

## **Fish Utilisation of restored intertidal habitats in a tidal backwater of the Thames estuary.**



*A report on a placement with the Environment Agency, Thames Region, in fulfilment of the requirements of the MSc in Aquatic Resource Management of Kings College London*

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# Executive Summary

## Background

This reports describes a work placement with the Fisheries Department of the Environment Agency, Thames Region between 11.06.7 – 06.09.07.

## Main Objectives

The main objectives of the study were to assess fish utilisation of two recently restored intertidal habitats (Creekmouth and A13) in a Thames tidal backwater. These data would contribute directly to a currently limited literature on the importance of small scale intertidal habitat within heavily urbanised estuaries.

## Results

A multiple method survey programme was used based on the best practice methods used to survey transitional water-bodies under Water Framework Directive (WFD). Fish utilisation within the Creekmouth site was also measured on gut fullness and content data. Quantitative estimates of fish density and biomass were calculated based on results from blocking seine net.

Both sites confirmed positive fish utilisation with the restored intertidal habitat. The gut analysis highlighted the feeding profitability produced within the reinstated marsh, with fish caught exiting the site on the ebb tide demonstrating significantly higher gut contents than those caught entering to site on the flood tide. *D. labrax* was found to feed predominately on freshwater invertebrates within these systems. A niche comparison between *D. labrax* and *R. rutilus* indicated their diets were more similar within the restored habitat than in the main channel.

Quantitative estimates of fish fry density within Creekmouth were highly variable, due to the spatial and temporal heterogeneity found during sampling. The atypical weather conditions of late July appeared to limit fish utilisation within these areas.

## Conclusions/ Recommendations

Small size restored intertidal habitat within heavy urbanised estuaries can function as relic marsh and provide a successful nursery and feeding grounds for juvenile fish. These fragmented habitats also help establish an intertidal migratory corridor through rivers, safeguarding fry from the high flows found in the central channel.

This emphasises the importance of intertidal estuarine habitat in achieving 'good ecological status/potential' under WFD, and the added value they could bring if included within the network of Marine Protected Areas (MPA) planned for around in the UK coast.

## **Acknowledgements**

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# 1. General Introduction

## 1.1. Environment Agency

The Environment Agency (EA; the Agency) was established under the Environment Act in 1995. The EA is a non-departmental public body of the Department for Environment, Food and Rural Affairs (DEFRA), and an assembly sponsored public body of the National Assembly for Wales (NAW). The EA currently employs approximately 12,000 people. The EA's budget of almost £900 million (EA, 2007) is predominately derived from three sources:

- Charging schemes e.g. abstraction licenses and rod licences
- Flood defence funded through local authorities
- Government grants through DEFRA and the NAW

The EA became functional on the 1<sup>st</sup> April 1996, with the principal task of protecting and improving the environment of England and Wales, whilst ensuring sustainable development. Under this brief the EA other responsibilities include (Environment Agency, 2004);

- Flood risk management
- Integrated pollution control and prevention (e.g. reducing industrial and sewage related impacts on rivers)
- Radioactive substances regulation
- Waste management (e.g. discharge consents)
- Water and land quality assessments
- Fisheries duties in coastal and freshwater systems
- Promoting recreation, navigation and conservation.

The EA's remit covers approximately 15 million hectares of land, including 36,000 km of river, 5,000 km of coastline and 2 million hectares of coastal water around England and Wales (DEFRA, 2006). The EA is managed by eight regional offices (seven of which are in England and one in Wales) to ensure national goals are delivered at a local level (Figure 1.1).



Figure 1.1: Regional Offices in England and Wales (Environment Agency, 2005)

Within estuaries and coastal waters management the EA has;

- Regulatory roles e.g. competent authority for the EC Directives such as the Water Framework, Bathing Water, Shellfish Water and Dangerous Substances Directives. As well as permit and compliance monitoring of effluent discharges, regulation of fishing for salmon, trout and eels to 6 nautical miles and regulation of flood risk management development.
- Monitoring and management roles for: coastal and tidal flood defence management, EC Directives and other international obligations, navigation and is the lead partner in the delivery of coastal saltmarsh and mudflat BAP Action Plans.
- Advisory and consultation roles for; ports, planning and development

## 1.2. Thames Region

The Thames Region of the EA is responsible for the protection of an area of 13,000 km<sup>2</sup>, defined by the River Thames catchment, from its source in Gloucestershire to its confluence with the North Sea in Essex (Figure 1.2). This area incorporates rural areas such as Wiltshire and Oxfordshire, as well as heavily urbanised areas such as London, Reading and Slough. The region has 5,330 km of main river channel and 896 km<sup>2</sup> of floodplain (Environment Agency, 2007). Although this area only covers approximately 10% of England and Wales, it is home to 12 million people (approximately one quarter of the total population). This puts a huge strain on local resources particularly water, as the region is naturally dry with an average precipitation of only 690 mm, compared with the national average of 897 mm (Environment Agency, 2007).

The Thames Region remains divided into three divisions (North East, South East and West) from historic boundaries created in the time of the National Rivers Authority (NRA). The placement was based at Crossness, one of six satellite offices in the region.





Figure 1.2: The River Thames catchment area (Environment Agency, 2007)

The main pressures in the Thames Region come from:

- Flooding; responsibilities for advising on new developments, constructing new flood defences, maintaining river channels and effective flood warning. These issues have been exacerbated with climate change and increasing sea-level rise.
- Waste; regulating, supervising and licensing local authorities and private companies, as well as trying to reduce the quantity of waste disposed in landfills.
- Water resource management; to provide sufficient water for a multitude of uses by managing abstractions, groundwater, drought and water quality.
- Conservation; escalating anthropogenic populations in the South East with increasing pressure to transform natural habitats (including the 146 Sites of Special Scientific interest; SSSI) into housing.
- Fisheries; to maintain, improve and develop fisheries under the Salmon and Freshwater Fisheries Act 1975, and promote recreational fishing to derive income from the rod licence.

### 1.3. Fisheries Department on Thames Region

The EA at Crossness has responsibilities for the tidal Thames (downstream of Teddington Lock to the Yantlet Line; a fishing boundary line between Crow Stone and London Stone), as well as all other lentic and lotic waterbodies in the region (including the Wandle, Hogsmill and Roding). In 1995 the Thames Region was also made responsible for sea fisheries in the tidal river upstream of Lower Hope Point. As a consequence the EA is now responsible for fisheries management (including rod licences), planning and consents under the Sea Fisheries Regulation Act (1996), the Salmon and Freshwaters Fisheries Act (1975), the Water Resources Act (1991) and the Environment Act (1995).

## 2. Routine Work Undertaken

### 2.1 Thames Tideway Fish Survey, 19<sup>th</sup> June 2007

The site is one of several sites used in the biannual shore based survey of the Thames between Richmond and Gravesend. The fisheries survey was conducted at the regular surveying site at Kew, on the area adjacent to the Strand on the Green. It is conducted at low water slack (a period of cessation in the strong flow of a current of water at low tide) which was at approximately 14.36pm. The monitoring involved the three standard survey methods now used in the WFD to survey transitional waters. They are:

- Shore seining using a 43 m by 2.5m net with 5mm fine mesh. Four sweeps are conducted.
- Beam trawling with a 2m beam trawl over 200m parallel to the seine netting site
- Kick sampling with standard FBA 1mm mesh net for 1 minute.

Originally the survey involved two different seine net sizes; however no notable difference was found in the species size or composition caught using the two different sizes. It was therefore modified to one 43 metre seine. During the survey, juvenile *Perca fluviatilis* (perch), *Platichthys flesus* (flounder) and *Leuciscus leuciscus* (dace) were recorded.

### 2.2. WFD Meeting to discuss ‘Good Ecological Status’, 21<sup>st</sup> June 2007

The Fisheries department at Crossness developed the methodology now regarded as best practice for WFD surveying of transitional waters. This uses multiple methodologies to generate a better representation of estuary dynamics. The WFD team (Steve Coates and Adam Waugh, over seen by Steve Colclough) from Crossness are currently involved in the training of regional teams on how to conduct these surveys. They are also involved in the development of a classification tool to help determine the ecological status of fish in all the 132 transitional waters in the UK. The classification tool is based on eight measurable factors which represent biological community, assemblage, structure and function (USEPA, 2000). The metrics are (Coates *et al.*, 2007);

- Species composition
- Presence of indicator species
- Species relative abundance
- Number of taxa which make up 90% of abundance
- Number of estuarine resident taxa
- Number of estuarine-dependent marine taxa
- Functional guild composition
- Number of benthic invertebrate feeding taxa

In accordance with the WFD time-frame the team are using five pilot estuaries, one of which is the Thames Estuary, to test the validity of the classification tool. The classification is

currently only based on two years data for the five pilot estuaries. At this early stage, the WFD classification will be heavily reliant on expert judgement, as statistical confidence will not be possible with such limited data. The Exe Estuary was used as the first pilot transitional water to assess the validity of the tool. The data collected from the survey programme was analysed using the classification tool, to find the estuary was achieving ‘good ecological status’. Expert judgement confirmed this by taking into account the physical attributes of the river and the key species present. This showed the ecological monitoring and analyses tool had passed the initial test.

### 2.3. Sampling a small piece of relic saltmarsh in the Deben Estuary, Suffolk, 28<sup>th</sup> June

The Deben Estuary is located within the Suffolk Coast and Heaths Area of Outstanding Natural Beauty (AONB). Suffolk has the greatest quantity of saltmarsh habitat remaining in any county in England and Wales, and the Deben Estuary contains 40% of Suffolk’s saltmarsh. The importance of this habitat for birds and plants is already recognised in the area; however no work has been done on its importance for fish.

The saltmarsh surveyed was a small section of relic marsh close to Woodbridge in the upper reaches of the Deben Estuary. As a leading expert in this field Steve Colclough was contacted by Suffolk and Coast Heaths organisation to sample the saltmarsh, which is currently under threat from erosion. It was hoped that by illustrating fish utilisation and the nursery functions of this marsh, it can be used to aid, protect and manage the saltmarsh from further anthropogenic induced damage.

The saltmarsh was surveyed by seine netting a creek channel, with a stop net placed at the end of the channel to prevent fish escaping (Figure 2). A typical fish community was found including two age classes of *Dicentrarchus labrax* (bass), *Chelon labrosus* (thick lipped mullet), *Liza ramada* (thin lipped mullet), *Pomatoschistus microps* (common goby) and a *Gasterosteus aculeatus* (three-spined stickleback). Fixed sample survey methods (e.g. the modified Surber sampler) were demonstrated, but not deemed suitable due to a high abundance of crabs, which would eat the captured fish.



Figure 2.1: Static block net deployed in Deben estuary saltmarsh catching fish leaving on ebbing tide and seine netting at high-water.

## 2.4. Thames Tideway Fish Survey, 3<sup>rd</sup> July 2007

The survey was conducted at the regular surveying site at Chelsea, on the area adjacent to the Battersea Church Road, as part of the biannual shore based fisheries monitoring of the Thames Estuary (Figure 2.2). This fieldwork was used to show EA Head of Fisheries Daffyd Evans, how the transitional water survey is conducted. The seine netting was conducted at low water slack tide, however at the time of sampling there were high fluvial flows making conditions difficult. During the survey the most abundant fish found was juvenile *L. leuciscus*, followed by juvenile *P. fluviatilis*, *P. flesus* and *D. labrax*. An *Anguilla anguilla* (eel) was also captured.



Figure 2.2: Seine net deployment and pulling in the seine net at Chelsea, River Thames

## 2.5. Saltmarsh sampling at Tollesbury Managed Realignment site, Essex

Tollesbury and Orplands managed realignment sites are two of the areas under investigation in the EU Interreg ComCoast (Combined functions in Coastal defence zones) fisheries PhD project by Leila Fonseca. Part of the UK involvement in the EU project is through three PhD's in fisheries, geomorphology and saltmarsh economics. Their aim is to help illustrate the wider benefits of a gradual transition between land and the sea.

This summer's sampling uses a large net, similar to a fyke net, in conjunction with stop nets to try and quantify fish utilising the sites for cost-benefit analysis. At Tollesbury (5.07.07), a piece of relic saltmarsh, just outside the breach was surveyed. The large fyke-like net was placed in the mouth of a small semi-discrete creek network, to ensure no movement to other areas of the marsh stop nets were placed around the periphery of the creek. The net in the entrance to the creek network was weighted down during the flooding tide to allow fish to enter the site and raised at high water slack to catch leaving the site during the ebbing tide. At low tide the fish were collected from the cod end, and stored to be weighed and measured. The catch consisted of primarily juvenile *D. labrax* and *Clupea harengus* (Atlantic herring). As the area of this discrete creek can be calculated, this data can be used to give a minimum estimate of fish utilisation per unit of area, a very useful tool in helping to establish the economic value of saltmarsh habitat.

The same large fyke-like net method was repeated at Orplands managed realignment on the 12.07.07 and 14.07.07. (Figure 2.3) The main fish species catch caught during sampling were *D. labrax*, *P. microps* and *Atherina presbyter* (sand smelt).



Figure 2.3: Orplands large fyke-like net and stop nets.

