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Erratum

In November 2018, an apparent inconsistency in the text of this report SAR 008 was discovered. Section 7.3, p.113 states (repeated in Executive Summary p4 and p5):

> "In line with international good practice, it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel as a 98%-ile";

whereas the original reference in Turnpenny and Liney 2007, in section 2.1.2 on p2-7 of that report stated:

> "in a river or estuarine channel, the plume should not occupy more than a certain percentage of the channel cross-section for more than a stated percentage of the time (e.g. a maximum of 25% of the cross-section for 95% of the time); the actual percentage figures may vary between regulatory agencies."

After careful investigation we the authors are satisfied that no new evidence (subsequent to the 2007 references) on thermal barriers had influenced the SAR008 recommendations on this subject area, and we have concluded that the reference to a 98th percentile was a typing error that had not been picked up in proof reading and that the correct figure should have been the 95th percentile that is found in the earlier source documents. On that basis this report is hereby reissued with the relevant references to a 98th percentile (itemised below) replaced by 95th percentiles.

On behalf of BEEMS Expert Panel: Signature

Name Dr. Andrew Turnpenny

Date 1st May 2019

Original SAR008 text where 98% has been replaced by 95%

Executive Summary

Recommendations for thermal limits - page 4

The retention of uplift values within future standards is essential in order to avoid fish acclimation issues, phenological changes and potential thermal blockage that might be associated with extreme values. Furthermore, in line with international good practice it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel as a 98%-ile.
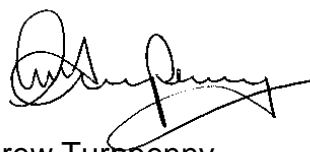
Regulatory considerations – page 5

Based on the evidence reviewed and presented within the current report, thermal standards are proposed for estuarine and coastal waters in England and Wales. In addition, it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel (as a 98%-ile).

7.3 Recommended boundary and uplift values – page 113

Under the WFD, boundary values have to be established to protect the most sensitive taxa; these would apply equally to the Habitats Directive. Given the rationale in this report, Table 32 sets out recommendations for temperature boundary values. In line with international good practice, it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel as a 98%-ile.

Quotes from original source references

1.   Turnpenny, Coughlan and Liney (2007). Review of Temperature and Dissolved Oxygen Effects on Fish in Transitional Waters, Report to Environment Agency. Section 6.2.3, p6-66. Jacobs.

 "With regard to regulation of thermal discharges, it is suggested that, for 95% of the time, the thermal plume should not be allowed to occupy more than 25% of the estuarine channel cross-section at any position; the 95% stipulation would be to allow for the unavoidable spread of the plume at slack water tides under some conditions. "

2.   Turnpenny and Liney (2007). Review and development of temperature standards for marine and freshwater environments. Report to Sniffer. Jacobs Engineering.

Section 2.1.2. p2-7.

 "Good practice design objectives for a thermal discharge can be summarised as follows:

- the outfall should be located and designed so as to minimise contact of warmed water with sensitive habitats under all hydraulic conditions;

- recirculation of the plume into the plant CW intake should be minimised (recirculation reduces plant thermal efficiency);

- in a river or estuarine channel, the plume should not occupy more than a certain percentage of the channel cross-section for more than a stated percentage of the time (e.g. a maximum of 25% of the cross-section for 95% of the time); the actual percentage figures may vary between regulatory agencies."

Section 7.2.2 p7-2.

"A number of countries reviewed in this study place limits on the allowable spread of the plume within river and estuarine channels, limiting the plume to 25% of the cross-sectional area of the channel. This is a sensible precaution to protect migratory channels. It is proposed that this limit should also be included in UK standards, such that the mixing zone should be contained within ≤25% of the channel cross-sectional area for 95% of the time. The 5% time allowance recognises the uncontrollable spread of the plume in estuarine channels under slack water conditions; this should allow ample opportunity for migratory fish to pass."

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Background

The abstraction and return of seawater used for cooling represents the most important environmental aspect to the marine environment of nuclear power station operation. The discharge introduces significant thermal energy (heat) to receiving waters, which will continue with little variation throughout the operational life of the station, which may exceed 40 years. Return cooling-water will typically be 8–10°C higher than background. A modest temperature rise adjacent to the discharge is inevitable with little practical opportunity for mitigation once the station is commissioned.

Although European and UK legislation includes *inter alia* thermal limits, there are currently no uniform standards available to the Environment Agency and other UK regulators for the control of temperature or thermal loads in Transitional (estuaries) and Coastal (TraC) waters. Current thermal controls have been derived mostly on pragmatic grounds and have worked tolerably well, but they have a number of fundamental flaws. Many of the current limits are either based on freshwater standards or rely heavily on an intuitive view on the behaviour of fish (and other marine organisms) with little scientific justification, whilst the potential impacts of a systemic rise in seawater temperature brought on by climate change has largely been ignored.

This report critically reviews the available scientific information from published and grey literature over the past 50 years. Based on data from numerous laboratory and field studies, detailed information is given on the thermal tolerance for a wide range of species which will be relevant in the context of the new nuclear build (NNB) programme in the UK. Context for this information is given in the earlier chapters, which review current legislation, discharge plume behaviour and possible influences of water quality issues.

The report provides the developer and regulator with:

➢ confidence that the local ecosystem will not be damaged by power station operation throughout the anticipated operating life in a context of systemic climate change;

➢ standards to ensure legitimate ecological conservation and fishing interests will be protected;

➢ assurance that investment in control measures provides environmental benefits.

In particular, this report provides an overview of all significant issues pertaining to thermal discharges to TraC waters with special consideration given to the requirements at Hinkley Point and Sizewell.

In essence, the review confirms the fact that adverse effects of cooling water (CW) outfalls are restricted to an area close to the plume, that temperature rises up to 3°C appear to be tolerable and that resulting temperatures less than 27°C have no clear deleterious impact on species in the receiving waters. However, in the longer term, changes in the local community may result as species with differing tolerances of elevated temperature show differing survival, growth and patterns of reproduction from those expressed under ambient conditions. Populations that persist in the receiving area of a heated CW effluent will acclimate to those new local conditions and may adapt in response to them. Superimposed on these changes, however, are community changes caused by species distribution alterations as the result of climate change.

Fish

Despite the very large amount of literature dealing with thermal discharges, very few large-scale mortalities have occurred which can unequivocally be related to high temperature. The temperature uplift outside the immediate discharge from power stations is normally within the limits of natural variation, so acute impacts are unlikely.

Most fish have thermal preferenda that defines the thermal niche where fish will seek to spend most of their time. Although they may, over time, modify their preferred range if there is an imperative – eg, food availability – thermal mixing zones could represent lost habitat for some species, though cold-water fish may be able to occupy the area beneath a buoyant plume and thermal addition may extend the season for which juvenile fish are able to occupy marginal areas.

Fish have a wide range of thermal sensitivities, with Arctic–Boreal affinities (cold-water species) being most likely to be disturbed by thermal discharges. Lethal temperatures, when used together with thermal mapping around discharges, indicate areas from which fish will be excluded, although they have a powerful ability to use behavioural thermoregulation in order to remain in areas of preferred temperature. Observed behavioural preferenda and avoidance thresholds, together with observed temperatures for optimum growth, therefore provide a better indication of how fish distribution is likely to be affected under warmer conditions.

As would be expected, these temperatures are lower for cold-water species. Since both warm- and cold-water species coexist in waters around England and Wales, the development of suitable boundary values needs to take account of both groups. Boundary values for upper-allowable temperatures therefore are best based on the more sensitive cold-water species, and therefore only this typology is considered. Despite this, it is accepted that with changing ambient temperatures due to climate change, cold-water species are being replaced by warm-water ones, hence altering the fish community.

The evidence of warm-water discharges causing 'thermal barriers' inhibiting migratory fish is equivocal, with evidence of salmon passing through rapid temperature changes as high as 10°C but a single failed attempt may inhibit migration altogether. Available data from monitoring of (cold-water) salmonid migrations through UK estuaries, especially evidence from more than two decades of monitoring in the Thames Estuary, provide no evidence of temperature-based water-quality barriers to diadromous fish migrations. Fish-tracking evidence shows that, during the highly motivated river entry phase, salmon will cross abrupt temperature changes as high as 9–10°C. Where such movements are inhibited, low dissolved oxygen (DO) concentration is invariably a factor; provided that Water Framework Directive (WFD) DO standards are maintained, this factor could be excluded. However, under some climate-change predictions for the future, temperature *per se* could become inhibitory to fish movement. It is recommended that this aspect and other synergistic responses between temperature and other water quality elements should be kept under regular review.

Far-field temperature rises from effluents (or global warming) might cause species living close to the upper limit of their thermal niche to move away. In the case of cold-water anadromous species such as salmonids or smelt, it is possible, but by no means proven, that they would fail to penetrate areas where the temperature is raised above their thermal niche limit to reach spawning grounds. The outcome will be a function of their ability to detect and avoid thermal changes, their motivation to spawn and their ability to swim though the affected area. If the temperature in a fish spawning area is artificially elevated, then spawning and early development could fail, since embryonic stages have narrower thermal niches than older life-stages. Effective regulation of temperatures in spawning areas is therefore critical.

Special consideration needs to be given to seasonal spawning areas, where for some species narrow temperature limits appear to be required for spawning. It is not clear whether some apparently narrow limits are merely artefacts of inadequate observation or of the background temperatures to which fish have become acclimated. The report reviews the impacts of discharges on spawning areas, although it is recommended that thermal discharges should be located so as to avoid important spawning areas.

Raised temperatures can make some waters more favourable as nursery areas, as is evidenced by the frequency of bass nursery areas around power station outfalls. This is an extension of natural behaviour, with juveniles of many fish species migrating in and out of shallow marginal areas of estuaries and sandy bays on the open coast to maintain temperatures close to their thermal preferenda in summer, and retreating into deeper, warmer coastal waters to overwinter.

Invertebrates, algae and marine plants

Most marine invertebrates have temperature thresholds that initiate maturation and spawning, and so elevated temperatures have the potential to delay spawning (in cold-water spawners) or initiate earlier spawning in warm-water spawners. Once settled, shellfish and other sessile epibenthic and infaunal species are unable to actively avoid areas of elevated water temperature, but intertidal species in particular already experience a large diurnal and seasonal temperature range. Many species have a short generation time, with an ability to adapt to changed conditions at a population level.

For an area of habitat within a given isotherm within an effluent plume, it likely that species inhabiting that area, and whose normal habitat range experiences that (raised) temperature naturally at that time of year, will not be significantly adversely affected. Where the tidally shifting interface between effluent water and the ambient water contacts an area of seabed (or shore) habitat outside the mixing zone, it would be appropriate that the temperature differential across that interface should not be greater than 3°C.

Within British coastal and estuarine habitats, it is recommended that the temperature of the effluent water outside the mixing zone should not exceed 30°C for more than six hours in any one week during summer, and not at all at other times of year. This numerical limit will aid the regulator when defining the mixing zone.

Under the Habitats Directive and WFD there is a requirement also to protect macroalgae and marine plants. The review indicates and summarizes the relatively poor evidence for the requirements of these taxa, but it should be noted that these are largely based on cold-water populations.

General conclusions

➢ Existing standards (even where they have statutory basis) are not considered to be a realistic guide for the protection of the marine environment; freshwater limits have been uncritically transferred to marine waters, and there appears to have been a reliance on intuition; for example, in many cases there is an assumption that a cooling-water discharge forms a thermal barrier to migratory fish, yet this is supported by little experimental evidence. Current standards are almost entirely based on known responses of fish and take no account of other marine biota or putative climate-change effects.

➢ Whereas previous thermal standards developed in the UK, primarily for application to freshwaters, have distinguished between salmonid and cyprinid waters, this distinction is inappropriate for TraC waters, which support a greater diversity of fish species. In marine waters, it is of greater relevance to consider thermal responses of species based on zoogeography, considering, for example, Arctic–Boreal and Lusitanian domains (cold- and warm-water species, respectively).

➢ In general, fish and other swimming organisms can detect and actively avoid warmed waters if they perceive them to be harmful, making it unlikely that they will suffer significant impact from the shorter-term temperature elevation. Equally, planktonic organisms in the receiving water will only be at risk as the effluent mixes and dilutes. It is therefore predominantly benthic organisms that are at risk of being affected by thermal effluents, but this risk is moderated by the buoyancy of the plume. Hence the importance of a good knowledge of the hydrodynamics of the receiving waters.

➢ While there is little evidence to suggest that existing thermal discharges have created barriers to cold-water migratory fish species such as salmon, sea trout, eel and smelt (which are also of stated conservation importance), future responses must be considered against a background of longer term water temperature rise associated with climate change, which is gradually reducing the metabolic headroom for cold-water species. The much larger flows of cooling water planned for NNB stations could also be expected to have a greater potential for

this type of effect, although planned temperature rises (ΔTs) will remain similar to those for existing thermal stations.

➤ Marine coastal invertebrate species around the British Isles experience natural year-to-year temperature variations greater than the total ΔT predicted by the Intergovernmental Panel on Climate Change over the next century. Given rapid generation times (eg, annual), they are likely to adapt to this environmental change. Therefore, for invertebrates it is not expected that seawater temperature rises hypothesized by climate-change models will have any significant synergistic effect with those of CW effluents.

➤ Combining fish zoogeographic data with sea-surface temperature data indicates the preferred temperature ranges of fish. Seasonal movements of fish away from fringe areas, and the ability of some fish to exploit cooler, deeper layers in some areas, may prevent them from experiencing the full ranges of temperatures inferred, as has been demonstrated, for example, in cod. Hence the zoogeographic information provides a useful indication of temperature preferences of species for which there are no laboratory studies or other measurements.

➤ Although the UK Technical Advisory Group on the WFD has been unable to accept recommendations for draft temperature standards for TraC waters due to insufficient data, the more extensive information reviewed in the present report, now including invertebrate data as well as an expanded fish database, allows the reconsideration of the temperature boundaries and allowable changes.

Recommendations for thermal limits

The review presents an extensive database of temperature tolerance which covers the ranges of biological sensitivity, and hence tentative boundary values are proposed for allowable maximum water temperature. Although specifically designed for sensitive fish species, the boundary values reflect the WFD normative definitions for all quality elements in TraC for High, Good, Moderate and Poor classes, respectively (see Table ES1). The boundary values are required to protect the most sensitive taxa; these would apply equally to the Habitats Directive. Table ES1 also recommends the maximum temperature uplift values (ie, exceedence above ambient) and recommends that these values should be used provided also that absolute temperature limits would not be exceeded. The retention of uplift values within future standards is essential in order to avoid fish acclimation issues, phenological changes and potential thermal blockage that might be associated with extreme values. Furthermore, in line with international good practice it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel as a 95%-ile.

The proposed thresholds reflect evidence of naturally higher summer temperatures in southern parts of the United Kingdom. However, preferred temperatures for reproduction indicate that additional seasonal limits would be more protective in areas used for reproduction. The proposed boundary values are also defined by the more sensitive species, which are well represented by fish.

Table ES1 Proposed temperature boundary values for all TraC waters outside the mixing zone

Typology	Normative definition boundary positions (as annual 95%-ile)			
	High/Good	**Good/Moderate**	**Moderate/Poor**	**Poor/Bad**
Maximum Allowable Temperature*	23°C	23°C	28°C	30°C
Maximum Allowable Temperature Uplift	+2°C	+3°C	+3°C	+3°C

*These are general figures for England and Wales; it may be appropriate to adjust where maximum summer background water temperatures are higher or lower.

Regulatory considerations

Under the EU Water Framework Directive, boundary values have to be established to protect the most sensitive taxa; these would apply equally to the EU Habitats Directive. Based on the evidence reviewed and presented within the current report, thermal standards are proposed for estuarine and coastal waters in England and Wales. In addition, it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel (as a 95%-ile). The coastal and estuarine power plants constitute a plan or project for which an Appropriate Assessment is required under the EU Habitats Directive, and so any possible effects of thermal discharges on conservation objectives have to be considered. Similarly, while the EU Marine Strategy Framework Directive will apply to marine areas offshore by >1 nm from the coast, any effects of the NNB on the descriptors of Good Environmental Status will also have to be considered.

The abstraction and return of cooling water represent the most important environmental aspects of nuclear power station operation to the marine environment. The discharge introduces significant thermal energy (heat) to receiving waters, which will continue with little variation throughout the operational life of the station, which may exceed 40 years. The temperature uplift is normally within the limits of natural variation, so acute impacts are unlikely. However, a modest temperature rise in the vicinity of the discharge is inevitable, with little practical opportunity for mitigation once the station is built. It is therefore essential that the thermal tolerances of key species are understood and that station design reflects the ability of the ecosystem to adjust to moderate chronic stress. The potential additive effects of long-term background temperature rises associated with predicted climate change must also be taken into account.

1 Introduction and overview

1.1 Background to the report

In support of their New Nuclear Build programme, British Energy/EDF Energy use a management framework – **BEEMS** (**B**ritish **E**nergy **E**stuarine and **M**arine **S**tudies) – to co-ordinate its research activities in estuarine and coastal waters. Within BEEMS, an Expert Panel was established to provide independent scrutiny of the scientific output. Members of the Expert Panel identified the lack of appropriate and defensible thermal standards as a significant gap in the knowledge base. Subsequently, EDF Energy requested the Expert Panel to prepare an independent report addressing the key issues of thermal tolerance and to suggest appropriate standards.

1.2 Remit of report

An adequate supply of cooling water is a fundamental requirement of a nuclear power station, as it is for any thermal power station. The laws of thermodynamics dictate that for every megawatt of power generated, up to two megawatts must be discarded as low-grade (low temperature) heat. In some cases this low-grade heat can be used in district heating schemes, commercial horticulture or aquaculture, but opportunities for the beneficial use of waste heat around isolated sites are limited. Thus a nominal 1,600 MWe nuclear generating station discards over 2,500 MW_{th} into the environment as waste heat.

Heat disposal from thermal power stations can be into the atmosphere via cooling towers or via direct cooling into surface water bodies and thence into the atmosphere. Direct cooling is the most efficient in energy terms and also minimizes greenhouse-gas emissions. In the BREF cooling water document (European Commission, 2001), once-through cooling is identified as BAT (Best Available Technique) in the EU for coastal power stations. Coastal waters or estuaries are thus the logical sites for direct cooling. All current UK nuclear stations, and many fossil-fuelled stations, use direct cooling, and are sited on the coast or in estuaries (TraC waters). Direct cooling is the preferred solution adopted by EDF Energy for their UK new-build programme.

However, there have to be constraints (temperature standards) on the temperatures of the waters that are discharged in order that there are no significant adverse effects on the existing ecology or on biological processes. The thermal (temperature) standards applied to New Nuclear Build (NNB) are critical to the use of direct cooling and fundamentally influence intake and outfall design and positioning. The temperature standards applied and the perception of their impact on the environment will be a significant consideration in the granting of permits to build and operate new capacity.

Although various pieces of European and UK legislation include *inter alia* thermal limits, there are currently no uniform standards available to the Environment Agency and other UK regulators for the control of temperature or thermal loads in TraC waters. The industry has developed operating regimes for thermal discharges, but these have no formal status.

Since the last generation of nuclear power stations was commissioned, there has been significant development in European legislation, notably implementation of two key European Directives: Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (usually referred to as the Habitats Directive), and Council Directive 2000/60/EC establishing a framework for the Community action in the field of water policy (the Water Framework Directive: WFD). Public expectation has increased and it is important that standards applied today are fit for the medium to long term (>50 years) if station operators are not to be burdened with an unsustainable environmental liability.

Current thermal controls (mostly pragmatic) have worked tolerably well; however, they have a number of fundamental flaws:

> there is inconsistency in the use of allowed maximum temperature and allowed temperature elevation above ambient;

> they make little distinction between full water column, bottom water or surface plume temperature;

> there is little or no understanding of the relative risks of short-term impact vs. chronic exposure (ie, the effects of exposure frequency and duration);

> no account is taken of seasonality – more or less restrictive standards may be appropriate at certain times of the year;

> some standards appear to be based on an intuitive view of fish behaviour despite a lack of experimental justification;

> they assume a static background and take no cognizance of predicted seawater temperature rise (associated with long-term climate change) during the operating life of NNB stations;

> they are generic and take no account of local fish or benthic assemblages at sites of planned NNB;

> temperature is taken in isolation ignoring potential secondary impacts (eg, dissolved oxygen levels or ammonia toxicity) which may be synergistic or antagonistic;

> the Habitats Directive 'review of consents' process applies the thermal standards from the Shellfish Waters Directive, even though these were developed for the limited purpose of maintaining commercial shellfish stocks.

Consequently, an uncritical adoption of these existing limits may prove unnecessarily precautionary in some locations, whilst in others it may threaten key local features and interests (including fisheries) which it is beneficial to protect. Therefore, there is a clear requirement for defensible and scientifically rigorous thermal standards that can be used to allow construction of NNB, and other power stations, without compromising the legitimate uses and conservation value of the adjacent marine environment.

This report reviews the available literature on thermal impacts on marine and estuarine biota, particularly fish, considers the potential impacts in combination with other water quality features, considers the extent to which behaviour of the cooling water (CW) discharge plume modifies risk, and proposes appropriate thermal standards.

The report provides the developer and regulator with:

> confidence that the local ecosystem will not be damaged by power station operation throughout the anticipated operating life in a context of systemic climate change;

> standards to ensure legitimate ecological conservation and fishing interests will be protected;

> assurance that investment in control measures provides environmental benefits.

In particular, this report provides an overview of all significant issues pertaining to thermal discharges to TraC waters, with special consideration given to the requirements at Hinkley Point and Sizewell.

1.3   General introduction to cooling water systems

Electric power plants require cooling water to complete the steam cycle. Low-energy steam from the turbines is condensed in large heat exchangers or condensers, with the condensed water being

reused in the steam cycle. In directly cooled power stations this process of condensation uses large quantities of cooling water, with a nuclear power station of 1,600 MW generating capacity requiring *c.*70 m^3 s^{-1}. Coastal water provides a suitable supply for cooling water, but it contains many biological species that may affect the cooling systems. Additionally, the increased temperature of the discharged water may affect the ecology of the surrounding area.

Within the cooling system, bacteria can cause slime build-up leading to reduced heat exchange efficiency and can initiate microbial influenced corrosion (MIC). The planktonic larvae of barnacles, mussels, oysters etc. can attach to all parts of the cooling system and form thick mats in conduits and other plant structures. Fish may be sieved out (impingement) or, in the case of larval and 0+ fish stages, entrained with varying levels of mortality. Uncontrolled colonization by either the micro- or macro-fouling organisms would cause reduced efficiency.

In the CW system, the offshore intake system comprises a conduit that runs out from land, below the seabed, to open in relatively deep water some distance offshore. The openings of the intake are usually close to the seabed to avoid drawing in the warmer surface layers, and may be equipped with some kind of concrete capping arrangement (sometimes known as the velocity cap) to reduce abstraction from the surface layers and cavitation. Such an arrangement will also help to reduce fish ingress, by eliminating vertical water currents. Coarse screening is used here to prevent the entry of large items such as tree branches and fishing nets which might otherwise obstruct the CW tunnels.

The heated water runs generally under gravity in large conduits to the discharge area, where there is a large sump to accommodate surges in the discharge during changes in outfall pumping rate. The heated discharge water is pumped to the outfall, which has several ports that direct the water vertically into the water column and thus aid dispersion and dilution. The outfall is generally located several metres above the seabed, in order to minimize scour around the structures.

In temperate regions, directly cooled coastal or estuarine power stations discharge cooling water typically between 8–12°C above the ambient water temperature (this *difference* being referred to as the ΔT)[1] into the receiving water body. Within the cooling system, steam from the low-pressure turbines is condensed back to water in the power station condensers, creating a vacuum that not only serves to draw the steam through the system, but also depresses the 'boiling point' of the water: condensation thus occurs at around 30–35°C. As a result, the cooling water at discharge is unlikely to rise much above 30°C.

The configuration of intake and discharge points is designed in relation to local hydrography, in order to minimize recirculation of effluent water before it has had sufficient time to cool.

The effluent, being warmer, is almost invariably less dense than the receiving water (notwithstanding possible salinity differences) and thus tends to rise to the surface, with a persistent interface between these two water bodies maintained by this density difference, although the diffuser system will be designed to aid mixing. Heat is lost from the effluent plume predominantly by radiation to both air and water, but also, and increasingly as it cools, by mixing with the receiving water, this latter effect being of more significance in more turbulent waters. These factors tend to localize the area of potential environmental impact.

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[1] ΔT, excess temperature, temperature uplift, temperature elevation and temperature rise are used interchangeably within this report.

2 A review of existing thermal standards

2.1 Introduction

Temperature regulations covering marine discharges in the UK are not well defined and have been under discussion and development for some time. One problem with any regulatory standard is that it is most consistent and enforceable if applied at 'the end of pipe' where it can readily be measured – that is, as an Emission Standard. An end-of-pipe regulatory standard applied universally across all similar types of plant will be termed a Uniform Emission Standard. This contrasts with what is of most relevance to biota, where the environmental levels in the water column, beyond the so-called mixing zone (MZ), are of key importance. The latter environmental level has long been referred to in the United Kingdom and in EU directives as an Environmental Quality Standard.

A simple definition of a mixing zone, as used by UK regulators, is the area around a discharge within which a regulator permits a quality standard to be exceeded.

The assessment of a discharge is best carried out by a combination of modelling and intensive sampling aimed at, and around, the discharge in the field. A compliance statistic is usually included in the consent (the discharge permission, licence or authorization) requiring a standard to be achieved for a percentage of the time, typically 95% or 98%. Occasional monitoring of spot samples to assess standards in the marine environment is of little practical use. There are so many variables during plume dispersal that it is almost impossible to estimate from where in the plume any sample comes, as the dilution process is far more complex than the simple models suggest. The size of the mixing zone may be subject to local regulation and may or may not be stipulated in the discharge licence. In Scotland, the Scottish Environmental Protection Agency (SEPA) has a guideline that mixing zones should occupy less than 50% of the cross-sectional area of an estuary channel or small embayment (SEPA, 2009) whilst for elsewhere in the United Kingdom a value of 25%, for 95% of the time, has been proposed by Turnpenny and Liney (2007). The regulatory purpose of confining the spread of the mixing zone is to maintain an open corridor for fish migration.

In their latest Better Regulation document (Environment Agency, 2010) the Environment Agency recommends that the thermal discharge should be designed to discharge in sufficient depth of water such that the thermal plume will not 'be attached to the seabed'. Similarly, the European Commission has published non-binding technical guidance on the definition of mixing zones for the WFD (European Commission, 2010b).

Most modern ecologically based regulations, such as the EU Water Framework Directive (Anon., 2000) and EU Habitats Directive (Anon., 1992), require assessments to be principally based on the measurement of the health of the biota directly. In the case of the latter directive the relevant biota are the habitats or species indicated in the conservation objectives for a designated area. It is clear that temperature may also affect chemical reaction rates, which may in turn have direct or indirect effects on biological systems in terms of lethality, breeding rates or growth. This is explored for relevant species in later chapters.

The distribution of an elevated-temperature discharge will follow most of the physical processes of any other buoyant discharge and is affected by the discharge pipe diameter, type of any diffuser system, discharge velocity, depth of the overlying water, relative densities of the discharge and receiving water, existence of any stratification, current speed, surface wave action and bed topography. There is, however, one key difference, in that heat is lost to (or gained from) the system at the interfaces between the discharge and the air or the sediment. Existing mixing zone guidelines are mainly based on conservative chemical dilution but could relatively easily be developed to consider how temperature might be best dissipated. This is explored in more detail in Chapter 3. Any standard laid down aims to provide an environmental level that is suitable for sustaining healthy biota. As discussed above, this usually has to be translated to an end-of-pipe standard that is the basis of design parameters and regulatory assessment.

2.2 Standards in the UK

Most existing standards in the UK are in place to provide protection for fish species, but, in line with modern thinking and regulation, future standards will be introduced that protect all marine and estuarine species, as required by the European WFD and the Habitats Directives. A comprehensive review of standards in the UK, Europe and beyond was carried out by Turnpenny and Liney (2007) for SNIFFER[2]. Although the emphasis was again largely on standards relevant to fish species, this remains a key source document. Regrettably little progress has been made in the United Kingdom since this report was completed.

Temperature standards can be set as either absolute values or as *x* degrees above or below ambient temperature. Because of the biogeographical differences between regions of the United Kingdom, and temperature shifts arising from climate change, specifying a rise above ambient may be the most suitable approach to help maintain a typical local fauna and flora. Absolute limits may also be required to cover extreme events. Although rate of change in temperature is important to fish behaviour, it has never been addressed by regulation. In the past, *ad hoc* standards have been based on the codified Shellfish Waters Directive (Anon., 2006a), where a guideline standard allowed a temperature variation of no more than 2°C, even though TAG (Technical Advisory Group) guidance (UKTAG, 2008b) acknowledges that there is 'a natural establishment and continued development of clam and oyster beds in transitional waters around the outfalls of power stations although temperature increases greater than 2°C are routinely observed'. In addition, standards derived from the Freshwater Fisheries Directive (Anon., 2006b) were used and modified to suit the marine environment. The Shellfish and the Freshwater Fish Directives will both be subsumed by the WFD in 2013, but future standards should offer no less protection than the original directives. Table 1 gives the existing imperative standards listed in the Freshwater Fisheries Directive. It is emphasized that the hydrodynamically simpler conditions in freshwaters and the ease with which river and lake mixing zones can be designed contrast with the greater complexity and variability in tidal estuarine and open coastal systems.

Table 1 Imperative standards for temperature required by the Freshwater Fisheries Directive

Parameter	Salmonids	Cyprinids	Notes
Temperature (°C)	Imperative Value Increment above upstream temperature should not exceed 1.5°C at the edge of the mixing zone Temperature should not exceed 21.5°C at the edge of the mixing zone for more than 98% of the time	Imperative value Increment above upstream temperature should not exceed 3.0°C at the edge of the mixing zone Temperature should not exceed 28.0°C at the edge of the mixing zone for more than 98% of the time	Sudden variations in temperature should be avoided Regional derogations from this standard are permitted. Species that require cold water for reproduction are protected by an upper limit of 10°C during the breeding season

The Habitats Directive has no specific temperature requirements, but requires that European protected habitats and species (as listed in the Directive and as stipulated in the conservation objectives in any adjacent designated area) be maintained or restored with strict protection of species listed on Annex IV of the directive. This Directive only applies if relevant species or habitats are to be found close to any site under investigation and in which case an Appropriate Assessment will be required to address the consequences of any plan or project on the species and/or habitats listed

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[2] Scotland and Northern Ireland Forum For Environmental Research

under the conservation objectives for a site. Hence, if such species or habitats are found nearby, then careful investigation is required to ensure that the tolerances of the species or habitats are well understood. While there is little doubt that investigation is required if the plan or project is within or adjacent to a designated site (a Special Area of Conservation [SAC], a Special Protected Area [SPA] or a European marine site [EMS]), recent European legal challenges have shown that because of the dynamic nature of the marine and estuarine environment and its species, then a plan or project can be away from the designated site and still be likely to have an effect on the relevant habitats and species.

TAG has recommended the interim standards as shown in

Table 2. A mixing zone is only permitted if it can be demonstrated it will not have an adverse effect on site integrity (WQTAG, 2006). However, Environment Agency guidelines suggest that any mean temperature change of more than 0.2°C in the water of the site would be a 'likely significant effect' and therefore would be subject to an Appropriate Assessment (Environment Agency, 2008). Again, the dynamic nature and high variability of estuarine and coastal waters will make it difficult to detect such an average temperature change.

Table 2 Temperature thresholds for assessing the impact of thermal discharges on SAC/SPA sites (if a site is designated as both an SAC and SPA then the most stringent condition applies)

Designation	Deviation from ambient	Maximum temperature
Special Protection Area (SPA)	2°C as a Maximum Allowable Concentration (MAC) at the edge of the mixing zone	28°C as a 98%-ile at the edge of the mixing zone
Special Areas of Conservation (SAC) – any area designated for estuary or embayment habitat and/or salmonid species	2°C as an MAC at the edge of the mixing zone	21.5°C as a 98%-ile at the edge of the mixing zone

A further complication is a requirement on the operator to consider other plans and projects that might have an effect 'in combination'; this is explored in more detail in a recent Department of Energy and Climate Change document (DECC, 2010).

The WFD is the key piece of prevailing regulation as it encompasses all waters within 1 nautical mile of the coast in England, Wales and Northern Ireland (3 nautical miles in Scotland) and all estuaries (included within the term transitional waters). The WFD requires that all transitional waters and coastal waters are at least at good status by 2015. Status is measured by a number of assessment tools associated with biological quality elements, as shown in Table 3.

Table 3 Biological quality elements within the WFD

Transitional waters				
Phytoplankton Composition and abundance Bloom frequency	**Macroalgae** Taxa and cover	**Angiosperms** Taxa and abundance	**Benthic Invertebrates** Diversity, abundance and disturbance-sensitive taxa	**Fish** Species composition and abundance and disturbance-sensitive taxa
Coastal waters				
Phytoplankton Composition and abundance Biomass Bloom frequency	**Macroalgae** Taxa and cover	**Angiosperms** Taxa and abundance	**Benthic Invertebrates** Diversity, abundance and disturbance-sensitive taxa	

Coasts and estuaries are divided up into water bodies of various sizes according to the degree of pressure that the body is under and the physical features that describe it. These water bodies are grouped into types according to common physical attributes both on a national and on a European scale. Biological status has to be assessed against a reference or benchmark condition and translated into an 'Ecological Quality Ratio' which allows the result for the water body to be classified according to a five-point scale. There has been common development of this approach throughout all member states, and the assessment tool has to be subject to an intercalibration exercise to ensure the classification has a degree of consistency throughout all member states of Europe (Hering *et al.*, 2010). Temperature is not directly attributed to a standard but is referred to in the so-called normative definitions for physico-chemical status shown in Table 4, as extracted text from the WFD.

Table 4 Selected normative definitions of physico-chemical criteria

High Status – Levels of salinity, pH, oxygen balance, acid neutralizing capacity and temperature do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions
Good Status – Temperature, oxygen balance, pH, acid neutralizing capacity and salinity do not reach levels outside the range established so as to ensure the functioning of the type specific ecosystem and the achievement of the values specified above for the biological quality elements
Moderate Status – Conditions consistent with the achievement of the values specified above for the biological quality elements

Despite the lack of a specific temperature standard it is clear that thermal changes could have a direct or indirect effect on virtually all of the biological quality elements, many of the chemical elements, and even some of the physico-chemical elements as well. However, again it is emphasized that there are large difficulties in attempting to define rigorously such physico-chemical parameters in highly dynamic and variable estuarine and coastal waters.

For cold and warm waters, the UK Technical Advisory Group for the WFD (UKTAG, 2008b) have produced some draft standards for rivers and suggest they be used as interim action levels for transitional and coastal waters until more specifically derived thresholds can be produced. Table 5 lists the maximum temperature that would be allowed as an annual 98th percentile at the edge of the mixing zone.

Table 5 Maximum temperature allowed at the edge of the mixing zone

Temperature (°C) annual 98%ile				
	High	Good	Moderate	Poor
Cold Water (Salmonids)	20	23	28	30
Warm Water (Cyprinids)	25	28	30	32

Note: outside the mixing zone a maximum ΔT of +3°C is allowed (+2°C where waters are of high status).

Chemical standards for the WFD, many of which are still under development, are a combination of new and existing standards, some nationally derived whilst others must be common to all states. Current standards are listed in TAG guidance documents.

New or existing standards for 'specific pollutants' generally are derived from annual means, while 'priority' and 'priority hazard' substances use maximum concentrations derived from an appropriate predicted no-effect concentration (PNEC). The TAG guidance documents (UKTAG, 2008a, 2008b, 2008c) give comprehensive lists of regulated substances, and some important and relevant contaminants are discussed in Chapter 4. Temperature can have an effect on a chemical contaminant's solubility, oxidation or reduction rate, or the interaction with biological tissue. If samples fail a chemical standard under the WFD the water body cannot be classified as Good or High. Table 6 andTable 7 illustrate a few selected chemical and physico-chemical standards developed for the WFD.

Table 6 Standards for dissolved oxygen and inorganic dissolved nitrogen (UKTAG, 2008a, 2008b)

Marine and transitional* dissolved oxygen levels must exceed 95% of the time	Dissolved inorganic nitrogen – winter mean
High 5.7 mg l^{-1}**	High 12 µmol l^{-1} for coastal and 20 µmol l^{-1} for transitional [#]
Good 4.0–5.7 mg l^{-1}**	Good 18 µmol l^{-1} for coastal and 30 µmol l^{-1} for transitional [#]
Moderate 2.4–4,0 mg l^{-1}	Moderate 27 µmol l^{-1} for coastal and 45 µmol l^{-1} for transitional [#]
Poor 1.6–2.4 mg l^{-1}	40.5 µmol l^{-1} for coastal and 67.5 µmol l^{-1} for transitional [#]
Bad 2.0–1.6 mg l^{-1}	

*A small shift in these values is allowed owing to reduced solubility of oxygen in more saline waters (see TAG Guidance (UKTAG, 2008a, 2008b)).

**2 mg l^{-1} minimum for intermittent discharges with a return period of 1 in 6 years for Good and 1in 3 years for Moderate status.

[#]Assuming coastal waters at salinity of 32 and transitional at 25. (Salinity is reported using the Practical Salinity Scale and thus has no dimensions or units). If transitional waters are turbid, a supplementary figure may be made (refer to the TAG guidance).

Table 7 Other selected Annex VIII chemical standards (UKTAG, 2008c)

Parameter	Qualifier	Limit value
Un-ionized ammonia µg l^{-1}	Annual mean	21
Chlorine µg l^{-1}	95%-ile	10 (total residual oxidant)
Zinc µg l^{-1}	Annual mean	40 dissolved
Copper µg l^{-1}	Annual mean	5 dissolved

2.3 European and international standards

As has been discussed, the key directives have not required a trans-European thermal standard, so it is of value to examine what individual countries inside the EU and beyond have developed for their saline waters. Turnpenny and Liney (2007) investigated the approach taken by a number of countries, and their findings are summarized in Table 8.

Table 8 Summary of standards for marine thermal discharges in other countries

Country	Marine surface water regulations in existence?	Regulation
United States	X	—
Canada	✓	Allow 1°C change from ambient
South Africa	✓	Allow 1°C change from ambient
New Zealand	X	—
Netherlands	✓	Allow 2°C change from ambient, with a 25°C maximum
Germany	X	—
Denmark	✓	Allow 1°C change from ambient at the edge of the mixing zone
Belgium	✓	Unknown
Finland	X	—
Sweden	X	—

A short informal survey in January 2011 of WFD contacts produced the following summaries:

➢ In France, on the Gironde Estuary, the difference in the intake and discharge cooling-water must be no greater than 11°C, with a maximum temperature of 30°C for the discharge (there is seasonal variation allowed, so in the summer it may be permitted to reach 36.5°C) and the standards vary for each power plant in France.

- Although Norway has no nuclear power stations they have standards for other thermal discharges, namely the ΔT at the point of discharge of 10°C and a ΔT outside the mixing zone of no more than +1°C. (Some flexibility appears to allow up to 3°C at particular sites).

- Sweden still does not seem to have a standard.

- In the Basque Country (Spain), 50 m or more from the discharge point in any 1 m layer within the water column the ΔT must not exceed +3°C or +1°C integrated throughout the water column.

