Appendix 13.1 – TR592

Benefits of the compensation package to the Estuaries qualifying feature of the Severn Estuary/Mor Hafren SAC: Typical fish assemblage species.

BENEFITS OF THE COMPENSATION PACKAGE TO THE ESTUARIES QUALIFYING FEATURE OF THE SEVERN ESTUARY / MÔR HAFREN SAC: TYPICAL FISH ASSEMBLAGE SPECIES

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BEEMS Technical Report TR592;

Benefits of the compensation package to the Estuaries qualifying feature of the Severn Estuary / Môr Hafren SAC: Typical fish assemblage species

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1 Background

In 2013, NNB Generation Company (HPC) Limited (NNB) was granted a development consent order (DCO) authorising the construction and operation of a new nuclear power station at Hinkley Point in Somerset. Once operational, Hinkley Point C (HPC) will abstract cooling water from the Severn Estuary (Figure 1). This abstraction will lead to the impingement of fish and other biota. The estuary is utilised by different life-history stages of fish at different times of the year. Most fish lost, owing to impingement, are the juvenile stages of marine species (BEEMS Technical Report TR583*, in prep.*).

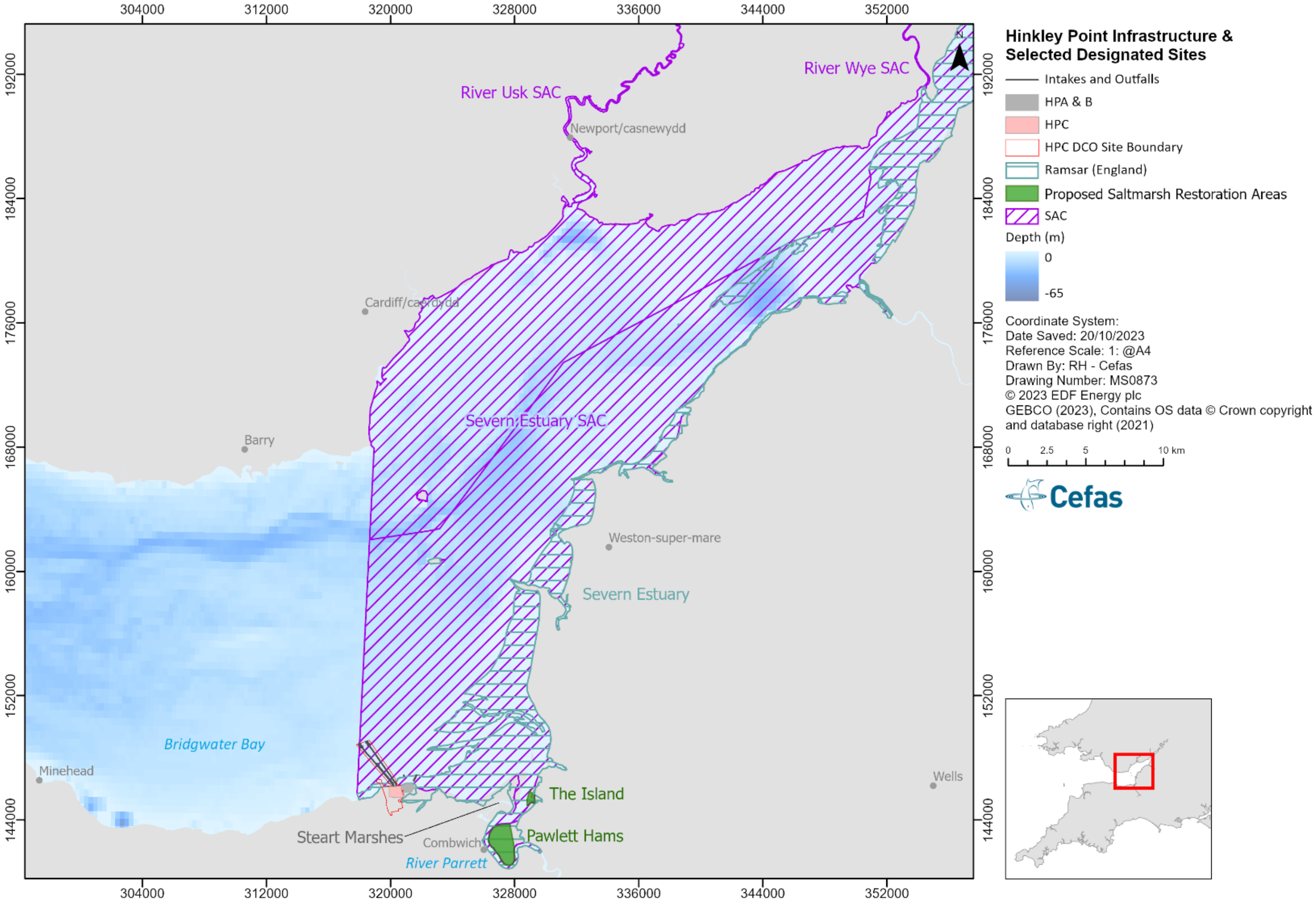

Figure 1 The Severn Estuary/Bristol Channel showing the location of HPC and the National Site Network sites relevant to the entrapment assessments.

To mitigate the impacts of abstraction on fish, three measures were incorporated into the design of the cooling water system for HPC, as consented by the HPC DCO in 2013, these were:

- Acoustic Fish Deterrent (AFD) system;
- Low Velocity Side-Entry (LVSE) cooling water intake heads with a capped head design; and
- Fish Recovery and Return (FRR) mitigation.

The LVSE intake heads and FRR remain part of the station design. However, on 15 February 2019, NNB made an application to the Environment Agency to vary the Water Discharge Activity (WDA) Permit to remove the requirement to install an AFD system. A public inquiry (an appeal for non-determination, hereafter the WDA Permit Inquiry) was held by the Planning Inspectorate in June 2021. The basis for the WDA Permit Inquiry was on the grounds that the Environment Agency were not able to conclude the project in the absence of an AFD would not adversely affect the integrity of the National Site Network and their qualifying features (Table 1). Seven species identified by the Environment Agency in their Appropriate Assessment[1] as being of concern in relation to the removal of the AFD formed the focus of the WDA Permit Inquiry. These were the marine species European sea bass, Atlantic cod, whiting, and Atlantic herring, and the Annex II / Ramsar Criteria 4 migratory species Atlantic salmon, allis shad, and twaite shad. The Defra Secretary of State (SoS)[2] agreed with the conclusions of the Planning Inspector[3] that adverse effects could not be ruled out beyond reasonable scientific doubt for the Severn Estuary/ Môr Hafren Special Area of Conservation (SAC) and Ramsar site, the River Usk SAC, and the River Wye SAC (Figure 1; Table 1).

Table 1 Species and habitats of European Sites of relevance to the WDA Permit Inquiry.

Relevant European site	Interest feature of concern
Severn Estuary SAC	**Annex I qualifying habitat:** **Estuaries** including the typical fish assemblage species of which Atlantic salmon; twaite shad; allis shad; Atlantic cod; European sea bass; Atlantic herring; and whiting were of relevance to the WDA Permit inquiry. **Annex II qualifying species** • Twaite shad
Severn Estuary Ramsar*	**Criterion 4 migratory fish assemblage species,** of which Atlantic salmon, allis shad and twaite shad were of relevance to the WDA Permit inquiry.
River Usk SAC	**Annex II qualifying species** • Atlantic salmon • Twaite shad
River Wye SAC	**Annex II qualifying species** • Atlantic salmon • Twaite shad **Present but not a primary reason for site selection** • Allis shad
* In his Decision Letter[2], the Secretary of State for Defra noted that: *"For the reasons given at IR11.39-46, the Secretary of State agrees with the Inspector that the agreed species of relevance, Atlantic cod, European sea bass, Atlantic herring and whiting, are not species to take into account when considering impacts on the Ramsar site. He notes that the Ramsar criteria (on which Ramsar sites are designated) draw a distinction between Criterion 8 (habitats), under which the Severn Estuary Ramsar site was designated, and Criterion 7 (criteria based*	

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[1] Environment Agency. 2020. Appropriate assessment of the application to vary the water discharge activity permit for Hinkley Point C (November 2020). https://ea.sharefile.com/share/view/s3ca03dcf62e34305a368cd5e85a25df0.

[2] Removal of acoustic fish deterrent conditions from Water Discharge Activity (WDA) Permit, Secretary of State for DEFRA Decision Letter: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1101961/hpc-decision-letter-220902.pdf).

[3] Removal of Acoustic Fish Deterrent Conditions from Water Discharge Activity (WDA) Permit, Inspectors Report: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1101903/environmental-permit-appeal-app-epr-573-hinkley-point-c.pdf.

Relevant European site	Interest feature of concern
on fish populations), which do not form part of the site's designation. The Secretary of State further agrees with the Inspector that this does not alter the position that the migratory species, Atlantic salmon, allis shad and twaite shad, are relevant features for assessment against Criterion 4 (criteria based on a wetland's role in supporting plant and/or animal species at a critical stage in their life cycles)."	

In relation to the Severn Estuary SAC, this conclusion was made due to the conservation objective including *'to ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring: ……..the structure and function (including typical species) of qualifying natural habitats'*[4]. The qualifying natural habitat here is the *Estuaries* qualifying habitat feature of the Severn Estuary SAC. The Severn Estuary European Marine Site Natural England and Countryside Council for Wales advice (given under regulation 33 of the Conservation (Natural Habitats) Regulations 2010 as amended and dated June 2009) refers to a fish species assemblage, a waterfowl species assemblage, and a vascular plant species assemblage as being relevant to the Estuaries qualifying habitat feature. The Severn Estuary/Bristol Channel is a large and complex water body that is home to over 100 species of marine, freshwater, and estuarine fish (Bird, 2008).

Impingement, in the absence of an AFD, is anticipated to increase. Effects from impingement and entrainment (collectively termed entrapment) will be assessed within an application for a variation of the existing 2013 DCO under the Conservation of Habitats and Species Regulations 2017 (Habitats Regulations). On the assumption that the assessment identifies a risk of adverse effects on integrity of the Severn Estuary SAC and Ramsar site (Criteria 4: migratory species) or other European sites linked to the Annex II qualifying migratory fish species (allis shad, twaite shad, and Atlantic salmon) then compensation measures are likely to comprise primarily of schemes to improve up/down stream fish passage through barrier easements or removal. The potential benefits to Annex II and Ramsar Criteria 4 migratory species of fish, are considered further in BEEMS Technical Report TR595.

On the assumption that the assessment identifies a risk of adverse effects on integrity of the Severn Estuary SAC linked to the Estuaries qualifying habitat feature, then habitat compensation measures including the creation and restoration of saltmarshes, kelp forests, seagrass beds, and native oyster reefs have been proposed[5]. These habitats are proposed to provide benefits to the Estuaries qualifying habitat feature, including a wide diversity of species and functional groups of fish associated with the Severn Estuary and Bristol Channel typical fish species assemblage.

As part of the DCO variation consultation, NNB have proposed a package of habitat compensation measures within the Severn Estuary and wider Bristol Channel. A summary of the proposals, as of October 2023 are provided below:

- Saltmarsh and associated habitats: approximately 340 ha, including:

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[4] European Site Conservation Objectives for Severn Estuary SAC (UK0013030): https://publications.naturalengland.org.uk/publication/6081105098702848

[5] In the context of this report the term *'restoration'* is applied to encompass the compensation package of different habitat types. Specific management measures may include a.) creation of new marine/intertidal habitat from terrestrial habitats (e.g., managed realignment to create new saltmarsh habitat at Pawlett Hams), b.) enhancement of existing habitat to improve ecological functioning (e.g., saltmarsh at 'the Island'), or c.) restoration of poor quality or locally absent pre-exiting habitat (e.g., seeding seagrass).

  - **Pawlett Hams habitat creation** – The proposed Pawlett Hams site is approximately 313 ha and is located on the east bank of the River Parrett opposite the village of Combwich. The site is approximately 3.5 km from the estuary mouth in close proximity to the existing Steart marshes wetland site and the Somerset Wetlands National Nature Reserve (Figure 1). The creation of saltmarsh and associated habitats through breaching of the soft landscape flood defenses and the excavation of new creeks will allow tidal waters to flood the low-lying areas of the Pawlett peninsula. The works are anticipated to be similar to the managed realignment scheme developed at Steart marshes and would provide ecological connectivity contributing to the overall coherence of the national network of protected sites.
  - **'The Island' habitat enhancement** - The Island is a 27 ha area located on the east bank of the River Parrett opposite the opening to WWT Steart Marshes wetland site and the Somerset Wetlands National Nature Reserve, approximately 1.2 km from the estuary mouth (Figure 1). Enhancement of existing saltmarsh and associated habitats would be achieved through lowering of the existing high-level marsh to create a range of habitats more amenable to fish usage including tidal creek, mudflat and lower-level saltmarsh. The indicative proposals at The Island include the excavation of a new creek system leading into the marsh with proposed extensions to the existing creek and three shallow pools at the heads of the creeks to create additional shallow mudflat areas.
- Seagrass: 5 ha – Compensation measures aim to deliver 5 ha of seagrass habitat in the Severn Estuary or the wider Bristol Channel.
- Kelp forest: 15 ha – Compensation measures aim to deliver 15 ha kelp forest habitat in the Severn Estuary or the wider Bristol Channel.
- Native oyster reefs: 1 – 2 ha – Compensation measures aim to deliver 1 – 2 ha of native oyster reefs in the Severn Estuary or the wider Bristol Channel.

Site selection of seagrass, kelp forest, and native oyster reef compensatory habitat will be undertaken in line with site feasibility studies and trials to identify optimal locations. In addition, assessments will be undertaken to determine the potential impacts of habitat creation/enhancement at each compensation site with measures proposed to avoid, minimise and where necessary, offset effects on existing habitats.

This report considers the compensation measures being proposed and how they may benefit the typical fish assemblage species. The report focuses on habitat restoration as the primary mechanism to offset losses to the typical fish assemblage species of the Severn Estuary SAC. The range of species that may directly or indirectly benefit from habitat creation are considered (Section 2), and estimates of fish production from the different habitats are provided (Section 3) with particular consideration paid to the four marine species of concern at the WDA Permit Inquiry: European sea bass, Atlantic cod, whiting, and Atlantic herring. Synergistic and indirect effects of the restoration measures as well as the potential benefits from fish passage improvements, as a result of the implementation of barrier removal and easement works in catchments draining into the Severn Estuary, are considered in Section 4.

Potential fish production rates in restored habitat are contextualised relative to total predicted impingement losses from the assemblage, to understand the extent of losses that may be offset as a result of compensation measures. The limitations and assumptions of approaches to predict fish production in different habitat types and production losses due to impingement, lead to uncertainties in the assessments. The limitations of the approaches in the context of the Severn Estuary are described (Section 3.1.1).

1.1 The typical fish species assemblage

The Severn Estuary/Bristol Channel is a large and complex water body from which over 100 fish species have been observed (Bird, 2008). Some fish species may spend their whole lives within the estuary environment, while other species may be more transitory, using the estuary for feeding, spawning or as a nursery area. The estuary is used as a migration route for species spawning in, or utilising, freshwater habitats of the river catchments that drain into the estuary.

Impingement monitoring at Hinkley Point B (HPB) provides a record of trends in the fish community in Bridgwater Bay. A total of 90 fish taxa have been observed in the Routine Impingement Monitoring Programme (RIMP) between 1981 – 2019, with an average of around 38 species typically recorded in each year. RIMP data collected at HPB between 1981 and 2008 identified an increase in the abundance of fish impinged that was attributed to improvements in water quality in the estuary and reductions in anthropogenic impacts such as abstraction (Henderson and Bird, 2010). As well as an increase in abundance, there have been increases in the number of different species observed that may be a result of climate change. Milder winters are likely leading to increased prevalence of fish for which the Celtic Sea is close to the northern extent of their distribution, while species favouring colder waters are still present (Henderson and Bird, 2010). The annual Comprehensive Impingement Monitoring Programme (CIMP) datasets identified 64 fish species in CIMP1 (2009/10) and 62 fish species in CIMP2 (2021/2022). Despite this high species richness, a few species dominated impinged abundance. Five species accounted for the top 90% of impingement numbers in CIMP1 (sprat, whiting, Dover sole, Atlantic cod, thin-lipped grey mullet), whereas eight taxa contributed to the top 90% of abundance in CIMP2 (sprat, Atlantic herring, whiting, gobies of the genus *Pomatoschistus* spp., Dover sole, poor cod, five-beard rockling, thin-lipped grey mullet) (BEEMS Technical Report TR583 *in prep.*).

The Severn Estuary fish assemblages, and the relative abundance of the species observed in the impingement records, are subject to large seasonal and interannual variability, due to recruitment events and environmental factors (Claridge *et al*., 1986). Sprat and whiting are species with the greatest abundance in the CIMP and RIMP datasets. Whiting abundance in the RIMP is typically low between April-June, peaking in winter. Similarly, sprat abundance peaks in the RIMP in January and December, with low abundance throughout the spring and summer (Henderson and Bird, 2010), a situation also reflected in the CIMP. Dab are another species that are impinged most frequently in winter and nearly absent over the summer months. Other species are most abundant in summer, including juvenile cod, Dover sole and flounder. Seasonal and interannual variation in abundance and the relative proportion of the fish species in the assemblage means the structure and functioning of the fish assemblage is dynamic. Summarising the fish assemblage, Bird (2008) noted that *“The picture that emerges for the fish assemblage in the Severn Estuary and Bristol Channel is one of complex interactions between different species of fish, their predators and prey. The system is characterised by remarkably consistent and robust seasonal cycles in the fish composition, but highly variable inter-annual patterns of abundance that are affected and influenced by a range of environmental variables”.*

The application of habitat-based compensation to offset losses from the typical fish assemblage species of the Estuaries qualifying feature of the Severn Estuary SAC is underpinned by the assumption that existing habitat is a limiting factor for fish production, and that recruitment to the system is sufficient to ‘fill’ any new habitat. These are reasonable assumptions for the assemblage over the longer-term, and habitat restoration projects are likely to alleviate a bottleneck leading to higher biomass and overall productivity. Habitat restoration may benefit the fish assemblage by enhancing biodiversity, increasing feeding opportunities, providing a nursery refuge from predation for juvenile stages, or providing spawning substrate for adults. Furthermore, biodiversity gains and enhanced production derived from the created habitats may be exported to adjacent habitats and indirectly benefit fish not directly associated with these habitat types. A mosaic of new high-quality habitat, alongside existing habitats, will provide a functionally linked *“seascape nursery”* environment for fish utilising the estuary (Nagelkerken *et al.,* 2015).

It is reasonable to assume that over the long-term an increase in high quality habitat would result in increases in production, particularly in years when recruitment to the estuary is not limiting. However, habitat availability will not be the only factor limiting system level production. Rates of recruitment of individual species to the estuary will vary from year to year because the annual supply of eggs and larvae is driven by changes in the reproductive output of the spawning populations and because environmental and meteorological conditions influence survival rates and the numbers of young fish transported to, and surviving within, the estuarine habitats. These factors can lead to order of magnitude differences in annual supply of early life stages of fish to estuarine habitats e.g., European sea bass, flounder, Dover sole and up to two orders of magnitude in sand gobies (Lourenço *et. al.,* 2023).

2 Habitat restoration effects on the typical fish assemblage species

Information from relevant studies of habitat restoration effects in similar geographic regions to the Severn Estuary has been collated to allow us to identify the species that may benefit from the proposed compensation package, and the feeding and functional roles they play in the assemblage.

Functional and feeding guild information is presented in Appendix A with further context provided below for specific species.

2.1 Saltmarsh

Saltmarshes are intertidal habitats that occur in sheltered coasts in areas of silt and muddy sediments. A continuum of halophytic (salt tolerant) vascular plant species grow at different salt marsh tidal elevations. Saltmarshes are productive habitats that act as carbon sinks, sequestering carbon dioxide from the air into plant material and into organic rich sediments (Hudson *et al.,* 2021). Saltmarshes typically establish between Mean High-Water Neaps (MHWN) and highest astronomical tide (HAT). Based on the MMO (2019) habitat descriptions, saltmarsh zonation can be characterised as:

- Mudflats occur at the base of saltmarsh between the levels of Mean Low Water Springs (MLWS) and MHWN. Mudflats are unvegetated and dominated by benthic associated microalgae (microphytobenthos). In some areas, seagrasses occur below saltmarshes.
- Lower and middle saltmarsh occurs between MHWN and Mean High Water Springs (MHWS). A range of halophytic grasses, rushes and shrubs grow at these marsh levels.
- Upper saltmarshes occur between MHWS and HAT and are dominated by perennial species.
- Transitional grassland occurs between HAT and approximately one metre above HAT (HAT+1).
- Grassland and other terrestrial habitats occur at elevations over one metre above HAT.

A network of creeks and lagoons intersperse lower and middle marshes contributing to the overall function of the saltmarsh habitat. Saltmarsh systems are driven by processes of sedimentation and hydrodynamics (Hudson *et al.,* 2021). Accretion and erosion leads to the evolution of saltmarshes. Over multi-decadal timescales, sea level rise would alter the marsh zonation at a given site.