## 2.6. Meeting with DEFRA, 10<sup>th</sup> July 2007

Steve Colclough and Martin Stark from the EA, and I, met with officials from (DEFRA) with three main aims;

- 1) To notify DEFRA of work being done within the EA regarding the importance of saltmarsh and coastal intertidal areas as nursery areas for juvenile marine and commercially important fish.
- 2) To discuss the possibility of some funds of the new European Fisheries Fund (EFF) being used in conjunction with other funding streams for habitat creation e.g. saltmarshes within managed realignment sites. This offers multi-functional benefits not only by providing nursery grounds to help address declining commercial fish stocks, but also assisting in flood storage capacity in relation to sea-level rise.
- 3) The possibility of using the EFF for fish and *A. anguilla* passes on rivers was also raised to help achieve 'good ecological status' under WFD whilst protecting BAP (Biodiversity Action Plan) species.

This discussion was timed to coincide with DEFRA drafting the Operational Programme for the EFF, to keep these new and important areas at the forefront of their mind, as well the conventional areas which received funding under the Common Fisheries Policy (CFP) such as decommissioning fishing vessels.

## 2.7. Tilbury Power Station Screen Monitoring, 11<sup>th</sup> July 2007

Tilbury Power Station is an old power station requiring a large intake of cooling water, which is pumped through a continually rotating band screen to remove debris including fish. Jets of water are used to remove the debris from the screen, which is then disposed off in trash skips. The EA and Zoological Society of London (ZSL) run a join project to monitor the organisms being taken in with the cooling water. The surveying involves the screen debris being intercepted at each of the four removable points using a fine mesh net. The debris is when emptied every half an hour between 10am and 3pm, every other Wednesday. The species are then identified, counted and measured (total length). On the day of sampling species recorded included large number of juvenile *P. flesus*, *D. labrax* and *P. microps*, as well as *C. labrosus*, *L. ramada*, *A. presbyter*, *C. harengus*, *Eriocheir sinensis* (Chinese mitten crab), *Carcinus maenas* (common shore crabs), *Solea vulgaris* (European Dover sole), and an *A. anguilla*.

## **Account of Original Work Undertaken**

- 3. An assessment of Fish Utilisation of restored intertidal habitats in a tidal backwater of the Thames estuary.**

## 3.1. Introduction

### 3.1.1. The Importance of Saltmarshes

A saltmarsh by definition is the higher, vegetated portion of the inter-tidal mudflat, which lies approximately between Mean High Water Neap (MHWN) and Mean High Water Spring tides (MHWS) where net sediment accumulation occurs (Doody, 2001). Some degree of shelter from wave action is necessary for sediment accretion, therefore saltmarshes are commonly found in inlets and estuaries. Initially, pioneer halophytic angiosperms stabilise and bind the mudflat sediments and reduce the water velocity through their stem network, aiding a succession to patchy vegetation stands (Mason *et al.*, 2003). These habitats support distinctive communities of halophytic fauna and flora such as *Salicornia* and *Spartina*, within dendritic creek systems which flood and drain with the tides (Little, 2000). Saltmarshes are highly productive systems and net exporters of energy. They have been likened to tropical rainforests, producing between 153g and 2,722 g m<sup>2</sup> y<sup>-1</sup> of green vegetation (Long and Mason, 1983).

Saltmarshes and their intertidal sediments are recognised as possessing a wide range of key biological (feeding and nursery), ecological and chemical (nutrient and carbon storage) functions, as well as acting as a primary flood defence system (Dixon *et al.*, 1998; Jickells *et al.*, 2003). Intertidal saltmarsh has been shown to attenuate wind wave height and tidal amplitude (Moller *et al.*, 1999). However, it is estimated 85% of British estuaries have lost up to 80% of their intertidal area through anthropogenic land claim for reasons such as agriculture, port developments, harbours, industry and housing (Atrill *et al.*, 1999). It was estimated that only 44,370 ha of saltmarsh remains in the UK (Burd, 1989), of which an estimated 100 ha is lost very year (Davidson *et al.*, 1991).

Intertidal habitat lost is also being exacerbated by sea-level rise (currently at 1-1.5 mm yr<sup>-1</sup>), resulting in a phenomenon called coastal squeeze. This is when habitat on the seaward side is lost through sea-level rise, because hard sea-line defences such as sea walls, prevent the marshes compensatory inland encroachment (Crooks and Turner, 1999; Hughes, 2004). It is estimated this currently results in the loss of 2% of English saltmarshes every year (Dixon *et al.*, 1998). This problem made worse in the South-east of England by post-glacial land subsidence from isostatic rebound (Colclough *et al.*, 2005). Controversial research also indicates saltmarsh development may also be limited by competition from herbivorous invertebrate infauna at the saltmarsh mudflat interface (Hughes and Paramor, 2001).

Saltmarshes have long been recognised as a key feeding and roosting ground for birds (Rupp and Nicholls, 2002). However, the importance of these areas for fish has only recently been documented in the UK, despite an extensive body of work on the issue in North America. In the US research, dating back to the 1970's, describes saltmarshes as vital components of local commercial and recreational fisheries, providing feeding, refugia and nursery areas for juvenile fish (Shenker and Dean, 1979; Boesch and Turner, 1984; Rountree and Able, 1992; Peterson and Turner, 1994; West and Zedler, 2000). Bell (1997) recognised a dramatic reduction in fish production was coupled with the loss of 68,000 ha of coastal habitat in the States between 1950 and 1970.



In Northern Europe the intrinsic difficulties of sampling in such large tidal regimes meant for a long time these habitats were not deemed to be important to fish (Elliott and Taylor, 1989; Laffaile *et al.*, 2000), Mathieson *et al.*, 2000). The significance of these high intertidal zones for fish is however now being realised in Europe. Elliott and Taylor (1989) reported the intertidal habitats in the Forth Estuary were nearly twice as productive as their subtidal equivalents (0.077 and 0.032 t per hectare respectively). Recent data on juvenile fish associations within these intertidal areas has incited the Environment Agency to recommend these areas as possible inland Marine Protected Areas (MPA) to help preserve fish stocks (Colclough *et al.*, 2002).

In the past these habitats were viewed as wastelands with no real value, and as a consequence anthropogenic encroachment onto these intertidal areas faced little public opposition. Encroachment is as the loss of land riverward of existing defences. Encroachment is biologically detrimental as it physically removes valuable foreshore habitat, and obstructs migrating species using selective tidal stream transport (Colclough *et al.*, 2002). It can also change the geomorphology of the system through increasing scour and flow. Another consequence of land encroachment is the dramatic reduction in flood storage capacity, a very serious issue in an era of increasing sea-level rise and storminess due to climate change. The removal and straightening of riverbeds has also been shown to dramatically reduce habitat structural complexity and heterogeneity (Atrill *et al.*, 1999). Anthropogenic altering, inhibiting and stopping of natural geomorphology processes has been shown to result in the loss of species richness and/or the disappearance of whole communities within estuaries (Claridge *et al.*, 1986; Elliott and Hemingway, 2002).

Saltmarshes are now designated as UK Biodiversity Action Plan (BAP) habitat and form part of DEFRA's High Level Target Habitat Series. This has resulted in initiatives requiring no further net loss of saltmarsh habitat, and a growing appreciation for protecting the UK coastlines with soft-engineering methods such as managed realignments, instead of previous hard-engineering options (Klein *et al.*, 1998; Leafe *et al.*, 2002).

A managed realignment is when the existing sea-defences are breached or removed and the line of sea defence is set back. This allows the establishment of new intertidal habitat, which in turn dissipates wave energy and increase flood storage capacity in an environmentally sustainable way (Rupp and Nicholls, 2002). Given the typical low productivity of coastal agricultural land in the UK, managed realignment is also considered economically effective (Bowers, 1999; Packham and Willis, 1997). Within the UK large scale managed realignment schemes have taken place in the Wash (Freiston), Humber Estuary (Paull Holm Strays) and Blackwater Estuary (Tollesbury, Abbots Hall and Orplands).

An EU Interreg project called ComCoast was set up in 2004 three years ago to as an information exchange network across Europe to discuss innovative solutions to climate change. In the UK data is being collected via three PhD projects to help illustrate the wider benefits of a gradual transition between land and the sea, and to promote flood defence with ecological benefits. Early findings from the PhD's shows that managed realignment sites in the Blackwater Estuary are successful new nursery habitats for fish, as well as acting as important pollution sinks. Initial reports from ComCoast are expected this autumn.

### 3.1.2. The Thames Estuary

The Thames Estuary extends 110 km from its tidal limit at Teddington Weir to the North Sea. The first human communities established on the banks of the River Thames approximately 250,000 years ago. The Thames Estuary has a long history of pollution and overfishing (Atrill *et al.*, 1999). From the beginning of the 1800's overfishing led to the collapse of the *A. presbyter* and *Salmo salar* (Atlantic salmon) fisheries, followed shortly by the collapse of the *A. anguilla* fishery from navigation locks and pollution (Naismith and Knights, 1993). Extensive riverside development and embankment creation in the 19<sup>th</sup> and 20<sup>th</sup> Century lead to the widespread loss of estuarine habitat (Colclough *et al.*, 2002). By the end of the 19<sup>th</sup> century *Alosa fallax* (twaite shad) had also been removed from the upper and middle reaches of the river (Whitfield and Elliott, 2002). The River reached an all time low in the 1950-1960's when anoxia from organic pollution left the middle reaches of the river biologically dead (Andrews, 1984). The introduction of sewage treatment improved the water quality of the estuary and led to a significant fish recovery from 1964 onwards (Colclough *et al.*, 2002). Today around 50 vessels commercially fish below Mucking, predominately for *C. labrosus*, *L. ramada*, *D. labrax*, *A. anguilla*, *C. harengus*, *S. vulgaris*, *Sprattus sprattus* (sprat), *Limanda limanda* (dab), *Merlangius merlangus* (whiting) and *Cerastoderma edule* (common cockle: Colclough *et al.*, 2002). There is also a substantial recreational fishery for *L. leuciscus*, *P. fluviatilis* *Rutilus rutilus* (roach) and *Abramis brama* (common bream) above Battersea, and *S. vulgaris*, *D. labrax*, *P. flesus*, *M. merlangus* and *A. anguilla* below Woolwich (Colclough *et al.*, 2002).

It is estimated between 1930-1980, 64% of the grazing marshes in the upper Thames Estuary were lost (Elkins, 1990), and now less than 1% of fragmented original bank and supralittoral foreshore still remains in the whole Thames Estuary as a result of encroachment (Colclough *et al.*, 2002). Encroachment, leading to the removal and loss of bankside and intertidal vegetation, has long been recognised in freshwater systems to substantially affect energy inputs and trophic dynamics (Hawkins *et al.*, 1982). But very little information exists of the effect of landscape changes within estuarine sections of rivers, despite them typically suffering the greatest morphological change (Davidson *et al.*, 1991; Petts, 1997). Improvements in the River Thames water quality have greatly improved the ecology within the estuary. However this recovery could now be limited by problems attributed to geomorphology change and encroachment pressure.

The general lack of interest in the ecology of the upper foreshore of urban estuaries may be due to them being viewed as highly impacted systems with little value or complexity (Boon, 1992). This is all set to change with the introduction of the WFD. This Directive requires that all Member States must achieve 'good ecological status' in all freshwater, transitional and coastal water-bodies by the end of 2015, excluding 'heavily modified waterbodies' which must achieve 'good ecological potential'. This is the first time transitional waterbodies (estuaries) have been included within required monitoring programmes.

Research on the Forth Estuary, Scotland, estimated encroachment in the area over the last 200 years could be responsible for a 66% reduction in local fish stocks (McLusky *et al.*, 1992). This is supported by research indicating round-fish fry in particular, closely associate with reed-beds and saltmarsh (Colclough *et al.*, 2002). The Thames Estuary is recognised as the largest *S. vulgaris* nursery on the east coast, and a significant nursery for *D. labrax*, and species of Gadidae and Clupidae (Colclough *et al.*, 2002; Thomas, 1998). This suggests

improving marginal intertidal habitat in the Thames Estuary, given its close proximity to the North Sea, could be seen as producing an inshore nursery habitat to aid fish production.

### **3.1.3. Aims**

- To assess fish utilisation (abundance and species richness) of two regenerated intertidal habitats in the tidal backwater of the Thames estuary.
- To quantitatively assess the feeding preference and gut fullness of *D. labrax* and *R. rutilus* on the flooding and ebbing tides, as a measure of intertidal utilisation.
- To assess if these small-scale man-made habitats function in the same generic way as relic intertidal habitat and managed realignments sites.
- To assess the importance of urban intertidal regeneration in relation to obtaining 'good ecological status/potential' for heavily urbanised waterbodies under the WFD and as potential inshore Marine Protected Areas (MPA) to help obtain sustainable fisheries.

## 3.2. Methods

### 3.2.1. Study Sites

This study focused on two recently regenerated sites in Barking Creek, on the tidal River Roding (Figure 3.1). The River Roding is a tributary to the River Thames, which rises to the north-east of London and flows south through east London, before joining the Thames in Barking. The dominant land-use in the surrounding area is commercial/industrial and extremely urbanised. These areas are protected from tidal and fluvial flooding by flood defences, and from extreme high tides by the Barking Barrier (part of the Thames tidal defences).



Figure 3.1: Barking Creek location within Thames Estuary and Creekmouth and A13 site location within Barking Creek (GoogleEarth, 2007).

The study sites form two of four enhancements undertaken by the Lower River Roding Regeneration Project, which was delivered by the EA using funding from the Government Sustainable Communities Fund. The project aimed to increase flood storage capacity on the river, whilst providing valuable BAP habitat. Important social factors included in the scheme were to improve the aesthetics of the riverside-area, improve access (via the creation of a new river pathway) and the provision of educational river awareness interpretation boards for the general public.