- In Germany, cooling water discharges must be at no more than 30°C and the ΔT no more than +10°C for an existing site, and +7°C (and 60% DO saturation) for a new development.

- In the Netherlands, the Rijkswaterstaat (Bruijs *et al.*, 2011) regulates CW discharges into marine waters to ensure that they do not exceed 30°C within a mixing zone bounded by the 25°C isotherm and that they do not occupy more than 25% of any narrow channel to allow passage of migratory fish. For shellfish waters, the maximum temperature is reduced to a maximum of 25°C with a ΔT of 2°C. Some short-term relaxation is allowed for no more than a week during exceptionally hot periods, if they should occur, in July or August only.

It is of course critical how and where these standards are measured and it is difficult to compare them without full contextual information. In particular, the site-specific considerations have to include the complex nature of the British coastline and the seasonal temperature differences experienced between different parts of the coastline, including the greater warming influence of the North Atlantic Drift on the south-west coast and the poor influence on the North Sea coast.

2.3.1   Developing regulation

Although the Marine Strategy Framework Directive (MSFD) (Anon., 2008) is at an early stage of development, and the exact definitions of Good Environmental Status for the 11 descriptors are only recently being proposed, the MSFD will not overlap with the WFD for the geographical area already covered by the framework directive (eg, it will not be enacted for benthic invertebrates within 1 nm of the national baseline). There are biota, such as cetaceans and birds, that are not covered by the WFD and potentially will be covered in the MSFD, so these may have to be considered in any future review of temperature effects even if the potential risk is low. Although this directive again includes no particular reference to temperature, it does refer to inputs of energy as needing to be controlled or removed (European Commission, 2010a). However, the directive concentrates on the ecological populations and communities and especially any deviations to their use of habitats, and it also, indirectly, considers adverse changes to habitats – each of these can be affected by thermal discharges. Generally this directive is more comprehensive in coverage than WFD, and it relates more to the ecological functioning of systems but lacks specific detail leaving it for national interpretation (Borja *et al.*, 2010). Hence coastal and marine industries will have to be prepared to consider the influence of their activities on the topics covered by the 11 descriptors of the MSFD.

2.4   Conclusion and summary

Despite the considerable amount of work that has been undertaken on chemical standards to protect all European marine biota, this is not reflected by comparable effort in developing thermal standards. Most of the temperature standards that have been produced are based on fish data, which is understandable because of considerable commercial and leisure interests in fisheries. While it would be difficult to claim that the temperature tolerances of all biota were known and documented, there is considerable data available that would allow provisional standards to be agreed. Any research project aiming to develop this approach must ensure that it covers the biogeography of all of the United Kingdom to allow for natural climatic and oceanographic differences. Ideally some flexibility should also be built in to allow for climate change and for the site-specific nature of different parts of the UK coast. It is not only the temperature of the receiving water that will change with any natural rise, but

the thermal discharge is likely to rise by an equivalent amount. Taking into account the current measured rate of change, seawater temperature rises in excess of 1°C can be expected over the life of any new station built in the next few years. There is a clear need to develop more comprehensive and justifiable critical levels for a wider biota in line with modern regulation.

3 Mixing zones and plume behaviour

3.1 Introduction

Power stations are often coastally situated in order that the thermal cooling can utilize the large volume of seawater, hence the most vulnerable area to the excess temperature is the inter-tidal and shallow water seabed. Similarly, estuarine and other locations with restricted water exchange will be the most vulnerable. By definition, estuaries contain variable proportions of riverine freshwater and hence are subject to large daily, monthly and seasonal temperature changes, given the relative differences in temperature between the freshwater and seawater entering the estuary. Additionally, they can be prone to extensive periods of constant high or low temperature and low salinity during river floods, depending on the season. Therefore, any biological impact will be dependent on a combination of salinity and temperature conditions, which can be described by hydroclimagraphs for any particular area (Elliott and McLusky, 2002). Such patterns show the influence of colder river water entering the estuary during the winter and warmer land runoff in the summer, an influence exacerbated in shallow estuarine areas. Coastal locations may not be affected by such large salinity and temperature variations, but will be more prone to the effects of weather, wind and waves. It is essential to define the ambient temperature absolute levels and variability at the intake onto which any discharges will be superimposed.

3.2 Background temperature field

The magnitude and distribution of the background temperature is important in that it provides a baseline of variation to which the biology is essentially adapted. It is also the baseline on which are superimposed the influence of any discharges. The background is responsive to the changing magnitudes of the thermal input from the sun and temperature retention of water bodies and thus reflects the effects of daily, monthly and seasonal cycles, together with the effects of weather and any stratification. The annual mean value for UK waters, which is likely to be in the range 5–15°C, will therefore be the averaged sum of these effects and will have large variations about the mean.

When assessing the impact of CW discharges, it is important to understand the wide natural variability in seawater temperature which occurs independently of any anthropogenic influences. The best long-term record of coastal water temperature in the British Isles is that for Port Erin on the southwest coast of the Isle of Man, where measurements have been made continuously at the same location for well over 100 years. With a small population and no significant industry, the Isle of Man data provide a good indication of natural variability and trends on an inter-annual and intra-annual basis.

Data for the past 60 years (1949–2008) have been selected to match the maximum operating period of the proposed NNB (Figure 1 &Figure 2) (the original data are held by the British Oceanographic Data Centre, Liverpool). This illustrates the natural variability in waters reasonably remote from the influence of shallow water and anthropogenic inputs.
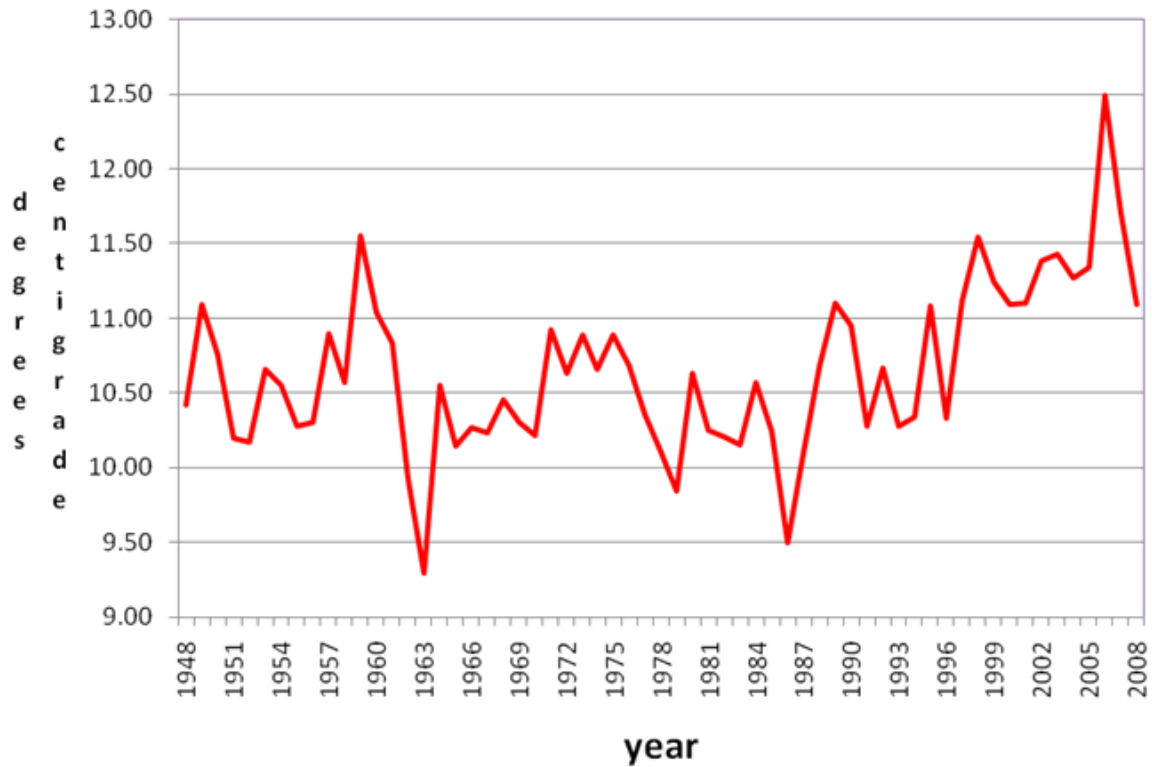
Figure 1 Annual mean seawater temperature at Port Erin (1949–2008)
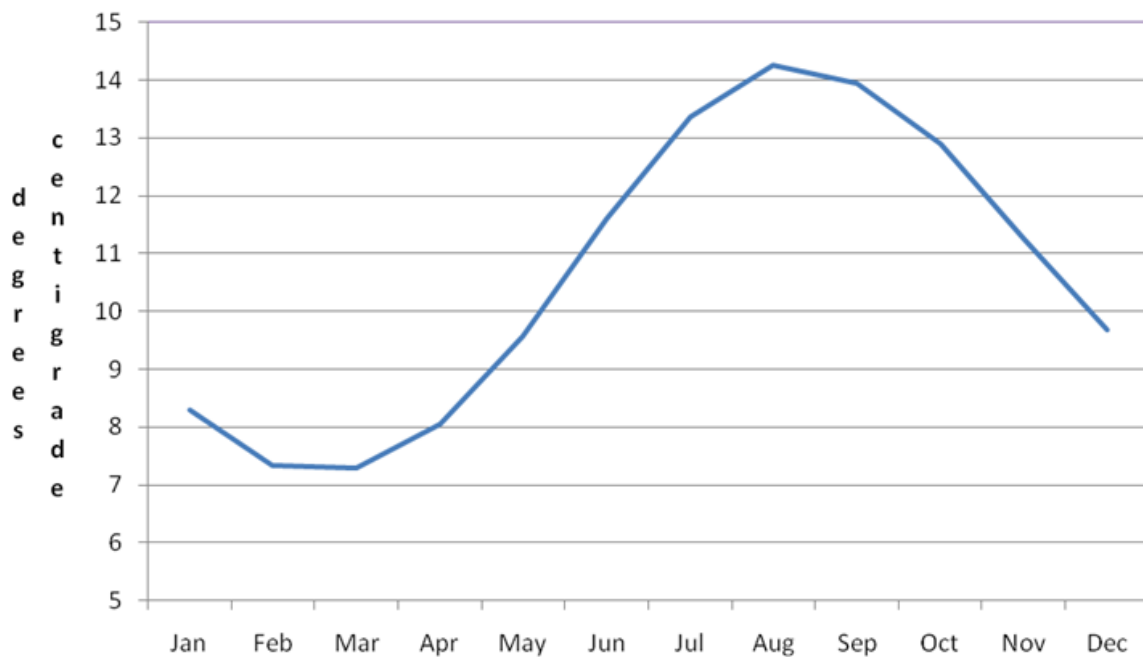
Figure 2 Monthly mean seawater temperature at Port Erin (1949–2008)

Diurnal/tidal effects. The difference between afternoon and night-time temperatures at the water surface overlying deep water will normally be within the range of 1–2°C, and normally within 1°C at the bed. However, the temporal variation may be modified by tidal water movement that can carry a regional temperature gradient across the point of interest. In shallower water, tidal currents and waves tend to maintain vertical homogeneity. The diurnal difference increases in shallow water until in

intertidal areas it can reach 10–15°C, because of the heating of shallow pools and the underlying mud when exposed, and rapid cooling at night. The timing of the tide and of insolation is therefore of significance and will vary with the lunar tidal phase. These features can create lateral differences across estuaries in moving from intertidal to shallow to deeper areas.

Monthly effects. Within the lunar month there can be spring-to-neap variations in temperature, largely because of the residual movement of water during the lunar cycle. These can be in the range 2–5°C. Weather will also be important in modifying the vertical differences in temperature, and there will be spatial gradients giving intermittent variations.

Seasonal cycles. The summer-to-winter temperature differences in surface water temperatures are likely to be in the range 5–20°C, depending on water depth. In areas of high tidal currents the surface to bed variations will be similar, but in deeper water a seasonal thermal stratification may develop in summer. This can be predicted in terms of critical values of the parameter $h \cdot u^{-3}$ where h is the water depth, and u the amplitude of the depth mean current. The temperature variation at the seabed is therefore less than that at the surface. In the case of the North Sea, the influence of this feature on the benthic biology was described by Glémarec (1973) who contrasted areas in terms of the thermal stability in which deeper areas were stratified seasonally whereas shallow areas were well-mixed in summer. Areas influenced by a river discharge are also subject to seasonal effects related to the temperature of the river water and runoff from the land. The difference in temperature between river and seawater will be minimal in the spring and the autumn, with seawater cooler than river water in the summer and warmer in the winter (see Figure 2). These features are shown in more detail by the hydroclimagraphs for the Forth Estuary (Figure 3), which shows the typical wedge-shaped pattern for an estuary with a more restricted temperature variation for the seaward part and a wider variation for the freshwater area (see McLusky and Elliott, 2004). Although such hydroclimagraphs have not been constructed for the Severn Estuary, it is expected that they will follow a similar shape.
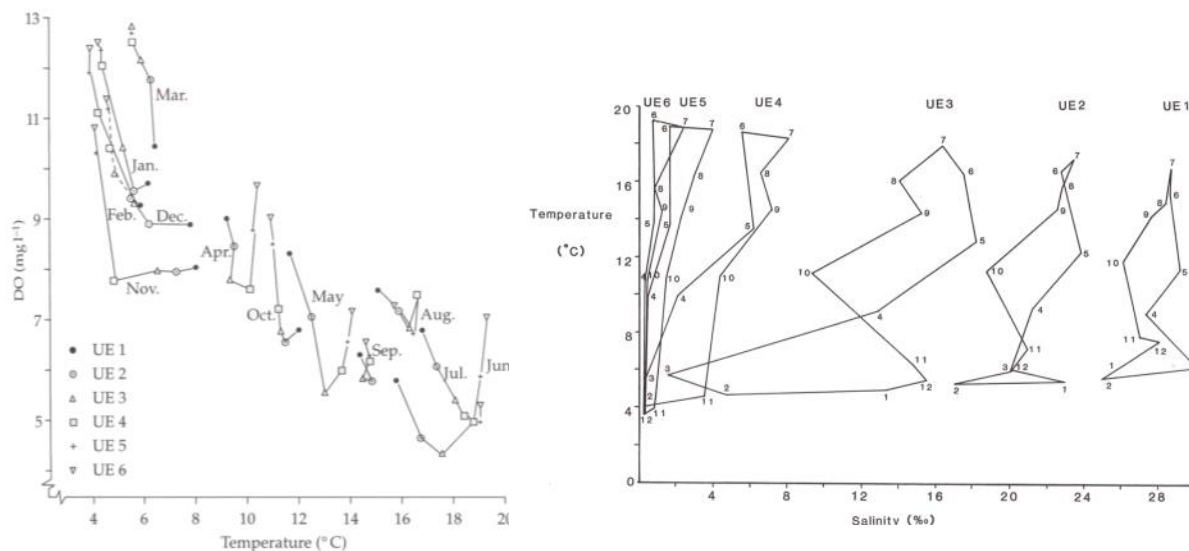
Figure 3 Hydroclimagraphs for the Forth Estuary; DO and temperature profiles (left) and temperature and salinity profiles (right)

Inter-annual variation. There is likely to be variation from year to year in temperature owing to changes in the weather patterns. This will normally be of a few degrees, but will be most apparent in comparisons of the maximum and minimum temperatures.

Climate change. Global climate model projections, although undergoing continuous improvements, remain too uncertain over a region such as Europe to draw more than very tentative conclusions. Nested regional climate model simulations also are too tenuous to provide firm conclusions about climatic change in Europe. Temperature changes in large parts of Europe will be a function not only of positive radiative forcing resulting from increases in the atmospheric concentrations of a number of greenhouse gases, but also of the counter-effects of aerosols; although the magnitude of the aerosol

effects is still quite uncertain, the regional negative forcing of sulphate aerosols in central Europe could offset almost half the positive radiative forcing of CO_2. Pachauri and Reisinger (2007), addressing six possible emissions scenarios, projected global average surface warming to range between 1.8°C and 4°C (ΔT) for 2099 compared with 1980–99, and projected a warming of 'about 0.2°C' per decade for the next two decades (see also Solomon *et al.*, 2007). The shelf seas around the United Kingdom are projected to be 1.5–4°C warmer by the end of the 21st century, with predicted warming particularly marked in the southern North Sea in autumn (UKCP09, 2011).

Relationship with salinity. Both temperature and salinity determine the water density, with a difference in salinity of about 2 having the same effect as a temperature difference of 5°C. As far as the physics of the water movement is concerned, it is the water density that is the crucial factor, again indicating the importance of defining the hydroclimagraphs for any area receiving heated power plant effluent. This produces considerable challenges in modelling the water flow and the temperature distribution resulting from thermal discharges, because of this 'feedback'. It will be most significant in estuarine areas, or in transitional waters experiencing discharge of heated lower salinity water. Density stratification may have the effect of producing a residual landwards water flow that will introduce cooler water near the bed. Thus it is also important to know the salinity field in the area.

3.3   Examples of monthly and inter-annual temperature

3.3.1   Hinkley Point power stations

Hinkley Point is located in the outer reaches of the Severn Estuary and is subject to a very high tidal range and consequently also high tidal currents. In a British Isles context, it receives a greater warming influence of the North Atlantic Drift, the remnants of the Gulf Stream, than the North Sea areas. There are extensive inter-tidal areas, and the mouth of the River Parrett is situated within a tidal trajectory of the outfall. Modelling and observation of the plume from Hinkley A and B power stations has shown that at high tide it is possible for the thermal plume to reach into the Parrett and, conversely, the Parrett discharge can reach the outfall during high river flows. Outfalls located further offshore would decrease the risk of this interaction.

Observations (Figure 4) taken near the intake location, show the characteristics of the temperature variation given in Table 9. Particularly noticeable are years of relatively low temperature range in 1988, and high range in 1995.
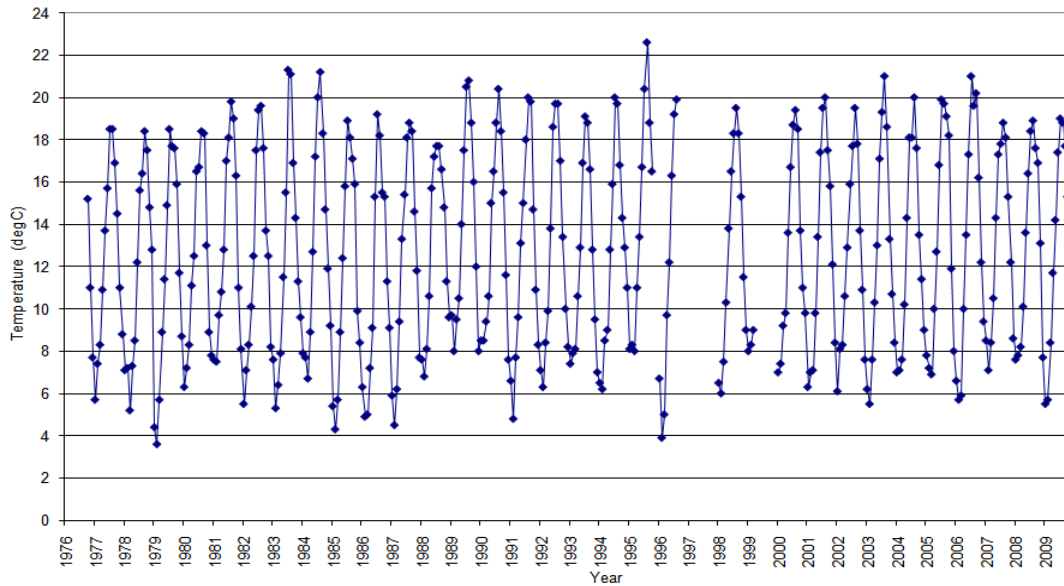
Figure 4 Monthly mean surface temperatures at Hinkley Point (data provided by British Energy Generation (UK) Ltd., BNFL, CEGB)

Table 9 Temperature variation (°C) from monthly mean values for Hinkley Point and Sizewell, with years of occurrence in brackets

	Hinkley Point	**Sizewell**
Min. annual mean	11.4 (1986)	10.1 (1967)
Max. annual mean	13.8 (1989)	14.4 (2006)
Max. monthly mean	22.6 (1985)	22.5 (2003)
Min. monthly mean	3.6 (1979)	2.8 (1985, 1986)
Interannual variation of monthly mean	~±1.2	~±1.2
Tidal/diurnal variation	~±1.0	~±1.0

3.3.2   Sizewell Power Station

Sizewell Power Station is on a coastal site exposed to waves from the north-east and south-east, but also being in the southern North Sea, it receives less influence from the North Atlantic Drift than the south-west of the British Isles. Locally, both intake and outfall locations are constrained by extensive offshore banks that can limit both the supply of cooling water and the dispersion of effluent. The most vulnerable areas for the impact of elevated temperature are the offshore banks.

Observations (Figure 5) taken in the vicinity of the intake location show the characteristics of the temperature variation given in Table 9. There has been a significant increase in annual mean temperature over the 43 years of the records, averaging 0.0825°C per year.
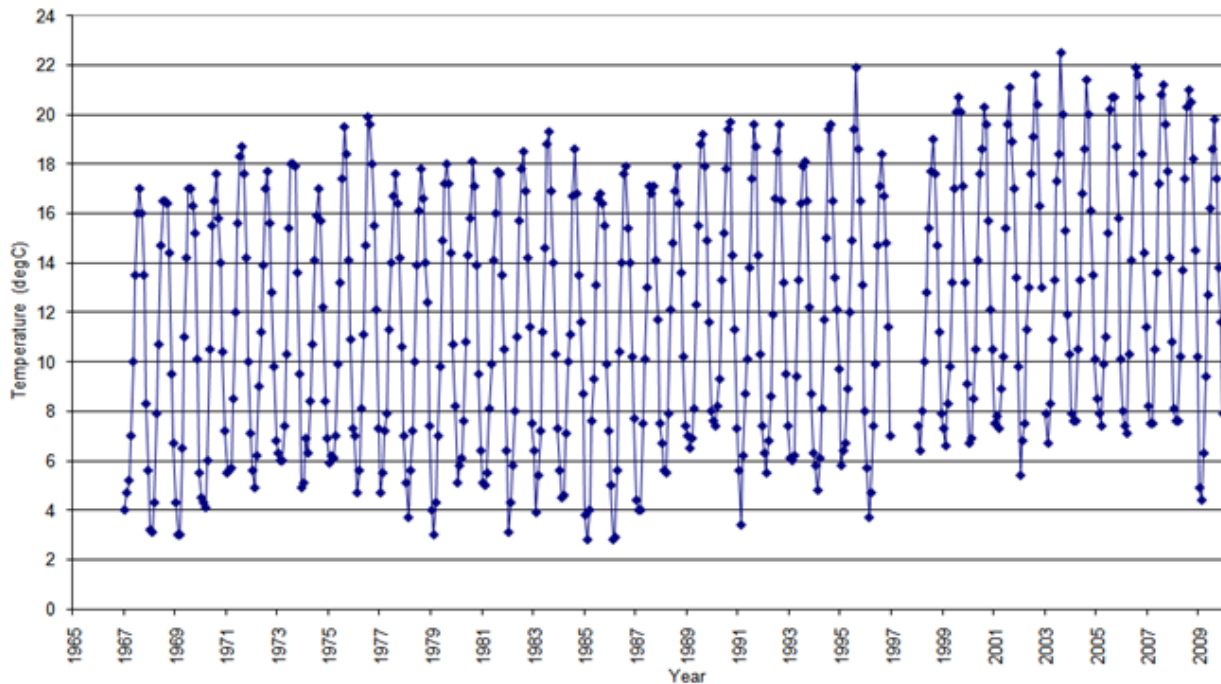
Figure 5 Monthly mean surface temperatures at Sizewell

3.4   Thermal plume behaviour

A plume is an active mixing zone in the vicinity of a power station where the properties of the receiving water are altered by the discharge from the power station outfall. It is normally recognized as a zone where temperature is higher than the ambient water. The classical descriptions of plume behaviour are normally derived from laboratory experiments into still water. These consider three zones within the plume: the near-field, the middle-field and the far-field (also known as the 'long-term field').

> *Near-field.* In the near-field the motion is dominated by the upward momentum induced by the discharge and the buoyancy resulting from the higher temperature, lower density fluids. Its characteristics can be influenced markedly by the design of any diffuser system – for example, a set of risers and diffuser nozzles that can spray the effluent into the water column and thus aid dispersion and dilution. The rising water has jet-like properties, and, if it rises to the surface, it can form a distinct elevation of the sea surface, a 'boil'. Mixing and reduction in temperature is caused by the entrainment of ambient cooler water into the plume. The plume therefore gains volume with distance, but maintains constant momentum. In deep water the plume will raise clear of the seabed and, if the water is stratified, is likely to stabilize at the level of neutral density, although, because of the upward initial forces, it can overshoot in the near-field. Generally the density difference is sufficient to take the plume to the surface.

> *Mid-field.* In the mid-field there is a transition to spreading; the vertical motion becomes horizontal, driven by the surface water slope produced by the boil, and the horizontal density gradient. The mixing is still mainly by entrainment because of shear at the density interface. The temperature distribution across the plume is likely to be Gaussian with distance from the centre, after averaging the turbulent fluctuations.

> *Far-field.* In the far-field, spreading and advection dominate, and the thickness of the plume will diminish significantly as it spreads. Mixing will occur due to shear at the density interface, with wind waves providing a variable additional input. Nevertheless, the distribution of temperature across the plume remains essentially Gaussian, when the turbulent fluctuations

are averaged. The reduction in temperature is now dominated by losses of heat to the atmosphere, and the wind becomes particularly important in controlling the rates of loss.

3.5   Plume behaviour in tidal flow

Continuous discharge into a tidal flow leads to more complex movements because of the changing rate and direction of movement of the ambient fluid.

> ➢ *Near-field.* Initially the plume is injected vertically into or across the flow and has no horizontal component of motion in the direction of the tidal current. In the near-field the plume will be subjected to fairly intense mixing because of entrainment into the rising plume, but the vertical velocities rapidly decrease and gradual acceleration occurs in the longitudinal sense as the discharge becomes influenced by the regional tidal pressure gradient.

> ➢ *Mid-field.* In the mid-field the vertical momentum will have been dissipated, but velocity gradients beneath the plume and at its edges will be strong, as the plume still travels relatively slowly within the ambient fluid. These gradients will induce continued acceleration of the plume, and entrainment of water from within it into the more turbulent ambient fluid, causing the plume to lose volume, and heat. The ambient fluid is likely to be more turbulent because of shear at the seabed.

> ➢ *Far-field.* In the far-field the plume is essentially integral in the flow and will respond to the tidal driving forces. The velocity differences between the plume and the ambient fluid decrease to the point that mixing is limited by the density gradient. Continued spreading of the plume involves a gradual decrease in thickness, which is generally of the order of a metre, and increasing importance of wave-induced mixing. Additionally, whenever there are spatial variations in bed shear, due to topographic irregularities, the plume can break up into filaments and eddies travelling at different rates.

As the tidal velocity varies, the extent and intensity of the mixing will vary drastically, as will the decrease of temperature. It will be noticed from the above that in the mid-field the sense of exchange of water during entrainment mixing will vary with the relative intensities of turbulence in the plume and in the ambient fluid. When the plume is relatively fast flowing, the entrainment is of ambient fluid into the plume, whereas when the tidal velocities dominate, plume water is entrained into the ambient flow.

As the tidal velocities decrease towards slack water, the magnitude of the near- and mid-field mixing will diminish and their relative extents alter. At slack water as the tide turns, there will be minimal mixing, and a large patch of low-density near-field water can form. Spreading will produce some entrainment into the patch, which will be moved away on the next phase of the tide, to mix further as the ambient fluid becomes more turbulent. When the discharge rate is high, the thermal timescale may be sufficiently long for the plume formed on one phase of the tide to persist and interact with that formed on the next; the water returns over the intake position and recirculation may occur. Further complexity arises when residual flows, storms and surges cause the water not to return over the outfall point and the plume influences a wider area. If the discharge of heated water is across the intertidal at low water, there is heat loss to the atmosphere, but no dilution occurs until the ambient water is reached.

Whether the water velocities in the plume are greater than those in the ambient fluid or not depends upon the relative strength of the discharge and the tidal flow. It is obvious that, in the short term, movement of the plume and consequent decreases in temperature are a complex interaction of the geometry of the outfall, the velocity and density of the discharge, the water depth and its spatial variation, the tidal velocities, seabed-induced turbulence, density of the receiving waters, waves, winds and atmospheric temperature. There are no theoretical methods for predicting the plume form and movement, apart from mathematical modelling of specific sets of conditions.

3.6   Observation of thermal plumes

Thermal plumes may be observed by measurement at sensors at fixed locations and by ship-borne techniques. The surface plume location can also be observed by remote sensing from aircraft or satellite-mounted sensors. Each technique has advantages and drawbacks.

Fixed sensors. Continuous fixed measurements have the advantage that the results can be averaged over discrete periods and the statistics of the durations and magnitudes of the temperature elevations determined. In particular, sensors can be located close to the seabed to monitor the temperatures relevant to benthic organisms. The main drawback is that only a few positions can be occupied, and for much of the time, especially in tidal flows, the sensors will not be within the plume. Accuracies of the measurements can be 0.1°C and resolutions 0.01°C.

Shipborne sensors. Sensors can be mounted or towed at several depths below the surface to provide profiles of temperature, and surveys can be undertaken of the horizontal distribution through the plume. Accuracies and resolutions are similar to those of fixed sensors. This technique is useful in following the plume for short periods, but suffers from the time taken to carry out the surveys. Thus, as the measurements are not simultaneous, contours cannot be accurately derived, and a truly instantaneous picture of the plume is not possible. Determination of the duration and magnitudes of the maximum temperatures at different locations are also not possible. Additionally, surveys cannot be carried out during extreme weather conditions. A combination of fixed sensors and shipborne surveys can be useful.

Remote sensing. The use of thermal band sensors from aircraft has the advantage of giving virtually simultaneous images of the complete plume extent at frequent intervals for large parts of individual tides. Ground-truth data are necessary for calibration of the sensors, and flights are often restricted by weather. Accuracies of about 0.1°C are achievable with resolutions of about 0.03°C. However, the results only apply to a thin surface layer of the water, normally only a few centimetres in thickness. The results are also averaged over an area of the surface, a pixel, generally a few metres square, determined by the sensor characteristics and the aircraft height. Satellite remote sensing consequently has a larger pixel size, generally several tens or hundreds of metres square, and the disadvantage of overpasses only every few days. However, despite there being needs for calibration, and being affected by cloud, clear overpasses can give useful semi-quantitative results. Accuracies of about 1.0°C are possible, with resolutions of about 0.5°C. The surface images though difficult to calibrate can be extremely revealing of the detailed extent and the interaction of the plume with topography. They are of especial use in helping to validate the results of hydrodynamic plume models.

Figure 6 presents an infra-red image showing the thermal plumes from the Sizewell A and B power stations on the Suffolk coastline. Note the two outfalls.

Figure 6 Outfall plume of Sizewell A&B Power Station showing flood tide (7:24 GMT and 9:20 GMT); slack water (11:21 GMT); and ebb tide (15:22 GMT) (source: Environment Agency)

3.7 Mathematical modelling

The application of mathematical hydrodynamic models provides an essential means of calculating the temperature distribution of a plume throughout tidal cycles for different weather conditions. There are many models of varying complexity and potential utility. Before applying the results, it is necessary to have a clear assessment of the limitations and sensitivities of the model outputs. With adequate calibration and validation, and testing of the sensitivities of the model outputs to input boundary conditions, and exchange coefficient parameterizations and magnitudes, estimates of the plume characteristics are an essential tool. Nevertheless, accuracies of the results are difficult to assess, and they have to be interpreted with care. It is advisable to use several 3D models to overcome the intrinsic differences of scale and parameterization of variables (ie, an 'ensemble approach').

Hydrodynamic models. Modelling of plumes has to be carried out based on 3D hydrodynamic models. The models should include tidal water movement, surges, wind and waves, and density stratification, together with reasonable mixing coefficients. As all of these processes act together, omitting any one will cause the results to be less applicable. The mixing coefficients need to be adequately validated against data. Good output results are generally considered achieved when elevation, current velocities and phases are within 10º of those observed, although this is often more difficult with phases. The grid size of the modelled area will generally be at least 50 m square.

Plume Models. To model a plume, the hydrodynamic model additionally needs specification of the locations of the intake and outfall structures, and the magnitudes and temperatures of the flows. Of particular importance is specification of the vertical level at which the intake cooling water is obtained in relation to the ambient thermal structure and how it may change over the tide. Additionally, specification of the outfall shape and associated form of the flow is important in the near-field. If the discharge occurs across the intertidal, the flow characteristics associated with dilution and cooling will vary with the tidal stage, as well as being very sensitive to waves. Coefficients of thermal exchange with the atmosphere are normally related to wind speed and atmospheric conditions, and those between the plume and the ambient fluid are specified in terms of turbulent-mixing criteria. There are few plume models that include all of the important processes; waves and winds are often omitted (and are poorly modelled by climatologists). Some models simulate the excess temperature produced by the discharge rather than the actual temperature.

The model outputs can be compared with observed measurements to provide thermal calibration, and coefficients and other boundary conditions altered to provide improved agreement. The tidal movement of the plume and its seasonal and weather-related variations can then be simulated. In estuaries the stratification and the mixing regime may alter significantly with tidal state and may interact with changes in water depth.

The value and importance of the results centres on the fact that various scenarios of tide, winds, waves and atmospheric conditions can be run for different discharge locations and magnitudes, and the plume outline and temperature changes at various locations can be examined. However, they should be treated as being indicative rather than offering true prediction, as the errors are uncertain. Particularly away from the core of the plume, temperature values and durations may be less accurate. Often the surface contours show intriguing patchy or necklace-like features, but it is sometimes unclear whether they are real or not.

3.8   Modelled thermal distributions

There are, of course, many distributions of the temperature within the mixing zone of the effluent plume that can be displayed, depending on the state of the tide, tidal range, wind force and direction, waves, and atmospheric conditions (temperature and humidity). Examples of typical results are shown in Figure 7 for Sizewell.
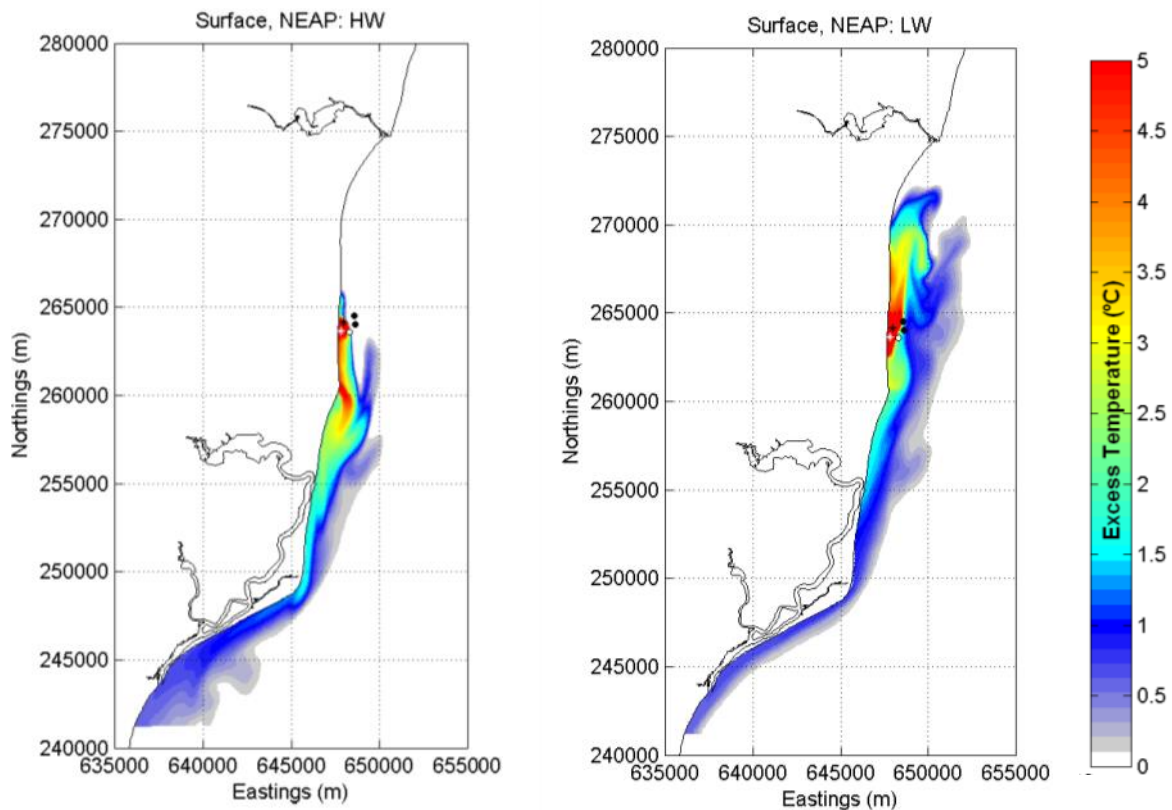
Figure 7 Modelled plume of Sizewell at both high and low water (BEEMS Technical Report No. 133)

3.9   Determination of critical temperature values

In physical terms the mixing zone is one where the plume is involved in mixing with the ambient seawater and where the temperature is constantly decreasing. Thus during a tide, and with different wind and weather conditions, the mixing zone would be carried over an envelope of the seabed (a zone of influence) at which the temperature would be at times elevated for different magnitudes and durations. The probability of a location being affected by a particular temperature rise will decrease with distance from the outfall and away from the direction of the tidal stream. Additionally, the temperature variation on the water surface would be different from that at the same location on the seabed.

As indicated previously, for regulation purposes, the mixing zone is defined by UK regulators as follows: 'The mixing zone represents the area (volume) of water adjacent to the discharge in which the EQS [Ecological Quality Standard] or other control parameter may be exceeded'. Consequently the mixing zone will encompass an area in which the temperature will vary considerably in magnitude and duration. Appropriate values for this definition need to be derived from observations or the output of plume modelling. Ideally a statistical approach to specification of temperature exceedence values needs to be used. This depends upon, for instance, the measured probabilities of tidal range, and surges, and wind and wave roses to determine frequencies of occurrence and probable duration. Worst-case scenarios, producing the highest excess temperatures, should be chosen for further exploration, although there may not always be sufficient information to carry this out. Alternatively, the observed and modelled temperature values can be displayed in a Geographical Information System (GIS) database as a series of maps. These can separately cover maximum values during spring and neap tides, seasonal effects, extreme weather conditions and other times of interest. The maps can be displayed as overlays to illustrate the temperature variation at different locations, and spatial and temporal interpolation effected. Similarly, maps of the variation of other significant parameters, such as salinity or turbidity, can be displayed. Comparison can then be made with the known sensitivity of organisms at the various locations to excess temperature and other factors.

4 Interactions with water quality

4.1 Introduction

Although the primary purpose of this report is to investigate the *direct* influence of temperature on the behaviour and survival of key TraC fish and other organisms, it is important to understand the effect of increased water temperature on general water quality. Increased temperatures will affect water chemistry in several ways:

➢ physical parameters – eg, reduced solubility and supersaturation of dissolved oxygen and nitrogen;

➢ altered kinetics – eg, plankton growth and microbiological activity;

➢ changes to speciation – eg, oxidation state and partition of metals between sediment and water.