At approximately 340 ha, saltmarsh is the primary habitat type being proposed for creation and enhancement as part of the compensation package. Saltmarsh habitat is proposed to be created from terrestrial habitats (e.g., Pawlett Hams) or through enhancement of existing habitat (e.g., ‘The Island’). Habitat creation/enhancement at Pawlett Hams and the Island would result in increased areas of intertidal habitat supporting the Estuaries qualifying feature of the SAC.

Saltmarshes are dynamic environments that flood on the incoming tide and drain as the tide ebbs. Fish access the marsh creeks on the flood tide to feed and potentially seek refuge from predators before moving off the marshes during the ebb tide. Saltmarshes are highly productive habitats providing organic plant material, a proportion of which can be utilised by coastal food webs (Bouchard & Leufeuvre, 2000). Organic material derived by saltmarsh production may be utilised by food webs *in-situ* or exported from the marshes by tidal processes or mobile fauna. Therefore, the elevation of the marshes and the degree of tidal inundation influences the benefits provided to fish and other receptors.

There are no European fish species that rely solely on saltmarshes for feeding. However, intensive feeding occurs within these habitats during periods of inundation. At Mont Saint-Michel, France, juvenile European sea bass enter saltmarsh creeks when they are flooded for 1 – 2 hours, which occurs during 43% of the tides in the cycle between neaps and springs, whereas the upper vegetated flats of the marsh are flooded by just 5 – 10% of the tides (Laffaille *et al.,* 2001). Mullet utilise saltmarsh primary production and detritus directly, exporting it from the saltmarsh system (Laffaille *et al.,* 2002). When fish entered the marshes to feed, over 90% had empty digestive tracts, whereas nearly all fish leaving the marsh contained food items in the digestive tracts (Lefeurve *et al.*, 1999). Lefeurve *et al.*, (1999) and Cattrijsse and Hampel (2006) and references therein, suggest daily consumption rates of fish foraging on tidal marshes are 4.5% but can reach 7 – 16% of body weight in common gobies, 7 – 8% for thin-lipped grey mullet and 10% for European sea bass. Despite the reduced period these habitats are inundated by the tide, high feeding activity may contribute a high proportion of juvenile fish energetic requirements. Temperature and life-history stage are important factors determining energetic demands and smaller fish and fish living at higher temperatures require more food per day than larger fish living at lower temperatures (Coche, 1997). In the Balearic Islands, juvenile grey mullet *Liza saliens* daily food requirements equated to 6% of body weight in August and by November of the same year reduced to 1.4% of body weight (Cardona, 1999), whilst the daily maintenance ration of the golden grey mullet *Mugil auratus* at 15°C is ~3% of body weight (De Silva, 1980). In the case of juvenile European sea bass, fed squid diets at a range of temperatures reflective of Welsh coastal waters, maintenance rations ranged between 0.7 and 2.3% of body weight per day and increased with temperature within the range 6-18°C.The maximum meal size increased from 4% body weight per day at 6°C to ~8% body weight per day at 18°C (Russel et al., 1996). Based on energetics presented in (Ehrenberg *et al.,* 2005), it is estimated that gobies of the genus *Pomatoschistus* inhabiting shallow subtidal soft sediment habitats of the Baltic Sea consumed approximately 5% of their body weight per day. Whilst food quality will vary, it is likely that a high proportion of daily dietary requirements may be achieved during saltmarsh foraging excursions.

Saltmarshes are regularly described as providing juvenile fish with refuges from predation. Cattrijsse *et al.,* (1997) suggest that saltmarsh creeks provide reduced fish predation pressure for juvenile brown shrimp However, reductions in juvenile fish mortality have not been quantified. In European marshes, infaunal productivity is high and this productivity may be exported from the marsh by fish and macrocrustaceans (Cattrijsse & Hampel, 2006). As well as acting as nursery habitat for over 40% of the fish species present in the Mira estuary, saltmarshes are also major exporters of organic material to the relatively oligotrophic adjacent waters (Costa *et al.,* 2001).

Studies from across Europe show varying levels of utilisation of saltmarshes by different species, commonly recorded species being European eel, flounder, and different goby species (Cattrijsse & Hample, 2006). A total of 32 species were identified during saltmarsh surveys at Mont Saint-Michel (Laffaille *et al.,* 2000b). Sand goby, Lozano's goby, European sea bass, thick-lipped grey mullet and sprat accounted for over 95% of numbers. Herring was present in nearly half the monthly samples but in low numbers and biomass. Biomass was dominated by thin-lipped grey mullet that accounted for 87% of the biomass (Laffaille *et al.,* 2000b). Saltmarshes may also support diadromous species with high conservation value. European eel are commonly recorded in European saltmarsh surveys (Appendix A, Table 6), whilst large numbers of juvenile Twaite shad enter saltmarsh creeks in the Wadden sea in late summer (Tulp *et al.,* 2017). Fish species associated with European saltmarshes are listed in (Appendix A, Table 6).

The large tidal ranges and physical challenges associated with surveying the saltmarsh environment mean few studies provide quantitative estimates of fish biomass or numbers in saltmarshes, and sample data is typically expressed in relative terms or as catch per unit effort (CPUE). Green *et al.,* (2012a) estimated the numbers of fish utilising Essex saltmarshes to be approximately 0.05 ind. m^{-2}. Seasonal utilisation of saltmarsh habitats can also lead to large within-year and between year variability in species composition and abundance (Laffaille *et al.,* 2000b; Green *et al.,* 2009; Fonseca, 2009).

Peak annual numbers of fish were reported by Fonseca (2009) at natural and managed realignment saltmarshes in Essex during July-August, when 0.56 ind. m^{-2} were observed (ranges of 0.08 to 2.70 ind. m^{-2}). Annual sampling recorded the presence of 13 species, six of which accounted for the top 94% of numbers and included common goby, European sea bass, golden grey mullet, thin-lipped grey mullet, sprat, and sand smelt. In the summer, the abundance of European sea bass (0-group and 1-group) was estimated to be 0.23 ind. m^{-2} with ranges of 0.04 to 0.95 ind. m^{-2} (Fonseca, 2009). European sea bass accounted for over 72% of the annual biomass. Most individuals were 0-group, however some older fish were also caught. European sea bass were the main contributors to summer biomass, ranging from 0.014 g m^{-2} to 4.54 g m^{-2} (Fonseca, 2009). Green *et al.,* (2009) recorded low numbers of European sea bass in similar saltmarsh sites in Essex between December and May. Abundance was typically greater in the period May to November (>0.1 ind. m^{-2}), peaking at over 0.5 ind. m^{-2}. Fourteen fish species were present, but the community was dominated by three species that made up over 90% of total catches and included the common goby (46%), juvenile herring (24%), and juvenile and larval stages of European sea bass (21%) (Green *et al.,* 2009). The abundance of juvenile herring contrasts with the results of Fonseca (2009) where herring accounted for approximately 1% of numbers and biomass. These annual differences at similar sites demonstrate the variability in saltmarsh assemblages, largely driven by variation in the timing and magnitude of annual recruitment events. For example, Colclough *et al.,* (2005) observed approximately 2,000 0-group herring in one seine net haul at Abbotts Hall managed realignment site in Essex.

Whilst these studies provide indicative estimates of numbers and biomass per unit area, direct comparisons between Essex saltmarshes and the potential habitat created in the Severn Estuary is hampered by differences in the environment, geographical location, and changes in the state of the fish populations between the time of sampling and implementation of compensation measures. Similarity in species composition between northern European tidal marshes of the Forth, Humber and Westerschelde is relatively low, and the proportion of functional guilds varies between marshes. The guilds present are typically estuarine resident species (ER), marine species (MA, MS and MJ) and varying proportions of diadromous species (CA) (Mathieson *et al.,* 2000).

Steart marshes, a tidal realignment site adjacent to Hinkley Point that was flooded in 2014, provides insight into the potential for created saltmarsh habitat in the Severn Estuary to benefit fish. Sampling at Steart marshes in spring/summer 2017 and 2018 revealed utilisation of the marshes by thin-lipped grey mullet, common gobies and sand gobies, and European sea bass (Stamp *et al,* 2023).

At least thirty of the species identified in HPB impingement sampling are also found in northern European saltmarsh habitats, with saltmarshes providing habitat for the juvenile stages of two of the species of concern associated with impingement; European sea bass and Atlantic herring. Furthermore, numerically abundant and/or high biomass species in the impingement records such as sprat, various species of gobies and thin-lipped grey mullet are also common saltmarsh species (Appendix A, Table 6).

The saltmarsh areas proposed as part of the compensation package at Pawlett Hams, adjacent to Steart marshes in the Parrett Estuary (Figure 1), would represent additional created habitat within the Severn Estuary SAC and Ramsar site, transforming terrestrial land, used primarily for agricultural practices, into new intertidal habitats consisting of a matrix of vegetated saltmarsh with channels, creeks, intertidal mudflats and pools. At ‘the Island’ immediately north of Pawlett Hams in the Parrett Estuary, existing habitat would undergo enhancement measures. The extent of these habitats and the time they take to deliver benefits to fish are an important consideration. In the habitat restoration model proposed by French McCay & Rowe (2003), the development of full ecological functioning of saltmarshes was estimated to take 15 years, although it was noted that vegetation develops rapidly, and fish populate created marshes at levels similar to natural habitat in as little as 2 to 5 years. Ecological functioning occurs at varying time scales for different trophic levels and ecological receptors. Some studies have observed fish densities in created marshes achieving similar levels to natural marshes within 5 years whereas other sites may take 15 years or more

(Strange *et al.,* 2002). The marsh morphology and period of tidal inundation of the habitat are important factors in determining benefits for fish in created habitats. Gut analyses of fish utilising realigned saltmarsh sites provide evidence that the created habitat affords feeding opportunities for fish (Green *et al.,* 2009; Fonseca, 2009; Howe *et al.,* 2014). Indeed, studies at the realignment sites at Steart marshes have demonstrated that within three years of flooding, the marshes provided feeding opportunities for European sea bass, thin-lipped grey mullet, and gobies, although not to the same extent as established marshes (Stamp *et al.,*2023).

In the case of European sea bass, juvenile stages are opportunistic feeders and show ontogenetic and seasonal changes in diet composition. In European saltmarshes their diet typically comprises of the amphipods *Corophium volutator* and *Orchestia* spp*.,* polychaetes worms *Nereis diversicolor* and *Nereis integer, brown shrimp Crangon crangon,* common shore crab *Carcinus maenas*, mysids, isopods and copepods (Cattrijsse and Hampel, 2006). Three years after tidal flooding at Steart marshes, the composition of prey species was similar between the natural and realignment sites, but the relative abundance of prey types varied (Stamp *et al.,* 2023). Feeding rates of thin-lipped grey mullet and European sea bass were 16% and 31% lower in realigned marshes than in natural saltmarshes. Realigned sites typically have lower vegetation density and organic matter, and this was suggested as the reason for lower abundance of herbivorous prey species in the stomachs of European sea bass on realigned marshes, where diets consisted predominantly of the amphipods *Orchestia gammarellus* and *Sphaeromatidae* isopods in summer (Stamp *et al.,* 2023).

In the Blackwater Estuary, managed realignment and established marshes provided valuable feeding opportunities for European sea bass. Juvenile fish leave both types of saltmarshes with significantly higher gut fullness. Small 0-group European sea bass fed predominantly on calanoid and harpacticoid copepods, whilst larger 0-group European sea bass fed on amphipods, calanoid copepods and the crab *Carcinus maenas* (Fonseca *et al.,* 2011). The feeding opportunities provided by the sites were related to the time that European sea bass could utilise the habitat. In contrast to the findings at Steart marshes, European sea bass had greater feeding opportunities at established sites and at managed realignment sites that had deeper creeks and more soft sediment, which allowed longer feeding times than at other managed realignment sites that had more vegetation (Fonseca *et al.,* 2011). Created saltmarsh may also be designed to enhance suitability for fish. Designing features that allow standing water to remain within realignments sites following high tide inundation would increase the available habitat area and may lead in increases in food availability, thereby promoting higher growth rates and survival (Colclough *et al*., 2005; Hudson *et al*. 2021).

2.2 Seagrass beds

In the UK and wider North Atlantic, the dominant seagrass species is the eelgrass *Zostera marina* (Short, 2003). Within the Severn Estuary and Bristol Channel, intertidal and subtidal *Z. marina* and the dwarf eelgrass *Z. noltei* are present.

There are no known surveys of the fish associated with seagrass within the Severn Estuary. Seine net surveys at Porth Dinllaen in North Wales over sandy sediments and over a 28-ha area of intertidal *Z. marina* identified 24 fish species in seagrass habitat, with densities ~4.6-fold higher than over sandy substrates (Bertelli & Unsworth, 2014). Pollack, sand goby and sand smelt were the most abundant species associated with seagrass. Cod and herring were also more abundant at the seagrass site than at the sandy site (Bertelli & Unsworth, 2014).

Seagrass ecosystems are well recognised as important nursery habitats, where juvenile and larval fish species avoid predation and take advantage of feeding opportunities. In a review of cod association with seagrass in the North-East Atlantic, average densities of juvenile cod were estimated at 0.02356 ind. m^{-2}. The presence of *Z. marina* likely enhances the growth and survival of Atlantic cod juveniles, increasing their viability and chance of reaching maturation (Lilley & Unsworth, 2014). The juvenile stages of predatory species such as the gadoids, Atlantic cod, whiting and pollock are also associated with seagrass, as are predatory species such as lesser spotted dogfish (Appendix A, Table 7).

In addition to providing refuge for early stages of fish and invertebrates, seagrass provides spawning grounds for species such as garfish *Belone belone*, stickleback *Gasterosteus aculeatus*, and herring *Clupea harengus* if located in the subtidal. After the loss of the entire subtidal seagrass fields in the Wadden Sea, due to disease in the 1930s, the absence of this important spawning grounds led to a decrease in local herring stocks. Similar observations were made in the White Sea during the 1960s and pointed to the negative effects of seagrass losses on herring populations (review Polte and Asmus 2006). Subsequently, Ivanova *et al.,* (2016) suggested that the recent expansion of seagrass beds in the White Sea was likely to be the reason for the rapid growth in the stickleback population, which is now the second most abundant White Sea fish species in terms of biomass, after herring.

Seagrass patches are also areas of low current velocities. Slowing down the water flow increases reproductive success of invertebrate broadcast spawners such as echinoderms, molluscs, cnidarians, and ascidians (Simon and Levitan, 2011).

2.3 Kelp forest

Kelp form multi-dimensional habitats from the intertidal zone to depths of >20 m. The habitats are made up of several different layers based on the holdfasts, the stipes, and the fronds of the kelp, and these in turn support epiphytic organisms (Burrows *et al*., 2014). In UK waters the predominant species making up kelp forest habitats are *Laminaria hyperborea* and *Saccharina latissimi* (Johnson *et al*., 2023).

Kelp act as habitat engineers by modifying the local physical environment, providing habitat for a wide range of marine organisms, with >1,800 species of flora and fauna recorded within UK kelp forests (Smale *et al*., 2013). These range from echinoderms and molluscs to larger crustaceans, such as crabs and lobsters and many different fish species. The complexity of kelp forests throughout the water column provides habitat for a range of species groups including pelagic species, benthopelagic gadoids including pollock and Atlantic cod, reef-associated species such as wrasse, and demersal species including European sea bass (Jackson-Bué *et al*., 2023). In addition to the gadoids, large bodied predatory species such as conger and lesser spotted dogfish, that contribute notable proportions of impinged biomass, are associated with kelp forests (Appendix A, Table 8).

In common with seagrasses, juvenile and larval fish species use the physical structure of the kelp as shelter to avoid predation and take advantage of the feeding opportunities presented. Kelp forests provide diverse and productive habitats for fish, acting as important areas for foraging and as nurseries for juvenile fish (Jackson-Bué *et al*., 2023).

Furness and Unsworth (2019) used baited remote underwater video (BRUV) to sample kelp and seagrass beds in west Wales and Scotland. The most prevalent species in kelp habitats were two-spotted goby, pollack and Ballan wrasse.

The findings of a review of species associated with kelp forests is provided in Appendix A, Table 8.

2.4 Native oyster reef

European native oyster (*Ostrea edulis*) reefs provide valuable ecosystem services, including enhanced fish production. In the USA and Australia, there is a wealth of research surrounding fish enhancements on oyster reefs, however, limited evidence is available from Europe. Species associated with oyster reefs in northern Europe are listed in Appendix A Table 9.

Oyster reefs are primarily found in shallow waters in depths of <10 m at densities of >5 individuals per m^{-2}. *O. edulis* are habitat engineers, forming densely populated reefs, consisting of both live oysters and dead oyster shells, the structure of which supports a wide variety of flora and fauna (Johnson *et al*., 2023). A variety of flora and fauna have been recorded on native oyster reefs, up to 246 species including fish species such as butterfish, five-bearded rockling and tompot blenny (Johnson *et al*., 2023). Atlantic cod and

European sea bass, species of concern following the variation to remove the AFD, are also associated with oyster reef habitats (Appendix A, Table 9). Oyster reefs are biodiverse habitats. Outside UK waters, studies in the United States found that 79 different finfish species were identified on oyster reefs, of which seven species could be classified as oyster reef residents (Coen *et al*., 1999). Oyster reefs support a range of both commercial and non-commercial species of fish and evidence suggests that they may support biodiversity and productivity at higher trophic levels in Australian estuaries (McLeod *et al.,* 2019).

Restoration projects in the UK are actively being undertaken to enhance native oyster reefs. Examples of native oyster restoration include, in the Blackwater estuary, restoration undertaken by the Essex Native Oyster Restoration Initiative (ENORI), in the Solent by Blue Marine Foundation (BLUE) and in the Mumbles on the Welsh Gower Peninsular by the Mumbles Oyster Company.

Research on oyster reef restoration projects in the USA and Australia can be used to infer potential enhancement of productivity in European waters, where the research is less well established (zu Ermgassen *et al*., 2020). Oyster reef restoration in the southeast United States has been shown to annually enhance production of fish and stone crabs by 2.57 kg per 10 m^{-2} on the restored reef habitat (Peterson *et al.,* 2003). The productivity provided by reefs in the USA is unlikely to be mirrored in a European context as European oyster reefs are less structurally complex and occur in a biogeographic region associated with different fish communities.

3 Quantifying benefits in terms of fish production

There is no standard approach to inform scales of compensation for fish losses due to abstraction by power stations and a range of scaling methods have been developed, particularly in the USA (e.g., Strange *et al.,* 2004). Habitat Production Foregone (HPF) or Equivalent Area of Lost Production (EALP) methods have previously been applied to relate losses of fish owing to power station entrapment (Turnpenny *et al.,* 2010). Recently, Brandthouse *et al.,* (2019) proposed the application of a Habitat Productivity Index (HPI) to allow both power station monitoring data and field data to be expressed in terms of productivity. In this section, the production of the habitats being proposed as part of the compensation package is considered prior to estimating the proportional impingement losses that would be offset.

3.1 Broad scale fish production in estuarine habitats

Broadscale fish production or production within specific habitats provides a means to quantify the benefits of habitat creation. Production estimates also enable like for like comparisons between gains and losses of young fishes in a created habitat and impingement sampling when the age and size distributions may differ (Section 3.3). Production may be used to estimate the amount of biomass a unit area of habitat can sustain over a given period and can be expressed in terms of $g\ m^{-2}\ y^{-1}$, where $1\ g\ m^{-2}\ y^{-1} \approx 1\ t\ km^{-2}\ y^{-1} \approx 10\ kg\ ha^{-1}\ y^{-1}$.

Within the Forth Estuary system, fish production based on the most abundant species has been estimated at 4.3 $g\ m^{-2}\ y^{-1}$ wet weight (WW), with nearly 54% derived from the most abundant species: flounder, whiting and Atlantic cod. The Forth Estuary had higher fish production rates than in the open North Sea (2.5 g WW $m^{-2}\ y^{-1}$), and the Baltic (0.3 – 4.2 g WW $m^{-2}\ y^{-1}$) with the exception of Kiel Bay (7.5 g WW $m^{-2}\ y^{-1}$), whilst considerably higher production estimates have been reported from the Skagerrak and Kattegat (approximately 11.7 – 23.4 g WW $m^{-2}\ y^{-1}$) [6] (Elliot and Hemmingway, 2002 and references therein).

In other estuarine systems, for example, in the Pyrmorye region of Russia, mean annual fish production rates of 0.17 – 4.27 g C $m^{-2}\ y^{-1}$ with peaks up to 9.36 g C $m^{-2}\ y^{-1}$ have been reported (Kolpakov, 2016). Following the conversions of Brey *et al.,* (2010)[7] these estimates equate to approximately 1.3 – 31 g WW $m^{-2}\ y^{-1}$ with maximum values of 68 g WW $m^{-2}\ y^{-1}$. Similar production values were collated for a range of temperate European estuaries reported in Cowley & Whitfield (2002) which ranged from 5.2 g WW $m^{-2}\ y^{-1}$ in the southern North Sea to 10 g WW $m^{-2}\ y^{-1}$ in the Wadden Sea. Production rates in Italy and the lagoon systems in the Bay of Cadez were higher, and sub-tropical and tropical locations can achieve productions rates an order of magnitude higher (Cowley & Whitfield, 2002 and references therein). Estimates from the Forth Estuary are at the lower end of the observed range for estuarine environments and production in the Severn may be greater, particularly given the abundance of juvenile sprat.

