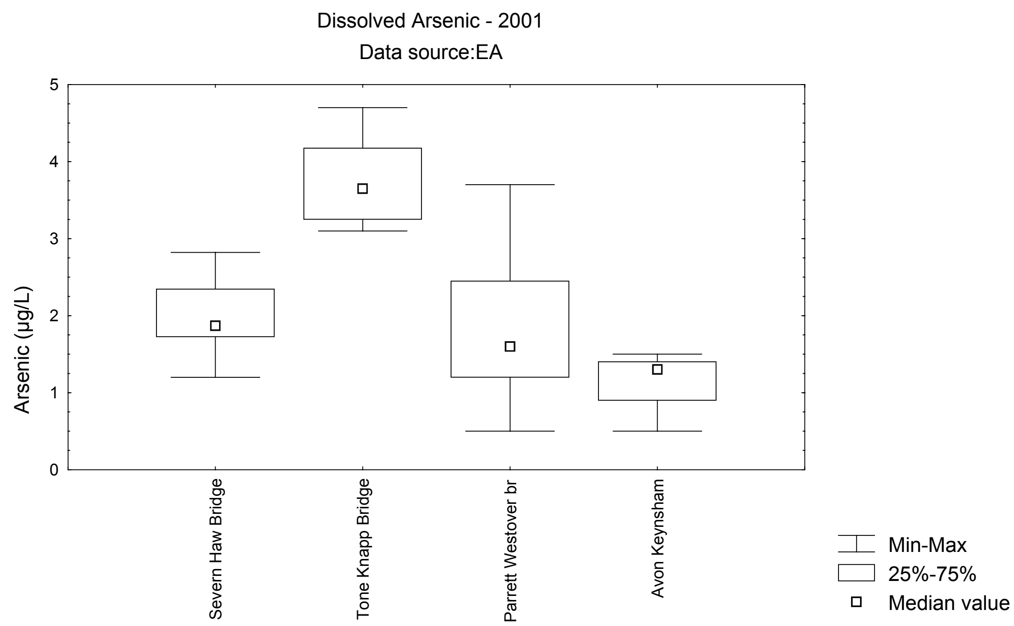


Figure 13. Concentrations of dissolved As µg l^{-1} at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data are for 2001. Data source EA.

The absence of riverine inputs is reflected in mean background concentrations across the entire Estuary (figure 14) which are well below the EQS for estuarine and coastal waters (25 µg l-1). Maximum values in the Severn East region are relatively high and suggest possible point source inputs here.

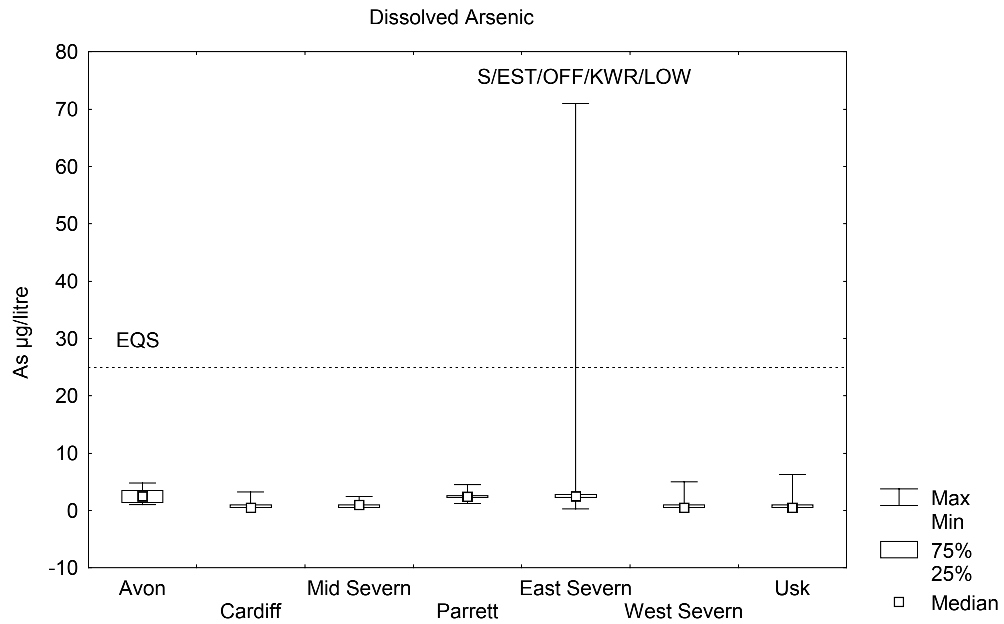


Figure 14. Concentrations of dissolved As µg l^{-1} in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Analysis of available data for individual sites reveals this pattern in more detail. Of the ten highest values in the database (table 20), nine are in the Severn East region. Four values are above the EQS value. The main area of concern would appear to lie in the Avonmouth region.

Table 20. Top ten highest individual As values in EA database for the Severn Estuary over the last decade

Sampling point name	NGR	Region	date	As µg l-1
S/EST/OFF/KWR/LOW	ST5100080250	severn e	9/9/91	71
S/EST/OFF/S.PILL/LOW	ST5160082300	severn e	9/9/91	69
S/EST/OFF/HMOUTH/LOW	ST5135080900	severn e	9/9/91	69
S EST KINGSTON SEYMOUR EQS HI	ST3836068839	severn e	7/16/96	26.6
KWR EQS SURVEY POINT	ST5130079800	severn e	7/5/94	17.2
KWR EQS SURVEY POINT	ST5130079800	severn e	9/21/92	14.6
ICI MIXED EQS SPOINT	ST5380083200	severn e	9/28/92	10.8
D/S ST.REGIS OUTFALL	ST5014087050	severn w	12/20/01	5
SEV EST OFF ICI HI	ST5210084100	severn e	10/1/96	8.4
S/EST/OFF/HMOUTH/LOW	ST5135080900	severn e	9/28/92	7.6

Invariably, however, these high values represent single extreme events, are more than 5 years old, and are probably outliers (see example for site 'SEVERN ESTUARY KINGSTON SEYMOUR EQS HIGH TIDE', figure 15). Average concentrations at each site lie below the EQS. Thus the mean value at the site with the highest observed concentration (71 µg l^{-1} - 'S/EST/OFF/KWR/LOW')[1] demonstrated total compliance with the EQS for As.

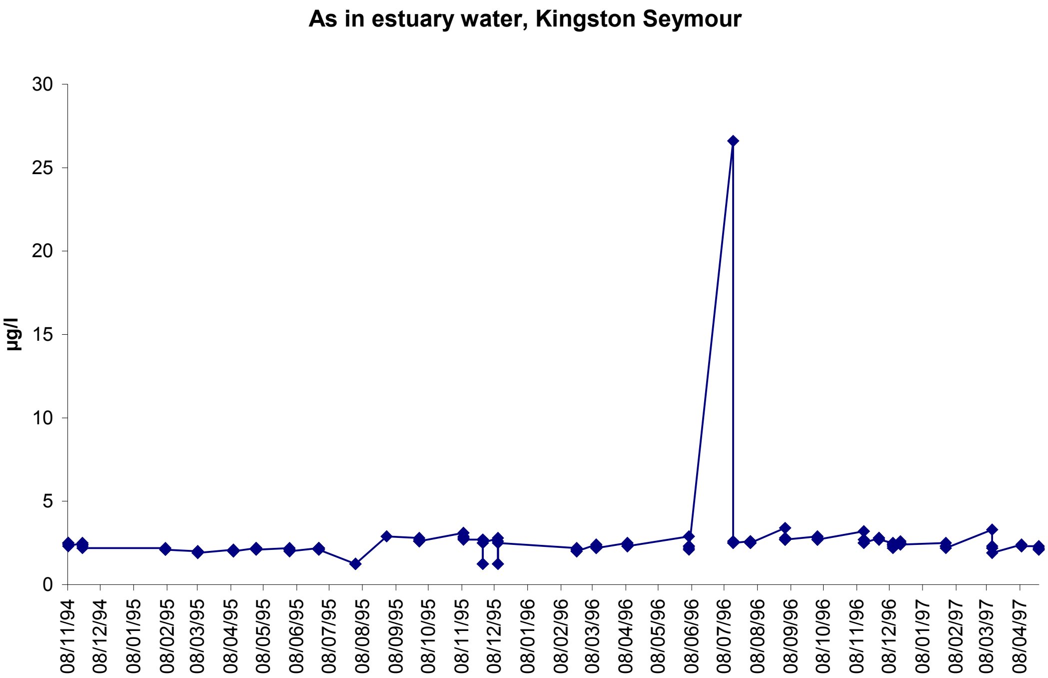


Figure 15. Dissolved As concentrations for EA site 'SEVERN ESTUARY KINGSTON SEYMOUR EQS HIGH TIDE' illustrating occurrence of single outlying value.

[1] "KWR" refers to Kingsweston Rhine. This is technically a watercourse but has no use other than as an effluent carrier. It receives discharges from Rhone Poulenc (now Rhoddia) and Britannia Zinc and is almost entirely contained within these sites. Above this there is virtually no flow. To make it suitable for stream life or "other uses" would mean tighter authorisation limits, cleaning up of the contaminated land and treatment of KWR sediments (EA pers comm.2002).

The only available summary statistics for discharges were for Cardiff Central, Cardiff East, Western Valley trunk sewer and Ystradyfodwg & Pontypridd trunk sewer (all data from the 1990s, all indicating annual averages below 2 µg l^{-1}). It is doubtful that on this evidence that a major update of consents is needed for As, though more careful monitoring seems warranted, particularly new sources in Avonmouth.

No obvious long-term trends were seen in the data.

Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in means at individual sites in the period between 1989-1990, but subsequently there were increases in means at a number of sites between in 1991 and 1992 suggesting this may represent natural variability rather than any systematic trend.

Cadmium

Riverine sources to the pSAC have been assessed by comparison of monitoring data for HMP, or their equivalent, in the various catchments (where 'reliable' data are available). The EQS for Cd in all fresh waters is 5µg l^{-1} and relates to 'total' rather than dissolved metal. Available summary statistics from the EA database for 2001 indicates mean values were less than 0.1µg l^{-1} (i.e. compliance) with occasional higher values (up to ~ 0.3 µg l^{-1} for the Severn at Haw Bridge), as shown in figure 16.

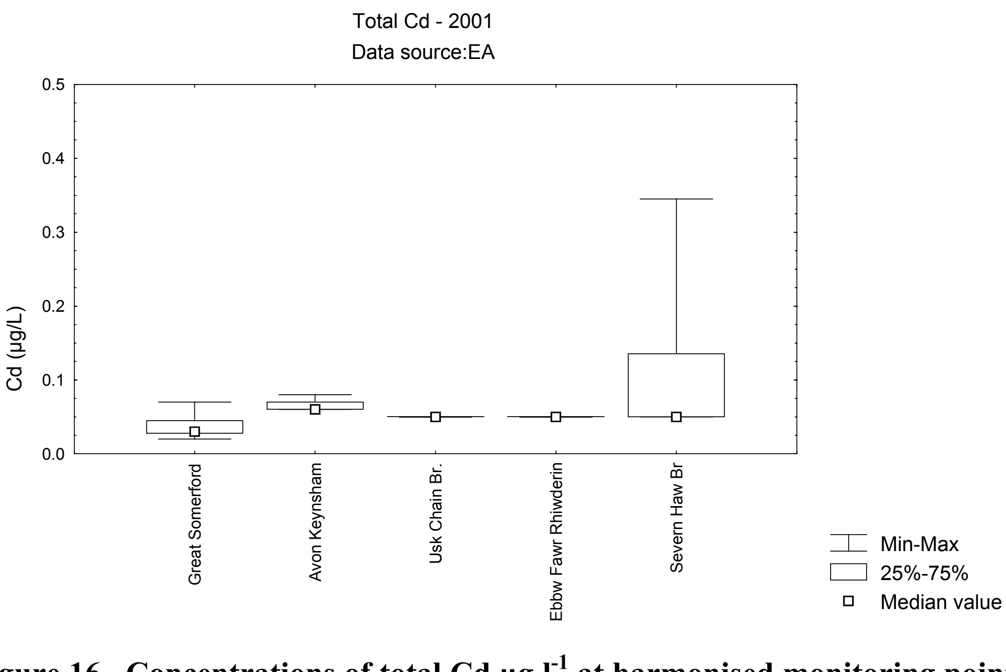


Figure 16. Concentrations of total Cd µg l^{-1} at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data are for 2001. Data source EA.

Data for dissolved Cd (figure 17) confirms low levels of Cd at the majority of sites. Even where elevated levels occur as in the Avon at Keynsham they are an order of magnitude below the EQS threshold.

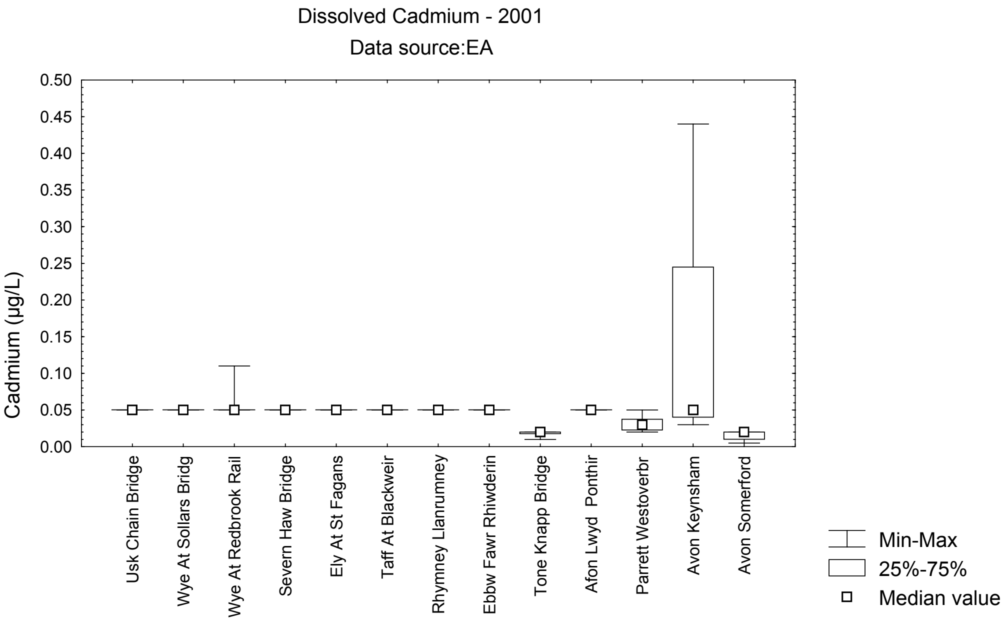


Figure 17. Concentrations of dissolved Cd µg l^{-1} at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data are for 2001. Data source EA.

Comparison of annual averages for dissolved Cd suggests improvements at most sites over the last decade (and no increases). It some cases however, it is uncertain whether these improvements are real or are due to lowered detection limits. For example a downward trend is implied in the data for Haw Bridge on the Severn which is partly artifactual due to the fact that (½) detection limits have dropped from 0.25 in 1990 to 0.05 in 2000 onwards.

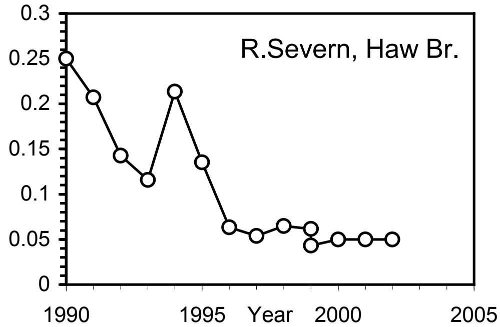


Figure 18. 'Temporal trend' for dissolved Cd data (µg l^{-1}) at Haw Bridge on the River Severn –partly artifactual due to lowered detection limit.

The pattern of Cd in estuarine water is similar to that for As. Mean background concentrations plotted on a regional basis are below the EQS for estuarine and coastal waters (2.5 µg l^{-1}) across the entire Estuary (figure 19). Maximum values in the Severn East region suggest possible small point source inputs in the Avon, Parrett and Usk regions but influence is greatest in the East Severn, again focused on Avonmouth. Analysis of available data for individual sites reveals this pattern in more detail. All ten highest values in the database are in the vicinity of the KWR survey point, close to the smelter and other discharges at Avonmouth.

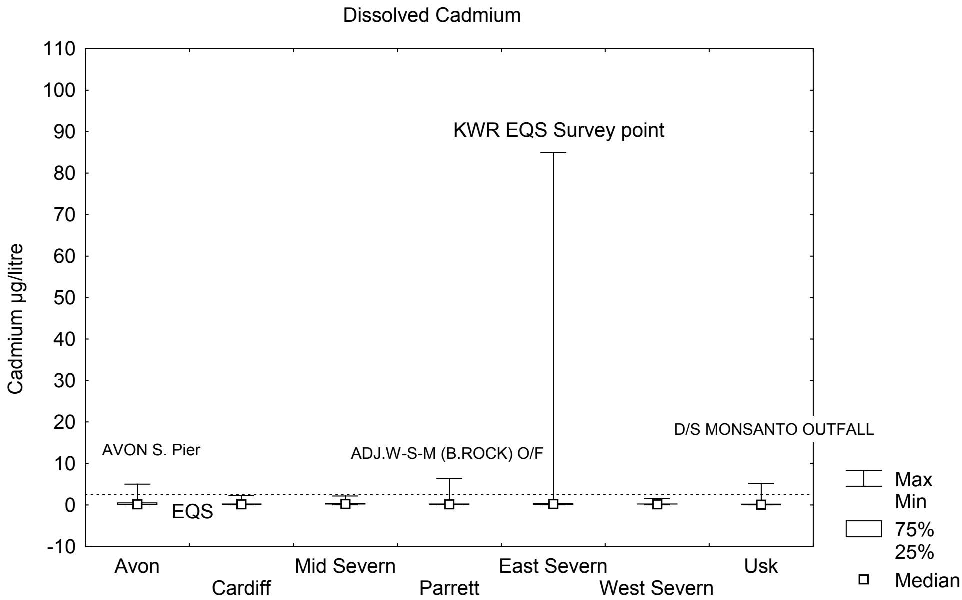


Figure 19. Concentrations of dissolved Cd µg l^{-1} in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

'KWR EQS Survey point' has an average Cd concentration of 47 µg l^{-1}, significantly above the EQS. However the data are for 1990 to 1994. Data for the 'KWR Survey point high water' illustrates extreme variability over short time scales (figure 20) presumably related to discharge rates and flow conditions. A note on more recent (self-monitoring) of the discharges at Avonmouth is given in section 5.1.1. In view of some of the occasional high values, further chemical (and biological) surveillance, and review of consents may be warranted here for Cd. Other sites where Cd values may occasionally exceed the EQS by a small margin are ADJ.W-S-M (B.ROCK) O/F; D/S MONSANTO OUTFALL and BRISTOL AVON AT SOUTH PIER (figure 19).

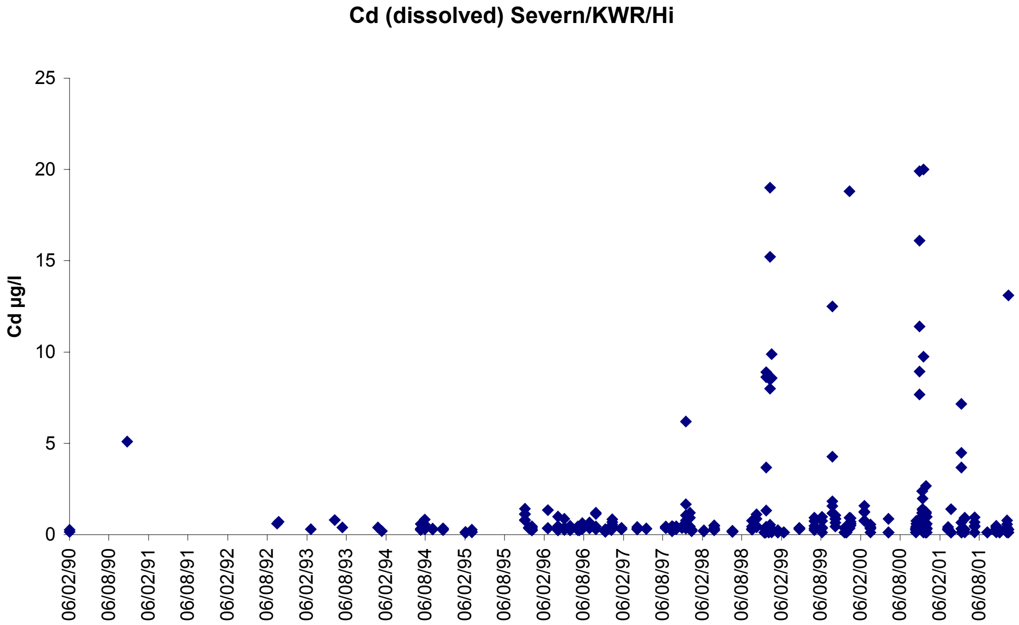


Figure 20. Dissolved Cd concentrations for EA site 'SEVERN ESTUARY OFF KWR AT HIGH WATER' **illustrating considerable temporal variation.**

Overall, meaningful data on discharges were not sufficiently extensive to evaluate point sources in terms of influences on estuarine water quality or biota. Available summary statistics for dissolved Cd are shown, for information, in table 21. At these particular sites there is little evidence to suggest Cd concentrations would be acutely toxic.

Table 21. Dissolved Cd in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Site	Year	Cd µg l^{-1}
BUILTH ROAD STW	1994	0.150
CARDIFF CENTRAL OUTFALL	1999	0.050
CARDIFF EAST PUMPING STA FINAL EFF DISCHARGE	1993	0.424
CARDIFF WEST PUMPING STATION	1997	0.575
HUNGER PILL STW	1996	1.583
ICI SEVERNSIDE WORKS BRISTOL ;MIXED DIS	1990	0.500
PONTHIR SEWAGE TREATMENT WORKS FINAL EFFLUENT	1996	0.350
RHYMNEY VALLEY TRUNK SEWER OUTFALL TO TIDAL SEVERN	1993	0.150

Examining temporal trend data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in means for dissolved Cd at individual sites in the period between 1984-1992 – presumably as a result of water quality improvement schemes in the 1980s. Virtually all sites complied with the EQS. However in the last survey examined, in 1997, this trend was reversed; sites upstream of Newnham church, in the inner Severn, exceeded the 2.5 µg l^{-1} annual average. This also merits further study.

Chromium.

Riverine sources to the pSAC have been assessed by comparison of monitoring data for HMPs, or their equivalent, in the various catchments (where 'reliable' data are available). The EA database indicates that mean values for 2001 are between 0.5 and 1.5µg l^{-1}, as shown in figure 21. The EQS for Cr in fresh waters suitable for salmonids ranges between 5 and 50 µg l^{-1} depending on hardness. Since most of the catchments examined probably have a hardness towards the upper end of the range a typical EQS value of 20 µg l^{-1} probably applies. In this case all the points examined would comply.

Despite uncertainties relating to detection limits, there are indications in reductions of annual average Cr values for Ebbw Fawr, Avon, upper Wye and the river Taff. In the lower Wye, at Redbrook Bridge this trend is reversed since 1995.

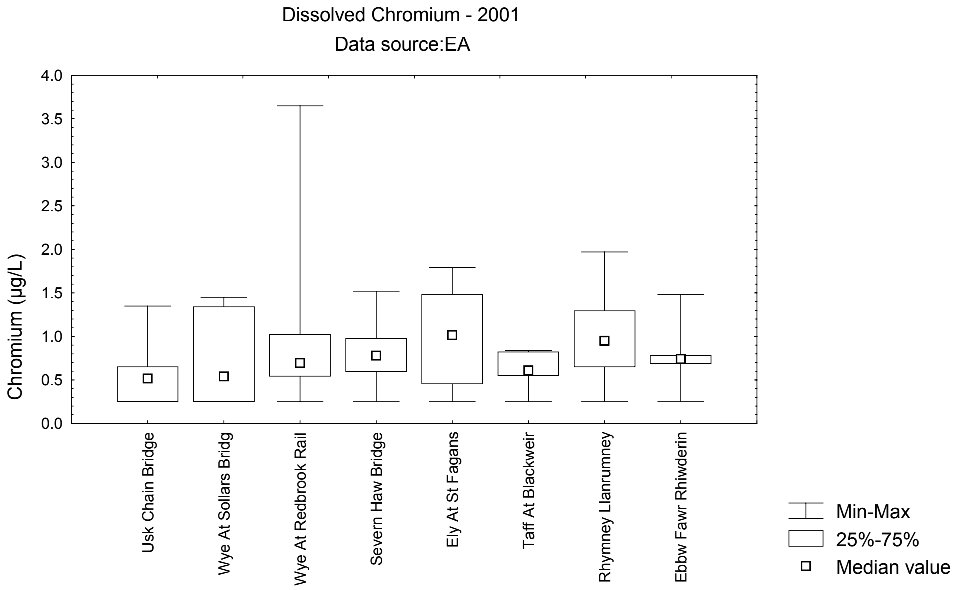


Figure 21. Concentrations of dissolved Cr µg l^{-1} at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data are for 2001. Data source EA.

The pattern of Cr in estuarine water plotted on a regional basis, shown in figure 22. Average values are below the EQS (15 µg l^{-1}) across the entire Estuary. Maximum values have sometimes exceeded the EQS in the Severn East region (ICI Mixed EQS sampling point) and in the Usk and Cardiff region, suggesting possible point source inputs. However these are a small number of samples, and are not recent data; i.e. they could be considered as outliers.

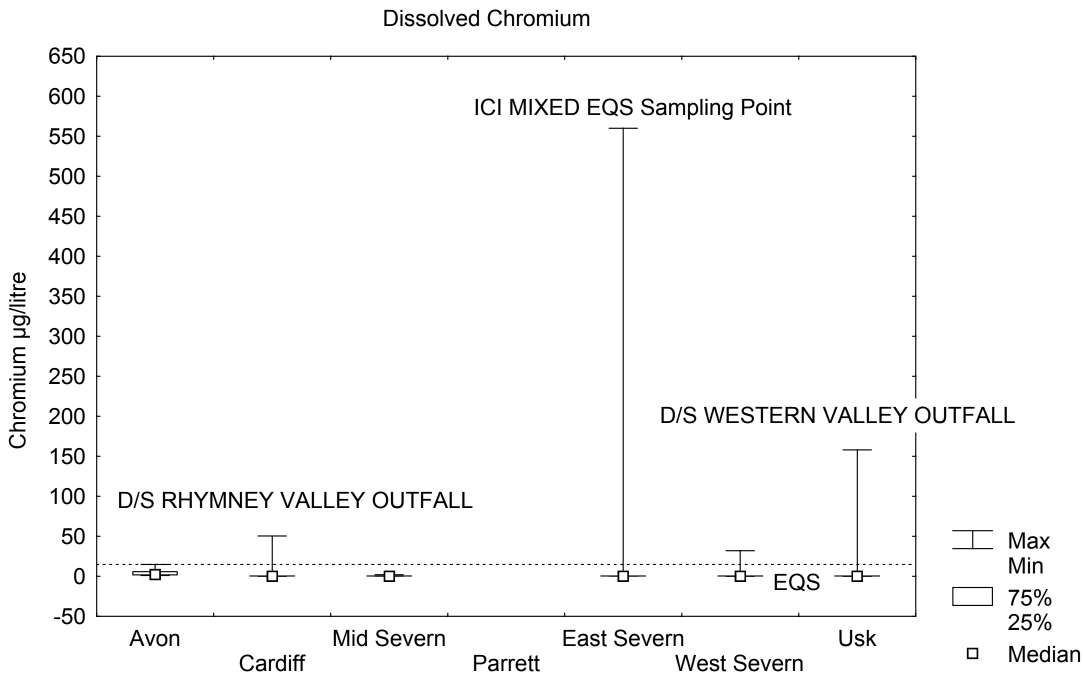


Figure 22. Concentrations of dissolved Cr µg l^{-1} in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Ten highest values for estuarine waters in the database are shown in (table 22).

Table 22. Top ten highest individual Cr values in EA database for the Severn Estuary over the last decade

Sample point Name	NGR	region	Date	Cr µg l^{-1}
ICI MIXED EQS SPOINT	ST5380083200	severn e	9/21/92	560
ICI MIXED EQS SPOINT	ST5380083200	severn e	9/28/92	400
D/S WESTERN VALLEY OUTFALL	ST2951880051	usk	3/22/00	158
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	10/7/97	70
D/S RHYMNEY VALLEY OUTFALL	ST2584477990	cardiff	7/16/98	50.5
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	1/27/97	46
S/EST/OFF/HMOUTH/HIGH WATER	ST5189380720	severn e	10/7/97	42
D/S ST.REGIS OUTFALL	ST5014087050	severn w	7/25/96	32
HOLESMOUTH EQS POINT	ST5190080800	severn e	2/6/90	26
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	1/27/97	24.5

Though highest recorded concentrations tend to appear for earlier years in the database, unequivocal indications of temporal trends for Cr in estuarine waters are not evident.

Examining a limited amount of data for individual sites in the helicopter surveys, Ellis (2002) indicate compliance with the EQS (stated to be 5 µg l^{-1}) below Newham Church, but failure upstream (1997).

Discharge figures supplied do not adequately address the sources implied from the observations of Cr in estuarine waters. For information, available summary statistics for dissolved Cr in trade and sewage discharges are shown in table 23. At these particular sites there is little evidence to suggest Cr concentrations would be acutely toxic; however more comprehensive data on these and other discharges are needed to assess possible influences on estuarine water quality and sublethal impact on biota.

Table 23. Dissolved Cr in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Site	Year of Sample	Cr µg l^{-1}
BUILTH ROAD STW	1994	7.75
CARDIFF CENTRAL OUTFALL	1997	6.43
CARDIFF WEST PUMPING STATION	1993	12.17
HUNGER PILL STW	1996	6.57
PONTHIR STW FINAL EFFLUENT	1995	5.81

Copper

Riverine sources to the pSAC have been assessed by comparison of monitoring data for HMP, or their equivalent, in the various catchments (where 'reliable' data are available). The EA database for 2001 indicates mean values between 0.5µg l^{-1} (in the Usk) and approximately 3 µg l^{-1} (in the Severn at Haw Bridge), as shown in figure 23. The EQS for Cu in freshwater is in the range 1 - 28 µg l^{-1} depending on hardness. Since most of the catchments examined probably have a hardness towards the upper end of the range, a typical EQS value of 10 µg l^{-1} probably applies. In this case all the points examined would comply.

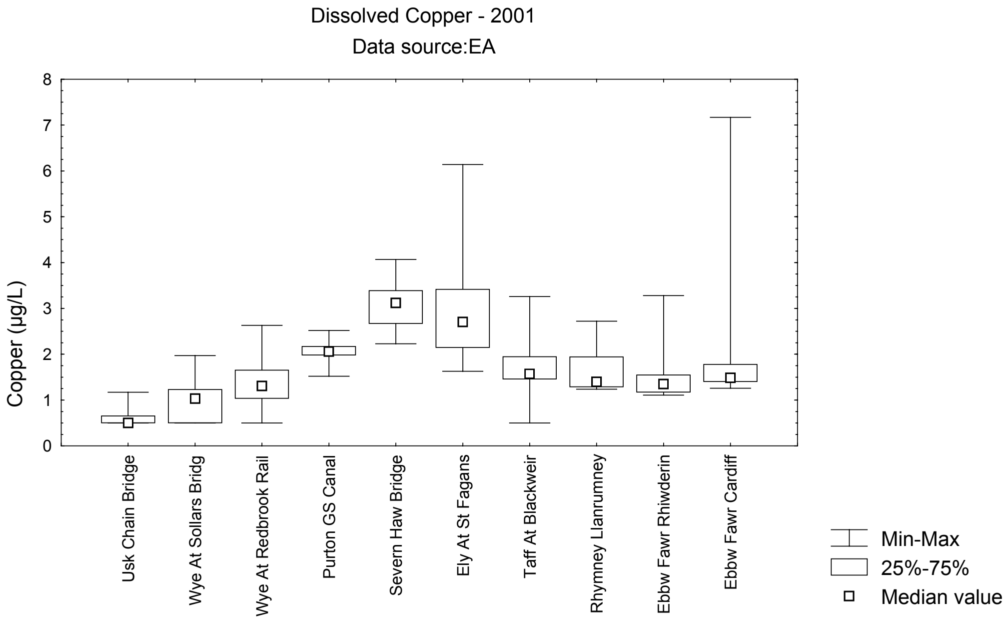


Figure 23. Concentrations of dissolved Cu µg l^{-1} at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data are for 2001. Data source EA.

Annual average dissolved Cu levels appear to have shown a significant ($P<0.05$) downward annual trend in several rivers including the Severn, Usk, Taff and Avon (figure 24). Similar trends are evident at Sollars Beach on the Wye, Sharpness Canal, Ebbw Fawr (Rhiwderin), and at sites on the rivers Tone, Huntspill, Axe, Afon Lwyd, Banwell and Parrett.

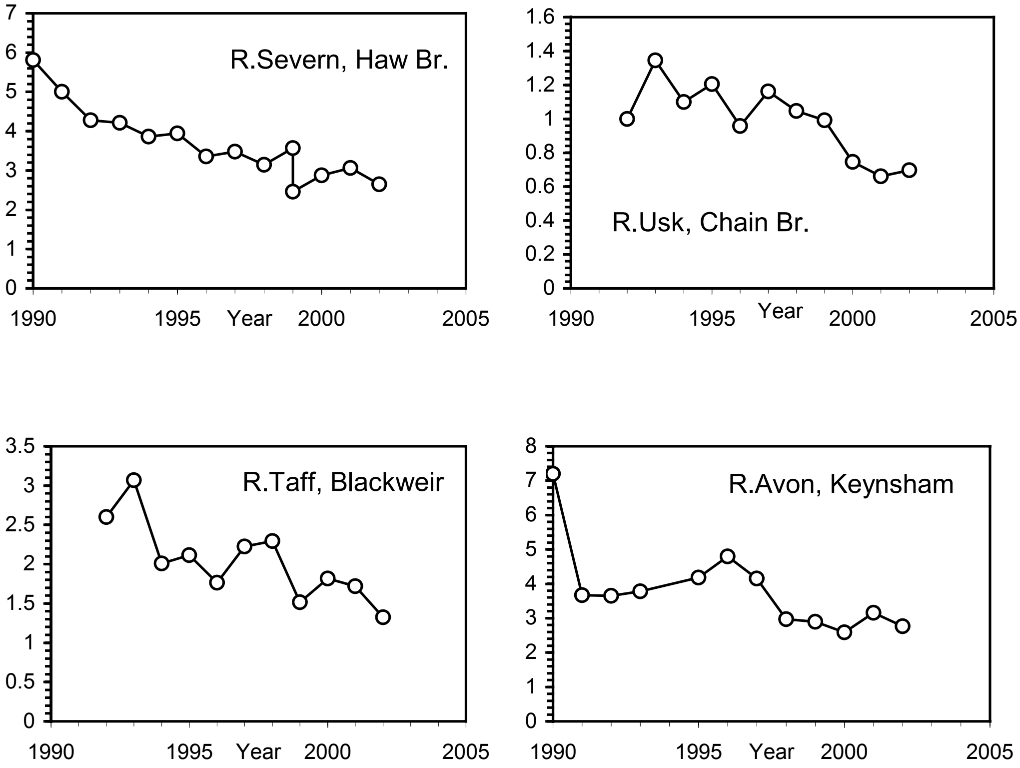


Figure 24. Trends in concentrations of dissolved Cu (annual averages, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding the Severn Estuary. Data source EA.