Hence there is the need to consider any synergistic (additive) and antagonistic (cancelling) effects of elevated temperatures on other water quality determinands and on the biological responses to these. If increased water temperatures degrade water quality, this may act as an additional stressor and needs to be considered if the effect of temperature on fish and other biota is to be fully assessed.

Another consideration is the risk that a temperature rise will, *in isolation*, cause a water quality standard to be exceeded. A simple example of this is ammonia, where a rise in water temperature will alter the ionized vs. un-ionized equilibrium, leading to a commensurate rise in un-ionized ammonia possibly above consented values. In other cases a more complex set of interactions may lead to increases in concentration of a given parameter – for example, temperature may enhance productivity, indirectly leading to pH changes, and this combination of factors could change the speciation of (toxic) metals. Potentially this is a very large subject, and it is neither practical nor appropriate within this document to explore in full the subject of thermal change on water chemistry. This chapter will identify the issues that are likely to be of importance in the context of new power station operation. Six main issues are considered:

➢ dissolved oxygen and nitrogen;

➢ primary productivity;

➢ ammonia;

➢ total residual oxidant and chlorination by-products;

➢ hydrazine;

➢ metal speciation.

4.2 Dissolved oxygen

4.2.1 Influence of temperature on the solubility of oxygen

The solubility of oxygen in water is determined by temperature, salinity, atmospheric pressure and depth. For UK coastal power stations, where water depths are modest (5–20 m), the pressure terms for depth and atmosphere can reasonably be ignored. Increasing temperature reduces the solubility of oxygen in water but increases the degree of saturation (Table 10).

Table 10 Impact of temperature on the solubility of oxygen

Temp., ^{0}C	8	10	12	14	16	18	20	22	24	26
Solubility (mg l^{-1})	9.75	9.32	8.92	8.56	8.21	7.90	7.60	7.33	7.07	6.83

Oxygen solubility also reduces with increasing salinity. In high salinity waters, although the solubility is reduced, the impact of temperature is smaller (Table 11).

Table 11 Impact of salinity and temperature on the solubility of oxygen

Salinity	Temperature, °C	Oxygen solubility, mg l^{-1}	% reduction
5	15	9.77	
	22	8.48	13.2
15	15	9.19	
	22	8.00	12.9
30	15	8.38	
	22	7.33	12.5

The solubility of oxygen in marine waters is calculated using the algorithm published by UNESCO (Weiss, 1981). For shallow waters (ignoring the pressure term) this is available as a simple Excel spread sheet via the BEEMS Data Centre[3]. For a given level of dissolved oxygen, increasing the temperature will increase the degree of saturation, as shown in Table 12.

Table 12 Impact of increasing temperature on seawater at salinity 30 and 7 mg l^{-1} dissolved oxygen

Temp., °C	8	10	12	14	16	18	20	22	24	26
% saturation	71.8	75.1	78.5	81.8	85.2	88.6	92.1	95.5	102.5	106.0

At the gill, fish respond to 'oxygen tension' (ie, % saturation), though it would be wrong to infer that the increased level of saturation caused by raised temperature somehow represents improved water quality. Many species are adversely affected when dissolved gases (including oxygen) become supersaturated – leading to 'gas bubble' disease, where small bubbles cover the gills and the exterior of the body. The ratio of dissolved oxygen to nitrogen under conditions of supersaturation is a complicating factor, with higher O:N ratios favouring survival. Large-scale mortalities have been attributed to extreme supersaturation (Woodbury, 1942), although a review by Weitkamp and Katz (1980) gives somewhat inconclusive results. From North America there is some experimental evidence that increased temperature reduces tolerance to supersaturation. Bouck *et al.* (1970) found that temperature increases in saturated water following exposure to supersaturation caused increased rates of blindness, whilst Nebeker *et al.* (1981) reported an increase of 2.7 times in mortality for steelhead trout (*Oncorhynchus mykiss*) and Chinook salmon (*Oncorhynchus tshawytscha*) in supersaturated water when the temperature increased from 8°C to 18°C, but no significant effect on coho salmon (*Oncorhynchus kisutch*) and sockeye salmon (*Oncorhynchus nerka*) under the same conditions. The experiments were conducted under conditions where mortality was not expected through temperature change alone. No recent data from UK TraC waters have been found.

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[3] BEEMS Data Centre, Centre for Environment, Fisheries and Aquaculture Science (Cefas), Lowestoft

4.2.2 Temperature impacts on oxygen consumption

Increasing temperature promotes the majority of biological and biochemical processes, including those that are mediated by bacteria and other micro-organisms. Microbial processes that can cause a reduction in DO with increasing temperature include:

➢ elevated Biochemical Oxygen Demand (BOD) of dissolved organic matter;

➢ increased respiration of cohesive sediments;

➢ increased rates of nitrification;

➢ increased primary productivity (as discussed in the next section, photosynthesis produces oxygen but also increases the ultimate organic load to the system).

All micro-organisms have a limited temperature range outwith which they cannot function. As a first approximation, it may be assumed that within their viable temperature range, activity increases logarithmically with temperature.

Portner and Knust (2007) have identified the pejus (ie, 'turning worse') temperature, beyond which the ability of animals to increase aerobic metabolism is reduced. This reduction is evident from the decline in aerobic scope, which is defined as the proportional difference between resting and maximal rates of oxygen consumption. The temperature range between the lower and higher pejus temperatures is much narrower than that between the critical temperatures beyond which the fish survives only for short periods.

4.3 Primary productivity

The detrimental impact on water quality of excessive growth of plankton is well documented, but less so for macrophytes. In theory, CW discharges causing a permanent uplift in water temperature (frequently not the case at coastal installations) may have an impact on primary production in the following ways:

➢ an earlier start to the plankton spring bloom;

➢ increasing growth rate (μ);

➢ changes to the kinetics of nutrient recycling.

In UK waters and especially in estuaries, light rather than temperature appears to be the limiting factor initiating primary production, but growth rates for phytoplankton increase with temperature. Bissinger *et al.* (2008) have derived the following relationship:

$\mu = 0.81e^{0.063T}$

where μ = maximum growth rate and T is temperature (°C).

Selecting an initial temperature of 15°C, an increase in temperature of 2°C would increase μ by 21% and a 7°C rise would increase μ by 56%.

Elevated temperatures have a less clear impact on macrophytes and angiosperms. There appears to be an optimum temperature for maximum growth, with reduced rates above and below this value. Herb and Stefan (2003) suggest that the steady-state biomass is independent of temperature but that the time taken to reach equilibrium is influenced by temperature.

Elevated water temperatures may accelerate denitrification, reducing the potential nutrient supply and productivity in systems where nitrogen is the limiting nutrient. In European temperate climate zones, phosphorus may be the limiting nutrient and, conversely, rapid recycling of phosphorus will be promoted by elevated temperatures.

Peperzak (2003) attempted to evaluate whether harmful algal blooms are likely to occur more or less often over the next 100 years in the North Sea as a consequence of future climate change. Climate change is expected to lead to an increase in extreme precipitation events (intense rainfall) in Britain, and this will result in sudden pulses of freshwater being released at the coast and hence intermittent salinity stratification in an area extending 30–40 km offshore. During such conditions, surface phytoplankton benefit from a decrease in salinity, greater availability of terrestrial nutrients, rapid increases in daily irradiance and higher water temperature, all of which are conducive to bloom formation. In addition, the relative direction of the North Atlantic Oscillation will influence rainfall patterns and thus the delivery of nutrients and freshwaters into estuaries.

At a global scale, satellite imagery has shown that warming of surface water is frequently reflected in reduced productivity. This may be a function of reduced vertical mixing limiting nutrient supply and is not indicative of conditions in UK TraC waters.

4.4 Ammonia

All fish excrete ammonia, but it is also toxic to them. Ammonia (as NH_3) is excreted across the gills, a process made more difficult if high ammonia concentrations are present in the surrounding water. Ammonia intoxification leads to stress and ultimately mortality. In solution, ammonia is present in an equilibrium between free ammonia and the ammonium ion:

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^-$$
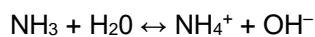
The equilibrium is influenced by pH, temperature and salinity; the fraction of un-ionized ammonia:

➢ increases with increasing temperature;

➢ increases with increasing pH;

➢ decreases with increasing salinity.

The principal toxic species is the free (un-ionized) ammonia, and UK and European saline water quality standards are based on un-ionized ammonia concentration. Under all normal conditions the equilibrium favours the ammonium ion, so a small change in the equilibrium can make a significant change in the concentration of un-ionized ammonia. In addition, synergistic effects must be considered; increased temperature may increase productivity and hence pH in the receiving water. Table 13 shows the sensitivity of the ammonia vs. ammonium equilibrium to temperature and pH.

Table 13 Concentration of un-ionized ammonia in seawater at salinity 30 and a total ammonia concentration of 1,000 µg l^{-1} NH$_3$-N

pH	Temperature, °C	Un-ionized ammonia, µg l^{-1}
7.8	10	9.5
	20	20.3
	30	41.0
8.0	10	14.9
	20	31.8
	30	63.5
8.2	10	23.4
	20	49.5
	30	97.0

Note on the determination of un-ionized ammonia: The interactions of the various parameters affecting the speciation of ammonia in seawater are complex. An algorithm for calculating the accurate concentration of un-ionized ammonia has been developed by Plymouth Marine Laboratory and is available as a simple Excel spread sheet via the BEEMS Data Centre.

The measurement of pH in saline waters needs to be undertaken with care. Conventional combination electrodes are slow to stabilize in salt water, which may cause errors if pH is constantly changing, as it may, for example, in an estuary. For accurate results, National Bureau of Standards (NBS) buffers – which were developed for freshwater applications – are unsuitable, and saline buffers are recommended.

4.5   Total residual oxidant and chlorination by-products

Chlorine, often in the form of sodium hypochlorite, is added to the CW circuit to prevent the settlement of growth of marine biota and avoid the blockage of heat exchangers, to control the development of bacterial slime and to maximize heat-transfer capacity. The complex chemistry of chlorine disinfection is discussed in detail elsewhere in the BEEMS documentation (eg, Taylor, 2006), and so only a brief description is given here.

'Chlorine' may be added as elemental chlorine (rare in the United Kingdom), in solution as (sodium) hypochlorite or produced *in situ* by electro-chlorination. In seawater the hypochlorous acid–hypochlorite (HOCl$^-$ OCl$^-$) mixture reacts rapidly with bromide to form hypobromous acid (HOBr), and much of the subsequent chemistry is bromine based (Figure 8). In the presence of high ammonia concentration, the ammonia/ammonium ion may compete with the bromide and form chloramines. Hypobromous acid/hypobromate will itself react with ammonia to form acutely toxic bromamines (Figure 9).

The actual disinfection is thus due to a complex mixture of oxidants, with hypobromous acid and a mixture of chloramines and bromamines the main constituents. In a typical seawater the halamines will normally be significant only when the Cl$_2$:N molar ratio is <1.5. The sum of this oxidizing (disinfecting) power is conventionally described as Total Residual Oxidant (TRO), expressed as a chlorine equivalent.
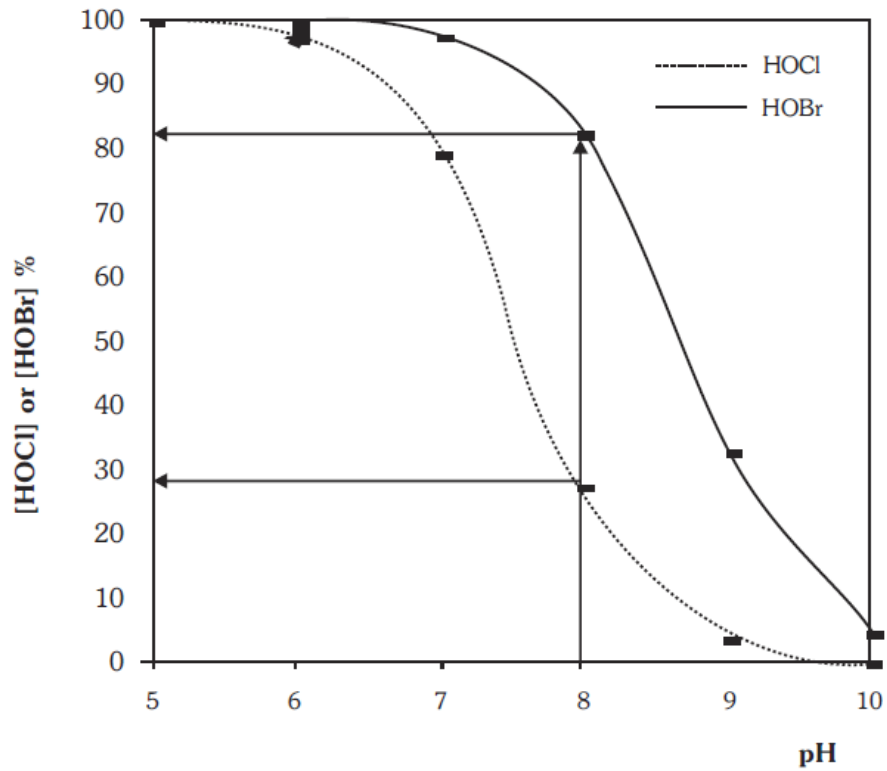
Figure 8 Comparison of the dissociation of hypochlorous acid and hypobromous acid with changing pH at 20°C (from Khalanski and Jenner, 2011)
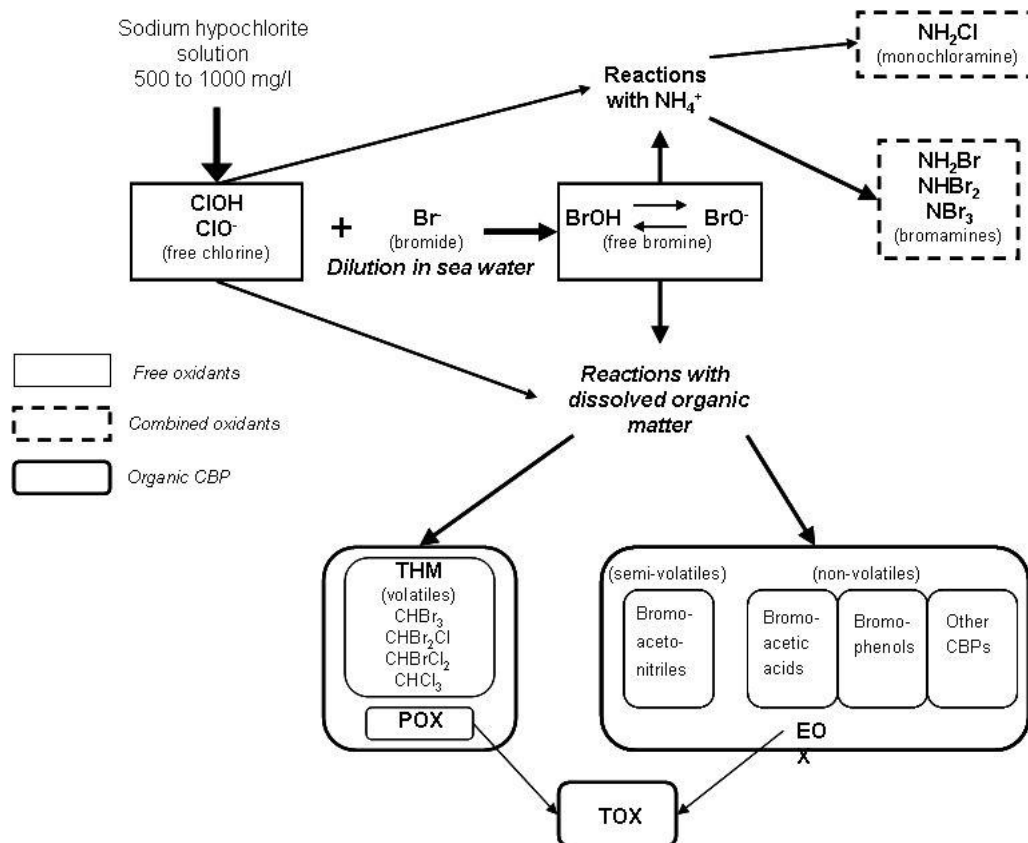
Figure 9 Chlorine- and bromine-based chemistry during chlorination of seawater (Taylor, 2006)

Oxidizing species further react with dissolved and particulate organic matter to form chlorine by-products (CBPs). When applied to the chlorination of saline waters, this term refers to several hundred chlorinated and brominated compounds. The principal compounds are:

➢ trihalomethanes – principally bromoform ($CHBr_3$);

➢ haloacetic acids – principally mono- and dibromoacetic acids;

➢ haloacetonitriles – eg, dibromoacetonitrile.

Halophenols and bromate have also been detected.

Oxidants formed by chlorination are not persistent in natural waters, and the major environmental concern is through the production of the numerous non-oxidizing compounds formed between 'chlorine' and the organic and mineral constituents of natural waters (Taylor, 2006; Khalanski and Jenner, 2011).

4.6   Effect of temperature

4.6.1   Total residual oxidant

The toxicity of TRO has been shown to increase with increasing temperature. The increasing susceptibility of organisms to seawater chlorination as water temperature increases may be partly due to changes in equilibrium, but the most important factor is the increase in metabolic rate with

temperature of all cold-blooded animals. Between 5°C and 35°C, each 10°C rise will increase metabolism by a factor ≥2.

4.6.2 Chlorination by-products

Increased temperature may be expected to increase the rate of CBP formation but also their ultimate degradation; it is not possible to speculate in general on changes in their maximum concentration, but significant changes are unlikely given the modest temperature rises expected, although the relative concentrations of individual compounds may change.

In the United Kingdom, chlorination is normally only undertaken when the risk of fouling is greatest, typically during late spring through to autumn. Even during this restricted period water temperatures can vary significantly, and this may be expected to influence the kinetics of CBP formation and survival. There is little or no information on the effect of temperature on the sensitivity of relevant organisms to CBPs.

4.6.3 Chlorine dioxide

Chlorine dioxide (ClO_2) is a very potent oxidative biocide, and it has been used on freshwater cooling systems (eg, urban cooling towers) where the formation of CBPs is unacceptable. It is often incorrectly claimed by water treatment companies that chlorine dioxide does not produce CBPs when used on saline systems. In reality, ClO_2 will oxidize bromide to bromine, and the subsequent bromine-based chemistry is similar to that arising when hypochlorite is used.

4.6.4 Hydrazine

Hydrazine N_2H_4 is normally encountered in aqueous solution as the hydrate: $N_2H_4 \cdot H_2O$. It is relatively stable but reacts rapidly with any oxidizing agents present. In taking into account the decay of hydrazine, increases in water temperature were found to enhance the toxicity of the compound for bluegill sunfish (Table 14) (Hunt *et al.*, 1981).

Table 14 Aquatic toxicity of hydrazine to bluegill sunfish (*Lepomis macrochirus*)

Temperature, °C	96-h LC$_{50}$ mg l^{-1}
10	1.6
15.5	1.0
21	1.2

Note: LC$_{50}$ = the concentration of a chemical which kills 50% of of the test organisms

These data need to be interpreted with care as the results are somewhat inconclusive and refer to a freshwater species that appears relatively tolerant of hydrazine. Henderson *et al.* (1981) indicate that for the eggs of the fathead minnow, the 48-h LC$_{50}$ was 0.1 mg l^{-1} and the 48-h NOEL (no observed effect limit) was 0.001 mg l^{-1}, but the data are for a single temperature (21°C). Furthermore, it is of note that the half-life of hydrazine added to natural seawater reduces with increasing temperature (Table 15).

Table 15 Degradation of hydrazine (12.8 mg l^{-1}) added to natural seawater (James, 1989)

Temperature, °C	Half-life (days)

0	117
10	125
20	2.5
35	0.83

The impact of temperature in this experiment was higher than that reported by Jingqiu *et al.*, (1994) for distilled water, where the rate at 20°C was 1.5 times than at 10°C, suggesting the combination of salinity and temperature may be important. The high water-solubility of hydrazine and its low log_K_{ow} water provides negligible potential for bioaccumulation. However, analysing hydrazine in natural waters at environmentally significant concentrations is difficult, and published data must be interpreted with care.

4.7   Metal speciation

Temperature influences the speciation of dissolved metals. In seawater a fraction of dissolved metal will be complexed and biologically inactive. The behaviour and sensitivity of different metals to temperature change varies according to the dominant complex, and increasing temperature may increase or decrease the proportion of bioavailable metal in solution. Table 16 groups the more common heavy metals into those with broadly similar behaviour. It should be noted that other factors – for example, pH and salinity – may have far larger impacts on speciation than on temperature alone.

Table 16 The impact of temperature on metal speciation (adapted from Byrne *et al.*, 1988)

Type	Metals	Increase or decrease in free metal with rising temperature	Sensitivity to temperature change
A	Al, Cr, Fe(III)	Decrease	Very high
B	Ag, Cu, Cd, Hg	Increase	Moderate
C	Cu	Decrease	High
D	Co, Fe(II), Mn, Ni, Pb, Zn	Decrease	Low

5 Thermal tolerances of fish

5.1 Introduction

5.1.1 Geographic range

This discussion is restricted to waters around the British Isles corresponding to WFD Ecoregions 1 (North Sea) and 4 (Atlantic), except where making comparisons with other geographic regions. Reference to present-day fish distribution at latitudes to the south of the British Isles (below 50ºN) can be helpful when considering likely alterations in species distributions associated with climate change.

Whereas previous thermal standards that have been developed in the United Kindom, primarily for application to freshwaters, have distinguished between salmonid and cyprinid waters, this distinction is inappropriate for TraC waters, where a greater diversity of fish species may be found. With reference to thermal responses of species, a more appropriate distinction can be drawn on zoogeographic lines, in relation to Arctic, Boreal and Lusitanian distributions (see Engelhard *et al.,* 2011). These biogeographic realms are imprecise, but a Lusitanian distribution covers warmer latitudes, approximately from the Mediterranean to the southern United Kingdom (ca 51°N); an Arctic distribution is self-explanatory (ca >60°N), and a Boreal distribution includes latitudes between these two; Arctic–Boreal (from the Arctic Circle through the northern temperate zone to the southern United Kingdom), and Boreal–Lusitanian distributions (north of the United Kingdom down to Iberia) are terms used to account for species that overlap these precise limits, thus reflecting a biogeographic continuum.

At any site, the fauna and flora include some species biogeographically close to their cooler limit of distribution (their northern limit in the Northern hemisphere), and others close to their warmer biogeographic limit (their southern limit in the Northern hemisphere). In British waters, while most species are Boreal (or Boreal-Lusitanian) and thus away from their temperature limits, populations of species showing a Lusitanian distribution find their northern (cooler) limit, and Arctic–Boreal species populations find their southern (warmer) limit; increases in the temperature regime may have a more significant effect on distribution on this last category. The migration of southern, Lusitanian species into British waters, and especially the waters of the south-west, has been documented and considered to reflect changing temperature conditions as the result of climate change (Stebbing *et al.*, 2002). Accordingly, increased temperatures adjacent to power plants can be hypothesized as attracting similar species. The migration northwards of cold-water species out of southern British waters has not yet been documented to the same extent.

5.1.2 Types of fish

5.1.3 Fish can be assigned to one of a number of 'ecological guilds'. For present purposes, the simple scheme of Elliott and Taylor (1989) is used (Conservation designations

Conservation citations arise from the Wildlife & Countryside Act, the EU Habitats Directive (in the case of European designated sites), the IUCN Red List or the UK Biodiversity Action Plan. The latter aims, among other things, to protect commercial marine species that are considered to be below safe biological limits (SBL) from over-exploitation by controlling Total Allowable Catch.

Table 17), although it is of note that these guilds have been modified in the light of recent information and by taking a wider geographical approach (Elliott, *et al.*, 2007; Franco *et al.*, 2008). Table 18 provides a shortlist of 34 fish species selected for consideration of thermal tolerance, including species of notable conservation importance and covering relevant ecological guilds. The present

review is predominantly concerned with fish that would be found in British WFD TraC waters. The species listed are intended to provide a good cross section of taxa suitable for the development of national surface water thermal standards. This core set of fish species forms the basis of the present review, although other species are referenced where useful data were found. While data are not necessarily yet available for all of these species, listing the full set of 34 species for each aspect of the review helps to identify areas where data are lacking.

The zoogeographic affinities of the various species are also shown in Table 19. In this document, the terms 'warm-water' and 'cold-water' are used to refer to fish of Lusitanean and Arctic–Boreal distribution, respectively. Table 19 takes this further, listing the northern and southern limits recorded for each species and indicating average summer high (September) and winter low (February) sea surface temperatures at the northern and southern limits of their ranges. This serves to illustrate the range of temperatures to which a species can be expected to adapt, although it should be recognized that records of a species in a particular area may be seasonal and therefore do not imply that they would experience the full temperature range shown for that area. Drinkwater (2005) makes this clear in the case of cod (*Gadus morhua*), for example; while the distribution of cod extends across waters that range from below zero temperature to >20ºC, they are normally found where water temperatures are between 0 and 12ºC, their preferred temperature range. This aspect is discussed further in Section 5.4.

In Table 19, 'warm'- and 'cold'-water species have been colour coded in pink or blue respectively, and this convention is continued where relevant in later tables throughout the report. Figure 10 is a map showing where the British Isles lie in relation to the various latitudes.

5.1.4 Conservation designations

Conservation citations arise from the Wildlife & Countryside Act, the EU Habitats Directive (in the case of European designated sites), the IUCN Red List or the UK Biodiversity Action Plan. The latter aims, among other things, to protect commercial marine species that are considered to be below safe biological limits (SBL) from over-exploitation by controlling Total Allowable Catch.

Table 17 List of ecological guilds, their abbreviated forms and notes on water body use (Elliott and Taylor, 1989)

Number	Ecological guild (abbreviated form)	Use of water body
1	Estuarine residents (ER)	Spend whole life in transitional water
2	Marine seasonal (MS)	Marine species with seasonal migrations to transitional water as adults
3	Marine juvenile (MJ)	Marine species using transitional water as a nursery area
4	Diadromous species (CA)	Species that use transitional waters during migrations between marine and freshwater habitats
5	Adventitious marine species (MA)	Marine species with no transitional water requirement
6	Adventitious freshwater species (FW)	Freshwater species with no transitional water requirement

Table 18 Fish species considered in the review of thermal tolerance, their ecological guild and their conservation status

Species	Common name	Guild, distribution	Conservation citations
Agonus cataphractus	Pogge	MA, A–B	
Alosa alosa	Allis shad	CA, LUS	HD; IUCN; UKBAP; WCA
Alosa fallax	Twaite shad	CA, LUS	HD; IUCN; UKBAP; WCA
Anguilla anguilla	Eel	CA, LUS	UKBAP
Atherina presbyter	Sand smelt	MJ, LUS	IUCN
Chelon labrosus	Thick-lipped grey mullet	MJ, LUS	IUCN
Clupea harengus	Herring	MJ, A–B	UKBAP-gp
Dicentrarchus labrax	Sea bass	MJ, LUS	
Dipturus batis	Skate	MA, LUS	IUCN
Engraulis encrasicolus	Anchovy	MA, LUS	
Gadhus morhua	Cod	MA, A–B	UKBAP
Lampetra fluviatilis	River lamprey	CA, LUS	HD; IUCN; UKBAP
Leuciscus leuciscus	Dace	FW, LUS	
Limanda limanda	Dab	MA, A–B	
Liparis liparis	Striped sea snail	MA, A–B	IUCN
Liza aurata	Golden grey mullet	MJ, LUS	IUCN
Liza ramada	Thin-lipped grey mullet	MJ, LUS	IUCN
Merlangius merlangus	Whiting	MA, A–B	UKBAP-gp
Mullus surmuletus	Red mullet	MA, LUS	
Osmerus eperlanus	Smelt	CA, A–B	UKBAP
Petromyzon marinus	Sea lamprey	CA, LUS	HD; IUCN; UKBAP
Platichthys flesus	Flounder	ER, LUS	
Pleuronectes platessa	Plaice	MA, A–B	UKBAP-gp
Pomatoschistus minutus	Sand goby	MA, LUS	
Psetta maxima	Turbot	MA, LUS	
Salmo salar	Salmon	CA, A–B	HD; IUCN; UKBAP
Salmo trutta	Sea trout/brown trout	CA, A–B	UKBAP
Sardina pilchardus	Sardine	MJ, LUS	
Scyliorhinus canicula	Dogfish	MA, LUS	IUCN
Solea solea	Dover sole	MA, A–B	UKBAP-gp
Sprattus sprattus	Sprat	MJ, LUS	

Species	Common name	Guild, distribution	Conservation citations
Syngnathus rostellatus	Nilsson's pipefish	MA, LUS	
Trisopterus luscus	Pouting	MA, A–B	

Notes: HD = Annex 2 of the Habitats Directive; IUCN=International Union for Conservation of Nature and Natural Resources Red List of Threatened Species (www.iucnredlist.org); WCA = Schedule 5 of the Wildlife & Countryside Act 1981 as amended; UKBAP = UK Biodiversity Action Plan; UKBAP-gp = grouped plan for commercial marine species. Distribution: A–B = Arctic–Boreal, LUS = Lusitanean.

Table 19 Latitudinal ranges of the 34 fish species listed in earlier tables and associated surface sea temperatures (SST)

| Species (Guild) | Common name | Latitude | Surface sea temperatures (–20°W) | | | |
| | | | September (°C) range | | February (°C) range | |
			Min	Max	Min	Max
Agonus cataphractus	Pogge	77°N – 43°N	–0.3	20.4	Ice	12.6
Clupea harengus	Herring	80°N – 33°N	–0.9	24.7	Ice	17.8
Engraulis encrasicolus	Anchovy	62°N – 37°N	12.4	23.5	7.4	15.5
Gadus morhua	Cod	80°N – 35°N	–0.9	24.3	Ice	16.2
Lampetra fluviatilis	River lamprey	69°N – 37°N	3.2	23.5	–0.9	15.5
Leuciscus leuciscus	Dace	72°N – 41°N	1.7	21.3	Ice	13.4
Limanda limanda	Dab	72°N – 43°N	1.7	20.4	Ice	12.6
Liparis liparis	Striped sea snail	81°N – 48°N	–1.0	18.0	Ice	11.0
Merlangius merlangus	Whiting	72°N – 35°N	1.7	24.3	Ice	16.2
Osmerus eperlanus	Smelt	70°N – 43°N	1.9	20.4	–1.0	12.6
Salmo salar	Salmon	72°N – 37°N	1.7	23.5	Ice	15.5
Salmo trutta	Sea trout/ brown trout	71°N – 34°N	1.8	24.6	Ice	17.2
Syngnathus rostellatus	Nilsson's pipefish	61°N – 43°N	13.0	20.4	7.4	12.6
Trisopterus luscus	Pouting	62°N – 25°N	12.4	24.7	7.4	22.0
Alosa alosa	Allis shad	61°N – 20°N	13.0	26.6	7.4	22.8
Alosa fallax	Twaite shad	70°N – 27°N	1.9	24.2	–1.0	21.6
Anguilla anguilla	Eel	75°N – 8°N	0.2	27.8	Ice	27.7
Atherina presbyter	Sand smelt	60°N – 14°N	13.4	27.5	7.8	24.7
Chelon labrosus	Thick-lipped grey mullet	65°N – 12°N	9.0	28.2	4.5	25.2
Dicentrarchus labrax	Sea bass	72°N – 11°N	1.7	27.8	Ice	25.5
Dipturus batis	Skate	69°N – 12°N	3.2	28.2	–0.9	25.2
Liza aurata	Golden grey mullet	64°N – 20°N	11.0	26.6	7.0	22.8
Liza ramada	Thin-lipped grey mullet	60°N – 20°N	13.4	26.6	7.8	22.8
Mullus surmuletus	Red mullet	62°N – 14°N	12.4	27.5	7.4	24.7
Petromyzon marinus	Sea lamprey	72°N – 25°N	1.7	24.7	Ice	22.0

| Species (Guild) | Common name | Latitude | Surface sea temperatures (–20°W) | | | |
| | | | September (°C) range | | February (°C) range | |
			Min	Max	Min	Max
Platichthys flesus	Flounder	72°N – 30°N	1.7	24.7	Ice	19.6
Pleuronectes platessa	Plaice	73°N – 20°N	1.5	26.6	Ice	22.8
Pomatoschistus microps	Common goby	64°N – 20°N	11.0	26.6	7.0	22.8
Pomatoschistus minutus	Sand goby	71°N – 35°N	1.8	24.3	Ice	16.2
Psetta maxima	Turbot	70°N – 30°N	1.9	24.7	–1.0	19.6
Sardina pilchardus	Sardine	68°N – 14°N	7.1	27.5	1.3	24.7
Scyliorhinus canicula	Dogfish	63°N – 12°N	11.5	28.2	6.7	25.2
Solea solea	Dover sole	67°N – 17°N	7.5	28.1	1.7	23.6
Sprattus sprattus	Sprat	66°N – 30°N	8.4	24.7	3.6	19.6

Shadings indicate cold-water (blue) or warm-water (pink) affinities of species. (Sources: fish distributional data from www.Fishbase.com; SST data from http://neo.sci.gsfc.nasa.gov/Search.html?datasetId=MYD28M)
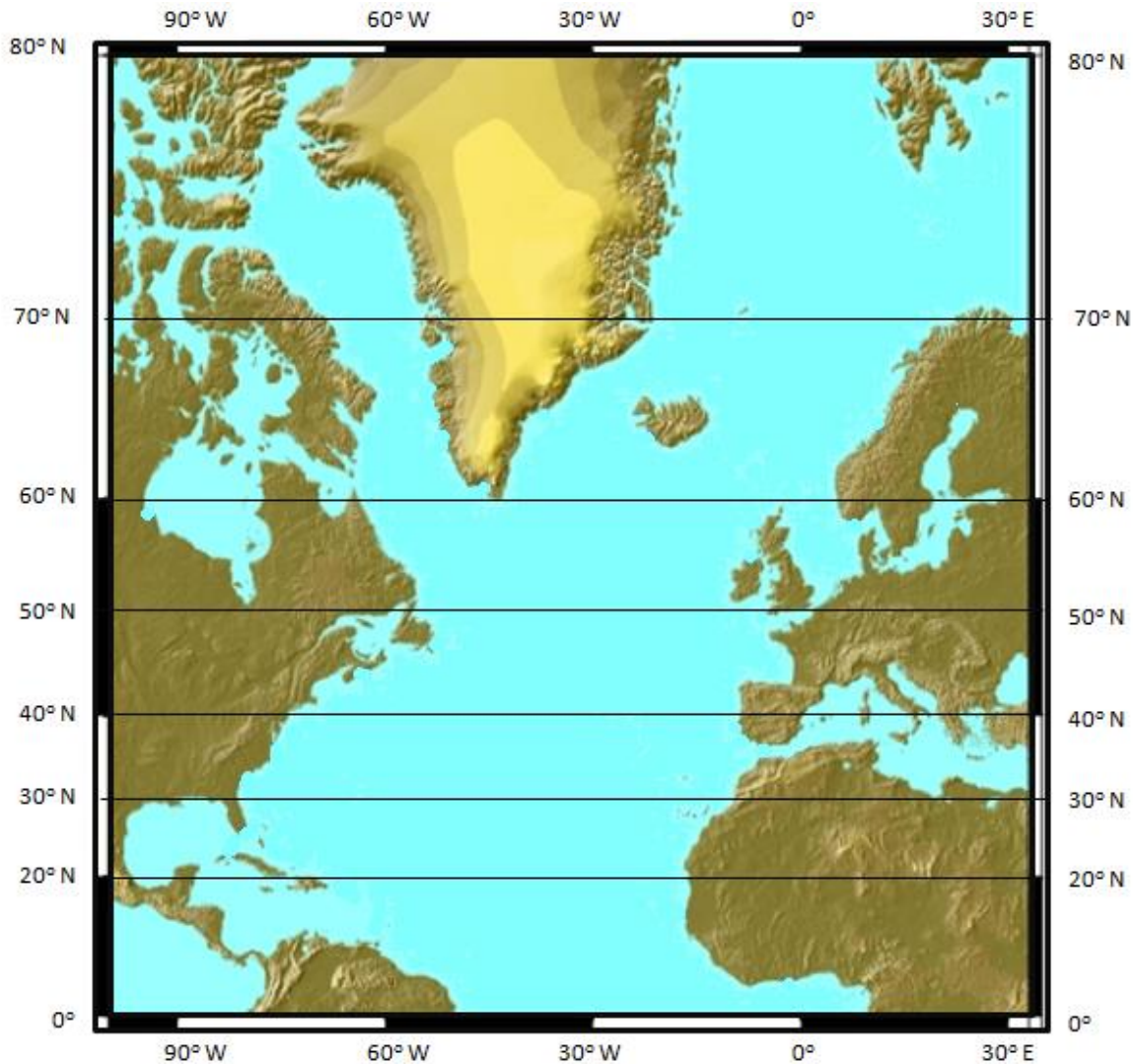
Figure 10 Latitudes in the North Atlantic

5.1.5 Water quality characteristics

As indicated in the previous chapter, other water quality characteristics, such as dissolved oxygen, salinity and toxicants, may be relevant to defining the response of fish to environmental temperature levels; these are discussed when relevant to illustrate principles. It is beyond the scope of this study to provide comprehensive data on interactions of polluting substances with water temperature.

5.2 Overview of thermal effects on fish

5.2.1 Thermal relationships in fish

When reviewing literature on thermal effects on fish, terminology can be confusing. Jobling (1981) provides the following explanation that 'acclimation temperature' refers to the temperature to which the fish have become adapted in the short term (ie, days). The thermal responses of fish relative to acclimation temperature can be divided into zones of tolerance, resistance and preference (Figure 11). The upper boundary is represented by the critical thermal maximum (CTM), which is a measure of thermal resistance determined by increasing the water temperature at a rate of 1°C min^{-1} until the

fish lose equilibrium, with survival times above CTM being close to zero. The upper and lower incipient lethal temperatures (UILT and LILT) represent the temperatures at which, theoretically, 50% of the population could survive indefinitely. The ultimate UILT (UUILT), LILT and CTM vary with acclimation temperature and the previous thermal exposure. These form the boundaries of the thermal tolerance zone (the shaded area in Figure 11). By choice, fish will spend as much time as possible within a certain preferred temperature range (thermal preferenda) at which they function best. Acute thermal preferenda are usually determined over a short period by their acclimation temperature, but, left in a temperature gradient for a long period, they will gravitate towards the optimum for the species (final preferendum). Jobling (1981) notes that fish will make voluntary forays into sub-optimal thermal conditions when it is advantageous to do so – for example, to gain access to food resources.
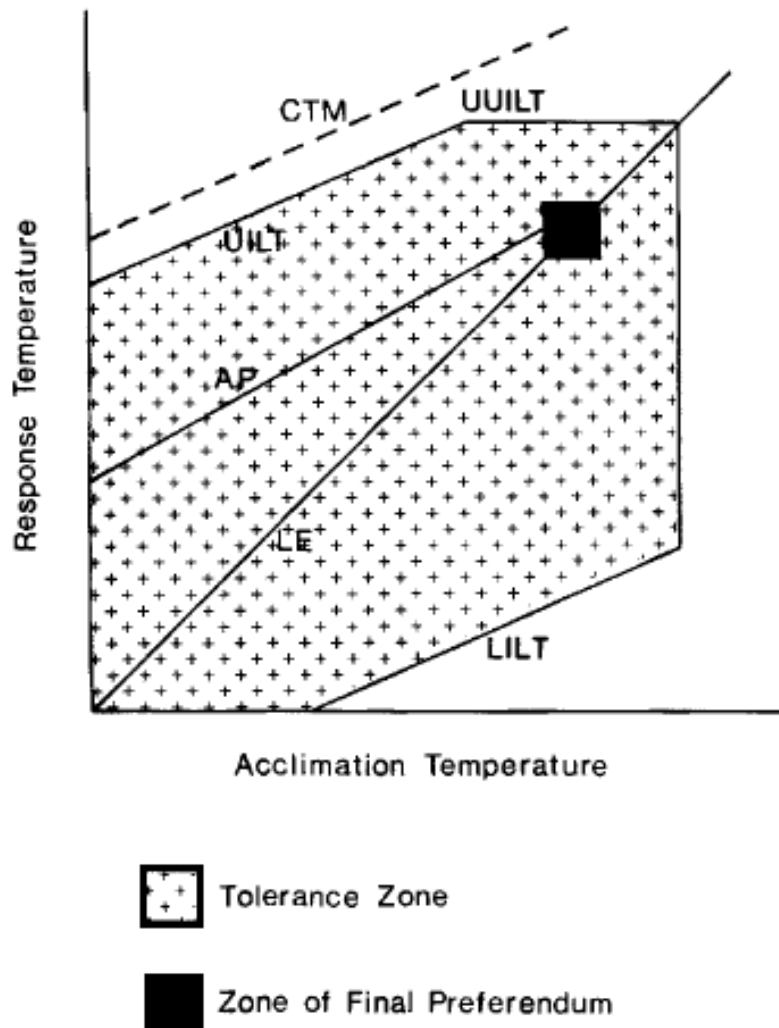
Figure 11 Temperature relations of fish

Note: CTM = Critical thermal maximum; UILT = upper incipient lethal temperature; LILT = lower incipient lethal temperature; UUILT = ultimate upper incipient lethal temperature; AP = acute thermal preferendum; LE= line of equality (Jobling, 1981).