Barking Creek is recognised as a valuable feeding and refuge area for a variety of fish species e.g. *P. flesus*, *A. anguilla*, *A. presbyter* and *D. labrax*, and also supports some commercial *A. anguilla* fishing (Colclough *et al.*, 2000). Enhancing and extending the upper intertidal habitat in this area should therefore benefit these fisheries.

The most easily accessed enhancement is Creekmouth (Figure 3.2). Prior to its regeneration, which started in October 2006 and was completed by March 2006, the Creekmouth site was a terrestrial grassland habitat with patches of scrub and the invasive *Fallopia japonica* (Japanese Knotweed). The scheme involved breaching the existing sea-defence to create a tidal backwater, with two areas of BAP habitat: a 0.1 ha tidal mudflat and a 0.9 ha of saltmarsh habitat. The intertidal area was created using a brushwood and coir revetment structure, and left to colonise naturally using the rivers own seed-bank. Additionally, by retreating the flood defence in this area an additional 15,000 m<sup>3</sup> of flood storage capacity was provided.



Figure 3.2: Creekmouth: before landscaping and flood defence improvement works, and after retreat of flood defence, construction of tidal backwater and landscaping works (EA Archive).

The A13 site was previously a species poor terrestrial scrubland habitat (Figure 3.3). The area was enhanced by the retreat and renewal of a 50 m section of flood defences. This created an additional 2000 m<sup>3</sup> of flood storage capacity and further 390 m<sup>2</sup> of BAP saltmarsh and mudflat habitat.



Figure 3.3: Land Nr 13: View of frontage before works (from Cuckholds Haven), and after retreat of flood defence and creation of intertidal zone (EA Archive).

### 3.2.2. Fieldwork Sampling

The sampling took place on the 16.07.07, 17.07.07, 26.07.07 and 27.07.07 due to logistical requirements involving the use of the boat and EA staff (Appendix 5.1). It used a range of different static and mobile sampling techniques, which take into account the dynamic nature of estuaries and the continuous redistribution and schooling behaviour of fry in the intertidal areas.

For the purpose of the study;

- Ebb tide is used to describe fish caught whilst exiting the intertidal vegetation on the ebbing (out-going) tide. This indicates the fish had been utilising the intertidal habitat during the high-tide.
- Flood tide is used to describe fish caught in close proximity to the intertidal vegetation trying to enter the sites on the flooding (in-coming) tide.

To catch fish entering the Creekmouth site on the flooding tide (Figure 3.4);

- 1) A static fyke net (200mm net with D ring opening and four internal hoops reducing to 280mm diameter and 15mm knotless mesh, connecting to a 460 cm central wall with 30mm knotless mesh) was placed on the peripheral intertidal vegetation on the outside of the Creekmouth site.
- 2) Mobile seine net (35 m by 2.5m net with 5mm knotless mesh) was deployed close to high water slack. The net was deployed via boat (with one end being held on shore) in the shape of a circle. The net is then hauled towards shore, where the lead line is then pulled up to form a purse with the fish trapped inside (Elliott *et al.*, 2002). This is an integral part of the WFD transitional fish survey kit, and has been demonstrated as one of the most effective ways of obtaining species diversity and population structures (Morrison *et al.*, 2002).

- 3) Push net (3 m by 1 m knotless 5mm mesh supported to two 1.5 m poles and pursed along the bottom). The net is kept taught and pushed quickly at a 45° angle into the marginal vegetation and then raised horizontally in the water column to net fish. This is specialised equipment designed to exclusively catch fry in marginal vegetation.

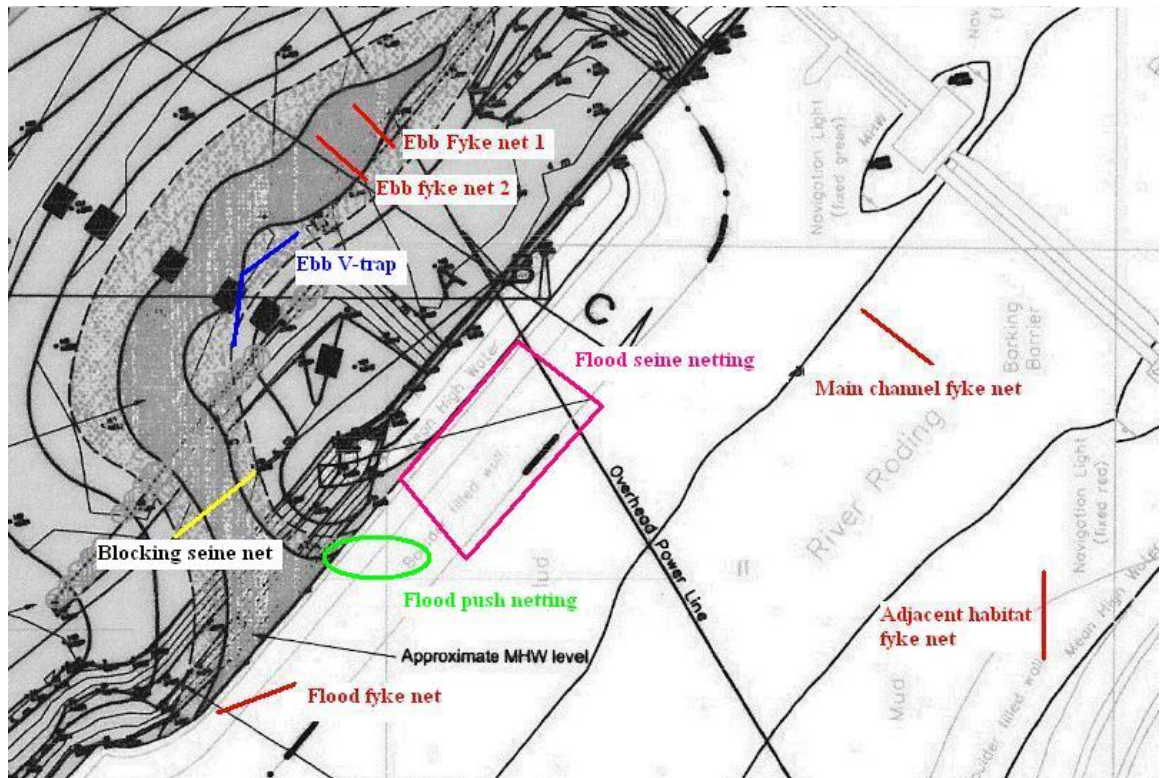


Figure 3.4: Location of methods used to sample Creekmouth regeneration site on the flood and ebb tides.

To catch fish leaving the Creekmouth site on the ebbing tide;

- 1) A blocking seine net (35 m by 2.5m net with 5mm knotless mesh) was used to completely block the entrance to the creek network. The net was hauled across the entrance on the creek (using a rope) just before high water slack and set in position to catch all fish leaving the site on the ebbing tide (Figure 3.5)
- 2) Static fyke nets, were set in the high intertidal vegetation within the Creekmouth, to see if fish were utilising the upper intertidal vegetation when available (Figure 3.6)
- 3) V-trap (500mm x 400mm x 2.5m, with 5m wings, 4mm knotless mesh and 1mm knotless mesh cod end) was also placed in the high intertidal area of the Creekmouth site.

Sufficiently large subsamples of marine and freshwater fish caught were retained for statistical analysis from the flooding and ebbing tides at Creekmouth. These were frozen as soon as possible for laboratory gut analysis. For comparative purposes a buoyed fyke net was placed on the bed of the River Roding to sample fish utilising the main channel and within some naturally-established older intertidal habitat adjacent to the Creekmouth site.

The nature of the A13 site meant it was not possible to set fixed sampling nets, therefore fish were caught on the flooding tide using the seine net method and on the ebbing tide using the push net.



Figure 3.5: Block seine net set at high water in the Creekmouth regeneration site.

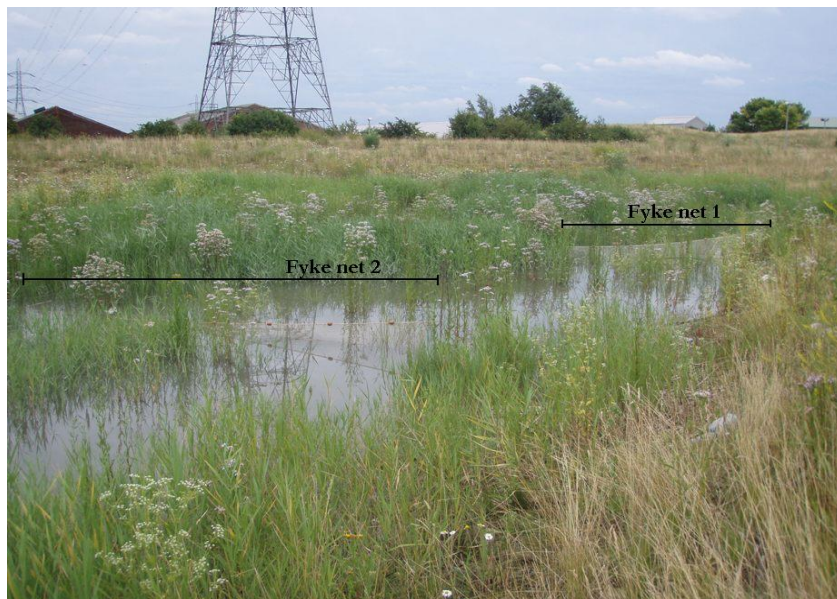


Figure 3.6: Two fyke nets set in the upper intertidal zone of the Creekmouth site during spring tides.



### 3.2.3. Laboratory Gut analysis

The gut analysis was adapted from Laffaille *et al.* (2001) methodology. First, the length to the nearest 1mm was measured. Total length (a measurement from most forward point of head to farthest tip of tail) was used for marine fish and fork-length (a measurement from most forward point of head to the centre of the fork in the caudal fin) was used for freshwater fish. The fresh weight (FW) was recorded to nearest 10mg. The gut was when removed under a Leica MZ95 binocular microscope, and the prey items were cleared of mucous and other unidentifiable material. Each prey item was when identified to the nearest taxonomic level possible, weighted and counted.

These data were used to calculate a number of indices;

- Mean Prey Biomass (MPB) also referred to as gut fullness, which is the ingested prey biomass as a proportion of the total biomass (Laffaille *et al.*, 2001).

$$\text{MPB} = (\text{TPW} / \text{FW}) \times 100$$

where TPW is total prey weight in gut (g) and FW is the fresh weight of fish (g).

- Frequency Occurrence (FO%) was calculated to show to percentage of total guts containing a certain prey species (Hyslop, 1980).

$$\text{FO}\% = (\text{FO}_i / \text{FO}_t) \times 100$$

where FO<sub>i</sub> is the number of guts in which the *i*th species occurs and FO<sub>t</sub> is the total number of guts analysed.

- Numerical (N%) and gravimetric (G%) composition of gut diet (Laffaille *et al.*, 2001);

$$\text{N}\% = (\text{NP}_i / \text{TP}) \times 100$$

$$\text{G}\% = (\text{WP}_i / \text{TWP}) \times 100$$

where NP<sub>i</sub> is total number of *i*th prey type, TP is total number of prey items ingested, WP<sub>i</sub> is wet weight of *i*th prey item and TWP is total wet weight of prey items in gut.

- Niche overlap was calculated between the *R. rutilus* and *D. labrax* on the flood and ebb tides, using the percentage similarity method for species *j* and *k* (*P<sub>jk</sub>*: Krebs, 1999). As a result if the *D. labrax* and *R. rutilus* diets were the same, the niche overlap would be 100%; because as the overlap declines, so does the percentage.

$$P_{jk} = [ \sum_{i=1}^n (\text{minimum } P_{ij}, P_{ik}) ] \times 100$$

where  $P_{ij}$  is the proportion resource  $I$  is of the total resource used by species  $j$  and  $P_{ik}$  is the proportion of the total resource used by species  $k$ .

Other studies have used composition indices such as the Main Food Index (MFI) to calculate primary and secondary food sources, however it is believed detail is lost within the generalisation of this index, so it is not used in this study (Berg, 1979; Hamerlynck and Cattrijsse, 1994). Mann-Whitney U-tests were used to compare gut fullness on the flooding and ebbing tides.

### 3.3. Results

#### 3.3.1. Creekmouth

Over the three sampling days (16.07.07, 17.07.07 and 27.07.07) at the Creekmouth regeneration site a total of 5,503 fish from 9 different families, belonging to 12 different species, were caught on the flooding and ebbing tides (Table 1.1). Of these, three were marine estuarine dependent species (*A. presbyter*, *D. labrax* and *L. ramada*), four were true estuarine species (*G. aculeatus*, *P. pungitius*, *P. minute* and *P. flesus*), one was a catadromous species (*A. anguilla*) and four were typical freshwater species (*R. rutilus*, *L. leuciscus*, *A. brama* and *P. fluviatilis*). All species were ubiquitous, apart from *A. presbyter*, *P. pungitius* and *A. Anguilla* which were only caught on the flooding tide; therefore the flooding tide had greatest species richness.

