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Evaluating options for sea trout and brown trout biological reference points

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Steve Killeen

Head of Science

Executive summary

Biological reference points (BRPs) provide a means of assessing the status of fish stocks against a defined reference condition. These assessments help to identify and prioritise management actions to ensure that fish populations reach a favourable status. The Environment Agency uses BRPs to assess salmon populations in rivers in England and Wales. However, a BRP method has not been applied to any other freshwater fish species in England and Wales, largely due to the dearth of species-specific data and information such as lifecycle variation.

The Salmon and Freshwater Fisheries Review (2000) and National Trout and Grayling Fisheries Strategy (2003) recommend that a BRP method be established for trout populations in England and Wales, as the trout is an important commercial and recreational resource.

The Environment Agency’s management objectives for trout are to:

- optimise freshwater production of trout;
- optimise recruitment to homewater fisheries, that is, rod and net fisheries (in stocks with a major migratory component);
- maintain and improve the diversity and fitness of stocks;
- consider sustainable catch potential (for both rod and net catches).

A further key consideration in the management of trout stocks is socio-economics; however, this does not fall within the scope of this project.

A number of factors make the use of BRPs for trout populations problematic, including biological, monitoring and management factors. In terms of the biology of the species, the trout exhibits two broad life-history types – the sea trout migrates to sea to feed and mature, while the freshwater resident trout completes its lifecycle in freshwater. In fact, many authors would argue that life-history tactics in trout cannot be classified solely as anadromous or freshwater resident, but rather as a continuum of life-history tactics in space and time. This idea is based on the principle that migration is likely to be a trade-off between the costs of migrating and the benefits of the environment, regardless of the distance and environment travelled.

The trout is a highly polymorphic and ecologically variable species whose life-history traits are phenotypically plastic in response to environment and genetic parameters. Because aquatic ecosystems present a continuous gradient of physical conditions, the trout can exhibit a continuum of life-history tactics to optimise individual fitness and population persistence. There is thus considerable variability in life-history tactics among individuals and populations of trout, with a number of different migration patterns described. Differences between the sexes occur, with males typically having a higher tendency to remain in the natal river, whilst females are more likely to migrate to sea. In setting biological reference points for trout, it will, however, be necessary to understand the contributions of both anadromous and freshwater-resident forms to total trout production.

A number of monitoring issues arise because the data required to develop and use BRPs for trout are not matched by current Environment Agency monitoring programmes. Data on rod catches of freshwater resident trout are largely lacking due to the current lack of a national catch-return system for non-migratory trout. Rod effort data are currently only available as ‘rod licence’ days for salmon and sea trout combined. Age data collected during routine juvenile surveys are limited in some areas. Freshwater age structure of trout is likely to be a key factor in understanding the causes and distribution of anadromy within individual catchments.

Management issues arise as a result of the different licensing, catch recording and other regulatory practices applied to sea trout and freshwater resident trout. For instance, regulatory controls for sea trout and freshwater resident trout often differ considerably, due mainly to the differences in locations and methods used by their respective fisheries. Whilst such an approach is inconsistent with the taxonomy of the trout, it does make sense in practical terms.

A single biological reference point method for trout is unlikely to satisfy all proposed management objectives, primarily because of the complexity of the trout's lifecycle, but also because of the way in which monitoring data is currently collected and in how the species is managed. A more suitable reference point approach would likely involve a hierarchy of diagnostic stages, with different reference points for different life stages, perhaps taking a similar form to the existing Salmon Lifecycle Model that incorporates all forms of monitoring into one assessment model. An equivalent Trout Lifecycle Model" would theoretically be possible, and would capitalise on the River Fisheries Habitat Inventory (SC040028) which was developed in support of the Salmon Lifecycle Model (SC020077).

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1 Introduction

This report explores the feasibility of establishing biological reference points (BRPs) for brown trout (*Salmo trutta* L.), as set out in the Environment Agency's National Trout and Grayling Fisheries Strategy (Environment Agency 2003). BRPs set reference levels of stock and fishery performance for management, monitoring and assessment purposes. The ideas outlined here have been used in the marine fisheries field for many years and more recently, have been applied to salmon fisheries in England and Wales and elsewhere (Crozier *et al.* 2003).

Nevertheless, the use of BRPs for trout is new and presents a number of difficulties that stem from the biology of the species and its monitoring and management. The principal biological issue is that the species has two broad life-history types – one that goes to sea as a juvenile (anadromy) to become sea trout and the other that remains as a freshwater resident (FR) brown trout. A further issue is that the data required to develop and use BRPs may not match current monitoring programmes. In terms of management, different licencing, catch recording and other regulatory practices apply to sea trout versus FR trout. These and other matters are discussed in the report, which explores options for the use of BRPs within the Trout and Grayling Fisheries Strategy.

The overall aims of the project are to:

- provide a framework of policy objectives and biological reference points;
- describe the distribution of migrant and non-migrant trout in England and Wales;
- establish a list of rivers for which biological reference points may be required;
- provide an assessment and statement of feasibility of possible approaches;
- establish the feasibility of each approach using test data on the River Tamar;
- provide a specification of national data needs, comparing current availability in order to evaluate development and costs of implementation;
- propose any further development of biological reference points for trout;
- outline the risks and uncertainties involved in setting biological reference points.

Box 1 Nomenclature

Trout nomenclature has been subject to intense debate for over a century and Ferguson (2006) discusses the issues. The term 'brown trout' is used by taxonomists conventionally to refer to *Salmo trutta* L, a species having many life-history types, including anadromy (Laikre *et al.* 1999). The term has been adopted in common parlance to mean the non-migratory form, which can be confusing. Ferguson (2006) suggests the use of the term freshwater resident for the non-anadromous form and sea trout for the anadromous form. Even that has complications, as there are degrees of anadromy and some sea trout parents (males) do not migrate to sea. In this report we use "brown trout" or "trout" to cover all morphs where the distinction is unimportant, "FR" to mean non-migratory trout, and "sea trout" to cover sea trout. This should cover most eventualities and where not, specific note is made.

1.1 Background

The trout (*Salmo trutta L.*) is widespread throughout England and Wales, occurring in both running and still waters. It is a polymorphic species that displays a range of migratory behaviour in different environments (Regan 1911; Trewavas 1953 and Thorpe 1987 cited in Harris 2002). Two major life-history types have been identified: trout that remain resident in freshwater for their entire lifecycle, and migratory sea trout which migrate to sea to feed and grow before returning to freshwater to spawn (Jonsson 1985; Harris 2002; Elliott 1994). It is generally accepted that these two life strategies represent the two extremes of a continuum of migratory patterns (Ferguson 2006). In many populations, sea trout interbreed with (typically male) FR trout, and offspring can exhibit either life-history trait (Hindar *et al*. 1991; Giger *et al*. 2006). Some of this life-history variation is known to have a genetic basis, but may also reflect adaptation to the environment (Environment Agency 2003). Much of the available literature suggests that anadromy in brown trout is best described as a 'threshold quantitative trait'; a trait controlled by multiple genes and by environmental influences that is expressed when this combination of factors exceeds a threshold level (Ferguson 2006).

The anadromous form is widely distributed in varying degrees of abundance (Harris 2002). In England and Wales, there are approximately 100 rivers for which sea trout rod catches have been reported. A large number of smaller coastal streams are known to have sea trout, but are not necessarily exploited. Sea trout generally make greater use of such small streams and tributaries for rearing, and the small size of these waters can make them more vulnerable to environmental pressures (Milner *et al*. 2006). It is therefore likely to be these smaller streams that require the greatest protection, but these are also likely to be the most difficult streams to manage effectively, owing to the paucity of available information on stocks.

The brown trout has been the subject of much research in recent years, and there is a wealth of literature on many aspects of its ecology and population dynamics. Despite this, there remains a number of unresolved questions on the ecology, life history and population dynamics of the anadromous form, the sea trout (Milner *et al*. 2006).

In recent years, the sea trout has become an increasingly popular alternative to salmon in many recreational and commercial fisheries in England and Wales. Indeed, with the recent decline in salmon stocks in the UK, the sea trout could now be considered the main target species of many fisheries (Harris 2002), with associated fisheries having considerable social and economic value (Mawle and O'Reilly 2006). Despite this, there is currently no systematic method for managing and conserving our sea trout stocks. The need to address the specific management requirements of sea trout was highlighted by an unforeseen and widespread decline in stocks on the West Coast of Scotland and in Western Ireland in the late 1980s and early 1990s (Gargan *et al*. 2006; Butler and Walker 2006). These stock declines are thought to have been caused by sea lice infestations from nearby salmon farms (Butler and Walker, 2006; Poole *et al.* 2006; McKibben *et al.* 2006). Reported catches of sea trout in England and Wales have been relatively poor for a number of rivers in the last few years.

In the early nineties, the National Rivers Authority began a national programme to improve the management of sea trout stocks. The programme involved a number of separate research initiatives, including analysis of historical catch records (Elliott 1992), literature reviews (Elliott *et al.* 1992), a study on the feasibility of establishing a gene bank for sea trout (Cross and Rogan 1992) and a review of information on sea trout stocks in England and Wales (Solomon 1995). Harris (2002) put together baseline data on sea trout stock characteristics in England and Wales, and Locke (2003) investigated the status and trends in sea trout rod and net catches since 1974.

Current drivers to improve the management of brown trout stocks frequently make reference to the use of biological reference points (MAFF and National Assembly for Wales 2000; Environment Agency 2003). The Fisheries Legislative Review Group (2000), for instance, recommended establishing sea trout conservation limits to aid the assessment of sea trout stocks, without prescribing how this might be done. The Environment Agency's Trout and Grayling Fisheries Strategy (2003) stated (Policy 8) that "*we will work to develop conservation targets for the abundance of wild trout and grayling stocks against which the status of these stocks can be assessed*". The Water Framework Directive (WFD) requires that standards be set on the ecological health of waters. The sea trout represents a unique 'sentinel' species in this respect, as the successful completion of its lifecycle requires good conditions in both freshwater and marine environments (Milner *et al.* 2006a). The use of trout BRPs may thus be an important complement to WFD monitoring. In the case of salmon, conservation limits have been incorporated into Habitats Directive-associated assessments in rivers designated as special areas of conservation.

1.2 What are biological reference points?

Biological targets, or reference points, have been used in the management of marine fisheries for a number of years, and a large number of different types of reference points have been developed for use in the management of different fish stocks (Crozier *et al.* 2003; Potter *et al.* 2003).

A biological reference point represents a way of assessing performance relative to some expected level based on system dynamics (Prevost and Chaput, 2001). The use of reference points for managing stock is based on the assumption that recruitment is dependent on spawning stock size. For salmonids, this premise is supported by a large number of studies on population dynamics (Prevost and Chaput, 2001).