When applying species-specific lethality data to a standard setting, it is preferable to choose a metric that is protective of the majority of the population; thus the UILT (based on a median response – it would better perhaps be called the UILT$_{50}$) would be a more suitable measure than the CTM (not survivable by any of the population). On the other hand, the 'UILT$_{90}$' (the highest temperature that

90% of the population could survive) would be even better. In practice, data based on uniform measurement standards and metrics are not available for a wide range of species, and so for the present a rather looser definition will have to suffice. Jobling (1981) also recognized this problem: in trying to define upper limits of temperature tolerance, he found that different authors presented either CTM, UILT or UUILT values and in his own review was obliged to adopt the term 'upper lethal temperature' as a catch-all. Although this inevitably reduces the precision of the data, for the purposes of standards setting, a safety margin can be added. The term 'upper lethal temperature' (ULT) is used in this sense in this document.

5.2.2 Temperature–DO interactions

As discussed in the earlier chapters, temperature plays a major role in determining DO levels, both via temperature-driven metabolic processes and as a determinant of oxygen solubility. Since fish are poikilotherms, their metabolic rate and oxygen requirements increase with temperature. In much of the evidence, temperature and DO effects are interlinked and cannot be treated independently, especially where DO levels are below a certain critical level. Similarly, Poxton and Allouse (1982) gave water quality criteria limits for marine species and indicated the varying sensitivities of different species. Following the earlier work, current WFD standards for DO are shown in Table 20. For the present purpose, it is recommended that DO should be considered in the context of temperature effects only where it falls below 5.7 mg l^{-1}. However, it is emphasized that dissolved oxygen tolerances also vary according to the energy expended by the fishes and other water physical characteristics – for example, lower DO concentrations will affect salmonids migrating through an estuary if they have to swim against strong currents, hence a summer estuarine migration (in warmer waters) of salmon grilse can result in an oxygen debt or even mortalities (Elliott and Hemingway, 2002).

Table 20 Dissolved oxygen standards as minimum allowable 5%-ile values for transitional and coastal waters (UKTAG, 2008a)

	Freshwater	Marine	Description
	5%-ile mg l^{-1}		
High	7	5.7	Protects all life-stages of salmonid fish
Good	5–7	4.0–5.7	Resident salmonid fish
Moderate	3–5	2.4–4.0	Protects most life-stages of non-salmonid adults
Poor	2–3	1.6–2.4	Resident non-salmonid fish, poor survival of salmonid fish
Bad	2	1.6	No salmonid fish. Marginal survival of resident species

5.2.3 Effect of salinity

Any effects of salinity upon the physiological responses to temperature are clearly important in the context of transitional waters. Although information on this aspect is sparse, Kinne (1960) showed that different levels of salinity can alter the thermal response of fish by lowering the optimum temperatures at lower salinities.

5.3 Thermal lethality

5.3.1 Temperature tolerance

A temperature of around 40–41°C can be regarded as the limiting CTM for all temperate fish. At lethal temperatures, proteins begin to denature and enzyme systems fail (Somero and Hofmann, 1997). In addition, sudden changes in temperature may kill fish due to 'thermal shock'.

In British waters, no species would tolerate temperatures as high as 40°C, although eel can survive to 38°C and some freshwater cyprinids can tolerate temperatures in the high-thirties (Langford, 1990).

 shows upper lethal temperatures for fish species commonly found in UK transitional and coastal waters. Since bulk temperatures (ie, outside any thermal plume) in Britain are unlikely to exceed around 25°C outside the mixing zone of thermal discharges, it is unlikely that fish kills would occur due to temperature alone. Even within thermal plumes where temperatures can reach 10–14°C above ambient, Langford's (1990) review of ecological effects of thermal discharges, concludes that '*in view of the vast amount of literature dealing with thermal discharges, very few large-scale mortalities have occurred which can unequivocally be related to high temperature*'.

It is of note that for 17 of the 34 species listed in

, no published lethality data were found. Important omissions from the thermal tolerance data for species that are considered a conservation priority (

) include river lamprey, smelt, whiting and sand-smelt. River and sea lamprey are related warm-water species (Wheeler, 1969), and the relatively high ULT of 31°C for the sea lamprey suggests that temperatures in UK transitional waters are unlikely ever to be lethal to river lamprey; similarly, sand-smelt has a Lusitanian distribution. Smelt, on the other hand, is a cold-water fish, closely related to the salmonidae and therefore is more likely to be vulnerable to high temperatures. This is supported by studies with smelt in the Thames Tideway: attempts to transfer adult smelt to the laboratory during August and September 2002 failed repeatedly, irrespective of capture and handling arrangements and oxygenation levels, until the river temperature fell below 18°C; after this, survival of transferred fish was close to 100%. It seems that while fish were able to live in the Tideway at temperatures of 18–22°C, there was insufficient metabolic headroom to cope with handling stress. (Turnpenny *et al.*, 2004).

With reference to power stations, it is generally ULTs that are of interest, but extreme cold can also kill fish. During severe winter conditions such as experienced in the 1960s and in 1996, sole were excluded from the shallows to the extent that mass mortality events were reported (Woodhead, 1964). A similar mass mortality event for sole was reported in early 1929 (Lumby and Atkinson, 1929), especially on the Terschelling ground off the Dutch coast. This coincided with seawater temperatures that were 5°C colder than was considered 'normal' for that time of year.

Table 21 Summary of upper lethal temperatures in fish

Species (Guild)	Common name	Eggs			Larval			Juvenile			Adult		
		AT, °C	ULT, °C	Source	AT, °C	ULT, °C	Source	AT, °C	ULT, °C	Source	AT, °C	ULT, °C	Source
Agonus cataphractus	Pogge												
Clupea harengus	Herring	?	20	Reid *et al.*, 1999	7.5–15.5 7–9	22–24 20.5–23.5	Blaxter, 1960; Yin & Blaxter, 1987	?	19.5–21.2	Reid *et al.*, 1999	? ?	22–24 19.5–21.2	Blaxter, 1960; Brawn, 1960
Engraulis encrasicolus	Anchovy				?	12–14	King *et al.*, 1978. In Brochier *et al.*, 2008						
Gadus morhua	Cod				7–9	15.5–18	Yin & Blaxter, 1987	12	23	Gamperl *et al.*, 2007	10	22.2	
Lampetra fluviatilis	River lamprey												
Leuciscus leuciscus	Dace												
Limanda limanda	Dab												
Liparis liparis	Striped sea snail												
Merlangius merlangus	Whiting												

Species	Common name												
Osmerus eperlanus	Smelt												
Salmo salar	Salmon	?	16	Elliott & Elliott, 2010	27	7 day: 27.8 10 min: 33	Elliott, 1991	21	7 day: 27.8 10 min: 33	Elliott & Hurley, 1997	?	28 (22–33)	Danie *et al.*, 1984; Jonsson & Jonsson, 2009
Salmo trutta	Sea trout/ brown trout)		14–16	Ojanguren & Brana, 2003				20	7 day: 24.7 10 min: 30	Elliott & Elliott, 1995	? ?	23 26.4	Cherry *et al.*, 1977; Alabaster & Downing, 1966
Syngnathus rostellatus	Nilsson's pipefish												
Trisopterus luscus	Pouting												
Alosa alosa	Allis shad (	?	30	Charles & Jatteau, 2010	?	35	Charles & Jatteau, 2010						
Alosa fallax	Twaite shad												
Anguilla anguilla	Eel										29	38	Sadler, 1979
Atherina presbyter	Sand smelt												
Chelon labrosus	Thick-lipped grey mullet												
Dicentrarchus labrax	Sea bass							?	32	Barnabé, 1991			

Species	Common name												
Liza aurata	Golden grey mullet												
Liza ramada	Thin-lipped grey mullet												
Mullus surmuletus	Red mullet												
Petromyzon marinus	Sea lamprey				25	31	Potter & Beamish, 1975						
Platichthys flesus	Flounder				?	21.5–24	Yin & Blaxter, 1987						
Pleuronectes platessa	Plaice											26–27	Waede, 1954. In: Fonds *et al.,* 1992
Pomatoschistus microps	Common goby												
Pomatoschistus minutus	Sand goby										15	30	Hesthagen, 1979
Psetta maxima	Turbot										?	28–30	
Sardina pilchardus	Sardine												
Solea solea	Dover sole	?	19–22	Fonds, 1979									
Sprattus sprattus	Sprat	?	14.7	Petereit *et al.*, 2008									

ULT = Upper lethal temperature; AT = Acclimatization temperature

Where different values or ranges of values are cited in

, this probably reflects acclimation temperatures (see Figure 11), but lower values could be due, for example, to stress induced by experimental handling. In either case, the higher values should be a better reflection of the upper lethal limit.

5.3.2   Lethal limits: summary

Evidence suggests that temperature *per se* is rarely a cause of fish kills in temperate waters. Summer temperatures in the warmest southern English estuaries (eg, the Thames) can reach 23–24°C. For most warm-water species, ULTs in the high twenties to thirties degrees centigrade provide adequate headroom for survival under these conditions, although the situation is more marginal for some cold-water species such as the salmonids and probably smelt. However, it is clear that the temperature could become critical for survival of these species under the more severe global-warming scenarios (UKCP09, 2011).

Although locally higher temperatures may occur around thermal discharges, fish are usually able to avoid exposure to lethal temperatures provided that the plume does not pervade the full channel width and depth.

Lethally low DO levels in transitional waters are very often associated with high summer temperatures owing to the reduced solubility of oxygen, increased microbial activity and Effective Oxygen Demand (EOD) and increased metabolic demands of the fish. In fish that are forced to live at temperatures close to lethal limits, hypoxic conditions that might not be lethal by themselves can reduce the metabolic 'headroom', increasing the risk of lethality. Salmonids and smelt appear most at risk in this respect.

5.4   Thermal preferenda

5.4.1   Thermal preferenda in fish

Arguably, thermal preferenda are the most important criteria in relation to thermal standards development. Early work on this subject is attributable to Fry (1947), who proposed that his 'final preferendum' paradigm would facilitate comparative studies of temperature preference among different animal groups. According to Fry's definition, the final preferendum is that temperature at which preference and acclimation are equal, and to which an animal in a thermal gradient will finally gravitate regardless of its acclimation temperature.

Brett's (1971) classic study of the thermobiology of sockeye salmon (*Oncorhynchus nerka*) demonstrated that the preferred temperature in this species (15°C) coincided with the optimum temperature for metabolic scope, cardiac scope, swimming performance and growth. Magnuson *et al*. (1979) proposed the concept of fish occupying a 'thermal niche', a band of temperatures in which they prefer to live: they found that most fish will spend two-thirds of their time at within ±2°C of their temperature preferendum and all of their time within ±5°C. This no doubt serves to keep fish within their optimum physiological temperature range (but note that while each species will exhibit a final preferendum as described in Section 5.2.1, their preferendum at any one time will be governed by acclimation temperature, which will alter seasonally). This mechanism is described by some authors as 'behavioural thermoregulation' (see Jobling, 1997).

The process of behavioural thermoregulation has been studied extensively in freshwater habitats, particularly with salmonids, which have amongst the lowest optimal temperatures for fish in temperate regions. Baird and Krueger (2003) demonstrated a reduction in metabolism of 20% when brook (*Salvelinus fontinalis*) and rainbow trout (*Oncorhynchus mykiss*) maintained position in areas 4°C cooler than the ambient water temperature, making a greater proportion of energy available for growth

and reproduction. Similarly, studies of behavioural thermoregulation of salmonids in Northwest America discovered up to 98% of salmon populations residing in only a few small pools 3–8°C lower than ambient temperatures (Berman and Quinn, 1991; Torgersen *et al.,* 1999; Baird and Krueger, 2003).

Behavioural thermoregulation has also been demonstrated in populations of Dover sole in an estuarine/marine environment. Vavoulis and Koutsikopoulos (2008) noted that if the body temperature of Dover sole is lower than their optimum temperature and they sense a cooling trend of water over the previous few weeks, then they would move to deeper, slower freezing waters. However, if the temperature is lower than optimum and they sense a warming of surrounding water, they move towards shallower, more rapidly warming areas. It may be surmised that by adding warmer water to these areas in summer (eg, from thermal plumes), the water temperature in shallow nursery zones could raise temperatures above the optimum temperatures of some flatfish and thus reduce occupation. However, such fish might equally benefit from a phenological change, whereby the period when they were able to remain in warmer intertidal areas becomes extended.

Magnuson *et al.* (1979) also proposed that fish will compete for resource use within a particular thermal niche. Therefore, at the individual river-basin level the dynamics of preference competition will be a major factor in determining fish community structure, and on a global scale it will determine the geographic range of the species.

Investigations into the preferred temperatures of species have been carried out mainly in the laboratory (see Table 22). The experimental method usually involves some form of gradient or choice chamber apparatus that allows fish to select from a range of offered temperatures. As with lethality data, preferred temperatures depend on the acclimation temperature and therefore vary seasonally. Owing to these inherent complexities surrounding thermal preferenda, including seasonality, acclimatization and food availability (Despatie *et al.*, 2001), many studies choose to quote 'avoidance' temperatures, which in general show less seasonal variation. In a comparison of temperature preferendum and avoidance of the sand goby over different seasons, Hestagen (1979) noted only 2°C difference in avoidance temperatures between spring and summer (4°C and 6°C, respectively). Contrastingly, the preferendum rises from 7.5°C in October to 13.5°C in May and 16.5°C in the summer. To reflect this, Table 22 includes both temperature preferenda (quoted relative to the corresponding acclimation temperature where possible) and/or avoidance temperatures. A later section elaborates on behavioural avoidance (Section 5.5).

Table 22 Summary of data on fish temperature preference (laboratory data except where indicated otherwise)

Species (Guild)	Common name	PREFERENCE		AVOIDANCE	
		Temp., °C	Source	Temp., °C	Source
Agonus cataphractus	Pogge				
Clupea harengus	Herring	Larvae – 16 Juveniles – 10–16 Adults – ≤10 (Thames)	Reid *et al.*, 1999 Power *et al.*, 2000		
Engraulis encrasicolus	Anchovy				
Gadus morhua	Cod	High food ration: 6.52 Low ration: 4.02 preferred habitat range: 0–12	Despatie *et al.*, 2001 Drinkwater, 2005	Juveniles: 9.3–16.0 (Depending on haemoglobin type)	Petersen & Jeffersen, 2003
Lampetra fluviatilis	River lamprey				
Leuciscus leuciscus	Dace			Juveniles: ∆T 8 above acclimation temp. (12)	Jacobs Engineering, 2008
Limanda limanda	Dab				
Liparis liparis	Striped sea snail				
Merlangius merlangus	Whiting	6–9	Loots *et al.*, 2011		
Osmerus eperlanus	Smelt			Juveniles: ∆T +4 above acclimation temp. (12)	Jacobs Engineering, 2008

Species (Guild)	Common name	PREFERENCE		AVOIDANCE	
		Temp., °C	Source	Temp., °C	Source
Salmo salar	Salmon	9–17	Jobling, 1981	23–24 Thames salmon successfully passed estuary	Alabaster and Gough,1986
				21.5 Thames migration ceased	Alabaster *et al.*,1991
				24.9 Axe migration ceased	Alabaster, 1989
				23.9 Frome migration ceased	Alabaster, 1991
				26.7 Dee migration ceased	Alabaster, 1990
				No avoidance below 19 except at low DO levels	Solomon & Sambrook, 2004
				Longest available dataset indicated no effect of temperature on entry into the Thames: controlled by DO only	Rosten *et al.* 2010
Salmo trutta	Sea trout /brown trout (AB)	8–17 16	Barton, 1996 Larsson, 2005	Adults: 20 Juveniles: 25	Alabaster & Downing, 1966; Cherry *et al.*, 1977. In Spigarelli *et al.*, 1983
Syngnathus rostellatus	Nilsson's pipefish	14.0–14.2	Power & Attrill, 2003		
Trisopterus luscus	Pouting	11.8			
Alosa alosa	Allis shad				

Species (Guild)	Common name	PREFERENCE		AVOIDANCE	
		Temp., °C	Source	Temp., °C	Source
Alosa fallax	Twaite shad	>21.5 (larvae)			
Anguilla anguilla	Eel	Acclimation ±3	Tongiorgi, *et al.,* 1986	Adults: ∆T +3 acclimation temp(12) Elver: no avoidance at up to ∆T +12 above acclimation temp. (9.5)	Tongiorgi *et al.,* 1986 Jacobs Engineering, 2008
Atherina presbyter	Sand smelt	11.2–13.5	Pombo *et al.,* 2005		
Chelon labrosus	Thick-lipped grey mullet				
Dicentrarchus labrax	Sea bass				
Dipturus batis	Skate				
Liza aurata	Golden grey mullet				
Liza ramada	Thin-lipped grey mullet				
Mullus surmuletus	Red mullet	16–19 (Mediterranean)	Maravelias *et al.,* 2006		
Petromyzon marinus	Sea lamprey	13.6 19.0 (17.8–21.8)	Reynolds & Casterlin, 1978; Holmes & Lin, 1994		
Platichthys flesus	Flounder				
Pleuronectes platessa	Plaice	16–17	Coutant, 1977		
Pomatoschistus microps	Common goby				
Pomatoschistus minutus	Sand goby	Summer – 16.6; Autumn – 7.5	Hesthagen, 1979	<4–6	Hesthagen, 1979
Psetta maxima	Turbot				
Sardina pilchardus	Sardine				

Species (Guild)	Common name	PREFERENCE		AVOIDANCE	
		Temp., °C	Source	Temp., °C	Source
Scyliorhinus canicula	Dogfish				
Solea solea	Dover sole				
Sprattus sprattus	Sprat				

Temperature preferences can also be sex-specific. Elasmobranchs and many flatfish often exhibit distinct and different temperature preferences in males and females, related to optimal conditions for egg/sperm development versus feeding (Swain and Morgan, 2001).

Understanding thermal preferenda in fish is important in that these figures determine to a large extent when and where species/life-stages will be found in TraC waters. For example, Hestagen (1970) reported a ULT for sand goby (*Pomatoschistus minutus*) of 31°C but an upper (summer) preferendum of 16.6°C. The author states that in the field (Oslofjord, Norway), sand gobies are not found at temperatures above 20°C. In autumn and winter the preferendum drops to around 7.5°C, and at temperatures below this gobies leave shallow areas of estuaries where there is a risk of cold temperature shock and move into deeper water. Juvenile bass exhibit similar autumnal emigrations from UK estuaries when the temperature falls below ~15°C, ceasing feeding at 10°C, and at temperatures below 5°C juvenile bass begin to die after a few weeks (Pickett and Pawson, 1994). These emigrations give rise to regular October/November peaks in impingement rates on power station screens (eg, at Oldbury, Fawley, Shoreham, West Thurrock and Kingsnorth power stations) as they migrate past the water intakes. This seasonal emigration is not seen in estuaries of south-west England such as the Tamar, in which higher winter temperatures are maintained by the Gulf Stream (Pickett and Pawson, 1994). Kingsnorth Power Station discharges its thermal effluent into a tidal creek. Juvenile and adolescent bass have been found to over-winter within the heated canal (Langford, 1983; Pickett and Pawson, 1994); the warm water effectively extending the bass's northern over-wintering geographic range. These observations are important since they demonstrate that for bass, movement out of the estuary is not an ecological requirement but a physiological imperative where winter temperatures are low (similarly for sole).

In a similar study, albeit as an indirect indication of behavioural avoidance, in assessing the estuarine conditions required to act as a water quality barrier and which fish may avoid, Pomfret *et al.* (1991) concluded that the flounder, *Platichthys flesus*, was likely to be absent from the upper areas of the Tyne and Forth estuaries when temperatures exceeded 14.5°C and DO was <7.5 mg l^{-1}. While these limits are considered respectively lower and higher than more recent evidence suggests for lethal limits, the values agree with those above for gobies and bass, and the study also illustrates the links between the water quality parameters of temperature and DO.

Attrill and Power (2004) describe an approach to looking at the relationship between thermal preferenda in estuarine fish and their occupation of estuarine habitat using the Magnuson *et al.* (1979) thermal resource model. Attrill and Power propose that within an estuary, partitioning of the thermal resource is primarily temporal rather than spatial. To explore this idea they undertook an analysis of fish catches over a 16 yr period (1977–1992) from the cooling water intake screens of West Thurrock Power Station, which is situated on the north bank of the Thames Estuary. By applying statistical models they were able to identify the median temperature 'used' by each species and the spread of values around this median, which they refer to as the 'thermal niche breadth' (median ±1 standard deviation). These are presented in Figure 12 for a selection of 16 fish species and a number of invertebrates. This provides a graphical summary of the thermal preferences of common British estuarine fish and crustacea. Further information on the feeding guilds shown in Figure 12 is given in Attrill and Power (2004). In Figure 13, the authors demonstrate a strong correlation between published experimental values for thermal preferenda and thermal niche values.
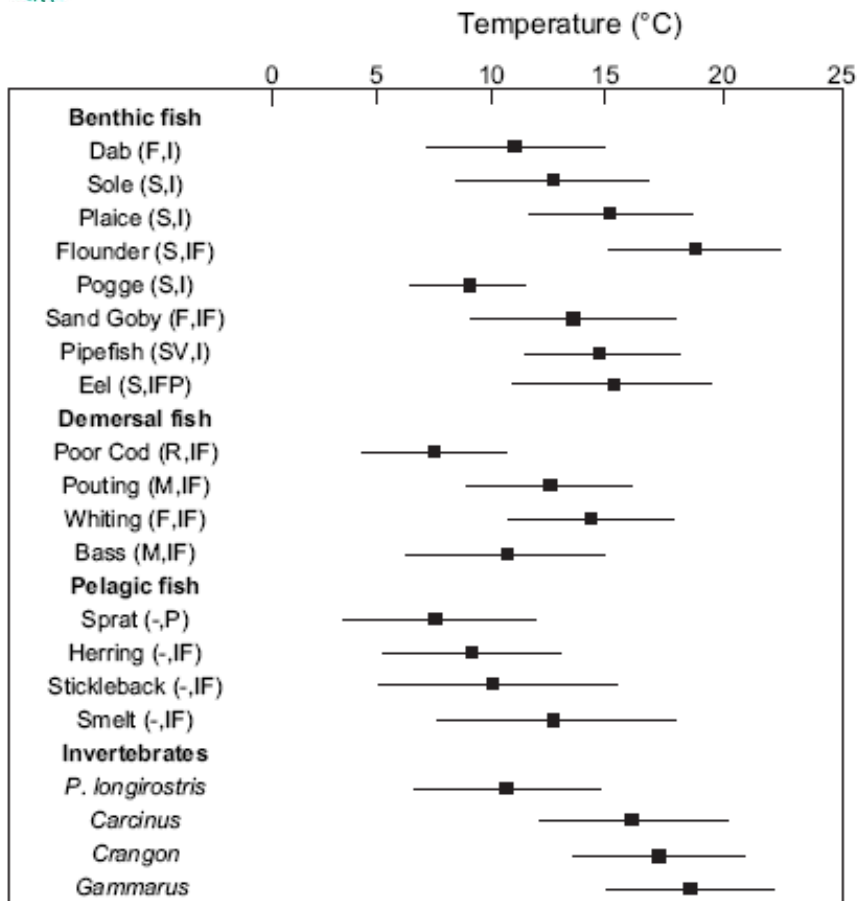
Figure 12 Habitat and feeding guild and thermal niche breadth characteristics of species sampled at West Thurrock (Attrill and Power, 2004)

Note: Thermal niche breadth (median ± SD) plotted as horizontal lines, with median temperature defined by black boxes. Species are grouped by vertical guild and then taxonomic similarity, with codes by each species name. Substratum guilds for non-pelagic species are defined as: S = sandy bottom for species living solely on sand, F = soft bottom for species living on sand, mud and/or fine gravel, R = rough bottom for species living on rocks stones and/or pebbles, M = mixed bottom for species that indiscriminately use any kind of substrate, and V = vegetation for species living amongst vegetation on a given substrate. Feeding guilds are defined in terms of dominant prey types as: I = invertebrates, F = fish, and P = plankton.
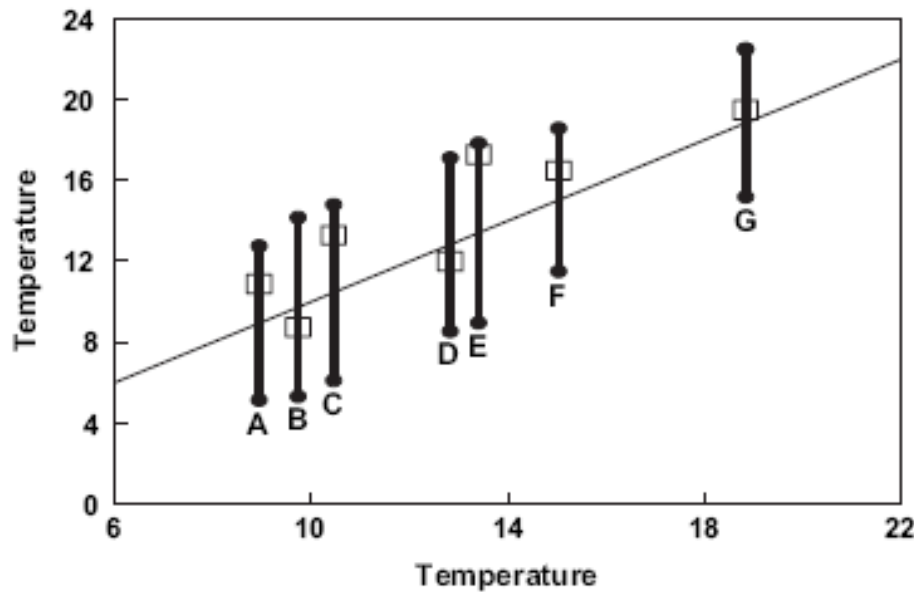
Figure 13 Plot of species thermal niche data (solid lines and circles) obtained from sampling at West Thurrock and literature-derived estimates of corresponding final temperature preferenda (Attrill and Power, 2004)

Note: The solid line defines the 45 line indicating the temperatures at which thermal niche median values (x-axis) and final temperature preferenda (y-axis) should coincide if identical. The correlation between the two sets of data was $r = 0.882$ (p-value = 0.005). Letters plotted below the thermal niche breadths define the species for which the niche breadth is plotted. A = herring, B = 3-spined stickleback, C = bass, D = sole, E = sand goby, F = plaice, and G = flounder.

A possible shortcoming of the Attrill and Power (2004) approach as a means of establishing thermal preferenda using power station data is that sampling was confined to a single point in the estuary (in the case of West Thurrock, this was mid-estuary). Taking the thermal niche shown for bass, for example, a median temperature of about 11°C, with a thermal niche breadth of 7.5–15°C is indicated (Figure 13). This relatively narrow and cool temperature range reflects the fact that juvenile bass pass the power station as they migrate into the estuary in spring/early summer and leave the estuary in autumn. They spend the summer months higher up the estuary out of the reach of the power station where summer temperatures commonly exceed 20°C. Elsewhere, bass will actively select areas of higher temperature, for instance in power plant outfalls. The approach might be better applied to multi-point sampling (eg, for a series of power stations, sampling from different points in the estuary). There is no reason why data should not be taken from power stations located on several European estuaries – indeed, this would be a better approach when investigating species requirements. In Britain, comparable long-term data have been collected at a number of the Severn-side power stations, and monthly data for three or more years have been collected from estuarine-sited power stations at Fawley (Southampton Water), Shoreham (Adur Estuary), Stallingborough (Humber Estuary) and Longannet (Forth Estuary) and possibly more. Nonetheless, Figure 13 implies a reasonable fit to laboratory data.

Given the large diurnal and tidal fluctuations of water temperature in the shallow margins of estuaries and coastal waters, it is clear that, whatever the preferred or acclimation temperatures of a given species and life-stage, individual fish have to cope with large short-term fluctuations in these environments. While they are protected to a large extent by their ability to behaviourally select preferred temperatures, it is also clear that some fish are well adapted to fluctuating environments. Section 5.9.1 refers to the improved growth of some species when in fluctuating rather than stable temperature conditions. In another demonstration of the natural ability of fish to adapt, Bulger (1984) showed that the saltmarsh fish *Fundulus heteroclitus* exhibited a daily rhythm of temperature tolerance which anticipated natural rhythmic fluctuations in temperature. When held in the laboratory, the physiological rhythm persisted for at least 30 days.

The use of data-storage tag technology offers unprecedented opportunities to investigate the thermal preferences of fish. Cod are known to be capable of moving large distances (*c.* 1,000 km) and hence could theoretically re-locate to anywhere in the North Sea. However, a study by Neat and Righton (2006) based on observations of the temperature experienced by 129 individual cod (using data storage tags) suggested that in the summer most of the individuals in the south experienced temperatures considered sub-optimal for growth. Cooler waters were easily within the reach of these cod, and a small number of individuals migrated to areas that allowed them to experience lower temperatures, indicating that the cod had the capacity to find cooler water. Most, however, did not, suggesting that the changing thermal regime of the North Sea might not yet be causing individual cod to seek cooler waters.

A larger study by Righton *et al.* (2010) used electronic tags to measure the thermal experience of 384 adult cod from eight different stocks in the north-east Atlantic. The data demonstrate that cod is an adaptable and tolerant species capable of surviving and growing in a wide range of temperate marine climates. The total thermal niche ranged from –1.5 to 19°C; this range was narrower (1–8°C) during the spawning season. Cod in each of the stocks studied had a thermal niche of approximately 12°C.

5.4.2   Thermal preferenda: summary

Thermal preferenda determine the thermal niche within a transitional or coastal water body within which fish will seek to spend most of their time. Areas occupied by thermal mixing zones could therefore represent lost habitat for some species, although this is rarely a large area. Also, cold-water fish may be able to occupy the area beneath a buoyant plume. On the other hand, thermal addition may extend the season for which juvenile fish are able to occupy marginal areas as they cool in autumn and winter.

A more important consideration is whether far-field temperature rises from effluents or global warming effects might push species living close to the upper limit of their thermal niche to move away or become impoverished. In the case of cold-water anadromous species such salmonids or smelt, it is possible, but by no means proven, that they would fail to penetrate areas where the temperature is raised above their thermal niche limit to reach spawning grounds. The outcome in such cases will be a function of, for instance, their ability to detect and avoid thermal changes, their motivation to spawn and their ability to swim though the affected area. Power station discharges may also create thermal 'stepping-stones', allowing warm-water species to penetrate new areas. These aspects are discussed in later sections.

5.5   Behaviour and migration

5.5.1   Thermal barriers: a reality?

Most laboratory work on fish behaviour in response to temperature choices has concentrated on determining thermal preferenda, and some avoidance of high water temperatures is implicit in these measurements. However, a more specific concern is as to how fish will behave when confronted with a step change or sharp gradient in temperature on meeting a thermal plume or far-field temperature rises caused by a thermal discharge. This is one of the most important regulatory aspects for consenting thermal discharges. In relation to European sites, WQTAG160 (WQTAG, 2006) takes a precautionary approach and *assumes* that fish will be reluctant to pass through thermal plumes:

'Migratory fish species will be an important element of estuarine habitat structure and functioning, and may even be cited as specific interest features in their own right. They require a significant "window of opportunity" to migrate upstream. The mixing zone is likely to be surface-biased but should not form a barrier to migration across the whole estuary, or block areas of the estuary through which fish are known to pass.'

Blockage by thermal plumes appears to be an intuitive rather than observed concept. The reasoning is that salmonids (the main migratory species of interest in past studies) are cold-water stenotherms and therefore avoid warm water. In fact, fish tracking studies carried out in rivers and estuaries in the United Kingdom and elsewhere in the world do not provide any clear evidence of thermal barriers (Langford, 1990), and where evidence suggests possible effects it has been confounded by other issues such as changes in – or absolute levels of – freshwater discharge and the levels of dissolved oxygen. As a cold-water group, salmonids are generally regarded as a sensitive indicator group for thermal concerns, so standards suitable for salmonids would also be protective of warm-water species such as eel, lamprey and shad.

A lesser known issue concerns juvenile migrations through estuaries. A common mechanism by which young fish undertake migrations in transitional waters is known as selective tidal stream transport (STST). STST was first demonstrated in plaice (*Pleuronectes platessa*) by Greer Walker *et al.* (1978) as a means by which flatfish can save energy by using the tide when it is flowing in the desired direction, whilst sitting out the reverse tide on the bed, and it has since undergone further research (eg, Hunter *et al.,* 2004). The process is one of vertical modulation of movement between slow-moving water at the bed and faster moving water above or to one side. STST has since been demonstrated for many other species; it is known to be important within the Thames Tideway as a fish dispersal mechanism (Naismith and Knights, 1988; Colclough *et al.*, 2002). The phenomenon is seen regularly during the summer months when a 'ribbon' of small fish such as dace (*Leuciscus leuciscus*), flounder (*P. flesus*), elvers (*Anguilla anguilla*) and smelt (*Osmerus eperlanus*) occupies the very shallow water margins of the channel. During migration, they may be heading either upstream or downstream, depending on species and habitat conditions. For example, in the case of elvers, the migration is towards freshwater, where eels spend most of their life growing to maturity. For other species the purpose may be simply to disperse the population over the available habitat. A recently proposed consenting of a thermal discharge on the Thames Tideway at Battersea raised questions about whether intrusion of the plume into shallow marginal areas might cause a barrier to juvenile migrations. To resolve such questions, it would be necessary to show (a) that the plume did not impinge on the intertidal foreshore, (b) that the temperature rise was not sufficient to cause a barrier or (c) that there were sufficient remaining migration paths or temporal windows of opportunity to ensure that the fish could pass. Until recently there was no scientific information on thermal avoidance thresholds in these life-stages, and the argument therefore hinged on opportunity. In considering this question, it should be recognized that fish living in such marginal areas will experience substantial temperature changes that occur naturally there.

5.5.2   Thermal avoidance: experimental studies

Experimental studies provide the option of controlling factors other than temperature and are therefore particularly helpful as an aid to interpreting field observations where other factors may come into play. A number of studies were carried out by Jacobs Engineering and its predecessors at Fawley Aquatic Research Laboratories (Southampton) (Wood and Turnpenny, 1994; Clough *et al.*, 2002; Jacobs Engineering, 2008).

The proposed construction of Shoreham Power Station (West Sussex) in the late 1990s led to a series of investigations into potential effects of the thermal plume. One concern was that sea trout (*Salmo trutta*) smolts exiting from the River Adur estuary might be deflected by the thermal plume and put at risk of delay or increased predation. Experiments were conducted with actively migrating wild sea trout smolts from the River Adur using saline water (salinity of 16) (Clough *et al.*, 2002). The experiments involved offering the smolts a choice between two parallel confluent streams of water, one at ambient temperature (*c.*11°C), and the other at a raised temperature (ΔT = <+1 to +8–10°C). The temperature of the warmed stream was raised in increments and the position of the heated side was reversed at each step. The median avoidance threshold was found to be +6°C, although in one experiment smolts were found to avoid temperatures rises of <1°C. This appeared to occur because the lead fish in the group, which may have been more sensitive, caused others to follow. The trials were conducted in a mobile trailer apparatus, allowing the work to be carried out on site at Shoreham using the actual dilution water.

Similar experiments, using the same trailer apparatus, showed that salmon (*Salmo salar*) smolts given a choice of water at ambient (16–17°C) or water with temperature raised between 0.3°C and 7°C, consistently chose the cooler water once the ΔTs exceeded 4°C (Wood and Turnpenny, 1994). Brown trout (*S. trutta*) in the same trials were found to be more sensitive to temperature changes, responding to a ΔT of only +1°C, similar to the more sensitive sea trout group reported by Clough *et al.* (2002). These experiments were conducted in freshwater.

The experimental work was extended on behalf of RWE Npower to include study of transitional water juveniles, including dace (*L. leuciscus*), smelt (*O. eperlanus*), elver (*A. anguilla*), flounder (*P. flesus*) and common goby (*Pomatoschistus microps*) (Jacobs Engineering, 2008). Given the choice of a base temperature averaging 12.3°C (9.5°C in the case of eels), or water incrementally raised by 2–12°C, only juvenile smelt and dace exhibited an avoidance reaction, initially observed at a ΔT of +4°C and +8°C, respectively, relative to the base temperature. Neither eel, goby nor flounder showed any significant response to the increasing temperature. In a separate study, Tongiorgi *et al.* (1986) demonstrated a clear preference of eels for water with a ΔT ≤ +3°C (though given the choice between temperatures 3°C lower or higher than acclimation, the lower was always selected).

5.5.3   Field studies on salmon migration

The tidal reaches of the Scottish Dee are characterized by a very stable and sharply defined salt wedge. In the River Dee, *S. salar* remained in the sea when water temperature exceeded 20°C rather than entering rivers (Hawkins, 1989). When salmon are running upstream, the freshwater side has a temperature of 18–20°C and the saltwater 7–10°C; the transition is across a matter of centimetres. Approximately 6 radio-tagged salmon were tracked, and of these most stayed in the salt wedge and followed it down to the bed, where bed roughness may have blurred the gradients, before they crossed into freshwater. However some of the salmon simply swam straight through the wedge front, seemingly oblivious of a 9–10°C temperature increase (A.D. Hawkins, pers. comm., in Coughlan, 1983). This strengthens the view from laboratory trials cited in Section 5.5.2 that salmonids are rather insensitive – or perhaps unresponsive – to sharp temperature fronts. Motivation is likely to be a key factor affecting the willingness of fish to cross thermal gradients or boundaries in field situations.