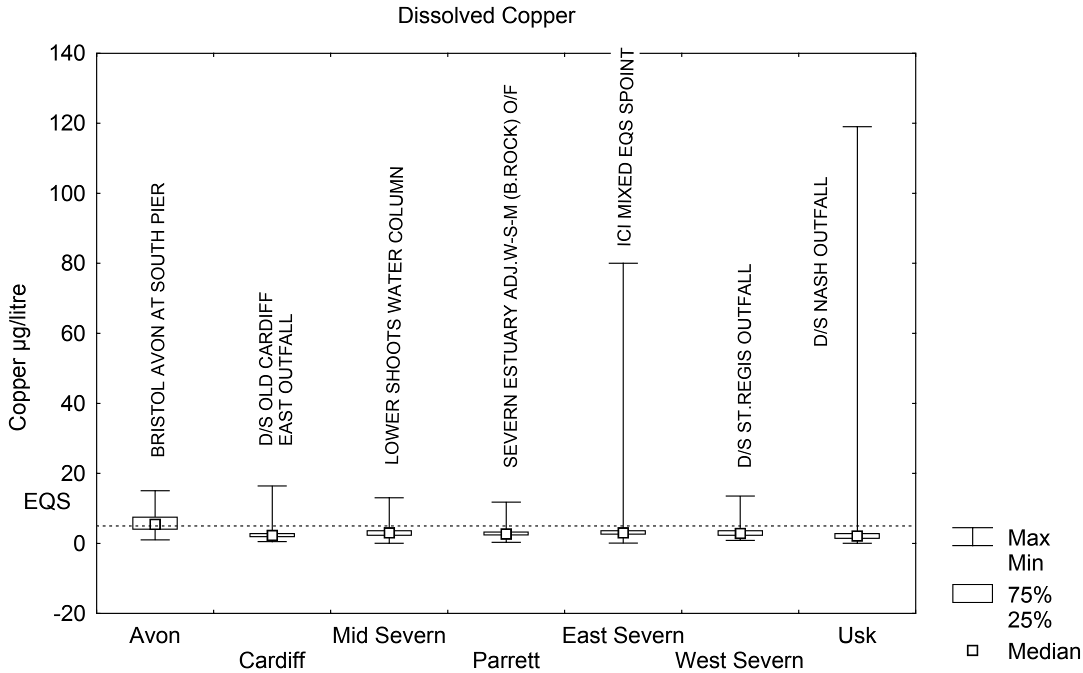


Figure 25. Concentrations of dissolved Cu µg l^{-1} in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Sites of maximum Cu concentration for each region are labelled. Data source EA.

The pattern of Cu in estuarine water, plotted on a regional basis is shown in figure 25. Average regional values are below the EQS (5 µg l^{-1}) across most of the Estuary. The mean for the Avon at south pier is 6 µg l^{-1} (though sampling appears to have ceased in 1994) and for a site downstream of the Nash outfall, on the Usk, the mean value for data between 1995 and 2001 is 7 µg l^{-1}. Maximum Cu values have sometimes exceeded the EQS in all regions, implying possible point source inputs (see table 24 for top ten values). Although these represent a small number of samples, and some are not recent data, nevertheless they should be reviewed further in view of the potential toxicity of Cu.

Analysis of compliance in the mid-stream survey data by Ellis (2002) indicates that three sites in the inner Estuary would have failed (Newham Church, Minsterworth Ham and Upper Parting) whilst annual averages in the remaining 22 sites along the Estuary were below the 5µg l^{-1} threshold.

Table 24. Top ten highest individual Cu values in EA database for the Severn Estuary over the last decade

Sampling point name	NGR	Estuary	Date	Cu µg l^{-1}
D/S NASH OUTFALL	ST3344084140	usk	09/06/01	119
ICI MIXED EQS SPOINT	ST5380083200	severn e	07/05/94	80
D/S NASH OUTFALL	ST3344084140	usk	11/21/00	69.5
HOLESMOUTH EQS POINT	ST5190080800	severn e	07/05/94	40
D/S MONSANTO OUTFALL	ST3456580051	usk	06/02/98	23.2
KWR EQS SURVEY POINT	ST5130079800	severn e	07/05/94	20
TRIB DITCH STUP PILL EQS POINT	ST5380082100	severn e	09/21/92	19.3
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	11/08/01	18.8
D/S NASH OUTFALL	ST3344084140	usk	02/27/01	18.5
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	08/01/95	17

No obvious distinctive long-term trends can be seen for Cu in estuarine waters. Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated some short-term improvements in water quality (26 stepwise decreases in means at individual sites) in the period between 1990-1993; subsequently there were 2 increases in means between 1995-1997.

Discharge figures supplied do not adequately address the sources responsible for the observed elevated Cu levels in estuarine waters. For information, available summary statistics for dissolved Cu in some trade and sewage discharges are shown in figure 26. These indicate potential sources from STWs. There appears to be little comparable data for major industrial effluents. More comprehensive geographical and spatial data for these and other discharges is needed to assess possible influences on estuarine water quality and sublethal impact on biota, since Cu is potentially one of the most toxicologically significant metals.

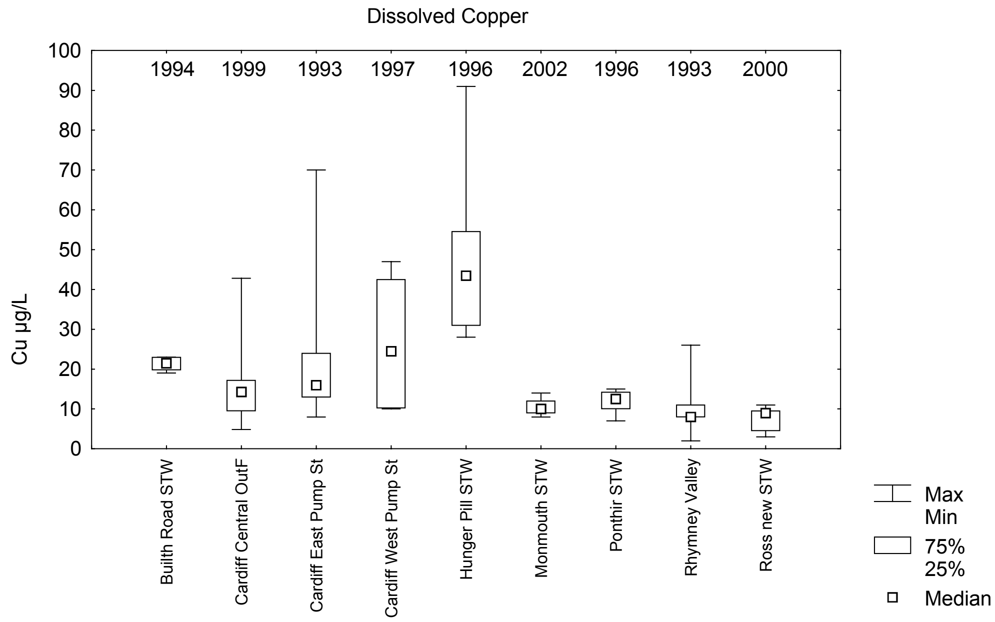


Figure 26. Dissolved Cu in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Iron

Concentrations of dissolved Fe in riverine sources to the pSAC based on comparison of monitoring data for HMP, or their equivalent, are shown in figure 27. The EQS for dissolved Fe is 1000µg l^{-1}. The data for 2001 indicate compliance for all catchments. (This also applies to some other sites not shown because of limited data for summary statistics in 2001, such as the Avon). Mean values were generally ten times lower than the EQS. No major temporal trends were observed in annual mean data for the last ten years.

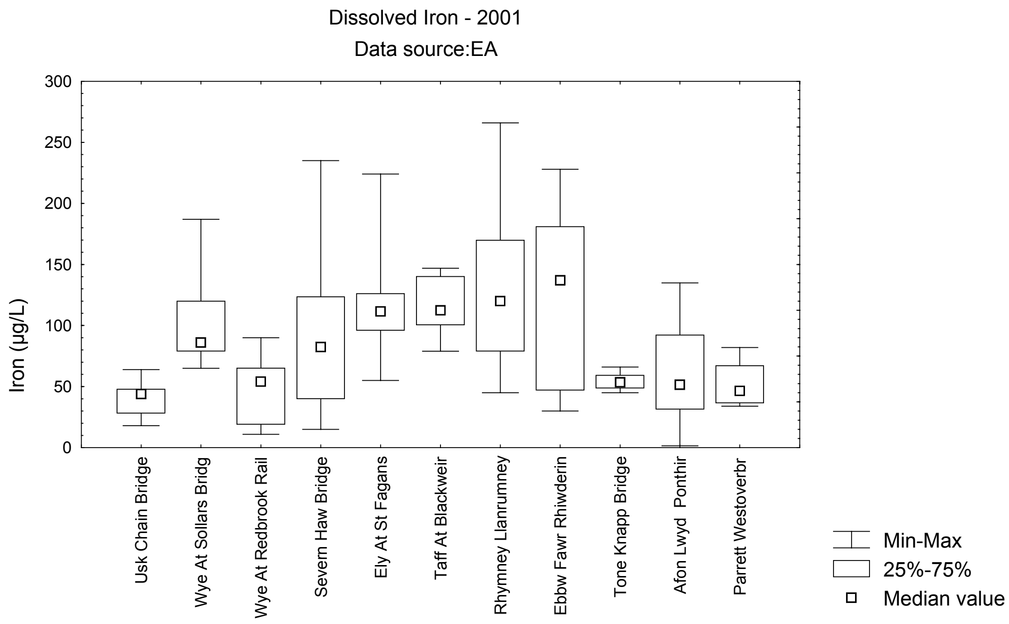


Figure 27. Trends in concentrations of dissolved Fe (annual averages, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding the Severn Estuary. Data source EA.

The pattern of Fe in estuarine water, plotted on a regional basis is shown in figure 28. Average regional values are invariably below the EQS (1000 µg l^{-1}) across most of the Estuary. Maximum Fe values have sometimes exceeded the EQS in the Severn east region, near Avonmouth, and, by a small margin, in the Usk, upstream of the Caerleon Roadbridge (see table 25 for top ten values). However, 'exceedences' represent a small number of samples, and most are not recent data.

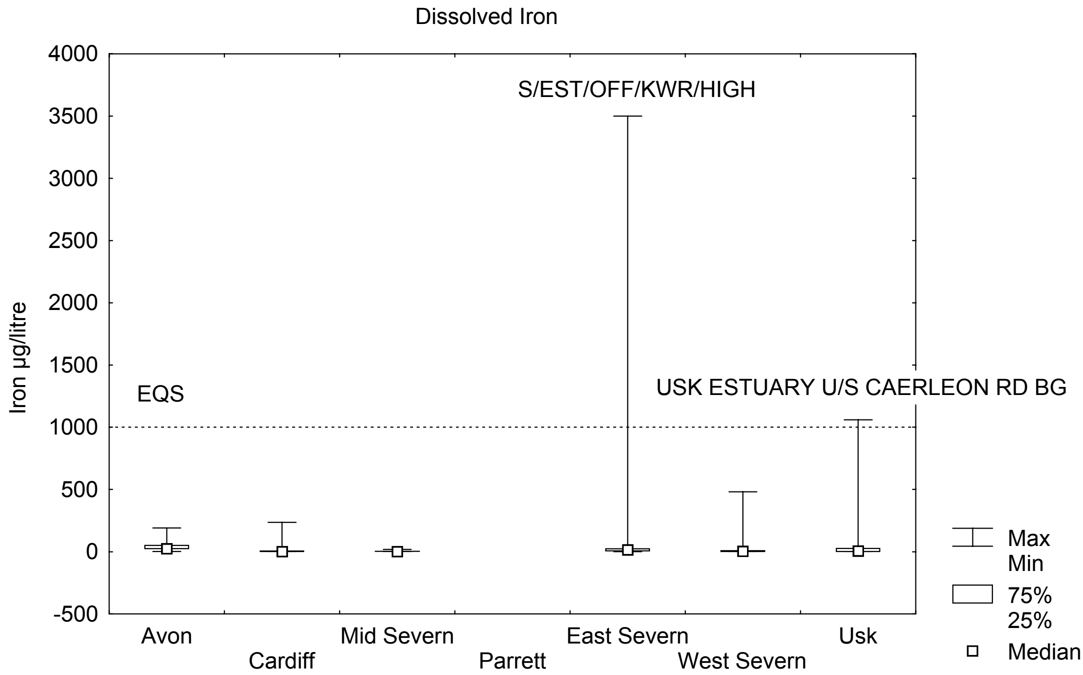


Figure 28. Concentrations of dissolved Fe µg l^{-1} in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Table 25. Top ten highest individual Fe values in EA database for Severn Estuary water over the last decade

Sample site name	NGR	region	date	Fe µg l^{-1}
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	05/01/95	3500
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	05/01/95	3000
USK ESTUARY U/S CAERLEON RD BG	ST3412090260	usk	03/02/93	1060
ICI MIXED EQS SPOINT	ST5380083200	severn e	09/21/92	690
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	05/31/95	600
SEV EST OFF ICI HI	ST5210084100	severn e	11/27/95	506
D/S ST.REGIS OUTFALL	ST5014087050	severn w	06/04/97	481
D/S ST.REGIS OUTFALL	ST5014087050	severn w	04/27/98	319
USK ESTUARY U/S CAERLEON RD BG	ST3412090260	usk	01/14/00	282
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	05/31/95	250

No distinctive long-term trends can be seen for Fe in estuarine waters.

Available summary statistics for dissolved Fe in trade and sewage discharges are shown in table 26. These represent the highest annual average concentrations for dissolved Fe in the supplied data-base but may not adequately address the sources responsible for the observed elevated Fe levels in estuarine waters. More comprehensive geographical and spatial data on these and other discharges are needed to assess possible influences on estuarine water quality and sublethal impact on biota.

Table 26. Dissolved Fe in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Site	year	Fe µg l^{-1}
B.S.C. WHITHEADS (Usk)	2001	987
HEREFORD EIGN STW	2002	133
HEREFORD ROTHERWAS STW	2002	129
LLYSWEN: NEAR HAY-ON-WYE	2000	148
R WYE AT BUILTH RAILWAY BRIDGE	2000	166
ROSS NEW STW (LOWER CLEEVE)	2000	104
WYELANDS ABSTRACTION POINT	2000	70

Nickel

Virtually all 2001 dissolved nickel data for HMPs (or equivalent fresh water sites) in the Avon, Tone, Parrett, Ebbw Fawr, Rhymney, Taff, Ely, Usk, Severn and Wye are at or below detection limits giving mean values, based on ½ DL, of 1.5-2.5 µg l^{-1}. The EQS for dissolved Ni ranges between 50 and 200 µg l^{-1} depending on hardness, therefore all the data points examined would comply. There are no unequivocal temporal trends for Ni, probably because many of the summary statistic data involve detection limit-derived values, and are therefore not considered meaningful.

The pattern of Ni in estuarine water, plotted on a regional basis is shown in figure 29. Average regional values are invariably below the EQS (30 µg l^{-1}) across the Estuary. Maximum Ni values have sometimes exceeded the EQS in the Severn east region, near Avonmouth and, by a small margin, in the Usk, downstream of the Nash outfall and at the Bedwin NMMP site, Severn West (see table 27 for top ten values). However, these represent a small number of samples (means for these sites are below the EQS), and most are not recent data.

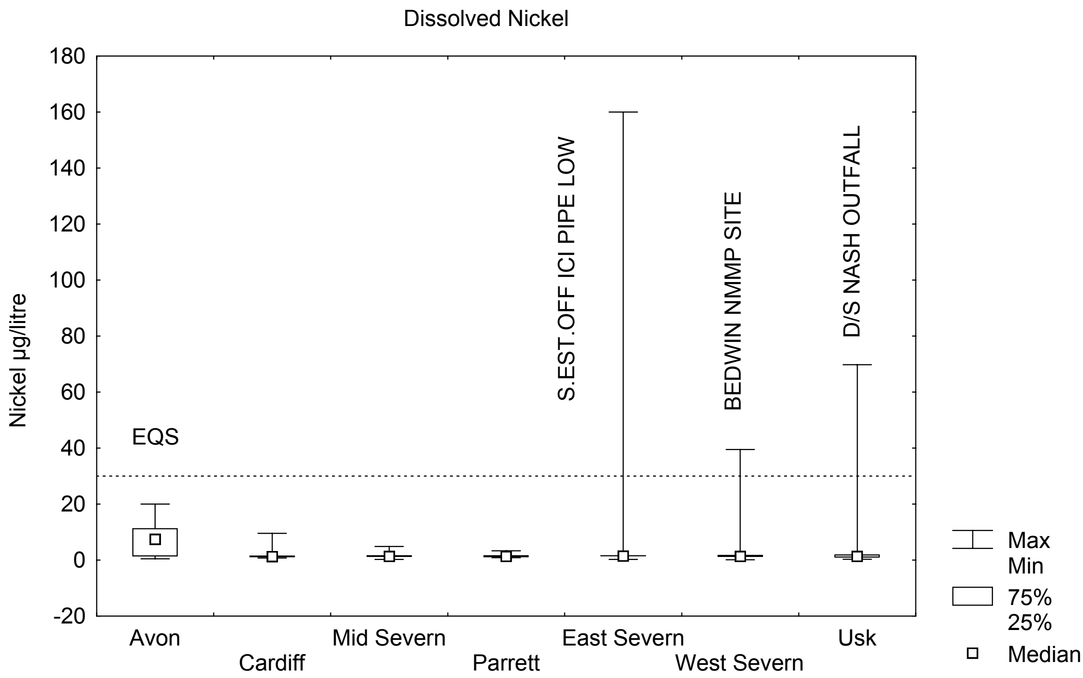


Figure 29. Concentrations of dissolved Ni (µg l^{-1}) in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Table 27. Top ten highest individual Ni values in EA database for the Severn Estuary over the last decade

Sampling point name	NGR	region	date	Ni µg l^{-1}
S.EST.OFF ICI PIPE LOW	ST5210084100	severn e	10-Apr-90	160
S/EST/OFF/KWR/LOW	ST5100080250	severn e	10-Apr-90	115
S/EST/OFF/HMOUTH/HIGH WATER	ST5189380720	severn e	10-Apr-90	110
S/EST/OFF/HMOUTH/LOW	ST5135080900	severn e	10-Apr-90	110
D/S NASH OUTFALL	ST3344084140	usk	06-Sep-01	69.8
HOLESMOUTH EQS POINT	ST5190080800	severn e	10-Apr-90	60
S/EST/OFF/KWR/HIGH	ST5131078950	severn e	29-Sep-95	60
S/EST/OFF/S.PILL/HI	ST5272382171	severn e	10-Apr-90	60
S/EST/OFF/S.PILL/LOW	ST5160082300	severn e	10-Apr-90	60
KWR EQS SURVEY POINT	ST5130079800	severn e	10-Apr-90	50

At the majority of sites there is no clear evidence of consistent temporal change in dissolved Ni concentrations. Upstream of the Caerleon Roadbridge, in the Usk Estuary there is a strong indication that concentrations may have been declining since the early 1990s (figure 30).

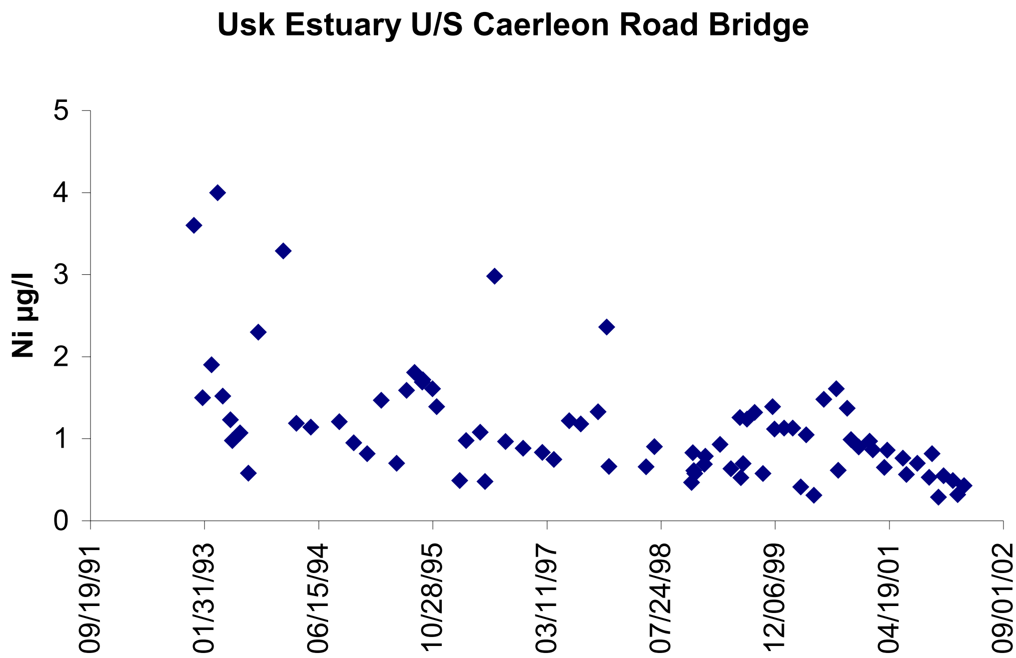


Figure 30. Temporal trend data for dissolved Ni, Caerleon Road Bridge, Usk Estuary.

Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in means in Ni concentration at individual sites (~25) in the period between 1990 –1993 whereas only one increase was observed. All examined records of sites complied with the EQS (1980-1997).

Available summary statistics for dissolved Ni in trade and sewage discharges are shown in table 28. These annual average concentrations do not appear to constitute a significant issue for the pSAC though the coverage of sites may not adequately address all the sources responsible for Ni levels in estuarine waters. More comprehensive geographical and spatial data are needed.

Table 28. Dissolved Ni in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Site	year	Ni µg l^{-1}
BUILTH ROAD STW	1994	7.8
CARDIFF CENTRAL OUTFALL	1999	3.6
CARDIFF E.PUMPING STA FINAL EFFLUENT	1993	24.4
CARDIFF WEST PUMPING STATION	1997	3.9
HUNGER PILL STW	1996	4.7
MONMOUTH STW. REDBROOK RD.	2002	1.5
PONTHIR STW FINAL EFFLUENT	1996	9.0
RHYMNEY VALLEY TRUNK SEWER TO TIDAL SEVERN	1993	2.7
ROSS NEW STW (LOWER CLEEVE)	2000	2.7

Lead

Virtually all 2001 dissolved lead data for HMPs (or equivalent fresh water sites) in the Avon, Tone, Parrett, Ebbw Fawr, Rhymney, Taff, Ely, Usk, Severn and Wye are at or below detection limits, giving mean values, based on ½ DL, of 1µg l^{-1} or less. Since the EQS for dissolved Pb ranges between 4 and 20 µg l^{-1} depending on hardness (and most of the studied rivers would have a standard toward the upper limit) all the sites examined would comply. There are no unequivocal temporal trends for Pb because many of the summary statistics involve detection limit-derived values, and are therefore not considered meaningful.

The pattern of Pb in estuarine water, plotted on a regional basis is shown in figure 31. Average regional values are invariably below the EQS (25 µg l^{-1}) across the Estuary. Maximum Pb values only exceeded the EQS in the Severn east region, near Avonmouth, and in one sample at Kingston Seymour (see table 29 for top ten values). All these data are several years old and the majority of exceptional high values could probably be classified as outliers. Means for these sites are below the EQS.

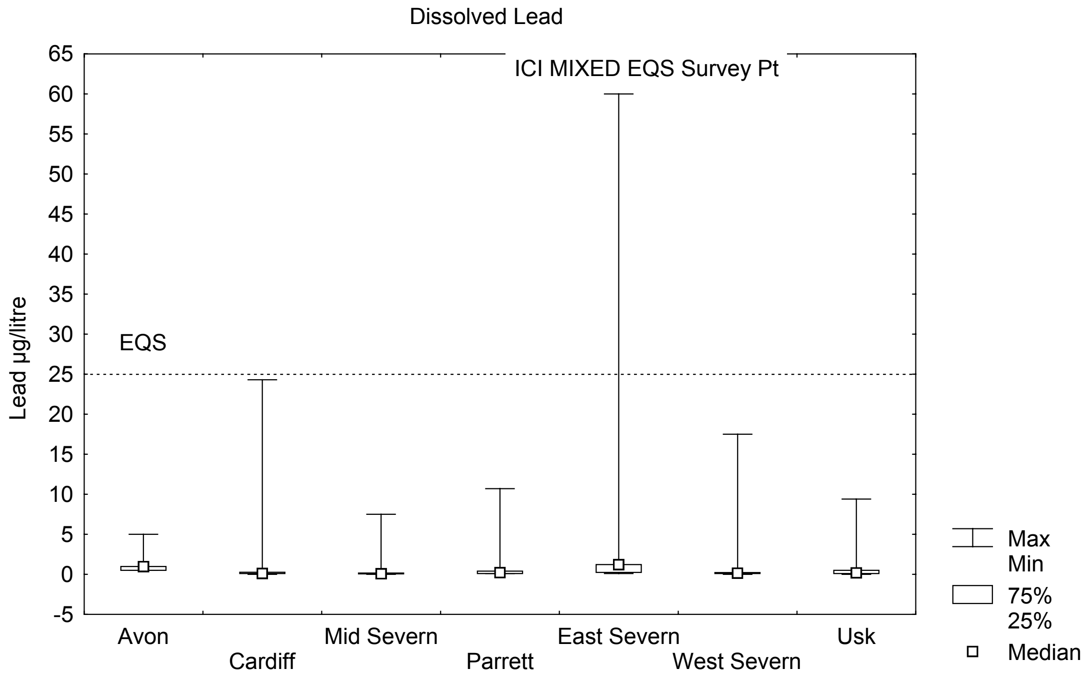


Figure 31. Concentrations of dissolved Pb (µg l^{-1}) in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Table 29. Top ten highest individual Pb values in EA database for the Severn Estuary over the last decade

Sampling point name	NGR	Estuary	date	Pb µg l^{-1}
ICI MIXED EQS SPOINT	ST5380083200	severn e	7/5/94	60
SEV EST OFF ICI HI	ST5210084100	severn e	7/10/96	34
S EST KINGSTON SEYMOUR EQS HI	ST3836068839	severn e	6/5/96	32
KWR EQS SURVEY POINT	ST5130079800	severn e	9/21/92	30
D/S OLD CARDIFF EAST OUTFALL	ST2273274940	cardiff	11/26/97	24.3
D/S ST.REGIS OUTFALL	ST5014087050	severn w	12/1/98	17.5
S EST KINGSTON SEYMOUR EQS HI	ST3836068839	severn e	4/9/96	16.4
S/EST/OFF/S.PILL/LOW	ST5160082300	severn e	4/10/90	11
S/EST/OFF/S.PILL/LOW	ST5160082300	severn e	4/10/90	11
SEVERN ESTUARY ADJ.W-S-M (B.ROCK) O/F	ST3056758682	parrett	3/13/97	10.7

At the majority of sites the data do not reveal long-term trends: one unusual exception is the mid-Severn sampling station at Redcliffe Buoy which suggests an overall increase during the 1990's, though concentrations are relatively low throughout (figure 32).

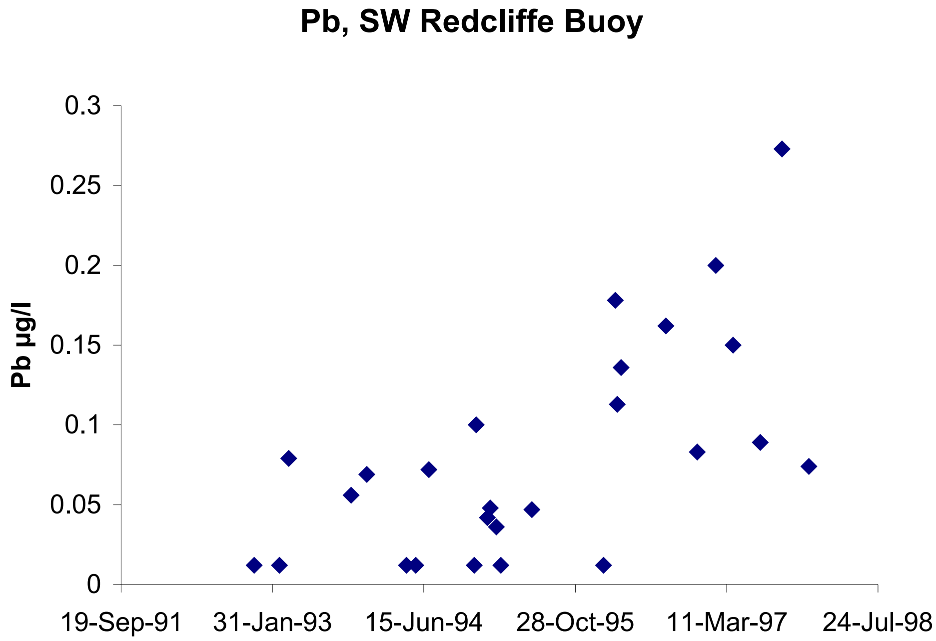


Figure 32. Temporal trend data for dissolved Pb, SW Redcliffe Buoy, Mid Severn Estuary.

Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in means in Pb concentration at individual sites in the period between 1992 –1993 though these were largely offset by increases in 1996, masking any overall trend. All examined records of sites complied with the EQS (1980-1997).

The most recent available summary statistics (annual averages) for dissolved Pb in trade and sewage discharges are shown in table 30. These concentrations could be of significance locally (particularly Hunger Pill and Cardiff central STWs). This coverage of sites does not adequately address all the potential sources to the pSAC. More comprehensive geographical and spatial data are needed in order to assess this.

Table 30. Dissolved Pb in discharges to the Severn Estuary (annual average for the latest year in which data supplied by EA)

Site	Year	Pb µg l^{-1}
BUILTH ROAD STW	1994	6.5
CARDIFF CENTRAL OUTFALL	1997	37.4
CARDIFF WEST PUMPING STATION	1997	9.7
HUNGER PILL STW	1996	85.2

It is difficult to establish temporal trend in discharges, based on such a limited data set, though it is worth highlighting the increase in Pb concentrations in the Cardiff central outfall that occurred between 1994 and 1997 (figure 33).

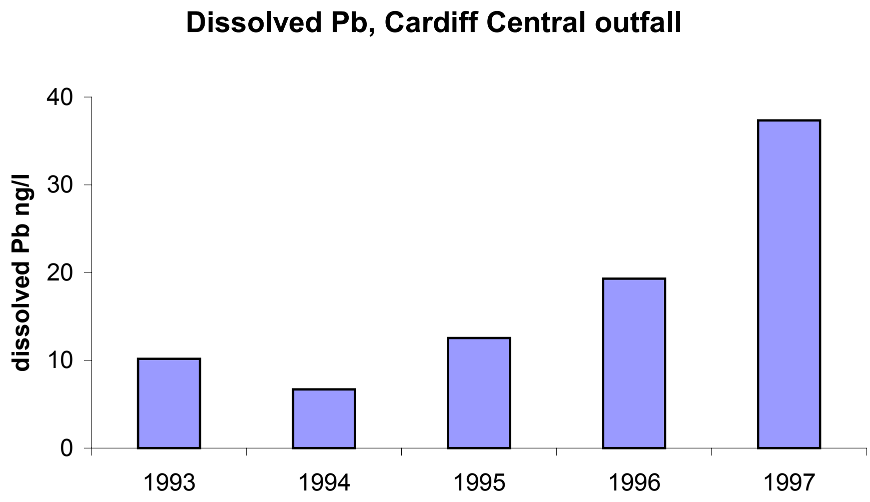


Figure 33. Temporal trend in annual average dissolved Pb concentration, Cardiff Central outfall.

Zinc

Total and dissolved zinc concentrations in riverine sources to the pSAC are shown in figures 34 and 35 (where 'reliable' data are available). The EQS (for total Zn) in fresh waters (suitable for salmonid fish), ranges from 8 to 120 µg l^{-1} depending on hardness (likely value for rivers in figure 34 are towards the upper end of the range – i.e. 75µg l^{-1}) . Summary statistics from the EA database for 2001, shown in figure 34, indicate that the Usk, Ebbw Fawr, Severn, and Avon are likely to comply with this standard.

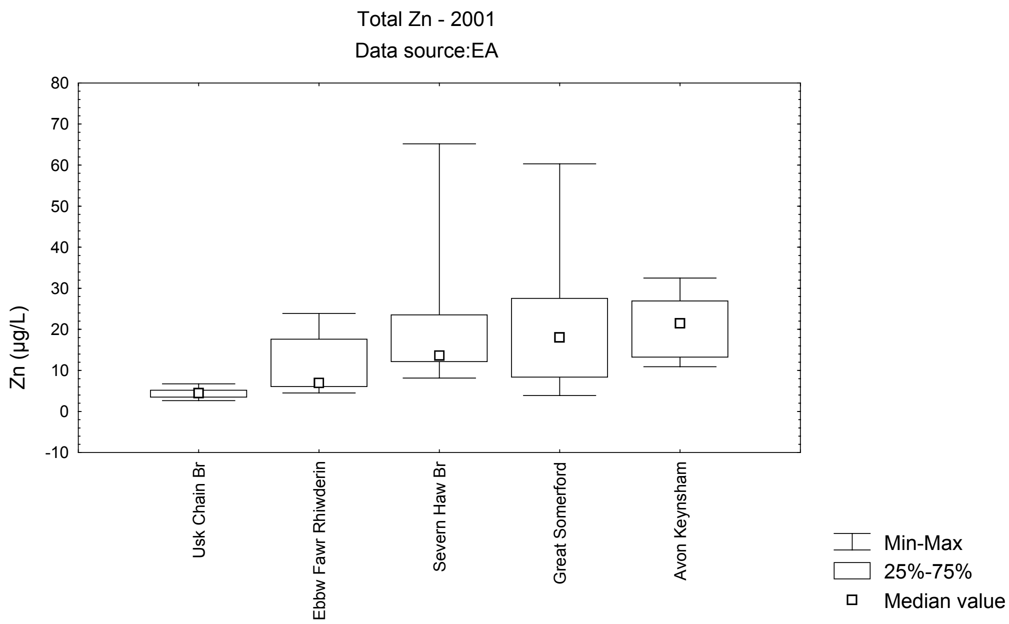


Figure 34. Concentrations of total Zn (2001 annual averages, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data source EA.

The slightly more extensive data for dissolved Zn, shown in figure 35 suggests comparable levels of Zn are likely to be found in other catchments around the Severn Estuary. Highest mean values (~ 15µg l^{-1}) were for the River Tone at Knapp Bridge.