Several types of biological reference points can be identified, representing potential control points at stages of the lifecycle and various fisheries, each having different management and assessment methods (Environment Agency 2003a). A biological reference point may be defined as a measure of stock size or fishing mortality designed to correspond with management or monitoring objectives.

Biological reference points take two forms: limits and targets. A 'limit' reference point is defined as a 'demarcation of undesirable stock levels or levels of fishing activity'. The ultimate management aim of a 'limit' reference point will be to ensure that there is a high probability that this boundary is not crossed. Using 'limit' reference points in management will therefore involve defining both the reference point and the required probability of avoiding undesirable stock levels (Potter *et al.* 2003). A 'target' reference point, on the other hand, typically represents a point to aim for to achieve a particular management objective. Target reference points may be used to take account of uncertainties, as is the case in salmon management, where 'management targets' (target reference points) are used to ensure that the spawning escapement of a stock exceeds the 'conservation limit' (limit reference point) with the required frequency.

In practice, 'limit' and 'target' reference points should not be regarded as alternatives; rather, fisheries may require more than one reference point in their overall management strategy (Potter *et al.* 2003).

1.3 Policy aims and management objectives

In order to set biological reference points and assess performance, management objectives must first be clearly defined. Four main objectives have been proposed for trout; these are to:

- optimise freshwater production of brown trout;
- optimise recruitment to homewater fisheries, that is, rod and net fisheries (in stocks with a major migratory component);
- maintain and improve the diversity and fitness of stocks;
- consider sustainable catch potential (for both rod and net catches).

One aspect that is not listed here is socioeconomics (although it is considered indirectly in the final objective on sustainable catch potential). Whilst this is a key consideration in the management of trout stocks, it does not fall within the scope of this project.

Each of these management objectives has links to policy statements within the National Trout and Grayling Strategy. The strategy outlines a number of management objectives for trout, but only some of these require quantitative assessment measures in the form of reference points. Table 1.1 outlines the main links with policy as well as the measures available for assessing performance against each of these objectives, and the existing monitoring programmes that may provide data for assessment.

Of the different management objectives outlined, each satisfies different policy aims, and each requires different measures of assessment. During the course of this project, it will therefore be necessary to consider different approaches to setting biological reference points and to establish which approach, or combination of approaches, best addresses all of the management objectives outlined. Whichever approach is adopted will need to be 'measurable', given the data that is currently available.

Table 1.1 Policy aims and management objectives

Objective	Policy links[1]	Measures	Current monitoring
Optimise freshwater production of trout	Policies 8, 22, 23, 24	- Abundance and structure - Growth - Biomass - Production	- Juvenile densities - HabScore - Age/size structure of juvenile populations - Adult catches (sea trout only) - Weight distribution of catches - Size/age composition of adult populations - Distribution of migratory/non-migratory trout - Sea age structure (index rivers only) - Juvenile age structure Adult catches
Optimise recruitment to homewater fisheries (rod/net)	Policies 9b, 10, 12, 13, 14, 23	- Spawning stock - Stock recruitment curves - Size of adult run	
Maintain and improve diversity and fitness of stocks	Policies 8, 16, 17, 27, 29	- Anadromy vs. non-anadromy. - Age structure in juveniles & adults - Biomass - Growth	
Sustainable catch potential (rod/net)	Policies 9b, 10, 12	- Catch (sea trout only)	

[1]Relating to policies in the Environment Agency's National Trout and Grayling Strategy

1.4 Difficulties in setting BRPs for trout

Before considering the possible approaches to setting biological reference points for trout, it is important to outline some of the potential difficulties involved.

1.4.1 Variation in stock characteristics

One of the biggest difficulties in setting biological reference points (BRPs) for brown trout is to determine an appropriate reference point that takes account of the large variation in stock characteristics around the country. It is important to consider genetic diversity within and between brown trout populations, recognising that many brown trout traits such as survival, growth rate, propensity for migration, disease resistance and temperature and pH tolerance have a genetic component as well as an environmental one. Although little is currently known of the links between genetic diversity and expression of phenotypic differences, the conservation of genetic diversity is nevertheless essential for populations and species to respond to environmental pressures.

A number of authors have described variations in the characteristics of sea trout stocks seen around the UK and this topic has assumed greater importance in recent years because of the new emphasis on managing the fisheries to maintain species biodiversity and the genetic integrity of individual stocks (Harris 2002).

Day (1887) and Regan (1911) examined differences in morphology, anatomy and behaviour of sea trout and suggested the occurrence of two separate ‘races’. Nall (1930) carried out extensive sea trout investigations and suggested that sea trout might be split into ‘east coast’ and ‘west coast’ types based on their different stock characteristics. Finally, Fahy (1978) proposed four distinct groupings of sea trout in Ireland based on patterns of growth in the sea and differences in adult survival rates, although Solomon (1995) suggested that much of this observed variability may actually be explained by differences in marine and freshwater environments rather than by genetic differences in the stocks examined (Harris, 2002).

More recently, Solomon (1995) described the characteristics of more than 80 rivers, including all rivers in England and Wales with significant runs and catches of sea trout, and Harris (2002) provided a detailed description of the structure and composition of adult stocks of sea trout from 16 study rivers in four regions of England and Wales.

Harris (2002) used this information to determine general stock ‘groupings’ based on five features believed to be significant in distinguishing one stock from another. These were: age at smolt migration, age at first return to the river as maiden fish, frequency of spawning, growth rates in the sea and the pattern and timing of runs of adult fish into the river. Four distinct stock groupings were determined, showing an apparent relationship between life-history characteristics and geographical location. The first group identified was represented by shorter lived, faster growing sea trout found in the majority of rivers studied in North East England. The second group was made up of shorter lived, slower growing sea trout stocks found in rivers of South West England. The third group was represented by the longer lived, faster growing sea trout of the majority of the Welsh rivers studied. And finally the fourth group, covering rivers in North West England, was made up of longer lived, slower growing sea trout stocks.

It is not clear to what extent these observed differences in sea trout stocks around the country are the result of genetic or environmental differences. It is possible that some of the differences in population structure may reflect differences in genetic composition, and that the groupings observed represent a number of post-glacial ‘races’ of trout. However, it is equally likely that most, if not all, of the features used to distinguish between stocks could be explained by environmental factors acting at some point in the lifecycle, affecting both the freshwater and marine stages (Harris 2002).

Management of both freshwater resident and migratory sea trout stocks will need to take account of the differences between stocks and ensure that this diversity is conserved. This will be essential in maintaining a robust stock able to withstand and recover from adverse factors in both freshwater and marine environments (Harris 2002).

1.4.2 Factors affecting anadromy and implications for BRPs

In order to set biological reference points for brown trout, it will be necessary to know the relative contributions of freshwater resident and sea trout to total trout production, irrespective of whether reference points are based on juveniles, adults or some combination of assessments (Milner *et al.* 2006a). We need to better understand the extent to which spawning sites overlap and where sea trout exist within a particular catchment, in order to assess the contribution of sea trout females to egg deposition and juvenile production (Charles *et al.* 2004). Understanding the causes of anadromy is a vital step in managing stocks and maintaining sea trout fisheries.

A great deal of research in recent years has focused on trying to better define the 'sea trout'. Although we now have a reasonable understanding of the ecology of the brown trout, the factors that influence anadromy are still relatively poorly understood.

Gross (1987) developed a migration model for understanding the evolution of migratory life strategies in fish and suggested that for a life-history pattern to exist, the gain in fitness from moving to a new habitat minus the cost of moving must be higher than remaining in one habitat. Migration to sea has a number of advantages, namely better feeding opportunities, which in turn can lead to increased fecundity and larger energy stores. However, there are a number of costs associated with this choice, such as increased chances of predation, parasites and energy costs. In river systems, the balance of costs and benefits often results in the co-existence of different life-history strategies within the same river (Ferguson 2006).

A number of studies suggest that different life-history strategies will be expressed along a gradient of migratory 'cost' within individual river systems. For instance, Jonsson and Jonsson (2006) examined migratory costs in brown trout in relation to the distance travelled from the sea to the spawning site. They hypothesised that variation in adult size and other life-history responses occur as a consequence of migratory cost in long-distant relative to short-distant brown trout migrants. Their results indicated that the proportion of stocks migrating to sea would be expected to decrease with increasing distance between the river mouth and the spawning site, although this relationship may be confounded by other influences such as inter-specific competition.

Bohlin *et al.* (2001) examined elevation as an index of migratory difficulty. At low elevations, densities of juvenile trout, taken as a measure of population productivity, were found to be higher for sea trout populations than for FR populations. This suggests that migration is beneficial for juvenile production. Conversely, juvenile densities were found to decrease with increasing elevation in sea trout populations, though the same was not true of freshwater resident populations, suggesting that increasing migratory costs reduce the fitness benefits of anadromy. Juvenile densities for both sea trout and FR trout populations in this study were found to be similar at an altitude of approximately 150 metres. This suggests a particular point at which the costs of migration are offset by the benefits.

The decision to migrate most likely involves a trade-off, balancing the growth, reproductive and mortality potential of different habitats. It remains unclear, however, as to what causes individual fish within a population to adopt one particular life strategy over another (Morinville and Rasmussen 2003).

There is considerable evidence that migratory behaviour is under genetic control and has high heritability (Ferguson 2006), and strong genetic control for residency seems to be particularly well developed for populations living above waterfalls (Northcote 1992). In streams where upstream migration is not restricted, however, genetic control appears to be less pronounced so that sea trout and FR trout occur within the same stock. Environmental conditions, particularly those affecting growth rate, can then affect the degree of residency expressed (Northcote 1992, and various authors therein).

Gross (1987) suggested that the relative abundance of food in freshwater versus marine habitats is likely to be the most important biological parameter influencing choice of life history. It is recognised that food and temperature strongly influence fish growth (Elliott 1976, 1982), and several authors have considered the link between food availability and the tendency for trout parr to smolt or remain resident in the river. For instance, a sea trout culture experiment carried out by Morgan and Pavely (1993) tested the theory that increasing the quantity of food available to progeny of sea trout not only promotes growth, but also causes fish to become sexually mature and affects the ability to osmoregulate in sea water - effectively producing brown trout from a sea trout stock. Olsson *et al.* (2006) tested the effect of food availability on the development of migratory and non-migratory body morphologies. The majority of brown trout were seen to migrate when food levels were low, but fewer did so when food was plentiful. They concluded that the decision to migrate is a plastic response, influenced by growth opportunities.