In tidal reaches of urbanized rivers, low DO levels can inhibit upstream salmon migration, and in some cases observed effects may relate to this factor rather than, or as well as, temperature. As indicated above, such effects are also confounded by current speed, based on river flows, and DO (Priede *et al.*, 1988). Extrapolated or actual data from several rivers showed that migration ceased at weekly or monthly mean temperatures of:

21.5°C	River Thames, UK	Alabaster *et al.* (1991)
24.9°C	River Axe, UK	Alabaster (1989)
23.9°C	River Frome, UK	Alabaster (1991)
26.7°C	River Dee, UK	Alabaster (1990)

Alabaster *et al.* (1991) suggest that the differences are not a racial or genetic effect but, rather, are because the concentrations of dissolved oxygen differ. For the Thames, their data suggest that a 1 mg l^{-1} fall in DO might be equivalent to a 4°C rise in temperature with regard to influence on salmon return. In a more recent study, which included a much more extensive set of Thames salmon return data from 1991 to 2005, Rosten *et al.* (2010) examined the relationship between river entry and levels of DO and water temperature using a general linear modelling technique. They found no statistically significant relationship with water temperature, explaining variation in the context of DO alone. Data suggested a mean annual DO requirement of 7.4 mg l^{-1} would be required to achieve the Thames Salmon Action Plan targets.

The figure of 21.5°C given by Alabaster *et al.* (1991) for the Thames appears also to be too low when compared with the earlier Alabaster and Gough (1986) data, which refers to observations made in 1984. These authors note that in August 1984, when most fish were caught (having succeeded in

their passage through the estuary), the water temperature was never lower than 19°C, and there was a length of estuary of at least 20–40 km where it exceeded 22°C. The data presented in their paper do not allow direct inference of the precise distribution of temperatures above 22°C, but Norris (2001) gives mean temperatures along the estuary for August 1984 as shown in Table 23.

Table 23 Mean temperatures along the Thames (Norris, 2001)

Location	Temperature °C
Belvedere	23.4
Littlebrook	22.8
West Thurrock	22.1
Northfleet	21.8
Tilbury	22.2

Since these data are based on power station intake records which are effectively continuous, taking into account the tidal and diurnal variation in temperature occurring, it suggests that salmon successfully migrating in August 1984 would have experienced long stretches of the estuary at temperatures in excess of 23–24°C in combination with the low DO concentrations (<5 mg l^{-1}) that prevail in the middle Tideway throughout August.

There have been several other studies of the patterns of entry of adult salmon into UK rivers. Most have been based on monitoring of fish traps, electronic fish counters or biotelemetry studies. Some studies refer to the relationship between entry time and water temperature and suggest that high temperatures may inhibit fish entry into freshwater. However, in practice it has been difficult to establish a causal link with temperature owing to strong inter-correlations not only with DO but also with other factors, including freshwater discharge, temperature and salinity.

Clarke *et al.* (1994) studied the movements of 260 salmon fitted with radio or CART (Combined Acoustic and Radio Transmitting) tags following release into the Tywi Estuary (Carmarthen Bay, South Wales) between the years 1988 and 1990 (see Figure 14). These years include wet and dry summers, yielding a range of temperature and water quality conditions.

R. Tywi Salmon Entry in Relation to Water Temperature
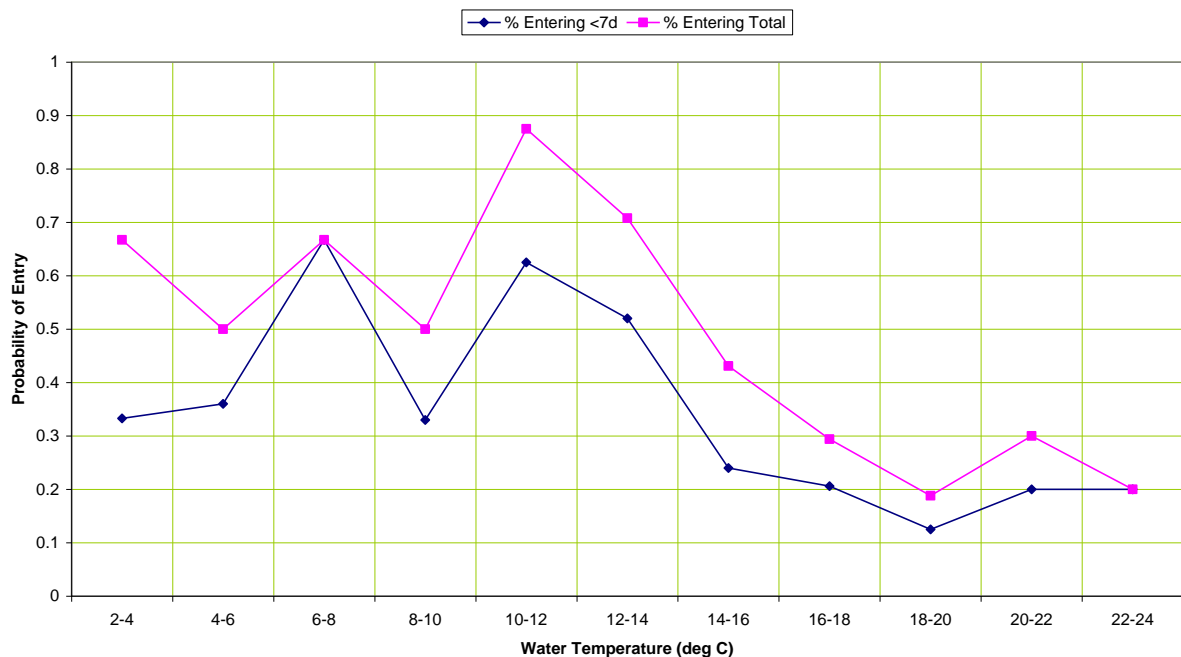
Figure 14 Probabilities of tagged salmon entering the River Tywi relative to the median temperature measured in the estuary during the first 7 days after tagging (Clarke *et al.*, 1994)

The Tywi Estuary is susceptible to DO sags (hypoxia) during the summer months, when DO levels are observed to dip below 5 mg l^{-1} and occasionally below 1 mg l^{-1}. Success of entry into freshwater was positively correlated with freshwater flow and DO and was negatively correlated with temperature. Figure 14 shows the relationship between fish entry and water temperature. The authors state that '*the scatter plot demonstrates no clear pattern between probability of entry and temperature below 14°C but a consistent decrease in the probability of entry was observed when the median temperature exceeded 14–16°C*'. It is likely, however, that the temperature relationship shown here is caused by a combination of high temperature and low DO and possibly other factors.

The results of similar salmon tracking studies reviewed by Solomon and Sambrook (2004) for estuaries in South West England include cases less susceptible to low oxygen, where effects may be more closely related to temperature. Of particular interest is a study carried out on the River Avon, which enters the sea at Christchurch (Dorset). The sample size of tagged fish (n = 437) was nearly twice that used on the Tywi (n = 260). The relationship between successful river entry and water temperature is shown in Figure 15. The authors state 'there is a suggestion of a plateau between about 13°C and 19°C, which includes the great majority of fish'. This is notably higher than the 14–16°C point at which successful salmon entries started to fall off in the Tywi study and may reflect the lesser involvement of hypoxia avoidance in the Avon. The authors state that DO levels were generally high in the Avon and 'unlikely to have a major effect on migration'. Even in this case, however, DO levels down to 2.4 mg l^{-1} were reported, albeit in a saline sump overlain by freshwater, with DO levels of the order of 7 mg l^{-1}. Hence the data may not be a completely reliable indicator of thermal avoidance. Nonetheless, the data suggest that the *onset* of thermal avoidance in salmon probably does not occur below a value of 19°C. Neither this study nor the Tywi study demonstrates an unequivocal causal relationship with temperature.

In the past, the potential significance of migratory fish turning back as a result of water quality barriers or insufficient freshwater flow may have been underestimated. It has generally been assumed that fish that were held up would stay longer in the outer estuary and make the attempt to ascend into freshwater when the conditions were more suitable. Both the Tywi study (Clarke *et al.*, 1994) and the

South West English rivers study (Solomon and Sambrook, 2004) showed that the probability of fish released into the estuary entering freshwater declines after 7–10 days (ie, if conditions are not suitable for their ascent into freshwater within this time, they may not enter at all). Solomon and Sambrook explain this in terms of 'missed physiological opportunity'. Their hypothesis proposes that upon entering the estuary, the fish prepare themselves physiologically for entry into freshwater and that they have a 'window of opportunity' in which they can successfully make this transition. If they fail to do so, then they might be physiologically compromised by remaining in seawater. They cite observations from the Tamar and Exe estuarine net fisheries that in years when returns to the rivers are poor, larger catches of lethargic fish with soft flesh are found and that the quality of the fish renders them barely marketable. Similar anecdotal observations have been made on the Thames Estuary (D. Clifton-Dey, pers. comm.).

When considering temperature as a potential barrier to river entry, the seasonality of migrations has to be taken into account. Compared to summer migrations, spring or autumn runs are less likely to be affected by far-field temperature rises, owing to the lower background temperatures. DO levels also tend to remain higher at these times of the year due to lower microbial activity.
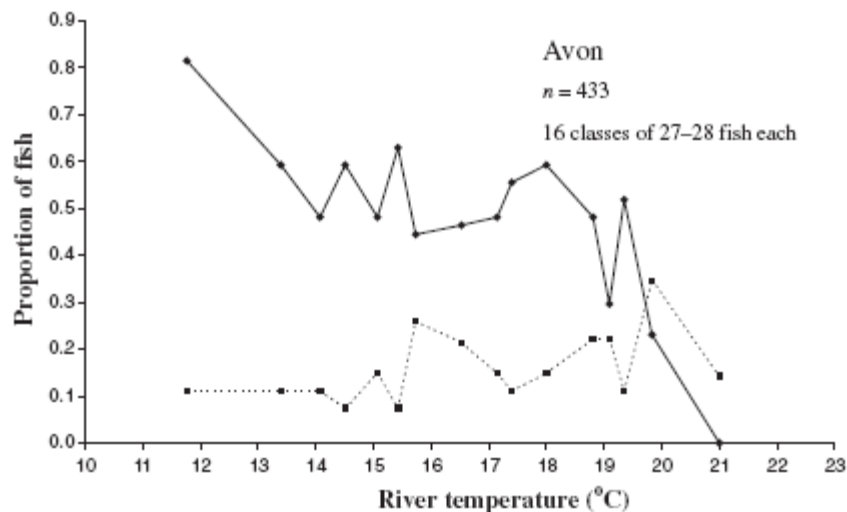
Figure 15 The relationship between river temperature on the day of tagging, and the proportion of tagged fish that entered the river within 10 days of tagging (solid line) and after more than 10 days (dashed line), for fish tagged at the mouth of the Avon Estuary. Each point represents the mean temperature and proportions for a group of 27 or 28 fish (Solomon and Sambrook, 2004).

5.5.4   The Environment Agency salmon migration model

Salmon stocks in the Rivers Test and Itchen which enter the sea via Southampton Water have reached critical levels in recent years, leading to the designation of the River Itchen as an SAC for Atlantic salmon. To reach these rivers from the English Channel and Solent, ascending adults must pass a series of power plant, chemical works and refinery thermal discharges. On the premise that the thermal discharges might inhibit the return of salmon to freshwater, the Environment Agency commissioned a salmon migration model (Seaby, 2003). The model considers the extent of intrusion across the estuary of the thermal plume (defined here as the area >+0.5°C above ambient background temperature). The model uses the Clarke *et al.* (1994) River Tywi data to estimate the probability that salmon will pass through the heated area, comparing this with the probability of them passing through the unheated area. To allow prediction of salmon probability of entry at temperature increments of 0.5°C, the WRc (Water Research Council) were asked by the Environment Agency to employ general linear modelling techniques, with the constraint that at temperatures less than or equal to 12°C, the probability of river entry was a constant (Figure 16).
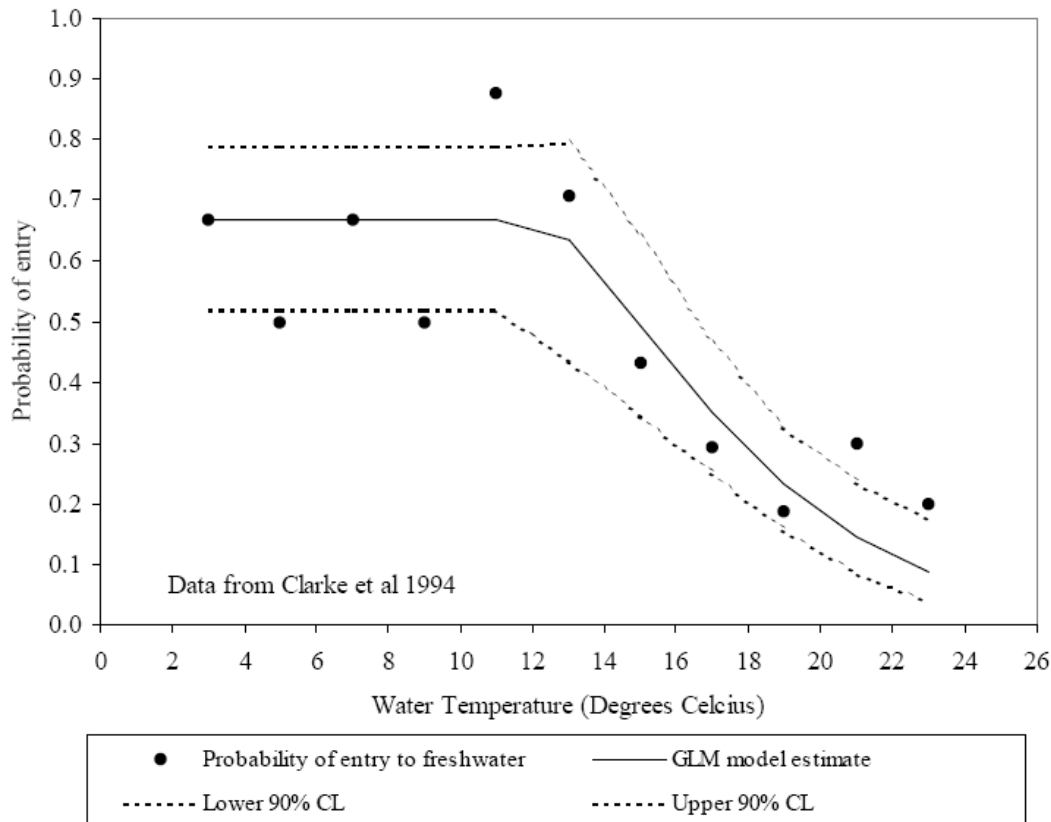
Figure 16 The General Linear Model of probability of river entry (data from Clarke *et al.*, 1994).

The model is summarized in the flow diagram (Figure 17). The model makes the assumptions on the behaviour of salmon listed below:

➢ salmon do not avoid thermal plumes;

➢ salmon migrate in the surface water layer of the estuary;

➢ salmon are equally spread across the width of the estuary;

➢ salmon only attempt to migrate once into an estuary.

The thermal relationship between salmon migrating and migration is similar in the estuary to that in River Tywi.

While there is a need to simplify for modelling purposes, the validity of some assumptions is questionable. As we have established, fish do preferentially select areas of warmer or colder water, and this point challenges the first assumption. Fish are also unlikely to be uniformly distributed across an estuary. Thus, when and where more detailed information becomes available it would be desirable to incorporate it into the model. Moreover, in view of the preceding discussion on the validity of interpreting the Tywi data in terms of temperature effects alone, the robustness of the final assumption is questionable and a relationship less influenced by DO effects would be more appropriate.

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┌─────────────────────────────┐
│  No. of fish arriving in     │
│  estuary per day             │
└──────────────┬──────────────┘
               │
               ▼
┌─────────────────────────────┐
│  Effect of temperature on    │──────────┐
│  upstream migration          │          │
└──────────────┬──────────────┘          │
               │                          ▼
               │              ┌─────────────────────────────┐
               │              │  Percentage of estuary       │
               │              │  affected by +0.5°C thermal  │
               │              │  plume                       │
               │              └──────────────┬──────────────┘
               │                             │
               │                             ▼
               │              ┌─────────────────────────────┐
               │              │  Effect of estuary           │
               │              │  temperature + 0.5°C on      │
               │              │  upstream migration          │
               │              └──────────────┬──────────────┘
               ▼                             ▼
┌─────────────────────────────┐ ┌─────────────────────────────┐
│  Number of salmon entering   │ │  Number of salmon entering   │
│  river without effect of     │ │  river with effect of        │
│  thermal plume               │ │  thermal plume               │
└─────────────────────────────┘ └─────────────────────────────┘
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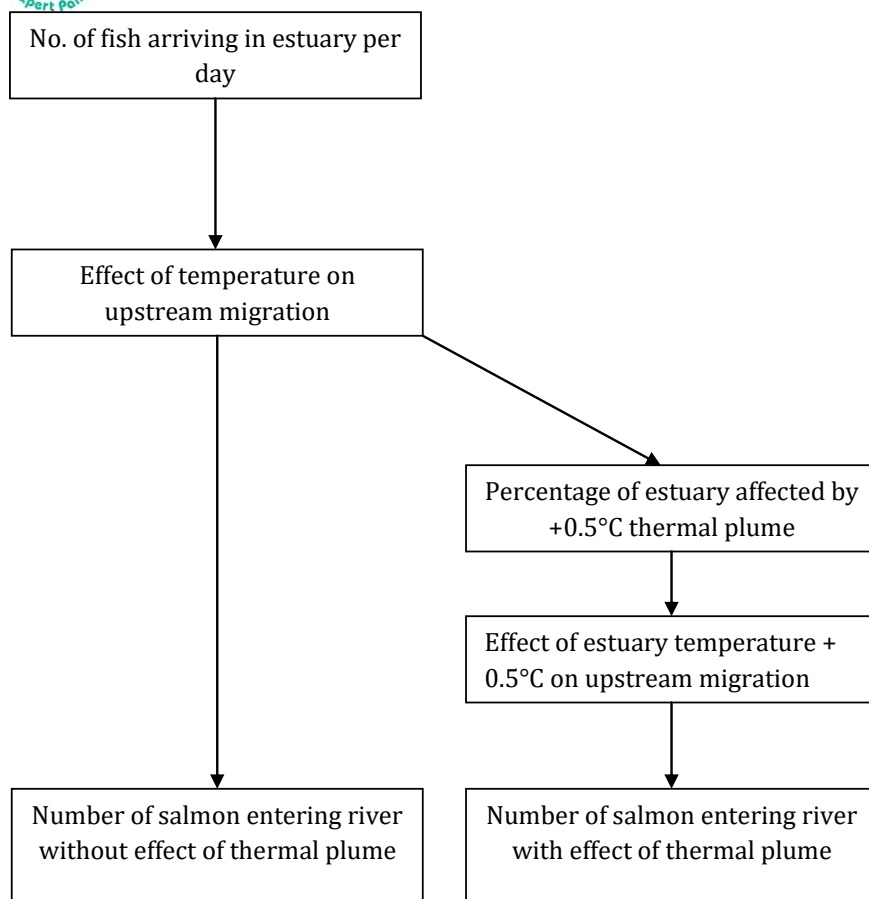
Figure 17 Flow chart of the Environment Agency model for assessing the effect of estuarine temperature on river entry of salmon (Seaby and Fewings, 2003)

5.5.5 Temperatures triggers for fish migration

Reported temperatures in transitional waters that appear to trigger various types of fish migration are listed in Table 24. These can be divided into migration for behavioural thermoregulation, migration for spawning into or through transitional waters by diadromous species such as smelt and shad, and entry of elvers into rivers. The former includes the already mentioned winter exodus of warm-water species such as bass, which risk death if they remain in cold estuarine waters (below 15°C) overwinter (Pickett and Pawson, 1994), and the exodus of cold-water species such as dab, which avoid parts of the North Sea during winter when it drops below 2.5°C (Bolle *et al.,* 1994).

Some weakly migratory species such as sole indicate little relationship between the timing of spawning and seasonal temperature changes (Horwood, 1993); however, species such as Pacific herring, which have temperature-dependent gonadal development (Ware and Tanasichuk, 1989), have been demonstrated to be directly affected by climatic changes. Sims *et al.* (2004) showed that the timing of flounder spawning migration from the Tamar Estuary to the English Channel was not fixed, but occurred around 1–2 months earlier when sea temperatures were lower by up to 2°C. This is likely to be because of the need to avoid low temperatures outside the optimal range and outside the range for gonadal development.

Claridge and Gardner (1978) also suggest that the seaward migration of juvenile shad occurs at river temperatures of <19°C, but this may be less of a trigger than the observed condition at the time in late summer when they move seawards.

Costa *et al.* (2002) found some inconsistency in reported effects of temperature on elver migration through estuaries. In the Thames Estuary, a threshold temperature of 14–16°C has been found to be the best predictor of elver migrations (White and Knights, 1997). However, research in continental

Europe led to the conclusion that a temperature differential between seawater and freshwater is more likely the trigger, or that it has no effect at all (Costa *et al.,* 2002).

The onset of seasonal migration is rarely as straightforward as an instant shift in temperature acting as a trigger (Comeau *et al.,* 2002). A computational analysis by Comeau *et al.* (2002) to understand likely proximate cues of migration for Atlantic cod suggested that a shift in autumn migration timing during the late 1980s to early 1990s was not initiated by a specific temperature or temperature regime over the summer. Instead, a widespread cooling of the bottom waters preceding the change in migration by at least one or two years was linked to the earlier migration. Another cause was suggested to be an abundance of their principle prey (herring and capelin), which led to a build up of energy reserves over the summer and thus an earlier readiness to engage in the autumn migration.

Any warming of transitional or coastal waters might be expected to affect the timing of when these migrations occurred; for those species that emigrate from estuaries overwinter, it might reduce the length of time that they are forced into deeper water, or eliminate it altogether as has been found recently in some bass habitats. For those catadromous species that depend on temperature for spawning migration such as flounder, warmer temperatures are likely to lead to later migration to the marine spawning grounds, with less physiological urgency to move from colder shallower rivers to the deeper warmer seas. On the other hand, for anadromous species such as smelt, warmer temperatures are likely to encourage earlier onset of migration for spawning.

Table 24 Summary of temperatures for triggering fish migration

Species (Guild)	Common name	Temp., °C	Type	Source
Agonus cataphractus	Pogge			
Clupea harengus	Herring			
Engraulis encrasicolus	Anchovy			
Gadus morhua	Cod		Temp. not trigger for migration	Comeau *et al.*, 2002.
Lampetra fluviatilis	River lamprey	< 12–16	Autumn upstream	Abou-Seedo & Potter, 1979
Leuciscus leuciscus	Dace			
Limanda limanda	Dab	2.5	Spawning / avoidance of winter temps	Bolle *et al.*, 1994
Liparis liparis	Striped sea snail			
Merlangius merlangus	Whiting			
Osmerus eperlanus	Smelt	> 5	Spawning	Hutchinson & Mills, 1987
Salmo salar	Salmon	10–12		Solomon, 1978
Salmo trutta	Sea trout /brown trout	10–12		Solomon, 1978
Syngnathus rostellatus	Nilsson's pipefish			
Trisopterus luscus	Pouting			
Alosa alosa	Allis shad	11	Spawning	Acolasa *et al.*, 2006
Alosa fallax	Twaite shad	10.6–12.3 < 19	Spawning Juvenile seaward	Aprahamian, 1988 Claridge & Gardner, 1978
Anguilla anguilla	Eel	14–16 (max. 18–20) 9–18	Elver migration through estuary	White & Knights, 1997 Vøllestad *et al.*, 1986
Atherina presbyter	Sand smelt			
Chelon labrosus	Thick-lipped grey mullet			
Dicentrarchus labrax	Sea bass	< 15	Marine overwintering	Pickett & Pawson, 1994

Species (Guild)	Common name	Temp., °C	Type	Source
Dipturus batis	Skate			
Liza aurata	Golden grey mullet			
Liza ramada	Thin-lipped grey mullet			
Mullus surmuletus	Red mullet			
Petromyzon marinus	Sea lamprey			
Platichthys flesus	Flounder	10–12.2	Seaward migration for spawning, Feb-Mar	Sims *et al.*, 2004
Pleuronectes platessa	Plaice			
Pomatoschistus microps	Common goby	5–7		Jones & Miller, 1966
Pomatoschistus minutus	Sand goby	5–7		Jones & Miller, 1966
Psetta maxima	Turbot			
Sardina pilchardus	Sardine			
Scyliorhinus canicula	Dogfish			
Solea solea	Dover sole	10–12		Dorel *et al.*, 1991
Sprattus sprattus	Sprat			

5.5.6 Effects on behaviour and migration: summary

Changes in the thermal regime of a transitional water body can change the behaviour of fish, with the potential of causing them to enter or leave at different times of the year. Changes in the timing of spawning migrations could have positive or negative effects on the population, for example, depending on any critical matching of the timing with other factors in the water body.

The idea that fish migrations may be blocked by thermal barriers is widely held, and thermal standards generally target the issue of thermal blockage of salmonid migrations. In reality, laboratory and field observations indicate that salmonids are rather insensitive to step changes in temperature, often willingly crossing ΔTs of several degrees Celsius, although most of these observations have been made at temperatures well below ULT values. With projected temperature rises over the next century as a result of climate change, metabolic headroom will be reduced and the combination of higher background temperatures, added to local temperature elevations around power stations, could become critical for migration of cold-water species, especially in the south of England.

Evidence from adult salmon tracking studies through estuaries is now abundant, along with data gathered from traps located above the tidal limit. A limitation in using this type of data to infer thermal barriers is that temperature is strongly inter-correlated with other factors such as river discharge and DO concentration, obscuring the causes of river-entry failures. At best, adult salmon have been shown to enter UK rivers at temperatures in the mid-twenties degrees Celsius; at worst, evidence may be taken to suggest that the probability of successful entry begins to decline at values above 12–14°C. These differences may be partly attributable to freshwater flow effects but are probably best explained in terms of a combination of high water temperatures and low DO values. Taking the relationship given by Alabaster *et al.* (1991) whereby a decrease of 1 mg l^{-1} in DO is equivalent to a temperature rise of 4°C, a decrease of, for example, 3 mg DO l^{-1} below the avoidance threshold of around 5 mg DO l^{-1} would account for the approximately 12°C disparity between the reported upper and lower thresholds of alleged thermal blockage.

With regard to juvenile fish that use the water margins of estuaries for selective tidal stream transport, a relatively large degree of thermal tolerance is to be expected in fish that use this naturally unstable thermal habitat. Laboratory evidence indicates that this is so, with no evidence of plume avoidance at temperature increments of <+3°C in any species so far tested, including cold-water stenotherms such as salmonids, sea trout and dace, and apparent indifference of some species to increments well above this value.

Modern developments in data-storage tag technology are providing new opportunities to study behavioural responses to the fish's thermal environment over large geographic ranges and timescales. Studies with cod show that they do not necessarily maximize their advantage by always seeking optimal thermal niches.

5.6 Fish swimming performance

5.6.1 Relevance of fish swimming performance

Loss of swimming performance is likely to harm fish populations ecologically in a number of ways, notably in acts of predation or avoidance of predation, and in migration against tidal or river flow. Migrational aspects assume particular importance in heavily modified estuaries such as the Thames, where progressive urban encroachment and bridge construction has narrowed the fluvial channel presenting an increasing challenge to small and weak-swimming fish. Swimming performance is also a factor in the ability of fish to avoid being drawn into water abstractions. In estuaries, these are mostly associated with CW intakes of power stations and refineries, but, in the future, desalination plants may also have an impact (Turnpenny and O'Keeffe, 2005).

5.6.2 Temperature effects

Fish swimming performance within a species is determined primarily by the length of the fish, but water temperature is probably the key factor that determines performance on a day-to-day basis (Beamish, 1978; Turnpenny and Clough, 2006). Stenothermal temperate fish such as the salmonids reach peak performance at temperatures of 15–20°C, whereas eurythermal species such as the eel (*A. anguilla*) peak at 25–30°C. Beyond this point, swimming performance again declines. At temperatures close to freezing, warm-water species such as cyprinids become torpid.

The effect of temperature on aerobic swimming can be demonstrated using Q_{10} (respiratory quotient) values, which determines the rate of increase in tail beat frequency for every 10°C temperature increase. Videler and Wardle (1991) looked at the Q_{10} (10°C) for muscle contraction time in cod at temperatures of 2–15°C and from regression analysis estimated the value $Q_{10} = 2.06$. This indicates an approximate doubling of speed for every 10°C temperature increment. They cite other work in which a muscle contraction Q_{10} of 2 has been found for sculpin (*Myoxocephalus scorpius* L.) and in amphibia (*Rana*, *Xenopus*). A similar value is also borne out, for example, in the critical swimming speed measurements for sand-smelt (*Atherina presbyter*) reported by Turnpenny and Bamber (1983), in which aerobic swimming speed increased from a median value of 2.70 body lengths per second (bl s^{-1}) at 5.9°C to 5.73 bl s^{-1} at 15.0°C.

Above an optimum value, physiological limitations come into play. Regression models fitted to swimming speed data often use logarithmic transformations of the temperature coefficient, which emulates the decline in the Q_{10} as the optimum is approached (but not the fall in absolute swimming speed once this point is exceeded). The sand-smelt data given by Turnpenny and Bamber (1983) illustrate this point (Figure 18).
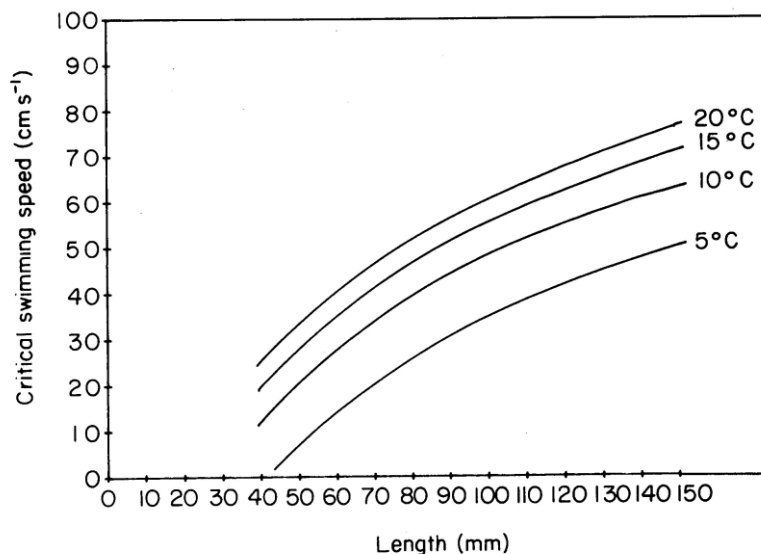
Figure 18 Critical swimming speed of the sand-smelt in relation to body length and temperature (Turnpenny and Bamber, 1983)

For warm-water species, optimum temperatures for swimming will be well above 20°C and therefore performance is unlikely to be limited severely by high temperatures in British estuaries. Stenothermal estuarine fish species whose optima are significantly lower than this notably include smelt (*O. eperlanus*) and salmonids. Figure 19 shows that smelt endurance (aerobic) and burst (anaerobic) swimming speeds measurements were higher at temperatures below 12°C than at temperatures above this value (Clough *et al.*, 2003).
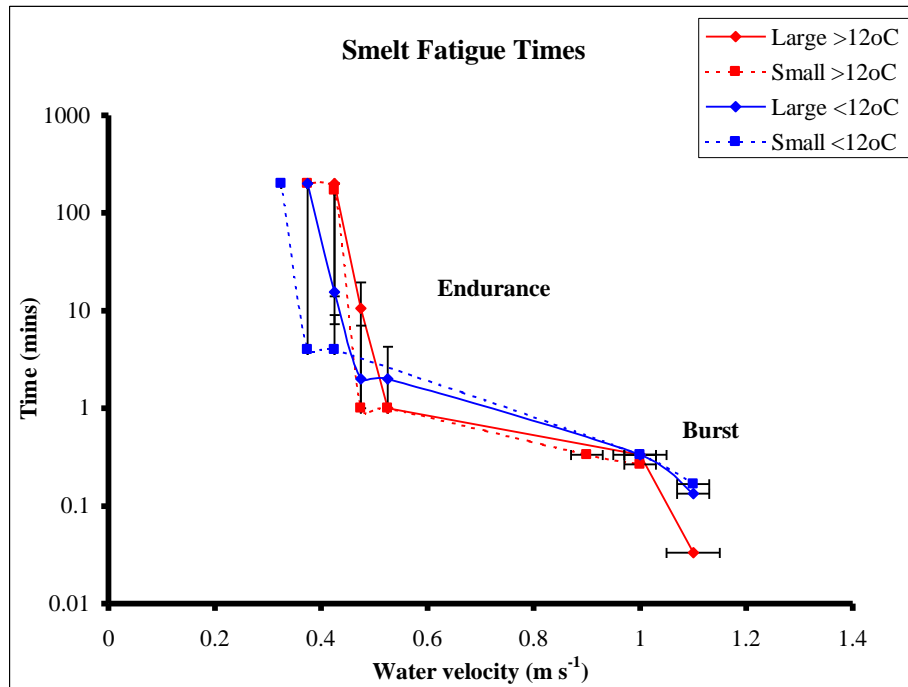
Figure 19 Smelt fatigue times in relation to water velocity and temperature

In Figure 19, vertical error bars on median endurance swimming data points represent upper and lower quartiles. Horizontal error bars on mean burst swimming data points represent 95% confidence limits (Clough *et al.*, 2003).

Sustained and burst swimming speeds of a variety of freshwater and migratory fish species have been investigated as part of an Environment Agency research programme, and data are accessible via the SWIMITv3.2 computer programme (Environment Agency R&D Project W2-029, "Swimming Speeds in Fish: Phase 2", 2003). This gives burst and sustained swimming speeds in relation to fish length and water temperature.

5.6.3 Effects on swimming performance: summary

Swimming performance affects the ability of fish to feed, avoid predation, avoid entrainment at water intakes and migrate. Water temperature strongly affects fish swimming performance, inducing torpor in warm-water species at low temperature and limiting both aerobic and anaerobic performance at high temperatures above the thermal preferendum. Swimming performance is optimal close to or at the preferendum. The effect of increased warming may therefore improve performance at lower seasonal temperatures – and conversely in warm summer conditions – although in Lusitanean species such as bass, summer warming may still push performance towards rather than away from the optimum.

The effects of temperature on fish migration through transitional waters may be less important than in rivers owing to the bidirectional tidal flow and the ability to use STST. Where thermal discharges or global warming cause long-term temperature increases, an overriding factor is likely to be the behavioural tendency of fish to seek out their preferred thermal niche.

5.7 Spawning and early life history

5.7.1 Temperature preferences for spawning

Effects of thermal plumes on spawning in the open sea are likely to be minimal, owing to fish mobility. Herring are a possible exception, being coastal gravel substratum spawners, with adhesive eggs that are deposited at preferred spawning locations (Wheeler, 1969). Thermal discharges into benthic spawning areas are normally screened out during the early stages of project environmental assessment.

Cefas Fish Sensitivity Maps (Coull *et al.*, 1998) give a summary of fish spawning and nursery areas in British waters and of spawning seasons. These provide useful information on which species are coastal spawners and therefore could be affected by thermal discharges, and those that spawn offshore and are therefore out of reach of discharge effects. Thus, while cod spawning, for example, is known to be sensitive to water temperature (Drinkwater, 2005), cod spawn mostly offshore. Known close-inshore spawning areas of cod are restricted to the east coast of Ireland and parts of the south-west of England. Other gadoid (cold-water) species also spawn mainly away from the coast. Inshore spawners, which are more potentially at risk, include mackerel (*Scomber scombrus*) and flatfishes such as Dover sole (*Solea solea*), lemon sole (*Microstomus kitt*), plaice (*P. platessa*) and flounder (*P. flesus*).

Estuarine spawners, or populations using affected harbours or marinas, are more at risk from thermal discharges, owing to the reduced rate of dilution. This can lead to a gradual build-up of background temperature. In intertidal mudflat areas, and adjacent subtidal areas that receive drainage from mudflats, this will add to already high peak summer temperatures. Future baseline temperature increases caused by climate change may further add to these. Inshore substratum spawners such as herring, smelt (*O. eperlanus*) and common goby (*P. microps*), and vegetation spawners such as sand-smelt (*A. presbyter*), would be at risk, particularly if the spawning adults were caused by high temperatures to avoid critical areas of spawning habitat. However, planktonic eggs of species such as Dover sole (*S. solea*) could also be exposed to the elevated temperatures. The thermal tolerance zone of eggs is narrower than for later developmental stages. For mid-temperate-zone fish, the zone of tolerance is typically 20–25°C, compared with only 11.5°C for embryonic stages (Rombough, 1997).

Spawning activity in estuaries has been described in the Thames by Colclough *et al.* (2002) and more generally in Western Europe by Elliott and Hemingway (2002). Relatively few fish species spawn in UK estuaries, and fewer still depend on estuaries entirely for spawning. Species that spawn in or close to transitional waters include smelt, common goby, sand goby (*P. minutus*), twaite shad, dace, sole, sea lamprey and flounder. Smelt and shad are often seen to spawn in the tidal freshwater or brackish reaches of rivers; for example, on the River Cree (Scotland) smelt spawn at Gatehouse of Fleet (head of tide) and in the Thames Estuary near Wandsworth (Colclough *et al.,* 2000), while twaite shad spawn on the River Usk (South Wales) above the town of Usk (~10 km above the tidal limit). Salmonids spawn entirely in freshwater but throughout the river system. Sand smelt will spawn in fully marine conditions but often spawn in estuaries or brackish lagoons – for example, in the lower estuary around Fawley in Southampton water (Turnpenny *et al.*, 1981), and in the Chesil Fleet, a brackish lagoon in Dorset (Bamber *et al.*, 1985). Herring are gravel substratum spawners and have adhesive eggs. Autumn-spawning herring stocks spawn offshore, but spring-spawning stocks deposit their eggs on gravel banks in shallow inshore areas, including estuaries, where they are more likely to be at risk from thermal discharges. Examples are the spring-spawning stocks in the River Blackwater Estuary, Essex (Fox and Aldridge, 2000), and in Milford Haven, Pembrokeshire (Clarke and King, 1985).

Timing of spawning in different water bodies is often affected by temperature. Environment Agency surveys in the closely associated Thames and Medway catchments have shown that smelt spawn later in the Medway, where spring temperatures of around 3°C lower are recorded. The mixing of stock from these two water bodies is thought to account for bimodal length–frequency distributions of 0-group smelt recorded in the Thames Estuary (S. Colclough, unpubl. data).

Once hatched, the larvae/post-larvae of these species would be at the same risk as other species, although if spawned within an estuary proportionally more larvae would be at risk of thermal shock or entrainment than would the young stages of marine spawners such as bass.

Table 25 provides data on preferred and limiting temperatures for the spawning of estuarine and marine fish. Values are only shown in the 'Lower' and 'Upper' categories where these are reported as known limits to spawning. Where a simple range of temperatures is shown, this is the range over which the species has been observed to spawn, but it does not necessarily imply limits.