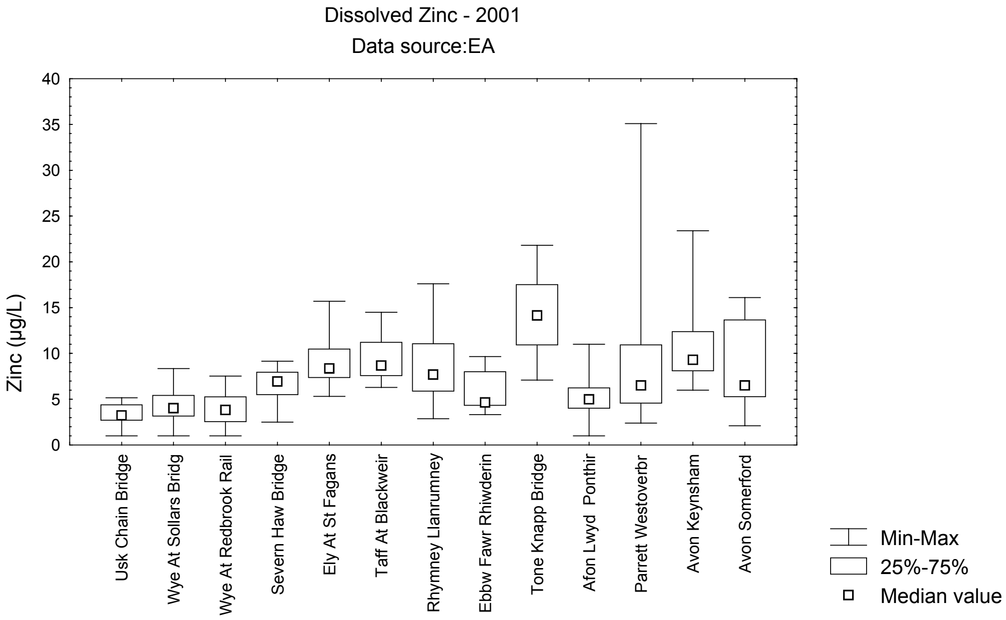


Figure 35. Trends in concentrations of dissolved Zn (2001 annual average, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding the Severn Estuary. Data source EA.

There are indications in reductions of annual average dissolved Zn values for Ebbw Fawr, and in the lower Wye and Severn .

The pattern of Zn in estuarine water, plotted on a regional basis is shown in figure 36. Average regional values are invariably below the current EQS (40µg l^{-1}) across the Estuary (though a proposed[1] revision to 10µg l^{-1} would place many values close to the standard). Extreme Zn values exceeded the EQS in several regions as indicated in figure 36, most notably in the Severn east region, near Avonmouth. Not surprisingly, at the KWR survey point near to the Britannia Zinc smelter[2], values have been recorded up to 15,000µg l^{-1} (all top ten values, all upwards of 900µg l^{-1}, are for this site). Mean reported values here were in excess of 5000µg l^{-1} prior to 1994. More extensive data for the KWR site at high water, up to 2001, puts the average at just over 100µg l^{-1}. Means for other sites labelled in figure 36 are below the EQS, albeit by a small margin (Old Cardiff outfall 11.3µg l^{-1}, W-S-M B.Rock 18µg l^{-1}, Nash Outfall 22.5µg l^{-1}) though few data are available for the last five years.

[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.

[2] Britannia Zinc will cease operations in March 2003. The discharge will cease in due course

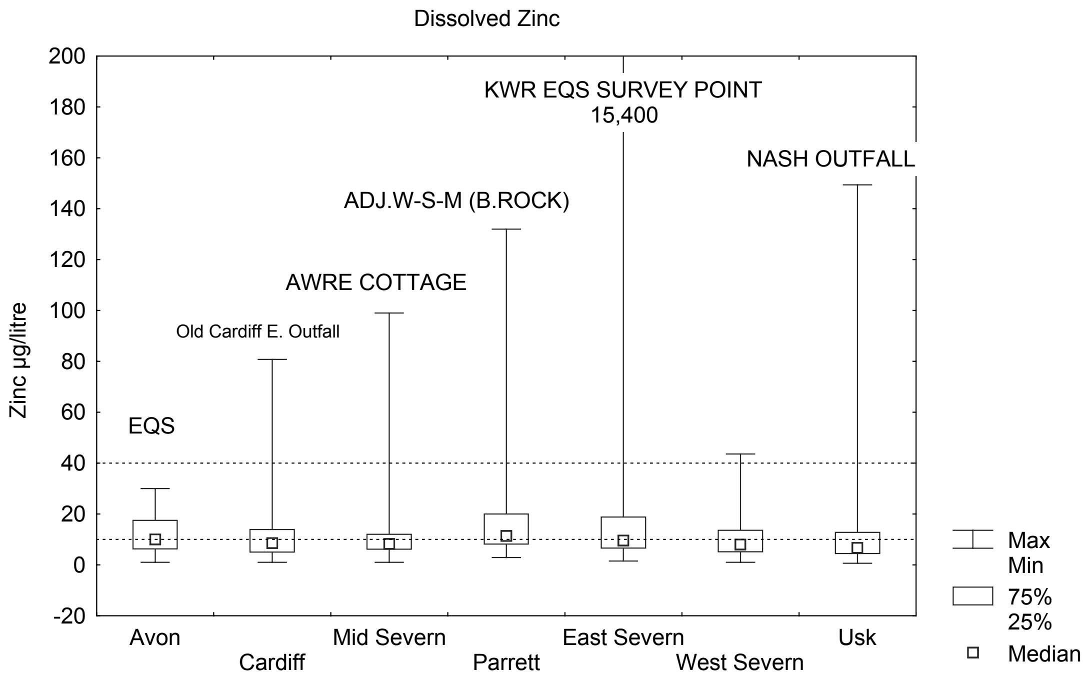
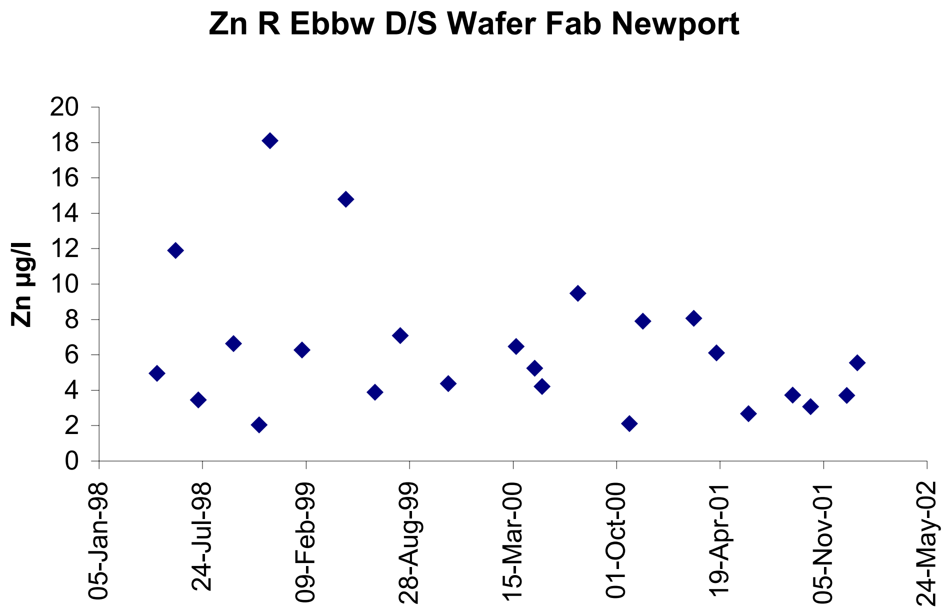


Figure 36. Concentrations of dissolved Zn (µg l^{-1}) in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA. Upper horizontal line represents current EQS of 40 µg l^{-1}, lower line proposed revised EQS of 10µg l^{-1} (Hunt and Hedgecott, 1992)

Data at the majority of sites do not establish the presence of long-term trends in dissolved Zn. There may be one or two exceptions such as in the R. Ebbw at Newport and in the Severn off the ICI plant, where reductions over time are indicated (figure 37).

Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in mean Zn concentration at individual sites (~10) in the period between 1992 –1995, whereas only one increase was observed. All examined records of sites complied with the EQS (1980-1997).

Despite indications of lower levels and compliance with the EQS in midstream samples, the major inputs, particularly in the Avonmouth area, should still be monitored carefully with regard to Zn.



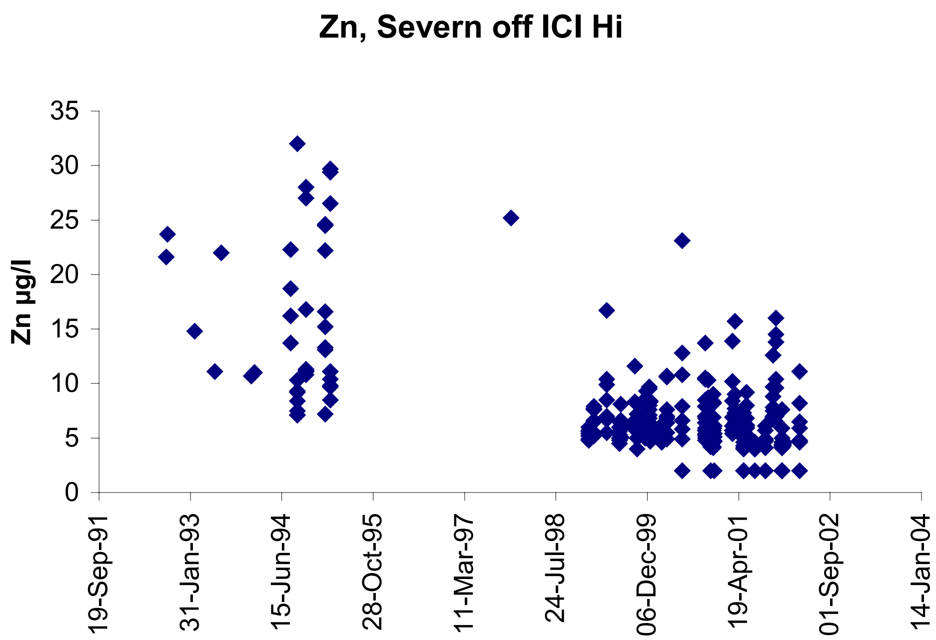


Figure 37. Temporal trend data for dissolved Zn, R. Ebbw at Newport (top) and in the Severn off the ICI plant (bottom)

Available summary statistics for dissolved Zn in trade and sewage discharges are shown in figure 38. Some of these concentrations exceed EQS values and could be of significance locally (particularly downstream of the BSC Whiteheads plant on the Usk). However, the coverage of sites does not adequately address all the sources responsible for Zn levels in estuarine waters. More comprehensive geographical and spatial data are needed, particularly in the Avonmouth area.

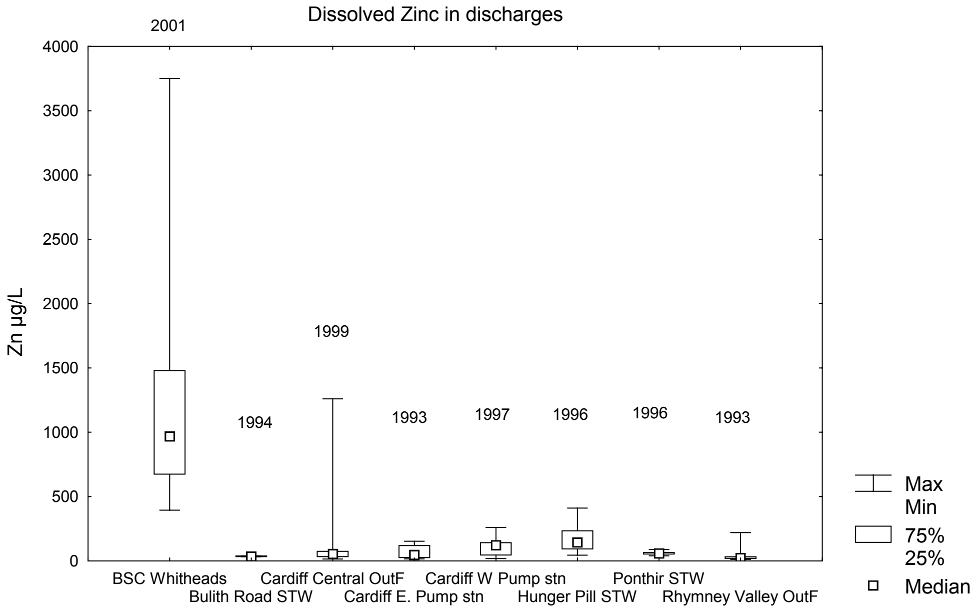


Figure 38. Dissolved Zn in some of the discharges to the Severn Estuary (summary statistics for the latest year in which data supplied by EA)

Mercury

Total mercury concentrations in riverine sources to the pSAC are shown in figure 39 (where 'reliable' data are available). The EQS for Hg in all fresh waters, based on total metal, is 1µg l^{-1}. Summary statistics shown in figure 39 indicate that the Usk, Ebbw Fawr, Severn, and Avon comply with the Hg standard by a considerable margin (almost two orders of magnitude). Highest values appear to occur in the Severn and Avon. Data are insufficient to assess temporal trends.

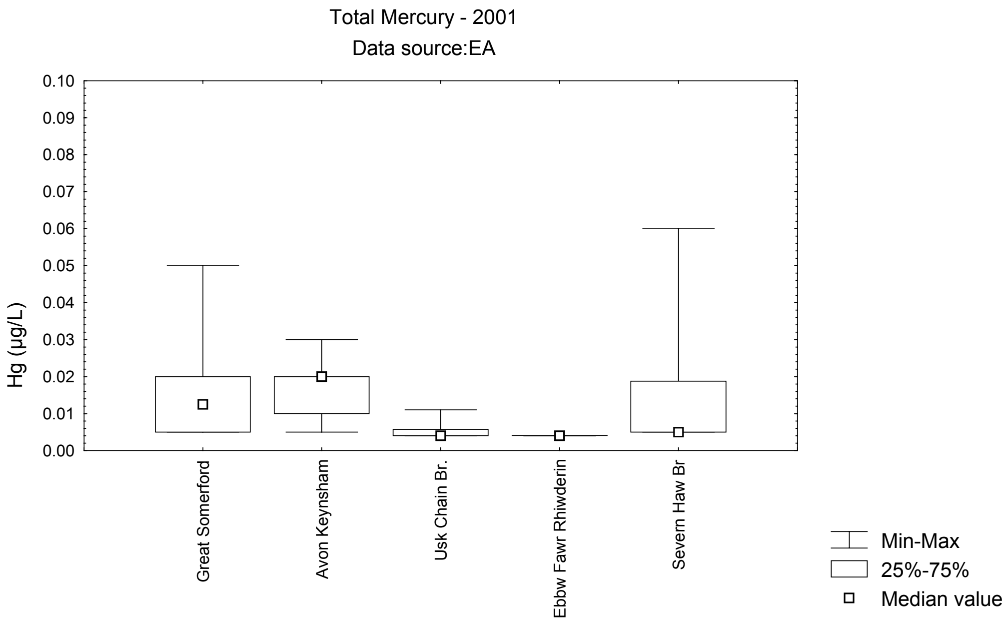


Figure 39. Concentrations of total Hg (2001 annual averages, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data source EA.

The pattern of Hg in estuarine water, plotted on a regional basis is shown in (figure 40). Average regional values are invariably below the EQS (0.3 µg l^{-1}). Extreme Hg values have in the past occasionally exceeded the EQS - downstream of the Monsanto outfall in the Usk, off the Severn Holesmouth monitoring station, High Water, (as indicated in figure 40), and also in one sample off an ICI pipe into the Severn. However, average values at these sites are approximately one order of magnitude below the EQS.

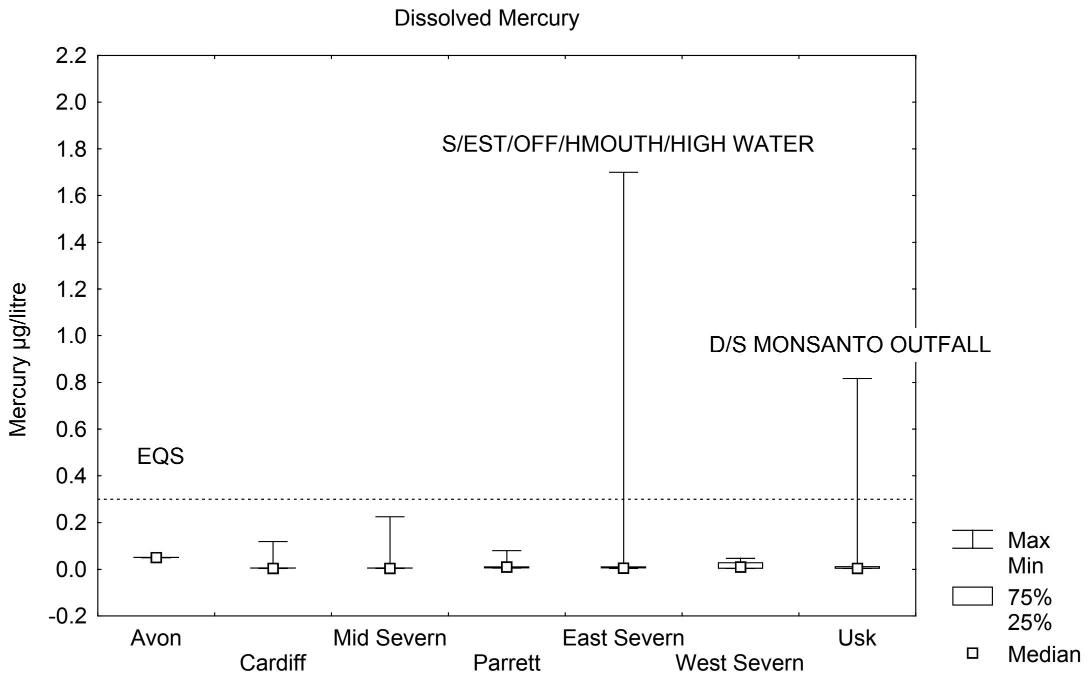


Figure 40. Concentrations of dissolved Hg (µg l^{-1}) in estuarine waters for different regions of Severn Estuary. Summary for the last decade. Data source EA.

Long-term temporal trends for Hg are not evident in the data.

Examining data for individual sites in the helicopter surveys, Ellis (2002) indicated a number of decreases in means in dissolved Hg concentration at individual sites in the period between 1990 –1992. However, these were largely offset by increases both prior to this, and afterwards (up to 1995), masking any overall trend. Virtually all midstream sites examined by Ellis complied with the EQS. However in the last survey examined, in 1997, sites upstream of Newnham church, in the inner Severn, exceeded the 0.3 µg l^{-1} annual average. The cause of this should be examined.

Statistics for Hg in trade and sewage discharges could only be found for two sites (Rhodia and ICI at Avonmouth) and data for both were below EQS values. More comprehensive geographical and spatial data are needed to assess sources to the Estuary.

7.1.2 Organotins

There are very few reliable TBT data for HMP sites or equivalent riverine sources to the pSAC, and the majority represent detection limit values, occasionally with 'detectable' concentrations as depicted in figure 41. In 2001, concentrations for the Wye, Ely, Severn, Parrett and Ebbw Fawr were below the freshwater standard (maximum concentration 0.02 µg l^{-1}) though only by a factor of two at some of these sites. Elsewhere a maximum value of 0.084µg l^{-1} TBT, above the EQS, was determined in one sample from the Avon at Keynsham though this appears to be uncharacteristically high as median value for the site was 0.002µgl^{-1}.

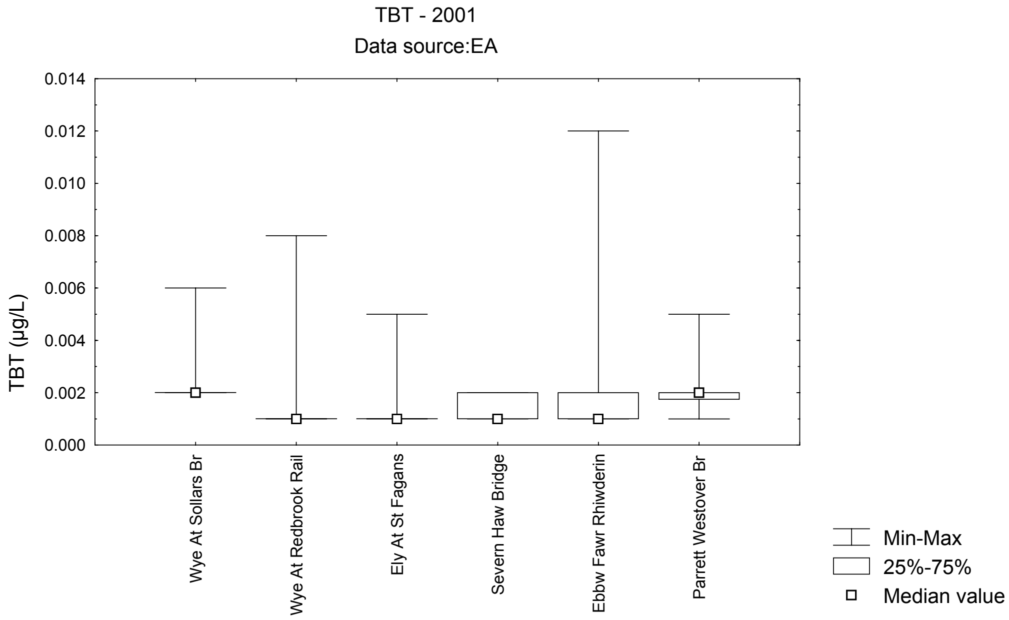


Figure 41. Concentrations of TBT (2001, µg l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data source EA.

Average concentrations in the Rivers Tone (Knapp Bridge) and Parrett (Westover Bridge) suggest that monitored sources at these sites have been declining in the last decade (figure 42; note values are in ng l^{-1}). At the start of this surveillance in 1994 maximum concentrations occasionally exceeded the EQS by up to two fold. No other trends in TBT in freshwater are discernable.

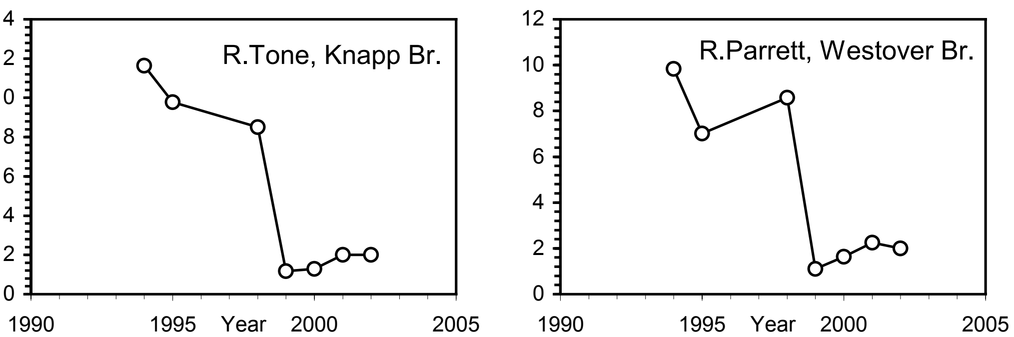


Figure 42. Temporal trends in annual average TBT concentrations (ng l^{-1}) at freshwater sites in the Rivers Tone and Parrett (Data source: EA)

Attempts at triphenyltin analysis (TPT) are recorded for the Tone, Parrett and Severn where values appear to be close to a detection limit of 0.012 µg l^{-1}, compared with the EQS of 0.02 µg l^{-1}.

It is not possible to establish accurately the pattern of TBT in estuarine water, since there only a limited amount of data, most close to detection limits, which may give a false picture. Also most monitoring concerns the western bank of the Estuary with few results for the eastern Severn. Summaries of available data for three sites in the Usk, West Severn and Cardiff area data are plotted in figure 43. Mean values for data collected between 1996 and 2002 ranged from 0.003-0.005 µg l^{-1} – just above the EQS (0.002 µg l^{-1}). Maximum values - on which the TBT standard is based – were exceeded by almost 40-fold downstream of the old Cardiff east outfall and the transporter bridge on the Usk and by 10-fold downstream of the St Regis outfall (near the western end of the new Severn road bridge).

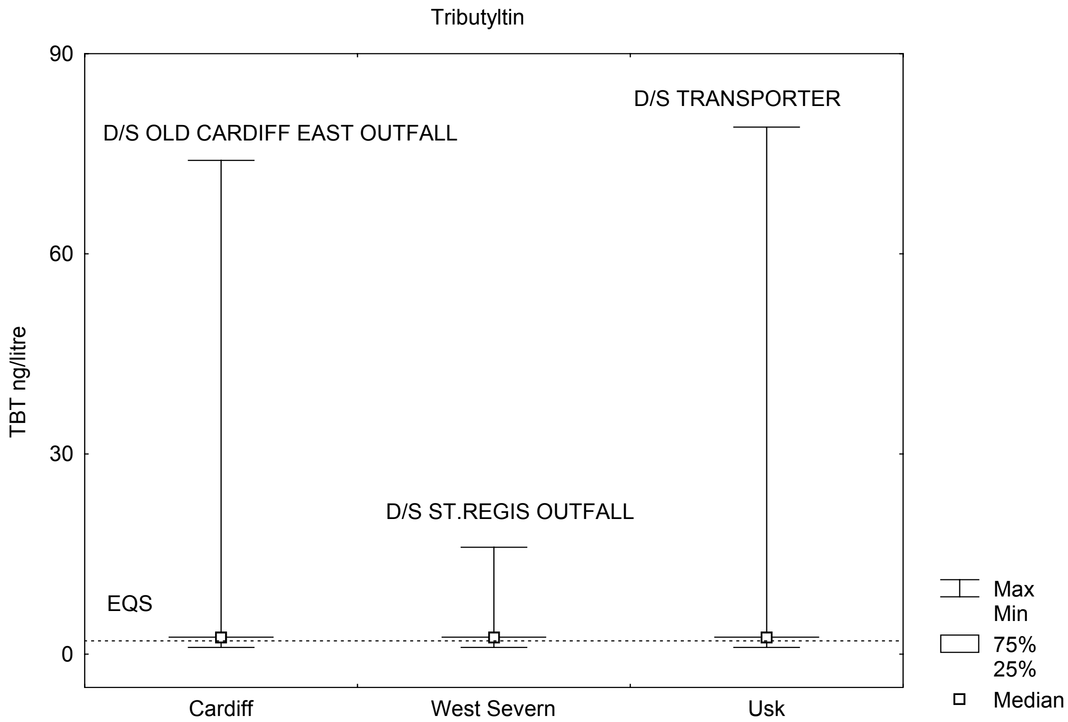


Figure 43. TBT in tidal waters for three sites in South Wales during the last decade. Data source:EA

Entries for the highest ten TBT values in the EA database are shown in table 31.

Table 31. Top ten highest individual TBT values in EA database for the Severn Estuary over the last decade

Sampling point name	NGR	Estuary	date	TBT ng l^{-1}
USK EST 1000M D/S TRANSPORTER	ST3200085700	usk	3/22/00	79
D/S OLD CARDIFF EAST OUTFALL	ST2273274940	cardiff	3/22/00	74
USK ESTUARY U/S CAERLEON RD BG	ST3412090260	usk	9/11/96	50
D/S WESTERN VALLEY OUTFALL	ST2951880051	usk	11/5/97	21
USK ESTUARY U/S CAERLEON RD BG	ST3412090260	usk	2/3/99	20
D/S MONSANTO OUTFALL	ST3456580051	usk	3/22/00	20
D/S CARDIFF CENTRAL OUTFALL	ST2091075500	cardiff	2/24/98	19
D/S RHYMNEY VALLEY OUTFALL	ST2584477990	cardiff	3/22/00	18
D/S WESTERN VALLEY OUTFALL	ST2951880051	usk	4/26/00	17
USK EST 1000M D/S TRANSPORTER	ST3200085700	usk	4/27/99	16

Although many of the entries in the database appear to be below detection limit, on the basis of table 31 and figure 43, more accurate monitoring of TBT is warranted in the Severn and its tributaries to establish sources and trends. The current data base contains some 300 entries for discharges of which only one-third are above detection limits. For guidance purposes only, these have been included in the summary statistics shown in figure 44. These merely indicate the potential for input from both STW and certain trade discharges and should not be seen as definitive. In particular, sites marked with an asterisk have few data. Elsewhere, there may be a few high values scattered amongst overall lower backgrounds. Because of the substantial amount of noise in the data, identification of temporal trends is difficult. There is an indication that the incidence of high values have been reduced since 1998 in the Monsanto and St Regis outfalls but the opposite may be true for CEDPS discharge, Cardiff (figure 45).

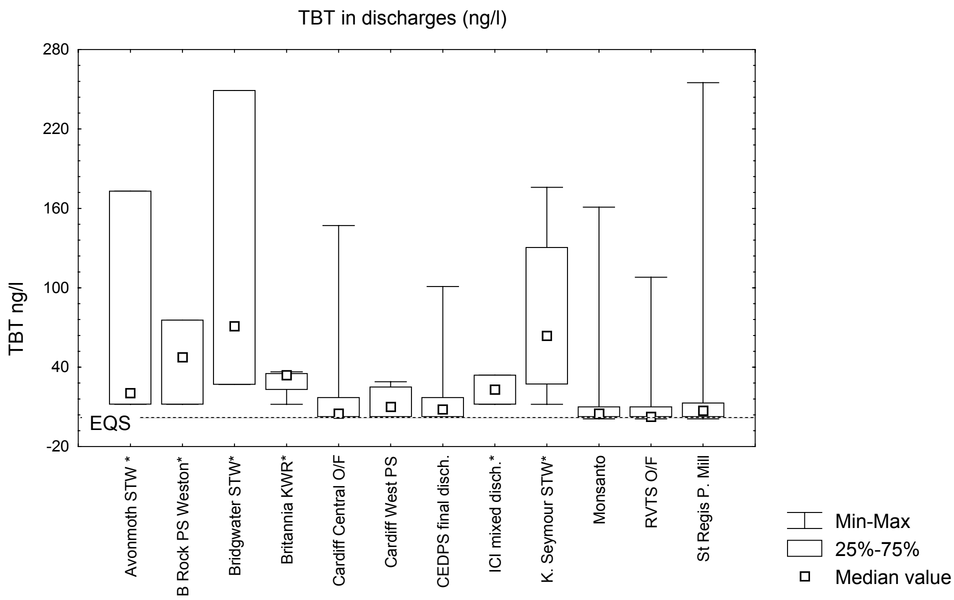
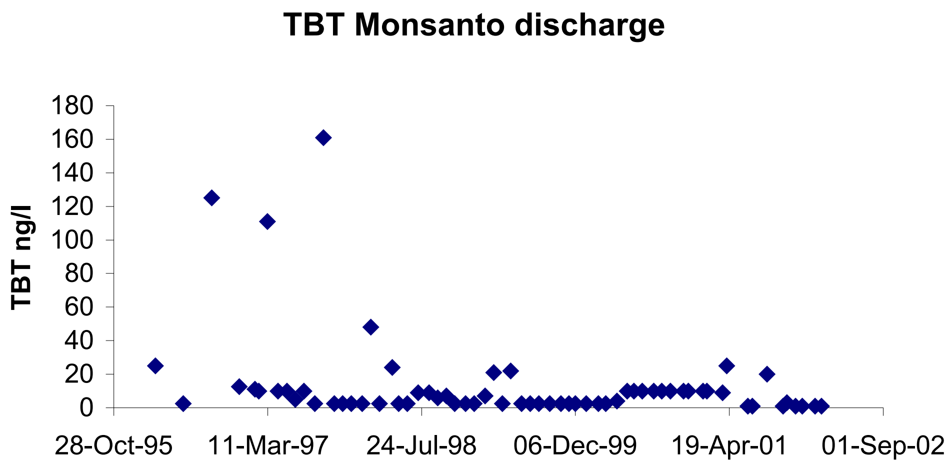
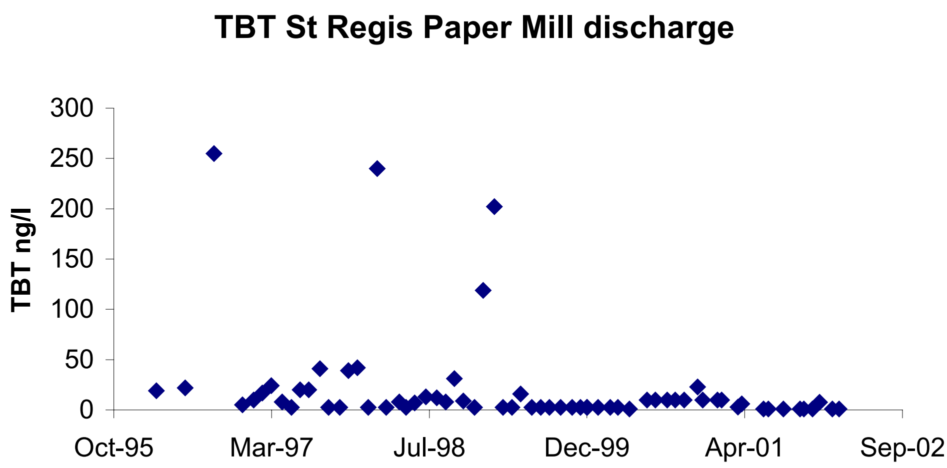


Figure 44. TBT in discharges. Summary of EA data, including half-DL values





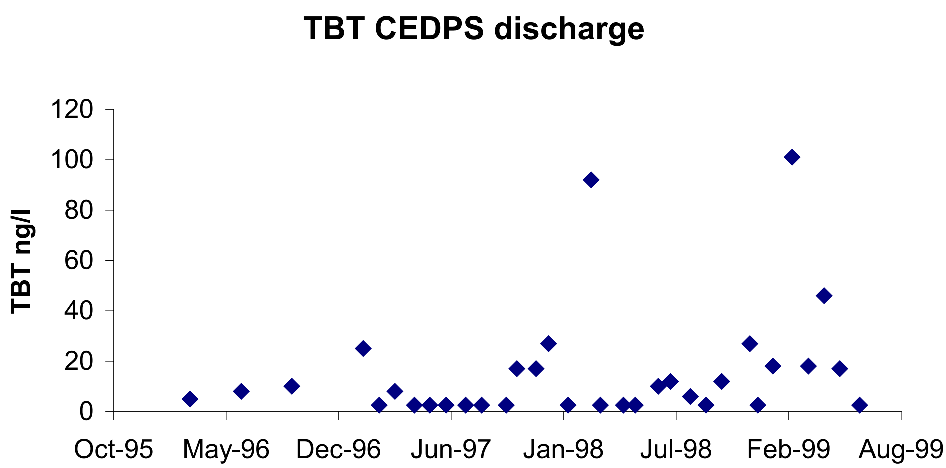


Figure 45. Temporal trends for TBT (ng l^{-1}) in discharges at selected sites.

7.1.3 Pesticides and Herbicides

Data for pesticides and herbicides in tidal waters of the pSAC are generally for the period early 1990s. Where they occur, (half) detection limits have been used in the derivation of summary statistics.

HCH (total); Gamma-HCH (lindane)

There are few results in the EA database for total hexachlorocyclohexanes (HCH), the majority of results are for gamma-HCH (lindane) used previously in insecticides and wood preservatives. The wide variation in detection limits is problematic in data interpretation (unless they are significantly below the EQS, detection limit data have not been considered here).