Bohlin *et al.* (1996) proposed that a critical threshold in body size must be reached for migration to be initiated and various authors (Jonsson 1985; Okland *et al.* 1992; Forseth *et al.* 1999) have observed that fast growers often migrate at younger ages than do slow growers. It is thought that food supplies in the natal habitat are likely to limit fast growers sooner than slow growers, meaning that they need to switch to richer feeding habitats earlier to ensure continued growth.

Forseth *et al.* (1999) examined growth in conjunction with energy intake and found that although sea trout consumed significantly more than FR trout, a larger proportion of the consumed energy was allocated towards metabolic costs, leaving less energy for growth compared to freshwater residents. The sea trout were therefore believed to be moving to richer feeding grounds as a result of their low growth efficiency, resulting from increased metabolic costs, but not necessarily low growth. Cucherousset *et al*. (2005) suggested that early life-history traits influence fitness by affecting survival probability later in life. Their study found that individuals with a low energetic rate tended to remain in the environment they were born in, whilst trout whose metabolic needs were higher tended to leave the brook to migrate downstream. If these fish were then unable to maintain their growth rate during their second year, they would move further downstream again, or in some cases would run to sea. The sources of variation in metabolic rate are not fully understood, but maternal and developmental effects are believed to play a part (Cucherousset *et al*. 2005).

Thorpe (1990) put forward the theory that ‘the fish’s primary objective is to reproduce as early as possible, and only secondarily to grow’. Since the conditions which favour rapid growth also favour early maturity (Alm 1959, Thorpe 1986 cited in Thorpe 1990), trout are therefore less likely to adopt a migratory life strategy if growing conditions in freshwater during the critical period in early spring are good. At the opposite extreme, in habitats with poor growth conditions, trout are more likely to smolt and emigrate. Intermediate conditions allow different proportions of both life strategies to be expressed, and whilst trout populations might be expected to produce both freshwater residents and sea trout in all years, the proportion would vary with the severity of winter/spring conditions (Thorpe 1990).

Olsson *et al.* (2006) showed that both migratory and non-migratory behaviour can be environmentally induced by reciprocally transplanting brown trout between two sections in a river. Migratory behaviour was seen to develop in a river section with high brown trout densities and low growth rates, whilst non-migratory behaviour developed in a section with low brown trout densities and high growth rates.

Additional environmental features believed to influence anadromy were studied by Northcote (1992). For instance, the type of stream bottom can be important in territory establishment, so higher densities of young are likely to occur in stream sections with coarse substrate and this may, in turn, promote a higher degree of residency. Residency may be less common in areas where spawning and rearing habitats are sparse and widely spaced, requiring more extensive movements of fish.

Ferguson (2006) notes that life-history pattern may shift with relatively small environmental changes. This has important implications in terms of potential changes to productivity following habitat alterations, or other management actions. It is also important to consider the possible life-history implications of climate change. Consideration therefore needs to be given to how such shifts in migratory habit can be accounted for in setting reference points, and this highlights the need to manage trout stocks as a whole rather than as separate anadromous and FR components.

Although sympatric populations of sea trout and FR trout are common, they are often unbalanced through sex-selective migration of females (Milner *et al.* 2006a); this is likely to reflect the fact that the benefits of migration are larger for female trout.

Female fertility increases exponentially with body size (Jonsson 1985, cited in Jonsson and Gravem 1985). Selection therefore tends to favour faster growing females and females thus have a stronger tendency to migrate to more productive environments (Northcote 1992; Cucherousset *et al.* 2005; Ferguson 2006). As a result, migrant female sea trout are likely to be the dominant source of total trout egg deposition in most rivers with a migratory trout component (Milner *et al.* 2006).

Trout males, on the other hand, may be reproductively fit both as small and large fish, and are therefore able to achieve reproductive success in freshwater after just a few years of growth. Large males are reproductively fit because they are able to drive smaller males away from females at the redd. However, small males are able to fertilise some of the eggs by darting in during spawning (Klemetsen *et al.* 2003).

Studies carried out by Jonsson and Jonsson (2006) found that gonadal mass in males decreased with the distance travelled to spawning grounds. This is likely to be the result of the lower energy reserves available for gonadal production in long-distance migrants. These sea trout males will have to compete with freshwater resident males on the spawning grounds, so this decrease in gonadal mass may be one factor favouring residents when migratory costs are large (Bohlin *et al.* 2001). No similar decrease was found in female gonadal allocation, suggesting that females are able to better conserve their gonadal allocation than males. This is likely to be because reproductive success in females is determined by the number and size of the eggs spawned, rather than through competition with other females.

In summary, the factors that trigger anadromy in trout are becoming better understood. However, in order to set reference points for trout as a whole, it is necessary to understand the distribution of spawning by sea trout and FR trout in river systems. A clearer picture is needed of the habitat and other physical features that may influence both the tendency to migrate and production parameters such as smolt age, or age at maturity. Such features might include catchment-scale connectivity of habitats, spawning distribution and population dispersal (Thorpe 1990), as well as physical characteristics such as water flow, temperature or latitude of streams (L'Abée-Lund *et al.* 1989; Jonsson *et al.* 2001). This information should ideally be presented as a national mapping system for trout which could form the basis for models that might predict sea trout and FR trout abundance.

A first important step in this process is improving our ability to distinguish between freshwater resident and sea trout. Although adult FR trout and sea trout can generally be distinguished from one another using colour, size and body form (Elliott 1994), the same is not true of eggs or juveniles (McCarthy and Waldron 2000). A number of techniques have been used to distinguish freshwater resident adults from sea trout adults in salmonid populations – namely carotenoid pigment profiling (Youngson *et al.* 1997), strontium contents of scales (Eek and Bohlin 1997) and microchemistry of otoliths (Kahlish 1990; Howland *et al.* 2001). It is likely that some of these techniques would be applicable to eggs or juveniles, but an easier and quicker technique would be a useful tool in helping to determine the contribution of anadromous females to juvenile production. One option that has been proposed as a rapid alternative is analysis of stable isotope ratios between tissues (McCarthy and Waldron 2000; Charles *et al.* 2004). This technique has been used by a number of authors to examine the reproductive contributions of anadromous and FR trout (Doucett *et al.* 1999; McCarthy and Waldron 2000; Charles *et al.* 2004; Ashton 2006, personal communication). Research into microchemistry and isotopic methods is also currently in progress (Alice Ramsey, personal communication)

1.4.3 Salmon/trout interactions

One of the practical problems discovered in the use of BRPs for salmon was the interactions between salmon and trout, and this is likely to apply equally to trout BRPs.

Salmon and sea trout have similar lifecycles: adults often spawn in the same areas in rivers and juveniles generally share the same freshwater habitat (Milner *et al.* 2006). If, as is likely, densities of one species influence the other, then this presents a number of problems in setting biological reference points for individual species.

A study by Milner *et al.* (2006) reviewed the potential impacts of interspecific competition between salmon and trout on biological reference points in rivers across the North East Atlantic Commission (NEAC) area. The study concluded that there is little evidence for effects of interspecific competitive interaction at fishery or catchment scale, even though these effects are quite clear at micro or macro-habitat scale. Nevertheless, the possibility of catchment-scale effects could not be eliminated, especially in smaller streams and rivers, and there is a need for further research to gain a better understanding of the processes involved. One area that requires careful consideration is the possible use of multi-species targets for salmon and trout.

1.4.4 Data availability

One of the key considerations in deciding how best to define biological reference points for trout is the availability of data, and how constraints on future data collection might influence the choice of approach adopted.

The availability of data is considered in detail in sections of this report, for each of the possible approaches outlined, but some generic points are worth considering here.

It is important to reflect on the level of information available for sea trout and FR trout stocks at different stages of the lifecycle, and to consider the sort of data available at different spatial scales (site level, catchment level and so on). For instance, the extent and quality of information on adult stocks varies considerably from catchment to catchment. Rod catch data have been collected consistently for a number of years and are available for the majority of the larger rivers in England and Wales. However, these only provide information on sea trout, since there is currently no national reporting scheme for adult FR trout catches. This creates a large gap in our knowledge that will not be solved without considerable investment and collaboration with a large number of anglers.

A few rivers in England and Wales have independent measures of sea trout run size (such as from traps and counters) whilst, at the other extreme, some smaller coastal streams have no measure of adult stock size whatsoever. Smaller sea trout streams collectively represent an important part of the total sea trout resource in terms of biodiversity and conservation value (Milner *et al.* 2006), and the complete lack of data on these streams therefore represents a considerable problem.

Finally, it is important to consider what might influence the Environment Agency's routine monitoring programmes in the future. For instance, implementation of the Water Framework Directive in England and Wales has led to a revision of current monitoring programmes, with the emphasis on a more risk-based approach to monitoring. It will therefore be important to define the data requirements of the various approaches to determining biological reference points for trout in the future.

1.4.5 Public understanding and perception of reference points

A further consideration in the process of determining BRPs for brown trout is the current public understanding and perception of reference points, and any 'lessons learnt' from the implementation of salmon conservation limits.

It is essential to ensure that the uncertainties involved in setting reference points and measuring compliance are adequately described.

Whilst they are often treated as exact, biological reference points are merely estimates with all the associated uncertainties. They are a useful management tool, but should also form part of a wider range of assessment tools to be used by fishery managers.

1.5 Possible approaches to setting BRPs

There are a number of possible approaches to setting biological reference points for trout and sea trout, using data based on different life stages. These options include assessing performance based on juvenile, smolt or adult densities, coupled with measures of habitat quality where available. The various options are summarised in Table 1.2 and are considered in more detail in the following section.

Table 1.2 Different approaches to setting biological reference points for trout

Reference point option	Data requirements	Feasibility
Number of juveniles	Electric fishing data	Possible
Number of juveniles/Habitat Quality Score	Electric fishing data + HabScore data	Possible
Number of juveniles/River Fisheries Habitat Inventory (RFHI) index	Electric fishing data + RFHI	Possible
Number of smolts/expected smolt output	Insufficient data	

Reference point option	Data requirements	Feasibility
Number of adults (FR trout)	Rod catch, CPUE	No data
Number of adults (sea trout)	Rod catch, CPUE	Possible
Number of adults (FR trout)/ habitat	Insufficient data	
Number of adults (sea trout)/ habitat	Rod catch, catchment scale habitat quality score	Possible but further work required

1.5.1 Habitat-based juvenile BRPs in which expected freshwater abundance is compared with observed

One possible approach to setting BRPs may be to assess the state of the juvenile trout population of a river. This approach offers benefits over other methods in that it targets the life stage which includes both freshwater resident and sea trout stock components.