Table 1.1: Creekmouth species summary

Group	Family	Species	Common name
<b>Marine Estuarine dependent species</b>	Atherinidae	<i>Atherina presbyter</i>	Sand smelt †
	Serranidae	<i>Dicentrarchus labrax</i>	Sea bass †*
	Mugilidae	<i>Liza ramada</i>	Thin lipped mullet †*
<b>True Estuarine species</b>	Gasterosteidae	<i>Gasterosteus aculeatus</i>	3-spined stickleback †*
		<i>Pungitius pungitius</i>	9 spined stickleback †
	Gobiidae	<i>Pomatoschistus minutes</i>	Sand goby †*
	Pleronectidae	<i>Platichthys flesus</i>	Flounder †*
<b>Catadromous (Migrants)</b>	Anguilidae	<i>Anguilla anguilla</i>	European eel †
<b>Freshwater species</b>	Cyprinidae	<i>Rutilus rutilus</i>	Roach †*
		<i>Leuciscus leuciscus</i>	Dace †*
		<i>Abramis brama</i>	Common bream †*
	Percidae	<i>Perca fluviatilis</i>	Perch †*

where \* was found on the ebbing tide and † was found on the flooding tide

At Creekmouth, on the ebbing tide, a total of 4,407 fish were caught from 9 different species. The first sample (16.07.07) yielded the greatest fish catch, with a total of 3,131 fish, from 7 different species (Table 1.2). Of this total abundance 92.6% (2,898 individuals) were of *D. labrax*, with fish lengths ranging from a minimum of 35 mm to a maximum of 65mm.

On the following day sampling (17.07.07), a total of 1,178 fish was caught on the ebbing tide from 8 different species. The composition of which was slightly different from the previous sample (16.07.07) with the addition of individuals from the species *A. brama* and *G. aculeatus*, and the loss of representatives from the species *P. minutes*. Of this total abundance 94.5% (1,113 individuals) were *D. labrax*.

This decreased to only 98 fish caught on the ebbing tide of the 27.07.07. These individuals were from 3 different species; *D. labrax* and the previously less dominant *L. leuciscus* and *R. rutilus*. The catch was dominated to a lesser degree (64%) by the marine estuarine dependent *D. labrax*.

The fyke nets captures illustrates that juvenile fish were utilising even the highest vegetated intertidal areas of the Creekmouth site. Fyke net 1, which was the highest in the tidal zone (Figure 3.5) caught *D. labrax*, *L. ramada*, *R. rutilus* and *P. minutes*, and the second fyke net, slightly lower in down in the high intertidal vegetation, caught *D. labrax*, *L. ramada* and *L. leuciscus*.

A total of 1,096 fish were caught from 12 different species on the flooding tide at Creekmouth (Table 1.2). On both the 16.07.07 and 17.07.07 catches were dominated by *D. labrax*, comprising of 93.2% (220 individuals) and 95.2% (672 individuals) respectively. These dynamics changed on the 27.07.07, when 69.1% (105 individuals) of the flooding catch were freshwater *R. rutilus*, and only 12.5% (19 individuals) were *D. labrax*. Despite this catch being predominately freshwater species; it did also have the only recorded presence of the marine estuarine dependent species *A. presbyter*.

It should be noted that species identification of early life stages of Cyprinidae fry is difficult in the field. This may have led to some misidentification between *L. leuciscus* and *R. rutilus* species.

Table 1.2: Creekmouth total ebb and flood species abundance (sectioned into methods used) from the 16<sup>th</sup>, 17<sup>th</sup> and 27<sup>th</sup> July 2007.

Date	Tide State	Method	Bass	Dace	Perch	TL † Mullet	Roach	Bream	Flounder	Sand goby	Eel	Sand Smelt	9-SS*	3-SS*	TOTAL
<b>16th</b>	<b>Ebb</b>	Fyke 1	28	-	-	4	1	-	-	1	-	-	-	-	34
		Fyke 2	115	1	-	38	-	-	-	-	-	-	-	-	154
		V-trap	74	-	-	16	25	-	-	-	-	-	-	-	115
		Block seine	2681	0	9	7	127	-	4	-	-	-	-	-	2828
		<b>Total</b>	2898	1	9	65	153	0	4	1	0	0	0	0	<b>3131</b>
<b>17th</b>	<b>Ebb</b>	Fyke 1	16	-	-	-	-	-	-	-	-	-	-	-	16
		Fyke 2	77	2	-	3	-	-	-	-	-	-	-	-	82
		V-trap	9	-	-	1	-	-	-	-	-	-	-	-	10
		Block seine	1027	3	8	3	37	1	6	-	-	-	-	1	1086
		<b>Total</b>	1113	5	8	7	37	1	6	0	0	0	0	1	<b>1178</b>
<b>27th</b>	<b>Ebb</b>	Block seine	63	8	-	-	27	-	-	-	-	-	-	-	98
		<b>Total</b>	63	8	0	0	27	0	0	0	0	0	0	0	<b>98</b>
<b>16th</b>	<b>Flood</b>	Seine net	89	-	1	5	2	-	-	1	-	-	-	-	98
		Push net	42	-	-	-	6	-	-	1	-	-	-	-	49
		Fyke	178	-	4	2	2	-	-	-	1	-	-	-	187
		<b>Total</b>	220	0	4	2	8	0	0	1	1	0	0	0	<b>236</b>
<b>17th</b>	<b>Flood</b>	Seine net	379	4	3	1	2	2	4	-	-	-	-	-	395
		Push net	17	4	-	-	-	-	-	-	-	-	-	-	21
		Fyke	278	2	1	2	-	2	7	-	-	-	-	-	292
		<b>Total</b>	674	10	4	3	2	4	11	0	0	0	0	0	<b>708</b>
<b>27th</b>	<b>Flood</b>	Seine net	19	17	-	-	105	7	-	-	-	1	2	1	152
		<b>Total</b>	19	17	0	0	105	7	0	0	0	1	2	1	<b>152</b>

where † = thin lipped mullet and \* = spined stickleback. Fyke 1 indicates net highest up the intertidal range

The nature of the blocking seine net deployed at high tide across the mouth of the Creekmouth system, means all fish utilising the area could be caught exiting the site. This allows a quantitative estimate of fish utilising this intertidal habitat to be estimated. This estimate should however be viewed with caution due to the allowing assumption and limitations:

- The estimate assumes all fish utilising the area during the high tide were caught in the block seine.
- The area of Creekmouth was calculated as 1,530 m<sup>2</sup>, using the inundated by Mean High Water Springs (MHWS) from scaled drawings of the site produced in the planning stages by Halcrow Group Limited. The height of MHWS is the year average of the two successive high waters of each month from the 24 hours period with the tidal range is at its greatest (Natural Environment Research Council, 2007). Tide tables were used to calculate the MHWS in terms of tidal height, to enable the percentage of area inundated on the tidal heights of the 16.07.07, 17.07.07 and 27.07.07 to be calculated.
- The biomass estimate assumes all the *D. labrax* caught were an average length of 47 mm (the mean value of ebbing *D. labrax* during sampling), and therefore an average weight of 1.08 g (based on length/weight collected during sampling). The average weight was calculated by fitting an exponential line of best fit and corresponding regression equation to the graphed length/weight relationship of *D. labrax* from the ebbing tide (Figure 3.7). The average length (47mm) was when substituted into the equation to give corresponding weight.

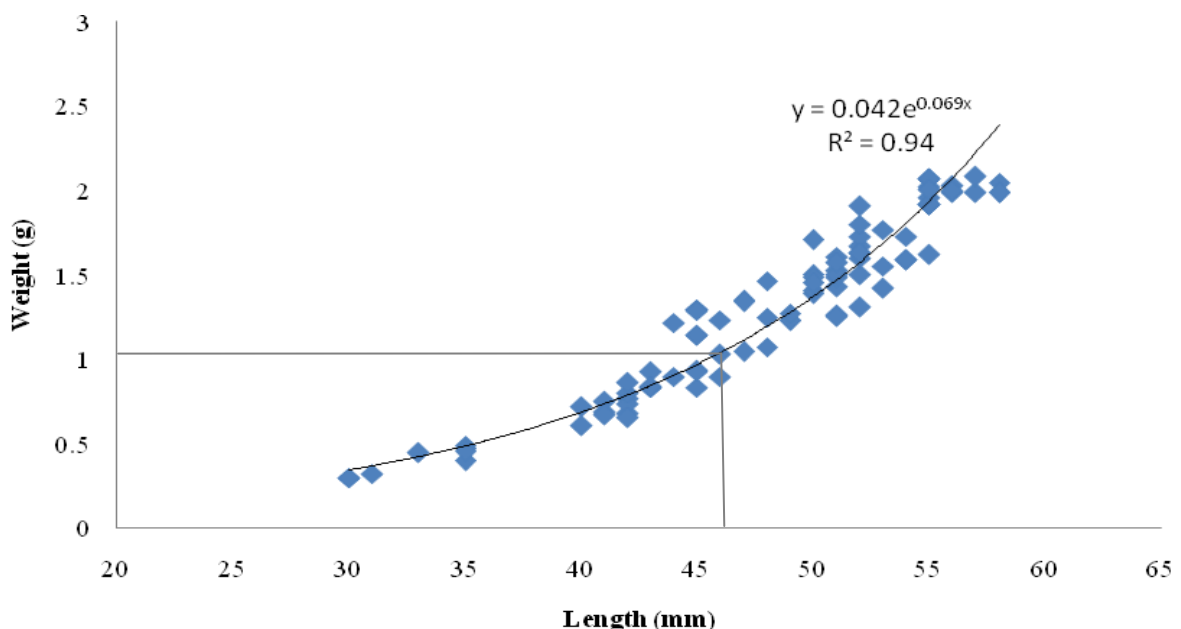


Figure 3.7: Length against weight relationship of *D. labrax* caught on the ebbing tide at Creekmouth.

The quantitative estimates for Creekmouth based on the assumptions stated, shows a large range in total fish fry density between the three survey days (Table 1.3). The maximum densities of fry fish, 2.17 fry per m<sup>2</sup> were found on the 16.07.07. Biomass estimates were conducted for *D. labrax* using length/weight relationships collected during sampling. This showed a maximum juvenile *D. labrax* biomass of 2.17 g m<sup>2</sup> on the 16.07.07. This could not be repeated for other species due to insufficient length/weight data.

Table 1.3: Quantitative estimates of the total fish density including the relative compositions of *D. labrax* and freshwater fish, and *D. labrax* biomass estimates from Creekmouth on the 16.07.07, 17.07.07 and 27.07.07.

Date	<i>D. labrax</i> (N)	<i>D. labrax</i> (m <sup>2</sup> )	<i>D. labrax</i> (g m <sup>2</sup> )	FW Fish (N)	FW Fish Density (m <sup>2</sup> )	Total (N)	Total fish fry Density (m <sup>2</sup> )
16th	2898	2.0072	2.1678	163	0.1129	3131	<b>2.1686</b>
17th	1113	0.7709	0.8326	51	0.0353	1177	<b>0.8152</b>
27th	63	0.0479	0.0518	35	0.0266	98	<b>0.0746</b>

where FW= freshwater, and N= abundance

Overall only one age class, this year juveniles (0+), was found for all of the species caught at Creekmouth site on the flooding or ebbing tide (Appendix 5.2). The differences in the length of *D. labrax* on the flood (Figure 3.8) and ebb (Figure 3.9) tides, were however statistically significant (F=456.7, df= 1, P <0.0001), with lower lengths occurring on the flooding tide.

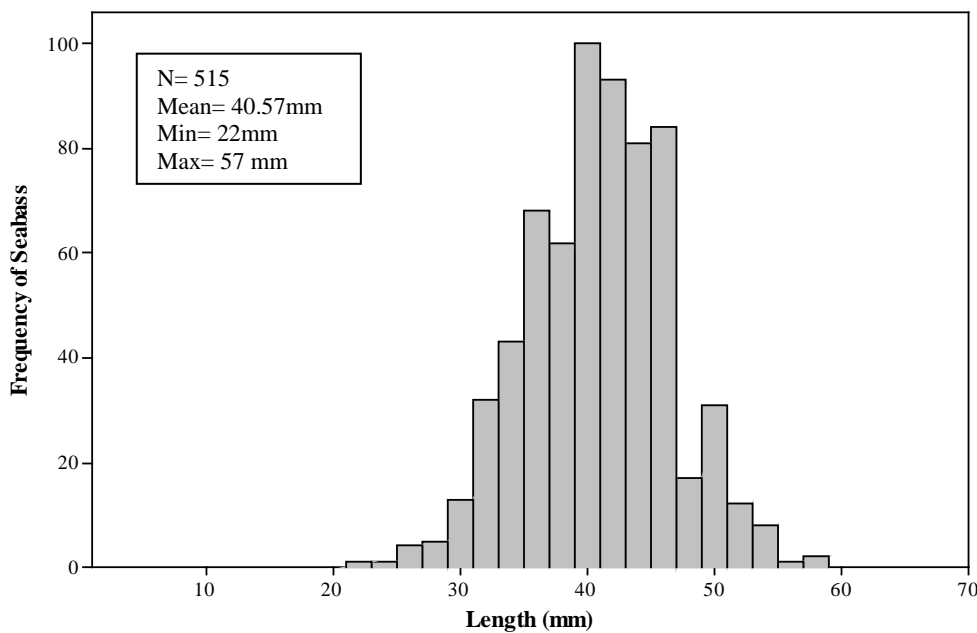


Figure 3.8: Histogram of total *D. labrax* lengths from Creekmouth on the flooding tide.