Some estuarine habitat types support higher fish production than others and a mosaic of ecologically connected habitats may enhance overall estuarine productivity for a range of species. Fish and mobile invertebrate productivity have been estimated to be greater in eastern oyster reef (*Crassostrea virginica*) habitats in the Atlantic (zu Ermgassen *et al.,* 2016) and saltmarsh edges, seagrass and oyster reefs in the Gulf of Mexico (zu Ermgassen *et al.,* 2021) relative to unstructured control habitats. Unit area production of fish and mobile invertebrates in south and mid-Atlantic oyster reefs was lower than in the Gulf of Mexico and was estimated to be 283.4 ± 57 g WW $m^{-2}\ y^{-1}$ (≈ t $km^{-2}\ y^{-1}$) and greater than in unstructured habitats (zu Ermgassen *et al.,* 2016). The high productivity estimates in these studies reflect the modelling approach that estimated production over the lifetime of the species, including growth once the fish leave the nursery

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[6] Reported as 2.5 – 5 g AFDW $m^{-2}\ y^{-1}$ (Elliot and Hemmingway, 2002) and converted to WW by applying 0.214 AFDW:WW for teleost fish following Brey *et al*., (2010).

[7] Applying the factors used in BEEMS Technical Report TR584 for water quality assessments based on Brey *et al.,* (2010) for herring: C: DW = 0.49, DW: WW = 0.279.

habitat. As such, they are not directly comparable with annual productivity estimates described in the previous paragraph. Nonetheless, these studies serve a clear illustration of the importance of high-quality habitat types for enhancing fish productivity. The following sections consider the potential for seagrass and kelp habitats to enhance fish production based on regionally applicable data sources. Estimates of habitat production of saltmarshes for fish regionally are not available, therefore simplified energy flow models are applied for saltmarshes as well as kelp forests and seagrass in Section 3.2.

3.1.1 Assumptions and limitations

There is uncertainty with regards to attempts to quantify the productivity of different habitats for fish and the following assumptions and limitations must be taken into account:

- Direct quantitative estimates of fish abundance are challenging to obtain in estuarine habitats and thus few existing measurements of biomass and production for the assemblage species in saltmarsh, seagrass and native oyster reef habitats exist. The Environment Agency sample transitional waters as part of the Water Framework Directive (WFD) monitoring and note that *"Estuaries are characterized by strong spatial gradients and temporal fluctuations in physico-chemical conditions. This, in turn, results in highly mobile fish populations, and communities that show considerable spatial and temporal variability. Not surprisingly, estuarine fish communities are difficult to survey and characterize accurately and precisely"* (Environment Agency, 2011). A range of data sources have therefore been considered but site-specific data for the Severn Estuary is lacking.
- Interpretation of existing measurements is complicated by fishes moving on and off different habitat types. In the case of saltmarshes, utilisation of the habitat changes with the tidal cycle, with seasons and with age of the fish. Assigning productivity to each habitat type is therefore extremely challenging when fish utilise a 'seascape' of habitat types. For example, In the Mira Estuary adult thin-lipped grey mullet utilised the upper reaches of the estuary but moved with the prevailing tide median distances of 6.2 km each tidal cycle (Almeida, 1996).
- Compensation through habitat creation assumes that habitat limitation is a bottleneck for production. Therefore, increases in habitat will increase production, rather than spread existing production over a larger area. The assumption that increases in habitat area will lead to increases in production is valid over the long-term at the scale of the assemblage but would be challenging to quantify for individual populations.
- There is also the assumption that larval recruitment is not limiting. Interannual variability in larval supply would be expected to drive differences in the assemblage of species utilising each habitat. This can be seen in the variability in herring abundance in Essex saltmarshes following recruitment events (Section 2.1.1) and the approximate ~9-fold differences between annual cod production rates in the Forth estuary (Elliot and Hemingway, 2002). In years of high recruitment, creation of additional habitat may support additional production, whereas in years of low recruitment fish may make less use of the created areas.

While recognising these uncertainties, available evidence has been used to provide estimates of the benefits, in terms of increases in production, that may be anticipated as a result of the proposed compensation package.

3.2 Estimates of habitat production

This section considers the potential abundance and production that may be achieved by specific habitat types. However, the transient nature of fish utilisation of different habitats and the export of production from high-productivity habitats, notably those in dynamic intertidal systems, to adjoining habitats is not quantified.

3.2.1 Estimated fish abundance on seagrass habitats

The productivity of seagrass habitats and the associated fish community is dependent on the species present, the coverage and density of the seagrass, and other site-specific factors. Fish production estimates in seagrass habitats within the Severn Estuary are not available. At Porth Dinllaen in North Wales, fish density in intertidal *Z. marina* was 36 ± 5.7 fish per seine (~60 m^2 sample area = 0.6 fish. m^{-2}), compared to 7.8 ± 1.6 fish over sand (Bertelli & Unsworth, 2014). Pollack, sand goby and sand smelt were the most abundant species with pollack densities approximately 3.3 fish per seine (~ 0.056 ind. m^{-2}). Cod and herring were also more abundant in seagrass than over sand. Herring associated with seagrasses occurred at lengths of approximately 8 cm but were observed up to 16 cm, comparable to the size range impinged at HPB. Pollack occurred at a range of sizes from 4-16 cm, occurring most frequently at 8 cm (Bertelli & Unsworth, 2014).

The fish density in seagrasses from at Porth Dinllaen suggests that one hectare of equivalent habitat may support approximately 6,000 fish (or a habitat enhancement of ~4,700 fish ha^{-1} over adjacent sandy habitats). On this basis, the proposed restoration of 5 ha of seagrass may support 30,000 fish or approximately 23,500 more fish than supported by 5ha of lower quality substrate.

Estimates of fish density on seagrass at Porth Dinllaen are derived from a single short-term study, in June and September 2012, outside the Bristol Channel and using semi-quantitative methods. In addition, the seagrasses at Porth Dinllaen are located within a protected bay in the Pen Llyn a'r Sarnau SAC and at 28 ha represent one of the largest seagrass beds in Wales (Bertelli & Unsworth, 2014). Restored habitat of any type would take time to become established and achieve similar functionality to an equivalent area of established habitat. It is therefore unlikely that seagrass habitat created through compensation measures would deliver the same fish production as a large established seagrass bed.

3.2.1.1 Estimated cod abundance on seagrass habitats

Lilley & Unsworth (2014) reviewed the benefits of the seagrass *Zostera marina* as a nursery habitat for juvenile Atlantic cod, a marine species of concern associated with impingement at HPC. Literature evidence pointed to high abundance and growth rates, enhanced feeding opportunities and reduced predation risk in seagrass habitats. The authors reviewed literature evidence of juvenile cod abundance in seagrass from across the North Atlantic and reported high variability in density estimates. The mean density of juvenile cod (0-group) in Northeast Atlantic seagrass was estimated at 235.6 ± 23.5 fish ha^{-1} (0.02356 fish. m^{-2}) (Lilley & Unsworth, 2014).

The creation of seagrass habitat is likely to support production of juvenile cod in the Severn Estuary. Any estimation of the total value of seagrass habitat must account for the potential of the habitat to increase growth rates or survival rates by providing refuge and wider feeding opportunities, as exist when energy exported from the habitat stimulates production in adjacent habitats. Applying the estimates from the Northeast Atlantic suggests that creation of 5 ha of seagrass could support 1,178 ± 117.5 juvenile cod, equivalent to 0.7 – 7.8% of the annual losses predicted for CIMP1 and CIMP2, respectively. Atlantic cod recruitment in the Celtic Sea was exceptionally high in 2009, the year of CIMP1 and fewer juvenile cod were impinged in CIMP2 (2021/2022) accounting for the range in estimates of cod losses predicted to be offset by the restoration of seagrass. Large interannual fluctuations in cod production have also been observed in the Forth Estuary (Elliot and Hemmingway, 2002).

3.2.2 Estimated fish production in kelp habitats

Eger *et al.,* (2023) described an approach to estimate the production of fish associated with kelp forests from the observed density of fish per m^2. In the approach described, fish were assumed to be 60% of the adult length, and a length to weight formula was applied to estimate biomass. Production was estimated based on a P:B ratio of approximately 0.4. The modest P:B ratio applied by Eger *et al.,* (2023) reflects the assumption that fish utilising kelp forests are closer to their maximum size.

Using supplementary data provided in Eger *et al.,* (2023), studies in southern Wales and Plymouth have been used to estimate fish production (Smale, unpublished data 2020 within Eger *et al.,* (2023)). The highest biomass taxa were pollock, wrasse and European sea bass. Based on the density and calculated biomass, and methods for estimating productivity, average fish production rates of between 800 – 1,600 kg WW ha^{-1} y^{-1} (80 – 160 g m^{-2} y^{-1}) were estimated for the kelp forests in southern Wales and Plymouth. The high productivity rates are consistent with the wider results from Eger *et al.,* (2023) who estimated that the global average fisheries production in kelp habitats was 2,380 kg WW ha^{-1} y^{-1} (238 g m^{-2} y^{-1}) for fish and invertebrates, of which approximately 38% is harvested.

The unit area fish production estimates derived from Plymouth and southern Wales suggest that once established, 15 ha of restored kelp forest may provide approximately 12,000 -24,000 kg WW of fish production per year. Site feasibility studies will identify the optimal location to undertake kelp restoration. However, it is likely that light limitation within the Severn Estuary SAC would result in sites for kelp projects being located seaward of the Severn Estuary SAC.

3.2.3 Aggregated Food Chain Model

The application of ecological modelling tools provides an alternative approach for estimating the production gains that may be delivered from compensatory habitats. To determine the requirements for compensation at the Salem nuclear power station in Delaware Bay, an Aggregated Food Web (AFW) model was used. The model described the flow of energy from primary producers to higher trophic levels (Balletto *et al.,* 2005). The basis for the compensation was that an increase in the area of saltmarsh habitats would increase food availability and thereby increase fish production.

The AFW is based on the concept of energy flows through trophic levels, and the ability to predict production of higher trophic levels based on primary production. In the approach the food web structure is simplified, and species and life stages aggregated to different groups (Table 2). The approach applied for Delaware assumed 50% of the above ground primary production was available to consumers as detritus and trophic transfer efficiency was 20% to subsequent primary consumers, secondary consumers and tertiary consumer trophic levels (Balletto *et al.,* 2005). Overall, this results in 2% of the primary productivity being converted into secondary consumer production, that was primarily fish.

Based on net above ground primary production rates of 300 g WW m^{-2} y^{-1}, secondary consumer production attributed to the compensatory habitat was assumed to be 6 g WW m^{-2} y^{-1}. In Delaware Bay, the fish species, bay anchovy, spot, weakfish, and white perch accounted for the majority of the secondary production, with terrapin and crab also contributing approximately 10% toward production of secondary consumers.

Table 2 Estimated production in the Aggregated Food Web model for Delaware Bay (Balletto *et al.,* 2005).

Group	Production	% of primary production
Above ground primary production (saltmarsh)	3,000 kg WW ha^{-1} y^{-1} (≈ 300 g WW m^{-2} y^{-1})	NA
Detrital pathways	150 g WW m^{-1} y^{-1}	50.0%
First level consumers (secondary production)	30 g WW m^{-2} y^{-1}	10.0%
Second level consumer	6 g WW m^{-2} y^{-1}	2.0%
Third level consumer	1.2 g WW m^{-2} y^{-1}	0.4%

Can trophic transfer efficiency approaches be applied to the Severn Estuary?

The estimates of trophic transfer efficiencies adopted in the AFW model may be used to approximate secondary consumer production where primary production is known. This approach is applied here to estimate production within seagrass beds, saltmarsh systems and kelp forests. Similar calculations for oyster reefs have not been made, however, with appropriate data on secondary production generated from the oyster reefs, it may be possible to estimate transfer to higher trophic levels. When approximating secondary consumer production based on primary production and transfer efficiencies, several assumptions are made.

i. Primary production estimates for established European saltmarshes and seagrass beds, and UK kelp forests, would apply to equivalent habitats in the Severn Estuary.

ii. Primary production estimates and trophic transfer efficiencies for established habitats apply to restored habitat. This assumption is likely to be unrealistic in the first years after habitat creation as lower production rates would be anticipated until the habitats become established.

iii. Trophic transfer efficiencies correspond to those assumed in Delaware Bay, with production of secondary consumers equivalent to 2% of primary productivity. The differences between USA and UK saltmarshes in relation to this assumption are considered in Section 3.2.3.1 and consideration is given to the proportion of the primary production that may be available to marine food webs. Trophic transfer efficiencies are challenging to measure and vary within and among habitats, as discussed below.

iv. Feeding behaviours of assemblage species in the Severn Estuary are expected to lead to trophic levels ranging from approximately 2 to 3 (species that are predominantly detritivores and planktivores) to over 4 (piscivores) (Table 3). In applying the Balletto *et al.*, (2005) conceptualisation of the food web, fish would be treated as secondary consumers and therefore at trophic level 4. The assumption that production of secondary consumers is entirely fish leads to overestimates in fish production at the secondary consumer levels as some of the production would be associated with invertebrates, and particularly large crustaceans. However, this overestimate would be counterbalanced by a proportion of fish production at trophic levels below 4 and some production from fish above trophic level 4.

Given these assumptions, estimates of production based on trophic transfer efficiencies are compared to estimates based on observations, when possible, to provide validation of the results.

Trophic transfer efficiencies are challenging to measure and vary within and among habitats. Appendix B considers the applicability of the trophic transfer efficiencies in relation to the above assumptions for the different habitat types.

3.2.3.1 Saltmarshes productivity estimates

Saltmarsh habitats occur in intertidal areas and are transitionally utilised by fish. Features such as pools and lagoons can be incorporated into the site design result in standing water at low tide thereby enhancing temporal utilisation and increasing *in-situ* production. However, deriving estimates of fish production from saltmarsh habitats with variable levels of inundation are inherently uncertain. Here fish production, based on saltmarsh macrophyte primary production and export, is estimated.

High production on the middle marsh is rapidly available for food webs, however, only a small proportion of upper marsh production is exported with most being mineralised within the saltmarsh itself (Bouchard & Leufeuvre, 2000). In the Aggregated Food Web model for Delaware Bay, 50% of marsh primary production was estimated to be available for detrital pathways (Table 2). However, the marshes in the USA differ from those in Europe as described below. Depending on tidal elevation, macro detritus exports derived from macrophyte production have been estimated to range between <1% up to 10% for different northern

European saltmarshes (Wolff *et al.,* 1979; Dankers *et al.,* 1984; and Bouchard & Leufeuvre, 2000). However, these values do not account for *in-situ* utilisation of macrophyte organic material or microphytobenthos in the sediment, that can play an important role in saltmarsh production.

Net aerial primary production rates have been derived for saltmarsh systems in Mont Saint-Michel in the low, mid, and upper parts of the marsh as 1,080, 1,910, and 1,990 g DW m^{-2} y^{-1}, respectively. This is equivalent to 490, 840, and 890 g C m^{-2} y^{-1} (Bouchard & Leufeuvre, 2000). The saltmarsh systems at Mont Saint-Michel are well established and created/enhanced marshes would take years to attain equivalent production rates. However, the primary production estimates are for the above ground macrophytes only and do not include, below ground production, or microphytobenthos. Microphytobenthos are the most important energy source for the saltmarsh consumers at Mont Saint-Michel (Creach *et al.,* 1997). In the Severn Estuary, microphytobenthos production has been estimated at 52.5 g C m^{-2} y^{-1} in mud and 17.5 g C m^{-2} y^{-1} in sand (Underwood, 2010).

Based on net above ground primary production, it is possible to estimate the flow of carbon through trophic levels by making assumptions relating to the availability of production and trophic transfer efficiencies. This assessment considers the upper and low/middle marshes separately as fish will not use these areas in the same ways. Upper marshes have low tidal inundation rates and fish would infrequently utilise the habitat thus have less access to the organic material produced.

This assessment takes the follows steps and makes the following assumptions:

a) Organic carbon availability is based on rates of above ground macrophyte production. It is assumed that 10% of primary production in the lower and middle marshes can be directly utilised as a carbon source for saltmarsh food webs or exported to adjacent coastal waters. The contribution of microphytobenthos and epiphytic microalgae production is not included in the above ground macrophyte production estimate. However, microphytobenthos production in muddy habitats (akin to marsh creeks and lagoons) in the Severn Estuary may be in the region of 52.5 g C m^{-2} y^{-1} in mud (Underwood, 2010) and is comparable to the estimate of available macrophyte production based on the assumption of 10% macrophyte primary production being available, b), below). In the upper marshes it is assumed that 1% of primary production is exported to support food webs in the adjacent coastal waters. At this stage of the conceptual design, the relative proportion of marsh vegetated areas compared to creeks and lagoons has not been fully determined. The estimate for available organic material at the base of the saltmarsh food web is considered conservative for established saltmarshes as vegetated areas would support epiphytic microalgae and microphytobenthos. It is noted that it takes years for a created saltmarsh to achieve similar functionality to natural marshes (Section 2.1).

b) Macrophyte net above ground productivity in the different marsh zones is estimated from Bouchard & Leufeuvre, (2000) as:

    a. Middle and lower saltmarsh production assumed to be 665 g C m^{-2} y^{-1}, with 10% of production readily available for food webs (66.5 g C m^{-2} y^{-1})[8].

    b. Upper marsh saltmarsh production assumed to be 890 g C m^{-2} y^{-1} with 1% of production available for food webs (8.9 g C m^{-2} y^{-1}).

c) Based on the current conceptual design for the two restoration sites approximately half the proposed areas for created/enhanced saltmarsh are at an elevation that would support upper saltmarsh. The other half would be low/middle saltmarsh and associated creeks and lagoons. On this basis the unit area production available to higher trophic levels is estimated as 37.7 g C m^{-2} y^{-1} ((66.5 g C m^{-2} y^{-1} ×

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[8] Average microphytobenthos production has been estimated at 52.5 g C m^{-2} y^{-1} in mud flats of the Severn Estuary (Underwood, 2010).

0.5) + (8.9 g C m^{-2} y^{-1} × 0.5)). Alongi (2020), estimated global export of particulate organic matter from saltmarshes at 60 ± 11 g C m^{-2} y^{-1} (median value; 31 g C m^{-2} y^{-1}. Dissolved organic carbon export was estimated to be higher at approximately 255 ± 55 g C m^{-2} y^{-1} (median value; 133 g C m^{2} y^{-1} (Alongi, 2020). This suggests that the value used for the calculations is very conservative.

d) Of the available primary production, the trophic transfer efficiency model assumes 10% is converted into primary consumer production and 2% into secondary consumer production (Table 2). As a validation exercise the estimated primary consumer production derived from this assessment can be compared to observed values. In the case of primary consumers, estimated production would equate to approximately 3.8 g C m^{-2} y^{-1}. In the saltmarsh systems of the Westerschelde Estuary, macrofauna biomass dominated by the amphipod *Corophium volutator*, the polychaete *Nereis diversicolor*, the bivalve *Limecola baltica* and *Oligochaeta* was positively correlated with salinity and ranged from average site values of 1.36 – 13.98 g AFDW m^{-2} (Hampel, 2003). Assuming a P:B ratio of 1.2, this may equate to production of approximately 0.7 – 7 g C m^{-2} y^{-1} [9]. The estimate of 3.8 g C m^{-2} y^{-1}, here, includes both *in-situ* production and production in the adjacent coastal waters stemming from exported organic matter. Therefore, the observed biomass and production estimate for *in-situ* macrofauna from the Westerschelde Estuary suggests the approach here is reasonable.

e) Conversion of primary production to secondary consumer production is estimated to be 2%, resulting in 0.754 g C m^{-2} y^{-1}. Secondary consumer production is assumed to be from fish, whereas a proportion would be from macrocrustaceans. This would result in overestimates of secondary fish production; however, fish production would occur at lower (e.g., detrital feeding mullet) and higher trophic levels (e.g., piscivorous European sea bass feeding at the marsh margins) as well, and this is not accounted for when all fish production is assigned to this one trophic level.

f) Carbon production can be converted to wet weight (WW) production by applying the factors used in BEEMS Technical Report TR584 for water quality assessments and based on Brey *et al.,* (2010) for herring: C: DW = 0.49, DW: WW = 0.279. This results in unit area wet weight estimates of fish production for saltmarsh habitat of 5.52 g WW m^{-2} y^{-1}.

g) The value of 5.52 g WW m^{-2} y^{-1} secondary consumer production is similar to the model estimate applied in the Salem example (6 g WW m^{-2} y^{-1}; Table 2). This would equate to 55.2 kg WW ha^{-1} y^{-1} and a total of 18,752 kg y^{-1} of fish production based on the approximate 340 ha area of saltmarsh proposed as part of the compensation package.