In salmon management, abundance of salmon fry and parr have been used to estimate either spawner numbers or smolt output, allowing information on stock-recruitment relationships to be derived (Kennedy and Crozier 1993; Bagliniere *et al.* 1993 cited in Walker *et al.* 2006). However, although a healthy juvenile trout population should provide the resource necessary for a healthy sea trout population, it is not currently possible to predict the effects of reducing or increasing juvenile abundance on smolt output or adult sea trout numbers due to the current inability to distinguish between the two forms at the juvenile stage (Walker *et al.* 2006). Juvenile targets do not provide the information required to confidently set exploitation controls for the adult fisheries.

An improved approach would involve considering juvenile abundance coupled with measures of habitat quality and predictions of carrying capacity, to provide an index of freshwater quality and of the extent to which carrying capacity is realised (Environment Agency 2003a). Poor juvenile abundance can reflect the impact of environmental factors or low recruitment. The habitat model HabScore (Wyatt *et al.* 1995) provides a ready-made mechanism for doing this, though its application at catchment level needs to be explored further, and the existing models require further refinement. However, HabScore predictions are based on the assumptions that neither water quality nor recruitment are limiting the populations. The recently developed River Fisheries Habitat Inventory provides a mechanism for classifying river habitat quality for salmonids on a catchment scale.

Habitat models will, in the first instance, have to be based on total trout populations, as it is currently not possible to distinguish between the two forms on morphological grounds (Milner *et al.* 2006a). Such an approach could be expanded on in the future with the development of tools to recognise sea trout and FR trout, and as further information becomes available on the habitat features that may influence anadromy.

The use of juvenile targets successfully addresses the first management objective: to optimise freshwater production of trout. The second objective to optimise recruitment to homewater fisheries is partially addressed in that, given low levels of exploitation, a healthy juvenile population should support a healthy adult population. However, as outlined above, it is currently impossible to predict the effects of reducing or increasing juvenile abundance on smolt output or adult sea trout numbers, so the effects on adult stock could not currently be measured. For this same reason, the fourth management objective (to maximise sustainable catch potential) is not successfully addressed by this approach. The third objective, to maintain and improve diversity and fitness of stocks, is difficult to address using juvenile targets, as it is impossible to distinguish between the freshwater and migratory components of the stock at this stage. However, smolt age composition may be taken as one potential measure of diversity.

1.5.2 Salmon-type egg deposition targets based on stock and recruitment

Biological reference points based on stock-recruitment relationships have been used successfully in salmon management in England and Wales since 1996, and currently form the basis of salmon management for all principal salmon rivers. Such reference points have been used successfully in specific management applications, such as in reviewing net limitation orders on certain rivers.

The BRP adopted for salmon is defined as the level of egg deposition representing the stock size at which yield should be maximised. The procedures used to calculate this reference point are covered in more detail in Section 5.1.

Because stock-recruitment relationships are not available for most rivers, the procedures used to set conservation limits for salmon in England and Wales are based on an estimated stock-recruitment relationship for the River Bush in Northern Ireland. Other countries such as Ireland have adopted alternative methods (Crozier *et al.* 2003).

Assuming that the population dynamics (initial or density independent mortality rate and smolting rate) of each stock are similar, the shape of the stock-recruitment curve is determined by the productivity of the freshwater habitat (which determines the height of the SR curve) and the 'replacement line' is defined by the marine survival rate. Thus, adjusting for these features allows the Bush model to be transported to other rivers (Environment Agency 2003a).

In the case of sea trout, published stock-recruitment relationships are available for three river systems: Black Brows Beck in Cumbria, the Bresle in Upper Normandy, and the Burrishoole system in western Ireland. One option may therefore be to review these existing relationships and determine if these could be 'transported' to other rivers as was done in the case of salmon. The River Dee in North Wales also has an extensive dataset for adult sea trout, and it may therefore be possible to derive a further (adult to adult) stock recruitment relationship for this river system.

However, Walker *et al.* (2006) outline a number of difficulties in doing this. The main issue is that the three river systems for which data are available are very different in their physical characteristics, productivity of their marine systems and life histories of the sea trout they produce. The population dynamics of sea trout stocks in England in Wales are generally far more variable than those of salmon and a wide variety of life-history strategies have been described (Solomon 1995, Harris 2002). The assumption used in the salmon model, that the population dynamics of stocks are similar, is therefore most probably not applicable to sea trout.

Another key problem in the application of salmon-type egg-deposition BRPs to sea trout is the fact that although some sea trout populations exhibit complete anadromy (Elliott 1994), others have a mixed life history, and FR trout freely interbreed with anadromous trout. There is a tendency for anadromy to be more prevalent in female fish (LeCren 1985, cited in Walker 2006). In order to transport a sea trout SR relationship between rivers, it may be necessary to understand the relative reproductive contribution of freshwater trout to sea trout runs (Walker *et al.* 2006). This is not currently possible, although a number of possible mechanisms for obtaining this information are currently being explored. Many brown trout populations are dominated by sea trout, so this issue may not be a problem for all rivers.

It is clear that this approach is not presently feasible, given current data limitations. However, with further advances in our understanding of sea trout, reference points based on salmon-type egg-deposition targets have the potential to address a number of our management objectives. Objectives 1, 2 and 4 are all clearly addressed by this approach, and objective 3 could potentially be addressed through ‘grouping’ of stocks and individual management of different groups with similar life-history strategies.

1.5.3 Catch-based targets

In considering which type of biological reference point is most applicable to brown trout, one of the key considerations must be the availability of data required to determine the appropriate reference point and assess the performance of stocks in relation to this.

Rod catch data are now widely available for the majority of sea trout fisheries in England and Wales, though the same is not true of FR trout fisheries. This type of data could therefore potentially form the basis of a catch-based biological reference point for sea trout, using catch as an index of both stock and fishery performance.

Previous studies have demonstrated a relationship between sea trout rod catches and catchment size and river flow (Milner 2006). Such relationships could be refined further by including some measure of channel width or catchment structure, and could eventually provide a basis for setting reference catch levels for individual catchments.

However, there are a number of potential problems with this approach. Firstly, catch-based targets ignore the contribution of FR trout which may be significant in some stocks (Walker *et al.* 2006), although this will no doubt vary from catchment to catchment. Rod and net catches also fluctuate considerably over time, due to factors such as stock size, fishing effort, recording accuracy and environmental, social and economic factors that influence the accessibility, availability and vulnerability of stock to fishermen (Environment Agency 2003a).

Clearly, given the current paucity of reporting of FR trout catches, this approach would only be applicable to sea trout. This option therefore only really addresses part of the second of our objectives – to optimise recruitment to homewater fisheries.

1.5.4 Combination approach

It is evident from the different possible approaches outlined thus far that there is no single solution. There are a number of potential problems with each approach, and none of them taken alone fully address the management objectives outlined above.

One of the primary aims of this project is therefore to provide an assessment and statement of feasibility of each of the possible approaches and recommend a way forward. It is likely that this will involve a number of approaches, combined in some way to provide the best possible assessment of the stock as a whole, and addressing all of the management objectives outlined.

2 Distribution of trout in England and Wales

One of the challenges in setting BRPs for trout is to develop reference points for both freshwater resident and anadromous stock components. In order to achieve this, it is important to determine the relative incidence and distribution of these two morphs in different catchments in England and Wales. This chapter examines available data on trout distribution and seeks to identify the main gaps in our understanding. The scope and list of rivers for which BRPs may be required is also considered.

2.1 Review of existing data on trout distribution

The distribution of brown trout has been the subject of a number of studies (Frost and Brown 1967; Wheeler 1969; Maitland 1972; Cresswell 1989; Maitland and Campbell 1992; Elliott 1994; Solomon 1995; Davies and Harding 2002; Gray and Mee 2002), all of which have provided a general overview of the distribution of trout stocks in England and Wales based on historic data.

A report prepared by Gray and Mee in 2002 established a baseline description and inventory of trout stocks in England and Wales, and calculated trout distribution quantitatively in terms of river length. The distribution of FR and sea trout in this study was determined subjectively, based on data gathered from Environment Agency databases and reports, checked by local expert staff. The study focused on rivers and main tributaries that feature on a 1:250 000 scale Ordnance Survey maps, and also included many of the smaller coastal streams for which little information is available. However, smaller tributaries supporting trout may have been discounted.

This report showed that brown trout were widespread throughout England and Wales, occupying 67 per cent of the total river length. Trout abundance was highest in the north and west, with 80 per cent of total river length in the North East, South West, Wales and North West supporting trout. Trout were least common in Thames and Anglian regions, occupying only 34 and 24 per cent of total respective river lengths.

Over 50 per cent of the rivers in the North East, South, South West, North West and Wales were found to be accessible to sea trout, although their presence was confirmed in less than half of these areas, including the many coastal streams which attract little angling interest. Isolated populations of FR trout (above impassable barriers) were found to occupy 25 per cent of the total river length in England and Wales (Gray and Mee, 2002). Man-made barriers to migration tend to prevail in the lowland catchments to the east (such as Anglian), and are typically flood defence structures. Barriers in the upper catchments of the north and west tend to be natural. Figures 2.1. to 2.5 show reproductions of the maps produced during this study.

The Environment Agency's National Trout and Grayling Strategy produced in 2003 also examined the distribution of brown trout in England and Wales, and defined 'native trout' waters as 'waters that have a significant natural production of trout, whether migratory or non-migratory, or from which there is ready access to other waters with such production'. Maps were produced showing the native trout waters for each area, based on local knowledge. The resulting national map is shown below (Figure 2.6). This differs from the trout inventory distribution maps, presumably due to the different definitions employed, since the definition of 'significant' trout production is subjective. It is worth bearing in mind that the trout inventory does not provide any quantitative assessment of trout numbers, rather a simple overview of distribution. The native trout waters designations, on the other hand, refer to a ‘significant’ production of trout.

Finally, it is possible to gain a good overview of brown trout distribution by simply examining records of trout from electric fishing surveys. The presence/absence of trout at sites surveyed during 2003-2005 is shown in Figure 2.7.

It is evident from the data outlined here that current information on the distribution of brown trout in England and Wales is inconsistent and often based on subjective assessments and classifications. Establishing an objective baseline of trout distribution (both freshwater resident and sea trout) will therefore be essential to devise an appropriate management strategy for trout. The National Trout Inventory provides a good baseline for this process, but further work is needed to determine the relative importance of FR trout and sea trout fisheries in certain rivers.

Recommendation 2.1: Establish an objective baseline of trout distribution (both FR trout and sea trout morphs) in England and Wales.

Catches of FR trout are not consistently reported, and it is therefore difficult to obtain an objective picture of the relative importance of migratory and non-migratory trout in different rivers. Angling pressure can vary considerably from river to river, and catch data may not be adequately representative of the true ratio of migratory and non-migratory forms. Artificial stocking of brown trout is widespread on a large number of rivers, with approximately 700,000 trout stocked into rivers in England and Wales on average every year (B Shields, personal communication). An overview of trout stocking in 2006 is given in Figure 2.8.