Summary statistics for concentrations of lindane in the rivers entering the SAC (in 2001) are shown in figure 46. Annual median values, were less than 2ng l^{-1} – below the EQS of 20ng l^{-1} - at all HMP sampling sites (or equivalent), though at Westover Bridge on the Parrett and Redbrook Bridge on the Wye individual samples may contain significantly higher concentrations. The use of lindane is currently being phased out in Europe following an EU decision in 2000 to ban it, however its use on food crops imported from other counties could result in lindane residues in sewage effluent. There is also the possibility of continued illegal usage.

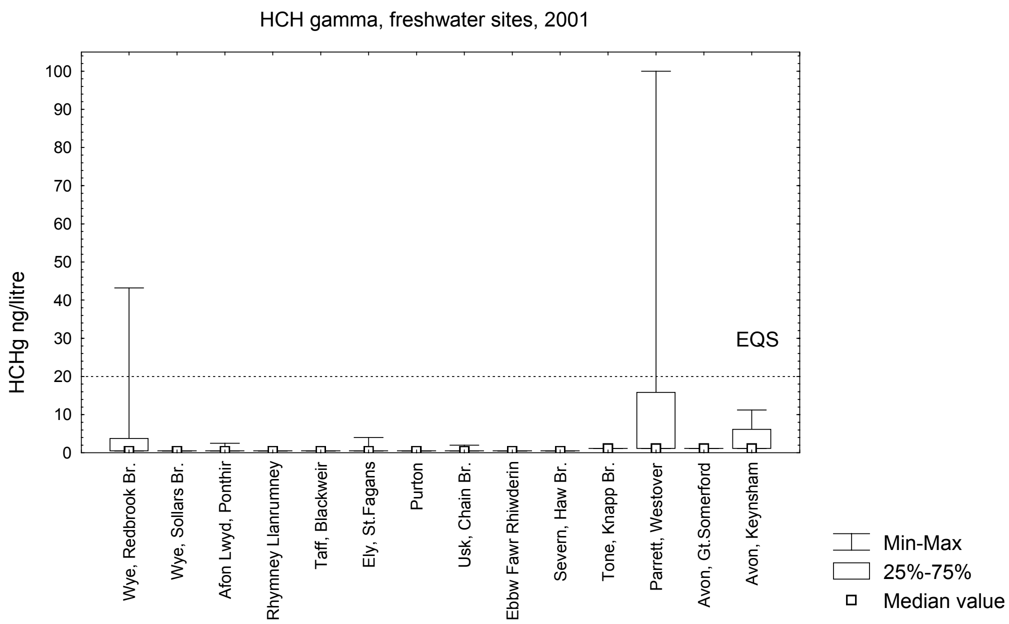


Figure 46. Concentrations of lindane (2001, ng l^{-1}) at harmonised monitoring points (or similar) in freshwater catchments feeding Severn Estuary. Data source EA.

Trends in lindane concentrations in effluent/discharge samples are particularly difficult to interpret and much of the data in the EA data-base are almost 10 years old. Table 32 is included only as an indication of where levels in discharges have exceeded the EQS in the past, and show that trade and domestic wastes are potential sources. However, there is broad evidence of decreasing usage as typified for Wellington STW in figure 47.

Table 32. Lindane concentrations in discharges (most recent available annual averages) which exceed 20ng l^{-1}.

EFFLUENT SAMPLE	Year	HCHg ng l^{-1}	Estuary
NASH STW - FINAL EFFLUENT	1996	396	Severn
BRIDGWATER STW	1996	210	Parrett
ST REGIS PAPER MILL	1996	175	Severn
CEDPS FINAL EFFLUENT DISCHARGE	1998	128	Severn
PONTHIR STW FINAL EFFLUENT	1995	124	Severn
CARDIFF CENTRAL OUTFALL	1994	116	Severn
HEREFORD ROTHERWAS STW.	1999	43	Severn
ISC ACID PLANT RHINE	1994	32	Severn
COLLEY LANE PS FINAL	1991	29	Parrett
WESTON SUPER MARE STW (UV)	1999	29	Severn

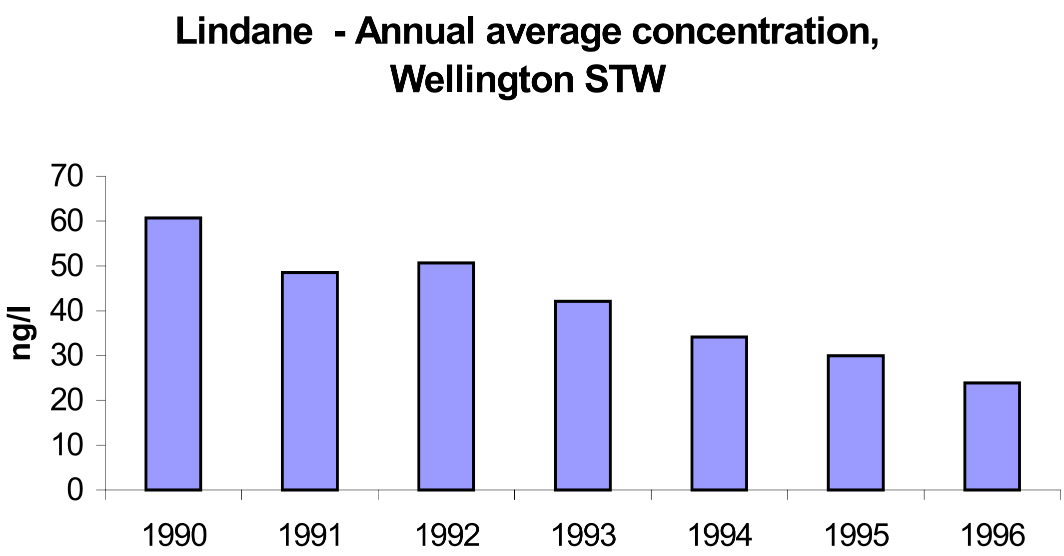


Figure 47. Temporal trends in concentrations of lindane (annual mean, ng l^{-1}) Wellington STW final effluent.

Overall distributions of lindane in tidal waters are shown in figure 48, as regional statistics. No viable data are available for the Cardiff or Severn West areas. In other regions median values are below the EQS. Highest concentrations are restricted to one or two extreme values (outliers?) at Holesmouth and KWR EQS survey points almost 10 years ago, elevating median values to 35 and 31ng l^{-1}. These sites are perhaps in need of re-evaluation. The few available data for total HCH, (Avon and Mid Severn) all fall below 10ng l^{-1}.

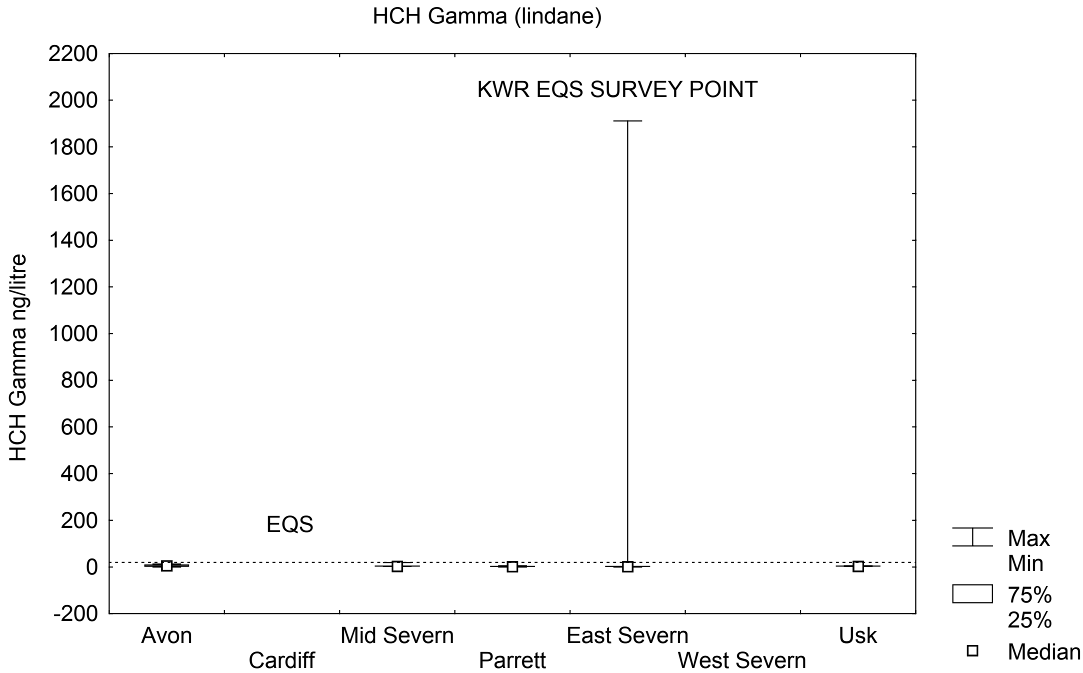
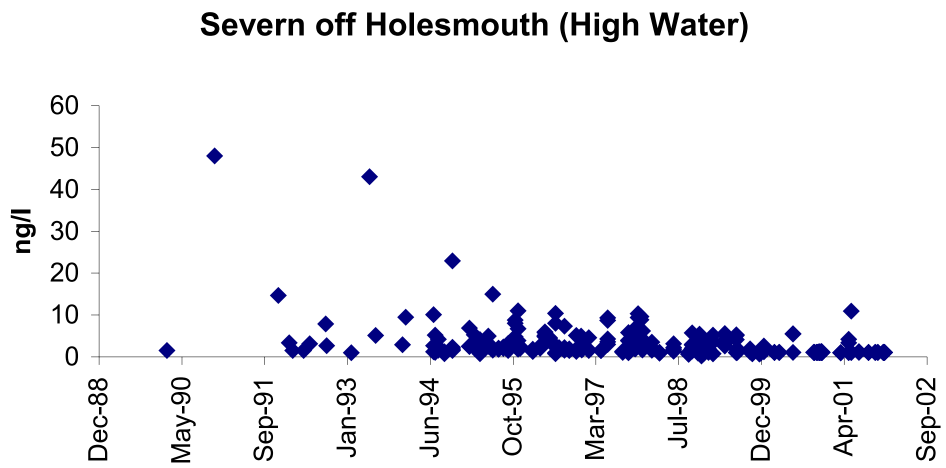


Figure 48. Lindane (γ-HCH) in estuarine waters for different regions of the Severn Estuary. Summary for the last decade. Data source EA.

There is only limited information on which to establish temporal trends in estuarine waters. One site off Holesmouth indicates that the incidence of high values has diminished in recent years (figure 49). Lower down the Estuary, at Kingston Seymour, background levels have remained relatively constant (~2ng l^{-1}) apart from some enrichment in summer 1996.



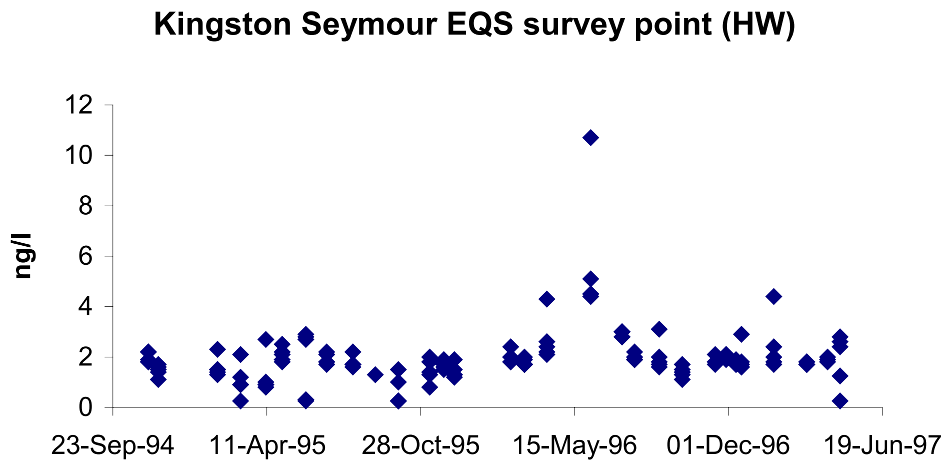


Figure 49. Temporal trend data for lindane, off Holesmouth (top) and in the lower Severn at Kingston Seymour (bottom)

DDT

Data for para, para-DDT (ppDDT) in freshwaters entering the SAC at HMPs or equivalent (see figure 11 for sites) were less than 1ng l^{-1} (expressed as annual average values for 2001) i.e. an order of magnitude below the EQS of 10ng l^{-1}. On this evidence, riverine sources of DDT do not appear to be a significant issue.

For tidal waters, summary statistics are presented (figure 50), for entries between 1990 and 1997 (the latest available samples). The majority of data were entered as <detection limits (usually 0.4 ng l^{-1} -well below the EQS). Median values (up to ~2ng l^{-1}) are thus invariably below the EQS implying little threat to biota. Sporadic high values have been recorded at KWR EQS Survey point (upto 21ng l^{-1}) and adjacent to Weston-Super-Mare Black Rock STW (7ng l^{-1}), but these do not significantly raise median values above levels found across the Estuary as a whole.

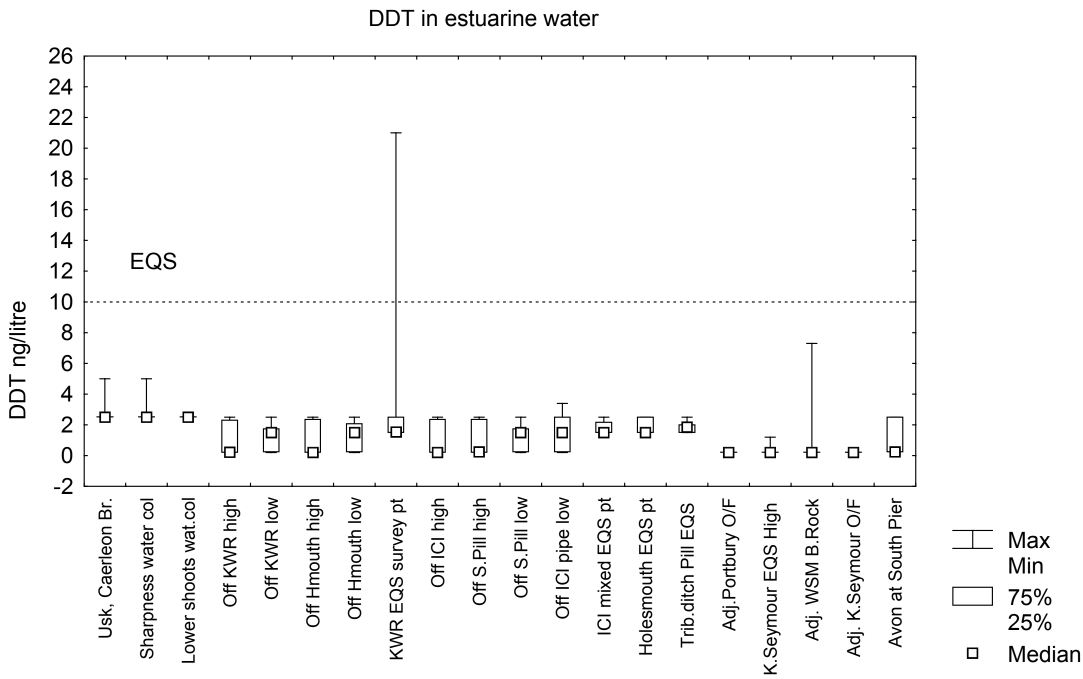


Figure 50. ppDDT in estuarine water samples (available data, 1990-1997)

From almost 1000 observations of DDT in discharges all but 22 are below detection limits which range from 20,000-0.4 ng l^{-1} and therefore are beyond statistical analysis. For information, those above the EQS for freshwater are listed in table 33. High values are all trade wastes prior to 1997. Inspection of the data suggests these are occasional incidents rather than sustained inputs and probably represent little threat other than, perhaps, in the vicinity of the discharge. However, elevated DDT levels at Black Rock PS Weston discharge do appear to coincide with those in adjacent tidal water samples (see above).

Table 33. ppDDT in discharge samples > 10ng l^{-1}

Sample point name	River/Estuary	NGR	Date	ppDDTng l^{-1}
ROF DITCH	Huntspill	ST3060044700	10/16/92	44.2
ROF DITCH	Huntspill	ST3060044700	11/02/92	34.2
BLACK ROCK PS WESTON	Severn	ST3250058500	07/04/97	23.3
ROF DITCH	Huntspill	ST3060044700	03/17/94	21.8
ROF DITCH	Huntspill	ST3060044700	11/16/94	21.4
UCB MAIN SEWER	Parrett	ST3094738641	11/16/93	19.6
ALB/WILSON COOL WAT.	Severn	ST4775077300	02/21/92	19.0
ROF DITCH	Huntspill	ST3060044700	01/20/95	13.8
ALB/WILSON COOL WAT.	Severn	ST4775077300	02/20/92	13.2
UCB MAIN SEWER	Parrett	ST3094738641	05/19/97	12.2

Dieldrin

In 2001, estimated concentrations of dieldrin in freshwaters entering the SAC at HMPs or equivalent (see figure 11 for sites) were less than 2ng l^{-1} (expressed as annual average values for 2001) i.e. lower than the EQS of 10ng l^{-1} by a factor of 5. Many observations were below detection limits. On this evidence, riverine sources of dieldrin do not appear to be a significant issue for the SAC.

For tidal waters, elevated concentrations of dieldrin have been recorded in and around the vicinity of discharges in the Avonmouth area, e.g the KWR (upto 3466 ng l^{-1} in 1991), Holesmouth and ICI mixed survey points. Again however these appear to be isolated cases and median values are below the EQS at all the monitored sites (see summary figure 51). The most recent values for dieldrin in tidal waters away from these sources were generally around the limit of detection (1ng l^{-1} or less).

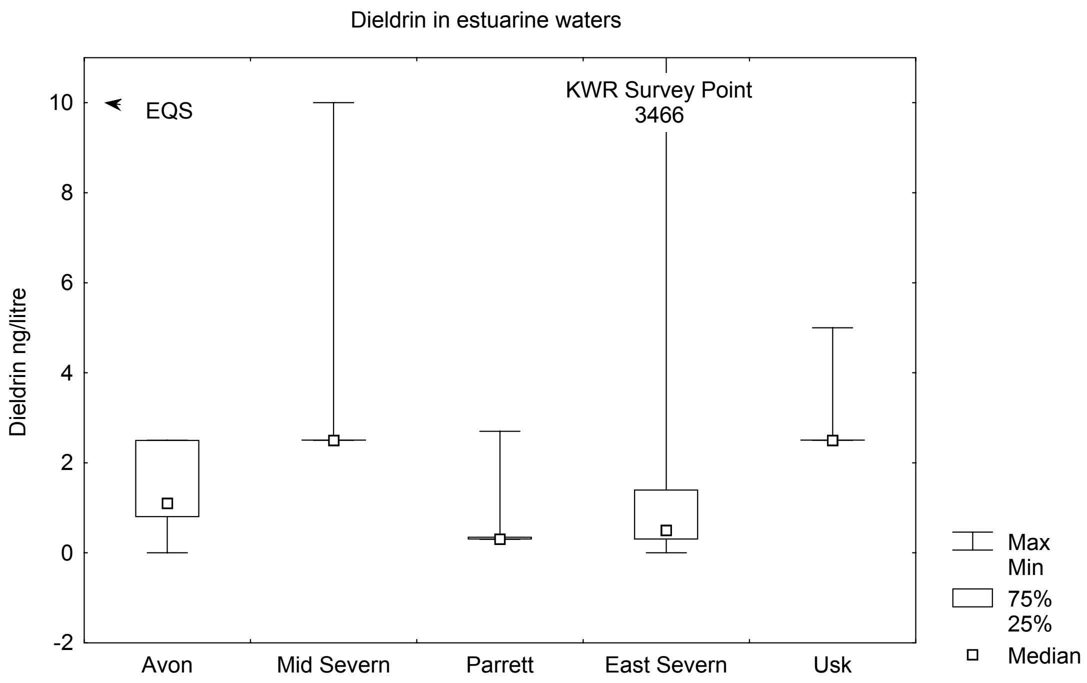


Figure 51. Dieldrin summary statistics for regional data, Severn Estuary

From more than 1500 observations of dieldrin in discharges only 12% were above detection limits which range from 50,000-0.4 ng l^{-1}. In view of this variation only values above DLs (detection limits) were examined further. For information, only average values which lie above the EQS for freshwater are listed in table 34. Inspection of the data suggests these are sustained inputs but probably represent little threat other than, perhaps, in the vicinity of the discharge. It is possible they could be the source of occasional elevated values in nearby estuarine water.

Table 34. Dieldrin in discharge samples > 10ng l^{-1} (means, all data above detection limit).

Discharge name	River/Estuary	NGR	last sample	Dieldrin ng l^{-1}
BLACK ROCK PS WESTON	Severn	ST3250058500	1998	14
COURTLEIGH SECURITIES LTD	Tone	ST1271421893	1996	29
ISC ACID PLANT RHINE	Severn (Avonmouth)	ST5270079400	1994	30
RHODIA CONSUMER SPECIALTIES	Severn (Avonmouth)	ST5240580535	1999	20

Aldrin

In 2001, estimated concentrations of aldrin in freshwaters entering the SAC at HMPs or equivalent (see figure 11 for sites) were less than 2ng l^{-1} (expressed as annual average values for 2001) i.e. lower than the EQS of 10ng l^{-1} by a factor of 5. Many observations were below detection limits. On this evidence, riverine sources of aldrin do not appear to be a significant issue for the SAC.

For estuarine waters the distribution pattern is similar to dieldrin – with concentrations ranges dominated by a small number of extreme values near Avonmouth discharges in the early 1990s and one sample in mid Severn (Lower Shoots); the most recent data for these sites (mid 1990s) indicate almost background levels. Elsewhere in the Estuary the vast majority of samples are below DLs (figure 52).

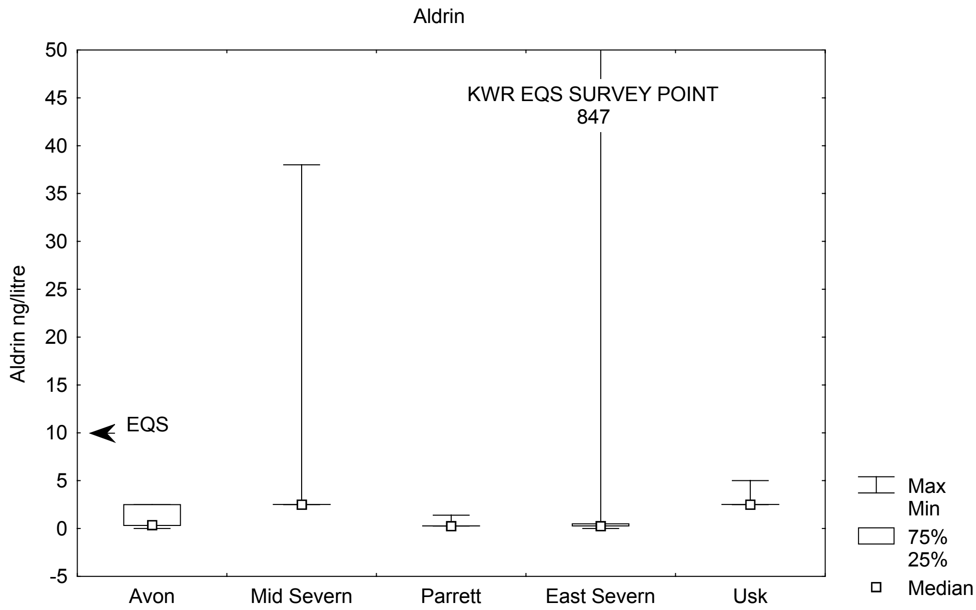


Figure 52. Aldrin. Summary statistics for regional data, Severn Estuary

From more than 1200 observations of aldrin in discharges only 19 were above detection limits which range from 50,000-0.25 ng l^{-1}. In view of this variation only values above DLs were examined further. For information, data are presented where average values approach or exceed the EQS for freshwater (table 35). It is possible these discharges could be the source of occasional elevated values in nearby estuarine water, but unlikely that they consistently affect water quality over a large area.

Table 35. Aldrin in discharge samples (means, all data above detection limit).

Discharge	River/Estuary	last sampled	Aldrin ng l^{-1}
BLACK ROCK PS WESTON	Severn	1993	31
COURTLEIGH SECURITIES LTD	Tone	1991	10
ISC ACID PLANT RHINE	Severn (Avonmouth)	1991	13
RHODIA CONSUMER SPECIALTIES	Severn (Avonmouth)	1991	10
ROF DITCH	Severn (Huntspill)	1997	8

Endrin

The EQS for endrin is 5ng l^{-1} in fresh and saline waters. In 2001, estimated concentrations of endrin in freshwaters entering the SAC at HMPs or equivalent (see figure 11 for sites) were less than 2ng l^{-1} (expressed as annual average values for 2001) i.e. lower than the EQS by at least 2-3 fold . Many observations were below detection limits. On this evidence, riverine sources of endrin do not appear to be a significant issue for the SAC.

Virtually all entries for tidal waters are recorded as below DL in the EA database and since this varies over an order of magnitude and a significant number or at or above the EQS, little insight into distribution or potential threat can be gleaned.

Highest individual values were at Avonmouth in the Severn (122 ng l^{-1} at KWS EQS survey point (in 1991) and 9.5 ng l^{-1} off Holesmouth (May 2001)). Using half DL values (the majority of sites) suggests no other possible 'hotspots' elsewhere within the site, though it is difficult to comment on compliance because of analytical uncertainty.

Only 14 from a total of 1300 observations of endrin in discharges were above detection limits which range from 50,000-0.3 ng l^{-1}. Half of these positive recordings were for Rhodia Consumer Specialties (up to 277ng l^{-1}) and ISC acid plant rhine (97 ng l^{-1}) implying that these discharges could be the source of occasional elevated values in nearby estuarine water. These sporadic elevated values were usually from 1995 or before however and are outnumbered by a values below detection limits. It is therefore unlikely that discharges consistently affect water quality over a large area.

Endosulphan (A and B)

The EQS for Endosulphan is 3ng l^{-1}. In 2001, reported concentrations in freshwaters entering the SAC at HMPs or equivalent (see figure11 for sites) were largely a function of detection limits and ranged from 0.5-2.5ng (for each isomer, expressed as annual averages). On this evidence, riverine sources of endosulphan do not appear to be a significant issue for the SAC.

Similarly, virtually all entries for tidal waters in the EA database are recorded as below DL. Since this varies over an order of magnitude and a significant number of are close the EQS, it is not possible to describe distributions or compliance meaningfully. Incorporating half DL estimates indicates that median regional values for the estuarine regions are below 1ng l^{-1} for the Avon, Parrett and East Severn, and of the order of 2.5ng l^{-1} for the Usk and mid Severn samples, though these reflect analytical rather than geographical variation. It seems unlikely that endosulphan isomers in estuarine waters are influencing the pSAC.

Only five from a total of 700 observations of Endosulphan A (and four for endosulphan B) in discharges were above detection limits (which range from 2800 – 0.6ng l^{-1}). Of the positive recordings for endosulphan A only those for Rhodia Consumer Specialties (206 ng l^{-1}), the UCB main sewer on the Parrett (20-25 ng l^{-1}) and ROF ditch (17ng l^{-1}) have exceeded the EQS, and then only on one or two sampling dates between 1995 and 1997. The sites at which Endosulphan B have been detected are different (Britannia Zinc Ltd KWR –7.4 ng l^{-1}; ISC acid plant rhyne – 18 ng l^{-1}; ROF ditch – 23ng l^{-1}; Cardiff central outfall –1140 ng l^{-1}) though they also represent single events between 1994-1996. These sporadic elevated values are outnumbered by values below detection limits. It is therefore unlikely that discharges consistently affect water quality other than on a very local level.

Atrazine and Simazine

Atrazine and Simazine are *s*-triazine herbicides and are on the UK red list of toxic compounds with a combined EQS of 2µg l^{-1} (annual average) or 10 µg l^{-1} (maximum allowable concentration. The total sum of the two triazines have been calculated for the HMP sites monitored in table 36. Though atrazine and simazine are present in these systems exceedences of the EQS did not occur. No data could be found for tidal waters.

Table 36. Atrazine and simazine: annual average and maximum concentrations (ng l^{-1}) at HMP sites, 2001.

	Simazine ng l^{-1}	Atrazine ng l^{-1}	Simazine + atrazine ng l^{-1}
		Annual averages 2001	
R PARRETT WESTOVERBR	49	23	72
R TONE KNAPP BRIDGE	14	15	29
AVON KEYNSHAM M	42	174	216
		Maximum concentrations 2001	
R PARRETT WESTOVERBR	291	50	341
R TONE KNAPP BRIDGE	26	23	49
AVON KEYNSHAM M	127	1050	1177

In discharges, only 22 and 26% of the records for atrazine and simazine are above detection limits (which range from 318 –10ng l^{-1}). Table 37 summarizes data for these positive recordings. In none of the discharges do concentrations appear likely to exceed the EQS and therefore these herbicides would not (on their own) be expected to exert significant effects in the environment.

Table 37. Atrazine and Simazine herbicides in discharges

Site	Estuary	Date	Atrazine ng l^{-1}	Simazine ng l^{-1}	Atrazine + simazine ng l^{-1}
AVONMOUTH STW (TIDAL P/S)	Severn	1996-98	65	86	151
BLACK ROCK PS WESTON	Severn	1995-1998	61	84	145
BRIDGWATER STW	Parrett	Dec-96	35		
ICI MIXED DISCHARGE (A'mth)	Severn	Nov-96	107	178	285
BRIDGWATER STW	Parrett	Dec-96			
BRITANNIA ZINC LTD KWR	Severn	Jun-95		150	
KINGSTON SEYMOUR STW	Severn	Dec-96		88	

7.1.4 PCBs

PCB congeners 28,52,101, 118, 138, 153 and 180 were analysed in freshwater samples from two HMP sites in 2001, the Avon at Keynsham and the Severn at Haw Bridge. All were below the detection limit of 5ng l^{-1}. In previous years relatively few values above the DL have been recorded from these sites (17) of which the highest was 369ng l^{-1} (PCB 101) in 1992.

Earlier samples (mid 1990s) from the River Usk (chain bridge) , Ebbw Fawr (Rhiwderin) and Tone were below detection limits. A single sample from the R. Parrett at Westover Bridge contained small amounts of PCBs 028 (11ng l^{-1}) and 052 (53ng l^{-1}).

PCBs in estuarine waters are recorded as below detection limits apart from one entry from Lower Shoots water column in the mid Severn in 1993 (194ng l^{-1} 'PCBs').

The only discharge with data showing significant values relates to the Monsanto effluent where annual averages ranged from 11.9-33.4 µg l^{-1} between 1992 and 1994.

The data on PCBs are not adequate to assess site characteristics or spatial trends, though the fact that most water samples are below detection limits implies little acute threat. However in view of their affinity for, and persistence in, sediments, more information on PCBs in this phase is probably needed. Particulate PCB concentrations recorded for NMMP sites at Bedwin Sands and Purton were 12-18 and 0.5-174 µg kg^{-1}, respectively, for years between 1994 and 1998. According to OSPAR guidelines (upper provisional value 10 µg kg^{-1}) these would be identified as areas of concern.

7.1.5 Hydrocarbons (Oil)

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

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

The EA database provided no information on 'hydrocarbon oils' at HMP sites or estuarine waters in the pSAC. A small number of freshwater samples mainly relating to discharges in the Bristol/Avonmouth area are shown in table 38. The median values are difficult to assess due to the absence of appropriate standards. Though inappropriate, the 'Bathing waters guideline EQS' of 300µg l^{-1}, (as the 90th percentile), may be used for reference purposes. Despite the fact that these appear to be industrial sites only two (Lydney and KWR mixed) would exceed the guideline.

Table 38. Hydrocarbon oils (µg l^{-1}) in streams

Site	Year	median (µg l^{-1})	90th percentile (µg l^{-1})
DRAIN DS LYDNEY IND ESTATE	2000	250	15410
INTAKES (RAW WATER) BRISTOL W/W	1998	50	100
KWR MIXED	1995	470	23738
STUP PILL D/S SOUTH SITE DISCHARGE	2000	200	300
STUP PILL DISCHARGE	1996	60	265
STUP PILL WASHING POOL LANE	2001	100	177
TRIB DITCH OF STUP PILL U/S BRICKWORKS	2000	100	260
TRIB DITCH STUP PILL U/S SEVALCO	2000	50	95

[1] It should be noted however that there are no designated shellfish beds in the Severn (the nearest are in the Taw Torridge estuaries): these guidelines are used merely as a benchmark for site characterisation in the absence of more relevant standards.

Values for discharges are included in figure 53. Though, again, entirely inappropriate, the 'Bathing waters guideline EQS' of 300µg l^{-1}, as the 90th percentile, is also included merely for reference purposes. For the majority of sites monitored (mainly 1992-1997) the 90th percentiles for hydrocarbon oil concentrations are above this 'threshold', as might be expected; significantly so for three of the discharges. With the exception of these three sites, median values are within the range 50-310 µg l^{-1}. At several sites hydrocarbon oil concentrations in discharges exceed the level (1000µg l^{-1}) at which tainting of fish and shellfish might be expected.

In the absence of recent data for concentrations in waters of the pSAC, we cannot comment, realistically, on the current water quality status with regard to hydrocarbons or their potential impact on estuarine biota. It is recommended that this situation be addressed, particularly in the vicinity of shellfish beds.

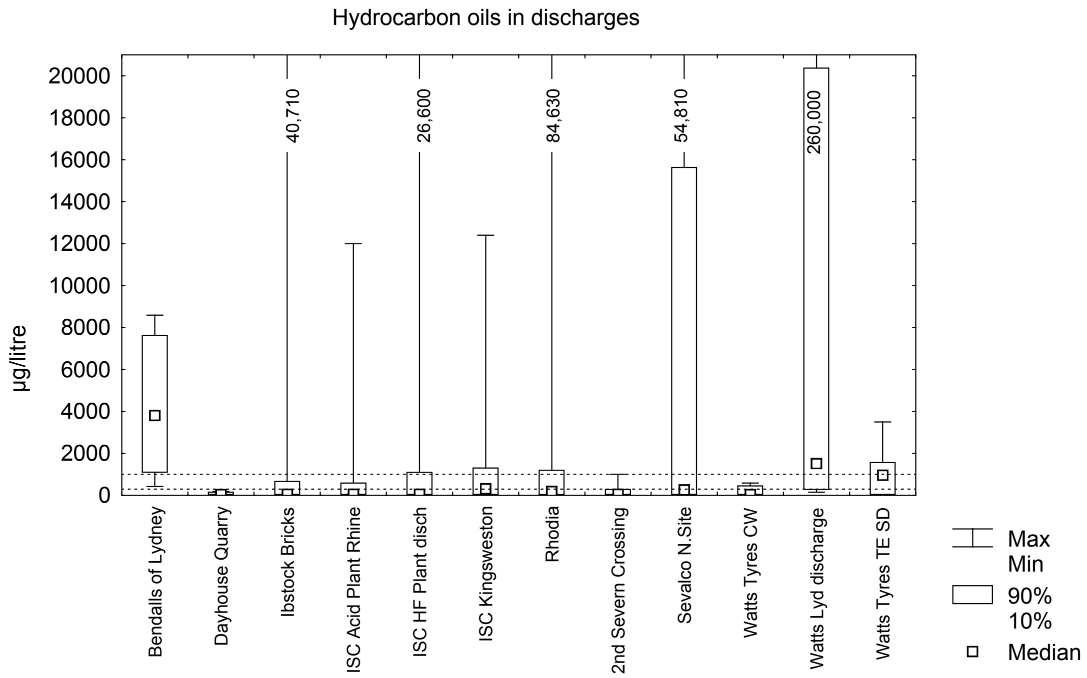


Figure 53. Hydrocarbon oils in discharges: Note boxes are 10th and 90th percentiles. Purely as an arbitrary measure, the Bathing waters guideline EQS of 300µg l^{-1} -as the 90th percentile– is shown as the lower dashed line. The upper dashed line represents a level (1000µg l^{-1}) at which tainting of fish and shellfish might be expected.