In Florida, saltmarshes have been estimated to support substantially higher levels of fish production of 31 g WW m^{-2} y^{-1} (>300 kg WW ha^{-1} y^{-1}), similar to previous estimates of estuarine fish production in the USA (Table 6 of Stevens *et al.,* 2006). Annual export of fish biomass from saltmarshes to adjacent estuarine waters was estimated to be 1-2% of primary production, providing an important trophic link between saltmarshes and the estuary (Stevens *et al.,* 2006)[10]. Strange *et al.,* (2004) reviewed examples of fish production from tidal marshes in the USA, reporting single species production estimates ranging from 12.5 to over 1,000 kg WW ha^{-1} y^{-1} (1.25 – 100 g WW m^{-2} y^{-1}), with total fish assemblage production ranging from 1,105 – 2,425 kg WW ha^{-1} y^{-1} (110.5 – 242.5 g WW m^{-2} y^{-1}). However, these high fish biomass and productivity estimates in examples from the USA may not be comparable to the UK. This is because USA saltmarshes are flooded for longer periods and retain subtidal channels at low water, whereas European marshes typically drain during the ebb tide (Cattrijsse & Hampel, 2006), furthermore the aforementioned

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[9] Based on a P:B coefficient of 1.2 and generic AFDW:DW conversion of 0.750 applied for all taxa and a C:DW conversion of 0.314 is an average including Mollusca, Annelida and Echinodermata from Brey *et al.,* (2010).

[10] Albeit in a different habitat and biogeographical region, the conversion from macrophyte primary production to fish production applied here is an order of magnitude lower (i.e., it is assumed ~10% of production is readily available for food-webs of which 2% is converted to fish biomass).

USA saltmarsh examples are in different biogeographic regions with higher rates of production. Primary production in USA saltmarshes is more readily exported to subtidal areas.

There are few fully quantitative density and biomass estimates of fish within UK saltmarsh habitats. Fish biomass estimates derived from the Blackwater Estuary in Essex (Section 2.1) suggest that fish production of 5.52 g WW m^2 y^{-1}, may not underestimate *in-situ* fish production within these systems. However, comparing direct measures of fish biomass in European saltmarshes does not take into account the transitory nature of habitat utilisation. Saltmarshes may thus provide energy sources to support fish productivity in adjacent habitats i.e., fish abundance within the saltmarsh is not a full reflection of the total fish production that the marshes support.

Cattrijsse and Hampel (2006) and references therein, suggest daily consumption rates of fish foraging on tidal marshes can reach 7 – 16% of body weight in common gobies, 8% for thin-lipped grey mullet and 10% of European sea bass. In European marshes crustaceans and fish may transfer organic matter from saltmarshes to adjacent coastal waters (Lefeuvre *et al*., 1999; Cattrijsse & Hampel, 2006). In one year, it was estimated that mullet alone exported 12 kg DW ha^{-1} of particulate organic matter in their guts from marshes in Mont Saint-Michel to the adjacent coastal waters (Lefeuvre *et al*., 1999). Furthermore, large European sea bass utilise the marsh edges during the ebb tide to feed on fry as they move off the saltmarsh and may enter marshes at night (Colclough *et al.,* 2005). Fish have been shown to follow the edge of the tide, migrating between intertidal and subtidal habitats. The utilisation of different habitats at different stages of the tide hampers metrics based on relative abundance in single habitat types. Saltmarsh may therefore result in indirect increases in fish production. These indirect ecosystem functions are recognised as important features of value for saltmarshes (e.g., UK NEAFO, 2014) but are challenging to quantify. However, ecosystem modelling has demonstrated the benefits of saltmarsh restoration in terms of ecosystem wide production in Delaware Bay (Frisk *et al*., 2011; Section 4.1.1).

Broad scale estuarine fish production rates often exceed the 5.52 g WW m^{-2} y^{-1} unit area production estimate derived above (Section 3.1). The precaution in the assessment used to derive this estimate and the comparisons with broadscale habitat types indicate it is likely to be conservative, particularly when exports of carbon from the marsh are considered. Due to the inherent uncertainty in the derivation of fish production estimates in the absence of locally relevant information a series of conservative measures have been employed and the resultant estimates are considered to be precautionary for established saltmarsh, however it is noted that there would be a delay prior to the marsh becoming fully established (Section 2.1). Further support of the precautionary nature of the approach applied here is the proximity of the estimate with those applied in the habitat compensation approaches for the Salem power station in Delaware Bay (Balletto *et al.,* 2005).

3.2.3.2 Seagrass bed productivity estimates

Annual primary production of *Z. marina* and associated epiphytes in California was estimated to be 328.5 g C m^{-2} y^{-1} (Penhale, 1977). Donovaro *et al.,* 2002 reported on benthic primary production of seagrasses including examples from the Mediterranean of intertidal *Posidonia* beds (169.3 – 300.0 g C m^{-2} y^{-1}), shallow subtidal *Cymodocea* (72.5 – 95.1 g C m^{-2} y^{-1}) and Atlantic intertidal *Zostera* beds (40.6 – 141.2 g C m^{-2} y^{-1}). Welsh *et al.,* (2000) observed instantaneous production rates in the Bassin d'Arcachon, French Atlantic coast for *Z. noltii* in February, May, and October of 0.09 g C m^{-2} h^{-1} to 0.4 g C m^{-2} h^{-1}. Seagrass beds have been shown to act as a carbon source for adjacent coastal habitats supporting food webs in a wider area, with average rates of carbon export of 24.3% reported (Unsworth *et al.,* 2019).

Primary production rates are highly variable, based on an intermediate value of 100 g C m^{-2} y^{-1} primary production for *Zostera sp.*, the flow of carbon through subsequent trophic levels and based on the assumed trophic transfer efficiencies is outlined below:

a) Assuming 2% conversion efficiency from primary production to the trophic levels supporting fish production results in 2 g C m^{-2} y^{-1}.

b) Expressed in terms of WW production[11] this equated to 14.6 g WW m^{-2} y^{-1}.

Fish production data for seagrass habitat in the Severn Estuary is not available, therefore these values are approximations.

Based on the derived unit area production estimates, 5 ha of restored seagrass may provide approximately 730 kg of fish production per year. However, it should be noted that seagrass habitats would be restored within existing estuarine habitat and as such would enhance production rather than provide entirely new production.

3.2.3.3 Kelp forest productivity estimates

Primary production in Norwegian kelp forest systems can exceed 3,000 g C m^{-2} y^{-1} (Norderhaug & Christie, 2011). Estimates of UK kelp primary production of approximately 425 g C m^{-2} y^{-1} have been derived, based on supplementary data provided in Eger *et al.,* (2023).

Using the UK macrophyte primary production estimates, a 2% overall trophic transfer efficiency to secondary consumer production (assumed to be fish) is 8.5 g C m^{-2} y^{-1}. When converted to wet biomass[12] this equates to production of 62 g WW m^{-2} y^{-1}. The trophic transfer efficiency approach indicates that 9,300 kg y^{-1} of fish production may be achieved as a result of the 15 ha of proposed compensation habitat.

Fish production of 62 g WW m^{-2} y^{-1} (620 kg WW ha^{-1} y^{-1}), derived from the trophic transfer approaches, is modest in comparison to production reported in Eger *et al.,* (2023) for the Northeast Atlantic. Direct fish production estimates from surveys in south Wales and Plymouth derived estimates of approximately 80 – 160 g WW m^{-2} y^{-1}(Section 3.2.2), this would result in 12,000 – 24,000 kg WW of fish production per year, once a kelp forest was established.

A summary of the habitat production estimates is provided in Appendix C.

3.3 Proportion of impingement losses offset

Sections 3.1 and 3.2 outline the benefits of the proposed compensation habitats for the fish species and estimate the fish production that such habitats may support. Here, the proportion of impingement losses offset by the proposed compensation package is assessed. This is achieved by converting impingement losses of all species in the assemblage, estimated in biomass terms, into annual production and comparing to estimates of annual production in the restored habitats.

Fish auditory sensitivity is dependent on physiology. Species have a wide range of hearing capabilities utilising both particle motion and sound pressure, therefore the effectiveness of acoustic deterrents are highly species specific (Putland & Mensinger, 2019). In the absence of an AFD, impingement rates are anticipated to increase, with some species being impacted by the change more than others. The removal of the AFD is not expected to have a meaningful effect on the entrainment fraction and no benefit of the AFD was applied in previous assessments of entrainment. This is because many individuals susceptible to entrainment, would be too small to actively avoid the intake even if they could sense the AFD. Furthermore, responses to juvenile fish to an AFD may vary from adult fish and few studies have determined age or size effects (Putland & Mensinger, 2019). Therefore, this section compares annual impingement losses for the whole assemblage to estimates of annual fish production in the restored habitat, to estimate the proportion of losses offset.

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[11] Applying the factors used in BEEMS Technical Report TR584 for water quality assessments based on Brey et al., (2010) for herring: C: DW = 0.49, DW: WW = 0.279.

[12] (herring: C: DW = 0.49, DW: WW = 0.27; Brey *et al.,* 2010)

The relationship between production and biomass of young fishes (the P:B ratio) allows biomass estimates by size and/or age to be converted into production estimates. Production estimates enable like for like comparisons between gains and losses of young fishes in a restored habitat and through impingement respectively, when the age and size distributions may differ. Production may be used in two ways to describe the benefits of restored habitat. First, to express the additional production of young fish, and other consumers, that results from the increased primary and detrital production in a restored habitat. Second, to express young fish numbers recorded by monitoring a restored habitat as young fish production. Production is typically estimated for the population, and there is no standardised approach for estimating production for individuals that make up a small subset of a population and are predicted to be impinged by HPC. At the population level, larger, long-lived species tend to have lower P:B ratios. At the individual and local level the relationship between production and biomass (P:B) is site and species specific and changes through ontogeny. For individuals or age classes, the early life-stages of a given species typically have a higher P:B ratio than later life stages. For example, in the North Sea ecosystem models, P:B ratios range from 0.2 to over 3 for adult and juvenile fish, respectively (Mackinson & Daskalov, 2007). In the Mondego Estuary in Portugal the fish community comprised of resident gobies and juvenile stages of European sea bass, flounder and common sole. The P:B ratio for each species varied from year to year but averaged 2.6 – 3.5 over a three-year period (Dolbeth *et al.,* 2008). In a review of 90 surveys of fish from 15 different estuary types in the Pyrmorye region of Russia between 2002 – 2015, mean P:B ratios of 0.2 – 3.2 were reported (Kolpakov, 2016).

The advantage of using production estimates derived from P:B ratios is that biomass data from the CIMP can be used directly and P:B ratios can be applied across a range of species, whereas other methods that consider production foregone over the life cycle of the species require detailed life-history information that is only available for a proportion of the species. Furthermore, annual production losses are more representative of the losses from the SAC when the juvenile fish utilise the estuary. A notable limitation when seeking to estimate production is that the age-specific data are rarely available in the literature and there is no established scientific consensus on a suitable method to estimate production in early life stages. Potential methods, rely on a series of assumptions that are not tested, and provide approximations of P:B with unknown uncertainty. P:B ratios from the literature for the 18 species accounting for the top 99% of predicted losses by HPC are provided in Table 3.

To account for uncertainties in estimating production, a P:B range of 2 – 4 has been applied to the biomass of the predominantly juvenile fish impinged. The application of the upper P:B ratio of 4 for all the total biomass of species incurring mortality is likely to lead to a highly conservative estimate of annual fish production losses owing to impingement. The weighted P:B, calculated to take account of the relative contribution of each species to the total biomass (using the highest results for the juvenile stages reported in Table 3), is less than 2 in both CIMP1 (1.94) and CIMP2 (1.52). The lower weighted P:B estimate for CIMP2 is reflective of the higher proportion of large-bodied species; conger eel and lesser spotted dogfish, in the impinged biomass. It is likely that P:B ratios towards the lower end of the range (2) are more reflective of the annual production of the assemblage within the Severn Estuary.

Table 3 Literature values for P:B ratios for the 18 species contributing to 99% of mitigated (Fish Recovery and Return (FRR) and capped head) biomass losses at Hinkley Point based on both CIMP1 and CIMP2. Trophic level estimates are obtained from Fishbase (Froese & Pauly, 2023), however dietary shifts through ontogeny may result in overestimates in the trophic level of some species impinged as juveniles. The species associated with each habitat type based on the literature reviewed in Appendix A is provided (✓ = present in the habitats based on literature, ✓✓ = commonly associated with habitat type in literature).

Species (Trophic Level)	**P:B**	**Site**	**Reference**	**Species associated with habitat types≠**			
				Saltmarsh	Seagrass	Kelp	Oyster
Whiting (4.4)	0-20cm: 2.36 Adult: 0.89	North Sea	Mackinson & Daskalov (2007)	✓	✓✓	✓	
Thin-lipped grey mullet (2.3)	Juvenile: 1.74 Adult: 0.54	Obetello Lagoon, Italy	Brando *et al.,* (2004)	✓✓			
Sprat (3.0)	2.28 Miscellaneous filter feeding pelagic fish: 4	North Sea	Mackinson & Daskalov (2007)	✓✓			
Atlantic herring (3.4)	Juvenile (0 – 1): 1.31 Adult: 0.8	North Sea	Mackinson & Daskalov (2007)	✓✓	✓		
Conger (4.3)	Large demersal fish: 0.55	North Sea	Mackinson & Daskalov (2007)			✓	
Atlantic cod (4.1)	0 – 40cm: 1.79 Adult: 1.19	North Sea	Mackinson & Daskalov (2007)	✓	✓✓	✓✓	✓
European flounder (3.3)	1.1*	North Sea	Mackinson & Daskalov (2007)	✓✓	✓		
Dover sole (3.2)	0.8* Juveniles: 2.1 – 2.9	North Sea Mondego Estuary, Portugal	Mackinson & Daskalov (2007) Dolbeth *et al.,* (2008)	✓	✓		

Species (Trophic Level)	P:B	Site	Reference	Species associated with habitat types≠			
				Saltmarsh	Seagrass	Kelp	Oyster
European sea bass (3.5)	Juvenile: 2.5 – 4.3 0.57	Mondego Estuary, Portugal Obetello Lagoon, Italy	Dolbeth *et al.,* (2008) Brando *et al.,* (2004)	✓✓		✓✓	✓
Lesser spotted dogfish (3.8)	Small sharks: 0.51	North Sea	Mackinson & Daskalov (2007)		✓✓	✓✓	
Five-beard rockling (3.5)	Small demersal fish: 1.42*	North Sea	Mackinson & Daskalov (2007)		✓	✓	✓✓
Thornback ray (3.8)	0.78*	North Sea	Mackinson & Daskalov (2007)		✓		✓
Bib (pout) (3.7)	Other gadoids (small): 2.5*	North Sea	Mackinson & Daskalov (2007)	✓	✓✓	✓	
Poor cod (3.7)	Other gadoids (small): 2.5*	North Sea	Mackinson & Daskalov (2007)		✓✓	✓	
Common sea snail (3.6)	Small demersal fish: 1.42*	North Sea	Mackinson & Daskalov (2007)	‡	‡		
Sand gobies (3.2)	Sand goby: 1.3 – 3.3 Common goby: 2.4 – 2.9	Mondego Estuary, Portugal	Dolbeth *et al.,* (2008)	✓	✓✓	✓	✓✓
Three-beard rockling (3.5)	Small demersal fish: 1.42*	North Sea	Mackinson & Daskalov (2007)		‡	‡	‡
European plaice (3.2)	0.85* 2.3 – 3.6	North Sea Mondego Estuary, Portugal	Mackinson & Daskalov (2007) Dolbeth *et al.,* (2008)		✓✓	✓	

* values based on the full population.

≠ The lack of presence of a species in a given habitat type from the literature reviewed does not imply the species is absent from the habitat or does not utilise the habitat type.

‡ Montagu's seasnail has been reported in saltmarsh and seagrass surveys. Three-beard rockling are likely to occur in habitat types where other rockling species have been identified.

With the application of the capped head to reduce vertical velocities at the cooling water intakes and FRR mitigation, the total biomass of all impinged species of fish, in the absence of an AFD, is predicted to be 45.8 t WW based on CIMP1 and 18.1 t WW based on CIMP2 (BEEMS Technical Report TR583 *in prep.*).

In CIMP1, the resulting estimate of annual production lost using a P:B ratio of 2 is approximately 91.7 t WW y^{-1}, the conservative P:B estimate for the whole assemblage leads to an estimated loss of to 183.3 t WW y^{-1}. In CIMP2, the likely estimate of production losses, applying a P:B ratio of 2, is 36.1 t WW y^{-1}, whereas the conservative (P:B = 4) scenario results in production losses of 72.2 t WW y^{-1}.

Whilst the results account for uncertainty in lost production due to impingement, by adopting conservative P:B ratios, the uncertainty in the production of restored habitats is treated in a precautionary way by using lower estimates of unit area productivity for saltmarsh, seagrass and kelp forest. Further, there is additional precaution because no account is made for synergistic and diffuse effects of restored habitat.

Once established, the estimates suggest that compensatory habitat may offset 31.4% of fish production losses estimated in CIMP1 and up to 79.8% in CIMP2. Applying conservative production loss estimates results in the compensatory habitat offsetting 15.7% in CIMP1 and 39.9% of fish losses in CIMP2 (Table 4). These predictions do not account for the benefit provided to the assemblage by barrier removal as proposed as a means to compensate losses of Annex II species/Criterion 4 migratory assemblage species (Section 4.3). Furthermore, export of energy sources from these habitats to adjacent habitats may result in higher production than observed *in-situ*, particularly in the case of saltmarshes where precautionary estimates have been applied to available primary production (Section 3.2.3.1). In producing these estimates, no account has been taken of the:

(i) design features of the restored habitats that specifically aim to enhance fish production in saltmarsh sites, such as pools of standing water at low tide;

(ii) the production benefits of native oyster reefs;

(iii) synergistic effects of the habitats in providing additional benefits to adjacent habitats; or

(iv) the benefits to some fish in the typical fish species assemblage of the removal of/easements to barriers.

Points (ii) and (iii) are considered in Section 4, below.

Table 4. Estimate of the level of fish production associated with different types of restored habitat and the proportion of impingement losses estimated to be offset by the habitats once established. Uncertainty in the impingement impact is incorporated with P:B ratios of 2 (likely) and 4 (conservative). Unit area habitat production is based on lower estimates.

Proposed habitat	**Area proposed (ha)**	**Estimated habitat fish production (kg ha^{-1} yr^{-1})**	**Proportion of losses offset: Conservative P:B**	**Proportion of losses offset: Likely P:B**
CIMP1: Estimated fish production losses 91.7 – 183.3 t y^{-1}				
Saltmarsh*	340	55.2	10.2%	20.5%
Seagrass	5	146	0.4%	0.8%
Kelp	15	620	5.1%	10.1%
Oyster reef	1-2	Unknown	NA	NA
Sub-total	**361 ha**		**15.7%**	**31.4%**
CIMP2: Estimated fish production losses 36.1 – 72.2 t y^{-1}				
Saltmarsh*	340	55.2	26.0%	52.0%
Seagrass	5	146	1.0%	2.0%
Kelp	15	620	12.9%	25.8%
Oyster reef	1-2	Unknown	NA	NA
Sub-total	**361 ha**		**39.9%**	**79.8%**

* Estimates for saltmarsh production assume approximately half the marsh area is upper saltmarsh, whilst half is middle or lower saltmarsh with intersecting creeks and lagoons.

4 Synergistic effects and indirect effects

The sections above outline the direct benefits of habitat restoration for the typical fish assemblage species through increases in the availability of nursery, feeding and spawning habitat that may deliver increases in productivity. This section considers the synergistic effects of creation of a mosaic of different habitat types and the indirect benefits of restoration of these habitats on fish. Finally, the potential benefits of barrier removal on the typical fish assemblage species are considered.

4.1 Synergistic effects

Fish inhabiting the estuary utilise a range of habitat types for feeding and refuge at different stages of the tidal cycle, seasonally and with ontogenetic shifts. Restoration of habitats may support production in adjacent habitats through export of energy and improvements in water quality (Section 4.2). NNB are planning to undertake feasibility trials to determine the optimal sites for restoration. It is recommended that site selection considers the proximity to adjacent subtidal and intertidal habitats that may a.) facilitate successful implementation of restoration measures and b.) increase the functional connectivity of the habitats providing a 'seascape' of habitats for fish to utilise thereby enhancing production. For example, the proposed saltmarsh restoration sites within the Parrett Estuary (Figure 1) within the Severn Estuary and Ramsar site are in close proximity to the existing WWT Steart marshes part of the Somerset Wetlands National Nature Reserve (NNR). The provision of additional restored habitat in this location will enhance ecological connectivity between the wetlands contributing to the overall coherence.