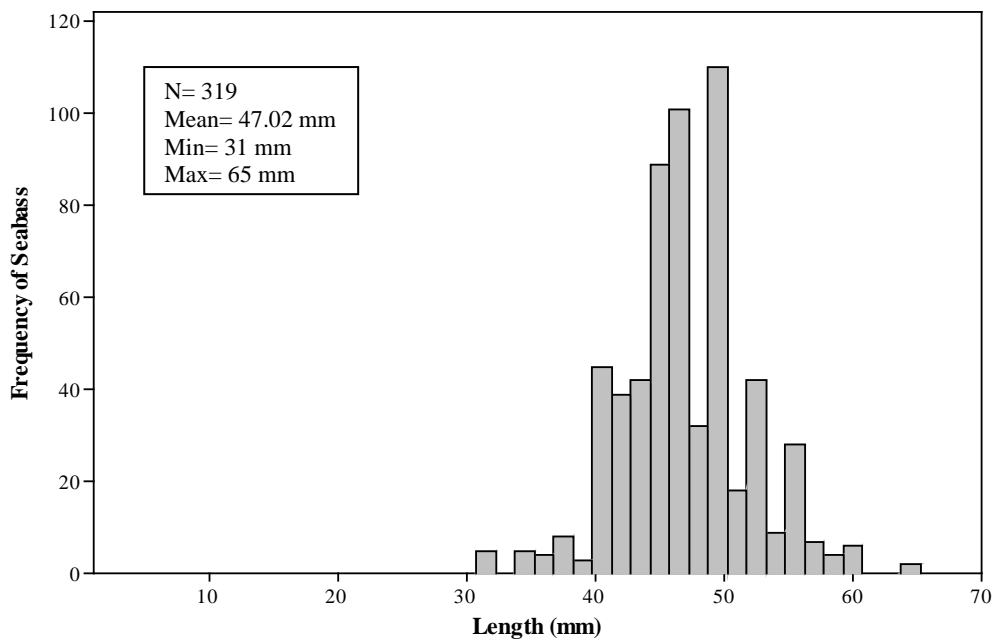


Figure 3.9: Histogram of total *D. labrax* lengths from Creekmouth on the ebbing tide.

Differences in length between the flood and ebb tides for *L. leuciscus* ( $F= 3.15$ ,  $df= 1$ ,  $P>0.05$ ) and *R. rutilus* ( $F= 0.33$ ,  $df= 1$ ,  $P>0.05$ ) were not significant. These were the only other species with sufficient data for statistical comparisons.

The fyke net sampling the adjacent natural intertidal marsh caught a total of 366 fish from 3 species (*R. rutilus*, *D. labrax* and *P. flesus*) on the two days of sampling (16.07.07 and 17.07.07: Appendix. 5.3). Of this total catch 97.3% of the fish caught were *D. labrax*, ranging in size from 31mm to 56mm. One 466mm adult *D. labrax* was captured; this was much larger than any specimen caught in the Creekmouth site (Figure 3.10).



Figure 3.10: Large *D. labrax* captured in adjacent intertidal marsh



The fyke net in the main river channel caught a total of 12 fish from 3 species (*R. rutilus*, *P. flesus* and *A. Anguilla*) on the two days of sampling (16.07.07 and 17.07.07: Appendix 5.4). Of this total catch 75% were *P. flesus*. This was the only site where no *D. labrax* were caught.

### 3.3.2. Gut analysis from Creekmouth

The gut contents of *D. labrax* from Creekmouth on the ebbing (N=89) and flooding (N=90) tide were analysed (Table 1.5). A total of 11 prey taxa were found in guts of fish caught on the ebbing tide (including unidentified Isopoda species, Diptera pupa, Isopod Sphaeromatid and Diptera Ceratopogonida) and 15 prey taxa from fish caught on the flooding tide (with the addition of Coleoptera, Hemiptera, Arachnida and Odonata). With a total of 16 prey taxa recorded over both tides. The main food source of *D. labrax* caught on the ebbing and flooding tide was an unidentified Diptera species, differentiated by a distinct green colouration (FO%= 97.87, G%= 55.01 and FO% 44.57, G%= 39.26 respectively). Further classification was not possible due to the deterioration of the specimens from the initial stages of gut digestion.

Table 1.5: *D. labrax*, gut content averaged from the three days sampling on the flood and ebb tides at Creekmouth (Appendix 5.5).

Species List	Flood (n=89)			Ebb (n=90)		
	FO (%)	N%	G%	FO (%)	N%	G%
Diptera (green)	44.57	33.10	39.26	97.87	49.06	55.01
Unidentified Isopoda	23.91	10.68	27.75	54.26	6.38	15.68
Diptera pupa	9.78	4.27	8.00	40.43	9.62	17.04
Calanoid copepods	13.04	16.73	0.02	22.34	9.10	0.01
Isopoda Sphaeromatid	9.78	7.83	13.31	20.21	3.56	5.71
Diptera Ceratopogonida	4.35	3.91	2.54	17.02	10.93	6.71
Harpacticoid copepods	9.78	14.95	0.00	17.02	10.20	0.00
Chironomid midge larvae	1.09	0.36	0.41	4.26	0.26	0.29
Gammarus	6.52	3.56	5.19	2.13	0.16	0.22
Other Diptera	0.00	0.00	0.00	2.13	0.10	0.12
Corophium	4.35	2.49	0.43	1.06	0.63	0.10
Arachnida	1.09	0.36	0.44	0.00	0.00	0.00
Hemiptera	1.09	0.36	0.55	0.00	0.00	0.00
Coleoptera	2.17	0.71	1.02	0.00	0.00	0.00
Terrestrial fly	1.09	0.36	0.42	0.00	0.00	0.00
Odonata	1.09	0.36	0.67	0.00	0.00	0.00

where: FO%= Frequency Occurrence, N%= Numerical composition and G% Gravimetric composition.

The gut contents of *R. rutilus* from Creekmouth on the ebbing (N=29) and flooding (N=27) tide were analysed (Table 1.5). A total of 10 prey taxa were found in guts of fish caught on the ebbing tide and 10 prey taxa from fish caught on the flooding tide (with the substitution of Unidentified Isopoda for Chironomid midge larvae ). A total of 11 prey taxa recorded over both tides, meaning the *R. rutilus* diet less diverse than the *D. labrax* diet. The main food source of *R. rutilus* caught on the ebbing tide was unidentified Isopod species (Ebb G%= 20.11) and on flooding tide was Diptera pupa (Flood G%= 21.33).

Table 1.4: *R. rutilus* gut content averaged from the three days sampling on the flood and ebb tides at Creekmouth

Species List	Flood (n=29)			Ebb (n=27)		
	FO (%)	N%	G%	FO (%)	N%	G%
Diptera (green)	13.79	8.43	15.76	40.74	17.58	16.25
Diptera Ceratopogonida	6.90	2.41	2.46	22.22	13.19	6.67
Calanoid copepods	10.34	12.05	0.02	3.70	1.10	0.0008
Harpacticoid copepods	13.79	37.35	0.01	14.81	17.58	0.0024
Diptera pupa	17.24	7.23	21.33	7.41	4.40	6.41
Isopoda Sphaeromatid	13.79	4.82	12.91	18.52	6.59	8.73
Corophium	10.34	4.82	11.07	6.90	5.49	6.24
Unidentified Isopoda	~	~	~	14.81	10.99	20.11
Chironomid midge larvae	3.45	1.20	2.21	~	~	~
Phytoplankton	62.07	13.06	11.13	72.41	15.19	14.88
Unidentified macrophytes	40.01	8.63	21.3	70.17	7.89	21.07

where: FO%= Frequency Occurrence, N%= Numerical composition and G% Gravimetric composition.

A niche percentage similarity comparison between the gut content of the estuarine marine species *D. labrax* and freshwater species *R. rutilus* produced a niche overlap of 39.99% on the flood tides and 50.72% on the ebb tides. This indicates a greater degree of niche overlap on the ebbing tide, suggesting the diets between the *D. labrax* and *R. rutilus* were more similar within the Creekmouth site than outside the site.

The differences between the gut fullness of *D. labrax* between the flood and ebb tide was highly significant for all three survey days with greater fullness on the ebb tides, 16.07.07 (W= 1087, P< 0.0001), 17.07.07 (W= 378, P<0.0001) and 27.07.07 (W= 170, P<0.0001: (Figure 3.11). Differences in the gut fullness were also significant between the ebbing (W= 276, P<0.01) and flooding tides (W= 289, P<0.005) on the 17.07.07 and 27.07.07, which could be due to the atypical high freshwater flows. The *R. rutilus* gut fullness between the ebbing and flooding tide from sampling on the 27.07.07 was also significant (W=488, df= 1, P<0.0001), with greater gut fullness on the ebbing tide.

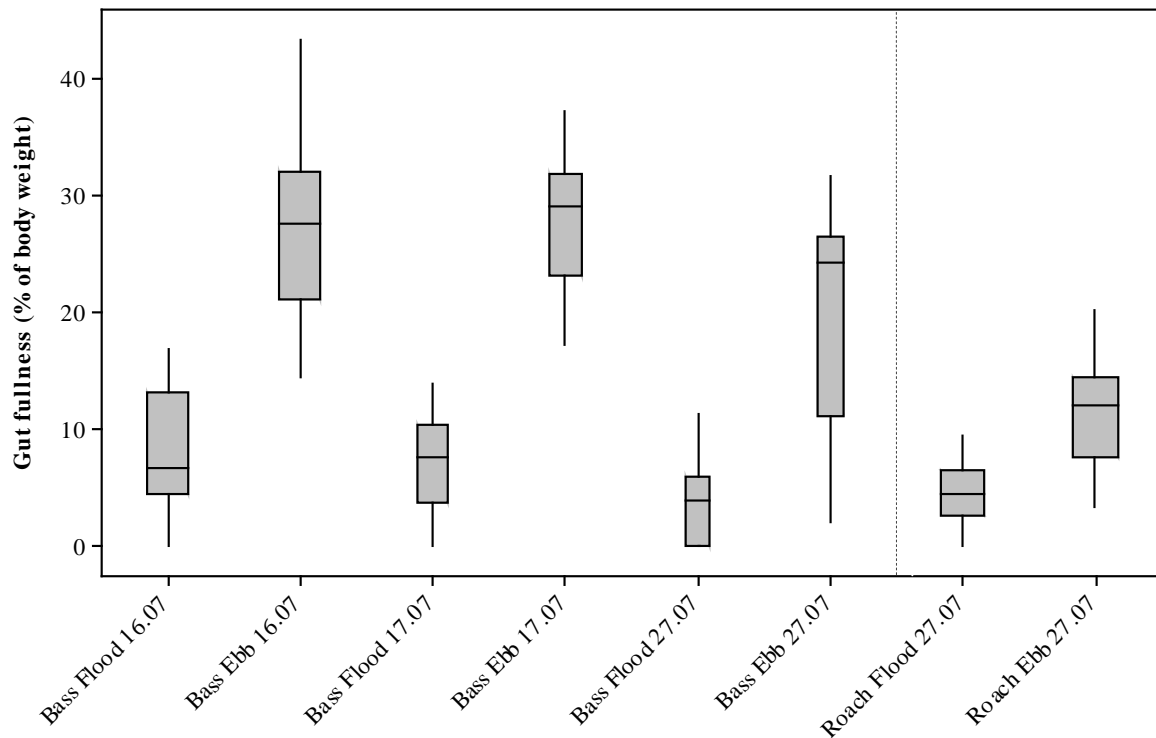


Figure 3.11: Box and whisker plots of the *D. labrax* and *R. rutilus* gut fullness on the flood and ebb tides, with box width proportional to sample size.

### 3.3.3. Vegetation at Creekmouth

The intertidal area created at Creekmouth was left to colonise naturally using the rivers own seed-bank (Figure 3.12). This method proved very successful with the rapid colonisation of many native species. In broad terms there are four distinct zones of vegetation present at the Creekmouth site (Webb, pers. comm.);

- The terrestrial zone which extends to approx 0.5 m below spring high tide level (using the high tide debris line as indicator of maximum high tide level). This comprised of common herbs, including *Lotus corniculatus*, *Plantago major*, *Tripleurospermum maritimum*, *Chrysanthemum leucanthemum*, *Artemisia vulgaris* and *Sanguisorba minor*.
- The marginal wetland zone, where tidal inundation is the main factor affecting species composition. This included *Aster tripolium*, *Apium graveolens*, *Beta vulgaris*, *Ranunculus scleratus*, *Elymus pycnanthus*, *Agrostis stolonifera* and *Atriplex hastata*.
- The final 2 zones are the lowest in the tidal regime in the area where maximum silt deposition occurs. These areas are dominated by emergent vegetation. These two vegetation zones are less distinct and are comprised of stands of *Phragmites australis*, and *Scirpus maritimus*, with the *S. maritimus* generally occurring furthest down the foreshore.



A)



B)



C)



D)

Figure 3.12: Some of the vegetation established at Creekmouth: A) Sea mayweed (*Tripleurospermum maritimum*): B) Sea aster (*Aster tripolium*) in flower: C) Common reed (*Phragmites australis*) and D) Sea aster (*Aster tripolium*).

### 3.3.4. A13 results

A single days sampling was conducted at the A13 site, on the 26.07.07. Between the flooding and ebbing tides 45 fish were caught from 4 different species (Table 1.6: Appendix 5.6). Of the fish caught 58% (26 individuals) were freshwater species (*R. rutilus* and *L. leuciscus*) and only 4% (2 individuals) were true marine estuarine dependent species (*D. labrax*). The species richness was greater on the flooding tide with 4 species recorded, compared with only 2 species recorded on the ebbing tide. The zone of vegetation was smaller and steeper at the A13 site. The lower (neap) tidal amplitude on the 27.07.07 (approximately 5.8m) meant that the vegetated zone was only inundated for a short period of time. This meant that the nets fished sub-optimally due to the limited period that fish could utilise the area.