Oil has also been cited in four other discharges, as Ekofisk or Fortes Hydrocarbons, both giving rather similar median values of a few µg l^{-1} (figure 54 shows data expressed as Fortes hydrocarbons).

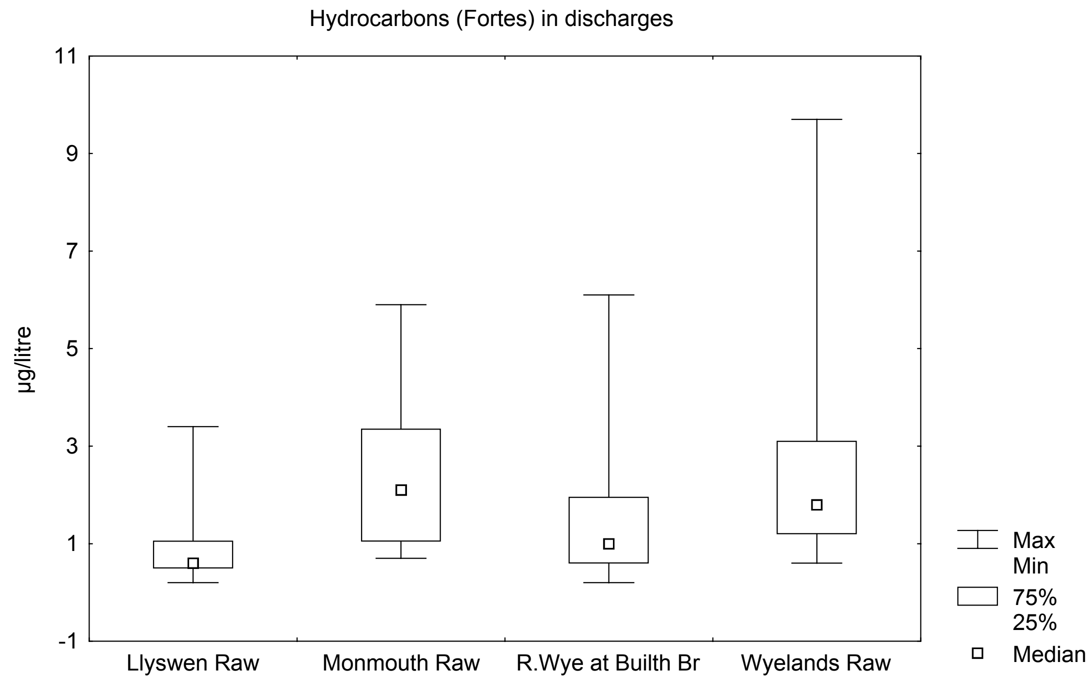


Figure 54 Oil in discharges, expressed as Fortes Hydrocarbons, (Data are for 1997-99)

7.1.6 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 only a limited amount of information in the EA dataset for total PAH concentrations in freshwater at the HMP points (2 sites). Summary statistics for Total PAH concentrations in the River Severn at Haw Bridge and the Gloucester Canal at Sharpness indicate annual median values close to 1 $\mu g\ l^{-1}$, though this often represents values close to detection limits. The median value was reported as 20$\mu g\ l^{-1}$ in 1993 but was probably biased by occasional high values.

There are only a few records of PAHs in estuarine water, mainly from the 1990s and limited to a small number of sites in mid Estuary, often downstream of discharges. The summary statistics (figure 55) indicate these have a substantial PAH concentration relative to the more open water near Severn Bridge (mean 2 $\mu g\ l^{-1}$). Concentrations greater than 1$\mu g\ l^{-1}$ (total PAHs) in estuaries are considered to be significant (Cole *et al.,* 1999), therefore total PAH concentrations are (or were) an important issue and should perhaps be surveyed more widely throughout the Estuary. This is backed up by the few available data for discharges which appear to highlight inputs in the Lydney area (figure 56), at least in the past.

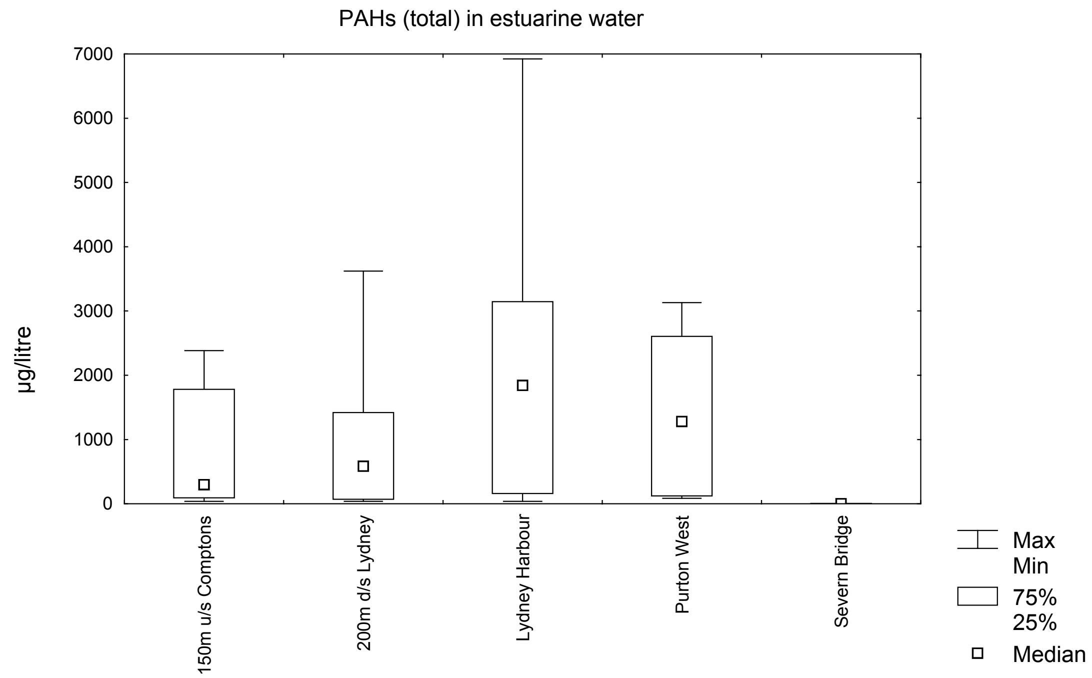


Figure 55. Total PAHs in estuarine water samples from the Severn Estuary. (Summary statistics, 1995 or 1996)

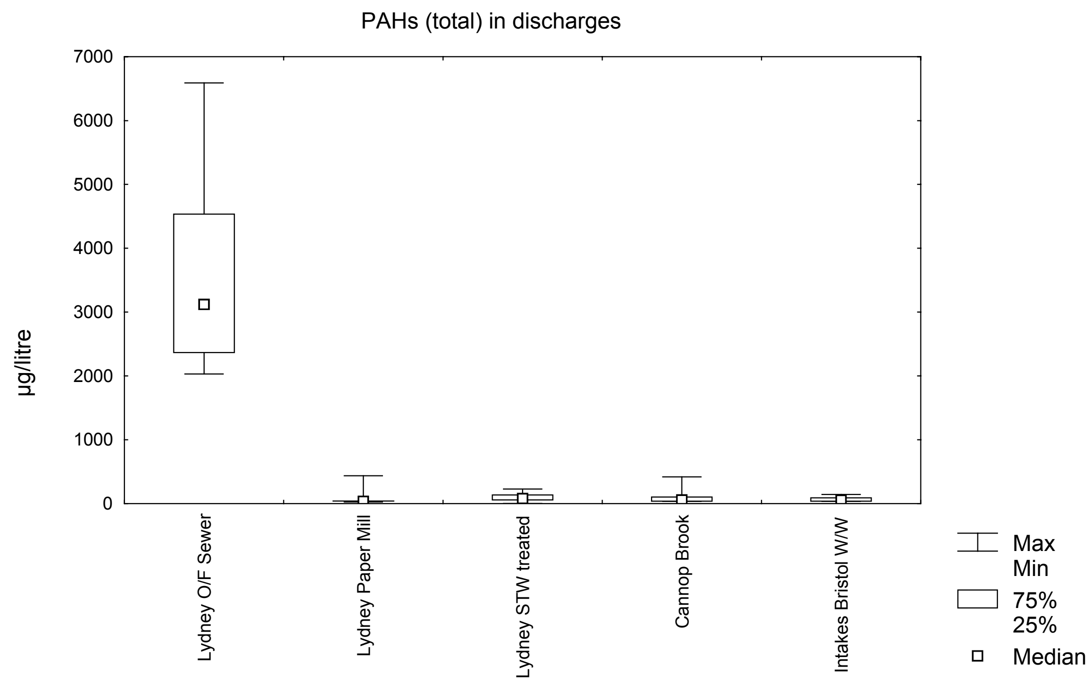


Figure 56. Total PAHs in discharges to the Severn Estuary.

Data on 12 PAHs were extracted from the database: naphthalene, chrysene, fluoranthene, pyrene, benzo(a)pyrene, benzo(b)fluoranthrene, benzo(k)fluoranthrene, ideno(123-cd)pyrene, benzo(ghi)perylene, acenaphthene, fluorene and anthracene

Measurements of individual PAHs in discharges focus on the Lydney area where naphthalene in paper mill and STW effluent were relatively high in 1995 (up to 15.3 and 290µg l^{-1} respectively). There followed a sharp decrease in concentrations and the most recent data indicate concentrations of 0.8 and 0.16µg l^{-1}, respectively. In 1995 the BSC[1] plant at Llanwern was also a major source of several PAHs as indicated in table 39 which lists contemporary data for other sites in the database for comparison.

Table 39. PAHs in discharge samples. Majority of samples are from May 1995*

	PAH concentrations µg l^{-1}				
	Lydney paper mill	**Lydney STW**	**Llanwrthwl STW**	**Lucas-Girling**	**BSC Llanwern**
naphthalene	15.3	290	<0.1	<0.1	nd
chrysene	0.228	0.526	0.014	0.024	nd
fluoranthene	0.47	0.917	0.038	0.187	50.35
pyrene	0.063	0.656	0.021	0.123	nd
benzo(a)pyrene	0.013	0.087	0.017	0.014	16.2
benzo(b)fluoranthene	0.01	0.127	0.024	0.02	20.4
benzo(k)fluoranthene	<0.01	0.042	<0.01	<0.01	10.01
ideno(123-cd)pyrene	0.066	0.147	0.039	<0.01	21.26
benzo(ghi)perylene	0.013	0.115	0.018	<0.01	7.79
acenaphthene	0.112	3.16	<0.01	nd	nd
fluorene	0.063	3.23	nd	0.877	nd
anthracene	0.079	0.846	<0.01	0.04	nd

* exception is Lucas Girling SW, sampled in 1993.   nd= not determined

Of more than 450 measurements for naphthalene in freshwater HMPs only four sites recorded concentrations of naphthalene above detection limits. These were the Avon at Keynsham (0.78 µg l^{-1}), the Severn at Haw Bridge (0.2µg l^{-1}), the Parrett at Westover Bridge (0.57 µg l^{-1}) and the Tone at Knapp Bridge (0.6-0.8 µg l^{-1}). Even less data are available for other PAHs as indicated in table 40.

There were only three samples for saline waters, near Weston-super-Mare; virtually all PAHs were below detection limits (0.1-0.3 µg l^{-1}) and hence, for naphthalene

[1] Llanwern is closing down. BSC ceased to exist some years ago, having been replaced by Corus. The plant stopped steelmaking in July 2001 and coke making in March 2002. Hence a potentially substantial source of metals and PAHs has ceased operation. Emissions will be in PARCOM reports and the Agency Pollution Inventory on web site.

below the EQS (5 µg l^{-1} annual average; 80µg l^{-1} maximum). Clearly these data are insufficient to characterise the threat from PAHs in waters across the pSAC.

Table 40. PAHs in rivers entering the pSAC. (HMP sites or equivalent)

	PAH concentrations µg l^{-1}			
	Severn Haw Br	**Parrett Westover Br**	**Tone Knapp Br**	**Avon Keynsham**
naphthalene	0.2	0.57	0.6-0.8	0.78
chrysene	0.005-0.03	0.03-0.04	nd	nd
fluoranthene	0.006-0.1	0.007	nd	nd
pyrene	0.006-0.07	0.005-0.01	nd	nd
benzo(a)pyrene	0.002-0.029	<0.002-0.005	nd	nd
benzo(b)fluoranthene	<0.005-0.027	<0.002-0.01	nd	nd
benzo(k)fluoranthene	0.001-0.014	<0.002-0.003	<0.002-0.014	<0.002-0.054
ideno(123-cd)pyrene	<0.005-0.085	nd	nd	nd
benzo(ghi)perylene	0.002-0.031	<0.002-0.01	0.002-0.027	<0.002-0.085
acenaphthene	<0.005-0.132	nd	nd	nd
fluorene	<0.005-0.738	nd	nd	nd
anthracene	<0.005-0.033	nd	nd	nd

nd= not determined.

7.2 Non-Toxic Contaminants

7.2.1 Nutrient Quality Criteria.

Nutrient concentrations vary with salinity, therefore measurements collected simultaneously from different regions within the Estuary, or from the same region but at different states of the tidal cycle, may show considerable differences and not be truly representative of water quality. To compound this difficulty, nutrient concentrations also vary throughout the year with freshwater flow. As yet there are no statutory water quality standards for nutrients in the UK and determination of the nutrient status of estuaries, and the ecological consequences, remain a notoriously contentious issue. To quote from the Agency's Technical Guidance for Water Quality: Review of Permissions to Discharge and New Applications (Habitats Directive) - 'Generally, it is impossible to calculate permit conditions in the absence of water quality standards…' and ' it is not easy to make a case or refuse or reject an application in the absence of such standards'. Therefore, judgement of nutrient status in the Severn Estuary pSAC, 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 Severn Estuary pSAC, and for initiating management responses:

- EU nitrates directive 91/676/EEC, on the protection of all waters against pollution caused by nitrates from agricultural sources, calls for the identification of all waters that contain **50mg l^{-1} nitrate**.
- The USEPA is still in the process of arriving at their national nutrient strategy but has for many years proposed a limit of **10mg l^{-1} nitrate** nitrogen for the protection of domestic water supplies (against overenrichment and impacts on human and animal health). A phosphorous criterion was reported some years ago in the EPA 'Red Book' as **0.1µg l^{-1} (as P)** to protect estuarine and marine organisms against the consequences of bioaccumulation (EPA, 1976). However, this was not established as threshold for eutrophication and is currently under review.

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

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

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 can then be grouped according to the following class boundaries (table 41):

Table 41. 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 Severn Estuary pSAC 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 Severn Estuary and comparisons with other areas
- validity of guideline values and classification schemes

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

Table 42. 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**

Cole *et al*., (1999) made a comparison of the nutrient status of UK estuaries, having extrapolated freshwater values (from seawater values) on the basis of conservative mixing. Using these criteria, the projected classification for TIN is below the UK average grading it as class B. Projected TRP is also relatively low resulting in a grade C (table 42).

Table 43. TIN and TRP in waters entering the Severn Estuary pSAC based on observations 2001. (Data source: EA)

River	Median TIN conc. (mg l^{-1}) in freshwater	GQA TIN class	Median TRP conc. (mg l^{-1}) in freshwater	GQA TRP class
Severn	3.69*	A	0.46	C
Parrett	5.57	B	0.58	C
Tone	7.42	B	0.65	C
Ely	-	-	0.37	B
Taff	-	-	0.17	A
Rhymney	-	-	0.05	A
Afon Lwyd	-	-	0.04	A
Wye	-	-	0.07	A
Bristol Avon	6.71	B	0.75	C
Ebbw Fawr	-	-	1.03	D
Usk	-	-	0.05	A

* 2000 data - not available for 2001
\- no data available

Actual EA data for TIN and TRP in waters entering the pSAC recently (2001) (where available) allows pSAC waters to be classified according to the same scheme (table 43).

The median TRP value for the Severn is approximately the same as the projected FW concentrations described by Cole *et al.,* (1999), and still results in a classification of grade C. The TIN value for the Severn is lower than that projected by Cole resulting in the classification of A, as opposed to B.

It is not known for certain whether the discrepancy is artifactual (a result of using real measurements as opposed to modelled values) or the result of genuine changes in water quality. However the latter explanation seems unlikely since there is no evidence of widespread temporal change in the Agency data. Nevertheless, this example serves to illustrate the problems of assessing nutrient status.

Information relating to some of the other rivers which feed the Severn is included in table 43. If projections are valid, some of the rivers could introduce nutrient rich waters into the Severn.

There are other schemes which estimate the nutrient status from freshwater load inputs, thus encompassing point source discharges to rivers. Dong *et al.,* (2000) calculate estuarine nutrient loads by multiplying annual 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; likewise estuarine sources make this type of estimate unreliable.

Figure 57 shows temporal trends in the nitrogen (total) and orthophosphate loads to the Severn Estuary (1991-19997) estimated by the EA (1999). There appears to have been a slight overall increase in the loadings during the period.

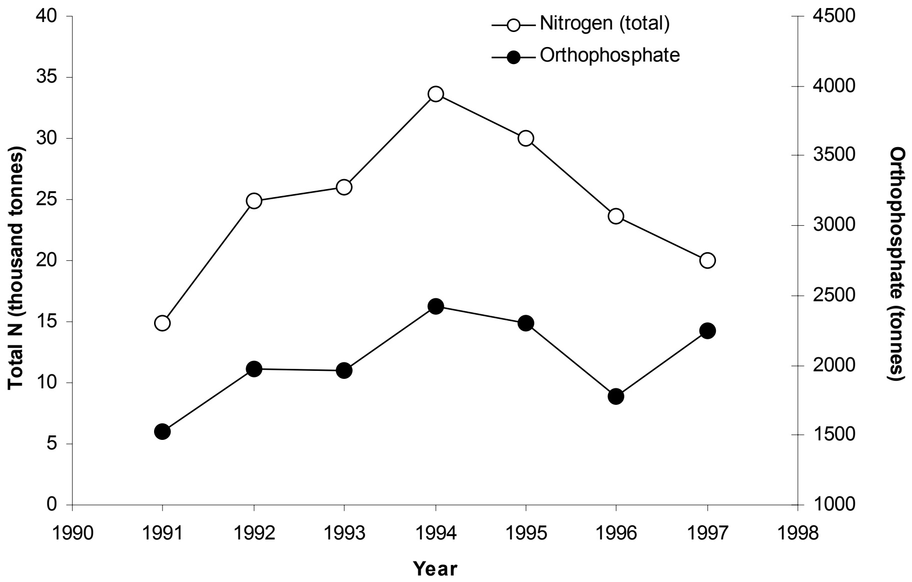


Figure. 57. Estimated loads of nitrogen (total) and orthophosphate to the Severn Estuary. Data source: EA (1999)

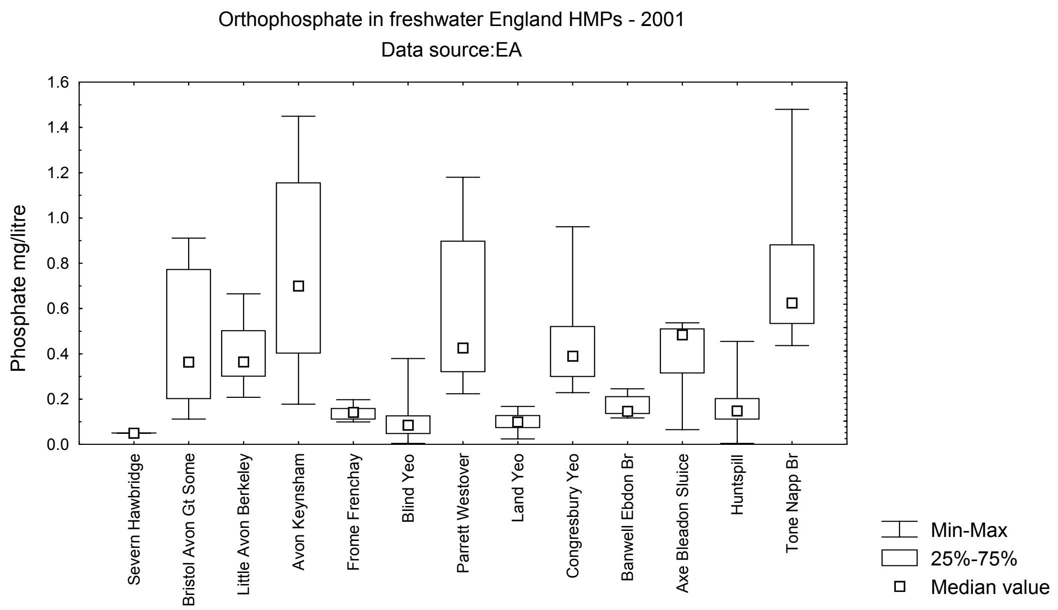
Historically, sources of P are considered to be industrial effluents, rivers and streams and domestic sewage, and for N, rivers (predominantly) with a significant component from discharges (Owens, 1984). The issue of whether or not to focus on nutrient concentrations in the tidal waters or loading criteria 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 pSAC and its catchment are treated separately in an attempt to further apportion sources of nutrients.

7.2.2 Phosphate

Riverine sources of phosphate entering the pSAC have been assessed by comparison of monitoring data for harmonised monitoring points (HMPs), or their equivalent, in the various catchments (figure 58 a, b). Many of the HMP sites are fairly high up on the rivers, but give an overall impression of orthophosphate levels introduced in freshwaters.

Median concentrations are somewhat elevated in some areas, notably the Avon, Parrett, Tone and Axe on the English side, and the Ely on the Welsh. Annual average concentrations have increased over the past 10-12 years at HMPs on the Frome, Ebbw Fawr (below Cardiff), Blind Yeo and the Usk, whilst annual averages for the Severn, Rhymney (at Llanrumney), Ebbw Fawr (Rhiwderin) and Congresbury Yeo have remained relatively constant. Concentrations at the remaining 14 freshwater sites have decreased over the period, significantly for many sites.

(a)



NB one measurement only for River Severn HMP

(b)

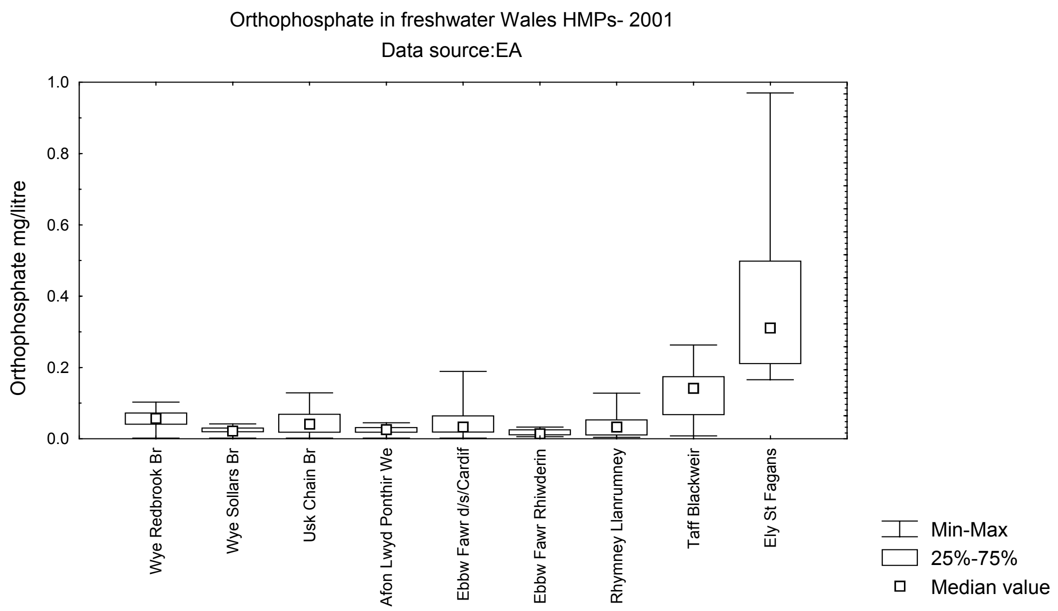


Figure 58. Dissolved orthophosphate at HMPs (or similar) in freshwater catchments feeding the Severn Estuary – (a) East shores (England). (b) West shores (Wales). Data are for 2001. Data source EA.

Data for orthophosphate in some of the discharges (2001) to waters of the pSAC are summarised in figure 59. Highest concentrations are in non-water company effluents. *However, it is stressed that the data are for concentration only and does not take into the account volume discharged.*

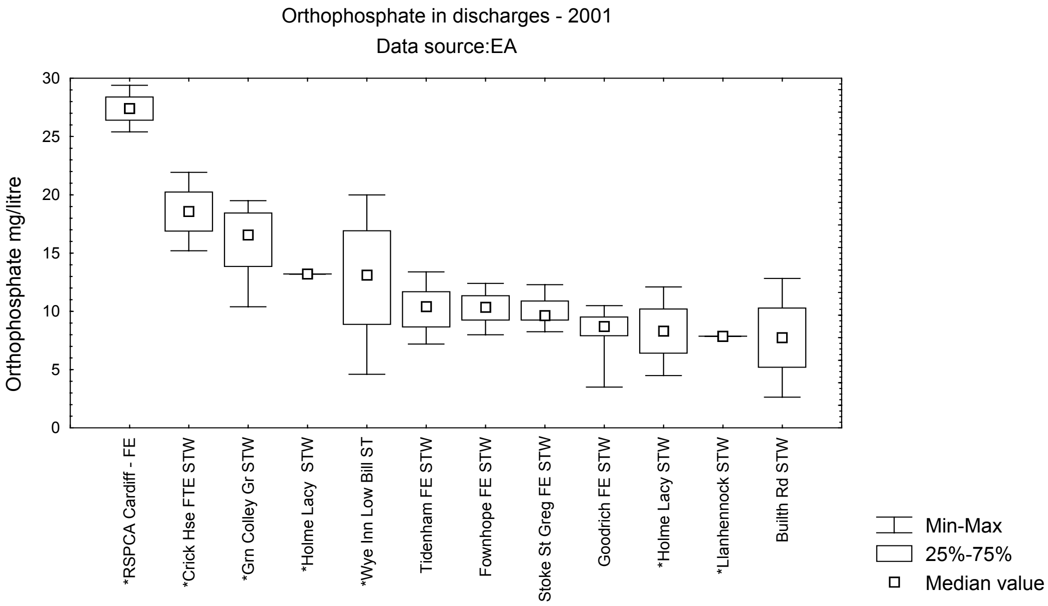


Figure 59. Dissolved orthophosphate in discharges to the Severn Estuary or tributaries – those with median concentration >7.7 mg l^{-1} shown. Data are for 2001. *denotes non-water company discharges. Data source EA.

Data for phosphate in tidal waters of the pSAC are generally for the period 1990-1997, as the EA ceased routine monitoring in 1997. Monitoring instead became targeted to particular estuaries (N.Cunningham, *pers comm.*), with sampling and analyses tailored to fulfil the requirements of several different directives. (Thus, for example, whilst figure 12 shows regional divisions and monitoring points therein, the data are not necessarily comprehensive for each site and each determinand).

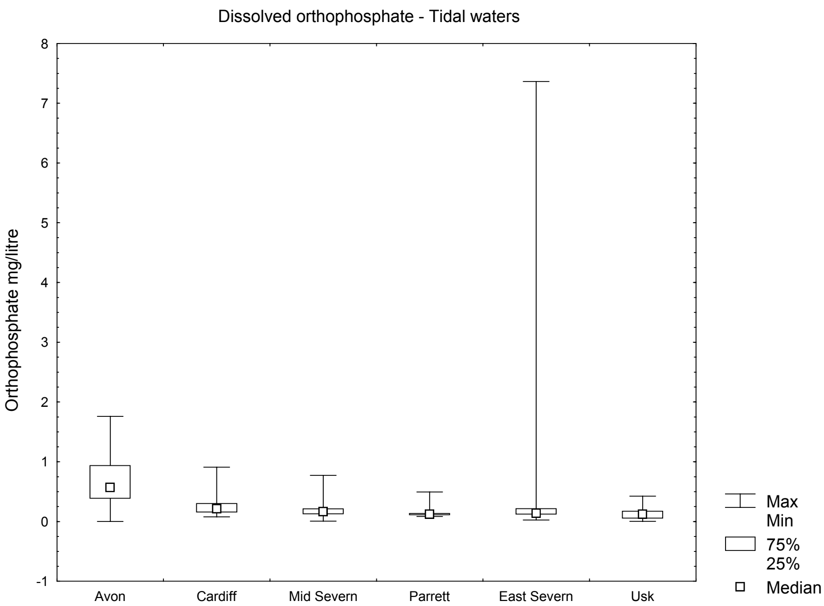


Figure 60. Dissolved orthophosphate in tidal waters for different regions of Severn Estuary. Summary data for Avon 1990-95, Cardiff 92-95, mid-Severn 92-98, Parrett 93-97, East-Severn 90-97, Usk 92-2001. Data source:EA

The pattern for orthophosphate in tidal waters plotted on a regional basis is shown in figure 60. Concentrations of orthophosphate reflect elevated concentrations in riverine and effluent discharge inputs to a certain extent. e.g. the Severn east and the Avon where freshwater levels are high (see figure 58). There are nine sampling points represented by the statistics for the Avon region, and median concentrations are very similar for each (0.38 – 0.98mg l^{-1}). The most elevated concentrations occurred in the Bristol Avon at Bath Bridge (mean, median and max, 0.92, 0.98 and 1.5mg l^{-1} respectively).

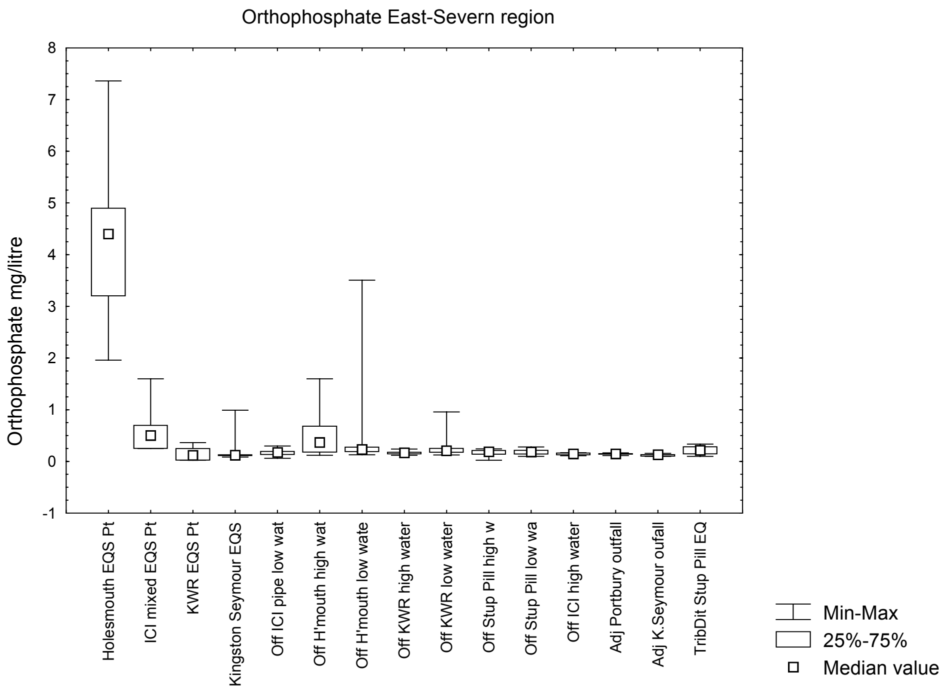


Figure 61. Dissolved orthophosphate in tidal waters for East-Severn region. Summary data 1990-1997. Data source EA.

Data for tidal waters of the east Severn region are summarised in figure 61. Sampling points in this region are predominantly along the shore of the Estuary and the more elevated concentrations occur off the industrialised area of Avonmouth (Holesmouth). Discharges with the highest concentrations of orthophosphate in the region are shown in figure 62. *Again it is stressed that these data represent concentrations only and not loadings (and may be incomplete).* There is no up-to-date information available as to the current status of discharges. However, the available data indicate that trade discharges from Albright and Wilson at Portbury Wharf represent a significant source of orthophosphate; the annual average concentration was 148mg l^{-1} in the early 1990s and although this figure had dropped to 73mg l^{-1} in 1996, it was still the highest in the region by far.

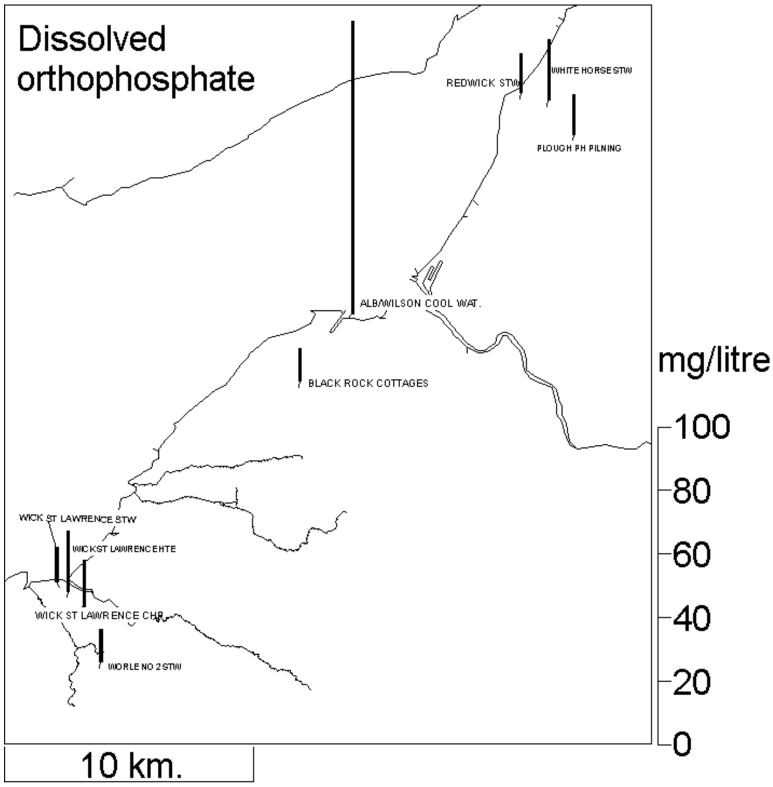


Figure 62. Dissolved orthophosphate in discharges to the east-Severn region. Summary data 1990-1997. Data source EA.

Calculated as elemental P, the approximate background for the tidal waters (25th percentile) in component regions is in the range 17.9 – 125µg l^{-1} (table 44), invariably above the 0.1µg l^{-1} criteria set by the EPA(US) to protect estuarine and marine organisms, but generally in the low-mid range reported by Parr *et al* (1999) for coastal waters (7 – 165µg P l^{-1}).