The following section outlines other potential sources of information that may further our understanding of the relative incidence and distribution of FR trout and sea trout.

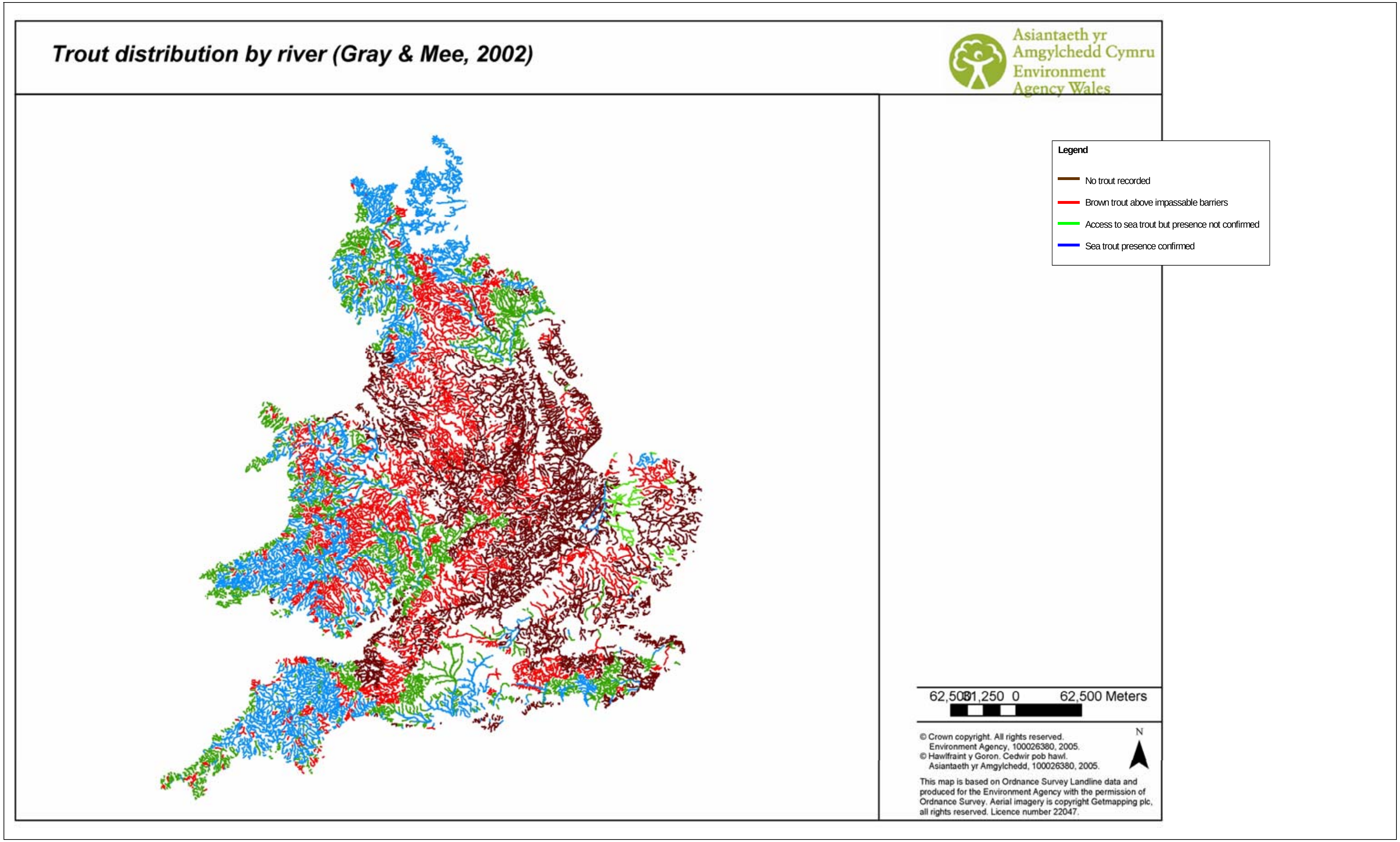

Figure 2.1 Distribution of trout in England and Wales (from Gray and Mee, 2002)

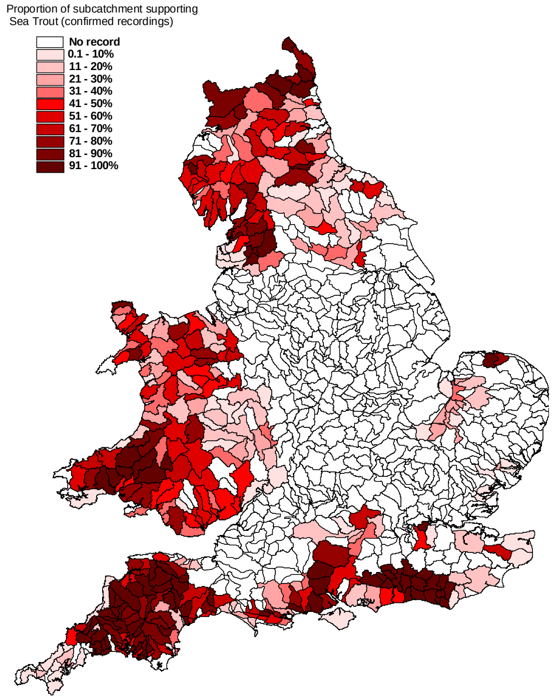

Figure 2.2 Proportional distribution of sea trout (confirmed presence) at sub-catchment level (from Gray and Mee, 2002)

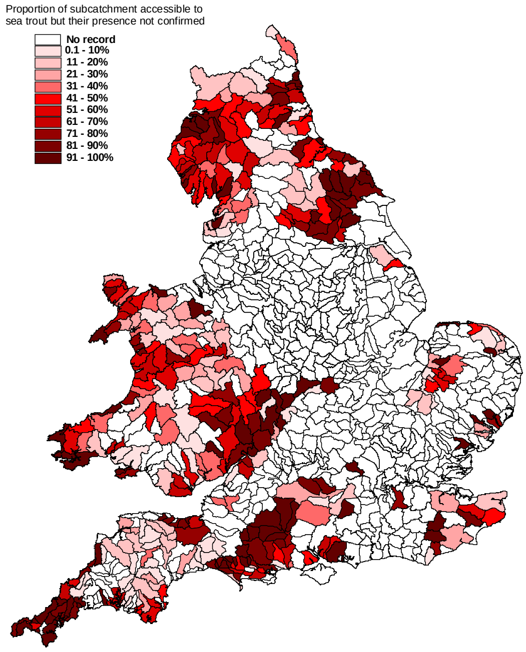

Figure 2.3 Proportional distribution of accessible river to sea trout (presence not confirmed) at sub-catchment level (from Gray and Mee, 2002).

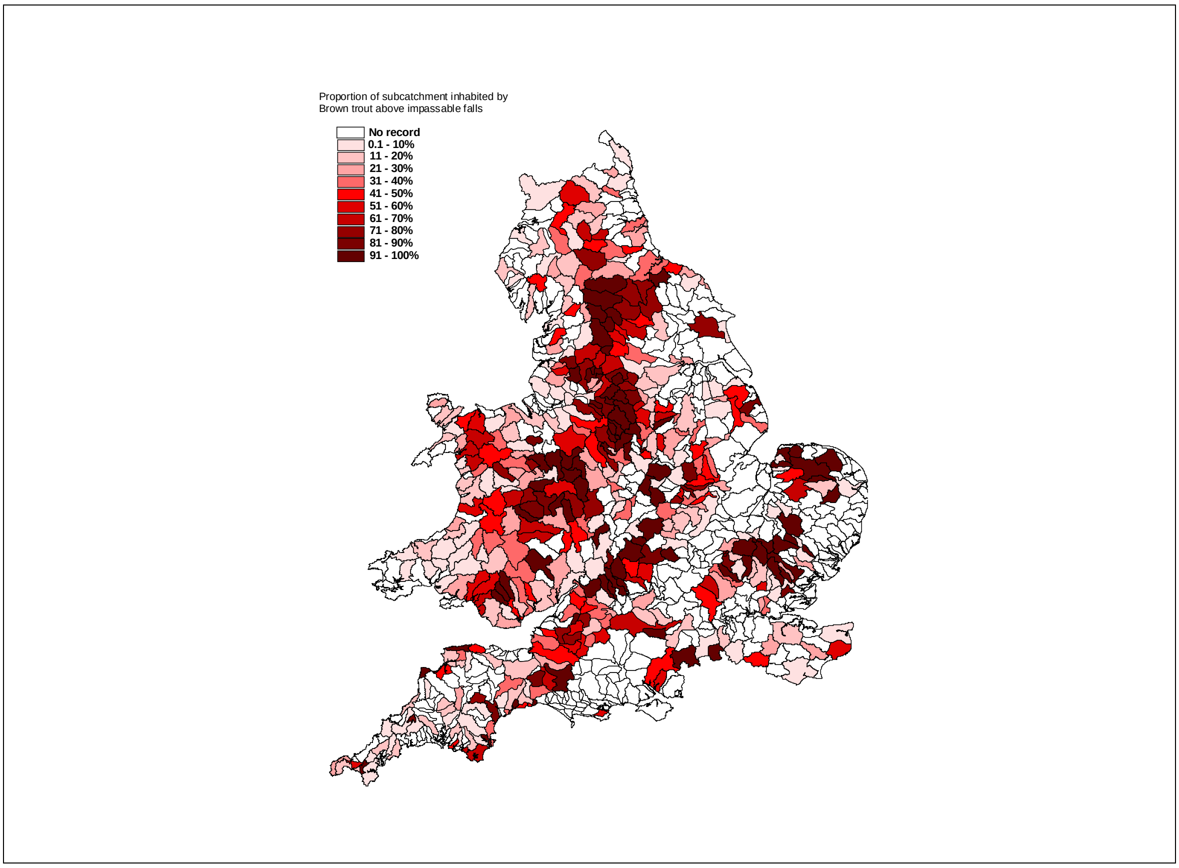

Figure 2.4 Proportional distribution of FR trout above impassable falls at sub-catchment level (from Gray and Mee, 2002)