Lyle and Maitland (1997) have described the spawning temperature threshold for smelt as 6°C in the River Cree in Scotland. Within the Thames Estuary, 25 yr fish survey data-sets allied to real-time water quality monitoring, indicate that smelt amass to spawn at Mucking at temperatures of 7–10°C, and actually spawn near Wandsworth in April when the temperature is between 11°C and 14°C (S. Colclough, unpubl. data.). Using the same data-sets, dace spawn contemporaneously at Wandsworth, and therefore at the same temperatures, and the threshold spawning temperatures for sand-smelt, which is recorded in June at Greenwich, is 17–19°C. Dover sole spawn between Gravesend and Southend at 9–12°C in April. In Southampton Water, Turnpenny *et al.* (1981) recorded serial spawning of sand-smelt starting in April–May at temperatures of ~10°C, continuing into summer temperatures to 17°C. Fonds and van Buurt (1974) have reported common goby as spawning at 10–20°C.

There appear to be no data-sets on high-temperature/low-DO effects on spawning, but this may not be a problem as most species spawn at temperatures where DO is unlikely to become limiting.

Table 25 Summary of temperature ranges for spawning

Species (Guild)	Common name	Temp., °C	Source
Agonus cataphractus	Pogge	Late autumn – early winter	
Clupea harengus	Herring	4.4–15	Ojaveer, 1981; Dempsey & Bamber, 1983
Engraulis encrasicolus	Anchovy	Surface: 14.5–19 Bottom: 12–15	Planque *et al.*, 2007
Gadus morhua	Cod	0–10, mainly 4–7	Brander, 1993
Lampetra fluviatilis	River lamprey	FW spawners	
Leuciscus leuciscus	Dace	11–14	S. Colclough, unpubl. data
Limanda limanda	Dab		
Liparis liparis	Striped sea snail	Jan.– March	
Merlangius merlangus	Whiting		
Osmerus eperlanus	Smelt	6–9 11–14	Lyle & Maitland, 1997 Colclough, pers. comm.
Salmo salar	Salmon	FW spawners	
Salmo trutta	Sea trout/brown trout	FW spawners	
Syngnathus rostellatus	Nilsson's pipefish		
Trisopterus luscus	Pouting	December – April	
Alosa alosa	Allis shad	FW spawners	
Alosa fallax	Twaite shad	FW spawners	

Species (Guild)	Common name	Temp., °C	Source
Anguilla anguilla	Eel	20–23	Haro, 1991
Atherina presbyter	Sand smelt	10–17 (Solent) 17–19 (Thames)	Turnpenny *et al.*, 1981; Bamber *et al.*, 1985 S. Colclough, unpubl. data
Chelon labrosus	Thick-lipped grey mullet		
Dicentrarchus labrax	Sea bass	10–12	Kennedy & Fitzmaurice, 1972
Dipturus batis	Skate		
Liza aurata	Golden grey mullet	20–22 (Caspian sea)	Avanesov, 1972. In Fazli *et al.*, 2008
Liza ramada	Thin-lipped grey mullet	19–21	S. Colclough, unpubl. data.
Mullus surmuletus	Red mullet		
Petromyzon marinus	Sea lamprey	FW spawners	
Platichthys flesus	Flounder	4–11	Rijnsdorp *et al.*, 2008
Pleuronectes platessa	Plaice	4–11	Rijnsdorp *et al.*, 2008
Pomatoschistus microps	Common goby	15–20	Wiederholm, 1987; Fonds & van Buurt, 1974
Pomatoschistus minutus	Sand goby	8–15	Wiederholm, 1987
Psetta maxima	Turbot	~14	Hara *et al.*, 2002.
Sardina pilchardus	Sardine		
Scyliorhinus canicula	Dogfish		
Solea solea	Dover sole	17–19 (Thames)	S. Colclough, unpubl. data
Sprattus sprattus	Sprat	9–14	Wahl & Alheit, 1988

5.7.2 Effects of temperature on recruitment

Power station-related temperature effects on fish in coastal waters are generally limited by the rapid dilution and cooling of the thermal discharge, confining the area of effect, and by the ability of most life-stages to avoid temperatures outside their preferred ranges. Exceptions would be where discharges affected confined marine habitats (eg, harbours) or where the thermal plume encroached onto the shoreline.

Although relatively few species spawn in estuaries, the early life-stages of many other fish move into or through estuaries from marine spawning areas (Costa *et al.,* 2002). The common species in Britain include herring, sprat, bass, Dover sole and plaice, as well as young gadoids such as poor cod (*Trisopterus minutus*), whiting (*Merlangius merlangus*) and pouting (*Trisopterus luscus*), for which transitional waters provide nursery areas. Elver (*A. anguilla*) also migrate through estuaries to get to freshwater. Species such as cod (*G. morhua*) may spawn along the coast before migrating offshore as juveniles or adults, and thus the thermal plume of a coastal power station could have an impact on such species.

Warm conditions in estuarine and coastal nursery waters (eg, flatfish nurseries in sandy bays) promote good year classes in many species, provided that they do not exceed optimal or lethal levels. Fonds (1979) found that the growth rate of newly hatched Dover sole larvae reached a maximum at 23–24°C, with growth very poor at 10°C and none at 7°C. As temperature increased, the time to metamorphosis decreased and occurred at a smaller size (at 22°C the time to metamorphosis was 2 weeks and the size was 9 mm; at 13°C the period was 4 weeks and the size was 10 mm; Poxton and Allouse, 1982). Passing through early developmental stages reduces the length of time that larvae/post-larvae are at risk from certain classes of predator. Warmer winter conditions also benefit some species by extending the length of the growth season and enabling them to remain further north than would otherwise be the case. The corollary of this is that cold-water species may be disadvantaged or suffer increased competition from Lusitanian species.

Costa *et al.* (2002) suggest that temperature is an important factor in attracting larval and post-larval fish into estuaries as well as affecting their survival and growth once they are there. Survival effects may in some cases be indirect – that is, by effects of temperature on predatory species. It is difficult to quantify such effects, but more warm-water species fare better in estuaries during warmer years. Holmes and Henderson (1990), for example, reported unusually high recruitment of bass, twaite shad, herring, pouting and red mullet (*Mullus surmuletus*) in the Severn Estuary following the exceptionally warm winter/spring of 1988/89.

In the case of cod, there is a well-established relationship between recruitment and sea temperature (O'Brien *et al.*, 2000; Clarke *et al.,* 2003; Beaugrand *et al.*, 2003). At the northern extremes, warming leads to enhancement of recruitment, whilst in the North Sea, close to the southern limits of the range, warm conditions lead to weaker-than-average year classes, and vice versa. During the late 1960s and early 1970s, cold conditions were correlated with a sequence of positive recruitment years in cod, haddock and whiting (Brander and Mohn, 2004) and subsequently high fisheries catches for a number of years to come (Heath and Brander, 2001). However, in more recent years, a warming climate has prevailed and year class strength has been weaker than average. Strong relationships have been observed between recruitment success and climatic variables in many other key fish and shellfish stocks that are critical to the UK fisheries economy, most notably whiting and haddock (Brander and Mohn, 2004), plaice (van der Veer and Witte, 1999) and herring (Nash and Dickey-Collas, 2005).

5.7.3   Effects of egg and larval entrainment at power stations

Thermal discharges from power stations may have effects of the type described above, and these effects may be over the zone of elevated temperatures (the thermal 'far-field'), but another risk is that of eggs and larval/post-larval fish becoming physically entrained into the CW intakes. They are then exposed to a combination of stressors, including hydrostatic and hydrodynamic pressure change, thermal shock, mechanical injury in screens, pumps and pipework and exposure to toxic biocides (usually chlorine- or bromine-based compounds).

The effects of these stressors on fish eggs and larvae/post-larvae, as well as of some crustacean species, have been investigated using a laboratory simulator (Entrainment Mimic Unit [EMU]; Turnpenny and Taylor, 2000). The results indicate that a notable proportion can survive this hostile treatment (Table 26). However, entrainment risk needs to be minimized in every case by careful siting of CW intakes away from areas of high larval density and by the use of appropriate intake technologies (Turnpenny and O'Keefe, 2005). The recent development of a new EMU (EMU2) for the BEEMS programme will allow entrainment mortality data be generated for an extended range of species and life-stages (Bamber and Turnpenny, 2011). EMU2 faithfully mimics the various stresses associated with passage of individual power station cooling circuits.

Table 26 Survival rates of entrained fish and crustacean from EMU CW passage simulation experiments (Bamber and Turnpenny, 2011)

Species	Life-stage	Entrainment survival rate at 0.2 ppm TRO and ~10°C ΔT	Prime causes of mortality
Bass (*Dicentrarchus labrax*)	Eggs	54%	Thermal stress
	Larvae	56%	Thermal stress and chlorine toxicity
Lobster (*Homarus gammarus*)	Larvae	92%	Mechanical stress
Shrimp (*Crangon crangon*)	Larvae	75%	Thermal stress and chlorine toxicity
Silver eel (*Anguilla anguilla*)	Larvae*	52%	TRO
Sole (*Solea solea*)	Eggs	93%	Pressure, thermal stress
	Post-larvae	8%	Thermal stress and chlorine toxicity
Turbot (*Psetta maxima*)	Eggs	93%	Pressure, thermal stress
	Post-larvae	30%	Thermal, mechanical and pressure stress

*Eel tested at 2 ppm TRO.

5.7.4 Spawning and early life history: summary

Temperatures suitable for spawning fall within a narrow range in species such as dace and smelt; for other species such as herring, whilst spawning may span a wider range of temperatures over its geographic range, the temperatures in British waters for spring-spawning stocks may already be towards their upper limit. If the temperature in a spawning area is artificially elevated, then spawning and early development could fail, since embryonic stages have a narrower thermal niche than older life-stages. Effective regulation of temperatures in spawning areas is therefore critical.

Once hatched, warmer temperatures can be beneficial by increasing developmental rates and shortening the time for larvae to pass through critical life-stages. This would be true (within limits) for all warm-water species, and possibly for cold-water species early in the season.

Ichthyoplankton entrained into power plant cooling systems will be subjected to thermal shock, along with mechanical and pressure-related stresses and, in many cases, biocide toxicity. Experiments conducted to date show that half or more of some species can survive entrainment, although the number of species and the range of power station exposure conditions so far examined remains limited. This is an important aspect requiring regulatory control.

5.8 Immunology and disease resistance

5.8.1 Temperature effects

The effects of environmental temperature on immunology and disease suppression appear to be complex. Fish in captivity can become stressed and more vulnerable to disease at both high and low temperature extremes. Rapid drops in temperature are generally tolerated better than sudden rises, although fluctuations of ~10°C in a day are not uncommon in rearing ponds (Noga, 2000). While not necessarily an indicator of what happens in the wild, because water quality and stock density conditions are unnatural, captive fish studies nevertheless point to what might happen *in extremis*.

High temperatures tend to favour certain pathogenic conditions, such as vibriosis (infection with pathogens of the marine bacteria genus *Vibrio*). For example, *Vibrio anguilarum* does not appear to infect farmed salmonids and turbot below 10–11°C, nor eels and flatfish below 15–16°C (Roberts, 1989). All marine fish are thought to be susceptible to at least one species of *Vibrio* and are most vulnerable at high salinities. On the other hand, wounded fish heal faster at high temperatures, although they are at greater risk of bacterial and fungal infections.

Crowding, organic pollution and other stressors can cause disease outbreaks (Noga, 2000). For example, *V. anguilarum* was very common in the eel (*A. anguilla*) farm that once operated using the thermal effluent from Hinkley Point Power Station on the Severn Estuary in Somerset. Here there was a combination of high temperature, crowding and high organic loading. Sea-reared salmonids are also very susceptible. Vibriosis can cause a variety of harmful conditions, including ulcers and muscle and eye lesions, often leading to death.

Jonsson and Jonsson (2009), in reviewing the likely affects of climate change on salmonids, drew attention to a number of temperature-related disease issues. These include effects of the myxozoan *Tetracapsuloides bryosalmonae*, which causes proliferative kidney disease (PKD), leading to mortality when water temperature exceeds 15–16°C. The virulence of *Aeromonas salmonicida* causing furunculosis, and sea lice *Lepeophtheirus salmonis* and *Caligus elongatus* killing salmonids at sea, also increases with water temperature.

Langford (1990) discusses the situation with regard to thermal discharges and lists numerous instances of increased disease and parasitic infections in warm-stressed fish living in thermal discharge zones.

Immunological research has shown that certain specific immune responses switch off at low temperatures, leaving the fish potentially vulnerable to infection; in salmonids, the critical low temperature has been found to be 4°C and in cyprinids, 14°C; however, non-specific defence responses such as phagocytosis tend to take over at lower temperatures (Le Morvan *et al.*, 1998).

5.8.2 Immunology effects: summary

In conclusion, it is likely that high summer temperatures, especially where combined with high organic loadings, may increase susceptibility of fish to pathogens such as *Vibrio* in the more saline lower reaches of an estuary, or in fully-marine harbours and coastal areas of low mixing, if fish remain there long enough to be affected. Under such conditions, stress on the fish is also likely to be exacerbated by lowered DO levels. The risk might be highest in summer-returning adult salmonids, where high temperatures and low DO levels may force fish to remain in the outer estuary until conditions are more suitable for entry. It has been noted that fish so-delayed are generally in poor condition (Solomon and Sambrook, 2004) and may be more vulnerable to pathogenic infections. On the other hand, elevated winter temperatures may help to maintain some species in an immunological 'safe zone'.

5.9 Diet and growth

5.9.1 Temperature effects on growth rates

The optimum temperature for growth is defined as the temperature at which growth rate is highest when the fish are reared under conditions of maximum or excess feeding (Jobling, 1981). There has been much work on optimum fish growth temperatures in the context of aquaculture, but there is increasing interest from an ecological perspective in the context of potential climate-change effects. A summary of data is shown in Table 27.

Growth rate increases with temperature up to the optimum growth temperature. Above this optimum, the growth rate of adults often declines, whereas in early life-stages it tends to reach a plateau (Rombough, 1997). Growth peaks at around 15°C in most cold-water species and in the twenties for most warm-water species, although Table 27 indicates that some warm-water species have optima below this. Kinne (1960) reported that the rates of food intake, food conversion and digestion in fish were highly temperature dependent, increasing by a factor of 2–3 times for a 10°C rise ($Q_{10} = 2$–3). Rombough (1997) investigated temperature effects on embryonic and larval growth rates over a range of fish species and found an embryonic Q_{10} of 3, falling to 2 for the larval stage.

Freitas *et al.* (2007) examined the relationship between temperature and growth rate in various marine fish and crustacean species to find optimum values. They also refer to 'performance breadth', a band within which growth is >69% of the optimum (Figure 20). This concept is not dissimilar to that of the 'thermal niche breadth' discussed in Section 5.4.1.

Table 27 Summary of data on optimum growth temperatures in fish

Species (Guild)	Common name	Eggs		Larval		Juvenile		Adult	
		Temp., °C	Source	Temp., °C	Source	Temp., °C	Source	Temp., °C	Source
Agonus cataphractus	Pogge							9.5	Power & Attrill, 2002
Clupea harengus	Herring					7 8–12	Saat & Veersalu, 1996; Reid *et al.*, 1999		
Engraulis encrasicolus	Anchovy			>20	Petitgas, 2008				
Gadus morhua	Cod			7.9 14–16	Jordaan & Kling, 2003 Otterlei *et al.*, 1999	8.2–15.4 depending on haemoglobin	Petersen & Steffensen, 2003	10–15ºC 14	Drinkwater, 2005 Freitas *et al.*, 2007
Lampetra fluviatilis	River lamprey								
Leuciscus leuciscus	Dace								
Limanda limanda	Dab					15–18	Bolle *et al.*, 2004	11.63 (modelled)	Attrill & Power, 2004
Liparis liparis	Striped sea snail								
Merlangius merlangus	Whiting							12.56 (modelled)	Attrill & Power, 2004
Osmerus eperlanus	Smelt							12	Keskinen *et al.*, 2009

Species (Guild)	Common name	Eggs		Larval		Juvenile		Adult	
		Temp., °C	Source	Temp., °C	Source	Temp., °C	Source	Temp., °C	Source
Salmo salar	Salmon	6	Peterson *et al.*, 1977			15.9 Parr 16–20	Elliott & Hurley, 1997 Jonsson & Jonsson, 2009	15.1 Post-smolt 13	Jobling, 1981 Jonsson & Jonsson, 2009
Salmo trutta	Sea / brown trout	14	Ojanguren & Brana, 2003.			16.1–17.4	Ojanguren *et al.*, 2001	13	Elliott, 1975
Syngnathus rostellatus	Nilsson's pipefish							14–14.2	Power & Attrill, 2003
Trisopterus luscus	Pouting							11.75 (modelled)	Attrill & Power, 2004
Alosa alosa	Allis shad	15–27	Charles & Jatteau, 2010					22–24	Freyhof & Kottela, 2008
Alosa fallax	Twaite shad			17–21.5 (preference)	Gerkens & Thiel, 2001				
Anguilla anguilla	Eel	-	-	-	-	16–19	Haro, 1991	22–23	Sadler, 1979
Atherina presbyter	Sand-smelt								
Chelon labrosus	Thick-lipped grey mullet								
Dicentrarchus labrax	Sea bass					22–24 23	Barnabé, 1991 Freitas *et al.*, 2007	24–27 23	Barnabé, 1991 Freitas *et al.*, 2007
Dipturus batis	Skate								

Species (Guild)	Common name	Eggs		Larval		Juvenile		Adult	
		Temp., °C	Source	Temp., °C	Source	Temp., °C	Source	Temp., °C	Source
Liza aurata	Golden grey mullet								
Liza ramada	Thin-lipped grey mullet								
Mullus surmuletus	Red mullet							13.6–23.8 (obs.)	Machias *et al.*, 1998
Petromyzon marinus	Sea lamprey	Max mass increase: 19	Rodriguez-Munoz *et al.*, 2001					15	Farmer *et al.*, 1977. In: Jobling, 1981
Platichthys flesus	Flounder							18 20	Fonds *et al.*, 1992 Freitas *et al.*, 2007
Pleuronectes platessa	Plaice							18 20	Fonds *et al.*, 1992 Freitas *et al.*, 2007
Pomatoschistus microps	Common goby								
Pomatoschistus minutus	Sand goby							12–18 20	Hesthagen, 1976 Freitas *et al.*, 2007

Species (Guild)	Common name	Eggs		Larval		Juvenile		Adult	
		Temp., °C	Source	Temp., °C	Source	Temp., °C	Source	Temp., °C	Source
Psetta maxima	Turbot			15	Jones, 1972	19.6 (Scottish)	Imsland *et al.*, 2000	14–16	
Sardina pilchardus	Sardine								
Scyliorhinus canicula	Dogfish	16	Thomason *et al.*, 1996						
Solea solea	Dover sole	Hatching: 13–15 Success: 10	Devauchelle *et al.*, 1987 Fonds, 1979	19–22	Fonds, 1979	20–25	Irvin, 1973; Fonds, 1976	14.17 (modelled)	Attrill & Power, 2004
Sprattus sprattus	Sprat			Max. survival: 6.8	Petereit *et al.*, 2008			6.74 (modelled)	Attrill & Power, 2004
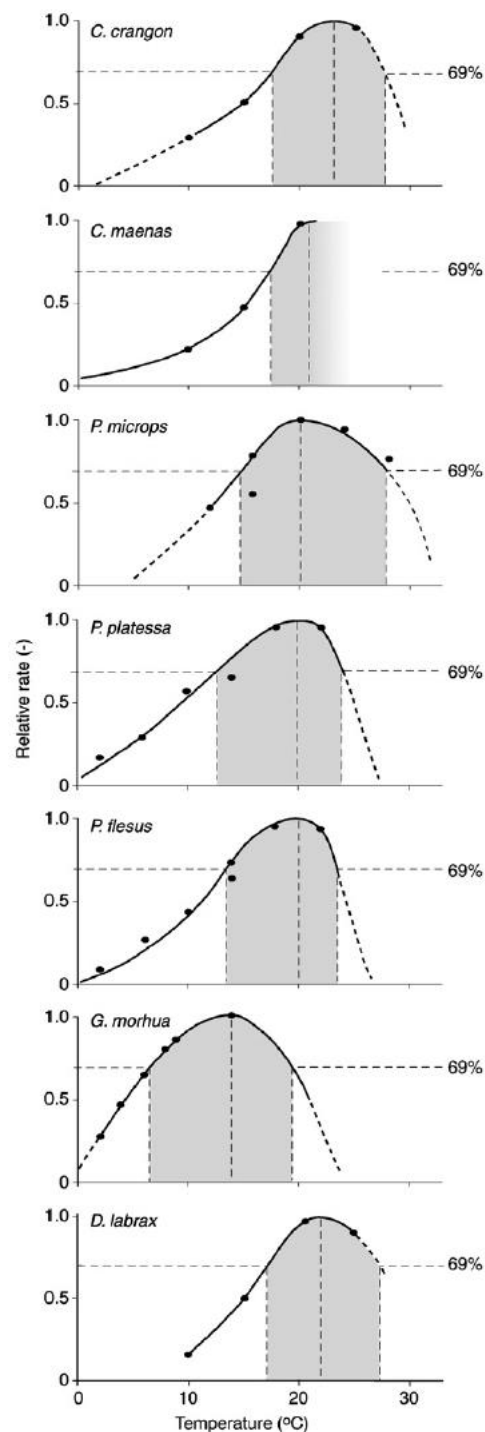
Figure 20 Optimal temperature (vertical line) (°C), temperature performance breadth (grey range) and tolerance range (total range of positive rate) for the crustacean species *Crangon crangon* and *Carcinus maenas*; the common goby *P. microps*; the flatfishes *Pleuronectes platessa* and *Platichthys flesus*; the gadoid species *Gadus morhua* and the sea bass *Dicentrarchus labrax*. Based on data of food intake and/or growth in relation to temperature. Rates are scaled to the maximum rate, which has been set to 1.0 (Freitas *et al.*, 2007).

In transitional waters, temperatures are considerably more variable than in rivers, lakes or the sea, being affected by tidal and diurnal effects. Growth can be faster in fluctuating temperature regimes ('thermocycling') than at static temperatures with the same mean temperature, but this enhancement applies only when the mean temperature is below the optimum (Jobling, 1997). Some power plants thermocycle to follow daily cycles of energy demand (eg, Combined Heat and Power (CHP) plants). It is now also common for power plants to operate intermittently as a result of the pool-bidding process of energy marketing.

Growth and food consumption are related to metabolic rate through energy expenditure or storage (Brett, 1970). Activity or stress may therefore affect the growth of fish at different temperatures as a result of increased energy demand.

Data for pogge given by Power and Attrill (2002) indicate optimum growth at 9.5°C, which appears low given their southerly limit of distribution at 43°N (Table 19). The data are based on sampling from a fixed point on the Thames (West Thurrock Power Station) and again highlight the need to consider data from a wide range of habitats. This caution should be applied when considering any thermal response data obtained from field studies.

Cod have been well studied and are a good example of a cold-water fish that is likely to be affected by temperature increases in UK waters. Drinkwater (2005) discusses optimum growth temperatures of cod in the context of global warming and cites optimum growth based on laboratory and field studies at temperatures between 10°C and 15°C. In an early study, Mckenzie (1934) showed that cod cease feeding at temperatures below 0°C and above 17°C. Brander (2000) found that weight-at-age of North Sea cod in the first year of life was strongly influenced by temperature. Older and larger cod have lower optimal temperatures for growth (Björnsson *et al.,* 2001). Pilling *et al.* (2007) examined annulus formation in the otolith (ear bone) of North Sea cod. The authors reported a change in the timing of annulus formation during warm versus cold periods (1985–86 contrasted against 1994–95). This study confirmed that southern North Sea cod do experience increased thermal stress during warmer years and that this manifests itself in slower growth rates.

The effect of rising seawater temperature on the growth of juvenile sole and plaice (*P. platessa*) was investigated by Teal *et al.* (2008) in the south-eastern North Sea, for the period 1970–2004. Increasing winter temperatures significantly increased the growing period of sole, a warm-water species that spawns in spring, but not of plaice, a temperate species that spawns in winter. The higher temperatures observed since 1989 positively affected the quality of the shallow coastal waters as a nursery area for sole but not for plaice. Between 1970 and 2004, mean lengths of 0-group sole increased significantly by over 2 cm.

5.9.2   Availability of food species

Since laboratory data on optimum temperatures for fish growth are based on feeding fish at maximum ration, it should be noted that effects on growth and feeding may arise indirectly due to temperature and hypoxia from an impoverished food supply.

5.9.3   Effects on growth: summary

Growth rates are found to increase, typically by a factor of 2–3 for a 10°C rise, up to a temperature optimum. Temperatures above the optimum value, especially in combination with chronically hypoxic conditions, can be expected to inhibit growth. Thermal performance breadth data, however, provide a useful indication of the temperature band in which fish can feed and grow near-optimally

Fluctuating temperatures are a natural phenomenon in estuaries and can be accentuated by thermocycling in some types of power plant, which can lead to growth rates that would exceed those attained at a stable temperature equivalent to the average value.

5.10 Fish population and community level effects

5.10.1 Localized effects of thermal discharges

5.10.1.1 The Kingsnorth thermal discharge

An unusual thermal discharge configuration is found at Kingsnorth Power Station on the Medway Estuary. Here the outfall enters a semi-natural creek system with a half-tide weir supporting a lagoon at its upper end. The outfall channel is some 3.5 km long and generates a tidally oscillating thermal gradient, with a ΔT of $\leq$ +10°C at its upper end, falling to ambient at its outer extremity. Temperatures in the canal can fluctuate by 8–10°C in a day.

This system has created a useful opportunity to study the effects of thermal discharges on estuarine biota, including benthos (Bamber and Spencer, 1984) and fish (Langford, 1983a, 1987). Approximately 20 of the 50 or so species of fish recorded in the Medway Estuary were also found in the canal (temperature range of 6–32°C) but only five were found in the lagoon and of these only bass (*Dicentrarchus labrax*) and golden and thin-lipped mullet (*Liza auratus* and *Liza ramada*) were found in every month surveyed (Langford, 1983a). Bass that entered the canal in July–September when they were still growing continued to do so, whereas later immigrants (October–January) that had already stopped growing in the colder estuary began to grow again, producing an unseasonal annulus on their scales (Langford, 1987).

Pickett and Pawson (1994) found throughout their study in the Medway that the warm water held a strong attraction for young bass at all times of the year. Post-larval bass arrive in the Thames Estuary relatively late compared to fish spawned earlier in the western English Channel and Bristol Channel. However, the 0-group in the Kingsnorth outfall canal grew more rapidly and reached a larger size (mean length 7.5–9.9 cm) than 0-groups sampled from unwarmed nursery areas at the same latitude (which typically reach 5–8 cm by October/November). By itself this size advantage may have a strong survival value, since there is strong circumstantial evidence that bass < 6 cm suffer high over-wintering mortality at the time when growth slows in the autumn. Irrespective of size, 0-group bass living in the Medway outside the warmed canal may suffer high winter mortality at temperatures <5–7°C. The fish stop feeding, spend long periods motionless and may become moribund. Without the warming influence of its power stations, the Thames Estuary and its tributaries have in the past been too cold to support first-year bass during many winters (Pawson and Eaton, 1999). Wheeler (1979) noted that until the time of his study, there had been few records of 0-group bass being taken from the Thames Estuary as a whole which were not associated with power stations (intake screen sampling). Nowadays, the Thames Estuary is recognized as the largest new bass nursery in the southern North Sea, with multiple waves of bass fry extending upstream as far as Richmond (Colclough *et al.*, 2002)*.

5.10.1.2 Port River-Barker Inlet Estuary, Australia

Jones *et al.* (1996) studied the effects of a thermal effluent on the fish community of an Australian estuary. This study showed rather similar findings to those at Kingsnorth, although in a radically different setting. The site is a semi-tropical estuary fringed with mangroves and has a salinity range of 35–41; the temperature at the site closest to the outfall ranged from 20 to 34°C, and the distant site 12 to 26°C. The number of species found, 41 in total, increased with distance from the outfall. During the summer/autumn period, thermal effluent only affected the species compositions in the inner estuary, whilst the estuary-opportunistic species *Aldrichetta forsteri, Arripis georgiana, Arripis truttacea* and *Hyporhamphus melanochir* avoided the area. During winter/spring the situation was reversed, with *A. forsteri* attracted to the warmer waters of the inner estuary. The extended growth season for this species, and a significantly higher growth rate for *Sillaginodes*

punctata, promoted a premature movement out of the inner estuary that the authors suggest could increase the vulnerability of these species to commercial fishing and to natural predators. The temperature-related loss of some seagrass from the inner estuary could have had indirect effects on the fish community.

5.10.2 Global warming: effects on zoogeography and phenology

Wood and MacDonald (1997) present a collection of papers on the subject of global warming implications for fish. Several of the papers from this book are cited individually in previous sections. Much of the content deals with experimental evidence on the detailed thermophysiology of fish. Whilst physiological evidence is of academic interest, the fundamental message from this and other work reviewed is that fish exhibit behavioural thermoregulation and compete for resources within their own thermal niche. This implies that fish will only be stressed by unsuitable temperatures when they are unable to access their preferred thermal niche, either owing to competition or by barriers.

The effects of global climate change on fish are seen at the macro-level in terms of geographic shifts in population boundaries. This is not a new phenomenon: Cushing (1995) reviews numerous cases of classic population cycles in fish related to oceanic cycles and events (eg, pilchard-mackerel fluctuations on the west coast of Britain related to Gulf Stream effects, and El Niño effects on Californian sardine fisheries). These are cases of population boundaries shifting transiently with time in response to temperature and current patterns. Murawski (1993) analysed the results of 24 years of trawl surveys in the North Atlantic and found that a 1°C rise in average water temperature led to a poleward shift in mackerel (*S. scombrus*) and herring distribution of 110 km. Over the past 30 years, there has been a progressive, well-documented northerly spread of bass along the west and east coasts of Britain, together with changes in spawning areas (Pickett *et al.*, 2004; Pawson, 2005). Global warming may be expected to lead to non-reversible shifts of this type in the United Kingdom, essentially a poleward shift in the distribution of all species as shown by Stebbing *et al.* (2002). Genner *et al.* (2004) have proposed that community changes will be more complex than this idea would suggest. Stebbing et al (2002) showed that while there seemed to be a greater inward migration of southerly species, there was a lesser outward migration of northerly species. The new order will reflect not simply the drifting of species into their favoured thermal niches but, rather, patterns controlled by inter-specific ecological interactions and dispersal capacity that will make species-level responses difficult to predict. Buckley and Durbin (2006) suggest that fish pursuing a particular prey species, which may itself be subject to climate-induced changes in distribution, may not be able to remain in their preferred thermal niche while doing so.

That the concept of a general poleward shift in response to climate change is over-simplistic is also illustrated by a study by Dulvy *et al.* (2008). These authors followed the tendency of North Sea fish to move into deeper, cooler waters as a result of temperature increases. Over a 25 yr period, North Sea winter bottom temperatures were shown to have increased by 1.6°C, and the whole demersal fish assemblage had deepened by around 3.6 m per decade. Latitudinal shifts also accompanied this movement. A useful pictorial summary of gross trends in fish movements associated with climate change is given in ICES (2011), shown in Figure 21.
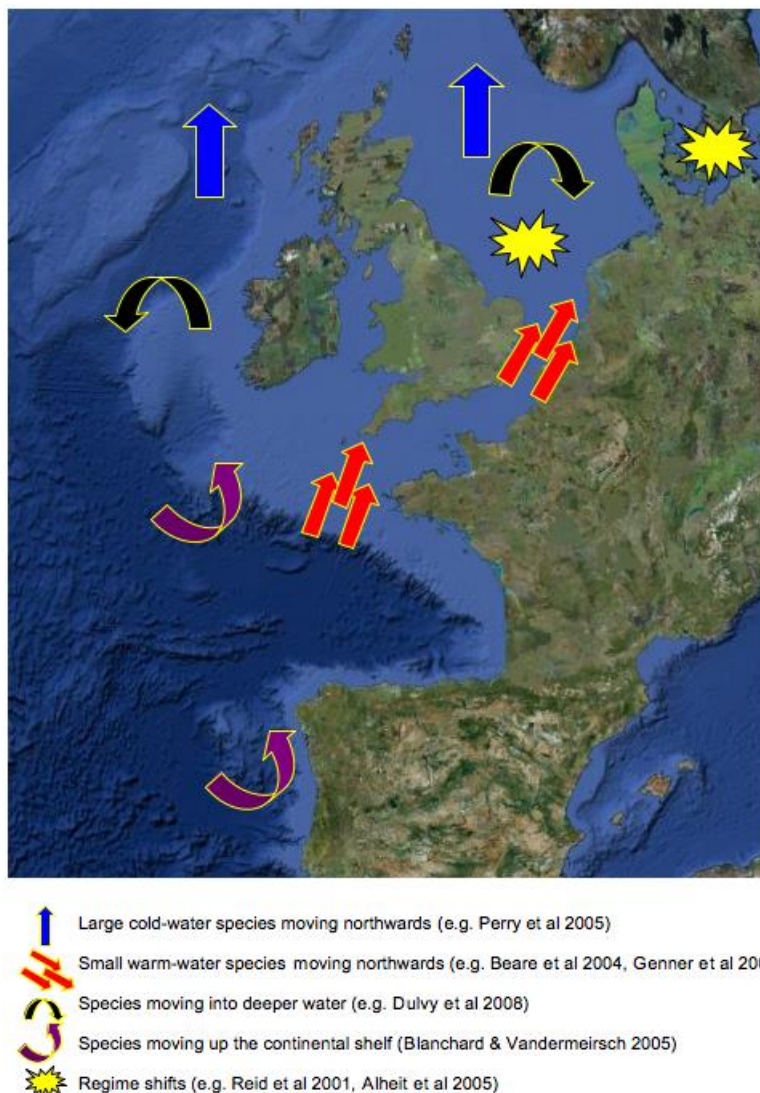
Figure 21 Reported climate-induced changes in distribution of species and composition of assemblage (ICES, 2011)

At the individual river-basin level, the first signs of temperature increase have already been witnessed in terms of increased sightings of southern visitors (Holmes and Henderson, 1990). Climate-related temperature increases may occur as a result of increased warming of shallow areas but also may be due to oceanic-scale features such as the North Atlantic Oscillation, which has forced warmer water into the North Sea. This has caused increases in Lusitanian species such as red mullet (*Mullus surmulletus*), pilchard (*Sardina pilchardus*) John Dory (*Zeus faber*) and snake pipefish (*Entelurus aequorius*) (Beare *et al.*, 2005; ICES, 2008). In general, rising temperatures will see the loss of some cold-water species that are on the edge of their thermal niches, and the gain of warm-water species (

Table 19). Reference to community structure in transitional water bodies at more southerly latitudes may give a good idea of what can be expected in the future, although this will also depend on finding a good match for other characteristics, such as size, geomorphology, tidal range and exposure. This theme has recently been developed by Engelhard *et al.* (2011), who present a categorization concept whereby species were assigned with respect to six ecotype classifications, according to biogeography, horizontal and vertical habitat preference, trophic guild, trophic level, or body size. Broadly speaking, in response to climate change

there were steady increases in abundance of species that were either Lusitanian, small-bodied, or low-/mid-trophic-level ecotypes, and generally declining, or only marginally increasing trends of most Boreal, large-bodied, or high-trophic-level ecotypes, or combinations of them.

For individual species, raising year-round temperature will tend to benefit fish growth and production, provided that the new temperature regime does not push the fish outside its optimum range. McCarthy and Houlihan (1997) illustrate this point using data on salmon from the River Alta in Norway (Figure 22). Salmon parr feed only at temperatures of >7°C. Under the predicted global warming scenario, this temperature is reached earlier in the year and extends later into the year, increasing the growing season by some weeks. Since the entire year will remain within the salmon's preferenda, higher temperatures are also predicted to increase growth rates. The second data-set from the River Nivelle, a more southerly river in France, shows that salmon potentially can grow under either regime all year round but that the period when the optimum growth temperature is exceeded is extended under the predicted global warming regime, suggesting that growth would be inhibited in this case. Whilst this example is not taken from a transitional water, the principle must hold good. Changes such as advancement, retardation or altering the duration of a seasonal ecological activity are known as phenological. A number of other examples of phenology are cited in earlier sections of this report – for example, in relation to juvenile sole migration (Section 5.4.1), and the extended growth season of bass at Kingsnorth (Section 5.10.1.1).

Jonsson and Jonsson (2009) provide a useful review of the thermobiology of salmonids in relation to likely climate-change effects, predicting extinction of populations at the southern extremities of their range. Atlantic salmon presently extend southwards to northern Spain and northern Portugal (latitude 37°N), but populations in southern British rivers can be regarded as being at particular risk, especially in estuaries such as the Thames, where low DO levels reduce metabolic headroom (Rosten et al., 2010). Pombo et al. (2002) indicate the declining salmonid populations in the Ria de Aveiro in Northern Portugal and suggest that this is a reflection of changing distributions due to climate change.
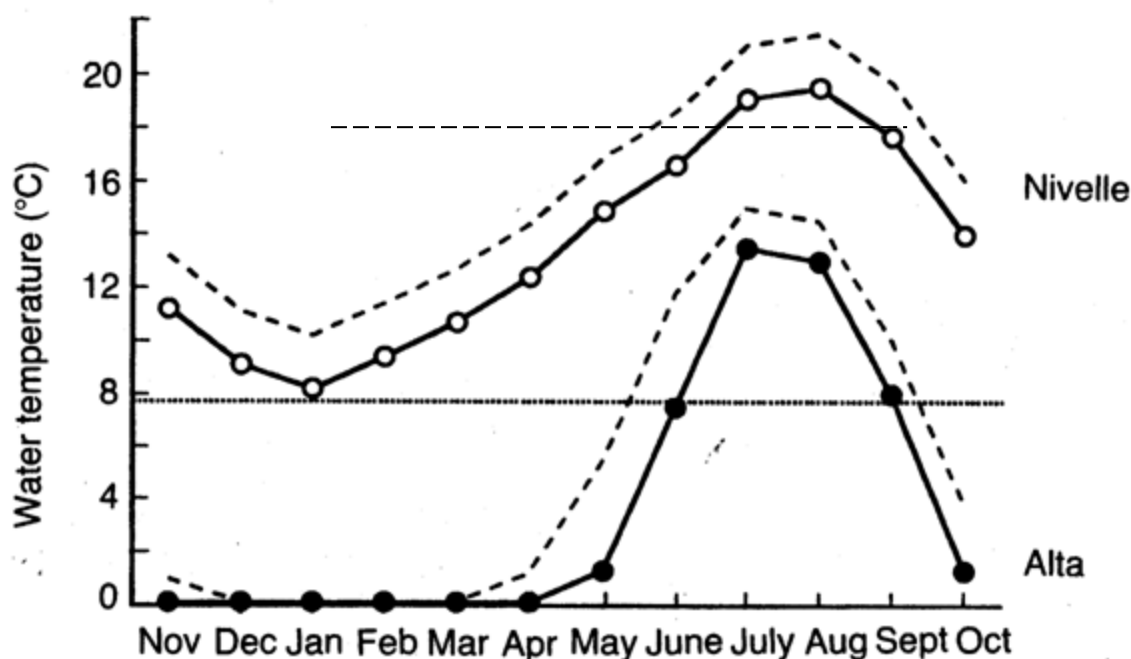
Figure 22 Average monthly water temperatures for two rivers from northerly (River Alta, Norway) and southerly (River Nivelle, France) parts of the salmon's geographic range. Note: The lower horizontal dotted line represents the minimum growth temperature and the upper dashed line represents the optimum growth temperature (adapted from McCarthy and Houlihan, 1997).