Table 1.6: A13 site species summary

Group	Family	Species	Common name
<b>Marine Estuarine dependent species</b>	Serranidae	<i>Dicentrarchus labrax</i>	Sea bass *
<b>True Estuarine species</b>	Gobiidae	<i>Pomatoschistus microps</i>	Common goby †
	Pleronectidae	<i>Platichthys flesus</i>	Flounder †
<b>Freshwater species</b>	Cyprinidae	<i>Rutilus rutilus</i>	Roach †*
		<i>Leuciscus leuciscus</i>	Dace †

where \* was found on the ebbing tide and † was found on the flooding tide

### 3.3.4. Environmental parameters

Data from a flow gauging station in the upper reaches of the tidal Thames clearly shows the large increase in freshwater flow between the sampling days in late July (Figure 3.13). Before the sampling on the 16.07.07 and 17.07.07 the freshwater input averaged between 50-100 cumecs per 15 min, however this increased to a peak of 280 cumecs per 15 min on the 27.07.07 at 1.45am, the day of sampling.

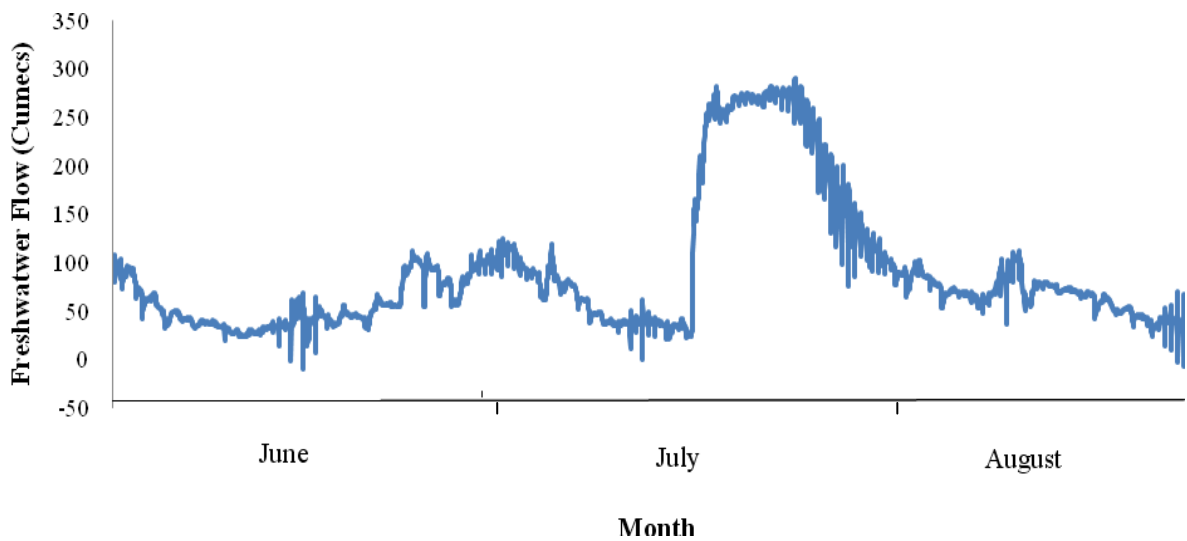


Figure 3.13: The freshwater flow (cumeecs) from Kington monitoring station between June-August 2007

Chloride is used as an indicator of the presents of seawater, because of the eleven chemicals which make up seawater chloride is the most abundant with approximately  $19.135 \text{ g kg}^{-1}$  of water, followed by sodium with  $10.76 \text{ g kg}^{-1}$  of water (Turekian, 1977). The data from an EA Automatic Quality Monitoring Station (AQMS) at Woolwich (closest station to the sites) clearly indicates a sharp decline in chloride from 3508 ppm on the flooding tide on the 19.07.07 to 147 ppm on the flooding tide on 23.07.07 (Figure 3.14). This shows the estuary was dominated by freshwater flow during sampling on the 26.07.07 and 27.07.07.

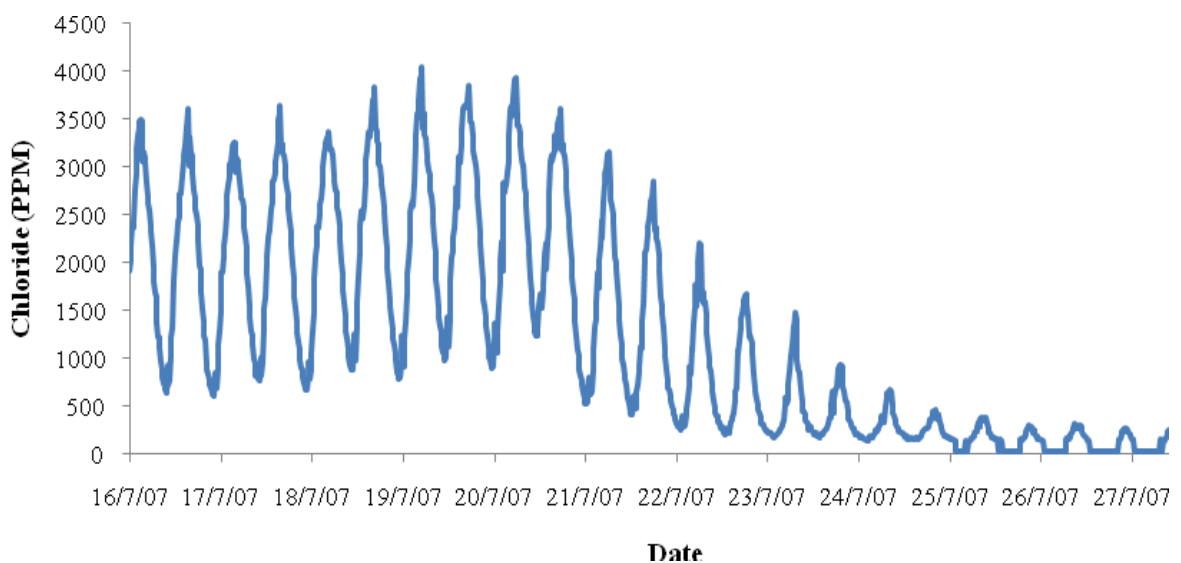


Figure 3.14: Chloride (ppm) from Woolwich monitoring station from 16.07.07-28.07.07.

The high rainfall events which begin on the 23.07.07 July increased the dissolved oxygen present in the water, from a mean of 44.7 % pre-rainfall to a mean of 64.2 % during the high rainfall (Figure 3.15).

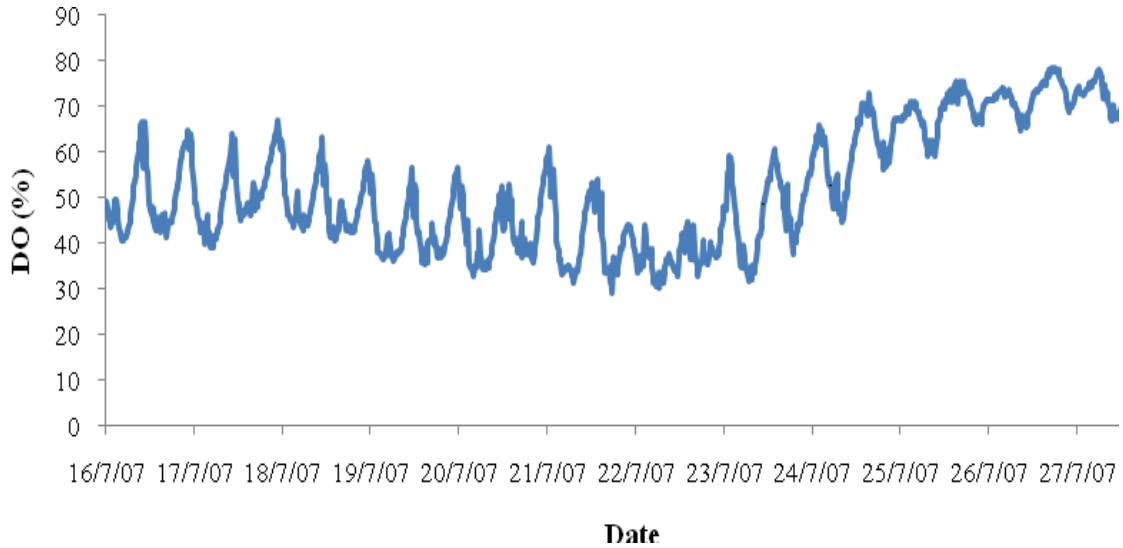


Figure 3.15: DO (%) from Woolwich monitoring station from 16.07.07-28.07.07.

The temperature was shown to decrease during the heavy rainfall events to an average of 18.5°C compared with a pre-rainfall average of 19.8°C (Figure 3.16).

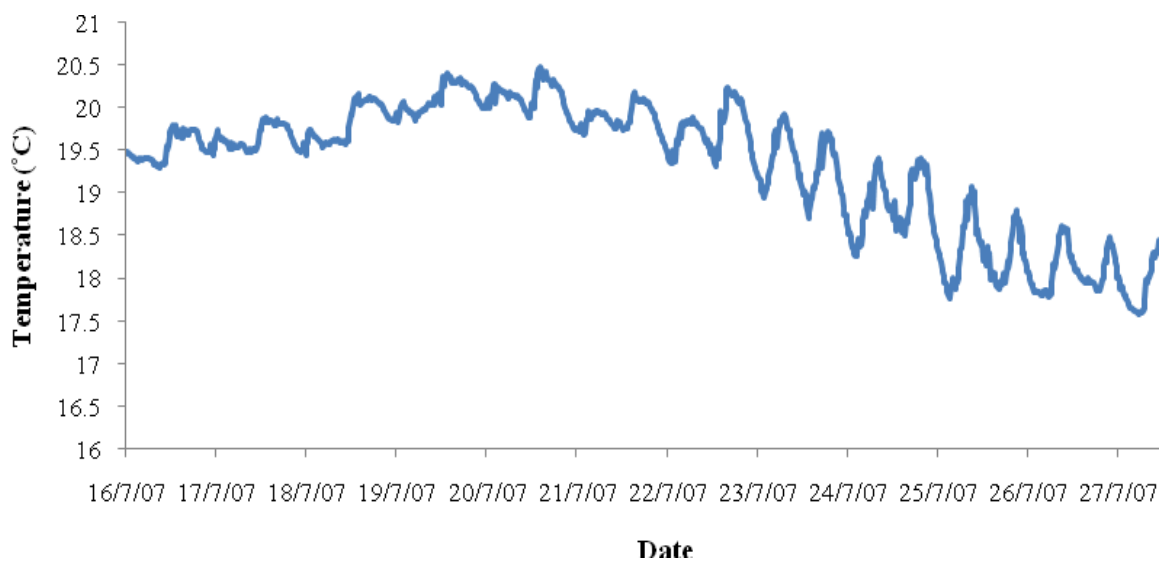


Figure 3.16: Temperature (°C) from Woolwich monitoring station from 16.07.07-28.07.07.

## 3.4. Discussion

### 3.4.1 Fish utilisation of regeneration sites

This study highlights the importance of regenerated intertidal habitat for juvenile fish. All fish caught at the Creekmouth and A13 sites were this year's juveniles, therefore from the 0+ age class. This corresponds with work by Shenker and Dean (1979) who reported it was predominately larval and juvenile fish using the intertidal creek habitats. Kelley (2002) even found 1+ *D. labrax* appeared to leave the shallow creeks systems following the arrival of the 0+ age group.

The length-frequency distribution of *D. labrax* shows slightly different length cohorts within the overall uni-modal distribution (Appendix 5.7). This supports previous research suggesting the juvenile *D. labrax* enter the estuary in multiple waves (Sabriye *et al.*, 1988). This could be the result of recruitment from different spawning grounds and/or due to females releasing ripe ova in two or three batches over a 2-3 week period (Mayer, 1987).

The high fish fry utilisation recorded at Creekmouth site supports the theory that optimal fish utilisation occurs in systems with gradual slopes and many drainage features which flood and drain on a regular tide (Peterson and Turner, 1994). The site also provides a shallow area in a side arm to the main Thames channel, meaning this area could function as a hideaway in periods of hypoxia in the main River Thames, as well as for food production and refugia (Moeller and Scholz, 1991). The shallow profile of the Creekmouth site could also attract 0+ *D. labrax*, as they are known to actively avoid low temperatures (Cabral and Costa, 2001). The AQMS at Woolwich recorded a mean temperature during sampling of 19°C. The optimum range for *D. labrax* growth is known to occur between 22-24°C (Barnabe, 1990). The water temperature with the Creekmouth site would have been greater than the main channel due to the decrease in depth.

Dominance of a small number of generalist fish species is regarded as a key feature of relic tidal marshes (Haedrich, 1983). The most common species recorded during sampling was *D. labrax*, with approximately 90.7% of the total abundance. *D. labrax* abundance is known to be highly variable within estuaries depending on year-class survival, which is believed to be strongly dependent on seawater temperature (Pickett and Pawson, 1994). The greatest recruitment years for *D. labrax* have typically resulted from a warm and settled spring (Kelley, 1979: 1986). These conditions were found in the South East of England this year, suggesting this has been an optimal year for *D. labrax* recruitment to inshore nurseries. Seasonal variation also plays a huge role in the presence and proportions of species utilising the marsh. The abundance of *D. labrax* shown in this study corresponds with this, as they are shown to arrive in estuaries (typically between 10-15mm) in May-June, approximately one month after being transported into coastal waters from offshore spawning (Kelley, 1988: Jennings and Pawson, 1992).

*P. flesus* are also shown to migrate up estuaries in May-June (Colclough *et al.*, 2002), yet very few were caught during sampling. This may suggest a poor recruitment year, or may be due to with the high riverine inputs, as it is believed their downstream migration in the autumn is stimulated by the heavy rainfall (Araujo *et al.*, 2000). It could however, result



from survey sample limitations, as some *P. flesus* was visually observed within the Creekmouth site.