The implementation of habitat restoration may also benefit the establishment/recovery of existing habitats through an increase in seed stock, thereby enhancing the probability of natural recruitment.

This 'seascape' approach would be anticipated to result in synergistic effects for fish production. Creation or enhancement of habitat would likely result in increased productivity of adjacent habitat types and increased ecological connectivity. Whilst synergistic effects have been described in the literature, quantifying effects is not currently feasible. Ecological modelling provides one such tool for predicting the ecosystem effects of restoration measures where monitoring data is available, giving an indication of the wider ecosystem level effects of restoration measures.

4.1.1 Ecosystem modelling

Ecosystem models provide insights into the effects of restoration in comparable systems internationally. For example, Frisk *et al.,* (2011) utilised Ecopath with Ecosim to estimate the ecosystem effects of saltmarsh habitat restoration within the wider 2,000 km^2 Delaware Bay system (an area approximately three times larger than the Severn Estuary SAC). The model, consisting of 47 functional groups including 27 species of fish, predicted that a 3% increase in the amount of saltmarsh habitat (45 km^2 restored habitat) resulted in increased total ecosystem biomass (across all trophic levels) of 47.7 t km^{-2} y^{-1} within the Delaware Bay system. Increases were not evenly distributed across all groups and whilst there was an overall increase in biomass, some species were predicted to have reduced biomass (Frisk *et al.,* 2011).

The model was parameterised based on information collected during long-term monitoring of key species in the Delaware Bay (1966 – 2003), literature values, and stock assessments. The simulations showed that the biomass of some species groups increased following restoration, whereas others reduced (Frisk *et al.,* 2011). The authors noted that the full dynamics of the ecosystem could not be captured within the model and assumptions had to be made in the cases of species that only spent a proportion of their time within Delaware Bay. The semi-enclosed environment of the Severn Estuary and inner Bristol Channel is more amenable for delineation than open coastal sites. However, parameterising biomass fluxes at the boundary of the model would be a notable complicating factor and the baseline data to inform the model at the scale of the Severn Estuary/Bristol Channel is not sufficiently resolved at present.

4.2 Indirect benefits

The proposed compensation habitats are likely to have a positive effect on physico-chemical water quality parameters of the estuary (including oxygen, nutrients and turbidity) by removing excess nutrients and reducing turbidity by trapping sediment. Improvements to these physico-chemical water quality parameters are likely to have a positive effect for fish as marine water quality is correlated with dissolved oxygen levels and influences the ability of fish to grow and reproduce (US EPA, 2021). The mosaic of habitats proposed to be restored within the Severn Estuary and Bristol Channel will be able to harbour species and encourage processes that filter large quantities of water and remove/bury nitrogen through sedimentation and denitrification (Preston *et al*., 2020; Gamble *et al*., 2021). The removal of excess nutrients from the local system through uptake of inorganic nutrients such as phosphates and nitrates will aid nutrient cycling locally (Dugan *et al*., 2011; Hudson *et al*., 2021).

The proposed compensation habitats and the species associated with them would also contribute to reducing the availability of contaminants in the local environment by absorption, trapping contaminated sediments, and by supporting microorganisms that can break down contaminants; along with consumption and biodeposition of contaminants by native oysters. The habitats naturally filter seawater including filtration and removal of pollutants, for example by absorption of heavy metals such as mercury, cadmium and uranium (Preston *et al*., 2020; Hudson *et al*., 2021; Umanzor and Stephens, 2022; Gamble *et al*., 2021).

Habitat restoration can have indirect benefits for other habitats or species of conservation concern (i.e., long-distance facilitation). For example, creation of native oyster reef can result in both direct and indirect protection of a species or the structure or function of other habitats of conservation concern (Overton *et al*., 2023). Restored oyster reef habitats would increase seed stock, thereby enhancing the probability of natural recruitment and supporting existing habitats. Measures to implement oyster reef restoration would need to adhere to a strict biosecurity management plan to ensure the risk of spreading diseases to existing reefs would be minimised.

Additional indirect benefits of the habitats proposed for restoration are that they can reduce the abiotic stressors on other intertidal habitats utilised by fish i.e., by increasing sedimentation rates and reducing wave attenuation (Overton *et al*., 2023). The proposed package of compensation habitat in the Severn Estuary and Bristol Channel could have indirect effects on the existing coastal intertidal habitats in the estuary by stabilising sediment, increasing surface elevation and reducing erosion from storms and waves (Løvås and Tørum, 2001; Fitzsimons *et al*., 2019; Gamble *et al*., 2021); thereby protecting other intertidal habitat beneficial to fish. All of which, will increase the resilience of the overall network of habitats that can be utilised by fish.

4.3 Barrier removal benefits to typical fish assemblage species

In addition to the proposed package of habitat creation / enhancement measures NNB is proposing to implement measures to alleviate barriers to migration. The primary purpose of barrier removal is to compensate for losses of the Annex II/Ramsar Criteria 4 species; Atlantic salmon, allis shad and twaite shad. NNB are in consultation with the SNCBs and stakeholders to identify weirs that currently act as barriers to species movement to improve network coherence for these species. Identification of priority sites and feasibility studies will determine weirs suitable for management works that may include weir removal or upstream/downstream easements such as fish passes or notches. The sites currently being considered include for feasibility studies include:

- Maisemore and Upper Lode weirs on the River Severn
- Trostrey Weir on the River Usk
- Manorafon Weir on the River Towy
- Three potential weirs on the River Lugg, a tributary in the River Wye catchment:

- Eyton
- Mousenatch, and
- Coxall

NNB are proposing to undertake fish passage improvements at three weirs as part of their compensation package. Maisemore Weir on the River Severn and Trostrey Weir on the River Usk are currently NNBs preferred proposals.

This section focuses on two barriers in the River Severn, Maisemore Weir (nr. Gloucester) and Upper Lode Weir (nr. Tewkesbury) (Figure 2). At the time of writing, discussions between NNB, the SNCBs and stakeholders are more advanced regarding these barriers and the availability of the Environment Agency Water Framework Directive (WFD) data in the upper Severn estuary and River Severn allows the potential benefits to the typical fish assemblage species to be described in qualitative terms.

Both Maisemore and Upper Lode weirs are man-made structures in the river designed to alter and control downstream flow. Maisemore Weir was built in 1871 and is situated north of Gloucester. It represents the first man-made barrier to migration encountered by migratory fish entering the River Severn catchment. Further upstream Upper Lode Weir, created in 1858, spans the width of the river and is located just downstream of the River Avon tributary joining the Severn. The distance between the Maisemore and Upper Lode weirs is approximately 16 km, equating to a wetted area (along the main channel) of approximately 104 ha. Both weirs are tidal. While Maisemore is within normal tidal ranges, Upper Lode is influenced by higher spring tides only (Davies, 2022). These weirs represent potential barriers to the movement of fish utilising transitional and coastal waters and those primarily residing in freshwater environments. Whilst Upper Lode has a notch style fish pass, improving transit through the weir, Maisemore Weir has no such fish pass and fish may migrate upstream of the weir only as the tide rises.

Thirty-one species were recorded from WFD monitoring sites in proximity to the weirs, with freshwater species making up the majority (Table 5 for full list of species). Eight species were present in all three sampling locations, these are three-spined stickleback, Atlantic salmon, chub, common bream, European eels and their elvers, perch, ruffe, and zander. These species are a mixture of diadromous and freshwater species. No marine adventurous, marine juvenile, marine seasonal or estuarine resident species were found in all three locations. Seven species were caught downstream of Maisemore Weir only, as expected these are predominantly marine and diadromous species and include black sea bream, river lamprey and their ammocoetes, European sea bass, sea lamprey, silver bream, thin-lipped grey mullet and twaite shad. Flounder, barbel and common goby were recorded downstream of Upper Lode Weir and were not present upstream. Grey mullet sp. was the only non-freshwater taxa observed solely above Upper Lode. The freshwater species observed only above Upper Lode were common carp, koi carp and mirror carp and tench (Table 5).

Whist thirty-one species were recorded from WFD monitoring sites, each sampling method used (Fyke netting, seine netting and electrofishing) has varying degrees of selectivity and effectiveness at capturing certain species of fish. Therefore, the absence of a species in the WFD data does not necessarily infer its absence from the survey area. For example, electrofishing, which should be a reliable method, has only been applied upstream of Upper Lode in predominantly freshwaters.

Weir removal, easement or installation of fish passes has the potential to benefit diadromous species other than the compensation target species of shads and Atlantic salmon, including European eel, sea trout, and sea and river lamprey. WFD fish monitoring data collected in the transitional and freshwater reaches of the Severn suggest that management works at Maisemore Weir may increase access to upstream habitat for the predominantly marine species such as thin-lipped grey mullet, European sea bass, flounder, and gobies all of which are common in the impingement record. Management measures would also increase interconnectivity for freshwater species.

Further upstream at Upper Lode Weir, reduced salinity may preclude movements of all but the euryhaline marine species, such as gobies and flounder from further passage, and measures are most likely to increase interconnectivity for freshwater species. The species that benefit from weir easement measures will be determined by the design of the structure and their different physiological characteristics, swimming abilities, body sizes, seasonal movements, and behaviours. Barrier easements have the potential to increase the availability of habitat for a range of fish from the assemblage and ease bottlenecks to movement that may limit productivity.

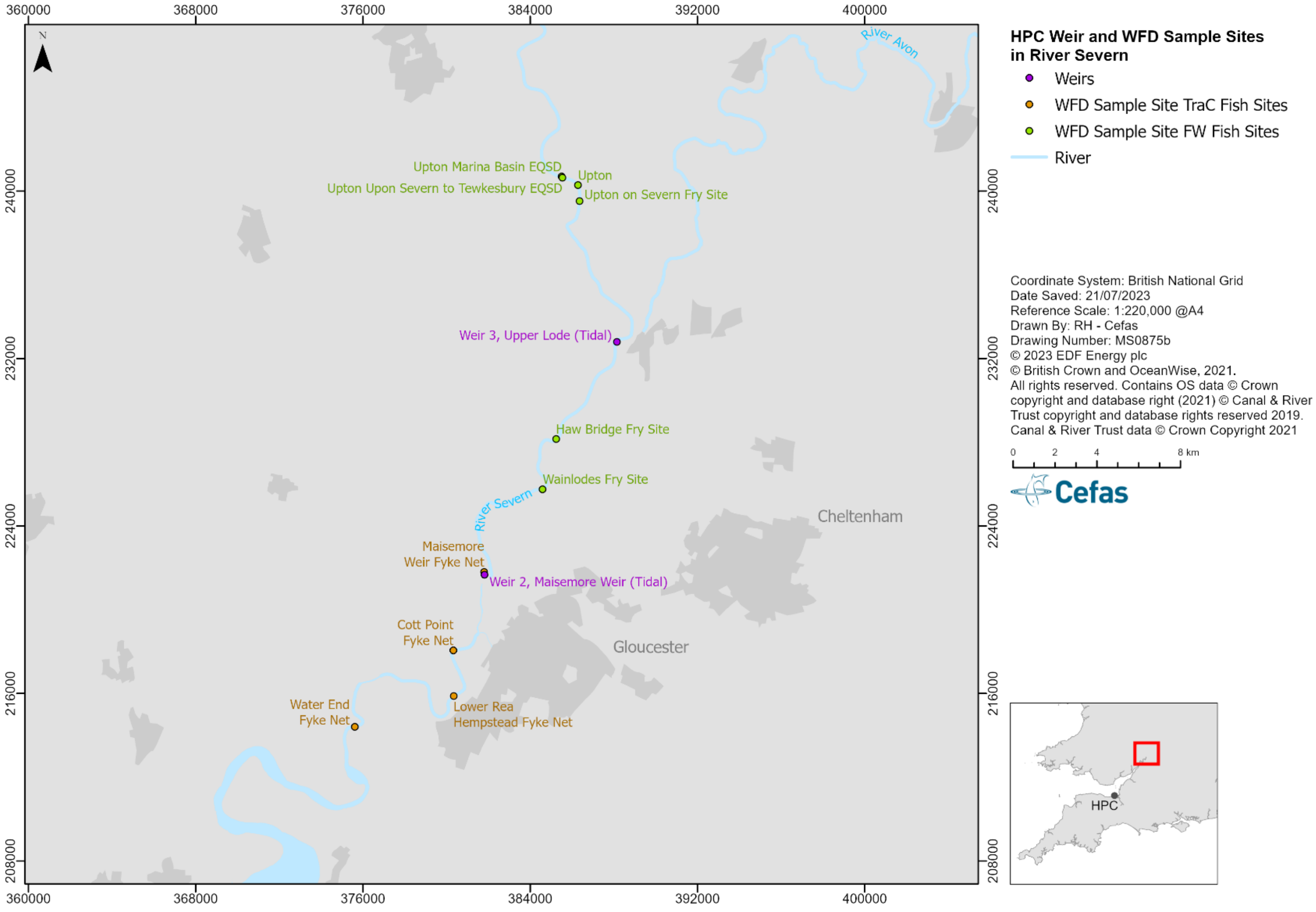

Figure 2 WFD Transitional and Coastal Waters (TraC) and Freshwater (FW) monitoring sites situated throughout the Severn Estuary SAC and River Severn. The locations of Maisemore and Upper Lode weirs are shown in purple.

Table 5. The species recorded in WFD monitoring sites (2000-2019), including the survey gear used and locations relative to Maisemore and Upper Lode weirs. Functional guilds from the UKTAG Guide to the Transitional Fish Classification Index (TFCI) Water Framework Directive: Transitional Waters. MJ = marine juvenile, MS = marine seasonal, ER = estuarine resident, FW = freshwater, CA = diadromous taxa.

Species		**Survey method**				**Location**		
Common name	**Scientific name**	**Functional guild**	**Fyke netting**	**Seine netting**	**Electro-fishing**	**Downstream of Maisemore Weir**	**Between Maisemore and Upper Lode weirs**	**Upstream of Upper Lode Weir**
3-spined stickleback	*Gasterosteus aculeatus*	CA	✓	✓	✓	✓	✓	✓
Atlantic salmon	*Salmo salar*	CA	✓	X	✓	✓	✓	✓
Barbel	*Barbus barbus*	FW	✓	X	X	✓	✓	X
Black sea bream	*Spondyliosoma cantharus*	MJ	✓	X	X	✓	X	X
Bleak	*Alburnus alburnus*	FW	X	✓	✓	X	✓	✓
Chub	*Leuciscus cephalus*	FW	✓	✓	✓	✓	✓	✓
Common [wild] carp	*Cyprinus carpio*	FW	X	X	✓	X	X	✓
Common bream	*Abramis brama*	FW	✓	X	✓	✓	✓	✓
Common goby	*Pomatoschistus microps*	ER	X	✓	X	X	✓	X
Dace	*Leuciscus leuciscus*	FW	X	✓	✓	X	✓	✓
European eel	*Anguilla anguilla*	CA	✓	✓	X	✓	✓	✓
European eels (elvers)	*Anguilla anguilla*	CA	✓	✓	✓	✓	✓	✓
Flounder	*Platichthys flesus*	ER	✓	✓	X	✓	✓	X
Grey mullet sp.	*Mugilidae*	CA	X	X	✓	X	X	✓
Gudgeon	*Gobio gobio*	FW	X	✓	✓	X	✓	✓
Koi carp	*Cyprinus carpio*	FW	X	✓	✓	X	X	✓

Species		**Survey method**				**Location**		
Common name	**Scientific name**	**Functional guild**	**Fyke netting**	**Seine netting**	**Electro-fishing**	**Downstream of Maisemore Weir**	**Between Maisemore and Upper Lode weirs**	**Upstream of Upper Lode Weir**
Minnow	*Phoxinus phoxinus*	FW	X	✓	✓	X	✓	✓
Mirror carp	*Cyprinus carpio*	FW	X	X	✓	X	X	✓
Perch	*Perca fluviatilis*	FW	✓	✓	✓	✓	✓	✓
Pike	*Esox lucius*	FW	✓	✓	X	✓	X	✓
River lamprey	*Lampetra fluviatilis*	CA	✓	X	X	✓	X	X
River lamprey	*Lampetra fluviatilis*	CA	✓	X	X	✓	X	X
Roach	*Rutilus rutilus*	FW	✓	✓	✓	✓	✓	✓
Roach x bream hybrid	*Rutilus rutilus x Abramis brama*	FW	✓	X	✓	✓	X	✓
Rudd	*Scardinius erythrophthalmus*	FW	✓	X	✓	✓	X	✓
Ruffe	*Gymnocephalus cernuus*	FW	✓	✓	✓	✓	✓	✓
European sea bass	*Dicentrarchus labrax*	MJ	✓	X	X	✓	X	X
Sea lamprey	*Petromyzon marinus*	CA	✓	X	X	✓	X	X
Silver bream	*Abramis bjoerkna*	FW	✓	X	X	✓	X	X
Tench	*Tinca tinca*	FW	X	X	✓	X	X	✓
Thin lipped grey mullet	*Chelon ramada*	CA	✓	X	X	✓	X	X
Twaite shad	*Alosa fallax*	CA	✓	X	X	✓	X	X
Zander	*Sander lucioperca*	FW	✓	X	✓	✓	✓	✓

5 Summary

Once operational, cooling water abstraction from HPC will result in entrapment of fish and other biota. In the absence of an AFD fish impingement is expected to increase and will be assessed within an application for a DCO Material Change application. Following the WDA Permit Inquiry, the Defra Secretary of State (SoS) concluded that adverse effects could not be ruled out beyond reasonable scientific doubt for seven species including the marine species European sea bass, Atlantic cod, whiting, herring, and the Annex II diadromous species Atlantic salmon, allis shad, and twaite shad that form part of a typical fish species assemblage of the '1130 Estuaries' qualifying feature of the Severn Estuary SAC. Atlantic salmon, allis shad, and twaite shad also form part of the Ramsar site Criterion 4 migratory fish assemblage.

NNB have proposed a package of compensation measures for the Annex II qualifying fish species/Criterion 4 migratory fish assemblage (in particular twaite shad, allis shad and Atlantic salmon) of the Severn Estuary SAC and other European sites for which they are qualifying features, by means of schemes to improve up/down stream migration. Provision of compensatory habitat, including approximately 340 ha of saltmarsh and associated habitats, 15 ha of kelp forests, 5 ha of seagrass beds, and 1 – 2 ha of native oyster reefs has been proposed to compensate for the '1130 Estuaries' qualifying feature of the Severn Estuary SAC including the typical fish species assemblage.

A literature review of evidence from UK and northern European seagrass beds, kelp forests, native oyster reefs and saltmarshes identified over 80 species that utilise these habitats to varying degrees (Appendix A). Many of these species are present in Bridgwater Bay and appear in the impingement record (BEEMS Technical Report TR583 *in prep.*). In general, saltmarshes primarily benefit detrital feeding mullet species, juvenile sea bass and gobies that feed on invertebrates, and zooplankton feeding species such as sand smelt, sprat and herring. These species represent estuarine resident, marine juvenile, marine seasonal and diadromous functional guilds. Seagrass beds, support a different array of species and are more beneficial to the marine juvenile stages of gadoids such as predatory pollack, cod, and whiting. Bib and poor cod, common in the impingement record, are also associated with seagrass. Seagrass also support benthic invertebrate feeding and zooplankton feeding taxa, along with a host of cryptic species including the Syngnathidae. The structural complexity of kelp forests means they support high biomass and a range of species throughout the water column, including cryptic and demersal species, reef-associated species including wrasse, benthopelagic species and pelagic species (Jackson-Bue *et al.,* 2023). These species represent different feeding and functional guilds. There is less data available on fish associated with native oyster reefs but these habitats support species such as butterfish, five-bearded rockling, tompot blenny and the species of concern (associated with HPC impingement), Atlantic cod and European sea bass. Furthermore, easements of barriers in the lower reaches of catchments, such as Maisemore in the River Severn may benefit species including thin-lipped grey mullet, European sea bass, flounder, and gobies as well as diadromous species allis shad, twaite shad, Atlantic salmon, European eel, sea trout, and sea and river lamprey.

Environmental factors and recruitment strength leads to seasonally and interannual variability in the relative abundance of the species observed in the impingement record in the Severn Estuary (Claridge et al., 1986). This inherent variation in species composition and plasticity in feeding behaviour and prey selectivity means the structure and functioning of the assemblage is dynamic. Restoration of habitats can provide direct benefits to a diversity of fish from different functional and feeding guilds, providing support to the structure and functioning of the typical fish assemblage. However, quantifying system level increases in fish biomass or production resulting from the habitat is challenging. There are examples of where habitat restoration has been shown to enhance ecosystem production, but increases are not evenly distributed across all species groups in the complex estuarine ecosystem. Some species would directly or indirectly benefit from restoration resulting in increases in biomass, whilst other species may incur reduced biomass (e.g., Frisk *et al.,* 2011).