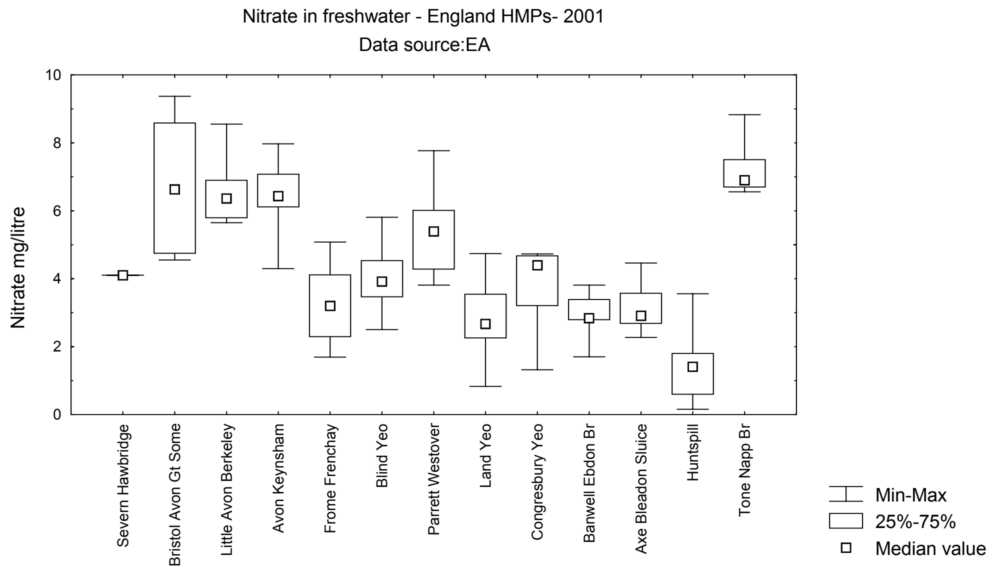
Table 44. Elemental P: 25th percentile for Severn Estuary regions. Data source:EA

Region	Elemental P 25th percentile µg l^{-1}
Avon	125
Cardiff	51.5
Mid-Severn	41.6
Parrett	36.5
East-Severn	39.8
Usk	17.9

Examining data for individual mid-estuarine sites in the helicopter surveys, Ellis (2002) showed that orthophosphate concentrations gradually increased from the seaward end of the Estuary, rising by ~7-fold or more at inner estuarine sites. Ellis also indicated a number of decreases in means for orthophosphate at individual sites in 1990 – presumably as a result of water quality improvement schemes in the 1980s. An increase in mean concentration for the inner Estuary in 1996 was the only increase observed.

7.2.3 Nitrate

(a)



NB one measurement only for River Severn HMP

(b)

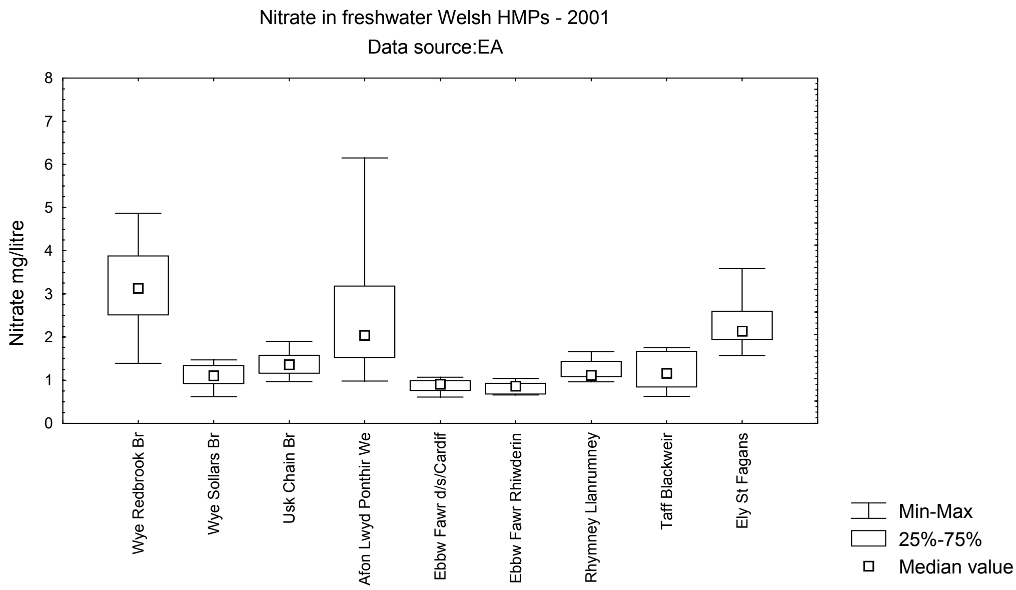


Figure 63. Nitrate in waters at HMPs (or similar) for freshwater catchments feeding the Severn Estuary – (a) East shores (England). (b) West shores (Wales). Data are for 2001. Data source EA.

Freshwater values for nitrate at HMPs or equivalent in the pSAC are shown in figure 63a,b. Concentrations appear to be elevated in several of the catchment areas, particularly the Avon and Parrett although median values (and 25th and 75th percentiles) are below the lower threshold of 10mg l^{-1}. For the Avon, at least, point source discharges in the catchment area probably contribute significantly to these high levels (see section 5.2.2). Over a 12-year period (1990-2001) annual average concentrations of nitrate have risen at HMPs on the Afon Lywd, Blind Yeo, Axe and Tone. On the Severn at Haw Bridge, nitrate concentrations dropped between 1990-98 but have been steadily increasing since. Levels have remained relatively constant at the Congresbury Yeo, Little Avon (Berkeley), the Wye (Sollars Bridge and Redbrook Bridge) and the Banwell at Ebdon Bridge. Nitrate in freshwater at the remaining 12 freshwater sites has remained relatively constant over the period.

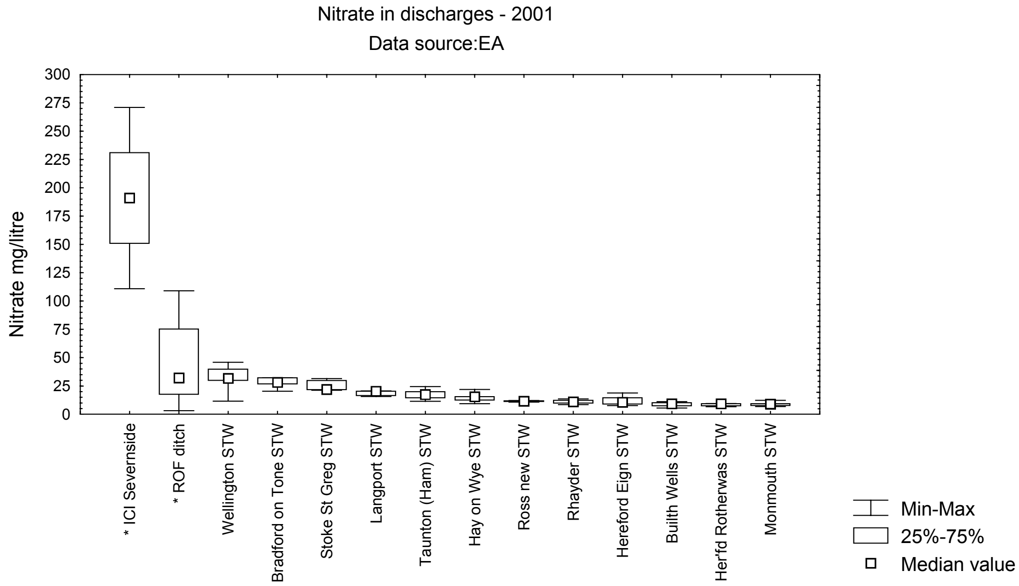


Figure 64. Nitrate in several of the discharges to the Severn Estuary or tributaries – those with median concentrations >9mg l^{-1} shown. Data are for 2001. *denotes trade discharges. Data source EA.

Figure 64 summarises data for nitrate in a selection of discharges within the Severn catchment. *Again, it is stressed that the data are for concentration only and does not take into the account volume discharged.*

Highest concentrations are in trade discharges (ICI Severnside (now Terra Nitrogen) and ROF Ditch[nr Huntspill]) which dominate the plot and presumably contribute significantly to the elevated concentrations in tidal waters of the Severn and Avon (figures 64 and 65). Concentrations of nitrate in some of the STW effluents are also elevated, e.g. Wellington and Bradford-on-Tone, which both discharge into the River Tone, and which probably contribute significantly to the relatively high levels at the HMP (Knapp Bridge – see figure 63a).

Nitrate in tidal waters, plotted on a regional basis, is shown in figure 65. Data are for the period 1990-99. There were insufficient values to include Cardiff and Parrett regions. Concentrations were relatively high throughout the Estuary. The Avon region represents 10 sampling sites, with median concentrations generally 6-7mg l^{-1}.

Maximum values for this region (up to 11.8mg l^{-1}) were recorded in the Bristol Avon at Coronation Road. The Mid Severn represents only two sampling sites, Lower Shoots, below the Severn Bridge, and Sharpness (mid channel) above the bridge.

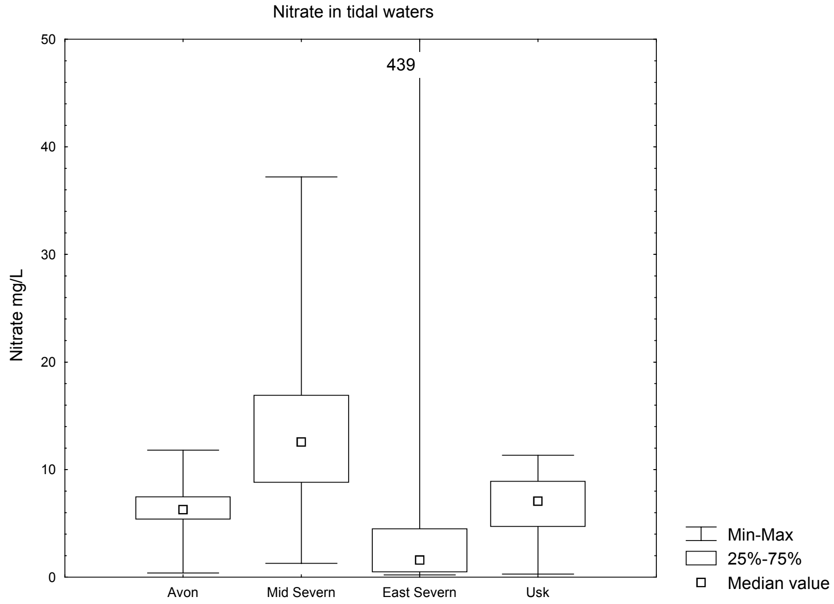


Figure 65. Nitrate in tidal waters for different regions of Severn Estuary. Summary data for Avon 1990-95, mid-Severn 94-98, east-Severn 90-94, Usk 95-99. Data source:EA

Figure 66 summarises nitrate data for the tidal waters of the east-Severn region. There are 11 sampling points represented by the data, median concentrations are generally between 1 - 4.5mg l^{-1}, although there are two notable exceptions, off ICI at an EQS sampling point, and a tributary ditch at Stup Pill in the same vicinity, which again indicates that the ICI discharge constitutes a major source of nitrogen.

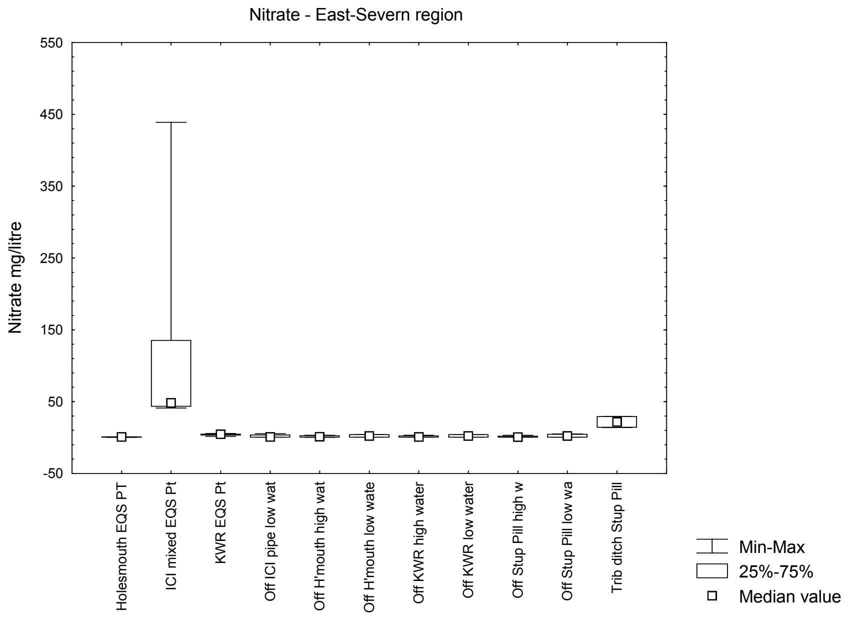


Figure 66. Nitrate in tidal waters of East-Severn region. Summary data 1990-1999. Data source EA.

Mean values for nitrate (expressed as N), for regions of tidal waters are shown in table 45. None are below the TIN value (0.144mg l^{-1}) considered to represent the threshold for hypernutrification (dependant on P levels) in coastal waters (North Sea Quality Status Report).

Table 45. Nitrate as N, mean for Severn Estuary regions.

Region	Mean mg l^{-1}
Avon	1.43
Mid-Severn	3.01
East-Severn	3.98
Usk	1.48

Expressed as N, most of the nitrate values for tidal waters are higher than 0.144mg l^{-1}, (in the range 0.049 – 99mg l^{-1}) and 84% are above the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations.

Examining spatial trends in the data for mid-estuarine sites in the helicopter surveys, Ellis (2002) showed a gradual increase in nitrate concentration from the outer (Bristol Channel) sites to the inner Estuary. Ellis also indicated a number of temporal decreases in means in nitrate concentration at individual sites (~10) in the period 1986 –1988 whereas only one increase was observed (outer Estuary - 1995). All but one record of sites examined by Ellis (20002) failed to comply with a 'proposed' EQS value of <210µg l^{-1} (1980-1997), and none of the helicopter monitoring sites conformed to the 'EQS' for TON (200 µg l^{-1}) although there is no indication as to the origin or derivation of these EQSs.

Thus, nutrient levels are elevated in tidal waters of the pSAC, particularly in the inner Estuary. Localised enrichment as a result of elevated concentrations in certain discharges may contribute to DO sags in this region (see section 7.2.4). Terra Nitrogen (Severnside) is an important source of nitrates (actual 758×10^3 kg yr^{-1}) Important sources of ammonia include Cardiff East (estimated 9173×10^3 kg yr^{-1}), Ponthir (estimated 1118×10^3 kg yr^{-1}), Nash STW (estimated 972×10^3 kg yr^{-1}), Terra Nitrogen (actual 613×10^3 kg yr^{-1}) and Gloucester STW (estimated 469×10^3 kg yr^{-1}) (Bird *et al.,* 2002).

It is generally acknowledged that concentrations of nutrients in the Severn are amongst the highest in UK estuarine waters (CEFAS, 2000), although due to the characteristics of the Estuary (tides, turbidity, flushing, etc.) there appears to be little evidence of the 'usual' problems associated with nutrient enrichment. However, effects on the rare species in the pSAC from the localised enrichment are largely unresearched, therefore it would seem that an increase in nutrients should be avoided, as a precautionary requirement. Changes to consents (quantities and location) should therefore be considered carefully to avoid the risk of further enrichment.

7.2.4 Ammonia

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

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

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

Unfortunately, monitoring data and summary statistics for ammonia in the Severn Estuary have not been provided by the Environment Agency. However mid-axial sampling and analysis for water quality parameters, including ammonia (N) have been conducted since the 1970s using helicopter surveys. Ellis (2002), summarised ammonia data (1980 to 1997) gathered in these surveys. Figure 67 shows the sampling locations and site codes used.

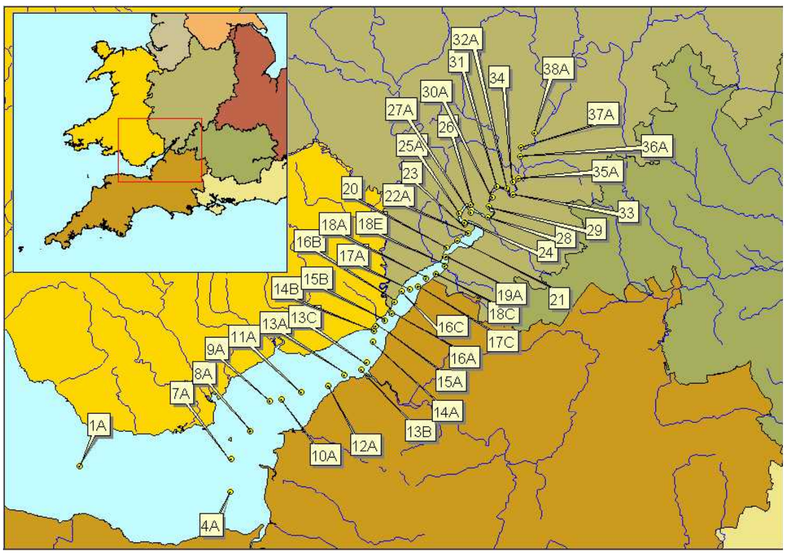


Figure 67 Sampling locations and site codes used in helicopter surveys (Ellis 2002)

Spatial trends for ammonia concentrations along the Severn Estuary are summarised in figure 68. The units are µg l^{-1}.

Mean values appear to range from ~30 to ~380µg l^{-1} and, as might be expected, are generally lower at the seaward end of the estuary, although there are peaks in concentration at sites 8A, 10A ad 12A, which are mid-channel sites between Cardiff and Portishead, and again, between 15 and 16B, under the first Severn Road Bridge (near Aust). There is also significant variation in ammonia concentrations at these sites. From site 18A, Hills Flats Buoy (upstream of Aust), mean ammonia concentrations show a gradual increase upstream, interrupted by another peak in mean concentration (300µg l^{-1}) at site 24 (Northington Pylons). The highest mean value (~380µg l^{-1}) (and 95th percentile) occurs at site 32A (near Minsterworth, and downstream of an STW), in the upper tidal estuary. Further up the estuary mean values decrease towards the freshwater Severn above Maisemore Weir.

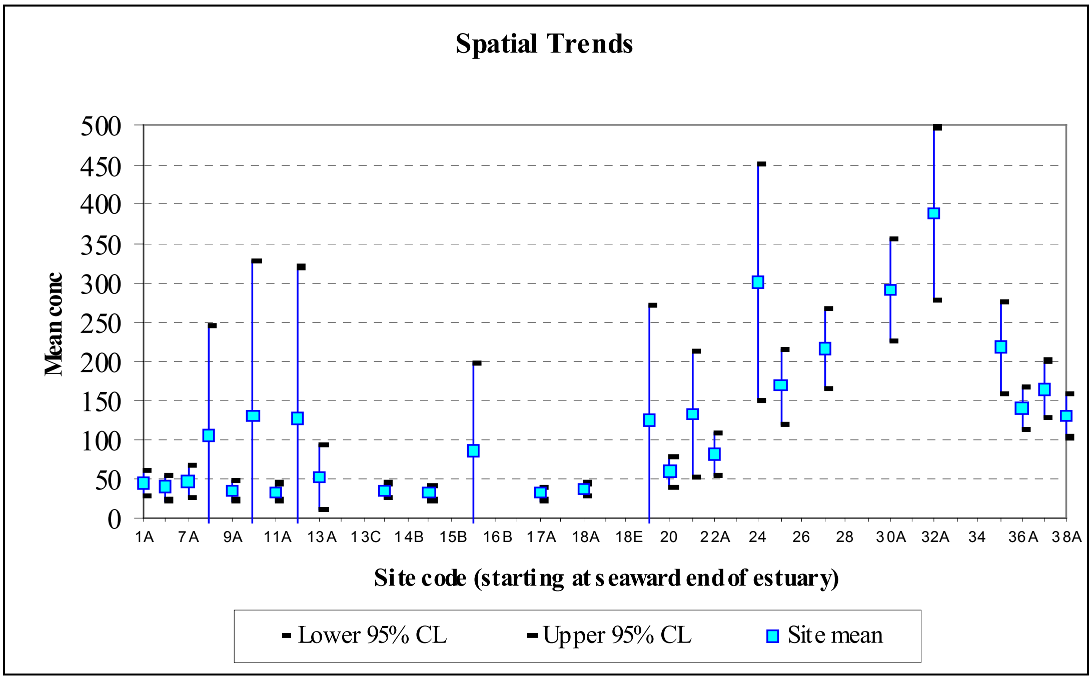


Figure 68 Spatial trends for ammonia (N) in the Severn Estuary. Units µg l^{-1}. Data for period 1980 to 1997. From Ellis (2002)

These values appear to be a product of data collected over 17 years, therefore do not necessarily represent the current status of the estuary with regard to ammonia. It must also be remembered that the samples were taken from mid-estuarine sites, and therefore do not include values for the tributary estuaries of the Avon, Usk, Parrett etc.

However, concentrations appear to be very high overall compared with other Southwest European Marine Sites. The range of mean annual values for ammonia (N) in tidal waters at sites in the southwest is shown in table 46.

Table 46 Minimum and maximum mean annual values for ammonia (N) for tidal water in South West European Marine Sites. The most recent year for which data is available shown for each estuary

Estuary	Range ($\mu g\ l^{-1}$)	
	Min	Max
Severn	**~30**	**~380**
Fal	8.2	179.0
Plymouth Sound and Estuaries	13.5	154.8
Exe	17.3	226.3
Poole Harbour	29.9	140.7
Fleet Lagoon	6.8	27.3

Figure 69 shows temporal decreases/increases in mean annual ammonia values for tidal waters of the Severn, and implies that there have been several reductions in ammonia concentration, notably in waters toward the seaward end of the estuary. Only one increase is shown since 1990, and this occurred in mid estuarine waters at No 1 Beacon (downstream of the new Severn Road Bridge) 1995/6.

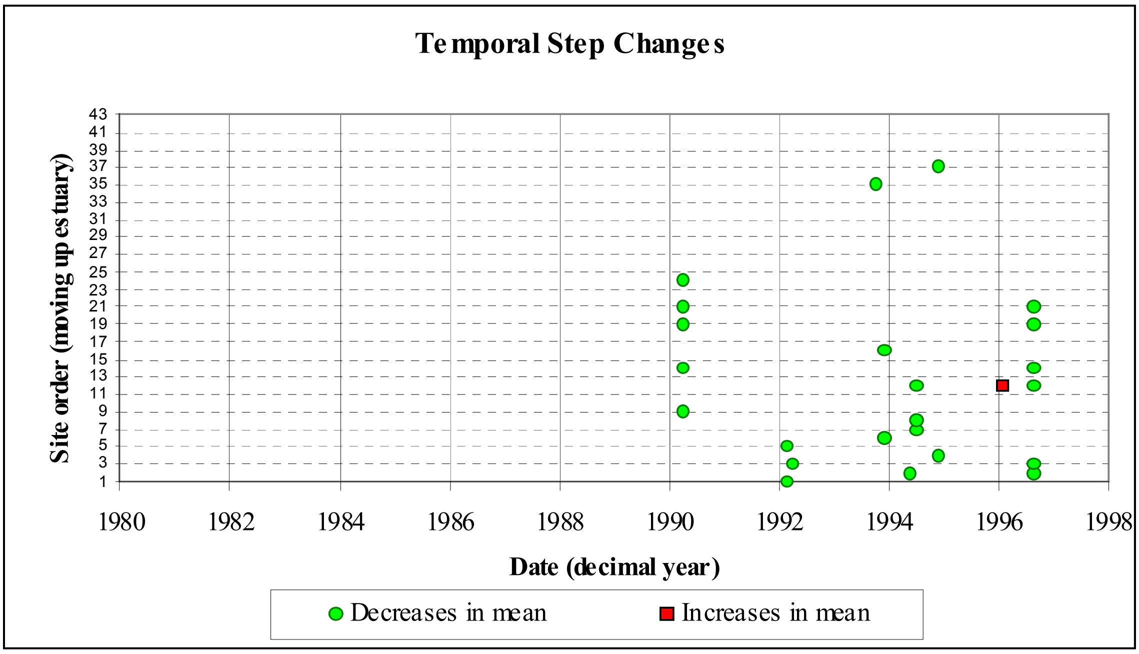


Figure 69. Temporal step changes in ammonia (N) concentrations in the Severn Estuary. From Ellis (2002)

Without information regarding ammonia concentrations for freshwater and discharge inputs to the Severn Estuary, it is difficult to establish exact sources of ammonia Generally the principal sources in estuaries are direct discharges from STW outfalls, however, for the Severn Estuary, it has been estimated that the contribution from industrial effluent almost equals that of STWs, and that >68% of the total ammonia inputs enter the more industrialised area of the estuary (Burnham on Sea to Aust) (see table 13 & 14 section 5.2.).

Some major sources of ammonia in the estuary are listed in annex 7. Terra Nitrogen (formerly Zeneca Ltd) discharge 613,000kg ammonia per year to the Severn Estuary (Severnside Avonmouth industrial site). Modelling studies have been undertaken 'to ensure that there should be little or no impact on estuarine quality' from the discharge of 1 - 2.5 million gallons per day (see section 9). Ammonia is considered as the most significant component in the effluent, toxicologically (Zeneca, 1997).

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 unionised ammonia - NH_3 (as N) l^{-1} - (≡ 26.3 and 37.4mg l^{-1} total ammonia (as N)), respectively (Allan *et al*., 1990). For the nauplius of the marine copepod *Tisbe battagliai*, Williams and Brown (1992) estimated a 96-hour LC50 of 0.787 mg NH_3 (N) l^{-1} (≡24.6mg total ammonia (N) l^{-1}), and tests on several life stages showed a No Observed Effect Concentration (NOEC) of 0.106mg NH_3 (N) l^{-1} (≡3.34mg total ammonia (N) l^{-1}). For invertebrates, toxicity appears to increase as salinity decreases (Miller *et al.,* 1990, Chen and Lin 1991), although more work is needed to establish whether this pattern is typical for all, or most, invertebrates (Nixon *et al.,* 1995). Several studies indicate that ammonia toxicity is greatest to early life stages of invertebrates.

The majority of ammonium toxicity data relates to fish, although most of the species tested are freshwater species, with many coarse fish appearing to be as sensitive to ammonia as salmonids (Mallet *et al.,* 1992). Acute toxicity of ammonia to fish increases at low dissolved oxygen concentrations in both fresh and marine water environments (Seager *et al.,* 1988, Nixon *et al.,* 1995). For this reason, the proposed GQA scheme for ammonia in estuaries was combined in a proposed joint scheme for dissolved oxygen and ammonia (Nixon *et al.,* 1995).

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

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

Note that the ammonia data are for ammonia (N), and values for unionised ammonia, NH_3 (N), would need to be calculated from the total data, taking account of pH, temperature, and salinity. As a rough guide; for a pH of 8.2, a temperature of 20˚C, and a salinity of about 30, 0.44mg l^{-1} total ammonia (N) relates to about 0.021mg l^{-1} NH_3 (N), which is the proposed EQS. There is not enough detailed information available to establish unequivocally whether this EQS is likely to be exceeded in the Estuary, though mean values are high (despite the efficient flushing characteristics of the estuary) suggesting that exceedences may could occur. The area of most concern would be the upper estuary near Minsterworth, where highest values have been recorded, and there are reports of oxygen sags (see section 7.2.5), and in other more enclosed regions such as the tributary estuaries.

7.2.5 Dissolved Oxygen

DO is measured in estuaries and coastal waters in terms of either a concentration (mg l^{-1}) or as a percent saturation (%). Table 47 shows recommended EQS values for saline waters derived from the review of Nixon *et al.,* (1995).

Table 47. Recommended EQSs for dissolved oxygen in saline waters (from Nixon *et al.,* 1995)

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

Various class thresholds for estuaries in England and Wales, based on DO over a continuous period of >1 hour were proposed by Nixon *et al.,* (1995) (see table 48) 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 48. 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 and salinity, 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. DO levels can also vary with temperature and salinity.

Increased or sustained turbidity in the water column may result in a reduction in algal (macroalgae and phytoplankton) growth rates due to reduced light availability. Subsequent adverse effects to zooplankton, benthic communities and fish populations (a general reduction in biodiversity) would be anticipated as particulates are suspended and re-deposited. An accompanying reduction in food availability may have secondary effects on higher trophic levels.

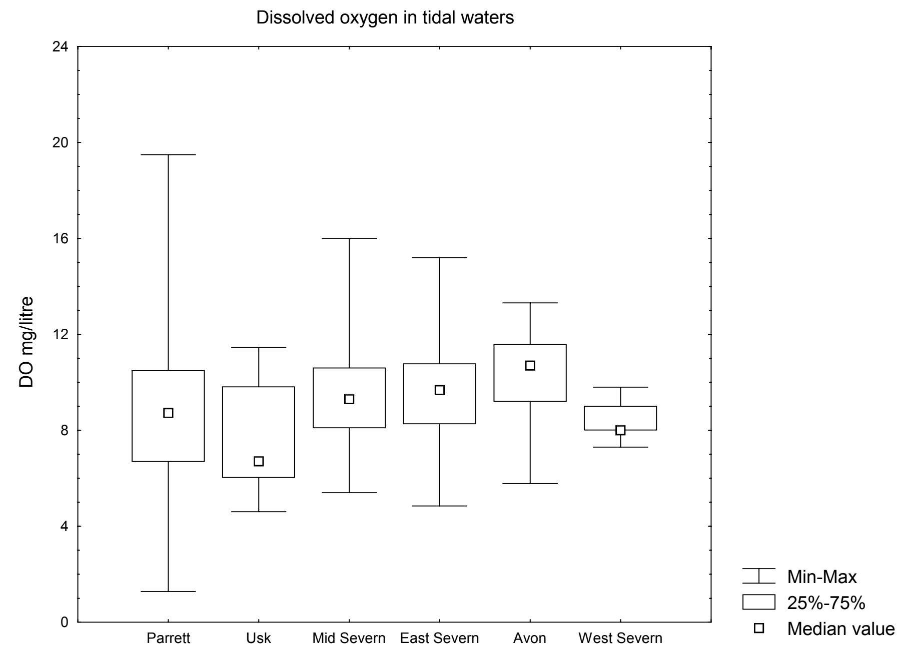


Figure 70. Dissolved oxygen in tidal waters for different regions of the Severn Estuary. Summary data for regions Parrett 1990-2001: Usk 1995-2001: Avon 1991-5, Mid Severn 1990-2002, E. Severn 1992-8, W. Severn 1994,. Data source EA.

Data for DO in tidal waters of the pSAC are summarised in figure 70. Overall median values for the regions are in the range 6.7-10.7mg l^{-1}, and are above the 5mg l^{-1} (median) recommended EQS value for saltwater life and for migratory fish (see table 47). However, minimum values fall below this figure in the Parrett, Usk and East-Severn (1.28mg l^{-1} for the Parrett).

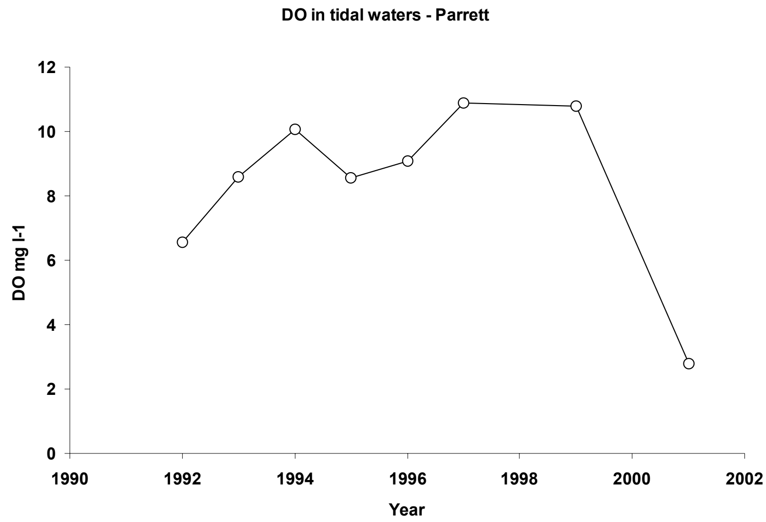


Figure 71. Temporal trend for mean annual DO in tidal waters of the Parrett. Data source:EA

Figure 71 shows temporal trends for mean annual DO concentrations in Parrett samples. However, the available data may not give an indication of DO in the region: values for the Severn shoreline are included prior to 1996, but for 1997, 1999 and 2001 the mean represents only three sites in the inner Parrett, namely, Blackbridge, downstream of Bridgwater STW, and downstream of British Cellophane.

There are no available data for the inner Severn, below Maisemore Weir, where DO sags have been reported (see section 5.2.5). However, examining spatial trends in data for mid-estuarine profiles in the helicopter surveys, Ellis (2002) show a sag at several sites in the inner Estuary, between Lower Dumball and Minsterworth, where concentrations drop to <9mg l^{-1} (<70% saturation). This sag is superimposed on a general profile which shows dissolved oxygen concentrations gradually increase from 9mg l^{-1} at the seaward end of the Estuary, to 11mg l^{-1} at the head of the Estuary.

Examining temporal trends, Ellis also indicated a number of increases in means for DO (% saturation) at individual sites between 1988-1995. Decreases were only observed, for the inner Estuary, in 1984 and 1987. Ellis found that since 1987, all sites have complied with the EQS (>5mg l^{-1} as the median and >2mg l^{-1} as 95th percentile). However, in terms of % saturation (as used under the shellfish directive), the majority of sites would have failed the EQS (>80% in 95% of samples) between 1981-1990, and more recently (1997) mid- and inner estuarine sites would still have failed. Fortunately, none of the Severn is designated as shellfish water.

7.2.6 Chlorophyll a

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

Reduced light levels due to large amounts of suspended solids may severely restrict phytoplankton growth in the Severn Estuary (see sections 5.2.3, 5.2.4), even where concentrations of nutrients may be sufficient for phytoplankton proliferation.

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 chlorophyll *a c*oncentrations for regions of the Severn are summarised in figure 72. Monitoring appears to have been restricted to freshwaters since 1998 therefore values are for the period 1992-8. Generally, chlorophyll *a* analysis is for April-September, although measurements taken in the winter months (Dec, Jan, Feb) are included for the Parrett region.