*P. flesus* were not caught in the high intertidal fyke nets, suggesting their distribution was limited to the main channel within the Creekmouth site. This could be because their movements were limited by problems exiting the Creekmouth main channel because of its steep profile. This is supported by previous work on the Greenwich millennium, a series of tiered banks, completed in 1990 as new tidal defences and intertidal habitat. Fish surveys on the terraces have demonstrated an abundance of fish fry utilising them, however limited demersal species due to problems manoeuvring the 90° angle frontage (Colclough, pers. comm.).

This study confirms juvenile *D. labrax* appear estuarine dependent, actively favouring shallow creeks and marshes in their first summer (Kelley, 1988). They were also shown to move onto the intertidal areas on the flood tides and return to deeper water on the ebb (Lyndon *et al.*, 2002). The likely cause for this behaviour is high foraging profitability and predator avoidance (Irlandi and Crawford, 1997). The systems offer refugia to the fry as they are too shallow for larger piscivorous fish to enter, due to their own risk of predation or stranding (Paterson and Whitfield, 2000). This phenomenon has however been shown to change at night, when reduced visibility decreases the risk to the piscivorous fish (Copp and Jurarda, 1993; Colclough *et al.*, 2005). The only large specimen of *D. labrax* (466mm) caught during this survey was found associated with the adjacent marginal saltmarsh, where it is likely it was predated upon juvenile fish utilising the marsh. As a large individual, this predator is likely to remain within the safety of the main channel.

A significant difference was found in the length of *D. labrax* between the flood and ebb tide at Creekmouth, with slightly larger fry found on the ebbing tide. This was an unexpected result, and not fully understood. Possible reasons for the difference could be due to sampling differences between the flood and ebb tides, as larger *D. labrax* are noted as better at avoiding mobile netting methods such as push and seine netting (Hume, pers. comm.). Or the larger *D. labrax*, as stronger swimmers may be more independent of the tides enabling them better access to the high food profitability within the site. Fish species are known to have the ability to condition themselves to utilise new habitat opportunities (Warbuton, 2003). Further data would be required to investigate this phenomenon further.

The success of the Creekmouth site for fish utilisation is clearly evident in the abundance of fish caught. The A13 site however had a significantly smaller catch. It is important to highlight the A13 survey results were limited by the methods used (e.g. no blocking net due to site topography), the tides (the site was not fully inundated during sampling) and particularly by the extreme freshwater flows. On the day of sampling at the A13 site the Thames Estuary and its tributaries were in an atypical state in terms of salinity and flow. The high freshwater flows preceding the 26.07.07 and 27.07.07 are likely to have resulted in the low fish utilisation recorded. The main water quality parameters known to effect the distribution of fish are salinity, temperature, turbidity and dissolved oxygen (Blaber and Blaber, 1980). Thiel *et al.*, (1995) found temperature was the best predictor of total abundance and salinity influenced species richness. The high freshwater flows are highly likely to be responsible for the dramatic reduction in *D. labrax*, for which optimal conditions are salinities between 13-17 psu and 22-28 psu (Cabral and Costa, 2001). This study clearly shows stochastic extreme conditions such as reduced salinity from freshwater storm flows can greatly affect both abundance and species richness within an estuary. Therefore, it is

likely this is not a clear or accurate representative of true fish utilisation of site, and further surveys would be required.

Surveying does not generally occur during atypical conditions such as the extreme freshwater flows, therefore no comparative data exists. It is likely that the distribution of the true estuarine and marine fish becomes restricted as the freshwater flow becomes dominant. Once the estuary returns to its typical summer-time freshwater-saline distribution, it is likely that the fish distributions also return to normal. Anecdotal observations by Agency Fisheries staff are that the converse of this is true in drought conditions, with marine and estuarine dependent species penetrating further up-river (Cousins, pers. comm).

The rapid success of the natural vegetation colonisation has helped the habitat become successful for fish by increasing the habitat structural complexity (Figure 3.17). Site topography, elevation and tidal inundation are critical to the success of vegetation at the site. Aquatic vegetation has been shown to be an important determinant of fish habitat selection (Grenouillet and Pont, 2001). Atrill *et al.*, (1999) found a clear link between biological diversity and habitat complexity, e.g. increased faunal richness demonstrates a clear correlation with diverse invertebrate communities. Sediment characteristics and, vegetation cover and heterogeneity, have also been shown to influence fish distribution through prey availability (Marchand, 1993). The gradual slope produced in the design seems to have aided rapid early vegetation colonisation, suggesting a strong seed-bank in the local area.



Figure 3.17: Creekmouth before vegetation establishment and after natural vegetation colonisation (EA Archive).

The importance of vegetation as refugia for juvenile fish was also evident when comparing the main channel to Creekmouth data. The main channel, devoid of vegetation and susceptible to the greatest flow, when surveyed only had larger *R. rutilus* (between 140-172mm), *P. flesus* (between 46-80mm) and an *A. anguilla* (700mm) present (Appendix 5.4). Of the main channel catch 75 percent were *P. flesus*, however this is a benthic species more susceptible to capture using a bottom set fyke net. Both *P. flesus* and round-fish juveniles use selective tidal stream transport to move up the estuary. *P. flesus* is able to avoid the strong currents and ebbing tide by lying flat on the riverbed, whilst round-fish fry need to seek areas of reduced velocity, normally associated with continuous tidal foreshore and marginal vegetation (Petrou, 1999). Data from the Great Ouse showed the lack of macrophytes to

provide this refuge and food supply, has had a detrimental effect on the recruitment of 0+ fish (Mann and Bass, 1997: Mann *et al.*, 1997: Copp, 1997). On the Thames Estuary geomorphology change such as dredging and encroachment (as narrowing rivers tends to increase flow) now means few marginal areas of the channel have a mean velocity below  $0.6 \text{ m s}^{-1}$ , the maximum sustainable swimming speed for *L. leuciscus* fry (Wagg, 1996: Colclough *et al.*, 2000: 2002).

### 3.4.2. Gut contents

*D. labrax* is a member of the family Serranidae. It is characterised by an anterior dorsal fin with eight or nine spiny rays, a posterior dorsal fin with one spiny ray and 12-13 branched rays and a caudal fin deeply notched with 13-17 rays and blunt head with large mouth (Wheeler, 1992). *D. labrax* is an estuarine dependent marine species and one of the most abundant and commercially important fish in European waters (Pickett and Pawson, 1994), with a range expanding from Scotland and Denmark south to Morocco and throughout the Mediterranean. *D. labrax* is however a K-selected species with slow growth and late maturity, and therefore deemed susceptible to over-exploitation (Kelley, 1988). The Bass Specified Areas Prohibition of Fishing Order 1990 (MAFF, 1990) prohibits fishing for *D. labrax* by fishing boat within 34 UK specified estuarine nursery areas when juveniles are present.

*R. rutilus*, is a member of the Cyprinidae family. It is characterised by a laterally flattened body, with a small head, small slightly oblique terminal mouth, reddish eyes and pelvic fins below the dorsal fin (Muus and Dahlstrom, 1971). The roach is one of the most abundant freshwater fish in lakes and rivers in the UK, partly due to its tolerance of pollution. Its wide spread distribution makes it economically important to angling. These two species due to their abundances within and around the Creekmouth site were used as representatives of the marine and freshwater systems in gut analysis.

The analysis of *D. labrax* and *R. rutilus* gut contents showed a clear functional linkage in the occurrence and affinity of fish fry to restored intertidal habitat. This determines it is not just a stochastic consequence of tidal transport (Simenstad *et al.*, 2000). Despite only being able to utilise the Creekmouth for 1-2 hours on the spring tide, *D. labrax* were shown to be extensively feeding on invertebrates, predominately Diptera, Diptera pupa and Isopod species. This shows a marine species utilising a transitional habitat and feeding predominately on freshwater invertebrates present in the reedbeds. *D. labrax* are known as opportunistic feeders (Pickett, 1989), however this data is different from previous research from the Mont Saint Michel Bay, France (Lafaille *et al.*, 2001) and Blackwater Estuary, UK (Fonseca, 2003) which showed 0-group *D. labrax* feeding mainly on true estuarine species such as *Crangon crangon* (the brown shrimp) and *Orchestia gammarellus* (the beach flea). However, data dating back to 1973 by Labourg and Stequert in France did find important quantities of insects in the diet of *D. labrax*. This illustrates the importance of these habitats within estuaries, as they offer functional overlap for both marine and freshwater species.

The Creekmouth and A13 regeneration sites are both at the top of the intertidal range meaning the 0+ fish could only use the intertidal habitat approximately 5% of the time.

Despite this the study confirmed previous research that primary and secondary production in these areas plays an essential role in the feeding of 0+ *D. labrax* (Laffaille *et al.*, 2001).

The analysis of gut fullness of the flood and ebb tide clearly demonstrated the juvenile *D. labrax* and *R. rutilus* were using the Creekmouth site as an active feeding ground. *R. rutilus* had median gut fullness relative to body weight on the flood tide of 4.53%, which increased statistically significantly to 12.07% on the ebb tide. An even greater statistical significance was seen with the *D. labrax*, with flood tide median of 6.41%, compared with an ebb median gut fullness of 27.14%, over four times greater. The differences in gut fullness are more significant than previous UK research on the Blackwater Estuary (Fonseca, 2003), and greater than the average 8% increase found in *D. labrax* body weight feeding in saltmarsh in Mont Saint Michel Bay, France (Laffaille *et al.*, 2000). West and Zedler (2000) however have found *Fundulus parvipinnis* (the California killifish) with access to saltmarsh consuming six times more than those with restricted access.

The invertebrate prey removed from the gut contents at Creekmouth were consistently nearly one order of magnitude heavier than the same species removed from the guts of *D. labrax* feeding in the Blackwater Estuary (Fonseca, pers. comm.). These magnified effects in gut fullness and prey size could be due to excess nutrients in the Estuary. Barking Creek is situated in close proximity to Becton sewage treatment works (STW) outfall pipe. Beckton is the largest sewage treatment works in the UK, serving sewage from 3.4 million Londoners every day (Thames Water, 2007). Treated effluent from STW are show to still contain up to 15 mg l<sup>-1</sup> of phosphate and 30 mg l<sup>-1</sup> of nitrate, which would be discharged via the outfall pipe. If this effluent was not adequately diluted before moving into this tidal backwater, it is possible it could cause enrichment of the area. The relatively low diversity of invertebrates found in the gut contents and dominance of Diptera species (commonly used as an indicator of organic pollution: Arimoroa, *et al.*, 2006) may also suggest nutrient enrichment. If artificial enrichment is occurring it appears to be having a positive effect on local flora and fauna, by increasing biomass and productivity. Fish abundance is shown to increase with local increases in the abundance of food e.g. invertebrates (Grenouillet and Pont, 2001). Pioneer vegetation may be more productive than relic marsh, indirectly contributing to a higher prey species production and biomass (Boesch and Turner, 1984). These reasons could explain the high fish utilisation within the Creekmouth site.

Seasonal variation is known to affect feeding ecology; depending on the prey is available (Bozeman and Dean, 1980: Claridge and Potter, 1983: Madon *et al.*, 2001), and the time of day, with higher gut vacuity between 8.00am and 8.00pm (Cabral and Costa, 2001). Temperature influences the ability of *D. labrax* to digest their food, which in turn affects their feeding rate (Russell *et al.*, 1996: Cabral and Costa, 2001), as does turbidity because *D. labrax* are visual predators. The high gut fullness therefore suggests optimal conditions (temperature and turbidity) were found within the Creekmouth site on the 16.07.07 and 17.07.07 which allowed maximum feeding benefits.

The niche overlap between *D. labrax* and *R. rutilus* showed a greater difference (40%) in diet between the flood tides. This is likely to be because of a greater selection of prey available or because the species were more able to feed on their preferred prey. Whereas on the ebb tide the diets may be more similar (51%) because the prey diversity or availability was lower within the Creekmouth site. Both species exhibited significantly high gut values, indicating that they are able to exploit these areas of restored habitat for feeding. The species utilising the study sites varied with the changes in flow and salinity, but this demonstrates that marine estuarine dependent and freshwater species both make opportunistic use of the resource.

*D. labrax* were shown to exploit the secondary production of the intertidal habitat, via invertebrate prey. *R. rutilus* feeding habits differed as they directly exploited the primary production of the intertidal habitat by feeding on plant material and phytoplankton, as well as feeding on secondary production.

Small changes in diet could also be seen between age ranges of *D. labrax*, with the smallest individuals feeding predominately on copepods. This is commonly observed as the smaller *D. labrax* are limited by their gape size (Fonseca, 2003).

### **3.4.3. The Value of Intertidal habitat**

McHugh (1966) estimated that two thirds of commercially important fish catch were dependent on estuarine habitat in their juvenile years. Research on the Forth Estuary estimated it supported approximately 0.5% of North Sea stocks of certain size-classes of commercially important species (Elliott *et al.*, 1990). Nixon (1980) produced evidence that commercial landing of estuarine dependent species correlated with marsh area available to them in the area. The reason for estuarine dependence in marine juveniles is not fully understood, but possible reasons include behavioural mechanisms to reduce osmotic stress (Lutz, 1975), reduced predation threat due to the increased turbidity and refugia (Blaber and Blaber, 1980) and/or feeding related gains due to high prey densities within estuaries (Elliott *et al.*, 1990).