Production was estimated for the range of compensatory habitats that may be created or enhanced, based on a number of comparative and model-based approaches. Given the absence of studies in the Severn Estuary, estimates drew on approaches and studies from a range of locations, and associated caveats and uncertainties are highlighted. Very limited regionally specific information is available to provide quantitative estimates of fish production in the habitats of interest. Furthermore, deriving unit area production estimates for intertidal habitats in dynamic environments is challenging due to the limitations of quantifying fish abundance *in-situ* and the likelihood that the habitats export energy to the adjacent habitats meaning *in-situ* abundance is not reflective of the total fish production that the habitat supports. Therefore, the estimates provided are indicative.

Estimates based on the available evidence suggested that, once established, the compensatory habitats proposed may offset 31.4% of the likely total fish production losses based on CIMP1 impingement estimates and up to 79.8% based on CIMP2 impingement estimates. Applying highly conservative production estimates for impinged fish results in estimates of the compensatory habitat offsetting 15.7% of total fish production losses based on CIMP1 impingement and 39.9% of fish losses using CIMP2 impingement. These estimates account for uncertainty in lost production due to impingement by applying conservative P:B ratios. However, the uncertainty in the production estimates for created habitats cannot be addressed directly. For this reason, lower estimates of unit area production for saltmarsh, seagrass beds and kelp forest have been assumed. In producing these estimates, no account has been taken of the:

(i) design features of the restored habitats that specifically aim to enhance fish production in saltmarsh sites, such as pools of standing water at low tide;

(ii) the production benefits of native oyster reefs;

(iii) synergistic effects of the habitats in providing additional benefits to adjacent habitats; or

(iv) the benefits to some fish in the typical fish species assemblage of the removal of/easements to barriers.

Habitat restoration projects at the scale of the package of measures to compensate impingement losses by HPC, are long-term initiatives. The scale, location relative to the SAC, and time for the compensation habitats to become established will determine the relative benefits and extent to which impingement losses are compensated over time. Whilst habitats such as saltmarsh provide benefits in perpetuity, full ecological functioning would take years to establish. For example, created saltmarsh has been estimated to take 15 years to achieve full ecological functioning, although vegetation develops rapidly, and fish populate created marshes at levels similar to natural habitat in as little as 2 to 5 years. Two years has been suggested as the period for created seagrass beds to perform the same ecological role as natural habitats (French McCay & Rowe, 2003). To determine the success of the compensation measures, NNB is preparing an Adaptive Monitoring and Management Plan (AMMP). The overarching aim of the AMMP would be to:

- Quantify the station impacts relative to impingement predictions;
- Determine appropriate compensation objectives including indicators and associated targets to determine success criteria.
- Provide evidence for the successful implementation of compensation measures;
- Set out a framework for additional monitoring and adaptive management should measures fail to achieve objectives.

None of the fish species impinged are reliant on a single estuarine habitat type throughout their life cycle and the ecological connectivity between adjoining habitats supports overall production. This is particularly true in the highly dynamic environments such as the Severn Estuary. Whilst the estimates of habitats production suggest the proposed package of habitat creation and enhancement would not completely offset all of the estimated losses due to impingement from the typical fish species assemblage of the Severn Estuary SAC qualifying Estuaries feature, the mosaic of proposed habitat types would support a diversity of fish species from different functional groups. High productivity habitat would offset a notable proportion of the fish production estimated to be lost due to impingement and provide synergistic benefits with existing habitats to enhance ecological connectivity of the estuarine ‘seascape’ and contribute to the overall coherence of the national network of protected sites. Created/enhanced habitats may be designed specifically to enhance fish biomass. For example, design features to create allow pools of standing water to remain within saltmarshes following high tide inundation. These measures may increase temporal utilisation of the habitats, thereby increasing feeding opportunities, growth rates and survival (Colclough *et al*., 2005; Hudson *et al*. 2021). Whilst recognising the benefits of habitat creation/enhancement, assessments forming part of the DCO Material Change will need to consider the potential impacts of habitat creation/enhancement at each site and identify measures to avoid, minimise, and where necessary, offset effects on existing habitats. With such steps appropriately undertaken, the provision of the package of compensation measures would have substantial benefits to the Estuaries qualifying habitat feature of the Severn SAC, including for the assemblage of waterfowl species and the assemblage of vascular plant species, as well as providing wider ecosystem services.

References

Almeida, P.R. 1996. Estuarine movement patterns of adult thin-lipped grey mullet, *Liza ramada* (Risso) (Pisces, Mugilidae), observed by ultrasonic tracking. *Journal of Experimental Marine Biology and Ecology*, *202*(2), pp.137-150.

Alongi, D.M. 2020. Carbon Balance in Salt Marsh and Mangrove Ecosystems: A Global Synthesis. *Journal of Marine Science and Engineering,* 8, 767; doi:10.3390/jmse8100767.

Balletto, J.H., Heimbuch, M.V. and Mahoney, H.J. 2005. Delaware Bay salt marsh restoration: mitigation for a power plant cooling water system in New Jersey, USA. *Ecological Engineering, 25*(3), pp.204 – 213.

Barnes, C., Maxwell, D., Reuman, D.C., and Jennings, S. 2010. Global patterns in predator–prey size relationships reveal size dependency of trophic transfer efficiency*. Ecology, 91*(1), pp.222 – 232.

BEEMS Technical Report TR583 (*in prep.*). Hinkley Point C – Entrapment Synthesis 2009 – 2010 (revised) and 2021 – 2022. Cefas, Lowestoft.

BEEMS Technical Report TR595. The potential benefits of barrier easement and removal on populations of shad and Atlantic salmon in the River Severn and selected Welsh rivers. Cefas, Lowestoft.

Bertelli, C.M. and Unsworth, R.K. 2014. Protecting the hand that feeds us: Seagrass (*Zostera marina*) serves as commercial juvenile fish habitat. *Marine Pollution Bulletin*, *83*(2), pp.425 – 429.

Bird, D.J. 2008. The biology and conservation of the fish assemblage of the Severn Estuary (cSAC). CCW Report No: CCW/SEW/08/1, 79pp.

Bouchard, V. and Lefeuvre, J.C. 2000. Primary production and macro-detritus dynamics in a European salt marsh: carbon and nitrogen budgets. *Aquatic Botany*, *67*(1), pp.23 – 42.

Brando, V.E., Ceccarelli, R., Libralato, S., and Ravagnan, G. 2004. Assessment of environmental management effects in a shallow water basin using mass-balance models. *Ecological Modelling. 172*, pp.213 – 232.

Brey, T., Müller-Wiegmann, C., Zittier, Z.M. and Hagen, W. 2010. Body composition in aquatic organisms—a global data bank of relationships between mass, elemental composition and energy content. *Journal of Sea Research*, *64*(3), pp.334 – 340.

Burrows, M.T., Kamenos, N.A., Hughes, D.J., Stahl, H., Howe, J.A. and Tett, P. 2014. Assessment of carbon budgets and potential blue carbon stores in Scotland’s coastal and marine environment.

Cardona, L., 1999. Seasonal changes in the food quality, diel feeding rhythm and growth rate of juvenile leaping grey mullet *Liza saliens*. *Aquatic Living Resources*, 12, pp. 263-270.

Cattrijsse, A. and Hampel, H. 2006. European intertidal marshes: a review of their habitat functioning and value for aquatic organisms. *Marine Ecology Progress Series*, 324, pp.293 – 307.

Cattrijsse, A., Dankwa, H.R. and Mees, J. 1997. Nursery function of an estuarine tidal marsh for the brown shrimp *Crangon crangon*. *Journal of Sea Research*, *38*(1-2), pp.109 – 121.

Christianen, M.J.A., Lengkeek, W., Bergsma, J.H., Coolen, J.W.P., Didderen, K., Dorenbosch, M., Driessen, F.M.F., Kamermans, P., Reuchlin-Hugenholtz, E., Sas, H. and Smaal, A. 2018. Return of the native facilitated by the invasive? Population composition, substrate preferences and epibenthic species richness of a recently discovered shellfish reef with native European flat oysters (*Ostrea edulis*) in the North Sea. *Marine Biology Research, 14*(6), pp.590 – 597.

Claridge, P.N., Potter, I.C. and Hardisty, M.W. 1986. Seasonal changes in movements, abundance, size composition and diversity of the fish fauna of the Severn Estuary. *Journal of the Marine Biological Association of the United Kingdom*, *66*(1), pp.229 – 258.

Coche, A.G. 1997. Simple methods for aquaculture: Management for freshwater fish culture ponds: ponds and water practices. *FAO Training Series*, No 21/1.233 pp.

Coen, L.D., Luckenbach, M.W. and Breitburg, D.L.1999, April. The role of oyster reefs as essential fish habitat: a review of current knowledge and some new perspectives. In *American Fisheries Society Symposium, 22*, pp.438 – 454.

Costa, M., Catarino, F. & Bettencourt, A. 2001. The role of salt marshes in the Mira estuary (Portugal). *Wetlands Ecology and Management*, 9, pp. 121 – 134.

Colclough, S., Coates, S., Dutton, C., Cousins, T. and Astley, T. 2003. The potential for fisheries enhancement associated with managed realignment. *Wetland valuation: State of the art and opportunities for further development*, p.42.

Colclough, S., Fonseca, L., Astley, T., Thomas, K. and Watts, W. 2005. Fish utilisation of managed realignments. *Fisheries Management and Ecology*, *12*(6), pp.351 – 360.

Cowley, P.D. and Whitfield, A.K. 2002. Biomass and production estimates of a fish community in a small South African estuary. *Journal of Fish Biology* 61, pp. 74 – 89.

Creach, V., M. T. Schricke, G. Bertru, and Mariotti, A. 1997. Stable isotopes and gut analyses to determine feeding relationships in saltmarsh macroconsumers. *Estuarine Coastal and Shelf Science,* 44, pp. 599 – 611.

Dankers, N., Binsbergen, M., Zegers, K., Laane, R., Rutgers van der Loeff, M. 1984. Transportation of water, particulate and dissolved organic and inorganic matter between a salt marsh and the Ems-Dollard estuary, The Netherlands. *Estuarine, Coastal and Shelf Science,* 19 (2), pp. 143-165.

Danovaro, R., Gambi, C. and Mirto, S. 2002. Meiofaunal production and energy transfer efficiency in a seagrass *Posidonia oceanica* bed in the western Mediterranean. *Marine Ecology Progress Series*, *234*, pp.95 – 104.

Davies, J. 1998. Bristol Channel and approaches (Cape Cornwall to Cwm yr Eglwys, Newport Bay) (MNCR Sector 9). *Marine Nature Conservation Review. Benthic marine ecosystems of Great Britain and the north-east Atlantic. Peterborough: Joint Nature Conservation Committee*, pp.255 – 295.

Davies, P. 2022. Movement ecology of two threatened anadromous species, twaite shad *Alosa fallax and* sea lamprey Petromyzon marinus, revealed by acoustic telemetry (Doctoral dissertation, Bournemouth University).

De Silva, S.S. 1980. Biology of juvenile grey mullet: a short review. *Aquaculture*, 19, pp. 21-36.

Didderen, K., Lengkeek, W., Bergsma, J.H., van Dongen, U., Driessen, F.M.F. and Kamermans, P. 2020. *WWF & ARK Borkum Reef Ground oyster pilot: Active restoration of native oysters in the North Sea-Monitoring September 2019* (No. 19 – 227). Bureau Waardenburg.

Didderen, K., van der Have, T., Bergsma, J.H., van der Jagt, H., Lengkeek, W., Kamermans, P., van den Brink, A.M., Maathuis, M. and Sas, H, 2019. Shellfish bed restoration pilots Voordelta, Netherlands: Annual report 2018. *Wageningen Marine Research*.

Dolbeth, M., Martinho, F., Viegas, I., Cabral, H. and Pardal, M.A. 2008. Estuarine production of resident and nursery fish species: conditioning by drought events? *Estuarine, Coastal and Shelf Science*, *78*(1), pp.51 – 60.

Du Pontavice, H., Gascuel, D., Reygondeau, G., Maureaud, A. and Cheung, W.W. 2020. Climate change undermines the global functioning of marine food webs. *Global Change Biology*, *26*(3), pp.1306 – 1318.

Dugan, J.E., Hubbard, D.M., Page, H.M. and Schimel, J.P. 2011. Marine macrophyte wrack inputs and dissolved nutrients in beach sands. *Estuaries and Coasts*, *34*, pp.839 – 850.

Eger, A.M., Marzinelli, E.M., Beas-Luna, R., Blain, C.O., Blamey, L.K., Byrnes, J.E., Carnell, P.E., Choi, C.G., Hessing-Lewis, M., Kim, K.Y. and Kumagai, N.H. 2023. The value of ecosystem services in global marine kelp forests. *Nature Communications*, *14*(1), pp.1894

Ehrenberg, S, Hansson, S. and Elmgren, R. 2005. Sublittoral abundance and food consumption of Baltic gobies. *Journal of Fish Biology,* 67, pp. 1083–1093

Elliot, M and Hemingway, K.L. 2002. Fishes in estuaries. *Blackwell Science, Oxford*, pp.636.

Environment Agency. 2011. Quantifying uncertainty in the Transitional Fish Classification Index tool. pp. 62.

Fitzsimons, J., Branigan, S., Brumbaugh, R., McDonald, T. and zu Ermgassen, P. 2019. *Restoration guidelines for shellfish reefs*. Deakin University.

Fonseca, L. 2009. *Fish utilisation of saltmarshes and managed realignment areas in SE England* (Doctoral dissertation, Queen Mary University of London).

Fonseca, L., Colclough, S. and Hughes, R.G. 2011. Variations in the feeding of 0-group bass *Dicentrarchus labrax* (L.) in managed realignment areas and saltmarshes in SE England. *Hydrobiologia*, *672*, pp.15 – 31.

French McCay, D.P., and Rowe, J.J. 2003. Habitat restoration as mitigation for lost production at multiple trophic levels. *Marine Ecology Progress Series,* 264, pp. 233 – 247.

Frisk, M.G., Miller, T.J., Latour, R.J. and Martell, S.J.D. 2011. Assessing biomass gains from marsh restoration in Delaware Bay using Ecopath with Ecosim. *Ecological Modelling, 222*(1), pp.190 – 200.

Froese, R. and Pauly, D. 2023. FishBase. world wide web electronic publication. *http://www.fishbase.org*. (02/2023).

Furness, E. and Unsworth, R.K. 2020. Demersal fish assemblages in NE Atlantic seagrass and Kelp. *Diversity*, *12*(10), p.366.

Galvan, K.A. 2008. The diet of saltmarsh consumers. *Louisiana State University and Agricultural & Mechanical College*. LSU Doctoral Dissertations. https://digitalcommons.lsu.edu/gradschool_dissertations/2233.

Gamble C., Debney, A., Glover, A., Bertelli, C., Green, B., Hendy, I., Lilley, R., Nuuttila, H., Potouroglou, M., Ragazzola, F., Unsworth, R. and Preston, J, (eds) 2021. Seagrass Restoration Handbook. Zoological Society of London, UK., London, UK.

Green, B.C., Smith, D.J., Earley, S.E., Hepburn, L.J. and Underwood, G.J. 2009. Seasonal changes in community composition and trophic structure of fish populations of five salt marshes along the Essex coastline, United Kingdom. *Estuarine, Coastal and Shelf Science*, *85*(2), pp.247 – 256.

Green, B.C., Smith, D.J. and Underwood, G.J. 2012a. Habitat connectivity and spatial complexity differentially affect mangrove and salt marsh fish assemblages. *Marine Ecology Progress Series*, *466*, pp.177 – 192.

Green, B.C., Smith, D.J., Grey, J. and Underwood, G.J. 2012b. High site fidelity and low site connectivity in temperate salt marsh fish populations: a stable isotope approach. *Oecologia*, *168*, pp.245 – 255.

Hampel, H. 2003. Factors influencing the habitat value of tidal marshes for nekton in the Westerschelde estuary. PhD dissertation, University of Gent, pp. 162.

Henderson, P.A., and Bird, D.J. 2010. Fish and macro-crustacean communities and their dynamics in the Severn Estuary. *Marine Pollution Bulletin*, 61, pp. 100-114.

Howe, E.R., Simenstad, C.A., Toft, J.D., Cordell, J.R. and Bollens, S.M. 2014. Macroinvertebrate prey availability and fish diet selectivity in relation to environmental variables in natural and restoring north San Francisco Bay tidal marsh channels. *San Francisco Estuary and Watershed Science*, *12*(1), pp.1 – 46.

Hudson, R., Kenworthy, J. and Best, M. (eds) 2021. Saltmarsh Restoration Handbook: UK and Ireland. Environment Agency, Bristol, UK.

Ivanova, T.S., Ivanov, M.V., Golovin, P.V., Polyakova, N.V., Lajus, D.L. 2016. The White Sea threespine stickleback population: spawning habitats, mortality, and abundance. *Evolutionary Ecology Research, 17*, pp. 301 – 315.

Jackson, E.L., Attrill, M.J., Rowden, A.A. and Jones, M.B. 2006. Seagrass complexity hierarchies: Influence on fish groups around the coast of Jersey (English Channel). *Journal of Experimental Marine Biology and Ecology*, *330*(1), pp.38 – 54.

Jackson-Bué, M., Smale, D.A., King, N.G., Rushton, A.G. and Moore, P.J. 2023. Spatial variability in the structure of fish assemblages associated with *Laminaria hyperborea* forests in the NE Atlantic. *Journal of Experimental Marine Biology and Ecology*, *564*, p.151899.

Jennings, S., Warr, K.J. and Mackinson, S. 2002. Use of size-based production and stable isotope analyses to predict trophic transfer efficiencies and predator-prey body mass ratios in food webs. *Marine Ecology Progress Series*, *240*, pp.11 – 20.

Johnson, C.L.E., Axelsson, M., Brown, L., Carrigan, K.H.O., Cordingley, A., Elliot, A.L., Downie, A., Gannon, L., Green, B., Jones, J., Marsh, M.K., McNie, F., Mills, S.R.A., Wallace, N.M., and Woods H.J. 2023. Marine Restoration Potential (MaRePo). Natural England Research Report JP054

Joint, I.R. and Pomroy, A.J. 1981. Primary production in turbid estuary. *Estuarine, Coastal and Shelf Science*, 13, pp. 303 – 316.

Koenigs C., Miller R.J., Page H.M. 2015. Top predators rely on carbon derived from giant kelp *Macrocystis pyrifera*. *Marine Ecology Progress Series 537*, pp. 1 − 8.

Kolpakov, N.V. 2016. Fish Production in Estuaries of Primorye. *Russian Journal of Marine Biology* 42 (7), pp. 551 – 567.

Laffaille, P., Brosse, S., Feunteun, E., Baisez, A. and Lefeuvre, J.C. 1998. Role of fish communities in particulate organic matter fluxes between salt marshes and coastal marine waters in the Mont Saint-Michel Bay. In *Oceans, Rivers and Lakes: Energy and Substance Transfers at Interfaces: Proceedings of the Third International Joint Conference on Limnology and Oceanography held in Nantes, France, October 1996* (pp. 121 – 133). Springer Netherlands.

Laffaille, P., Lefeuvre, J.C. and Feunteun, E. 2000a. Impact of sheep grazing on juvenile sea bass, *Dicentrarchus labrax* L., in tidal salt marshes. *Biological Conservation*, *96*(3), pp.271 – 277.

Laffaille, P., Feunteun, E. and Lefeuvre, J.C. 2000b. Composition of fish communities in a European macrotidal salt marsh (the Mont Saint-Michel Bay, France). *Estuarine, Coastal and Shelf Science*, *51*(4), pp.429 – 438.

Laffaille, P., Lefeuvre, J.C., Schricke, M.T. and Feunteun, E. 2001. Feeding ecology of o-group sea bass, *Dicentrarchus labrax*, in salt marshes of Mont Saint Michel Bay (France). *Estuaries*, *24*, pp.116 – 125.

Laffaille, P., Feunteun, E., Lefebvre, C., Radureau, A., Sagan, G. and Lefeuvre, J.C. 2002. Can thin-lipped mullet directly exploit the primary and detritic production of European macrotidal salt marshes? *Estuarine, Coastal and Shelf Science*, *54*(4), pp.729 – 736.

Lefeuvre, J.C., Laffaille, P., and Feunteun, E. 1999. Do fish communities function as biotic vectors of organic matter between salt marshes and marine coastal waters? *Aquatic Ecology,* 33, pp.293-299.

Lilley, R.J. and Unsworth, R.K. 2014. Atlantic Cod (*Gadus morhua*) benefits from the availability of seagrass (*Zostera marina*) nursery habitat. *Global Ecology and Conservation*, *2*, pp.367 – 377.

Lourenço, S,, Bueno-Pardo, J., Vaz, A., Primo, A.L., Costa, F., Pardal, M.A., Martinho, F. 2023. Short and long-term temperature variations drive recruitment variability in marine and estuarine juvenile fishes. Marine Pollution Bulletin 192. https://doi.org/10.1016/j.marpolbul.2023.115093

Løvås, S.M. and Tørum, A. 2001. Effect of the kelp *Laminaria hyperborea* upon sand dune erosion and water particle velocities. *Coastal Engineering*, *44*(1), pp.37 – 63.