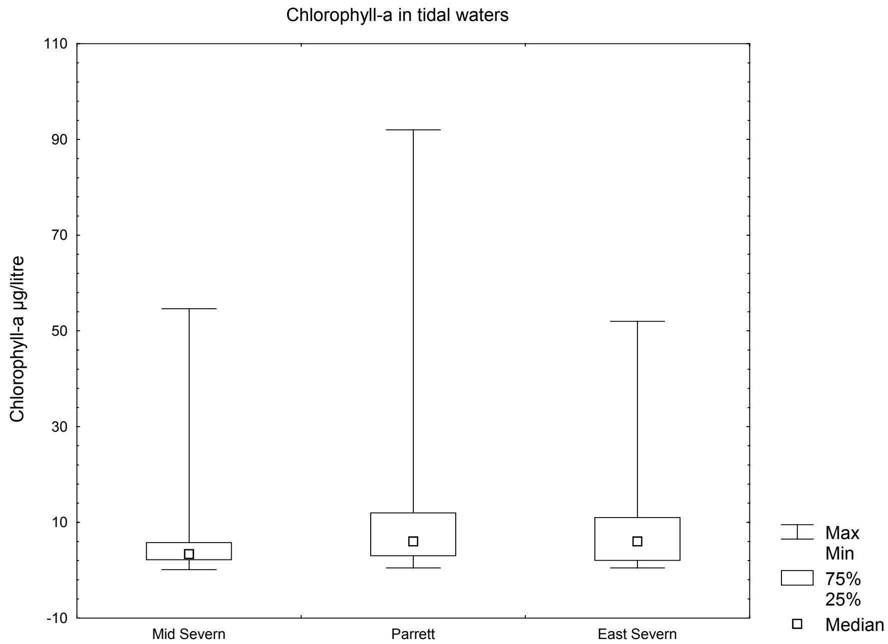


Figure 72. Chlorophyll *a* in tidal waters for different regions of the Severn Estuary. Summary for period 1992-1998. Data source EA.

Median mean and maximum values for the period are shown in table 49. Despite the occurrence of high concentrations, especially in the Parrett region (Brean) mean values for the period do not exceed 10µg l^{-1}, although during the period annual averages are up to 13µg l^{-1} for the Parrett during the early 1990's.

Table 49. Mean, median and maximum concentrations of chlorophyll *a* in regions of the Severn Estuary 1992-1998

Region	Chlorophyll *a* µg l^{-1}		
	Mean	Median	Max
Mid-Severn	4.7.	3.3	55
Parrett	9.7	6	92
East-Severn	7.8	6	52

Recent (2001) values for chlorophyll *a* in freshwaters entering the pSAC are shown in figure 73. Highest concentrations are recorded in the Avon with a maximum of 104µg l^{-1} in June (note also that relatively high nutrient levels were recorded here). Elevated chlorophyll *a* concentrations (presumably freshwater phytoplankton species) also occur in several of the rivers and streams throughout the various catchment areas. Potential effects to the pSAC would be due to die-off of these species (release of DOC) if they entered more saline waters in any great numbers.

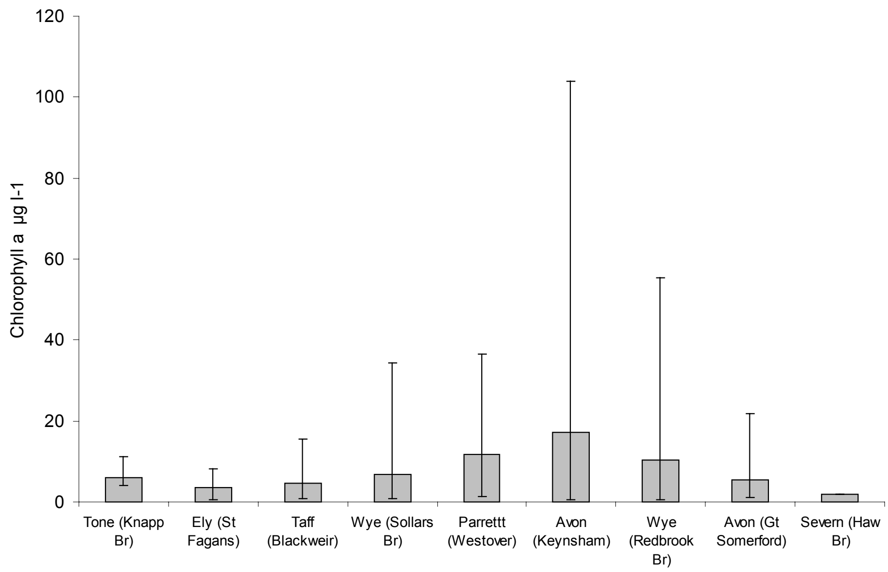


Figure 73. Mean annual chlorophyll *a* concentrations at HMPs or similar in the Severn Catchment. Error bars show minimum and maximum values. Data are for 2001 except Severn Haw Br - 1999. Data source:EA

Examining data for individual sites (1980-1997) in the helicopter surveys, Ellis (2002) found highest chlorophyll *a* concentrations in the inner Estuary, upstream of Sharpness (mean values up to 25µg l^{-1}). Such concentrations are indicative of plankton blooms. Ellis also showed that inner estuarine sites have consistently failed to 'comply' with a 'proposed' EQS (<10µg l^{-1}) since 1980.

7.2.7 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 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 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'. Clearly, they are not of much relevance with regard to benthic biota of the Severn Estuary.

Two principal methods are used by the EA for quantifying turbidity in the Severn Estuary pSAC: suspended solids (units mg l^{-1}, at 105°C), and light scattering, measured using a turbidimeter calibrated with Formazin (units Formazin Turbidity Units, FTU). Data for the tidal waters and discharges are largely recorded in terms of suspended solids, therefore for the purposes of this report this determinand has been used to assess turbidity in the Estuary.

Median values for regions are in the range 81-336mg l^{-1} with the highest median value for the mid-Severn region (figure 74). Maximum values occur in the Avon and east Severn regions (consistent with the findings of Parker and Kirby, 1982) where values up to 72540, and 3800mg l^{-1} occur, respectively.

To put these levels into some perspective, Cole *et al* (1998) cited typical annual values for mean suspended solids (105°C) around the English and Welsh coast as 1-110mg l^{-1}, and suggested that anything >100 mg l^{-1} could be considered high. Annual average values for the mid-Severn region are in the range 143-1225 mg l^{-1}.

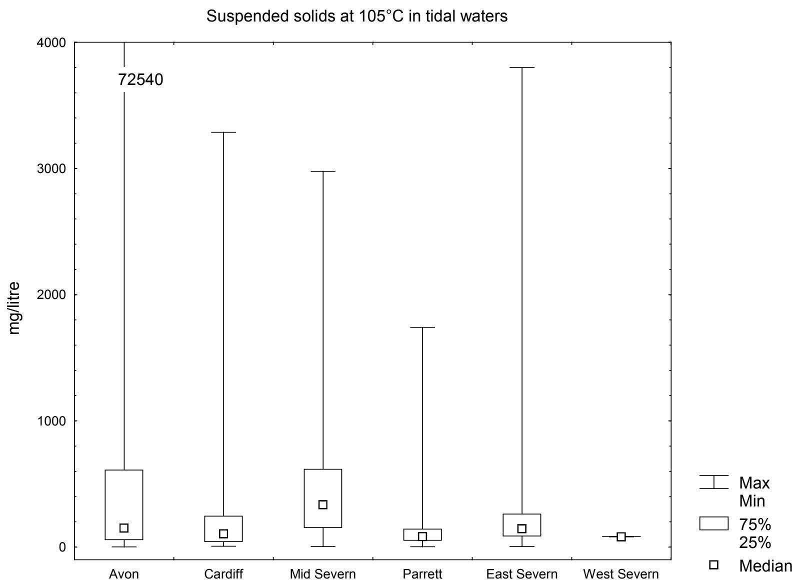


Figure 74. Suspended solids (105ºC) in tidal waters for different regions of the Severn Estuary. Summary for the last decade. Data source EA.

The Severn is naturally a highly turbid Estuary due its to the physical shape, tidal regime and flow rates (see section 5.2.4). The principal source of turbidity is often quoted as being sediment resuspension (Parr *et al.,* 1998) and peak levels are generally confined to a discrete area in the mid-upper reaches of the system, which moves up and down with the tide (Cole *et al*., 1999). The level of suspended solids depends on a variety of factors, including: substrate type, river flow, tidal height, water velocity, wind reach/speed and depth of water mixing (Parr *et al*., 1998). For the tidal waters of the Severn Estuaries pSAC, peak turbidity levels (the turbidity maximum) occupies the whole of the Estuary, east of Bridgwater Bay (Dyer, 1984). However, suspended solids in discharges may contribute to this load locally.

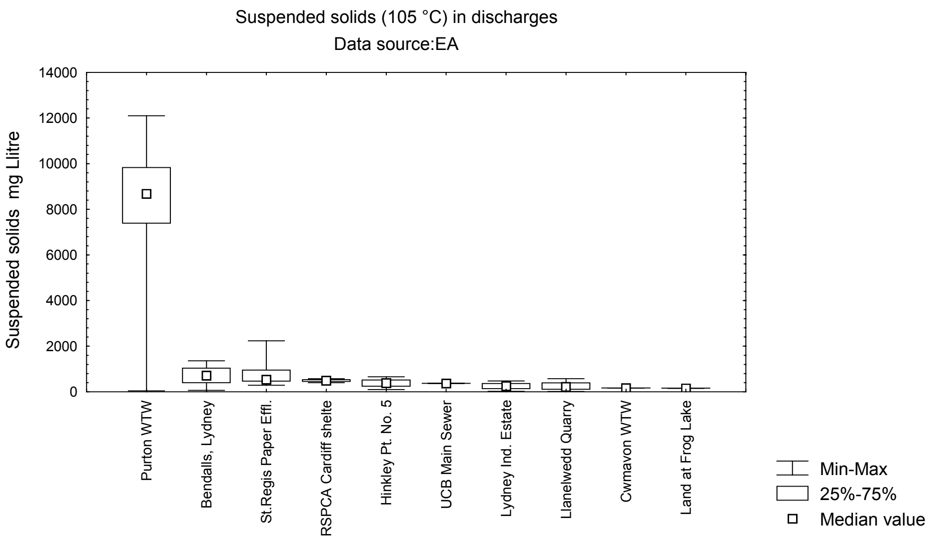


Figure 75 Suspended solids (mg l^{-1} @105°C) in discharges (10 highest concentrations) to the Severn Estuary 2001. Data source :EA

Data for suspended solids in discharges are summarised in figure 75 (*Again it is stressed that these figures represent concentrations and not loadings*). Of these, trade effluent from Purton WTW stands out as a major contributor of suspended solids in discharges to the Estuary (mean, median and max 7893, 8680 and 12100mg l^{-1}, respectively). Concentrations of suspended solids in this discharge have been rising over the last decade (figure 76), although the discharge volume has been reduced by approximately a third from 2000 to 600 cubic metres per day, therefore the loading has not increased.

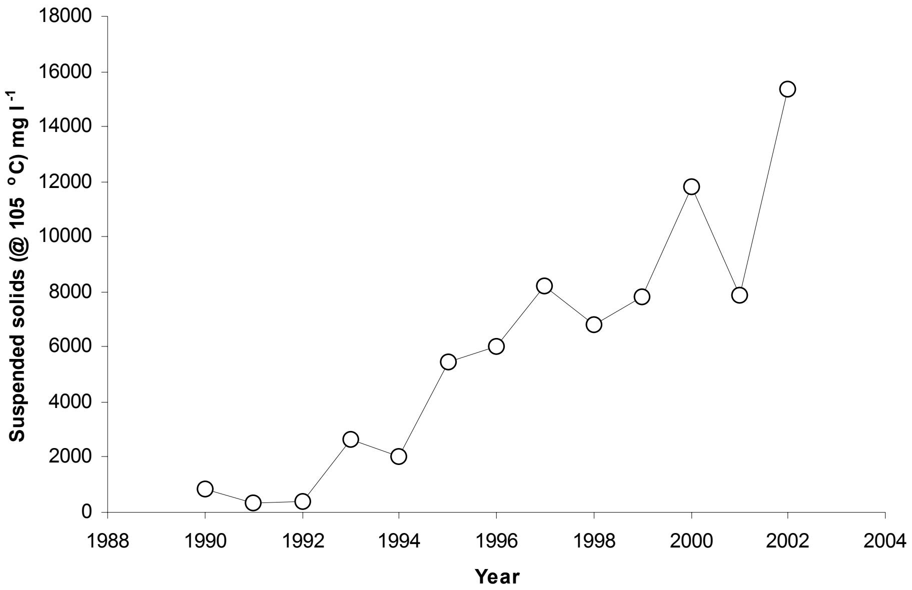


Figure 76. Temporal trends for mean annual suspended solids concentration in trade effluent (process water) from Purton WTW. Data source :EA.

Biodegradation of organic particles can exert an oxygen demand on the sediment, reducing available oxygen to infaunal animals and changing many of the chemical processes within the sediment. Deposition of organic particles may also increase the load of toxic substances to the sediment, as many of these are associated with organic particles (Cole *et al.,*1998).

8. SEDIMENT STATUS AND QUALITY STANDARDS

8.1 Metals

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

The energetic environment of the Severn Estuary has a significant influence on metal distributions. In effect the strong tidal mixing and formation of fluidised bed layers produces considerable homogeneity, dispersing many contaminants from their source and resulting in shallower pollution gradients than might be anticipated. This can be seen in the maps for Hg, Cd, As, Cr, Cu, Pb and Zn in figures 77 and 78.

At present there are no environmental quality standards for sediments applicable in the UK. However, several guidelines on sediment quality are emerging, and CEFAS has cautiously recommended the Canadian/US effects-based approach (CCME ,1999; Long *et al*., 1995). Threshold Effects Levels (TELs - affecting the most sensitive species) and Probable Effect Levels (PELs - likely to affect a range of organisms) are derived from published toxicity data for a variety of substances in sediments (laboratory and field exposures). TELs are proposed as an Interim Sediment Quality Guideline (ISQG) value. As yet these guidelines have not been validated in the UK, though for many List I substances of the Dangerous Substances Directive, a ‘standstill’ provision applies whereby the concentration of the substance in sediments

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

(and organisms) must not increase with time. Sediment quality is also important under the remit of the Habitats Directive (attainment of Favourable Conservation Status - FCS) which may require improvements to sediments at the site in order to secure long-term sustainability.

There are a number of caveats to the application of guidelines, as discussed by Grimwood and Dixon (1997) in the context of List II metals, including possible fundamental differences in sediment geochemistry (discussed in the footnote on the previous page) and the use of test species which are not indigenous to the UK, in deriving thresholds. Nevertheless, in the absence of any UK standards, interim guidelines adopted by Environment Canada (CCME 1999; see Annex 5) serve as a rough indication of the risk to biota from sediment contaminants and identify instances where efforts should be made to minimise further inputs of these substances to the pSAC.

The status of metals in sediments from the Severn are, as indicated above, based on MBA's own published and unpublished records. These data have been collected at intervals over the last 30 years in connection with various research projects and were not intended as a monitoring programme. 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 (without further normalisation – see footnote) and are based on concentrated nitric acid digests (Langston *et al.,* 1994a,b).

With the possibility of biological effects in mind, we have summarised data for metals in inter-tidal sediments from around the Severn Estuary pSAC by classifying sites 'according to the interim sediment guideline criteria for each metal (figures 77 -78). Green bars denote sites where no harm to the environment is predicted (below ISQG's), grey bars denote sites where effects cannot be excluded (between ISQG's and PEL's) and red bars represent sediment concentrations where harmful effects are expected (above PEL's).

Figures 77-78 emphasise the rather homogenous distribution of metals in fine sediments throughout the site. There are few sharp gradients or step changes (with the possible exception of Cd). In sediments of the pSAC all of the seven metals are, to some extent (usually in the mid-Estuary), in concentration ranges where adverse effects to biota cannot be excluded and at some sites harmful effects would be expected (red bars).

Arsenic and copper sediment concentrations generally fall between ISQG and PEL values i.e. within the range where effects ' cannot be excluded' (designated as grey bars in figure 77) rather than where effects 'would be expected'. Concentrations of As at two of the sites are within the ISQG guideline values, according to CCME criteria (green bars).

Many Cd values, particularly in the outer Estuary, fall below the ISQG value (green bars in figure 77), and all are below the PEL. Sediments from a substantial set of sites, particularly in the mid section of the Estuary, around Avonmouth, Cardiff and Newport, fall between these values (grey bars –'effects cannot be excluded').

Unusually, Cr levels are generally lowest in the inner Estuary where most values fall below the ISQG value (green bars in figure 78). Slightly higher concentrations occur at a number of sites in the mid-outer Estuary (grey bars – biological effects cannot be excluded), though none exceed the PEL value where effects would be expected.

The majority of Hg and Pb values fall within the 'grey zone' though at two sites, in the Cardiff area, Pb values fall above the Probable Effects Level, albeit by a small margin (figure 78). There is a higher incidence of Zn concentrations above the PEL, particularly in the mid-section of the Estuary. These values are widely spread and clearly 'contamination' by Zn is effectively widely spread throughout the Estuary as a result of a) the dispersal of fines, and b) possibly a major component from the atmosphere (Owens, 1984). Again, however, values exceed the PEL by a relatively small margin (usually less than a factor of two), rather than by orders of magnitude.

Thus, sediments at a number of sites in the Severn pSAC contain levels of metals which are likely to exert pressures on biota, though these are probably chronic rather than acute.

The fact that a significant number of metals in sediments sometimes exceed baselines (i.e. at those sites represented by grey or red bars) presumably signifies both an anthropogenic as well as geogenic component to the sediment loading. As far as we are aware no attempts have been made to distinguish these components on a catchment scale, though clearly this would be a useful exercise in relation to evaluation of contaminant sources. Though not as heavily mineralised as parts of Devon and Cornwall there are geological anomalies within the Severn basin (e.g. Cd in the Mendips). 'Fingerprinting' catchment sediments would help to establish true baselines.

Based on a relatively small sample set supplied by the EA, Ellis (2002) also concluded that Severn Estuary sediments have exceeded the Environment Canada guidelines for As, Cd, Cr, Cu Pb, Hg and Zn on various occasions since 1990. In the latest analysis (1997) only As, Cd and Hg were above the recommended limits. However, it is not known whether these represent surface sediments, specific size fractions, nor is the method of digestion given. Furthermore, the usual caveats apply regarding the ecological significance of 'non-compliances'. More recent evidence is needed to harmonise the monitoring strategy and to establish temporal trends and for sediments. This is seen as a particularly important issue in terms of meeting standstill requirements for sediments under the Dangerous Substances Directive, and attainment of Favourable Condition (Habitats Directive), and may partially drive the requirement to minimise further inputs via aqueous discharges.

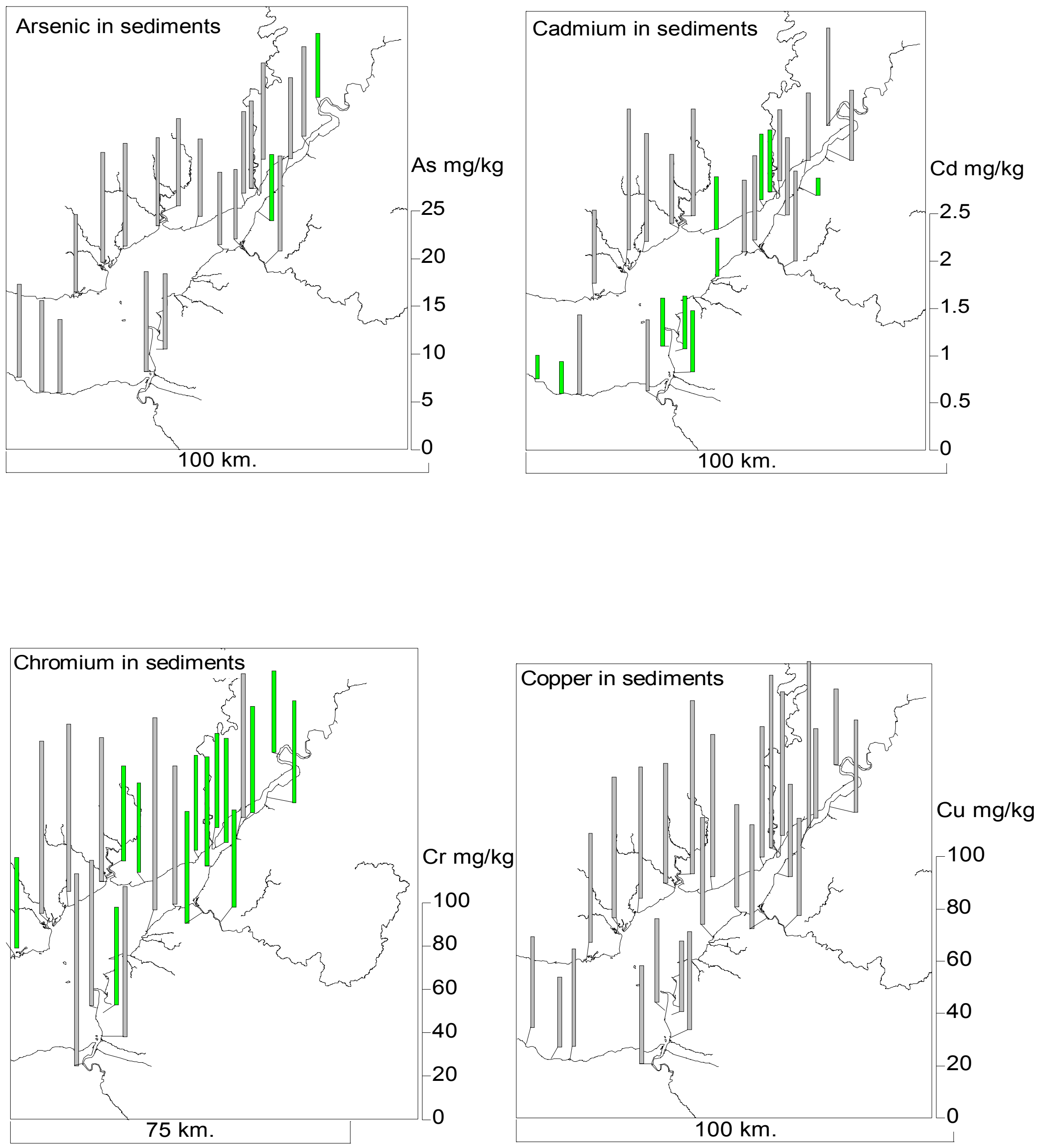


Figure 77. Arsenic, Cadmium, Chromium and Copper in sediment. Classification of the Severn Estuary pSAC based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected.

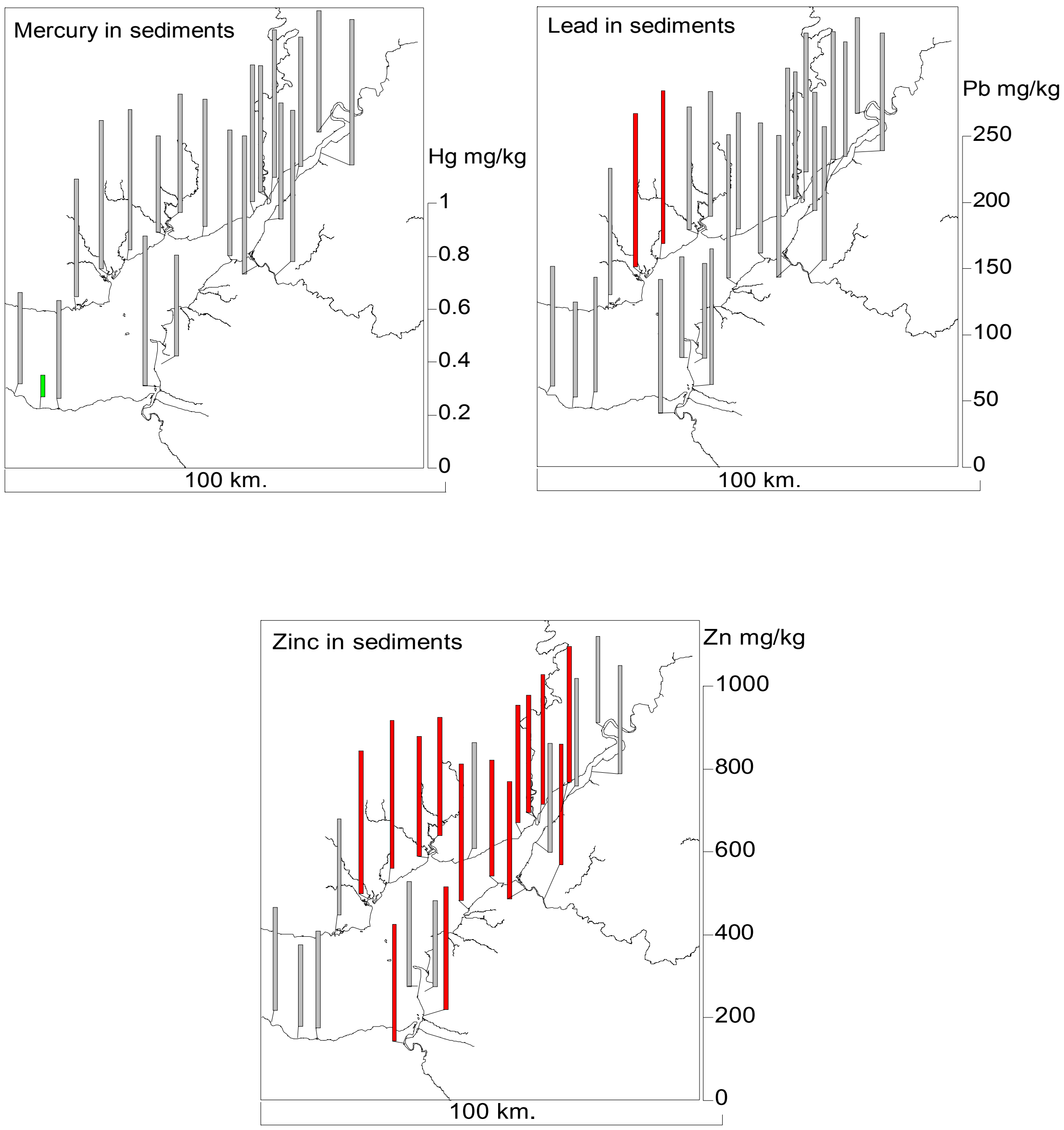


Figure 78. Mercury, Lead and Zinc in sediment. Classification of the Severn Estuary pSAC based on interim marine sediment quality guidelines (ISQG's) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected.

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

Unfortunately the EA data-base, as supplied, contained insufficient information on PCBs, PAH's, pesticides, (e.g. DDT, DDE, dieldrin) triazine herbicides and organotins to give a realistic impression of distribution and sources of contamination in sediments of the pSAC. If no such data exists it is recommended that future sampling programmes incorporate sediment contaminants.

The NMP surveys of sediments in the early 1990s highlighted that PCB concentrations in the Severn Estuary were amongst the highest in the UK. Not surprisingly, the highest reported levels of PCB 153 – up to 25µg kg^{-1} median value - were in an area close to where PCBs were manufactured in the past. The quality guideline (ISQG) for Arochlor 1254 (a mixture of more than 200 different PCB congeners) is 63µg kg^{-1} and the probable effect level (PEL) is 709µg kg^{-1}. For total PCBs the thresholds are 21.5 and 189µg kg^{-1} (see section 5 for further discussion of published studies).

PAHs were also observed by NPMMG (1998) to be relatively high in the Severn Estuary with concentrations for total PAH occurring within the range 1000-10,000 µg/kg dry weight. Elsewhere in the UK concentrations ranged from barely detectable in offshore sands (27-488µg kg^{-1} dry weight around Scotland) up to 35,400µg kg^{-1} in the Tyne Estuary. One site in Milford Haven contained an extreme value of 93,000µg kg^{-1} in 1996, no doubt due to the combined presence of the oil terminal and the Sea Empress spill that year. PAHs concentrations in the Severn often exceed ISQG and threshold effects guideline values and occasionally the PEL criteria (see section 5 for further discussion of published studies).

9. MODELS

Ecosystem models: Techniques for numerical modelling of water quality in the context of ecosystem change were studied by IMER, Plymouth, in the late 1970s and early 1980s in response to proposals to build the Severn Barrage. GEMBASE enabled preliminary simulations to be made of the levels of primary and secondary productivity that might be expected in pelagic environment, together with their subsequent seasonal and annual variability, following barrage construction (Radford, 1979, 1981; Radford and Joint, 1980). There are related papers on validation (Radford & Ruardij, 1987) and thoughts on how models may be useful to minimize monitoring (Radford & West, 1986). This work highlighted the need for further developments particularly concerning the supply and redistribution of sediment and how resultant benthic communities might develop and respond. The continued development of numerical modelling of environmental quality and the ecosystem, using for example the estuarine simulating shell ECoS (Harris, *et al*., 1991) still remains a priority.

Water quality models: An early simulation of the distribution of Cd in the Severn Estuary was performed using the HYDROBASE model (Radford *et al*., 1981). This used salinity data to develop a simple steady-state diffusion model capable of predicting the average distribution of any conservative material discharged from a point source. The same model enabled salinity distributions to be predicted for any given river flows, and was therefore used to simulate the effects of a future Severn barrage. It is interesting to note in this context that the likely impact of a proposed barrage, and accompanying restriction on flushing, would be to increase concentrations of conservative elements such as Cd and Ni by 56 and 65 %, respectively. If similar increases were also observed for Zn and Pb it is doubtful that this would result in any EQS non-compliance for this particular group of metals. In contrast, the standard for Cu would probably be breached (Owens, 1984).

There have also been attempts to predict the axial distribution of ^{137}Cs in relation to salinity: the observed distribution of ^{137}Cs resulting from known sources in the Severn Estuary, permits the testing of simple predictive models for the one-dimensional distribution of non-conservative substances in the Estuary. It generalizes previous formulations and dispersion coefficients to permit variation of the net runoff with position along the Estuary (Uncles, 1979; Rattray and Uncles 1983). More wide-ranging models which predict levels of artificial radionuclides in different environmental materials have been described by Coll, 1988.

On behalf of the Agency, HR Wallingford have carried out simulations of discharges from the Avonmouth STW. These studies assess the impact of continuous secondary treated effluent and storm discharges on faecal coliforms in designated bathing waters at Clevedon (HR Wallingford, 1999). Additional work was also performed to assess the dilution of contaminants in the discharge. These employ HR Wallingfords TELEMAC-2D model of tidal currents in the Severn and the PLUME-RW model to look at pollutant dispersal. The model represents pollutant discharges as regular releases of discrete particles each representing a defined quantity of pollutant which can decrease with time representing decay processes such as bacterial mortality or adsorption of contaminants on to particulates. Simulations of a model contaminant (set initially at 0.288mg l^{-1}) indicated that dilutions of 10^4 (~0.03 µg l^{-1}) occurred at approximately 12km to the NE and 20km to the SW of the discharge point, on spring tides. During neaps an equivalent plume might extend approximately 8km to the NE and 14 km to the SW.

Zeneca Ltd (formerly ICI) had previously (1974) developed a simple one-dimensional simulation to show that the effluent from their Severnside plant, at Avonmouth was diluted sufficiently 'to ensure that there should be little or no impact on estuarine quality'. This was re-developed in 1989 to predict tidal water movement and dilution of the discharges, and hence to enable users of the site to identify possible zones of impact (see section 5.1.1). Ammonia was considered a principle concern and the required dilutions to meet the EQS of 21 $\mu g\ l^{-1}$ unionised ammonia were estimated at 240 under normal conditions and 1500 for the maximum consented discharge. The model was used to predict the mixing zones outside which such dilutions would be achieved. At spring tides the impact zone was estimated at 3.2Km to the NE and 0.3Km to the SW assuming x240 dilution requirements under typical discharge rates, and 6Km to the NE and 7.2Km to the SW assuming x 1500 dilution (if maximum consented amounts were being discharged). At neap tides plumes would be shorter - extending 0.4 Km to the NE and 0.3Km to the SW (x240 dilution), and 3.6 Km to the NE and 5.4 Km to the SW (x1500 dilution).

Both Zeneca and HR Wallingford models indicate a fairly thin plume running parallel and close to the English shoreline.

Further insights into water quality and hydraulic modelling in turbid systems, based on the Severn Estuary can be found in Larsen *et al.*,1992; Osment , 1992; Allen, 1988, and Suckling and Ryrie,1990.

In an attempt to investigate the role of particulate BOD (slowly degrading detritus) on the oxygen balance of the Parrettt Estuary, Maskell (1985) applied a one-dimensional, cross sectionally averaged, mathematical model using data from field surveys carried out by the Hydraulic Research Station during spring and neap tides. The model was calibrated to simulate the processes of tidal propagation, saline intrusion and sediment transport during a period of steady low fluvial flows. Simulations included various tidal conditions and effluent flows. The model predicted that the BOD of naturally occurring marine detritus in the Estuary which is periodically resuspended, would be more than the BOD due to dissolved and particulate effluent discharges to the Estuary. DO levels were predicted to fall below 60% saturation during spring tides.

The major source of detritus was considered by Maskell (1985) to be dead phytoplankton from the more turbid parts of the Severn Estuary. However, modelling carbon flow in the Severn, Randerson (1986) considered phytoplankton production to be relatively low and estimated that more than 90% of autochthonous carbon inputs to the Estuary could be attributed to *Spartina* marsh and mudflat systems. Allochthonous sources of organic carbon were considered by Randerson (1986) to arise principally from sewage discharges.

10. CONCLUDING REMARKS: SEVERN pSAC

10.1 Biological Status

The exceptional tidal range and classic funnel shape make the Severn Estuary unique in Britain and rare world-wide. Large tidal-currents are a dominating feature providing a mechanism for transport of particles; frictional stresses result in variations in bed-types (and associated contaminant loadings), which in turn dominate the distribution of benthic macrofaunal communities in the pSAC. In particular, variations brought about by changes in sedimentation patterns over the spring-neap tidal cycle (deposition of fine sediment in the upper reaches during neaps – erosion of fines and deposition of coarser particles during springs), and seasonal influences thereon, are likely to be important modifying factors for biota.

Generally, reports suggest that the Severn as a whole supports a relatively impoverished fauna and flora. High turbidity means that algal productivity is low, though organic carbon may be enriched and BOD high in fluid muds. These may disperse at spring tides to produce DO sags. In those areas frequently covered by turbid layers colonisation, by filter-feeders in particular, is likely to be sparse. Much of the sub-tidal Severn mud is impoverished and even some sandy areas may be impoverished because of extreme mobility of silts at spring tides[1]. The fate of the large quantity of sediment that is resuspended and recirculated on each tide will clearly play a major role in the mobility, bioavailability and impact of associated contaminants.

Reports based on the numbers and diversity of fish entrained on power station intake screens suggest that there is a long-term trend of increasing species richness, with warmer water species becoming more abundant. However the abundance of several fish species (Twaite shad, eel, pout, hake, poor cod, thornback ray, plaice and sea snail) is apparently declining in the Severn. Reports indicate that several contaminants (organochlorine compounds, PCBs, metals) have been found in the tissues of fish from the Severn, however empirical evidence for the impact of such contaminants is remarkably difficult to elucidate. Concerns have been expressed that the environmental quality of the Estuary could affect the passage and survival of migratory fish, particularly shad. Twaite shad, which spawns in the Wye, Usk and Severn, are reported to have been lost from several rivers (Environment Agency, 1999). Intuitively, larval and juvenile stages are likely to be the most contaminant-sensitive phases. Shad in the Severn Estuary may also be vulnerable to sediment contamination either directly, or through their diet of benthic invertebrates and fish. An alternative concern is that the sensitive olfactory responses of anadromous fish could be disturbed by metals and other contaminants. There is presently no information on effects and levels of contamination in young shad in the Estuary (Bird, 2002) and until further evidence is produced, causes of declining migratory fish remain speculative. A monitoring programme, perhaps using individuals entrained on water-intake screens could be used to establish tissue concentrations of metals and organic contaminants.