However, despite the obvious value of intertidal habitats little work has been done to economically quantify the value of them (Blaber and Blaber, 1980; Haedrich, 1982). Costanza *et al.*, (1997) estimated that estuaries as one of the most expensive habitats to recreate due to the large amount of functions they perform both naturally and for humans, and valued them accordingly at \$22,832 per hectare per year. Bell (1997) estimated the value of an acre of saltmarsh to a recreational fin fishery alone in Florida laid between \$981-6471 per year, depending on location.

The Rio Earth Summit in 1992 called for a precautionary principle to be used to safeguard and manage our natural resources and habitats. Many believe that success also requires accurate economic valuation, to help politically weight the importance of these habitats in relation to other options (Ledoux and Turner, 2002). This can be achieved by conducting a cost-benefit analysis (CBA) to give a value of the economic efficiency of potential schemes in monetary terms. However, it should be remembered that it is difficult to assign costs and benefits to environmental functions, and therefore this should only be use as an aid in decision making (Turner *et al.*, 2001).

Saltmarshes, in particular are very dynamic and heterogenic habitats, so making short-term quantitative estimates is often highly inaccurate. The range of density estimates found in the quantitative fish assessments of Creekmouth demonstrates this. Many factors could be influencing the distribution of *D. labrax* within the estuary such as depth, salinity, temperature, turbidity, prey availability and/or tidal cycles etc (Cabral and Costa, 2001). The observations obtained from Creekmouth should therefore be viewed with caution, as it is only based on three days of sampling, one of which was atypical. Further monitoring would be required to better predict the fish utilisation in monetary terms throughout the year. This

initial study is important in illustrating that even small areas of marsh have economic value. A very crude calculation to show this can be made if you simplify and assume that:

- A survival rate of 50% of the 2.17 fish fry per m<sup>2</sup>, the highest density recorded during sampling at Creekmouth. This is the average survival rate of larval *D. labrax* in artificial conditions with appropriate environmental conditions and feeding strategies (Chatain, 1997). This does not take into account the effect of atypical environmental conditions or predation.
- The Creekmouth site (and its intertidal habitat) helped achieve 50% survival of *D. labrax* to adulthood, by providing increased refugia and feeding profitability during the vulnerable early life stages.
- The restored area of intertidal foreshore habitat resulted in an increase in fish production of the estuary, and not just displacement from other resources.
- The commercial value of *D. labrax* at £5.64/ kg (DEFRA, 2002), and the average size of *D. labrax* caught commercially as 36 cm and approximately 800kg (Colclough, pers. comm.).

Under these very broad assumptions the Creekmouth site could help in the survival of 1.085 fish per m<sup>2</sup> to a commercially catchable size, providing an economic commercial value of the marsh of £4.90 per m<sup>2</sup>. If survival rates decrease to 10%, the value of the marsh would be £0.98 per m<sup>2</sup>. Using these two different survival rates (10% and 50%) values the intertidal habitat within the Creekmouth site, for *D. labrax* production only, are between £ 1,498- 1,660 respectively.

The diverse range of functions and species associated with saltmarshes make them very challenging systems to economically value accurately (Ledoux, 2003; Loomis *et al.*, 2000). Taking into account this, CBA for managed realignments on the Blackwater Estuary has shown they provide positive economic advantages compared against conventional flood defence, despite valuing habitats function as uncertain. The CBA model assumed sedimentation rates between 1.5-6 mm y<sup>-1</sup> and carbon credits of £7 t<sup>-1</sup> (Shepherd *et al.*, 2007). This positive economic value, despite undervaluing habitat value, is possible due to the huge range of other functions associated with intertidal areas.

Evidence from the Humber Estuary shows the removal of intertidal saltmarsh and mudflat markedly reduces the carbon and nutrient trapping capacity on the estuary (Andrews *et al.*, 2000; Jickells *et al.*, 2000). With the UK now required to implement large scale decreases in nutrient inputs reaching the North Sea under the OSPAR (Oslo- Paris) Convention and EC Nitrates from Agricultural Directive (91/676/EEC), this in itself has huge monetary value. Global pressure to reduce carbon footprints, also highlights the need to value and assess the carbon storage abilities of intertidal areas within coastal management schemes (Andrews *et al.*, 2006). Unpublished work from a ComCoast PhD project has also recently found intertidal areas are important in removing atmospheric particulate matter, in particular PM<sub>10</sub>; particles of 10 µm or less (ComCoast, 2007). This means under the right conditions creating new intertidal habitat in urban areas could also improve air quality. Intertidal areas also have huge economic value in protecting our coastline, with saltmarsh protecting seawalls valued at £14,800 per hectare (King and Lester, 1995). Other economic values through diversification of saltmarsh include; shellfish and salad crop production (harvesting *Salicornia*) and premium lamb grazing.

### 3.4.4. The Management of Intertidal habitat

The success of the Barking Creek intertidal habitats even within a heavily modified water-body should aid future similar developments. In terms of fisheries benefits each part of intertidal habitat within estuaries could indirectly contribute to commercial offshore fisheries through increased recruitment (Tinch, 2003) and directly contribute to recreational fisheries (Bell, 1997). The significance of benefits to recreational sea-angling should also be flagged up, with recreational sea-angling generating approximately £28 million for the Welsh economy in 2000, compared with approximately £8.8 and £11.8 million from inshore and offshore commercial fishing respectively (Colclough *et al.*, 2004). The impacts of climate change also appear to make the UK very suitable to *D. labrax* production; therefore increasing their preferred nurseries could result in larger *D. labrax* fisheries in the future.

The importance of these inshore intertidal areas for marine fish suggests they should be considered as candidates for Marine Protected Areas (MPA). The UK is committed under OSPAR to designate a network of MPA's to help preserve biodiversity and fish stocks. Including estuaries within these plans seems a logical step in helping to protect valuable nursery habitats and feeding grounds. The nature of these inshore areas would also make them easier to manage and enforce than many offshore MPAs. The MPA network should be working to help safeguard different stages of commercial fish's life history, helping achieve maximum recruitment through nursery protection can only help sea fisheries in the future. The new European Fisheries Fund (EFF) could be used in conjunction with this and other funding streams to help create intertidal habitat. This would offer multi-functional benefits, by providing nursery grounds to help address declining commercial fish stocks, and well as assisting in flood storage capacity in response to sea-level rise.

The knowledge gained in recent years of the value of intertidal habitat in estuaries alone will not safeguard them. In an economically driven country, such as the UK, estuarine encroachment pressure continues to grow. The Habitats Directive does supply a statutory requirement to obtain compensatory habitat if the proposed development can be shown to have a deleterious impact on the site. However, this protection only occurs in Natura 2000 designated sites e.g. SSSI. Statutory protection is currently lacking on undesignated sites. The Environment Agency is developing a National Encroachment Policy to help resist further encroachment into estuaries. This will require new a habitat creation scheme to compensate for encroachment permitted to go ahead. The value of fragmented foreshore habitats is however very hard to establish. A relationship between the size of foreshore habitat and species diversity has been found (Simberloff and Abele, 1982), but this does not take into account the corridor function of small fragmented intertidal areas.

If avoidance of negative environmental impacts cannot be achieved, mitigation is a workable solution for estuarine habitat. However, structural complexity means achieving an identical replacement is very difficult (Ledoux *et al.*, 2000). A method currently being used in the USA is mitigation banking. This is when the value of an intertidal area is quantified as credits to compensate for the losses to habitat function resulting from development. All developer credits are deposited in an account, and therefore can be used at a later date collectively to produce larger scale restoration projects. This scheme is cost-effective as it reduces incremental losses from individual small-scale projects (Ledoux *et al.*, 2000). However, this should not always be done at the expense of small-scale projects, as there is a clear conservation benefit in maintaining a corridor of estuarine foreshore within heavily urbanised rivers (Atrill *et al.*, 1999). Future restoration projects require a holistic and integrated

approach, to view habitat creation within the wider ecological picture and help to re-establish natural migratory corridors by joining fragmented habitats within estuaries.

In 1994 the Thames Estuary Management Plan sponsored by English Nature (now Natural England) set up the Thames Estuary Partnership. This is a consensus building body, with representatives from a wide range of stakeholders on the Thames, including; the Kent and Essex Sea Fisheries Committee, the Port of London Authority and EA. The partnership has been successful in developing a Tidal Fisheries Management Action Plan to help integrate fisheries management for a sustainable future (Colclough *et al.*, 2002). Independent facilitation bodies like this are also vital in managing systems such as estuaries with so many stakeholders. The implementation of similar stakeholder liaison groups of other large estuaries would help achieve holistic management.

The latest re-modelling of water policy led to development of WFD in December 2000. WFD is seen as a new, integrated approach to the protection, improvement and sustainable use of Europe's rivers, lakes, coastal waters, groundwater and transitional waterbodies (Barrera, 2003). WFD has instigated a move away from previous chemically dominated monitoring systems, to a new predominately biological based monitoring system. The Directive introduces ecological objectives to protect and restore (where possible) the structure and function of aquatic ecosystems, through holistic ecosystem management, e.g. with the use of river basin management planning systems (Olsen, 2001). The over-riding prerequisite of the Directive is that Member States are required to achieve 'good ecological status' in all waterbodies by the end of 2015. 'Heavily modified waterbodies' will be exempt from this, but will be required to reach 'good ecological potential. Under these new guidelines the preservation and restoration of intertidal habitats within estuaries could be required to obtain 'good ecological status/potential'. The high association of fish with the habitats studied in this report and with ComCoast, could be developed as an argument under WFD to protect all extant saltmarsh as a necessary element within existing good ecological status/potential (Colclough, per. comm.).

A phenomenon recently noted on the Thames Estuary is the development and extension of intertidal vegetation on hard-engineering structures (Colclough, pers. comm.). The roughness and complexity of the materials used appears to have allowed the accretion of sediment and consequent succession to vegetation. Studies have shown greater invertebrate richness on brick walls compared to concrete or metal walls due to increased roughness (Atrill *et al.*, 1999). The data collected in this study and supporting literature would strongly suggest that fish fry would be utilising these habitats during the tidal cycle, and possibly as part of a migration corridor. If the fish fry utilisation could be proven to use these artificially produced habitats, it raises the question of whether these habitats are contributing to 'good ecological status' and if so should the old engineering structures themselves be considered under WFD. The reason for the recent spread in intertidal habitats up rivers such as the Thames is not fully understood, but could be due to climate change or improved water quality.

### **3.4.5. Future Work**

To further the finding of this study a greater range of temporal and spatial variations could be surveyed to better illustrate fish utilisation within these sites. Temporal monitoring could include differences between tidal cycles, diel cycles, climatic events, neap/spring tides, as



well as reproductive cycles and life history durations (Simenstad *et al.*, 2000). Nocturnal sampling at different light intensities would also be useful to assess foraging success. Long term temporal monitoring of gut contents from the Creekmouth site would show if productivity declines after an initial disturbance peak. Extending the monitoring to incorporate atypical conditions would also greatly benefit the limited knowledge of fish distributions in the river during this time.

Spatial monitoring and visual observations within Creekmouth could be used to more accurately define fish utilisation within the site. This would require physical (e.g. vegetation type and cover, marsh height and topography) and chemical (e.g. local depth, water temperature and salinity) data to be collected. In addition, monitoring sediment accretion levels within the creek could help assess the areas long term effectiveness for flood storage and fish utilisation.

An extremely beneficial continuation of this project would be to maintain quantitative monitoring of the Creekmouth site through a range of seasons. Long term quantitative fish utilisation data in terms of biomass and density per m<sup>2</sup>, would be a powerful tool in assessing the value of these intertidal areas more accurately. Investigating fish association and utilisation in the newly vegetated hard-engineering structures would also be very beneficial in influencing future management on the Thames.

Fundamentally the success and continual development of other restoration sites or large scale managed realignments requires post-appraisal monitoring of schemes already implemented. Without assessing the success or short-coming of these sites, our knowledge cannot continue to grow and better inform future projects.

### 3.5. Conclusions

The role of estuaries as nursery and refugia for juvenile fish is now well known on a qualitative basis, however little quantitative data is available. The range of commercially important species utilising these areas, highlights the necessity to quantify their importance for ensure future protection. Fisheries assessments such as this help facilitate economic justification for further regeneration projects within heavily urbanised estuaries.

The species utilising the study sites were shown to vary with the changes in fluvial flow, demonstrating that both marine estuarine dependent and freshwater species make opportunistic use of the resource. The gut analysis showed functional overlap between these species and significantly favoured feeding opportunities within the Creekmouth site.

Studies just as this are very important as European saltmarsh research has tended to focus on fish utilisation of extensive areas on saltmarsh such as the Blackwater realignment sites, which do not reflect the condition of much on the remaining UK saltmarsh, which is small and fragmented (Hughes, 2004). This study has shown the fish communities in these smaller saltmarsh areas, despite being located in heavily urbanised estuaries, are very diverse and productive. It highlights the fact that intertidal habitat restoration requires a holistic management approach to help connect fragmented habitat and produce a migration corridor to support fish communities (Simenstad *et al.*, 2000).

Estuaries are highly dynamic systems with continual larvae/ juvenile fish immigration and emigration from fluvial and marine habitat. The protection and creation of new intertidal habitats in these areas, possibly through the WFD or MPA's, have the potential to increase juvenile fish recruitment and production and consequentially improve commercial and recreational fisheries in the area.

Saltmarshes have the ability to achieve a range of functions from increasing biodiversity to improving air quality and flood defence simultaneously. These areas should therefore be considered as a key management tool in coastal protection and enhancement in the future.

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## 5. Appendix