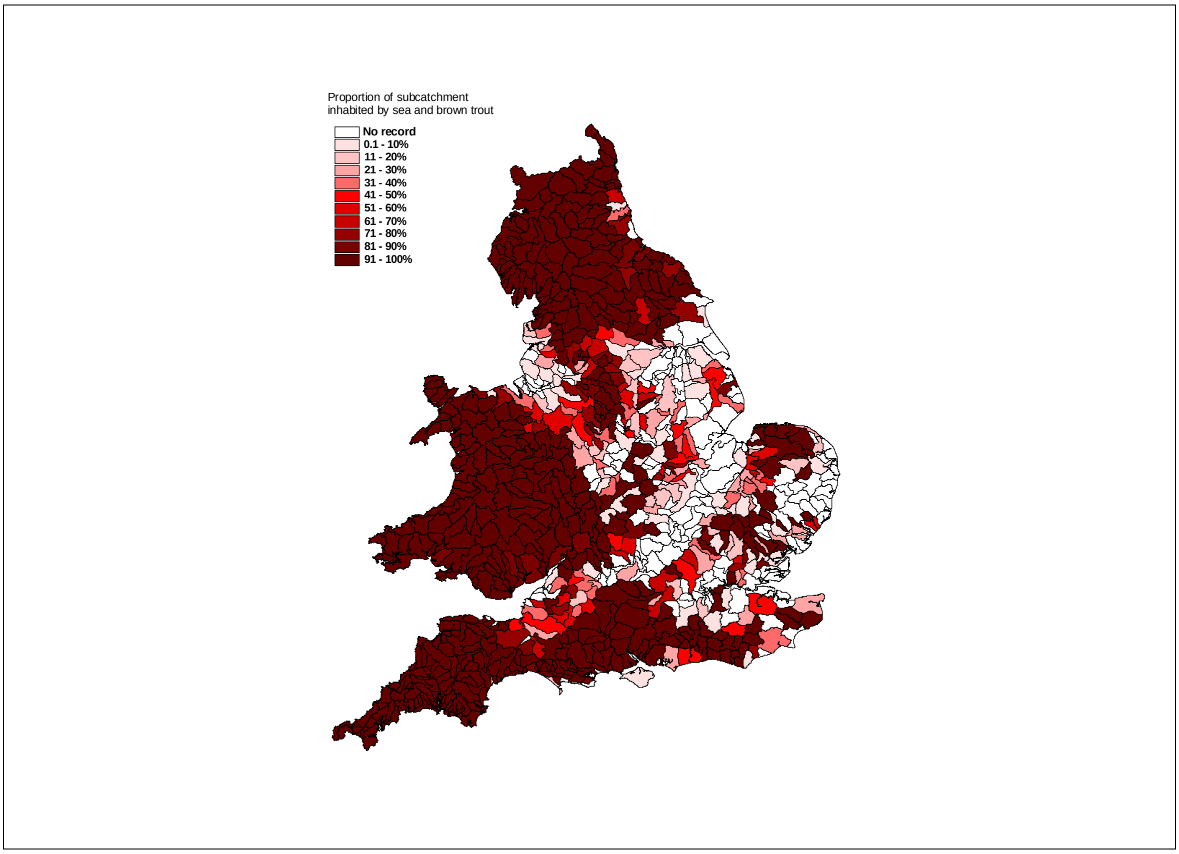

Figure 2.5 Proportional distribution of sea and FR trout at sub-catchment level (from Gray and Mee, 2002)