Mackinson, S. and Daskalov, G. 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. *Cefas Science Series Technical Report*, 142, pp.196.

Mathieson, S., Cattrijsse, A., Costa, M.J., Drake, P., Elliott, M., Gardner, J. and Marchand, J. 2000. Fish assemblages of European tidal marshes: a comparison based on species, families and functional guilds. *Marine Ecology Progress Series*, *204*, pp.225 – 242.

McCormick, H., Salguero-Gómez, R., Mills, M. and Davis, K. 2021. Using a residency index to estimate the economic value of coastal habitat provisioning services for commercially important fish species. *Conservation Science and Practice*, *3*(5), pp.e363.

McLeod, I.M., Boström-Einarsson, L., Creighton, C., D'Anastasi, B., Diggles, B., Dwyer, P.G., Firby, L., Le Port, A., Luongo, A., Martínez-Baena, F. and McOrrie, S. 2019. Habitat value of Sydney rock oyster (*Saccostrea glomerata*) reefs on soft sediments. *Marine and Freshwater Research*, *71*(7), pp.771 – 781.

MMO. 2019. Identifying sites suitable for marine habitat restoration or creation. A report produced for the Marine Management Organisation by ABPmer and AER, MMO Project No: 1135, February 2019, 93pp.

Nagelkerken, I., Sheaves, M., Baker, R. and Connolly, R.M. 2015. The seascape nursery: a novel spatial approach to identify and manage nurseries for coastal marine fauna. *Fish and Fisheries*, *16*(2), pp.362 – 371.

Natural Capital Committee. 2013. The State of Natural Capital: Towards a framework for measurement and valuation.

Norderhaug, K.M., and Christie, H. 2011. Secondary production in a *Laminaria hyperborea* kelp forest and variation according to wave exposure. *Estuarine, Coastal and Shelf Science. 95*: pp.135 – 144.

Norderhaug, K.M., Christie, H., Fosså, J.H. and Fredriksen, S. 2005. Fish-macrofauna interactions in a kelp (*Laminaria hyperborea*) forest. *Journal of the Marine Biological Association of the United Kingdom*, *85*(5), pp.1279 – 1286.

Overton, K., Dempster, T., Swearer, S.E., Morris, R.L. and Barrett, L.T. 2023. Achieving conservation and restoration outcomes through ecologically beneficial aquaculture. *Conservation Biology*, p.e14065.

Penhale, P.A. 1977. Macrophyte-epiphyte biomass and productivity in an eelgrass (*Zostera marina* L.) community. *Journal of Experimental Marine Biology and Ecology*, *26*(2), pp.211 – 224.

Peters, J.R., McCloskey, R.M., Hinder, S.L. and Unsworth, R.K.F. 2015. Motile fauna of sub-tidal *Zostera marina* meadows in England and Wales. *Marine Biodiversity*, *45*, pp.647 – 654.

Peterson, C.H., Grabowski, J.H. and Powers, S.P. 2003. Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. *Marine Ecology Progress Series*, *264*, pp.249 – 264.

Polte, P. and Asmus, H. 2006. Intertidal seagrass beds (*Zostera noltii*) as spawning grounds for transient fishes in the Wadden Sea. Marine Ecology Progress Series, 312, pp.235 – 243.

Preston, J., Gamble, C., Debney, A., Helmer, L.D., Hancock, B. and zu Ermgassen, P. 2020, November. European native oyster habitat restoration handbook. Zoological Society of London.

Radford, P.J. 1994. Pre- and post-barrage scenarios of the relative productivity of benthic and pelagic subsystems of the Bristol Channel and Severn Estuary. *Biological Journal of the Linnean Society*, 51, pp. 5 – 16.

Rosenberg, A.A., Fogarty, M.J., Cooper, A.B., Dickey-Collas, M., Fulton, E.A., Gutiérrez, N.L., Hyde, K.J., Kleisner, K.M., Kristiansen, T., Longo, C. and Minte-Vera, C.V. 2014. Developing new approaches to global stock status assessment and fishery production potential of the seas. *FAO Fisheries and Aquaculture Circular*, (1086), pp.1 – 175.

Russel, N.R., Fish, J.D. and Wootton, R.J. 1996. Feeding and growth of juvenile sea bass: the effect of ration and temperature on growth rate and efficiency. *Journal of Fish Biology*, 49, pp. 206-220.

Sas, H., Kamermans, P., Van der Have, T.M., Christianen, M.J.A., Coolen, J.W.P., Lengkeek, W., Didderen, K., Driessen, F., Bergsma, J., van Dalen, P. and van Gool, A.C.M. 2018. Shellfish bed restoration pilots: Voordelta. The Netherlands: Annual report 2017. *Wageningen Marine Research*.

Short, F.T. 2003. *World atlas of seagrasses*. Univ of California Press.

Simon, T.N., Levitan, D.R. 2011. Measuring fertilization success of broadcast-spawning marine invertebrates within seagrass meadows. *Biological Bulletin, 220*, pp.32 – 38.

Smale, D.A., Burrows, M.T., Moore, P., O'Connor, N. and Hawkins, S.J. 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and evolution*, *3*(11), pp.4016-4038.

Stamp, T., West, E., Colclough, S., Plenty, S., Ciotti, B., Robbins, T. and Sheehan, E. 2023. Suitability of compensatory saltmarsh habitat for feeding and diet of multiple estuarine fish species. *Fisheries Management and Ecology*, 30(1), pp.44 – 55.

Stevens, P.W., Montague, C.L., and Sulak, K.J. 2006. Fate of fish production in a seasonally flooded saltmarsh. Marine Ecology Progress Series, 327, pp. 267-277.

Strange, E., Galbriath, H., Bickel, S. Mills, D., Beltman, D., and Lipton, J. 2002. Determining Ecological Equivalence in Service-to-Service Scaling of Salt Marsh Restoration. *Environmental Management,* 29(2), pp. 290–300.

Strange, E., Allen, D., Mills, D. and Raimondi, P. 2004. Research on estimating the environmental benefits of restoration to mitigate or avoid environmental impacts caused by California power plant cooling water intake structures. *California Energy Commission, PIER Energy-Related Environmental Research, Sacramento, California Available at. http://www. swrcb. ca. gov/rwqcb3/water_issues/programs/diablo_canyon/docs/09_09_05_staff_report/item15_attachment5. pdf*.

Tulp, I., Bolle, L. J., Dänhardt, A., de Vries, P., Haslob, H., Jepsen, N., ... van der Veer, H. W. 2017. Fish. Wadden Sea Quality Status Report 2017. Wilhelmshaven: Common Wadden Sea Secretariat.

Turnpenny, A.W.H., Coughlan, J., Ng, B., Crews, P., Bamber, R.N. and Rowles, P. 2010. Cooling water options for the new generation of nuclear power stations in the UK Technical Report SC070015/SR3. *Environment Agency, Bristol, UK*.

UK NEAFO. 2014. Work Package 1: Natural Capital—Fisheries asset check. Annex 4: Saltmarsh-Fisheries Case study. In: Dickie, I., Cryle, P. & Maskell, L., 2014. UK National Ecosystem Assessment Follow-on. Work Package Report 1: Developing the evidence base for a Natural Capital Asset Check: What characteristics should we understand in order to improve environmental appraisal and natural income accounts? *UNEP-WCMC, LWEC, UK*. Available at: http://uknea.unep-wcmc.org/LinkClick.aspx?fileticket=dNp%2BvIjmXi4%3D&tabid=82

Umanzor, S. and Stephens, T. 2022. Nitrogen and Carbon Removal Capacity by Farmed Kelp *Alaria marginata* and *Saccharina latissima* Varies by Species. *Aquaculture Journal*, *3*(1), pp.1 – 6.

Underwood, G.J.C. 2010. Microphytobenthos and phytoplankton in the Severn estuary, UK: Present situation and possible consequences of a tidal energy barrage. *Marine Pollution Bulletin,* 61, pp. 83-91.

Underwood, G.J.C., Kromkamp, J. 1999. Primary production by phytoplankton and microphytobenthos in estuaries. *Advances in Ecological Research*, 29, pp. 93 – 153.

Unsworth, R.K., Nordlund, L.M. and Cullen-Unsworth, L.C. 2019. Seagrass meadows support global fisheries production. *Conservation Letters*, *12* (1), pp.e12566.

Unsworth, R.K.F., Peters, J.R., McCloskey, R.M. and Hinder, S.L. 2014. Optimising stereo baited underwater video for sampling fish and invertebrates in temperate coastal habitats. *Estuarine, Coastal and Shelf Science*, *150*, pp.281 – 287.

US EPA. 2021. United States Environment Protection Agency. Salish Sea Marine Water Quality. Available (online) at: https://www.epa.gov/salish-sea/marine-water-quality (Accessed 25 September 2023).

Warwick, R.W. 1984. The Benthic Ecology of the Bristol Channel. *Marine Pollution Bulletin*, 15 (2), pp. 70-76.

Water Framework Directive. 2014. UKTAG Guide to the Transitional Fish Classification Index (TFCI) Water Framework Directive: Transitional Waters.

Welsh, D.T., Bartoli, M., Nizzoli, D., Castaldelli, G., Riou, S.A. and Viaroli, P. 2000. Denitrification, nitrogen fixation, community primary productivity and inorganic-N and oxygen fluxes in an intertidal *Zostera noltii* meadow. *Marine Ecology Progress Series*, *208*, pp.65 – 77.

Wolff, W.J., Van Eeden, M.J. and Lammens, E.H.R.R. 1979. Primary production and import of particulate organic matter on a salt marsh in the Netherlands. *Netherlands Journal of Sea Research*, *13*(2), pp.242 – 255.

Zu Ermgassen, P.S., Bonačić, K., Boudry, P., Bromley, C.A., Cameron, T.C., Colsoul, B., Coolen, J.W., Frankić, A., Hancock, B., van Der Have, T.M. and Holbrook, Z. 2020. Forty questions of importance to the policy and practice of native oyster reef restoration in Europe. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *30*(11), pp.2038 – 2049.

Zu Ermgassen, P.S., DeAngelis, B., Gair, J.R., Ermgassen, S.Z., Baker, R., Daniels, A., MacDonald, T.C., Meckley, K., Powers, S., Ribera, M. and Rozas, L.P. 2021. Estimating and applying fish and invertebrate density and production enhancement from seagrass, salt marsh edge, and oyster reef nursery habitats in the Gulf of Mexico. *Estuaries and Coasts*, *44*, pp.1588 – 1603.

Zu Ermgassen, P.S., Grabowski, J.H., Gair, J.R. and Powers, S.P. 2016. Quantifying fish and mobile invertebrate production from a threatened nursery habitat. *Journal of Applied Ecology*, *53*(2), pp.596 – 606.

Zuercher, R. 2022. Pelagic-benthic coupling in kelp forests of central California. *Marine Ecology Progress Series*, 682, pp.79 – 96.

Appendix A Fish species associated with each habitat

This Appendix provides a summary of a review of literature on species association with different habitats from the UK and northern Europe. It is not an exhaustive review but collates information from relevant studies in similar geographic regions to the Severn Estuary. The lack of presence of a species in a given habitat type in the tables below does not imply the species is absent from the habitat or does not utilise the habitat type.

Key to codes:

Functional guilds: MA = marine adventurous, MJ = marine juvenile, MS = marine seasonal, ER = estuarine resident, FW = freshwater, CA = diadromous

Feeding guilds: D = detritus, O = omnivore, BI = benthic invertebrate, Z = zooplankton, P = piscivorous.

Feeding guild information provides a way to classify species based on the primary dietary components of adult fish. This may mask the complexity of feeding interactions during ontogenetic diet shifts. Plasticity of feeding interactions and ontogenetic diet shifts is not accounted for. Invertebrates provide a large component of the diet of the juvenile stages of the piscivorous fishes impinged. Whilst many species classified as benthic invertebrate feeders also prey on small fish.

A.1 Saltmarsh

Table 6 Finfish species associated with saltmarshes from available northern European literature. Common saltmarsh species contributing more than 90% of abundance or biomass are shown in bold (from Laffaille *et al.*, 2000a and b; Green *et al.*, 2009 or Fonseca 2009).

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Common goby	***Pomatoschistus microps***	**ER**	**BI**	**1, 3, 4, 5, 6, 7. 8, 9, 11, 12, 15**	✓
Sand goby	***Pomatoschistus minutus***	**ER**	**BI**	**1, 4, 6, 11, 12, 15**	✓
Lozano's goby	***Pomatoschistus lozanoi***	**MA**	**BI**	**11, 12, 15**	× (Recorded RIMP)
European sea bass	***Dicentrarchus labrax***	**MJ**	**P**	**1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 14, 15**	✓
Herring	***Clupea harengus***	**MJ**	**Z**	**1, 4, 8, 11, 12, 15**	✓
Thin-lipped mullet	***Chelon ramada***	**CA**	**D**	**4, 6, 7, 11, 12, 13, 15**	✓
Thick-lipped mullet	***Chelon labrosus***	**MS**	**D**	**1, 4, 8, 11, 15**	×
Golden grey mullet	***Liza aurata***	**MS**	**D**	**7, 11, 12**	×
Sprat	***Sprattus sprattus***	**MS**	**Z**	**1, 7, 11, 12, 15**	✓
Sand smelt (adult & larvae)	***Atherina presbyter***	**MJ**	**Z**	**1, 4, 7, 8, 9, 11, 12, 15**	✓
European eel	*Anguilla anguilla*	CA	P	4, 11, 12, 15	✓
3-spined stickleback	*Gasterosteus aculetua*	ER	P	1, 4, 11, 12, 15	✓
Transparent goby	*Aphia minuta*	ER	Z	1, 11, 12	✓
Crystal goby	*Crystallogobius linearis*	MA	BI	11, 12	✓
Pogge (armed bullhead)	*Agonus cataphractus*	ER	BI	11	✓
Flounder	*Platichthys flesus*	ER	BI	4, 5, 11, 12, 15	✓

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Lesser sandeel	*Ammodytes tobianus*	ER	Z	1, 11, 12	✓
Garfish	*Belone belone*	MS	P	1, 11	✓
Snake pipefish	*Entelureus aequoreus*	MA	Z	1	✓
Broad-nosed pipefish	*Syngnathus typhle*	ER	Z	1	×
Short-snouted seahorse	*Hippocampus hippocampus*	MA	Z	11, 12	×
Black-striped pipefish	*Syngnathus abaster*	ER	Z	11, 12	×
Nilsson's pipefish	*Syngnathus rostellatus*	ER	Z	11, 12	✓
Greater pipefish	*Syngnathus acus*	ER	Z	15	✓
Viviparous blenny	*Zoarces viviparus*	ER	B	15	×
Horse-mackerel (scad)	*Trachurus trachurus*	MA	P	11	✓
Shore clingfish	*Lepadogaster lepadogaster*	MS	P	11	×
Gobies	*Gobiidae*	NA	NA	15	✓
Brill	*Scophthalmus rhombus*	MA	P	11	✓
Atlantic cod	*Gadus morhua*	MJ	P	15	✓
Whiting	*Merlangius merlangus*	MJ	P	15	✓
Saithe	*Pollachius virens*	MA	P	15	×
Montagu's seasnail	*Liparis montagui*	MA	BI	11	✓ - Common sea snail and sea snail indet.
Plaice (larvae)	*Pleuronectes platessa*	MJ	BI	1, 11, 12, 14, 15	✓
Bib (pout-whiting)	*Trisopterus luscus*	MJ	BI	11	✓
Dab	*Limanda limanda*	MJ	BI	11, 12	✓
Dover sole	*Solea solea*	MJ	BI	11, 12, 14, 15	✓
Dace	*Leuciscus leuciscus*	FW	Z	4, 5	×
Common bleak	*Alburnus alburnus*	FW	Z	15	×
Zander	*Stizostedion lucioperca*	NA	NA	15	×
Roach	*Rutilus rutilus*	FW	Z	4	✓
Sea trout	*Salmo trutta*	CA	P	15	× (Recorded in RIMP)
European smelt	*Osmerus eperlanus*	CA	Z	4, 15	×
Ninespine stickleback	*Pungitius pungitius*	CA	Z	15	×
Lesser weever fish	*Echiichthys (Trachinus) vipera*	MA	BI	11, 12	✓

References: 1. Green, *et al*., (2009). 2. Natural Capital Committee, (2013). 3. Cattrijsse *et al*., (1997). 4. Colclough *et al*., (2005). 5. Colclough *et al*., (2003). 6. Stamp *et al*., (2023). 7. Fonseca, L., (2009). 8. Green *et al*., (2012a). 9. Green *et al*., (2012b). 10. Laffaille *et al*., (2000a). 11. Laffaille *et al*., (2000b). 12. Laffaille *et al*., (1998). 13. Laffaille *et al*., (2002). 14. McCormick *et al*., (2021). 15. Mathieson *et al*., (2000).

A.2 Seagrass bed

Table 7 Finfish species associated with seagrasses from selected available literature for Northern Europe. Species predicted to be impinged at HPC are shown in blue. Common seagrass species contributing the greatest abundance or biomass from Bertelli and Unsworth (2014), Unsworth *et al.,* (2014) and Furness and Unsworth (2019) are shown in bold.

Common name	**Scientific name**	**Functional Guild**	**Feeding Guild**	**Reference**	**Present in either CIMP record**
Pollack	***Pollachius pollachius***	**MJ**	**P**	**20, 21, 23, 24, 25**	✓
European Plaice	***Pleuronectes platessa***	**MJ**	**BI**	**20, 23, 24, 25**	✓
Sand goby	***Pomatoschistus minutus***	**ER**	**BI**	**21**	✓
Sand smelt	***Atherina presbyter***	**MJ**	**Z**	**23, 24**	✓
Poor cod	***Trisopterus minutus***	**MA**	**BI**	**21, 23, 24**	✓
Whiting	***Merlangius merlangus***	**MJ**	**P**	**20, 23, 24, 25**	✓
Gobies	***Gobiidae***	**-**	**-**	**24**	✓
Fifteen-spined stickleback	***Spinachia spinachia***	**ER**	**Z**	**21, 23, 24**	✓
Sandeels	***Ammodytidae***	**ER**	**Z**	**23, 24**	✓
Lesser spotted dogfish	***Scyliorhinus canicula***	**MA**	**P**	**23, 25**	✓
Common goby	***Pomatoschistus microps***	**ER**	**BI**	**21, 25**	✓
Dragonet	***Callionymus lyra***	**MA**	**BI**	**21, 23, 24, 25**	✓
Atlantic cod	*Gadus morhua*	MJ	P	20, 22, 23, 24, 25	✓
Bib (pout-whiting)	*Trisopterus luscus*	MJ	BI	20, 21, 23, 24	✓
Greater pipefish	*Syngnathus acus*	ER	Z	21, 23, 24	✓
Ballan wrasse	*Labrus bergylta*	MA	BI	21, 23, 24	✓
Saithe	*Pollachius virens*	MA	P	20, 23	×
Five-bearded rockling	*Ciliata mustela*	MS	BI	21, 23	✓
Thick-lipped mullet	*Chelon labrosus*	MJ	Z	20	×
Herring	*Clupea harengus*	MJ	Z	20	✓
Thornback ray	*Raja clavata*	MA	BI	21	✓
Brill	*Scophthalmus rhombus*	MA	P	20	✓
Flounder	*Platichthys flesus*	ER	BI	25	✓
Dover sole	*Solea solea*	MJ	BI	21	✓
Dab	*Limanda limanda*	MJ	BI	25	✓
Goby, Painted	*Pomatoschistus pictus*	MA	BI	21	✓
Sand goby (*Pomatoschistus spp)*	*Pomatoschistus spp*	MA/ER	BI	21	✓
Two-spotted goby	*Gobiusculus flavescens*	MA	BI	21	×
Corkwing wrasse	*Symphodus (Crenilabrus) melops*	MA	BI	21	×
Goldsinny wrasse	*Ctenolabrus rupestris*	MA	BI	21	× (Recorded in RIMP)
Cuckoo wrasse	*Labrus mixtus*	MA	BI	21	× (Recorded in RIMP)
Rock cook	*Centrolabrus exoletus*	MA	Z	21	✓
Baillon's wrasse	*Symphodus bailloni*	-	-	21	×

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Snake pipefish	*Entelurus aequoreus*	MA	Z	21	✓
Broadnosed pipefish	*Syngnathus typhle*	ER	Z	21	×
Lesser sandeel	*Ammodytes tobianus*	ER	Z	21	✓
Clingfish	*Lepadogaster candelloni*	-	-	21	×
Shore clingfish	*Lepadogaster lepadogaster*	MS	P	21	×
Small-headed clingfish	*Apletodon microcephalus*	MA	BI	21	×
Montagu's seasnail	*Liparis montagui*	MA	BI	21	✓ - Common sea snail and sea snail indet.
Dory (John dory)	*Zeus faber*	MA	P	21	✓
Lumpsucker	*Cyclopterus lumpus*	MS	BI	21	✓
Longspined bullhead	*Taurulus bubalis*	MA	BI	21	×
Black seabream	*Spondyliosoma cantharus*	MJ	BI	21	× (Recorded in RIMP)
Lesser weever fish	*Echiichthys (Trachinus) vipera*	MA	BI	23	✓

References: 20. Bertelli and Unsworth., (2014). 21. Jackson *et al*., (2006). 22. Lilley and Unsworth., (2014). 23. Peters *et al*., (2015). 24. Unsworth *et al*., (2014). 25. Furness and Unsworth., (2019).