A study of the changing fish community in the Bristol Channel was published by Henderson (2007), using data derived from the CW intake screens of Hinkley Point Power Station. This study identified two periods of discrete change in the fish community over the past 25 years. The first change occurred in the late 1980s and involved a shift in the relative abundance of the 'permanent' members of the community. This abrupt event coincided with observed changes in the plankton of the north-east Atlantic (Beaugrand, 2004) and was correlated with the winter North Atlantic Oscillation. A second discrete change, affecting the total species assemblage, occurred in the early 1990s. This was marked by a sudden alteration in the set of 'occasionally occurring' species. After 1993 these were dominated by species with distributions centred to the south of the Bristol Channel, whereas northerly cold-water species disappeared. This change was correlated with average seawater temperature (and possibly the Atlantic Multidecadal Oscillation rather than North Atlantic Oscillation).

5.10.3 Population and community effects: summary

The thermal preferences of different species are a key factor in sorting the suite of species that will occupy a particular environment. On a macro-scale this is seen in terms of zoogeography, but species are found to zonate within a given water body according to temperature preferences where the thermal gradient is sufficiently strong.

Population shifts in exploited pelagic species such as herring, pilchard and mackerel associated with climatic cycles have been known for centuries and have had major economic impacts on local fishing communities. Similar effects and wider changes in the structure of fish communities could be associated with global warming; bulk temperature rises due to thermal power stations or other industries affecting large parts of smaller estuaries can produce a similar and additive effect on a local scale.

6 Thermal tolerances of shellfish and other epibenthic and infaunal species

6.1 Introduction

The potential ecological impact of thermal effluents has been the subject of increasing investigation over the last four to five decades, and the various environmental studies have been progressively reviewed (see, eg, Naylor, 1965; Water Research Centre, 1976; Talmage and Coutant, 1980; Langford, 1990; for a more comprehensive bibliographic list, see Langford, 1983a, chap. 5).

In the shorter term, potential environmental impacts of this raised temperature relate to the plume itself. In the longer term, in restricted water bodies such as estuaries (as a component of transitional waters *sensu* the Water Framework Directive: Anon., 2000), rias, fjards, fjords or coastal saline lagoons, there may be residual effects from an overall and maintained rise in temperature.

To a large extent, nektonic organisms can detect and actively avoid thermally enhanced waters if they perceive them to be deleterious, as a result of which they are unlikely to suffer significant impact from the shorter term temperature elevation. Equally, planktonic organisms in the receiving water will only be at risk as the effluent mixes and dilutes, thus as the ΔT decreases (ie, the amount of any exposure to the ΔT is in a direct negative relationship to that ΔT). It is therefore predominantly benthic organisms that are at risk of being affected by thermal effluents.

This chapter addresses the potential impacts of discharged cooling water on the aquatic invertebrates, macroalgae and plants of marine and estuarine waters. While there may be other impacts owing to the effects of the CW flow *per se*, such as scour, these are outside the remit of the present report and are not considered herein: direct and synergistic (with temperature) effects of that aspect of a CW discharge are properly addressed on a site-specific basis.

6.2 The effects of raised temperature

The direct effects of thermal discharges on the marine and estuarine environment fall into four categories:

➢ the change in temperature *per se* (the mean temperature in relation to 'normal' ambient water temperature), which relates to the ΔT;

➢ the absolute temperature, insofar as this can, in some cases, approach the thermal death point (upper incipient lethal temperature) of the exposed animals, and otherwise interfere with enzymatic functions or triggers;

➢ short-term fluctuations in temperature: where the discharge water meets the cooler receiving-water there is normally a sharp interface or 'temperature front'; where the receiving waters are tidal, this temperature front will be moved tidally in relation to the seabed or shore, on a regular daily cycle of twice per day;

➢ thermal barriers: it has been suggested that the temporary interface between effluent and receiving waters may give rise to barriers to the migration of aquatic species.

These are discussed separately below.

6.2.1 Change in temperature

As emphasized throughout this review, the area under impact from the effluent plume is warmer than the surrounding area. All marine species will have a preferred temperature range, evolved in relation to the normal temperature regime of their geographic range (see Chapter 5). Species and populations of species nearer their cooler limit of distribution may be expected to gain advantage from the effects of a thermal discharge, while those near their warmer limit may be deleteriously affected. Where a deleterious impact from raised temperature results in significant loss of a species, it can be difficult to distinguish the effects of mean temperature from the effects of absolute temperature. These aspects will be illustrated here using selected and representative species.

6.2.1.1 Sabellaria alveolata

The Lusitanian polychaete *Sabellaria alveolata* (L.) (the honeycomb-reef worm) builds a sand tube, and in colonies these tubes develop into communal reefs from the lower shore into the sublittoral (to depths of 20 m or more) (eg, Wilson, 1971; Warwick and Uncles, 1980; Bamber, 1997; Warwick, 2008). At Hinkley Point Nuclear Power Station(s), on the Bristol Channel, *S. alveolata* colonizes the shore at low water and develops substantial reefs which are significantly larger (up to 1 m across) within the flow of the CW effluent water than they are elsewhere along the shore (where they are generally up to 15 cm across, but also merging) (Figure 23).

Figure 23 *Sabellaria alveolata* reef within the outflow of the CW discharge from Hinkley Point Power Station (metre rule for scale)

The biology of *S. alveolata*, including tube-building, growth and feeding, is suppressed at low temperatures (<5°C; Y. Gruet, unpubl. data); this species is thus constrained by low winter temperatures, water temperatures naturally falling below 5°C in midwinter and air temperatures (during low-tide emersion) below 0°C, and indeed can be killed by frost when exposed at low tide.

In an intensive study of the reefs at Hinkley Point (Bamber and Irving, 1994, 1997), the difference in size was attributed to suppression of tube growth and increased mortality in normal low-shore populations during winter months when water temperatures can fall below 5°C and air temperatures (during low tide) below 0°C; this temperature control appears to be the main restriction on the upstream limit of distribution of this species in the Severn Estuary (rather than the also-limiting reduction in salinity). Specimens of *S. alveolata* growing at the power station outfall, by contrast, experienced water temperatures at or above 10°C (normally <13°C in midwinter) and are protected from frost during low tide by radiation from this water; they are thus able to survive in colder winter periods, and continue feeding and growth (including tube-building).

As a further benefit, the resultant increased density of crevicial niches in these larger reef units allows significantly larger densities of such species as the isopod *Cyathura carinata* (Krøyer), the amphipod *Melita palmata* (Montagu), the shore crab *Carcinus maenas* (L.) and the syllid polychaete *Syllis gracilis* (Grube).

6.2.1.2 *Cyathura carinata*

The anthurid isopod *Cyathura carinata* (Krøyer) is a European species which has, over time, spread northwards from its origin in the Mediterranean (Burbanck, 1961), now occurring as far north as the Baltic. A population of *C. carinata* living within 100 m of the outfall at Kingsnorth Power Station on the River Medway Estuary, Kent, was studied monthly over two years (Bamber, 1985a); the water temperature at this site was between 8°C and 10°C warmer than the normal estuary temperature.

Analyses of the age-structure, recruitment and life histories of *C. carinata* from Biscay, the Kiel Canal and the Baltic demonstrate that, as this southern species has moved north, the lower winter temperatures have produced constraints on the life-history pattern. Growth is reduced or absent through the winter in Baltic populations, such that not all individuals (probably as few as 50%) are sufficiently large to breed at 1 yr old (Figure 24); there is thus a selection pressure to increase longevity in northern populations, as is also found in the sibling species *Cyathura polita* (Stimpson) on the North American Atlantic coast. This increased longevity acts to encourage protogynous hermaphroditism until conditions reach the point where 1+ age-class maturity is minimal, when hermaphroditism is no longer an advantage, and the sex ratio returns to 1:1.
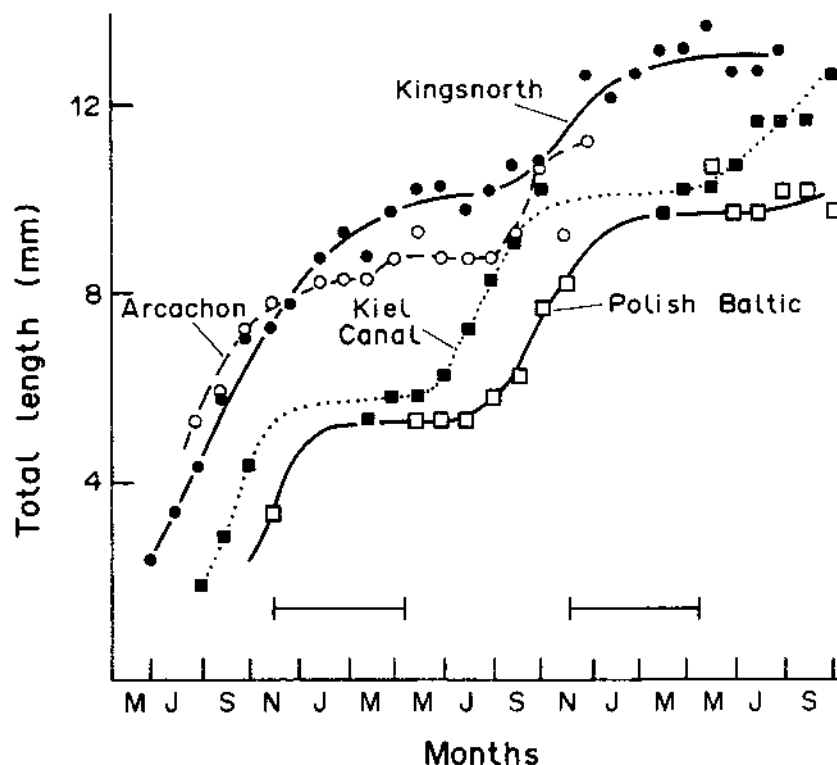
Figure 24 Growth curves for *Cyathura carinata* populations from Kingsnorth Power Station discharge canal (solid circles), compared with populations from Arcachon (open circles), the Kiel Canal (solid squares) and the Polish Baltic (open squares) (after Bamber, 1985); horizontal bars indicate winters

Thus southern populations, with good winter growth, live to 1+ with 90% maturity at 1 yr old, summer recruitment and a low female bias in the adult sex ratio; northern populations show no winter growth, live to 2+ with only 50% maturity at 1 yr old, and have a high female bias in the adult sex ratio. The Kingsnorth population showed the 2+ longevity and highly biased sex ratio of a (genetically) northern population. However, the enhanced temperature regime of the power station CW outfall allows continuous winter growth (Figure 24) and early summer recruitment, resulting in a successful population with over 90% of 1+ adults being mature and breeding. Clearly, the decade during which the population had lived in the power station effluent area was not in any way sufficient to change the 'northern evolution' of its life-history strategy, but the raised winter temperatures have allowed this southern species to develop the most dense (by an order of magnitude) population recorded from British waters (Table 28).

Table 28 Densities (no. per m^2) of British populations of *Cyathura carinata*

Site	Maximum density	Reference
Poole Harbour	200	Spooner & Moore, 1940
Tamar Estuary	380	Spooner & Moore, 1940
Towy Estuary	496	Howells, 1964
Severn Estuary	40	Boyden & Little, 1973
Kingsnorth	4,040	Bamber, 1985
	(mean 538)	

6.2.1.3 *Urothoe brevicornis*

A similar enhancement of an amphipod population was found by Barnett (1971), who compared breeding and growth of the sand-dwelling amphipod *Urothoe brevicornis* from the beach adjacent to Hunterston Power Station, Scotland, with those of a control population 3 km away. The Hunterston animals showed an earlier breeding season, with earlier recruitment (by up to 2 months) and more prolonged growth than the control population (Figure 25). As a result, fully grown adults at Hunterston were some 20% larger, although they showed no difference in growth rate. Pearson and Barnett (1987) similarly studied the population dynamics of other species in the Hunterston area and concluded that a summer spawner such as the bivalve *Tellina* (now *Angulus*) *tenuis* extended its spawning period due to the thermal discharge, whilst a cold-water spawner, the lugworm *Arenicola marina,* had a delayed spawning and even may have not spawned in some winters due to the unnatural elevation of the thermal discharge.
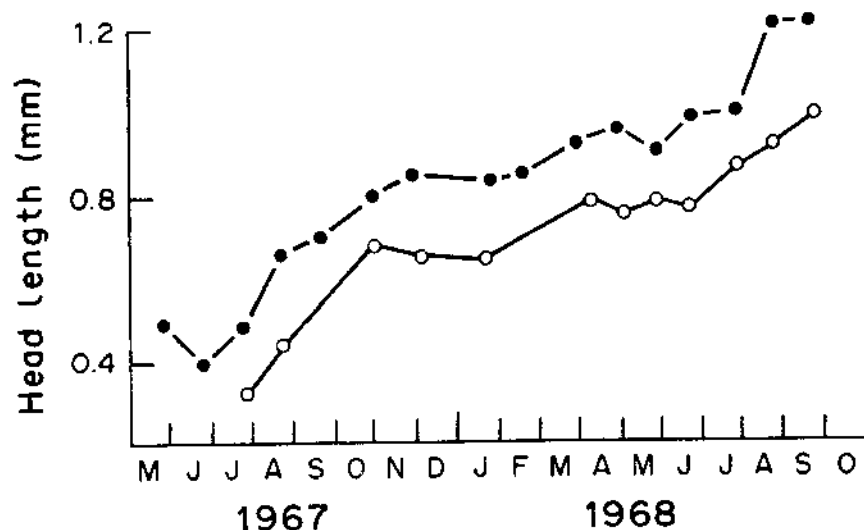
Figure 25 Growth curves for *Urothoe brevicornis* for the 1967 cohorts at Hunterston Power Station beach (solid circles) and the control beach at Millport (open circles) (redrawn after Barnett, 1971)

6.2.1.4 Barnacles

The European barnacle *Semibalanus balanoides* (L.) is an Arctic–Boreal species, and its distribution is somewhat constrained by the fact that, for successful reproduction, it requires a period of temperature <10°C to allow successful fertilization (Crisp, in Naylor, 1965).

Population densities of this species were studied along a rocky-shore transect away from the outfall headworks of Wylfa Power Station, Anglesey, in 1971 (Bamber, 1989). This species showed notable reductions of percentage cover near the outfall (Figure 26). There was a clear trend of increasing density away from the outfall, with a significant reduction within 200 m of the discharge headworks.
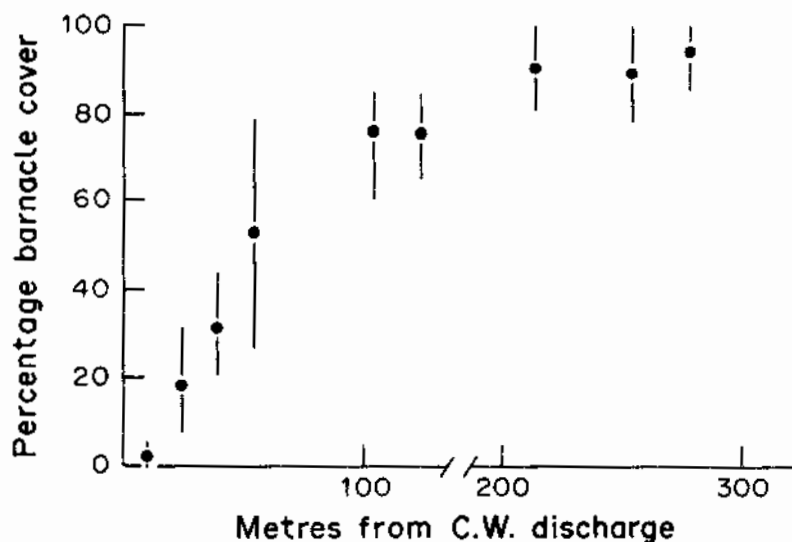
Figure 26 Percentage cover of *Balanus balanoides* (means with ranges; 5 replicates per station) for a transect of 10 sampling stations on the rocks leading away from Wylfa Power Station CW discharge (from Bamber, 1989)

While actual temperature data were not available, extrapolation from other situations suggests that at 200 m distance, the surface waters encroaching on this littoral zone carry a ΔT of some +6°C to +8°C. That some specimens of *S. balanoides* (mean of 2.4% cover) were surviving at the station nearest to the headworks suggests that this is an effect of mean temperature rather than absolute temperature mortality. Such a gradient response is characteristic of a mean-temperature effect. Similarly, Straughan (1980a, 1980b), studying the littoral fauna around Estero Bay and King Harbor Power Stations in California, United States, examined the distribution of two barnacle species of the genus *Chthamalus*. She found that sites affected by both thermal discharges favoured the more southern, warmer-water species *Chthamalus fissus* over the northern, cooler-water species *Chthamalus dalli*.

6.2.1.5 Oligochaetes

While littoral species are naturally subject to greater temperature extremes and variation than are sublittoral species, the fauna of this zone still includes species whose tolerances to raised temperature are different, particularly species of the low-littoral extending into the sublittoral. A study of the benthos of Kingsnorth Power Station found two dominant species of the oligochaete genus *Tubificoides* living in the CW outfall canal. A 3 yr monthly survey of six sites along the length of the canal (from adjacent to the outfall to 4 km downstream) found that the common estuarine species *Tubificoides benedii* (Udekem) showed a gradient

response of decline from the control site (DC4) to the discharge outfall, while the congeneric *Tubificoides amplivasatus* Erseus increased in density towards the outfall (Figure 27).
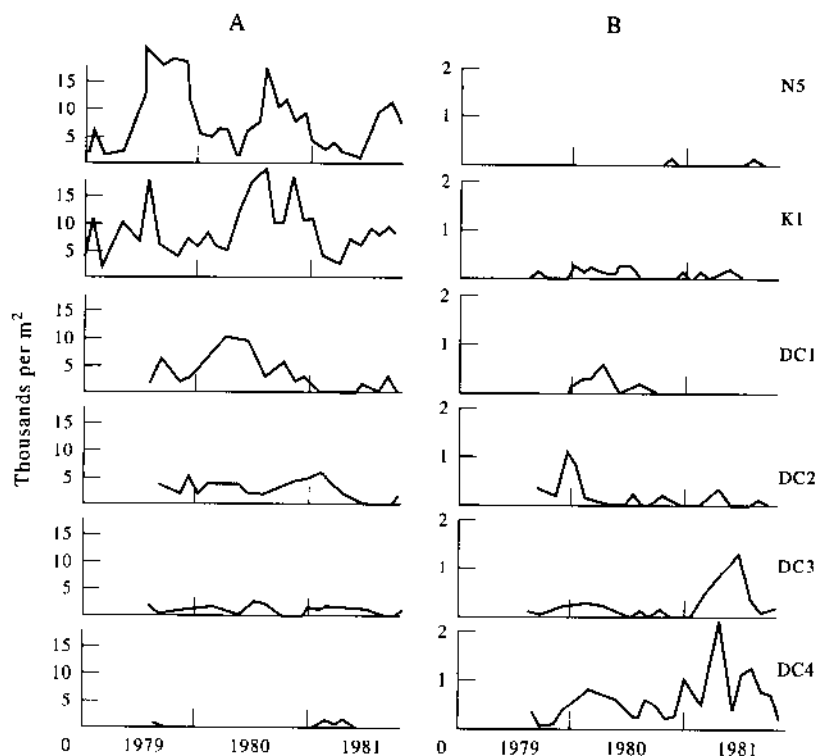
Figure 27 Monthly densities of (A) *Tubificoides amplivasatus* and (B) *Tubificoides benedii* in the CW discharge canal of Kingsnorth Power Station, at six sampling sites reflecting a gradient of mean ΔT from 9.2°C at N5 to 0°C at DC4 (after Bamber and Spencer, 1984)

Paradoxically, and appreciating that the distributions of most tubificid oligochaetes are poorly known, *T. amplivasatus* appears to be a northern species, mainly recorded from Scandinavia, where *T. benedii* is also dominant.

6.2.1.6 *Shore zonation*

A broader effect of the change in mean temperature brought about by heated CW effluents is found in the zonation of species across shores on which such effluents impinge. Species inhabiting the littoral zone are naturally exposed to greater ranges and extremes of temperature than are species restricted to the sublittoral. Species of the upper shore are subject naturally to wider extremes of temperature (and thus higher maximum temperatures) and for a longer time (when uncovered) than are species from the lower shore (and mid-shore species are of course intermediate). Although, when covered by the tide the shore temperature will reflect that of the water, once uncovered exposure to insolation, wind and frost can lead to a radical and rapid change in temperature. During summer low tides in Milford Haven, Spencer (1970) measured temperature rises on the uncovered sand-flats of 0.2°C min^{-1}, with a temperature range in September (1966) of 15°C to 26°C on the shore, compared with an ambient water temperature of 16.5°C. On the mud-flats of the Medway Estuary (adjacent to the Thames) Walters (1977, in Bamber, 1990) measured an annual range from –4.5°C to 32.5°C. The higher the shore zone, the longer the exposure to atmospheric (rather than aquatic) conditions and thus the more extreme are the conditions described above.

As a result, the species that inhabit this zone are tolerant of (and adapted to) such temperature variations (within natural timescales).

The overall effect of a thermal elevation is to displace cool-stenothermal or low-littoral species (Bamber and Spencer, 1984) and cause a downward-shifting of the shore zonation, as less tolerant lower-shore species are displaced, while mid-shore species can move lower on the shore to a more acceptable thermal regime, and similarly upper-shore species can move down to the mid-shore as that has become vacated. These changes result from effects on physiology, on recruitment failure and simply mortality.

An extreme example of this downward shift in littoral zonation was found at Hinkley Point Power Station, situated on the highly turbid waters of the Severn Estuary/Bristol Channel (Figure 28) (Bamber and Coughlan, 1987). The CW outfall discharges across the shore within a cut channel through the stratified bedrock. At high tide the buoyant effluent plume disperses at the surface into the estuary, tending downstream as the tide falls; at low tide it meets the receiving water at the mouth of this channel. As the tide rises, the effluent is lifted out of the channel and impinges directly across the rock surfaces upstream of the channel.

Figure 28 Hinkley Point outfall channel from the air, showing seaward-facing limestone slopes colonized by fucoid algae (dark areas)

Thus, the seaward-inclined flat rock surfaces downstream of and away from the channel support dense beds of fucoid algae, from *Pelvetia canaliculata* on the upper shore through *Fucus vesiculosus*, to *Ascophyllum nodosum* on the mid- to lower shore and *Fucus serratus* on the lower shore; however, the rocks upstream of the channel appear almost devoid of algae for a distance of some 100 to 200 m from the channel. This is the result of the algae being eliminated from their favoured zone by the increased and fluctuating temperature; *F. vesiculosus* is present on the mid-shore, but neither *A. nosodum* nor *F. serratus* can colonize lower on the shore owing to the lack of light penetration in these highly turbid waters.

A similar down-shore movement of the littoral fauna and flora on rocky shores was recorded by Straughan (1980a, 1980b) for shores in California, generally restricted to an area within 500 m of the effluent flow.

This down-shore shift in vertical distribution extends into the sublittoral; Bamber and Spencer (1984) found that most of the sublittoral benthic species in Kingsnorth Power Station outfall channel were excluded, and were replaced by littoral invertebrates.

This increased tolerance of higher-shore taxa to their more variable environment is also expressed in other variable habitats. There is evidence that estuarine species, and those of coastal saline lagoons, are more tolerant of thermal stress than are fully marine species (eg, the congeneric isopods *Idotea chelipes* vs. *Idotea emarginata*; Naylor, 1965). This degree of differentiation in environmental tolerance as a result of evolutionary selection also extends to intraspecific differences – that is, between different populations of the same species.

The bivalve *Macoma balthica* (L.) has been studied sufficiently to show the complexity of interpopulation differences in environmental tolerance. Beukema and Dekker (2003) studied this species in the Wadden Sea and showed that, as well as colonizing as onshore and offshore populations, it migrates between littoral and sublittoral habitats, with the degree of offshore migration greater during cold weather. Egg productivity is greater on the low-shore or in the infralittoral than on the high shore (van der Meer *et al.*, 2003). However, Dorjes *et al.* (1986) showed that Wadden Sea populations of *M. balthica* fluctuate in response to temperature, but intertidal populations show a greater stability than do subtidal populations (in accord with the findings of Bamber and Spencer, 1984; see above). Bachelet (1986) studied in more detail recruitment in the intertidal population and found that recruitment was not related to environmental factors (including temperature) but was controlled by biological interactions (predation, competition).

6.2.2   Absolute temperature

Examples of an absolute temperature effect are less easy to detect, as they will commonly be expressed as an absence of a species, and thus less clearly detectable. During the study of the Kingsnorth Power Station outfall benthos mentioned above (Bamber and Spencer, 1984), one of the more abundant species of the littoral and infralittoral mud was the amphipod *Corophium volutator*. This species has been shown to become hyperactive at higher temperatures and will leave its tubes to swim upwards into the water column at temperatures over 25°C (Gonzales and Yevitch, 1971). In spring and early summer, the population of *C. volutator* near the Kingsnorth outfall headworks achieved densities of the order of 25×10^4 individuals per square metre; numbers then declined markedly as summer progressed and water temperatures increased above 27°C, and the species was absent by August (recruiting again the following autumn/winter).

Tolerance of absolute-temperature stresses often differ for different life-stages of a species. The CW outfall canal of Kingsnorth Power Station (see above) supported a dense population of the boring bivalve *Petricola pholadiformis*, a North American species introduced into the United Kingdom in the late 19th century. Adults of this species bore into clay, mud, peat, etc., and have a life span of about ten years; the juveniles (spat) initially settle in mud or muddy sand, then migrate to firmer substrata into which they bore. During the first ten years of the operation of Kingsnorth Power Station, settled adults of *P. pholadiformis* were present in clays in the outfall canal, where they grew to maximal sizes; after ten years all were dead, having reached the end of their normal life span (ie, senescent) (Bamber, 1985b). During this period, while spat recruitment was recorded annually in the muds downstream, no recruitment occurred in the canal, so by the time of the senescent demise of the existing adults, the population had completely disappeared. While it is not possible to determine whether the lack of recruitment was due to a lack of settlement of juveniles at the prevailing temperatures, or to mortality of those which did settle, clearly the high temperatures prevailing in the canal were not deleterious to the adults, but were to the juveniles.

The concept of different tolerances for different stages was developed by Rasmussen (1973), who gives the relationship between temperature and spawning and settlement of many estuarine and coastal invertebrates. His extensive data show each species to have thresholds for spawning, larval development in the water column and settlement to the substrata. These data can be used to indicate the effects of a thermal-discharge-modified seasonal temperature curve on the spawning and development of the species found adjacent to north-west European coastal and estuarine power plants.

More extreme effects of raised temperatures occur when the temperature approaches levels at which species cannot survive – the thermal death point or upper incipient lethal temperature. ULT levels have not commonly been calculated for invertebrates (and less so for macroalgae and plants) but are generally around 30 to 33°C (regardless of latitude) (eg, Bamber, 1990). In summarizing the lethal temperatures for a large number of invertebrate species, Welch and Lindell (1980) found that the statistical mode for the group lay between 35°C and 40°C. Some examples are collated in Table 29.

The effectiveness of ULT levels is given by the fact that this absolute-temperature effect is utilized by some power stations in Europe and the United States for biofouling control: cooling water is temporarily recycled within the CW system to kill off fouling organisms. For example, La Spezia Power Station, Italy, recycles water at 35°C for 10 h to successfully control fouling by serpulid polychaetes, barnacles and mussels (Jenner *et al.*, 1998). At Eems Power Station, Netherlands, application of water at 38°C for 30 min was sufficient to kill the barnacle *Balanus crenatus* and the mussel *Mytilus edulis*.

Table 29 Upper incipient lethal temperatures for various marine invertebrates

Species	Stage	Acclimation temperature, °C	UILT, °C	Reference
Mytilus edulis (Common mussel)	Juvenile		6 hr: 33 (75%) 11 hr: 27 (50%)	LeBlanc *et al.*, 2005
	Adult		5 days: 32.5 Instant: 40–45	Jones *et al.*, 2009
Cerastoderma edule (Edible cockle)	Larvae		35	Kingston, 1974
	Adult	25	6 hours: 37 96 hours: 31	Ansell, 1981
Ostrea edulis (European oyster)	Adult	15	4 days: 38 6 hours: 40	Piano *et al.*, 2002
Crassostrea gigas (Pacific oyster)	Juvenile	15	32.5	Carvalho-Saucedo, 2003[a]
	Juvenile	22	37.5	Carvalho-Saucedo, 2003[a].
	Adult	20	1 hr: 43 (100%) 39–45 (50%)	Rajagopal *et al.*, 2005
Homarus gammarus (European lobster)	Larvae		34	Gruffydd *et al.*, 1975
Crangon crangon (Common shrimp)	Adult		30	Freitas *et al.*, 2007
Euphausiacea			mean 25.1	Yeung, 1983
Anostraca			mean 30.0	Yeung, 1983
Eucopepoda			mean 30.3	Yeung, 1983
Decapoda			mean 32.9	Yeung, 1983
Amphipoda			mean 34.4	Yeung, 1983

[a]In Sicard *et al.* (2006).

6.2.3 Tidally shifting temperature interface

The rapid movement of a temperature front across the seabed represents a greater stress than does the raised mean temperature, although it is limited to a smaller area. The thermal interface from the effluent at Kingsnorth Power Station (see above) was monitored at 5 min intervals over 18 months by Bamber and Spencer (1984). At the site, the temperature front progressed the entire length of the 4 km outfall creek on a tidal cycle, causing a variation of up to 12°C at the seabed within 15–30 min (Figure 29, DC1, DC2). At the control site (Figure 30, DC4) a residual front of 3°C was found on the first flood of the recirculating tide.
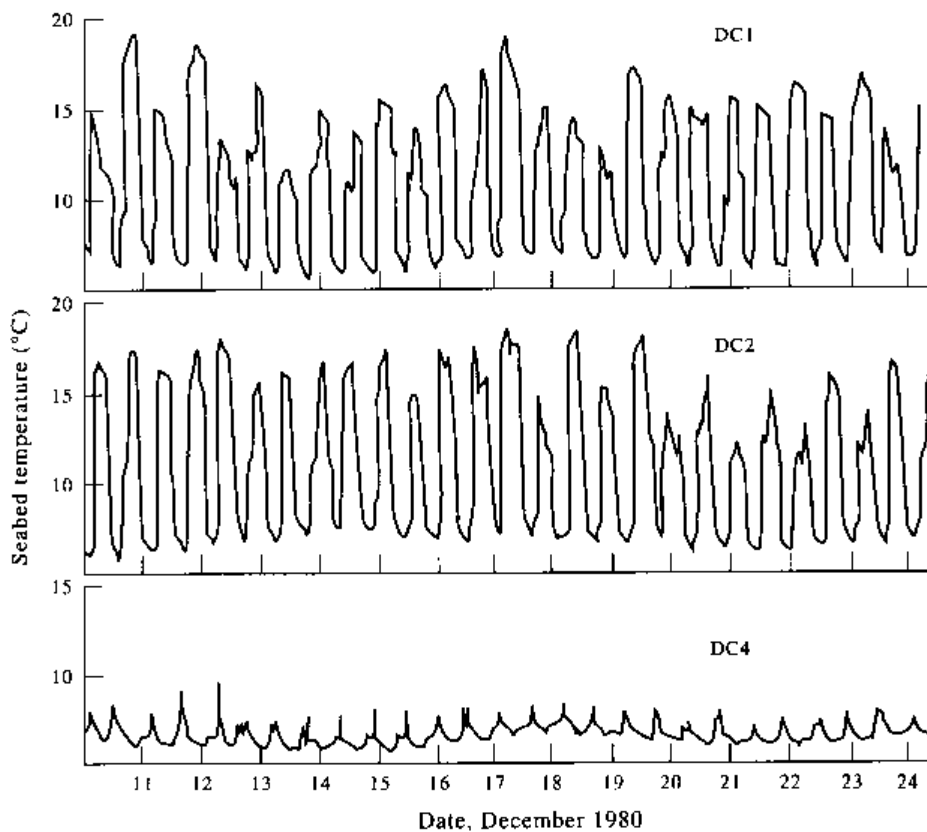
Figure 29 Continuous seabed temperatures in Kingsnorth Power Station CW discharge canal over two weeks in December 1980, starting at high tide, for sampling sites DC1 (1 km from the outfall), DC2 (1.7 km) and the control site DC4 (4 km)
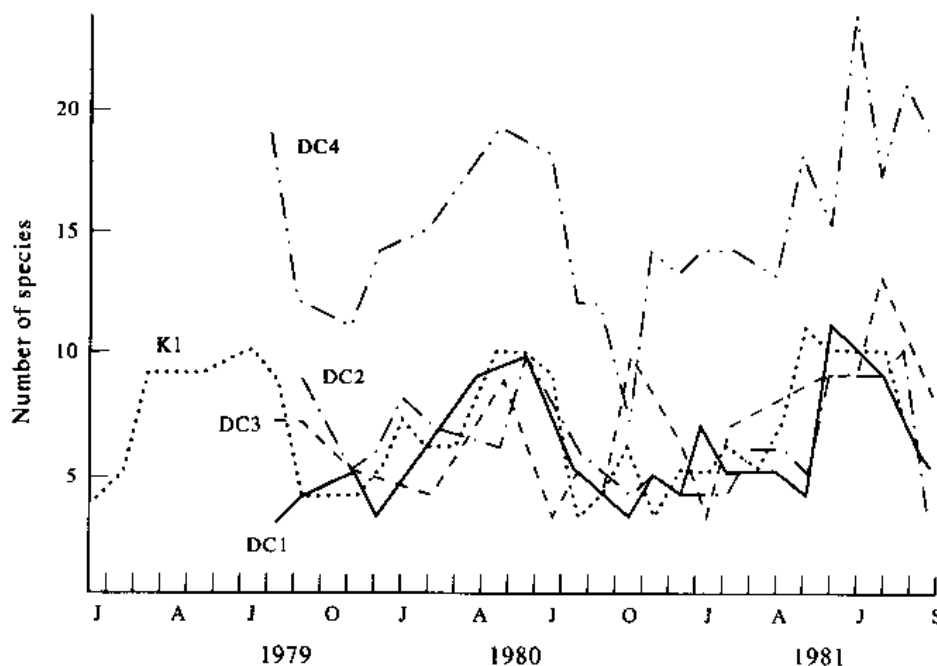
Figure 30 Numbers of benthic infaunal species in Kingsnorth Power Station CW discharge canal over three years; sampling sites K1 and DC1 to DC3 are within the canal, DC4 is the control site (after Bamber and Spencer, 1984)

As a result, the number of species present at each of four studied sites along the length of the creek was consistently half of that at the control site (Figure 30), irrespective of the gradient of mean temperature (see above). The species not eliminated by this interface stress were all species whose range included the littoral zone – that is, species naturally more tolerant of temperature fluctuation. Equally, the results from the control site indicate that a tidally moving thermal front of up to 3°C was tolerable to these species. That the species eliminated by the tidally shifting plume-interface included all molluscs and the ostracod *Eusarsiella zostericola* suggests that such extreme temperature stress may interfere with the physiology of shell formation, but no work has been done on this.

6.2.4   Thermal barriers

Generally, migratory marine invertebrates are not constrained to narrow corridors; thus thermal barriers to their movement would only apply in restricted water bodies such as estuaries, and then only where the plume extends right across the water body.

Littoral species that are excluded by the direct impact of an effluent plume (eg, *Gibbula umbilicalis* at Hinkley; Bamber and Coughlan, 1987) would also not attempt to recolonize, but this is not really an example of a thermal barrier as they would not survive in the impacted area if they were to migrate into it.

Species migrating on the seabed would be below a surface and less-dense effluent plume, so would not experience a problem, unless the less-dense plume water were somehow in contact with the seabed right across a restricted waterway, a most unlikely occurrence in estuaries and not possible in the open sea.

Species that disperse via planktonic larvae can only be affected by an effluent plume as it mixes with the receiving water, and thus as it dilutes; as they are not capable of gross directed movement, but rather float within the water body, the concept of thermal barriers does not apply. In practice such larvae are produced in

very high numbers, and suffer a high natural mortality (mainly through predation), so any deleterious impact in what must be localized areas of water would not be significant.

Nektonic migrants in the water column, such as mysids, the pelagic prawn *Pasiphaea sivado*, or the swimming crab *Polybius henslowi* tend to be widespread in the water column. All of these occur in the vicinity of Hinkley Point Power Station. The larger decapods can detect and avoid temperatures perceived to be deleterious but are not restricted to surface waters and have a sufficient area of water body in which to navigate around a plume. Mysids show extensive seasonal migrations in the Severn Estuary and the Bristol Channel, moving both upstream and downstream of Hinkley Point, but studies on their seasonal patterns have shown that their position in the Estuary is constrained by salinity rather than by temperature (Bamber and Henderson, 1994). Hence, in practice, thermal barriers are only a potential issue with fish (eg, Bamber, 1994), as addressed in Chapter 5.

6.3   The influence of climate change

Ocean temperatures increase more slowly than land temperatures because of the larger effective heat capacity of the oceans and because the ocean loses more heat by evaporation. The Northern Hemisphere warms faster than the Southern Hemisphere because it has more land. Changes in regional climate are expected to include greater warming over land, with most warming at high northern latitudes, and least warming over the Southern Ocean and parts of the North Atlantic Ocean (Pachauri and Reisinger, 2007). Current models of climate change range in prediction from decreases to increases in seawater temperatures around the British Isles (eg, Section 11.3.3.1. in Solomon *et al.*, 2007, UKCP09, 2011).

Most marine animals have comparatively short generation times, mainly annual or even iteroparity within one year. As a consequence, evolutionary pressures act rapidly to select for genotypes more adapted to environmental changes, including temperature rise.

The northern sandy-beach amphipod *Bathyporeia sarsi* reaches the southern (warmer) limit of its distribution on the south coasts of England. In a study of a beach community in Hampshire over 11 years, Bamber (2005) found *B. sarsi* to be the dominant macrobenthic species; the population bred twice per year. Three-monthly running-mean seawater temperatures over the years 1988–99 varied between –2.2 and +3°C in comparison with a 5 yr average temperature baseline for the site (eg, Figure 31).
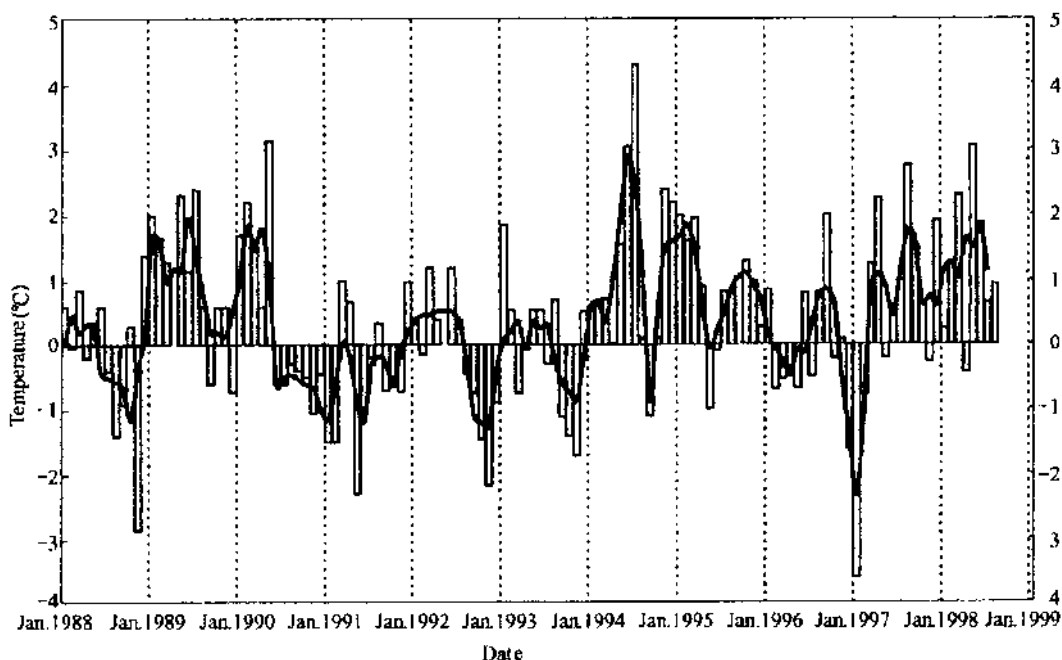
Figure 31 Deviations (ΔT) from the 5 yr mean seawater temperatures at Stanswood Bay, Hampshire (monthly, vertical bars) with connected 3 yr running mean deviation, for the years 1988–99

During warmer years, expressed by the average ΔT between November and March each year, the overwinter population size of *B. sarsi* was half as dense as in cooler years, and summer recruitment was delayed by one month and again generated about half the density recorded during cooler years. During cooler years, the population recovered.