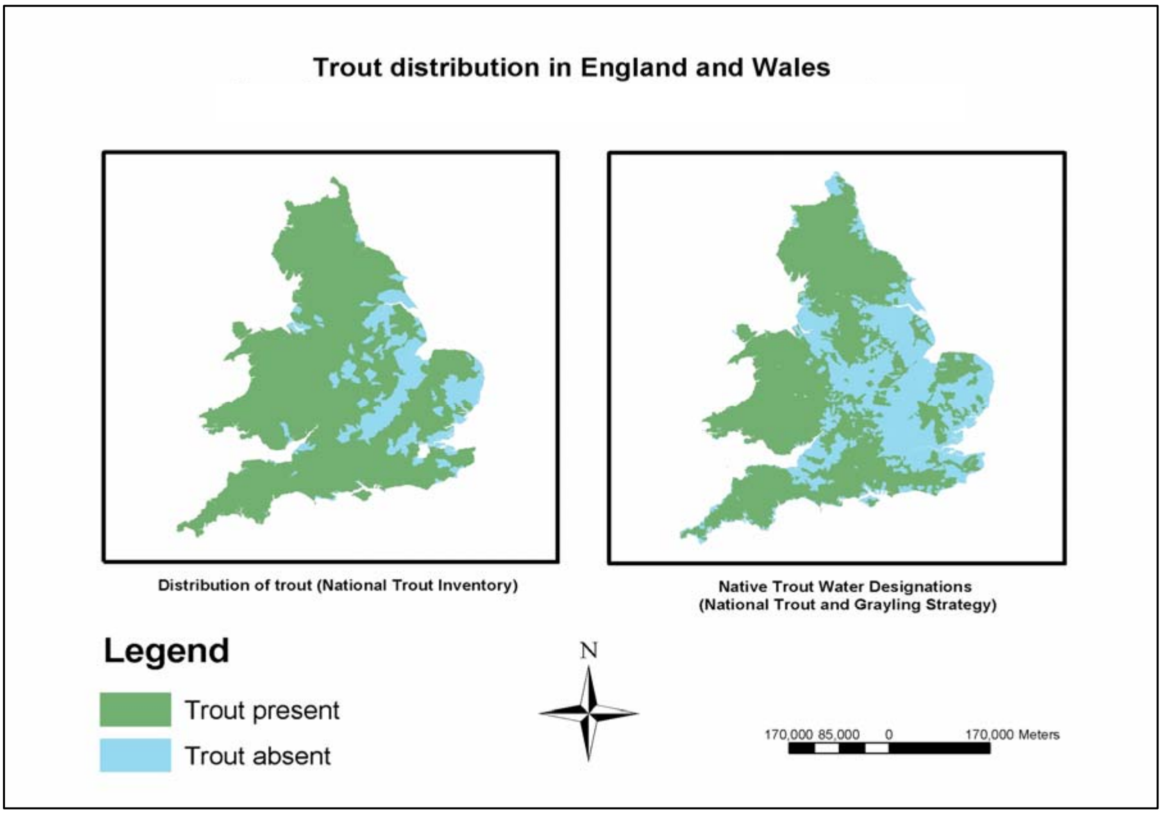

Figure 2.6 Trout distribution in England and Wales

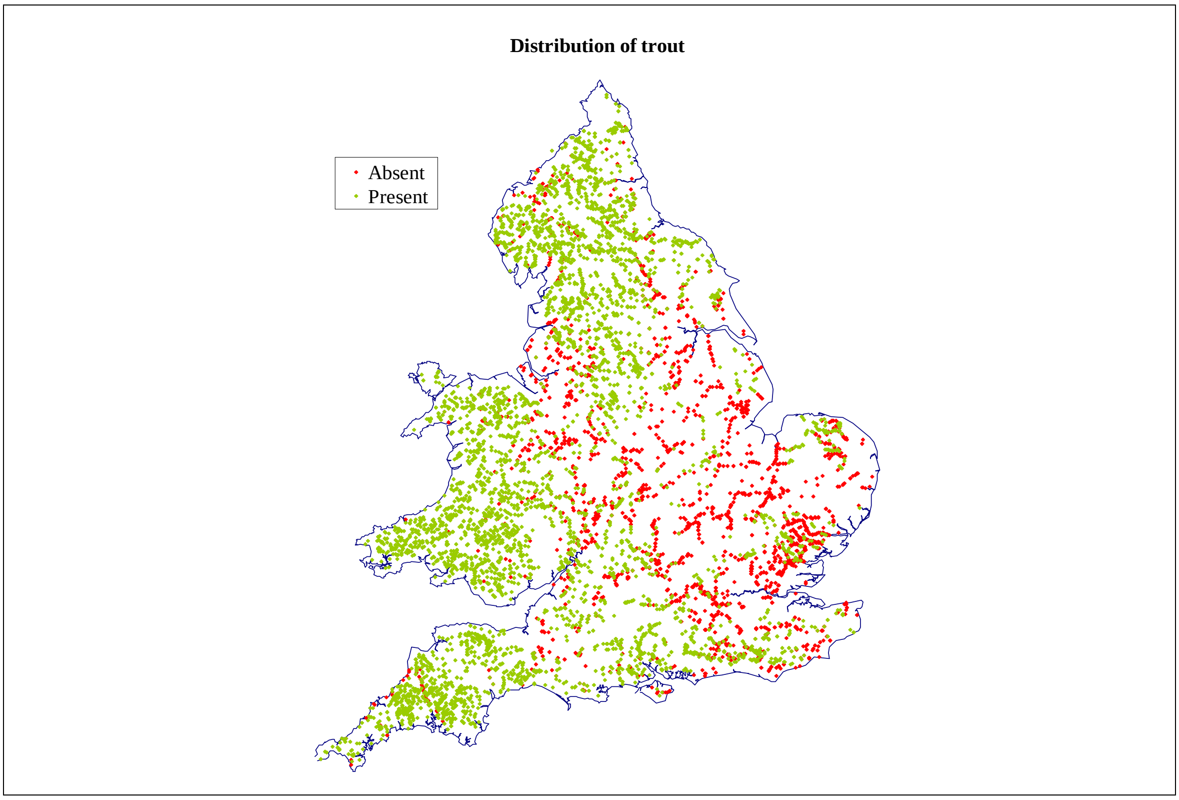

Figure 2.7 Distribution of trout in 2003-2005 based on juvenile survey data

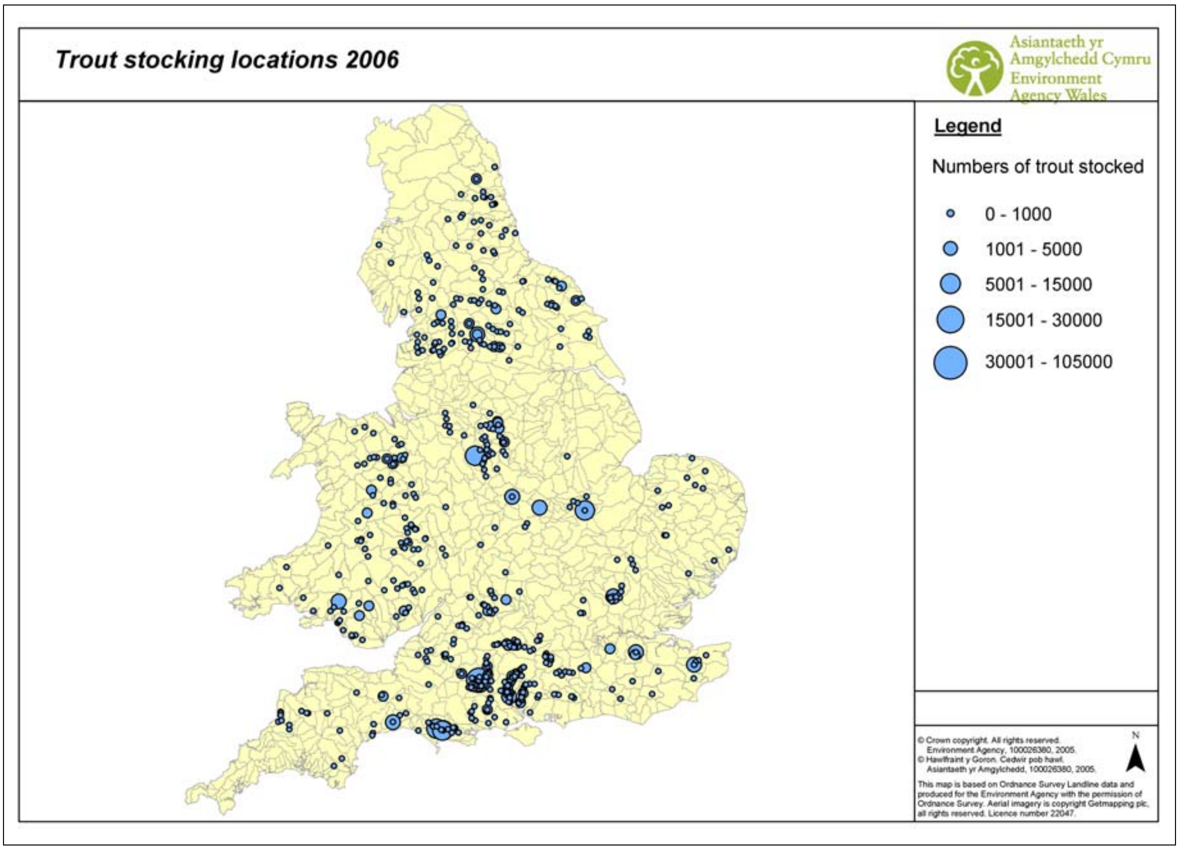

Figure 2.8 Brown trout stocking in England and Wales during 2006

2.2 Further sources of information on trout distribution

The type of catchment-level assessment described in Section 2.1 does not adequately describe the distribution of migratory and non-migratory stock components, and a much finer scale of assessment is needed to provide a true picture of relative incidence of anadromy within individual catchments. Unfortunately, data to support such a fine-scale assessment are scarce.

Potential sources of information might include the location of obstructions to migration, age structure signals or the use of isotopic ratios to determine migratory parentage.

Information on the location of obstructions to migration is available for the majority of catchments in England and Wales and was reviewed recently (B Wilson, personal communication). The National Trout Inventory also provides a good overview of where FR trout populations occur above impassable barriers (Figure 2.4).

Age structure signals from juvenile trout data may provide a clue to the relative incidence of anadromy for a particular site or reach. The general thinking behind this approach is that the proportion of older age classes of trout is likely to be lower at sites where the incidence of anadromy is higher, since the majority of juvenile trout in anadromous populations will tend to migrate to sea by age 2+. In practice, it may be difficult to use this approach for certain catchments in England and Wales due to the way juvenile data is collected nationally. Whilst some areas do record individual fish ages and lengths, others simply record data for 0+ or >0+ fish, making it impossible to extract information on age structure without going back to the raw data. Detailed information on individual fish lengths and ages is currently only believed to be available for about a third of all catchments surveyed under the national monitoring programme, with the most detailed data typically available for rivers in the north east, south and south west of the country. Age frequency data are essential to the understanding of the distribution of anadromy within individual catchments and would be necessary for the effective application of juvenile reference points. The benefits of routinely collecting this type of information properly cannot be overstated.

Recommendation 2.2: Improve quality of age data collected during routine juvenile fisheries surveys.

Various authors have shown that variations in early growth or growth efficiency in trout may be associated with migratory or resident life histories, and information on growth in freshwater will therefore be a key factor in understanding the distribution of anadromy within individual catchments. Understanding of the relationship between water temperature and growth performance is well established for brown trout in freshwater, and this relationship is of particular interest in considering the possible life-history consequences of climate change. It is clear, however, that routine collection of such data would require considerable modification of current monitoring programmes.

Finally, data on isotopic ratios can help to identify sea trout spawning areas and the relative contribution of freshwater-resident trout and anadromous sea trout to juvenile production. For instance, the Centre for Ecology and Hydrology is currently sponsoring research at Exeter University to quantify the costs and benefits of different life-history strategies in trout. The study will aim to determine the parentage of individual trout in Tadnoll Brook, a small tributary of the River Frome, and examine the influence of parentage (and life-history strategy of the parents) on the survival, growth, behaviour and subsequent life-history strategies of offspring.

The Environment Agency is a partner in research currently being carried out at Bangor University, using scale/otolith microchemistry data to identify fish of anadromous parentage and subcatchment origin of fish in the Conwy and/or Dee river systems.

Such research initiatives will go some way to furthering our understanding of the causes and relative incidence of anadromy within individual catchments. The next logical step in this process will be to try to identify whether juvenile sea trout production correlates with particular habitat features, with the aim of ultimately developing models that predict the abundance of the two separate life-history morphs.

Recommendation 2.3: Continue to promote collaborative research into the causes and relative incidence of anadromy in England and Wales.

2.3 Rivers for which BRPs may be required

Brown trout are widespread throughout England and Wales, occurring in a number of different habitats and facing a variety of pressures. An important consideration in setting BRPs is the scope of rivers for which reference points may be required. Key considerations are:

- the scale at which reference points might be applied;
- whether to focus on principal trout rivers, as was the case with salmon;
- the importance of conserving genetic diversity and variation in life strategies;
- what data might be used in assessing compliance against BRPs.

2.3.1 Application scale

An important consideration in implementing biological reference points for trout will be the scale of application, and what ‘management units’ should be used. These might be reaches, sub-catchments, catchments, or even river basin districts, to tie in with the Water Framework Directive.

It is likely that the different management objectives outlined in Chapter 1 will require reference points with different scales of application. For instance, optimisation of juvenile production will most likely be measurable at site to sub-catchment level, whilst optimising recruitment to homewater fisheries and maximising sustainable catch potential are more likely to apply at a catchment scale. Finally, conserving the diversity of stocks implies a requirement for management at a finer scale, ensuring that the populations which underpin the fisheries are protected.

However, it will be necessary to consider at what scale reference points are practicable, given the data that is currently available. Although the local population is the basic unit of production and therefore the preferred unit of management, we must consider that populations of trout in England and Wales are likely to be too numerous to allow management at population level.

Youngson *et al*. (2003) suggest that this difficulty may be addressed by ‘combining populations in fisheries management units that comprise interchangeable, nested groupings of populations that are both genetically and biologically meaningful’. Such groups are akin to ‘evolutionary significant units’ (ESU). An ESU may be defined as a population or group of populations that merit a separate management or conservation strategy. ESUs are often divided into management units (MU) which represent different populations, or stocks, that are demographically separated and important for the long-term viability of the ESU (Allendorf and Luikart 2007 in Ostergren 2007).

This approach is potentially flexible since interchangeable groupings can be derived relating to particular management targets, so these may apply on a catchment or sub-catchment level. However, it will be important to understand the extent to which a catchment-based reference level can ensure conservation at a finer scale.

2.3.2 Definition of ‘principal trout rivers’

In the production of Salmon Action Plans, assessment of compliance with the conservation limit was restricted to ‘principal salmon rivers’. These were defined as ‘all rivers supporting net fisheries or with mean annual rod catches in excess of 30 salmon or 100 sea trout' (E Black, personal communication). It may be appropriate to focus management efforts for trout in a similar manner, using, for example, a threshold catch of 100 sea trout.

Such an approach would enable management efforts to be targeted at the main sea trout rivers. However, it would not consider the relative importance of the smaller, unfished sea trout streams. Cresswell (1989) recognised the value of such minor sea trout rivers in Wales and pointed out that, collectively, these populations represent a significant proportion of the total trout resource, in terms of biodiversity and conservation value. It is crucial to recognise that this approach focuses management efforts on sea trout as opposed to FR trout.

2.3.3 Genetic diversity

It is important to maintain a diverse stock structure in different rivers, to ensure the conservation of different genetic strains and diversity that help stocks make the most of their home river environments.

There is much diversity within brown trout that can not be placed in a traditional taxonomic framework of species and sub-species (Ferguson, 2004). Trout are known to exhibit considerable genetic variation within and between streams, but few studies have characterised this variation or quantified the levels of genetic differentiation between rivers and catchments. As such, conservation of genetic diversity can only be based on the conservation of genetic differences within and among populations. Many brown trout traits such as survival, growth rate, propensity for migration, disease resistance and temperature and pH tolerance all have a genetic component as well as an environmental one. Conservation of genetic diversity is therefore essential in allowing populations and species to respond to environmental pressures. Although it may not always be possible to link specific genetic differences to phenotypic differences, it may be important to monitor genetic diversity in isolation on the basis that high levels of genetic variation are likely to be beneficial.

For many trout stocks, the longevity and multiple spawning of individuals, the different run timing patterns, and the contributions of resident and migratory fish all contribute to buffering recruitment against the impacts of short-term environmental changes to habitats (Walker *et al.* 2006). Sea trout stock structures are generally more robust than those of salmon in that they exhibit a risk-averse life-history strategy in their pattern of divided migration to sea and subsequent return of adults to the river to spawn for the first time. This pattern potentially limits the impact of any adverse factors affecting survival in freshwater and marine environments in any given year, effectively cushioning the stock from collapse and allowing a faster rate of recovery (Harris 2002).

The conservation of such diversity within stocks will thus clearly be important in maintaining the robustness of stocks. It will be important to understand to what degree trout production and stock structure vary throughout a single catchment and how a catchment-scale reference point can ensure conservation at a finer scale (Harris 2002).

Recommendation 2.4: Monitor genetic diversity of trout stocks where possible.

2.3.4 Data availability

The approach to setting BRPs that is eventually adopted must be ‘measurable’ based on available data.

Table 1.1 outlines the data that is currently available to help assess performance against different management objectives. It will also be important to consider the frequency of monitoring in relation to the desired frequency of assessment.

Data availability and frequency of monitoring for each of the possible approaches to setting BRPs for trout are considered in more detail in Chapters 3 to 6.

3. Habitat-based juvenile reference points

This chapter considers the feasibility of adopting a juvenile-based biological reference point for trout. Juvenile data may be examined either in isolation or in conjunction with some measure of habitat quality or 'carrying capacity' applied at either site, reach or catchment level, allowing management to focus on both the fisheries and on the environmental quality of rearing habitats in freshwater.