A.3 Kelp forest

Table 8 Finfish species associated with kelp forests from European literature. No quantitative measures of abundance were available therefore, species have been ordered by presence in different studies.

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Pollack	*Pollachius pollachius*	MJ	P	*41, 42, 43*	✓
Ballan wrasse	*Labrus bergylta*	MA	BI	*41, 42, 43*	✓
Corkwing wrasse	*Symphodus (Crenilabrus) melops*	MA	BI	*41, 42, 43*	✓
Goldsinny wrasse	*Ctenolabrus rupestris*	MA	BI	*41, 42, 43*	× (Recorded in RIMP)
Cuckoo wrasse	*Labrus bimaculatus*	MA	BI	*41, 42, 43*	× (Recorded in RIMP)
Rock cook	*Centrolabrus exoletus*	MA	Z	*41, 42, 43*	✓
Atlantic cod	*Gadus morhua*	MJ	P	*41, 42*	✓
Longspined bullhead	*Taurulus bubalis*	MA	BI	*41, 42*	×
Rock gunnel / butterfish	*Pholis gunnellus*	ER	BI	*41, 42*	×
Mackerel	*Scomber scombrus*	MA	Z	*41, 42*	×
Two-spotted goby	*Gobiusculus flavescens*	MA	B	*42, 43*	×
Lesser sandeel	*Ammodytes tobianus*	ER	Z	*42, 43*	✓
European sea bass	*Dicentrarchus labrax*	MJ	P	*42, 43*	✓
Thicklip grey mullet	*Chelon labrosus*	MS	D	*42, 43*	×
Tompot blenny	*Parablennius gattorugine*	MA	B	*42, 43*	✓
Lesser spotted dogfish	*Scyliorhinus canicula*	MA	P	*42, 43*	✓
Snake pipefish	*Entelurus aequoreus*	MA	Z	*42, 43*	✓

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Common dragonet	*Callionymus lyra*	MA	B	*42, 43*	✓
European Plaice	*Pleuronectes platessa*	MJ	BI	*41*	✓
Whiting	*Merlangius merlangus*	MJ	P	*41*	✓
Saithe	*Pollachius virens*	MA	P	*41*	×
Five-bearded rockling	*Ciliata mustela*	MS	BI	*41*	✓
Bullrout / Short-spined sea scorpion	*Myoxocephalus scorpius*	ER	P	*41*	×
Pogge (armed bullhead)	*Agonus cataphractus*	ER	BI	*41*	✓
Topknot	*Zeugopterus punctatus*	MA	BI	*41*	✓
Lumpsucker	*Cyclopterus lumpus*	MS	BI	*41*	✓
Atlantic horse mackerel	*Trachus trachurus*	MA	P	*41*	✓
Starry ray	*Raja radiata*	-	-	*41*	×
Spiny dogfish	*Squalus acanthias*	MA	P	*41*	×
Norway pout, bib, poor cod	*Trisopterus spp*	MA/MJ	B	*42*	✓
Shore rockling	*Gaidropsarus mediterraneus*	MA	B	*42*	✓
Fifteen-spined stickleback	*Spinachia spinachia*	ER	Z	*42*	✓
Sand goby	*Pomatochistus spp*	MA	-	*42*	✓
Conger eel	*Conger conger*	MA	P	*42*	✓
Broadnosed pipefish	*Syngnathus typhle*	ER	Z	*42*	×
Bib (pout-whiting)	*Trisopterus luscus*	MJ	B	*43*	✓

References: 41. Norderhaug *et al*., (2005) 42. Jackson-Bué *et al*., (2023) 43. Eger *et al*., (2023) and data sources therein including Smale (2020) unpublished data for Wales and Plymouth.

A.4 Native oyster reef

Table 9 Finfish species associated with native oyster reefs from European (North Sea) literature. No quantitative measures of abundance were available, therefore species have been ordered by frequency of presence in different studies.

Common name	Scientific name	Functional Guild	Feeding Guild	Reference	Present in either CIMP record
Rock gunnel / butterfish	*Pholis gunnellus*	ER	BI	30, 31, 32, 33	×
Five-bearded rockling	*Ciliata mustela*	MS	BI	30, 31, 33	✓
Tompot blenny	*Parablennius gattorugine*	MA	BI	30, 32, 33	✓
Two-spotted goby	*Gobiusculus flavescens*	MA	BI	30, 32, 33	×
Black goby	*Gobius niger*	ER	BI	30, 32, 33	✓
Gobies	*Gobiidae*	-	-	31, 32	✓
Bullrout / Short-spined sea scorpion	*Myoxocephalus scorpius*	ER	P	30, 33	×
Big-eyed sand-smelt	*Atherina boyeri*	ER	P	30, 33	✓
Longspined bullhead	*Taurulus bubalis*	MA	BI	30, 33	×
Atlantic cod	*Gadus morhua*	MJ	P	30, 33	✓
European sea bass	*Dicentrarchus labrax*	MJ	P	30	✓
Dab	*Limanda limanda*	MJ	BI	31	✓
Tadpole fish	*Raniceps raninus*	ER	O	31	× (Recorded in RIMP)
Pogge (armed bullhead)	*Agonus cataphractus*	ER	BI	31	✓
Goldsinny wrasse	*Ctenolabrus rupestris*	MA	BI	31	× (Recorded in RIMP)
(Striped) red mullet	*Mullus surmuletus*	MA	BI	31	✓
Dragonet	*Callionymus lyra*	MA	BI	31	✓
Thornback ray	*Raja clavata*	MA	BI	32	✓

References: 30. Christianen *et al*., (2018). 31. Didderen, *et al*., (2020). 32. Didderen *et al*., (2019). 33. Sas *et al*., (2018).

Appendix B Assumptions of trophic transfer efficiency approaches: can the approaches be applied to the Severn Estuary?

Trophic transfer efficiencies are challenging to measure and vary within and among habitats. In this section, the applicability of the trophic transfer efficiencies in relation to the assumptions detailed in Section 3.1.1 are considered.

B.1 Primary production as a resource

In the context of saltmarsh and kelp forests primary production is taken as production of macrophytes alone. In these habitats other sources of primary production would exist, such as microphytobenthos (MPB), and the habitats will harbour consumers that utilise production transported through the habitat by currents and tides.

In saltmarshes high above ground and below ground production occurs and there are fluxes of particulate organic matter (POM), dissolved organic matter (DOM) and inorganic nutrients between the saltmarsh and surrounding coastal environments (Hample, 2003; Laffaille *et al*., 1998). Above ground primary production of 1,080 – 1,990 g DW m^{-2} y^{-1} (equivalent to ~ 490 – 890 g C m^{-2} y^{-1}) has been reported in the saltmarshes of Mont Saint-Michel Bay, France at different marsh heights.

Studies at Plum Island Estuary, Massachusetts demonstrated that overall, infauna of *Spartina* saltmarsh habitats relied on phytoplankton and microphytobenthos (MPB) as dominant food resources. Generally, detritus from the vascular marsh plants, *Spartina alterniflora* and *S. patens*, was of limited dietary importance to many consumers. Instead, microalgae, including epipelic (MPB) and epiphytic diatoms and phytoplankton were dominant basal resources in the saltmarsh food web (Galvan, 2008). MPB are also the most important energy source for the saltmarsh consumers at Mont St. Michel (Creach *et al.,* 1997).

In kelp forests in the eastern Pacific Ocean the relative input of primary production coming from kelp and phytoplankton was studied through the diet of diverse fish species. Phytoplankton-derived carbon uptake by fish was ~65% in species that forage in the water column and ~50% in fish foraging on or near benthic substrates (Zuercher, 2022). Another study from Pacific kelp forests proved that the phytoplankton-derived carbon comprises 60 – 80% of fish tissue carbon with kelp-derived carbon accounting for 20 – 40% (Koenigs *et al.,* 2015).

In the Severn Estuary phytoplankton production is low due to the highly turbid environment. The primary production of phytoplankton in the inner Bristol Channel was estimated at 6.8 g C m^{-2} y^{-1} (Joint & Pomroy, 1981). MPB form biofilms on intertidal mud and sand and within saltmarsh creeks. The average production of MPB within the Severn Estuary is 52.5 g C m^{-2} y^{-1} in mud and 17.5 g C m^{-2} y^{-1} in sand with an average value of 33 g C m^{-2} y^{-1} (Underwood, 2010). The combined production of phytoplankton and microphytobenthos might be estimated as 59.3 g C m^{-2} y^{-1} in mud and 24.3 g C m^{-2} y^{-1} in sand. Intertidal microphytobenthos production in North European estuaries typically ranges from 29 C m^{-2} y^{-1} to over 300 C m^{-2} y^{-1} (Underwood & Kromkamp, 1999). Primary production in the Severn Estuary is relatively low. Therefore, the additional primary production provided by restored habitats has the potential to provide an important additional food source supporting food webs.

B.2 Transfer of primary production to first level consumers:

Some species of fish such as mullet, that are commonly found in saltmarsh systems, utilise primary and detritic production directly, exporting it from saltmarsh system (Laffaille *et al*., 2002). Other species feed on benthic invertebrates or are piscivorous and therefore rely on the flux of carbon through the food chain.

For three predominating subtidal benthic communities of the Bristol Channel ("*Venus", "Abra*" and "*Modiolus*") production was estimated to be 25.8, 14.2 and 34.1 g AFDW m^{-2} y^{-1} (Warwick, 1984). Using the generic conversion AFDW to DW of 0.75, and from DW to carbon as 0.314, the respective carbon production of these three communities might be estimated as 10.8, 5.9 and 14.3 g C m^{-2} y^{-1}. The *Venus* community is associated with sands, *Abra* with muddy bottoms, and *Modiolus* with hard bottoms (Davies, 1998). These sediment types occur around Hinkley Point with mud predominating in the immediate vicinity (BEEMS Technical Report TR136a[13]). In the Inner Channel, where Hinkley Point is situated, high turbidity limits subtidal benthic production with estimated benthic production of 1.30 g C m^{-2} y^{-1} in deposit feeders and only 0.024 g C m^{-2} y^{-1} in suspension feeders (Radford, 1994).

Hamon grab samples of the subtidal sediment within Bridgwater Bay identified low biomass of benthic infauna at approximately 0.5 g AFDW m^{-2} (equivalent to approximately 0.21 g C m^{-2}) (BEEMS Technical Report TR136a). In the intertidal muddy sediments, higher benthic biomass has been estimated at 29.9 – 92.0 g WW m^{-2} (BEEMS Technical Report TR360[14]). This may equate to production of approximately 2.4 – 7.3 g C m^{-2} y^{-1} [15]. Accordingly, the available evidence suggests the intertidal benthic production in the mudflats is approximately 5 – 14% of the average MPB production reported in intertidal mudflats in the Severn Estuary by Underwood (2010) (Table 2).

In the saltmarsh systems of the Westerschelde Estuary, macrofauna biomass dominated by the amphipod *Corophium volutator*, the polychaete *Nereis diversicolor*, the bivalve *Macoma baltica* and *Oligochaeta* was positively correlated with salinity and ranged from average site values of 1.36 – 13.98 g AFDW m^{-2} (Hampel, 2003). Assuming a P:B ratio of 1.2, this may equate to production of approximately 0.7 – 7 g C m^{-2} y^{-1} [16]. The benthic fauna in the Westerschelde are consumed by flounder, sea bass, gobies, and juvenile shrimp (Hampel, 2003).

In highly productive Norwegian kelp forest systems, where primary production can exceed 3,000 g C m^{-2} y^{-1}, Norderhaug & Christie (2011) conservatively estimated secondary production rates by first-level consumers of 3 – 8% of the total primary production. The kelp associated invertebrate species consisted of polychaetes, isopods, amphipods, decapods, gastropods bivalves and echinoderms with production rates ranging from 68 (SE ±18) g DW m^{-2} y^{-1} – 308 (± 64) g DW m^{-2} y^{-1} along an exposure gradient[17]. The authors suggested that the relatively low values of trophic transfer efficiency, which they note are typically closer to 20% (double that applied in Table 2), may be due to the kelp associated fauna not being food limited and only a proportion of the primary production being utilised *in-situ* with the remainder exported to adjacent coastal systems (Norderhaug & Christie, 2011).

In the case of seagrass, the efficiency at which primary production is converted up the food web is highly variable. In productive Mediterranean *Posidonia* systems 3 – 5.3% of total primary production was converted to meiofauna production consisting of copepods, polychaetes and turbellarians. Higher rates in intertidal

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[13] BEEMS Technical Report TR136a. A comparison of macrobenthic fauna and sediment characteristics from Hamon grab and Day grab samples taken at Hinkley Point, 2010. Cefas, Lowestoft.

[14] BEEMS Technical Report TR360. Intertidal mudflat monitoring in Bridgwater Bay. Cefas, Lowestoft.

[15] Based on a P:B coefficient of 1.2 and a general WW:DW conversion of 0.21 and a C:DW conversion of 0.314 as an average value for including Mollusca, Annelida and Echinodermata from Brey *et al.,* (2010).

[16] A generic AFDW:DW conversion of 0.750 is applied for all taxa and a C:DW conversion of 0.314 is an average including Mollusca, Annelida and Echinodermata from Brey *et al.,* (2010).

[17] Equating to approximately 20 – 97 g C g m^{-2} y^{-1} after applying general DW:C conversion factor of 0.314 as used in BEEMS Technical Report TR584 from Brey *et al*., (2010).

Zostera beds in the Atlantic have been reported elsewhere where meiofauna trophic transfer efficiencies are 15 – 32% of primary production (Danovaro *et al*., 2002).

Based on these literature values, it is clear that high quality kelp, seagrass and saltmarsh habitats have the potential to provide enhanced secondary production in comparison to the existing intertidal and subtidal communities. A proportion of the production from high quality habitats may be exported to adjacent intertidal and subtidal habitats supporting higher secondary production in a wider area.

B.3 Transfer to higher consumers including fish:

In temperate ecosystems trophic transfer efficiencies between trophic level 2 and trophic level 4 have been estimated to be 8.1% ± 2.4 (Du Pontavice *et al*., 2020), whereas median transfer efficiencies of 10% for each trophic level up to level 4 were proposed by Rosenberg *et al.,* (2014) based on over 200 Ecopath with Ecosim (EwE) models.

Trophic transfer efficiencies have been shown to decrease with increasing trophic level and body size (Barnes *et al.,* 2010). In central North Sea food webs, trophic transfer efficiencies of all fauna (excluding seabirds), from 2 g to over 2 kg WW, was assessed to be between 3.7% to 12.4%. However, when larger fauna were excluded, and trophic transfer efficiencies were considered only for fauna 2 g – 512 g and 2 g – 256 g, trophic transfer efficiencies increased to 12.4 % and 27.1%, respectively (Jennings *et al*., 2002). In the habitat restoration model proposed by French McCay & Rowe (2003) trophic transfer efficiencies were defined as follows:

- 20% for fish and invertebrates < 200 g.
- 10% for fish 200 g to 1 kg.
- 4% for fish > 1 kg.

Typical trophic levels of the fish species impinged in the greatest biomass at Hinkley Point range from 2.3 for detrital feeding thin lipped grey mullet (Froese & Pauly, 2023), to 3 for sprat, 3.4 for herring, and over 4 for the predominantly piscivorous species. The fish impinged are predominantly small juvenile stages and over 99% of the herring, whiting, cod and seabass impinged in CIMP1 and CIMP2 were individuals below 200 g. Created habitats are expected to support a range of different taxa and provide nursery habitat for the juvenile fish that utilise the estuary. The juvenile fish in the created habitats are expected to have higher production to biomass ratios and lower trophic levels than larger fish. It is therefore feasible that trophic transfer efficiencies tend from 10% towards 20%. However, trophic transfer from secondary production to subsequent higher trophic levels would not be entirely apportioned to fish, with macrofauna, and particularly large crustaceans, competing for available resources.

In the Salem example, crabs and terrapins accounted for approximately 10% of the production at the secondary consumer level with the remaining productivity apportioned to finfish (Balletto *et al.,* 2005). The transfer of production between trophic levels in the complex food web in the Severn Estuary, and the proportion of production transferred to fish relative to large crustaceans is unknown. It is clear from impingement monitoring that there are large interannual variations in the biomass of finfish relative to macrocrustaceans, therefore production of these two groups is likely to vary over time with the relative proportions changing.

As a starting position, the trophic transfer efficiencies in Table 2 are applied with the assumption that 2% of primary production is converted to fish production. There is a high degree of uncertainty in estimating the proportion of primary or secondary production that would support fish production. Therefore, fish production estimates derived using these methods must be treated as approximations and the results and assumptions are challenged with literature evidence for each habitat type.

Appendix C Summary of production estimates

Table 10 Area estimate approximations of the productivity delivered by single habitat types.

Habitat type	Approach	Production metric	Extent proposed	Production estimate	Context	Source
Saltmarsh	Application of model values from Salem.	Secondary consumer production estimate, assumed to be fish, derived from Salem Aggregated Food Web model: 6 g WW m^{-2} y^{-1}.	**340**	20,400 kg WW y^{-1}	The example is not site specific and assumes same fish production rates as Delaware Bay model. Saltmarsh production may be exported to adjacent habitats providing greater benefits.	Based on transfer efficiencies from the aggregated food web model for Salem (see Balletto *et al.*, 2005).
Saltmarsh	Trophic transfer efficiencies (TTE).	TTE based on 2% of available saltmarsh macrophyte primary production being converted into secondary consumers secondary consumers at a rate of 5.52 g WW m^{-2} y^{-1}.	**340**	18,752 kg WW y^{-1}	Proportion of secondary production by macrocrustaceans and fish is unknown. Saltmarsh production may be exported to adjacent habitats.	Section 3.2.1.
Seagrass	Trophic transfer efficiencies (TTE).	Assuming fish production is approximately equivalent to 2% of primary production: Seagrass primary production assumed to	**5**	730 kg WW y^{-1}	Sparse regional data of seagrass primary production and fish biomass/production to validate predictions. Assessment underpinned by initial production rates and TTE	Seagrass production based on a range of sources (Section 3.2.1).

Habitat type	Approach	Production metric	Extent proposed	Production estimate	Context	Source
		be 100 g C m^{-2} y^{-1} resulting in fish production of 14.6 g WW m^{-2} y^{-1}.			assumptions.	
Seagrass	Direct numbers comparison.	Porth Dinllaen, North Wales: seagrass habitat supports ~6,000 fish ha^{-1} with a habitat enhancement of ~4,700 fish ha^{-1} over adjacent sandy habitats.	5	30,000 fish or approximately 23,500 additional fish.	The seagrass in the Pen Llyn a'r Sarnau SAC in Porth Dinllaen, North Wales is a large well-established site. Restoration projects unlikely to achieve equivalent ecological production.	Bertelli & Unsworth (2014)
Seagrass: cod	Enhancement in juvenile cod numbers comparison.	Mean density of juvenile cod (0-group) in the Northeast Atlantic estimated at 235.6 ± 23.5 fish ha^{-1}	**5**	1,178 ± 117.5 juvenile cod	Annual recruitment and estuarine production of juvenile cod can vary by an order of magnitude between years.	Lilley & Unsworth, (2014)
Kelp	Fish productivity comparison.	Estimate of fish production following the methods in Eger *et al.,* (2023) from studies in southern Wales (average for two sites ~160 g WW m^{-2} y^{-1}) and Plymouth (average for three sites ~82.5 g WW	**15**	12,000 - 24,000 kg WW y^{-1}	Site suitability studies and trials would be required prior to scaling up, records for kelp exist in rocks reefs on both sides of the outer Bristol Channel, west of the SAC.	Smale (2020) – unpublished data collated as part of the database for Eger *et al.,* (2023). The approach to biomass and productivity makes assumptions discussed further in

Habitat type	Approach	Production metric	Extent proposed	Production estimate	Context	Source
		m^{-2} y^{-1}).				the text.
Kelp	Trophic transfer efficiencies (TTE).	Assuming fish production is 2% of primary production: Kelp primary production ~ 425 g C m^{-2} y^{-1} as an average UK estimate. Secondary consumers assumed to be fish biomass. Kelp C production converted to fish biomass using equations from Brey *et al.,* (2010).	**15**	9,300 kg WW y^{-1}	TTE predictions for secondary consumers are < 65 g WW m^{-2} y^{-1}, lower than UK estimates of fish production (above) and typical values for the NE Atlantic reported in Eger *et al.,* (2023).	Kelp primary production estimates derived from UK data collated as part of the database for Eger *et al.,* (2023). Assessment underpinned by initial production rates and assumed TTE rates.