What is notable here is that a proportion of the *B. sarsi* population remains at the site during 'adverse' warmer periods and survives the conditions. As it is these individuals that are responsible for the main annual recruitment in the summer, there will thus be a continuous evolutionary selection for those genotypes that are more tolerant of warmer temperatures, as these individuals will produce more offspring more often. Hence it is not valid to presume that temperature rises of the order of 1–2°C per decade will have a deleterious effect on such a species; *B. sarsi*, a short-lived species at the thermal limit of what we should regard as its current range, clearly shows the potential to adapt to rising temperature.

In fact, this will be the case with most marine coastal species in the seas around the British Isles which live in an environment that shows year-to-year temperature variations greater than the total temperature rise predicted by the Intergovernmental Panel on Climate Change (IPCC) over the next century; given rapid generation times (eg, annual), they would be expected to adapt to this environmental change. There is therefore no reason to believe that levels of seawater temperature-rise hypothesized by climate-change models will have any significant synergistic effect with those of CW effluents. In fact, the only case where an increased background temperature (ie, of the CW intake water) can combine with the CW system thermal enhancement to cause a significant deleterious effect is where this combination raises the effluent temperature to UILT levels (see Section 6.2.2), a scenario easily mitigated by proportionate constraint on the ΔT across the power station condensers (eg, by an increased CW flow rate).

6.4 Phenology

The thermal input to habitats exposed continuously or continually to a heated CW effluent will maintain an underlying higher temperature regime throughout the year, which can potentially influence behavioural and physiological processes in organisms living in that habitat. In temperate climates, this raised temperature can alter the timing of seasonality, as well as interfering with low-temperature-physiology events.

The study on the amphipod *U. brevicornis* by Barnett (1971) cited in Section 6.2.1.3 showed an earlier onset of breeding, with earlier recruitment (by up to 2 months) and more prolonged growth than the control population, although with no difference in growth rate. Similarly, Pearson and Barnett's (1987) study on *Tellina* and *Arenicola* illustrates the consequences of thermal elevation in relation to spawning thresholds and seasonal spawning times. Equally, the complex modification of the life-cycle of the isopod *C. carinata* in a heated effluent is described in Section 6.2.1.2, wherein the advantageous thermal regime offered by the CW effluent conditions allowed a species highly modified to cold winters to breed more, live longer and achieve a very high population density.

The gradient response of the shore barnacle *S. balanoides* in response to mean temperature has been described in Section 6.2.1.4. This northern species requires low winter temperatures for the onset of gametogenesis, and it has been found that temperatures >10°C can delay fertilization indefinitely in this species (Crisp, in Naylor, 1965). While populations in the path of a CW discharge may be locally reduced in their density, they will be affected further by an inhibition of their reproduction.

The detailed study of *B. sarsi* described in the previous section indicated that the spring population recruitment showed a negative correlation with overwinter temperature deviation, and a positive correlation with the size of the population during the previous summer. Recruitment during warmer years was earlier, and the size of the autumn recruitment greater, probably reflecting faster growth to maturity in higher temperatures (Bamber, 2005).

Overall, it is to be expected that the effects of raised temperatures will be expressed most conspicuously through their influence on reproductive processes, with resulting changes in season of maturation, increases (or decreases) in generation time, time to maturity and, of course, recruitment. As an example of geographical (latitudinal) large-scale changes in growth rate and longevity, Elliott and McLusky (1985) showed the patterns for the bivalve *M. balthica*. With increasing latitude, and thus decreasing temperatures, the growth rate decreased and the longevity increased. Given that each community is composed of many species each with differing responses, the responses within a community will be as diverse as the species within that community, and thus unpredictable.

At the same time, evolution will act on localized populations to select for genotypes that are adapted better to the local conditions, including those modified by a CW effluent. Population adaptation through evolutionary selection was shown in the Wadden Sea for the introduced oyster *Crassostrea gigas*. Reise (1998) and Wehrmann *et al*. (2000) recorded the first autochthonous larval settlement of this species, in a habitat where it had been introduced under the assumption that water temperatures were too low to allow reproduction, and therefore the species would not become invasive. There are recent indications of the introduced oyster breeding in south-west England as the result of spat from cultured organisms, again suggested as the result of both acclimation to local conditions and increased warming of waters (K. Hiscock, pers. comm.).

6.5 Xenobionts

There has been popular conjecture that higher temperatures encourage colonization by introduced (exotic) species. Xenobiotic species are continually being brought to the waters of the United Kingdom, by drift in oceanic currents, on sea-turtles and by anthropogenic means such as in the ballast water or on the hull-fouling of ships (see Eno *et al.*, 1997). These are variously referred to as non-native species, alien species, and – in the case of those that settle successfully and expand to have a deleterious impact on native species

– invasive species. Elliott (2003) used the term biological pollutants for those invasive species that create harm in the marine system. As some of these arrivals do so by natural means (ie, not by anthropogenic means), the preferred term is xenobiotic species (or xenobionts) (Elliott, 2003; Olenin *et al.*, 2007). It can also be difficult to distinguish between those carried into native waters by vessels, by flotsam or by other animals, and those that are merely spreading their native range, as all species are attempting to do continuously.

There are examples of species from outside the United Kingdom settling at sites within the influence of power station CW effluents. These are invariably examples of species introduced rather than immigrants, as such species would have to cross large distances through colder waters on the off-chance of finding suitable warmer conditions such as those found around a power station. As an example of this, the Mediterranean crab *Brachynotus sexdentatus* was recorded in Swansea Dock at the time that Tir John Power Station was discharging its effluent into that dock, but it subsequently died off once the artificial heating stopped (Naylor, 1965; Clark, 1986). In the same dock, the northwest Atlantic crab *Neopanope sayi* appeared and has remained after the cessation of the thermal effluent; the temperature range at Swansea is within the crab's natural range, and its appearance there was more a reflection of the ability of ships to transport xenobionts from dock to dock than any thermal effluent effect.

Barnacles are readily transported in ship hull-fouling, to the point that many species now show a worldwide distribution within their preferred ranges of latitude. The pan-subtropical species *Amphibalanus amphitrite* has been recorded regularly from sites in the United Kingdom, invariably from harbours and similar that are warmed artificially by thermal effluents. Recorded settlements do not survive overwinter in normal UK temperatures, and those at power station discharges do not persist after closure of the station (eg, Coughlan, 1977).

Mercenaria mercenaria, the hard-shell clam, is an introduced species in restricted areas on the south coast of England. It has been suggested, but not confirmed, that it's presence and reproductive cycles have been influenced by the presence of warmer water from power stations (Ansell *et al.*, 1964; Barnett, 1972).

6.6   Effects on primary productivity including algal blooms

The influence of increased water temperature on primary productivity and the potential for encouraging microalgal blooms is discussed in Section 4.3. In practice, any increased production and reproduction of algae in response to thermal discharges at open coastal locations and in larger estuaries will not result in localized concentration of the algae, as there is sufficient water exchange to prevent the build-up of blooms. Confined water bodies such as harbours would be more at risk from bloom formation, especially where nutrient levels are not limiting, owing to their more limited water exchange.

Some data for macroalgal temperature tolerance are given by Lüning (1984), but these are based on cold-water populations from Helgoland, so are of limited application. For example, their quoted ULT for *Chorda filum* of 23°C is inappropriate to those dense populations occurring on more southerly shores such as those of the Isles of Scilly or the Bay of Biscay, where such a temperature normally naturally occurs in summer. Breemer (1988) pointed out that, in some species, ecotypic differentiation in thermal responses over the distribution range influenced the location of geographic boundaries, and Lüning *et al.* (1986) explore the tolerance of *Chondrus* species by population, giving a ULT of 28°C for *Chondrus crispus*. Some of these data were collated by Langford *et al.* (1998).

The complexity of temperature control on the distribution of macroalgae was demonstrated by Breeman (1988), who found differential tolerance of different life-stages as well as differential temperature requirements for processes such as reproduction. He states that while Amphi-Atlantic (Arctic-) temperate species survived at subzero temperatures, in species with isomorphic life histories not specifically requiring low temperatures for reproduction, southern boundaries are set by lethally high summer temperatures on both sides of the Atlantic; none of these species survived temperatures over 30°C (which precludes their tropical occurrence). In most species with heteromorphic life histories (or crustose and erect growth forms),

low temperatures were required for formation of the macrothalli, so on European coasts southern boundaries to distribution are set by winter temperatures too high for the induction of macrothalli.

Selected lethal and sub-lethal toxicity data relating to temperature, based largely on cold-water populations as described above, is provided for selected marine algae species in Table 30 and Table 31, respectively.

Table 30 Selected lethal toxicity data relating to temperature (Langford *et al.*, 1998)

Lethal	
Species	**Threshold, °C**
Chorda filum	23
Chondrus crispus	28
Cladophora spp.	30–35
Chorda filum	23
Laminaria digitata	23
Laminaria saccharina	23
Laminaria hyperborea	18
Dumontia contorta	24
Fucus serratus	28
Lomentaria articulata	28
Desmarestia aculeata	25

Table 31 Selected sub-lethal toxicity data relating to temperature (Langford *et al.*, 1998)

Sub-lethal		
Species	**Effect**	**Threshold (°C)**
Fucus	Decline	5–7 >ambient
Ascophyllum nodosum	Cease releasing gametes	15
Chorda filum	Infertility	15
Chondrus crispus	Reproduction limit	15
Cladophora spp.	Reproduction limit	25
Chorda filum	Reproduction limit	15
Laminaria digitata	Reproduction limit	10
Laminaria saccharina	Reproduction limit	15
Laminaria hyperborea	Reproduction limit	8
Dumontia contorta	Reproduction limit	12
Lomentaria articulata	Gamete production	15
Desmarestia aculeata	Reproduction limit	15

6.7   Discussion and conclusions

The sections above give examples demonstrating that warmer-water species are more tolerant of higher temperature stresses than are colder-water species and that species whose distribution includes the littoral zone are more tolerant than those from the sublittoral. In addition, within species, different populations are adapted to different thermal tolerance as a result of selection within their ambient habitat.

It is the interrelationship between this evolutionary history, potentially resulting in some preadaptation to thermal-effluent temperature conditions, and the three main stress parameters of those conditions (mean, absolute and fluctuation in temperature) that will govern the response of marine organisms to power station CW discharges. Other than at outfall systems of a semi-enclosed configuration, significant thermal effects are restricted to within 200–500 m of the discharge, with some subtle effects possible slightly further afield. Within this affected range, it can be predicted that eurythermal species from the littoral, from estuaries and

coastal lagoons, and/or those from warmer biogeographic ranges will be unaffected or favoured, until ULTs are reached, while stenothermal species from the sublittoral, fully marine and/or cooler biogeographic areas will be deleteriously affected. The extent and degree of any impact (deleterious or otherwise) will be proportional to the volume, velocity and ΔT of the discharged effluent, other things being equal.

This leads to the question of what actual levels of temperature, or of temperature rise, are potentially deleterious. Actual data from field studies are limited (largely to those examples quoted above), owing to the history of underfunding of longer term studies necessary to distinguish the effects of a thermal discharge from the natural variation in temperature that occurs in marine and estuarine habitats. Equally, laboratory studies are few and must be interpreted carefully owing to the unrealistic environment and conditions in which the test organisms are maintained. These limitations equally limit the confidence in general conclusions, while at the same time the known wide range of thermal tolerance shown by the spectrum of marine non-vertebrate taxa itself constrains the confidence in generalizations.

> The **absolute temperature** (see Section 6.2.2) will be deleterious as it approaches the ULT. For those non-vertebrate species in UK waters for which there are data, this would appear to be around 30–33°C over relatively extended periods. It is therefore necessary to ensure that the CW flow rate is sufficiently high to avoid such temperatures at the edge of the mixing zone of the CW effluent plume, and to ensure that the effluent temperature is not exacerbated by excessive recirculation of effluent water at the intake.

> There are also examples of non-lethal but severe effects at lower temperatures – eg, the mass emigration of *C. volutator* described above; temperature limits for these effects will be species specific, and so hardly predictable.

> Note that the absolute temperature is not the same as any mid- to long-term average temperature.

> The short-term **fluctuations in temperature** resulting from tidal shifting of the plume, identified above as the greatest impact of a CW effluent, albeit restricted to a limited area of the shore or seabed, have been shown to be acceptable at a ΔT of 3°C in English waters. Modelling of proposed effluent plumes must therefore of necessity model both the tidal movement of such an isotherm in order to determine its presence (if at all) on the seabed, and the area of impact of any tidally shifted interface of >3°C.

> A tolerable level of **change in temperature** *per se* is more difficult to define as, as has been seen above, deleterious impacts are expressed at different temperatures for different species and even for different life-stages. Equally, when an effluent-enhanced thermal regime is advantageous to a species present, the success of such a species (eg, enhanced productivity, or increased population size) may itself have a deleterious impact on its competitors.

> It is shown above that a sudden change in temperature of 3°C on a tidally shifting plume is acceptable; therefore, acceptable levels of a more general rise in temperature will be higher than this. However, the gradient response in tubificid oligochaetes (Figure 27) demonstrates that changed temperature regimes will have some influence. These effects will be species-specific, and even population-specific, as defined in the opening paragraph of this section, and hence can only be interpreted on a site-specific basis.

Such legislation as exists for marine and estuarine species is generally founded not on information from *in situ* tolerance of temperatures, but on laboratory results (on material from unspecified populations) and derived from freshwater data.

The Shellfish Directive (Anon., 2006a) arbitrarily states in Annex I: 'A discharge affecting shellfish waters must not cause the temperature of the waters to exceed by more than 2°C the temperature of waters not so affected.' It is apparent from the foregoing that this is naïve, as we have seen that a 3°C rise is acceptable. Furthermore, any tolerance is species-specific for a given habitat.

Thus the European oyster, *Ostrea edulis*, is a southern species whose native range includes the Mediterranean. In UK waters, it gains benefit, in terms of growth rate and reproduction, from raised temperatures in these waters, whereas a population living in Norwegian fjords requires a temperature of 25°C for spawning (FAO FIGIS, 2006). Newell *et al.* (1971) showed that in *O. edulis* the improvement of

filtration following warm acclimation offsets metabolic energy losses even at low food concentrations. Even if energy losses not accounted for in their study (at temperatures up to 30°C) were substantial, a positive index of energy balance could be maintained during the summer months at ration levels that commonly occur in inshore waters. Hence, populations either established or implanted (as in a fishery) into waters influenced by a CW discharge will be automatically acclimated to a warmer temperature regime, and would be expected to do better than those individuals not exposed to the raised temperature.

Particular concern in impact assessment is given to species of conservation importance (eg, species listed in the Habitats Directive; Anon., 1992), which is to say species that by definition are rare, threatened or fragile. Unfortunately, this means that investigations on the thermal tolerance of such species have not been undertaken, owing to the difficulty either of obtaining material for experimental work (and obtaining the relevant licences) or of studying such species in the field.

The available data reviewed above confirms that adverse effects of CW outfalls are restricted to an area close to the plume, that temperature rises up to 3°C appear to be tolerable, and that resulting temperatures of less than 27°C have no clear deleterious impact on species in the receiving waters, but, in the longer term, changes in the local community may result as species with differing tolerances of elevated temperature show differing survival, growth and patterns of reproduction from those expressed under ambient conditions. Furthermore, populations that persist adjacent to a heated CW effluent will acclimate to those new local conditions and evolve in response to them.

7 Final summary and conclusions

The abstraction and return of cooling water represent the most important environmental aspects of nuclear power station operation to the marine environment. The discharge introduces significant thermal energy (heat) to receiving waters, which will continue with little variation throughout the operational life of the station, which may exceed forty years. The temperature uplift is normally within the limits of natural variation, so serious acute impacts are unlikely. However, a modest temperature rise in the vicinity of the discharge is inevitable, with little practical opportunity for mitigation once the station is built. It is therefore essential that the thermal tolerances of key species are understood and that station design reflects the ability of the ecosystem to adjust to moderate chronic stress. The potential additive effects of long-term background temperature rises associated with predicted climate change must also be taken into account.

7.1 General conclusions

➢ Existing standards (even where they have statutory basis) cannot be relied upon to be a realistic guide for the protection of the marine environment; there has been uncritical transference of freshwater limits to marine waters, and there appears to have been a reliance on intuition – eg, in many cases there is an assumption that a CW discharge forms a thermal barrier to migratory fish, yet there is little experimental evidence to support this. Current standards are almost entirely based on known responses of fish and take no account of putative climate-change effects.

➢ Whereas previous thermal standards developed in the United Kingdom, primarily for application to freshwaters, have distinguished between salmonid and cyprinid waters, this distinction is inappropriate for transitional and coastal waters, where a much wider variety of fish species may be found. With reference to thermal responses of species, a more appropriate distinction can be drawn on zoogeographic lines, considering, for example, Arctic–Boreal and Lusitanian tendencies (cold- and warm-water species, respectively).

➢ To a large extent, fish and other swimming organisms can detect and actively avoid warmed waters if they perceive them to be harmful, making it unlikely that they will suffer significant impact from the shorter term temperature elevation. Equally, planktonic organisms in the receiving water will only be at risk as the effluent mixes and dilutes. It is therefore predominantly benthic organisms that are at risk of being affected by thermal effluents, but this risk is moderated by the buoyancy of the plume.

➢ While there is little evidence to suggest that existing thermal discharges have created barriers to cold-water migratory fish species such as salmon, sea trout, eel and smelt, the future situation must be considered against a background of longer term water temperature rise associated with climate change, which is gradually reducing the metabolic headroom for cold-water species. The much larger flows of cooling water planned for NNB stations could also be expected to have a greater potential for this type of effect, although planned temperature rises (ΔTs) will remain similar to those for existing thermal stations.

➢ Marine coastal invertebrate species in the seas around the British Isles live in an environment that shows natural year-to-year temperature variations greater than the total ΔT predicted by the IPCC over the next century. Given rapid generation times (eg, annual), they would be expected to adapt to this environmental change. Therefore, for invertebrates there is little reason to believe that levels of seawater temperature rise hypothesized by climate-change models will have any significant synergistic effect with those of CW effluents.

➢ The juxtaposition of fish zoogeographic data with sea-surface temperature data provides a good practical indication of the preferred temperature ranges of fish. Seasonal movements of fish away from fringe areas, and the ability of some fish to exploit cooler, deeper layers in some areas, may mean that

they do not actually experience the full ranges of temperatures inferred, as has been demonstrated, for example, in cod. With these caveats, the zoogeographic information provides a useful indication of temperature preferences of species for which there are no laboratory studies or other measurements.

➢ In March 2008, the UK Technical Advisory Group on the WFD published draft temperature standards for surface waters. UKTAG was unable to accept recommendations for TraC waters as it felt that there were insufficient data to support the recommendations. The more extensive information reviewed in the present report, now including invertebrate data as well as an expanded fish database, makes it appropriate to reconsider the temperature boundaries and allowable changes.

7.2   Recommendations for thermal limits

The temperature tolerance tables in Appendix C display the ranges of biological sensitivity drawn from Chapters 5 and 6. Based on the information shown, tentative boundary values are proposed. The boundary values are required to reflect the WFD normative definitions for all quality elements in TraC waters shown in Appendix B.

The draft proposals prepared for temperature boundaries by Turnpenny and Liney (2007) on behalf of UKTAG envisaged limits of 20°C, 23°C, 28°C and 30°C for High, Good, Moderate and Poor classes, respectively. Where appropriate, boundaries have been moved in the light of further information. Only one change has been recommended, which is a shift from 20°C to 23°C for the maximum allowable value suggested for High-status TraC waters. This arises not from new biological information, but from evidence of naturally higher summer temperatures in southern parts of the United Kingdom.

Preferred temperatures for reproduction shown in Appendix C indicate that additional seasonal limits would be more protective in areas used for reproduction.

As can be seen in Appendix C, the proposed boundary values are defined by the more sensitive species, which are well represented by fish.

7.2.1   Fish

Fish exhibit a wide variety of thermal sensitivities, those of Arctic–Boreal affinities (cold-water species) being the most likely to be disturbed by thermal discharges. Lethal temperatures, when used in conjunction with thermal mapping in the vicinity of discharges, provide an absolute indication of areas from which fish will be excluded. However, fish have a powerful ability to use behavioural thermoregulation in order to remain in areas of preferred temperature. Observed behavioural preferenda and avoidance thresholds, along with observed temperatures for optimum growth, therefore provide a better indication of how fish distribution is likely to be affected by areas of thermal enhancement.

As would be expected, these temperatures are lower for cold-water species. Since both warm- and cold-water species coexist in waters around England and Wales, the development of suitable boundary values needs to take account of both groups. Boundary values for upper-allowable temperatures therefore are best based on the more sensitive cold-water species, and therefore only this typology needs to be considered.

Special consideration needs to be given to seasonal spawning areas, where for some species narrow temperature limits appear to be required for spawning. It is not clear whether some apparently narrow limits are merely artefacts of inadequate observation or of the background temperatures to which fish have become acclimated. The information in Appendix C can be used in the interim to guide appropriate discharges into spawning areas. In principle, however, thermal discharges should be located so as to avoid important spawning areas.

Available data from monitoring of (cold-water) salmonid migrations through UK estuaries provide no evidence of so-called thermal blocks to diadromous fish migrations. This includes long-term (>20 years) observations of salmon movements through the Thames Estuary. Fish-tracking data from elsewhere have shown that, during the highly motivated river entry phase, salmon will cross abrupt temperature changes as high as 9–10°C. Where such movements are inhibited, low DO concentration is invariably a factor; provided that WFD DO standards are maintained, this factor could be excluded. However, according to some climate-change predictions, temperature *per se* could become inhibitory to fish movement. This aspect should be kept under regular review.

Appendix C recommends limits for allowable maximum water temperature that are appropriate to fish, based upon the expanded database presented in this report. The previous proposals of Turnpenny and Liney (2007) included a requirement to limit temperature uplift to +2°C for High-status TraC waters and to +3°C for other classes, provided also that absolute temperature limits would not be exceeded. The retention of uplift values within future standards is essential in order to avoid fish acclimation issues, phenological changes and potential thermal blockage that might be associated with extreme values. The previously recommended values remain appropriate.

7.2.2 Other taxa

For an area of habitat within a given isotherm within an effluent plume, it is entirely reasonable to assume that species inhabiting that area, and whose normal habitat range experiences that (raised) temperature naturally at that time of year, will not be significantly adversely affected.

Where the tidally shifting interface between effluent water and the ambient water contacts an area of seabed (or shore) habitat outside the mixing zone, it would be appropriate that the temperature differential across that interface should not be greater than 3°C.

Within British coastal and estuarine habitats, it is recommended that the temperature of the effluent water outside the mixing zone should not exceed 30°C for more than 6 h in any one week during summer, and not at all at other times of year. This numerical limit will aid the regulator when defining the mixing zone.

7.2.3 Marine plants

Under the Habitats Directive and WFD there is a requirement also to protect marine plants. The only available guidance is that reviewed in WQTAG160, the raw data from which were summarized in Section 6.6.

The lethal limit data are summarized in Appendix C, but it should be noted that these are largely based on cold-water populations.

7.3 Recommended boundary and uplift values

Under the WFD, boundary values have to be established to protect the most sensitive taxa; these would apply equally to the Habitats Directive.

Given the rationale in this report, Table 32 sets out recommendations for temperature boundary values.

In line with international good practice, it is also recommended that the mixing zone should not occupy more than 25% of the cross-sectional area of an estuarine channel as a 95%-ile.

Table 32 Proposed temperature boundary values for all TraC waters outside the mixing zone

Typology	Normative definition boundary positions (as annual 95%-ile)			
	High/Good	**Good/Moderate**	**Moderate/Poor**	**Poor/Bad**
Maximum Allowable Temperature*	23°C	23°C	28°C	30°C
Maximum Allowable Temperature Uplift	+2°C	+3°C	+3°C	+3°C

*These are general figures for England and Wales; it may be appropriate to adjust where maximum summer background water temperatures are higher or lower.

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Appendix A Glossary

Acclimation (temperature)	The process of the individual organism adjusting to a gradual change in temperature over a short period of time, under laboratory conditions
Arctic Boreal	Refers here to cold-water preference species
Biota	All plants and animals
BREF	Best Available Technique Reference Document
CW	Cooling water
Dissolved Oxygen (DO)	The elemental oxygen in water
Eurythermal	Tolerates a wide range of temperatures
Fjard	Lowland fjord
IPCC	Intergovernmental Panel on Climate Change
K_{OW} = Octanol	Water partition coefficient
LOEL	Lowest observed effect level
Lusitanean	Refers here to warm-water preference species
MZ	Mixing zone
NBS buffers	US National Bureau of Standards buffers are designed for use in freshwater
NOEL	No observed effect limit
Phenology	Timing of biological processes
Planktonic organisms	The collection of small or microscopic organisms that float or drift in great numbers in fresh or salt water, and serve as food for fish and other larger organisms
PNEC	Predicted no effect concentration
Preferendum	Limit of preferred range
SNIFFER	Scotland and Northern Ireland Forum for Environmental Research
Stenothermal	Tolerates only a narrow range of temperature
Thermocycling	Temperature changes resulting from cyclic operation of generating plant, usually over a 24 h period
TraC Waters	Transitional (estuarine and brackish lagoons) and coastal waters
TRO	Total residual oxidents
UKTAG	WFD UK Technical Advisory Group
ULT	Upper lethal temperature
WFD	Water Framework Directive
Xeniobionts	'Strange organism': exotic, introduced, or alien species

Appendix B Normative definition of WFD ecological status classification

General definitions for rivers, lakes, transitional waters and coastal waters

Element	High Status	Good Status	Moderate Status
General	There are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type under undisturbed conditions. The value of the biological quality elements for the surface water body reflects those normally associated with that type under undisturbed conditions and show no, or only very minor, evidence of distortion. These are the type-specific conditions and communities.	The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body under undisturbed conditions.	The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity, and are significantly more disturbed than under conditions of good status.

Waters achieving a status below moderate shall be classified as poor or bad.

Waters showing evidence of major alterations to the values of the biological quality elements for the surface water body type, and in which the relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions, shall be classified as poor.

Waters showing evidence of severe alterations to the values of the biological quality elements for the surface water body type, and in which large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent, shall be classified as bad.

Definitions for high, good and moderate ecological status in transitional waters

Biological quality elements

Element	High status	Good status	Moderate status
Phytoplankton	The composition and abundance of the phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type-specific physicochemical conditions.	There are slight changes in the composition and abundance of phytoplanktonic taxa. There are slight changes in biomass compared to the type-specific conditions. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water. A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	The composition and abundance of phytoplanktonic taxa differ moderately from type-specific conditions. Biomass is moderately disturbed and may be such as to produce a significant undesirable disturbance in the condition of other biological quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
Macroalgae	The composition of macroalgal taxa is consistent with undisturbed conditions. There are no detectable changes in macroalgal cover due to anthropogenic activities.	There are slight changes in the composition and abundance of macroalgal taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water.	The composition of macroalgal taxa differs moderately from type-specific conditions and is significantly more distorted than at good quality. Moderate changes in the average macroalgal abundance are evident and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.
Angiosperms	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities.	There are slight changes in the composition of angiosperm taxa compared to the type-specific communities. Angiosperm abundance shows slight signs of disturbance.	The composition of the angiosperm taxa differs moderately from the type-specific communities and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa.

Definitions for high, good and moderate ecological status in transitional waters

Biological quality elements

Element	High status	Good status	Moderate status
Benthic invertebrate fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.	The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions. Most of the sensitive taxa of the type-specific communities are present.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the type-specific conditions. Taxa indicative of pollution are present. Many of the sensitive taxa of the type-specific communities are absent.
Fish Fauna	Species composition and abundance is consistent with undisturbed conditions.	The abundance of the disturbance-sensitive species shows slight signs of distortion from type-specific conditions attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements.	A moderate proportion of the type-specific disturbance-sensitive species are absent as a result of anthropogenic impacts on physicochemical or hydromorphological quality elements.

Definitions for high, good and moderate ecological status in coastal waters

Biological quality elements

Element	High Status	Good Status	Moderate Status
Phytoplankton	The composition and abundance of phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type-specific physicochemical conditions.	The composition and abundance of phytoplanktonic taxa show slight signs of disturbance. There are slight changes in biomass compared to type-specific conditions. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the quality of the water. A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	The composition and abundance of planktonic taxa show signs of moderate disturbance. Algal biomass is substantially outside the range associated with type-specific conditions, and is such as to impact upon other biological quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
Macroalgae and angiosperms	All disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present. The levels of macroalgal cover and angiosperm abundance are consistent with undisturbed conditions.	Most disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present. The level of macroalgal cover and angiosperm abundance show slight signs of disturbance.	A moderate number of the disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are absent. Macroalgal cover and angiosperm abundance is moderately disturbed and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.
Benthic invertebrate fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.	The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions. Most of the sensitive taxa of the type-specific communities are present.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the type-specific conditions. Taxa indicative of pollution are present. Many of the sensitive taxa of the type-specific communities are absent.

Appendix C Summary tables of thermal tolerances

KEY:

| FFD (22) | ──── | **Good (23)** | ──── | **High (23)** | ──── | **Mod. (28)** | ──── | **Poor (30)** | ──── |

Temperature tolerance of fish

Species	Common name	Criterion	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Salmo salar	Salmon	Lethal																				Salmon															
Gadus morhua	Cod												Larval										Adult														
Engraulis encrasicolus	Anchovy										Larval																										
Salmo trutta	Sea trout												Eggs									Juvenile / Adult															
Clupea harengus	Herring																																				
Pleuronectes platessa	Plaice																																				
Pomatoschistus minutus	Sand goby																									Adult											
Platichthys flesus	Flounder																						Larval														
Psetta maxima	Turbot																						Adult														
Sprattus sprattus	Sprat													Eggs																							
Petromyzon marinus	Sea lamprey																							Larval													
Alosa alosa	Allis shad																							Eggs		Larval										
Solea solea	Dover sole																																				
Dicentrarchus labrax	Sea bass																									Juvenile										
Anguilla anguilla	Eel																																				

Species	Common name	Criterion	Temperature °C
			5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39
Agonus cataphractus	Pogge	Optimal Growth	
Limanda limanda	Dab		Adult / Juvenile
Osmerus eperlanus	Smelt		Adult
Syngnathus rostellatus	Pipefish		Adult
Salmo salar	Salmon		Eggs / Juvenile / Adult
Gadus morhua	Cod		Juvenile / Larval
Merlangius merlangus	Whiting		Adult
Engraulis encrasicolus	Anchovy		Larval >20
Salmo trutta	Sea trout		Adult / Eggs / Juvenile
Clupea harengus	Herring		Juvenile
Trisopterus luscus	Pouting		Adult
Pluronectes platessa	Plaice		Adult
Pomatoschistus minutus	Sand goby		Adult
Platichthys flesus	Flounder		Adult
Psetta maxima	Turbot		Adult / Juvenilee
Sprattus sprattus	Sprat		
Alosa fallax	Twaite shad		Larval
Petromyzon marinus	Sea lamprey		Adult / Egg
Alosa alosa	Allis shad		Egg / Adult / Egg
Solea solea	Dover sole		Egg / Adult / Larval / Juvenile
Atherina presbyter	Sand smelt		
Mullus surmuletus	Red mullet		Adult
Scyliorhinus canicula	Dogfish		Egg
Dicentrarchus labrax	Sea bass		Juvenile / Adult
Anguilla anguilla	Eel		Juvenile / Adult

Species	Common name	Criterion	Temperature °C																																		
			5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Syngnathus rostellatus	Pipefish	Preferred										▓																										
Salmo salar	Salmon	Preferred					▓	▓	▓	▓	▓	▓	▓	▓	▓																							
Gadus morhua	Cod	Preferred	▓	▓																																		
Salmo trutta	Sea trout	Preferred				▓	▓	▓	▓	▓	▓	▓	▓	▓	▓																							
Clupea harengus	Herring	Preferred							Larvae / Juveniles																													
Trisopterus luscus	Pouting	Preferred							▓																													
Pleuronectes platessa	Plaice	Preferred												▓																								
Pomatoschistus minutus	Sand goby	Preferred			Autumn									Summer																								
Alosa fallax	Twaite shad	Preferred																> 21.5 (larvae)																				
Petromyzon marinus	Sea lamprey	Preferred									▓					▓	▓																					
Atherina presbyter	Sand smelt	Preferred						▓	▓	▓																												

Species	Common name	Criterion	Spawning temperature (°C)
Osmerus eperlanus	Smelt	Spawning	11–13
Salmo salar	Salmon	Spawning	Freshwater spawners - not applicable
Gadus morhua	Cod	Spawning	5–10
Leuciscus leuciscus	Dace	Spawning	8–9
Lampetra fluviatilis	River lamprey	Spawning	Freshwater spawners - not applicable
Engraulis encrasicolus	Anchovy	Spawning	Surface temp 14–20
Salmo trutta	Sea trout	Spawning	Freshwater spawners - not applicable
Clupea harengus	Herring	Spawning	5–16
Platichthys flesus	Flounder	Spawning	11–13
Pomatoschistus minutus	Sand goby	Spawning	8–16
Psetta maxima	Turbot	Spawning	14
Alosa fallax	Twaite shad	Spawning	Freshwater spawners - not applicable
Petromyzon marinus	Sea lamprey	Spawning	Freshwater spawners - not applicable
Alosa Alosa	Allis shad	Spawning	Freshwater spawners - not applicable
Liza aurata	Golden grey mullet	Spawning	20–22
Pomatoschistus microps	Common goby	Spawning	16–22
Atherina presbyter	Sand smelt	Spawning	11–19
Solea solea	Dover sole	Spawning	17–19
Dicentrarchus labrax	Sea bass	Spawning	10–12
Anguilla anguilla	Eel	Spawning	20–23

Species	Common name	Criterion	Temperature C
			5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Liparis liparis	Striped sea snail	Geographic																																				
Agonus cataphractus	Pogge																																					
Limanda limanda	Dab																																					
Osmerus eperlanus	Smelt																																					
Syngnathus rostellatus	Pipefish																																					
Salmo salar	Salmon																																					
Gadus morhua	Cod																																					
Merlangius merlangus	Whiting																																					
Leuciscus leuciscus	Dace																																					
Lampetra fluviatilis	River lamprey																																					
Engraulis encrasicolus	Anchovy																																					
Salmo trutta	Sea trout																																					
Clupea harengus	Herring																																					
Trisopterus luscus	Pouting																																					

Species	Common name	Criterion	Temperature °C (green band)
Pleuronectes platessa	Plaice	Geographic	5–26
Pomatoschistus minutus	Sand goby		5–24
Platichthys flesus	Flounder		5–24
Psetta maxima	Turbot		5–26
Sprattus sprattus	Sprat		5–26
Alosa fallax	Twaite shad		5–26
Petromyzon marinus	Sea lamprey		5–24
Alosa alosa	Allis shad		5–26
Liza aurata	Golden grey mullet		7–26
Liza ramada	Thin-lipped grey mullet		6–26
Chelon labrosus	Thick-lipped grey mullet		5–28
Pomatoschistus microps	Common goby		7–27
Solea solea	Dover sole		5–28
Atherina presbyter	Sand smelt		7–27
Mullus surmuletus	Red mullet		5–27
Sardina pilchardus	Sardine		5–28
Dipturus batis	Skate		5–28
Scyliorhinus canicula	Dogfish		6–28
Dicentrarchus labrax	Sea bass		5–27
Anguilla anguilla	Eel		5–28

Temperature scale (°C): 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39

Temperature °C — Criterion: Avoidance

Species	Common name	Criterion	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Osmerus eperlanus	Smelt	Avoidance																						Juveniles													
Salmo salar	Salmon																																				
Gadus morhua	Cod								Juveniles															Juveniles													
Leuciscus leuciscus	Dace																								Juveniles												
Salmo trutta	Sea trout															Adults								Juveniles													
Pomatoschistus minutus	Sand goby																																				
Anguilla anguilla	Eel													Adults										Elvers													

Temperature °C — Criterion: Migration

Species	Common name	Criterion	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Osmerus eperlanus	Smelt	Migration																																			
Salmo salar	Salmon							Smolt							Adult								Adult														
Lampetra fluviatilis	River lamprey																																				
Salmo trutta	Sea trout																																				
Pomatoschistus minutus	Sand goby																																				
Platichthys flesus	Flounder																																				
Alosa fallax	Twaite shad																																				
Alosa alosa	Allis shad																																				
Pomatoschistus microps	Common goby																																				
Solea solea	Dover sole																																				
Dicentrarchus labrax	Sea bass																																				
Anguilla anguilla	Eel																																				

Temperature tolerance of marine plants

KEY:

FFD (22) ━━━	Good (23) ━━━	High (23) ━━━	Mod. (28) ━━━	Poor (30) ━━━

Species	Common name	Criterion	Temperature °C
			5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39
Chondrus crispus	Carragheen	Lethal	
Cladophora spp			
Chorda filum	Thong weed		
Laminaria digitata	Oarweed		
Laminaria saccharina	Sugar kelp		
Laminaria hyperborea	Cuvie		
Dumontia contorta			
Fucus serratus	Serrated wrack		
Lomentaria articulata			
Desmarestia aculeata	Landlady's wig		

Temperature tolerance of marine invertebrates

KEY:

FFD (22) ▬	Good (23) ▬	High (23) ▬	Mod. (28) ▬	Poor (30) ▬

Species	Common name	Criterion	Temperature °C																																
			5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Mytilus edulis	Common mussel	Lethal																													🟥	🟥	🟥	🟥	🟥	🟥	🟥
Cerastosderma edule	Edible cockle																													🟥	🟥	🟥	🟥	🟥	🟥	🟥
Ostrea edulis	European oyster																																			🟥
Crassostrea gigas	Pacific Oyster																														🟥	🟥	🟥	🟥	🟥	🟥
Crangon crangoon	Common shrimp																											🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥	🟥
Homarus gammarus	Lobster **LARVAE**																												🟥					🟥	🟥	🟥	🟥