It has been suggested that numbers of the common eel (*Anguilla anguilla*) may continue to decline without vigorous conservation effort. Again, the impact of

[1] It is interesting to note that by reducing turbidity a future Severn Barrage, if constructed, would theoretically increase primary productivity and the diversity of bottom fauna)

contaminants on Severn eels has not been studied extensively but could be useful in determining sub-lethal effects. A study on metallothionein induction and 'metal fingerprinting', for example, has been shown to provide valuable information about the ecotoxicological status of eel populations, with regard to metals, in the Thames Estuary (Langston *et al.,* 2002; see annex 6). Studies on EROD induction in the same samples might provide a useful indication of the impact of organic contaminants (see, for example, Rotchell *et al*., 1999; and section 10.2, below).

Warwick *et al* (1989) noted that populations of several invertebrate species were dominated by smaller individuals in the Severn than in other SW estuaries, suggesting a shorter life-span. Again, it is not clear whether is be attributable to the harsh environmental regime of the Estuary or other factors. Concentrations of some metals (Ag, Cd, Hg) are relatively high in some Severn invertebrates compared to UK baseline values and local effects on macrofauna have been described for sites close to discharges around industrialised Avonmouth. Although deleterious effects do not appear to occur widely, metal bioaccumulation could provide an exposure route for organisms higher up the food chain such as fish and birds (see section 5.1.1).

Diversity indices

Diversity indices are frequently used to assess the effects of environmental degradation on the biodiversity of natural assemblages of organisms. MPMMG (1998) included a summary of principal univariate statistics to examine biological diversity within benthic communities at coastal and estuarine sampling sites around the UK including the Severn Estuary:

Abundance - Contaminated sites (such as the Tyne and Wear) contained highest densities (>10000 m^{-2}) dominated by a restricted range of principal species (*Polydora*, *Capitella* and *Ophryotrocha*), probably indicating the effects of organic enrichment. The Severn and the Solway contained the lowest abundances (<6100 m^{-2}) compared with the average abundance for UK estuarine sites in the survey (6449 m^{-2}).

Species richness is indicated by the number of taxa per $0.1m^2$, and the pattern across UK estuaries was similar to that of abundance with the Severn and Solway containing the fewest species.

Shannon-Weiner diversity index expresses the relationships between the occurrence of species and the apportioning of individuals among those species. The Severn estuarine sites again scored very low (~1), as did sites further out into the Bristol Channel. Evenness is a complementary measure of the allocation of individuals across species: low values are associated with samples from sites numerically dominated by only one or two species which is generally indicative of stressed communities. The evenness statistic for the Severn was ~0.5, close to the mean of 0.503 for estuarine sites. For the Bristol Channel sites the score was ~0.5, compared to the average of 0.738 for coastal waters.

It is evident from these indices that biodiversity in the Severn Estuary is relatively low in comparison with other UK estuaries. However, MPMMG acknowledges that low diversities at sites such as the Severn emphasises the effect of the tidally dynamic environment.

Abundance Biomass Comparison - Warwick *et al.,* (1989) used the Abundance Biomass Comparison (ABC) method (Warwick *et al.,* 1986) to analyse faunal distribution and community structure in several estuaries, including the Severn in an effort to establish whether observed patterns resulted from the effects of natural environmental variables, or whether they were affected by some unnatural disturbance such as chemical pollution, organic enrichment from sewage, frequent bait digging etc. This method depends on the fact that the distributions of biomass among species in marine macrobenthic communities show a differential response to disturbance, which can be demonstrated by the comparison of k-dominance curves for abundance and biomass. In the Severn, seven out of the twelve sites investigated appeared to be completely undisturbed. Sand Bay (upper and mid), and Clevedon (S) were classified as moderately disturbed, whilst Lydney and Frampton in the inner Estuary were grossly disturbed.

Thus, the pSAC is generally characterised by its relatively low biodiversity which, on current evidence, is attributable to its unique physical conditions. The large tidal-currents which dominate the Estuary create a relatively challenging environment for faunal communities and play a large part in determining the abundance and distribution of benthic macrofaunal communities in the pSAC.

Oyster embryo assays

As one means of assessing biological impact MPMMG have conducted oyster embryo assays which essentially involved the collection, (twice annually, summer and winter) of surface water samples from estuaries in the UK. These were incubated in the laboratory with a fixed number of freshly fertilised embryos of *Crassostrea gigas*. The developing embryos integrate the adverse effects of many contaminants in the sample and if the water quality is poor, they fail to develop normally to D-shaped larvae. Severn water was tested in this assay for three consecutive years between 1993 and 1995, and the percentage net adverse response was low, but increased every year culminating with 10% embryos failing to develop normally in 1995. Failure to reach the normal larval state can be caused by everything from lethal effects to subtle interferences with embryonic development, but the endpoint is a simple expression of water quality in comparison to a clean reference sample of seawater. Thus water quality of the pSAC appeared to be relatively good, although the gradual deterioration in 'biological condition' should be monitored carefully using this and other bioassays (see annex 6). No specific compound is implicated as being responsible for the (relatively low) failure rate in oyster larvae, although, in reality, it is probably due to a combination of contaminants and not one single compound (see below).

Application of the classification scheme drawn up by the National Water Council depicts most of the pSAC as being good or fair, with a small number of sites near urban conurbations (Cardiff, Weston) being of poorer quality. However this is a generalised subjective scheme (based on biological quality, dissolved oxygen and aesthetics) and would be insufficiently sensitive to reflect all but the most dramatic change in water quality. Hence the need to examine chemical data more systematically.

10.2 Chemical Status

Because of the dominance of physical forcing agents in the pSAC, and sometimes limited chemical data, there is little evidence available to indicate modifications to biota due to contaminants, except near large discharges. A major challenge for the future is therefore to establish more subtle, reliable and pragmatic means of assessing such impact (discussed in greater detail in section 10.3). Until more appropriate biological effects monitoring is put in place to answer this challenge it is only possible to assess, subjectively, possible impact of individual contaminants, or groups of contaminants, based on best available evidence. Some of the major water and sediment quality issues are discussed below and briefly summarised in table 50.

Table 50. A Summary of Water and Sediment Quality issues in the Severn Estuary pSAC. (Findings for each of the numbered 'contaminant categories' are explained in more detail in the accompanying text).

'contaminant'	Area	Potential Sources	Most vulnerable features/biota
1) Organotins (TBT, TPT?)	Poorly defined. Probably localised to the mouths of sub-estuaries, embayments and docks eg Cardiff, Newport, Avonmouth	Shipping? STWs? Docks? Sediments?	Molluscs (primarily gastropods, also some bivalves
2) Metals (As, Cu, Cd, Hg, Zn, and perhaps Ag)	Mid-upper Estuary	Trade discharges (nb Avonmouth), major STWs and rivers, landfill, sediments	Invertebrates (primarily molluscs and crustaceans), species composition, larval fish and birds
3) Nutrients, DOC, (Chlorophyll *a*)	Localised – inner Estuary, embayments, sub-estuaries n.b. Avon and east Severn regions	Sewage and trade discharges Diffuse sources	Phytoplankton, Invertebrates, fish (estuarine and migratory, esp. early life stages), seabirds, mammals, *Zostera* bed (and associated fauna).
4) Low DO	Localised -inner estuarine areas: Severn, Usk, Parrett, Wye	O_2 demand in sediments, perhaps enhanced by discharge outfalls. Also related to freshwater flows	Migratory fish, invertebrate communities
5) Hydrocarbons: PAHs, - eg benzo(a)pyrene, phenanthrene	Poorly defined, probably localised– embayments, sub-estuaries	coal mining, fossil fuel combustion, run-off, discharges, sediments	Benthic invertebrates and fish (NB those in contact with sediment)
6) Pesticides and herbicides eg. DDT lindane, dieldrin, endosulphan,triazines, other endocrine disruptors	Poorly defined, probably very localised	Rivers and occasional discharges sediment	Invertebrates (esp. crustacea), fish, sea birds and mammals, species composition, *Zostera*
7) PCBs	Localised – sub-estuaries eg lower Usk	Historic, industrial, Sediments	Benthic invertebrates and food chain bioaccumulation
8) Radionuclides	Cardiff	Pharmaceutical discharge; sediments	infauna, demersal fish

1) Organotins

Currently there is very little evidence of impact. Data on organotins in the Severn are minimal and sources, sinks and biological effects need further study. Analysis of dredge spoils in and around the pSAC suggest there may be localised reservoirs of TBT near major conurbations such as Newport and Cardiff, and presumably elsewhere. Processes including physical resuspension and bioturbation could remobilise these sinks. Furthermore, TBT in such contaminated sediments is likely to be available and potentially harmful to deposit-feeders and infauna.

2) *Metals*

The Severn receives the largest input of Cd of all UK estuaries, principally from the smelting industry at Severnside, where for much of the 1990s two of the UKs largest discharges (one sewage and one industrial) were situated. Significant quantities of other metals (Zn, Cu, Hg, As) are also discharged from these and other sources in the area. There are indications that environmental concentrations of Cd may have decreased in recent years. This broadly agrees with trend data on Parcom loadings estimates. There have been no major changes observed for other metals. EQS values are sometimes exceeded in the mid-upper Estuary and continued surveillance is therefore recommended.

Metals in sediments exceed sediment quality guidelines widely, though exceedences of probable effects levels are rare. It would be useful to distinguish between anthropogenic and geogenic components and to assess their relative significance, by comparing 'elemental fingerprints' with sediments from appropriate catchments (upstream of anthropogenic sources).

Bioindicator studies with species such as worms and clams have demonstrated significant bioaccumulation of silver, cadmium and mercury, above normal, presumably of anthropogenic origin. Mussels in the Severn Estuary also contain some of the highest levels of Cd in the UK: in the 'upper' range of guideline values in the OSPAR classification scheme. In the past, elevated body burdens in invertebrate species have been reflected in residues in birds and other predators near industrial centres of the Severn. More work needs to be done to establish the physiological and ecological significance of these body burdens and to confirm any temporal improvements. Some of the sub-lethal indicators described in Annex 6, notably metallothionein induction, may provide more sensitive measures of impact than recording the presence or absence of species and may also give indications as to which chemicals are most responsible for impact. Those metals currently perceived as being most important, toxicologically, are Cd, Cu , Zn, As, Hg and perhaps Ag.

3) *Nutrients*

Algal growth in oligotrophic estuaries is characteristically restricted by P towards the freshwater end-member grading towards nitrogen limitation at the seaward end. Nutrient levels are high in the Severn Estuary (MPMMG 1998 and data presented in this report), however, the lack of light resulting from the large suspended sediment loadings is suspected of limiting algal growth. Thus nutrient enrichment alone does not signify eutrophication and has to be considered alongside chlorophyll-a measurements (as an indicator of primary productivity). The latter are not excessive within the pSAC itself, though the limitations imposed by suspended solids presumably may not apply further out into the Bristol Channel. There is evidence of localized nutrient enrichment, coinciding geographically with areas where suspended solids are high and DO is low (Avon, east-Severn, Parrett). Indications are that point source inputs may contribute significantly to this problem.

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 behaviour, transformations and fluxes are also affected by environmental conditions, particularly low O_2. Thus as SOD and BOD levels vary across the seasons, 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 become 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.

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

4) *Dissolved oxygen*

Oxygen sags have been reported intermittently, both in the upper 20km of the Severn Estuary itself and in the estuaries of several of the principal rivers opening into the Severn (Usk, Ebbw, Parrett and Wye). These probably originate from the high densities of particulate organic matter of natural and anthropogenic origins, which oscillate tidally within the Estuary, but are focused hydrodynamically in the low salinity region.

Low DO levels can have a significant impact on migratory fish; due to the energetic demands of the upstream migration and the increased activity that takes place at spawning, the oxygen requirements of anadromous lampreys and migratory teleosts are very high at these times in the life cycle (Claridge & Potter 1975). Since the maximum amount of oxygen that will dissolve in water is inversely related to temperature, even in clean turbulent water, oxygen concentrations can fall to below the minimum level for survival. Although River lampreys reproduce in the spring, Sea lampreys and both species of shad spawn in early- to mid-summer when water temperatures are higher and when the amount of oxygen available in rivers is much lower than at other times of the year. Consequently, any factors that lower oxygen tensions will have a significant impact on fish populations at these critical times (Bird 2002) and should be minimized.

5) *Hydrocarbons and PAHs*

A number of hydrocarbon compounds including PAHs are present, locally, in elevated concentrations. Sources include a combination of fossil fuel combustion, shipping, urban run-off, STW and various point-source and diffuse discharges from industrialised areas (e.g. drains near Lydney Industrial Estate and Kingsweston Rhine). Coal bearing strata of south Wales (and associated, historic, mining activity), together with the oil bearing shales exposed on the southern coast of the Estuary (Bridgwater Bay) also make a contribution to PAHs in the Severn system. The relative significance of sources is difficult to quantify and may vary in different areas, though the principal component appears to be anthropogenic.

Moderately high levels of total PAHs appear to be common in sediments but decrease outside the Bristol Channel. For some individual PAHs concentrations occasionally exceed ISQG and PEL criteria.

PAH exposure induces the production of detoxifying enzymes (P450-associated) that are present in many tissues, which results in the formation of hydrophilic metabolites (Stegeman and Hahn, 1994). Levels of one of these enzymes, ethoxyresorufin-O-deethylase (EROD), in fish from the Severn Estuary has been investigated (Rotchell *et al.,* (1999). Eels *Anguilla anguilla* and flounder *Pleuronectes flesus* were both found to show elevated hepatic EROD activity which was partly attributed to seasonal variation. However, the authors also considered that the high levels of PAHs (and PCBs) reported in some Severn sediments were significant and might contribute to the observed EROD activity. PAH (e.g. benzo(a)pyrene) accumulation has also been described in molluscs from the pSAC suggesting that the induction of EROD should be investigated further in relation to bioavailability of these compounds.

The liver is an important site for PAH metabolism and the metabolites thus produced are secreted into the bile and stored in the gall bladder before being excreted (Britvic *et al*., 1993: Varanasi and Stein 1991). Bile metabolites of PAHs have been identified in three species of fish (common eels - *Anguilla anguilla*, flounder - *Pleuronectes flesus*, and conger eels - *Conger conger*) from the Severn Estuary (Ruddock *et al*., 2002). Of the three species investigated, common eels contained the highest metabolite concentrations.

Since sediments contain much higher PAH concentrations than water, species like eels, which spend much of their time buried in muddy sediments are particularly susceptible to PAH exposure (van Schooten *et al*., 1995). Dietary preferences of *A. anguilla* may result in higher exposure to PAHs than the other species (Wheeler 1969).

Thus, hydrocarbon contamination of sediments has implications for benthic marine organisms, and also for the species of interest for which Severn has been recommended as a pSAC: although adult sea and river lamprey are parasitic, feeding on blood and tissues of teleost fish and marine mammals, larval lamprey (ammocoetes) are filter feeders and extract diatoms, algae and other microorganisms from the water overlying their burrows (Moore and Beamish 1973). Diatoms are known to incorporate aliphatic, and to a lesser extent, aromatic hydrocarbons (Thompson and Eglinton, 1976, 1979). The duration of larval life for lampreys depends on the availability of food, but is approximately 5-6 years for the Sea lamprey (Hardisty 1969; Lowe *et al.*, 1973) and just over 4 years for the River lamprey (Hardisty and Huggins 1970; Potter 1980), representing an extended period for possible exposure, both from feeding and directly from sediments.

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

Currently there is very little evidence of impact. Only occasional elevated levels are reported (e.g. DDT in sediments and eels; dieldrin, triazines and endosulphan in water). The general issue here is that there appears to be insufficient data available to characterise the threat from the majority of these compounds. If such threats do occur, these would be expected to be largely localised issues. Rivers probably introduce the largest loadings but there are indications from the EA data that occasional high pesticide concentrations can occur in some discharges. Allen *et al*., (2000) highlight the lack of information on endocrine disruption in the Severn Estuary, flagging, as a high priority, research into the presence and effects of pesticides and other potential endocrine disruptors.

7) *PCBs*

In the marine environment, PCB contamination is related to past industrial production (e.g. at Newport on the Usk) and usage patterns - coupled with disposal and subsequent transport with fine sediment. According to a report from CEFAS commissioned by the Agency, marine sediments are classified as being 'heavily contaminated' close to river mouths and docks in the Severn Estuary/Bristol Channel, 'contaminated' at offshore sites in the Severn Estuary and Swansea Bay, and 'slightly contaminated' in the centre of the Bristol Channel (Reed and Waldock, 1998). PCBs have been found to be present in freshwater eel tissue at sites throughout Wales, but highest concentrations were found in the lower reaches of industrialised catchments in the south-east (Weatherly *et al.,* 1997). Although other fish from Cardigan Bay and the Severn Estuary do not appear to be seriously contaminated, high concentrations have been found in marine mammals from these locations. This appears to be evidence of high accumulation in predators at a considerable distance from PCB sources.

8) *Radionuclides*

In recent years the disposals of organic tritium from a radiopharmaceutical plant at Cardiff (currently under review) have resulted in anomalously high concentrations of tritium in sediments and benthic biota such as flounders and mussels. The consequences have, to date, not been considered a threat for humans (RIFE 2001). Clearly, however, this is an issue for further research since relatively little is known of the bioavailability, assimilation pathways and effects (e.g. genotoxicity) of organically-bound tritium on marine life.

10.3 Future Research Requirements

For a large number of determinands it is difficult to ascertain with any certainty whether or not they are influencing the site's characteristics and, hence, Favourable Conservation Status[1]. The fragmented composition of much of the available environmental data - much of which has been collected for compliance purposes rather than to provide a rigorous interpretation of nature - prevents all but a first-order approximation of the status of the site. This needs to be addressed in order to progress our understanding of how environmental quality, and in particular anthropogenic inputs, are affecting the status of the pSAC.

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

Traditionally, surveillance of chemical parameters in water has often been carried out by the EA for the purposes of compliance monitoring and is not necessarily intended for survey purposes (or for the type of characterisation being undertaken here). This should perhaps be reviewed in future to maximise the value of the information. Furthermore, available chemical data, mainly supplied by the EA, contain insufficient recent information on e.g. metals, PCBs, PAHs, and pesticides to give accurate impressions of fluxes from rivers and discharges, or the current distribution, sources and sinks of contamination in sediments of the pSAC. It is particularly important that future sampling programmes incorporate more information on sediment contaminants.

Remarkably, there are scarcely any published records of organotin in water, sediment or biota from the pSAC. Since organotin compounds can arise from various sources in addition to shipping, and in view of their well-documented endocrine disruptive effect, the distribution and impact of butyl and phenyltins may require further assessment.

For metals, the use of validated suite of indicator species, at a selection of key reference sites, is clearly a useful way of estimating bioavailability around the system. Valuable baseline information is already in place regarding the (historic) distributions of many of these species and their metal burdens. The current status of the SAC needs to be defined in similar fashion, and at intervals in the future, to ensure bioavailability does not increase. The same organisms may also be useful for the surveillance of organic pollutants and should be tested accordingly.

Metal concentrations in sediments presumably consist of both anthropogenic and geogenic components which have yet to be distinguished. Methods are becoming

[1] Favourable Conservation Status (FCS). The Habitats Directive requires the site to acheive a condition where species/habitats are sustainable in the long term. EN has produced Favourable Condition tables to aid this process, which encompasses a number of attributes, including the extent and biological quality of the interest features (summarised in annex 1 for the Severn Estuary pSAC, SPA).

available which address this problem and the consequences for bioavailability. It is recommended that these be applied to the Severn pSAC.
In recent years, techniques have also been developed to assess sub-lethal biological impact in greater detail, which would allow screening of the pSAC, including possible problem discharges (and leachate from landfill sites), together with diffuse sources in sediments. 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. This would ideally 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 such procedures are used in combination in well-designed survey programmes, they can provide insights into which pollutants are responsible for environmental degradation. They may also be useful in addressing the long-standing problem of additivity/ synergism. A major criticism of many current statutory monitoring assessments, whether using comparisons with EQS values, or some other marker, is that they address only single contaminants at a time. Even if individual chemicals do comply with such values (as most appear to do in the Severn) it does not necessarily mean the environment is healthy. Biological effects may occur if several contaminants act together. Hydrocarbon residues alone contribute a myriad of individual compounds, most of which have additive toxicity. The majority of outfalls contain a particular cocktail of chemicals whose true impact can usually only be assessed through a site-specific evaluation, taking into consideration the interactions that occur between different components. Only by incorporating biological-effects monitoring, alongside chemical surveillance can we make substantial progress towards understanding and managing such complex environmental issues.

A similar integrated approach is equally amenable for measuring long-term trends e.g. in the assessment of recovery. There are examples cited in this review which indicate that environmental conditions are improving (for some contaminants) and other observations which herald biological revival in the Severn Estuary pSAC. It would be exciting, scientifically, and rewarding from a management perspective if these two events could be linked on a sound mechanistic basis. The outcome would provide for more reliable and objective site characterisations in the future.

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* SEVERN ESTUARY DATA SET – reports based on data from Hinckley Point Power Station (see section 6)

12 ANNEXES

Annex 1. Severn Estuary pSAC, SPA: Summary Of The Interest Features

Severn Estuary SPA: Summary of the interest (or qualifying) features, and conservation objectives, (adapted from English Nature and Countryside Council for Wales, 2001)

Internationally important populations of regularly occurring Annex 1 bird species (Bewicks swan *Cygnus columbianus bewickii*)

- Maintain numbers as % of GB and NW European population

Habitat
- Intertidal mudflats and sandflats
  - maintain extent
- Saltmarsh
  - maintain extent
  - maintain >25% cover of soft leaved herbs and grasses (*Agrostis stolonifers, Glyceria fluitans, Alopecururs geniculatus*) during winter

Internationally important assemblage of waterfowl (the Estuary supports over 20,000 wintering wildfowl)

- Maintain numbers

Habitat
- Intertidal mudflats and sandflats
  - maintain extent
  - maintain presence and abundance of prey species
- Saltmarsh
  - maintain extent
  - maintain abundance of soft leaved and seed bearing plants (*Puccinelli maritima, Saliicornia, Agrostis and Atriplex)*
  - maintain vegetation height of <10cm
- Shingle and rocky shores
  - maintain extent
  -maintain presence and abundance of intertidal invertebrates

Internationally important populations of regularly occurring migratory species (including shelduck, dunlin, redshank, European white-fronted goose, gadwall)

- Maintain numbers

Habitat
- Intertidal mudflats and sandflats
  - maintain extent
  - maintain presence and abundance of prey species
- Saltmarsh
  - maintain extent
  - maintain abundance of soft leaved and seed bearing plants (*Puccinelli maritima, Saliicornia, Agrostis and Atriplex)*
  - maintain vegetation height of <10cm
- Shingle and rocky shores
  - maintain extent
  -maintain presence and abundance of intertidal invertebrates

Annex 1. (cont.)

Summary of the interest features for which the Severn Estuary has been recommended as a pSAC

Atlantic salt meadows (*Glauca-Puccinellietalia maritimae)*

- **For which the area is considered to be one of the best areas in the UK**

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

Mudflats and sandflats not covered by seawater at low tide

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

These are intertidal mud and sand sediments on a shore that are exposed at low tide but submerged at high tide. Many sites are important feeding areas for waders and wildfowl.

Sandbanks which are slightly covered by seawater all the time

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

Sandbanks permanently covered by seawater to depths of up to 20 metres below the low water can include muddy sands, clean sands, gravely sands, eelgrass *Zostera marina* beds, and maerl beds (carpets of small, unattached, calcareous seaweed).

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

Estuaries

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

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.

Annex 1. (cont.)

Severn Estuary pSAC: Summary of the interest features for which the Severn Estuary has been recommended as a pSAC.

***Alosa alosa* (Allis shad)**

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

The allis shad is a medium-sized fish of coastal waters and estuaries of the western Mediterranean and north-east Atlantic coasts. It spawns in rivers but has become rare due to over-fishing, pollution and obstructions to migration.

***Alosa fallax* (Twaite shad)**

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

The twaite shad is a fish that occurs in western European coastal waters. It enters lower reaches of rivers to spawn It has become rare due to over-fishing, pollution and obstructions to migration

***ampetra fluviatilis* (River lamprey)**

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

e river lamprey is a primitive jawless fish bling an eel. Confined to western t migrates from the sea to spawn in silt eds of many rivers in the UK. One population in the UK is, however, known to live entirely in prey is absent from some rivers because of pollution and barriers to igration.

***Petromyzon marinus* (Sea lamprey)**

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

The sea lamprey is a primitive jawless fish resembling an eel. It is the largest of the lampreys found in the UK. It inhabits north Atlantic coastal waters and migrates to spawn in rivers. It has widespread distribution within the UK, although populations have declined due to pollution and barriers to migration.

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 Coles *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 Coles *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 streptococci	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 Coles *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 (e.g. Pipe *et al*, 2000; Raftos and Hutchinson, 1995).

EROD (ethoxyresorufin-O-deethylase) is a marker for the activity of the mixed function oxidase (MFO) system, whose induction is usually associated with exposure to, and the detoxification of xenobiotics such as PAHs and PCBs. Occasionally these transformations may produce deleterious side effects due to the formation of carcinogenic or genotoxic compounds (e.g. formation of benzo(a) pyrene diol epoxide from the benzo(a) pyrene). Genotoxicicity assays (see below) may help to establish this possibility

Metallothionein (MT) induction and associated changes in metal metabolism are specifically induced by metals and are sufficiently sensitive to be used to detect elevated levels of bioavailable metal in the field or in arising from metals in discharges (e.g. Langston *et al*., 2002).

Genotoxicity-The Comet Assay - The single cell gel-electrophoresis (comet) assay is ideal for screening for possible genotoxicity associated with point-source and diffuse inputs to the system.

The CAPMON technique - Cardiac activity in bivalve molluscs and decapod crustaceans – Heart rate provides a general indication of the metabolic status of mussels and crabs. The CAPMON technique (Depledge and Anderson, 1990) permits the non-invasive, continuous monitoring of cardiac activity using infra-red sensors attached to the shell.

Tolerance Studies - More widespread investigations of community tolerance to establish their adaptation to contamination levels. Mapping the genetic composition of tolerant populations of individual species (*Hediste, Littorina* and others) in relation to induction of detoxification systems (such as EROD and metallothionein) should also be considered. This could add an interesting temporal dimension to biological monitoring – e.g. in determining the consequences of anticipated improvements in environmental quality (arising from planned schemes, standstill provisions of the Dangerous Substances Directive, or as required under the Habitats Directive to achieve Favourable Condition).

Toxicity Studies on sensitive species - Toxicity has been studied in a relatively small number of species to date. It would be useful to examine subtle sublethal-effects in some of the less well represented and perhaps sensitive species. Also to include sediment bioassays to look at growth and survival of juvenile bivalves.Compare responses in Severn 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 Warwick et al., 1998.

It is stressed that the above procedures have been selected primarily with regard to their ease of use, low cost and relevance to known environmental problems. Ideally, all components to the scheme need to be synchronised and run in tandem to achieve best value and to provide the most useful information on causal links and mechanisms. The results will assist environmental managers in identifying those consents and activities which most require attention and hopefully may help to decide on the best options for action.

Annex 7. Estimated quantities of contaminants in some major discharges to the Severn

(From data cited in Bird, 2002)

Contaminant	Source	Quantity (kg/yr)	E (estimate) or A (actual)*
Arsenic	Britannia Zinc	70	A
	Walpole Landfill	58	E
	Hempstead Landfill	18	E
Cadmium	Britannia Zinc	454	A
	Avonmouth STW	479	E
	Purton WTW	164	E
	Rhodia Organique Fine	32	A
	Walpole Landfill	29	E
	Solutia	13	E
Chromium	Cardiff East	74,532	E
	Orb Electrical	14,600	E
	Avonmouth STW	11,680	E
	Cog Moors	4,058	E
	Nash STW	1,268	E
	Gloucester STW	781	E
Copper	Cardiff East	28,666	E
	Cog Moors	10,011	E
	Avonmouth STW	8,760	E
	Nash STW	8,453	E
	Ponthir	4,188	E
	Gloucester STW	1,562	E
	Orb Electrical	1,460	E
	Solutia	913	E
Lead	Cardiff East	17,199	E
	Cog Moors	7,170	E
	Nash STW	4,227	E
	Ponthir	2,794	E
	Britannia Zinc	1,417	A
	Gloucester STW	1,562	E
Mercury	AstraZenica	314	A
	Ponthir	140	E
	Avonmouth STW	58	E
	Walpole Landfill	15	E
	Gloucester STW	7.8	E
	Hempstead Landfill	3.7	E
Nickel	Cardiff East	154,798	E
	Orb Electrical	14,600	E
	Avonmouth STW	58,400	E
	Gloucester STW	1,562	E
	Terra Nitrogen	251	A
Zinc	Cardiff East	68,799	E
	Avonmouth STW	58,400	E
	Cog Moors	43,834	E
	Gloucester STW	7,811	E
	Ponthir	8,382	E
	Regis Paper Mill	6,658	E
	Nash STW	4,227	E
	Lydney STW	3,361	E
	Terra Nitrogen	2,447	A
	Portbury Wharf	1,781	E
	Britannia Zinc	972	A

[cont…..]

[…………Annex 7. cont.]

Contaminant	Source	Quantity (kg/yr)	E (estimate) or A (actual)*
Total halogenated organics	Rhodia Organique	6,339	E
Fornaldehyde	Borden Chemicals	36,500	E
Monohydric phenols	Borden Chemicals	10,950	E
Toluene	Borden Chemicals	18,250	E
Vinyl chloride	Borden Chemicals	6,570	E
HCH Gamma	Portbury Wharf	400	E
PCBs	Solutia	26	E
Ammonia	Cardiff East	9,173,000	E
	Ponthir	1,118,000	E
	Nash STW	972,000	E
	Terra Nitrogen	613,000	A
	Gloucester STW	469,000	E
Nitrates	Terra Nitrogen	758,000	A

*** A-** Actual values for 2001 from the Environment Agency's South West Region or **E –** estimations calculated from discharge consents (maximum discharge allowed for each contaminant x the total flow). The latter are likely to be overestimates, since they are based on the theoretical maximum discharged rather than that actually discharged.

Halogenated pesticides are commonly listed in discharge consents and include the pesticides, aldrin, dieldrin and endrin. From the information available, STWs are the main source of these compounds with maximum annual discharges that are typically less than 1 kg yr^{-1}.

Annex 8. Water company improvements in the Severn Estuary pSAC

(Information source - EA and SWW)

The 'Clean Sweep' programme is a programme of investment started when the industry was privatised in 1989. Asset Management Plans (AMPs) were instigated (timescale below).

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

Severn Estuary – SE (English side)

Under AMP2, secondary treatment and UV disinfection were incorporated at Kingston Seymour under the Bathing Water and UWWT Directives. Secondary treatment was incorporated at Avonmouth STW. Thornbury STW was uprated to provide secondary treatment as required by the UWWT Directive. All schemes were completed during 2000.

In AMP3, increased secondary treatment capacity and storm storage is identified at Avonmouth STW during May 2003 under the Bathing Water and UWWT Directives. One storm overflow at Clevedon and 2 at Portishead are due for improvement by April 2003, while approximately 100 CSOs (combined sewage outfalls) will be improved in the Avonmouth and Bristol catchments by April 2004. These include both direct and indirect inputs to the tidal Avon.

From 1987 to April 1999, the effluent from Weston-Super-Mare was screened and disinfected by chlorination during the bathing season and discharged through the Black Rock outfall close to the Bathing Water at Uphill. As required by the UWWT and Bathing Water Directives, secondary treatment and UV disinfection was incorporated during AMP2. In addition, major improvements to the sewerage have been undertaken to ensure that the frequency of storm discharges from the Black Rock outfall are limited to 3 per bathing season (on average). The new STW for Weston-Super-Mare providing secondary treatment and UV disinfection was commissioned prior to the 1999 Bathing Season.

Since July 1994 the primary treated effluent at both West Huntspill and Bridgwater STWs has been disinfected by chlorine during the bathing season. In AMP2, these works were identified for improvements under the Bathing Water Directive and UWWT Directive with upgrading or the provision of secondary treatment and UV disinfection. Both schemes were completed in 2000.

In AMP3, eight storm overflows in Bridgwater and a storm overflow in Burnham have been identified for further improvements under the Bathing Water and UWWT Directives. Improvements are due by April 2005.

From 1991 to April 1998, treatment at Minehead STW was primary only, with disinfection by chlorination during the bathing season. As required by the UWWT Directive, secondary treatment was incorporated within AMP2, and UV disinfection was also installed to comply with the requirements of the EC Bathing Water Directive. Additionally, where necessary, improvements were made to the sewerage system to reduce spill frequencies to 3 spills per bathing season. The treatment scheme was completed in October 1999.

Improvements to a storm overflow at Minehead (Blenheim Road) by April 2002 have been identified under the Bathing Water and UWWT Directives in AMP3. Primary treatment was installed to treat the crude discharge from Watchet in 2000, during which time 4 CSOs were also improved. As part of AMP3, the STW will be improved to provide secondary treatment for these flows, and flows from Donniford by the end of 2002.

Severn Estuary - NW (Welsh side)

Almost the whole catchment represented by South East Area, Environment Agency Wales drains into the Severn Estuary. Therefore all Water Company improvements will have had some impact upon the Estuary.

Continuous Discharges
There were three major UWWTD STW schemes within AMP2/3 that discharge directly into the Estuary.

Nash STW – Covering Newport, Chepstow and the surrounding areas. 5 existing STWs (Magor, Caldicot, Caerleon, Christchurch and Nash) and several crude outfalls, were amalgamated into one new secondary treatment works at Nash with a PE of 118,000. The STW was completed in 1998 and all flows were transferred to them by March 2002.

Cardiff STW – Four crude outfalls from trunk sewers (Cardiff East, Cardiff Central, Western Valley and Rhymney Valley)were diverted to a new secondary treatment STW with a PE of 888,000. All flows were transferred by September 2001 and received crude treatment, secondary treatment was brought online in March 2002.

Cog Moors STW - Crude discharges from West Cardiff and Barry were diverted to a new secondary treatment STW with a PE of 163,000. A long sea outfall was also provided to give protection to the 3 designated Bathing Waters within the Barry Area. Secondary treatment was in place in April 1998.

Intermittent Discharges
Given the nature of sewerage provision in much of the area with trunk sewers out to the coast, there are numerous CSOs and Pumping Stations, many of which have been identfied as unsatisfactory and scheduled for improvement. The key schemes along the coast have been at Newport, Cardiff and Barry.

Newport – AMP2 -5 intermittents improved; AMP3 – 16 will be improved.
Cardiff – AMP2 – 11 intermittents improved; AMP3 – 15 will be improved.
Barry - AMP2 – 13 intermittents improved; AMP3 – 78 will be improved.

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)

Back Cover photograph: The Severn Bore
Proffesor H Peregrine, CEGF - Centre for Environmental
and Geophysical Flows
University of Bristol.

"When the boar comes, the stream does not swell by degrees, as at other times, but rolls in with a head... foaming and roaring as though it were enraged by the opposition which it encounter"
– Thomas Harrel (1824), on the Severn Bore

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