The inability to distinguish between sea trout and freshwater resident trout in the freshwater stage means that an approach based on juvenile data will need to consider total trout populations. This may change in the future with the development of tools to recognise the identities and location of sea trout and FR trout, but this is speculative at this stage (Milner *et al.* 2006). An underlying assumption of a juvenile-based approach must therefore be that a healthy juvenile population should provide the resource to support a healthy adult population, given low levels of exploitation.

Key considerations in a juvenile-based approach include:

- how to determine 'carrying capacity', and its different possible definitions;
- the availability and quality of juvenile data upon which to base such an assessment.

3.1 Availability of data

3.1.1 Juvenile salmonid data

Frequency and resolution of monitoring

The Environment Agency routinely monitors the abundance of juvenile trout throughout England and Wales. The current monitoring programme has been in place since 2001 and has involved the monitoring of approximately 340 river sub-catchments representing 29,000 km of river over the past five years. For all major sea trout and principal brown trout rivers, temporal monitoring has been carried out annually at a resolution of approximately one survey site for every 40 km of river. Spatial monitoring has been carried out on a five-year rolling programme at a resolution of around one site for every 5 km of river. The programme also includes 'sentinel' sites, which have been monitored once every five years and represent areas of river which support salmonids but do not fall under any of the main migratory and brown trout fishery categories.

The monitoring programme has recently been under review to bring it in line with the requirements of the Water Framework Directive. The main changes to the programme, effective from 2007 onwards, involve the discontinuation of the sentinel programme, and reduction of the spatial programme by 25 per cent. The frequency of spatial monitoring has now changed to a six-year cycle.

Survey methodology

Three types of survey methods are routinely used in the national monitoring programme to assess salmonid populations:

- ‘quantitative’ surveys provide an estimate of the size of the trout population, using a population model such as the Maximum Weighted Likelihood equation of Carle and Strub (1978);
- ‘semi-quantitative’ surveys provide minimum density figures, and may be calibrated to provide equivalent quantitative population estimates;
- ‘five-minute fry’ surveys provide catch per unit effort figures and are typically used on larger river sites where conventional survey methods are impractical.

3.1.2 Freshwater production data

A number of models can be used to assess the potential freshwater production or carrying capacity of a site or river reach. Such models predict expected fish abundance based on ‘pristine’ conditions. Other models, such as the Fisheries Classification Scheme (FCS), are calibrated on a random selection of survey sites and therefore predict expected fish abundance based on sites with similar habitat, expressed in relatively simple terms of width and gradient parameters.

In setting habitat-based juvenile reference points, it will therefore be important to consider whether it is more appropriate to set reference points based on pristine conditions, using models such as HabScore or RFHI to predict carrying capacity, or on existing conditions, using models such as FCS to reflect the expected abundance of fish based on sites of a similar width or gradient.

HabScore

HabScore is a multiple regression model that predicts expected juvenile salmonid populations for individual sites, given the quality of the habitat, accessibility to migratory salmonids and the position and topography within the catchment. Values for each site are generated as a Habitat Quality Score (HQS), which is a measure of the habitat quality expressed as the expected long-term average density of fish (number per 100 m^2). Values are produced for 0+ salmon and trout (fry), >1+ salmon and trout (parr), and >20 cm trout. The HQS value assumes that neither water quality nor recruitment are limiting. Thus, the HQS provides an indicator of the potential of the site and population of juvenile salmonids expected to preside given pristine conditions, and may therefore be interpreted as carrying capacity.

HQS confidence intervals are generated by the HabScore model. These figures represent the lower and upper 90 per cent confidence limits for the HQS, expressed as number per 100 m^2. In addition, a Habitat Utilisation Index (HUI) is generated. The HUI is a measure of the extent to which the habitat is used by salmonids. It is based on the difference between the observed density and that which would be expected under pristine conditions (HQS) (Barnard 1999).

HabScore surveys have been incorporated into the Environment Agency’s routine monitoring programme since 2001, and by the end of the survey season in 2006, HabScore surveys should be available for all temporal and spatial monitoring sites nationally. In practice, however, a number of surveys have been omitted from the survey programme, due to a variety of factors including access problems and resourcing issues (see Table 3.1 for details of available HabScore surveys by catchment). Further work may be needed to achieve a more extensive spatial coverage in certain catchments if HabScore data is to be used in developing BRPs for trout.

Fisheries Classification Scheme (FCS) & River Fisheries Habitat Inventory (RFHI)

The Environment Agency, in conjunction with WRc, has developed a Fisheries Classification Scheme (FCS) to compare fishery data, assess fishery health and communicate results on a national basis. The classification has been designed to take account of broad habitat types and allows comparisons to be made between sites.

Two levels of classification are used: the 'absolute' classification, which classifies observed fish abundance, and the 'relative' classification , which classifies observed abundance relative to the expected abundance for a given river type, defined by width and gradient. Because the Fisheries Classification Scheme is calibrated on a random selection of survey sites, the outputs reflect the expected abundance of fish, rather than trying to predict pristine carrying capacity.

However, although the existing FCS is able to provide a classification of fish abundance relative to the expected abundance for a given river type, it is not currently able to provide a direct classification of expected abundance or 'habitat quality'. This is a gap that the new River Fisheries Habitat Inventory 3 model intends to fill.

The RFHI models describe the expected abundance of fish in a water body by means of a frequency distribution that is characterised by three distinct parameters; the 'prevalence' or probability that a species is present, the average abundance at sites where the species is present, and the variability in abundance at sites where the species is present. The RFHI models track the changes in these three parameters in response to geographic location and environmental gradients, and these outputs may be used to estimate an overall measure of 'habitat quality', defined as the expected abundance, against which observed abundance may be compared (Wyatt 2006).

Summary

Table 3.1 provides a summary of data availability for HabScore and juvenile survey data for all catchments routinely monitored under the Environment Agency's national juvenile salmonid monitoring programme. HabScore data availability is based on data recently submitted by areas as part of a national data collation exercise (and therefore assumes that areas have submitted all data that is available). Juvenile data availability is based on data collated as part of the recent national monitoring programme review. The spatial sites listed refer to semi-quantitative surveys only, since HabScore data is not collected for timed fisheries surveys (figures taken from the National Fisheries Population Database - NFPD). The availability of individual records of juvenile fish age/length is also taken from NFPD for 2005/2006 and is based on temporal sites only. All figures are based on available national data at the time of publication.

Table 3.1 Juvenile data availability England and Wales

River catchment	Average sea trout rod catch 2001-2005	Number juvenile surveys (temporal)[1]	Number juvenile surveys (spatial)[2]	Individual records of juvenile age/length available?	Number of HabScore surveys available
Teifi	4,000	12	85	N	12
Tywi	3,614	16	63	N	45
Tyne	1,981	8	167	Y	150
Lune	1,877	11	72	Y	81
Wear	1,725	8	107	Y	28
Dyfi	1,651	4	42	N	30
Mawddach	1,370	2	15	N	2
Ribble	1,191	6	72	Y	61

River catchment	Average sea trout rod catch 2001-2005	Number juvenile surveys (temporal)[1]	Number juvenile surveys (spatial)[2]	Individual records of juvenile age/length available?	Number of HabScore surveys available
Avon (Hants)	1,183	11	37	N	ND
Camel	1,108	7	7	Y	35
Border Esk	1,075	13	84	N	74
Fowey	1,060	7	21	Y	24
Clwyd	1,007	5	44	N	5
Dwyfawr	931	1	9	N	1
Nevern	875	2	9	N	8
Rheidol	730	1	8	N	4
Cleddaus	727	6	47	N	13
Ehen	689	2	16	N	3
Dart	677	6	36	N	ND
Ogmore	673	3	14	N	7
Teign	637	5	26	N	ND
Itchen	633	2	16	Y	ND
Glaslyn	606	1	8	N	1
Neath	556	3	23	N	2
Conwy	548	3	22	Y	11
Eden	543	25	154	N	29
Taw	509	14	36	N	ND
Dysinni	482	1	9	N	ND
Loughor	458	5	34	N	6
Tamar	436	20	83	Y	48
Aeron	433	2	12	N	3
Esk (Yorks)	428	3	27	Y	ND
Kent	409	2	16	N	19
Taf	399	6	41	N	8
Coquet	382	7	79	Y	72
Avon (Devon)	378	2	13	Y	6
Llynfi	352	6	1	N	4
Tavy	340	6	19	Y	6
Axe	340	6	23	N	ND
Derwent	331	6	48	Y	ND
Tawe	323	2	12	N	2
Dee	303	13	94	N	9
Torridge	302	9	41	N	ND
Test	245	6	48	Y	5
Usk	243	14	103	Y	40
Frome	204	3	12	Y	6
Dwyryd	203	9	2	N	ND
Ystwyth	189	2	11	N	2
Lynher	186	3	4	Y	19
Ogwen	152	1	3	N	2
Afan	150	2	10	N	12
Looe	143	2	10	Y	ND
Esk (Cumbrian)	124	1	6	N	2
Irt	112	1	8	N	ND
Duddon	109	2	12	N	ND
Sussex Ouse	102	13	13	Y	ND
Plym	102	3	11	Y	3
Aln	100	2	31	Y	22

River catchment	Average sea trout rod catch 2001-2005	Number juvenile surveys (temporal)[1]	Number juvenile surveys (spatial)[2]	Individual records of juvenile age/length available?	Number of HabScore surveys available
Otter	95	3	20	N	ND
Gwendraeth	94	3	14	N	2
Taff	90	4	30	Y	22
Ellen	85	2	8	N	ND
Yealm	85	5	6	Y	6
Tees	83	12	80	Y	ND
Seiont	79	1	3	Y	3
Leven	68	2	17	N	1
Artro	52	6	4	N	ND
Erme	49	1	8	N	ND
Lyn	47	2	1	N	ND
Wye	43	28	178	Y	33
Exe	39	19	59	N	ND
Wyre	38	2	18	Y	19
Stour (Hants)	32	ND	4	N	ND
Piddle	32	3	7	N	ND
Severn	32	248	155	N	40
Rhymney	25	ND	3	N	ND
Calder	9	1	17	N	4
Gwyrfai	7	10	3	Y	3
Thames	1	162	29	N	ND
Crake	-	1	6	N	8
Hodder	-	5	57	N	38

[1]Temporal sites are surveyed annually
[2]Spatial sites are surveyed every six years
ND – no data supplied

3.2 Quality of data

3.2.1 Juvenile salmonid data

The scope of the Environment Agency's core salmonid electric fishing monitoring programme is to provide information on temporal and spatial trends in fish abundance, and to allow spatial comparisons at sub-catchment scale. Individual catchments are now monitored in full once every six years. A smaller network of sites is monitored annually to provide information on temporal trends. However, the current network of survey sites has only been monitored consistently for the past five years. A considerable amount of historical survey data exists, but a lack of consistent annual data precludes the study of temporal variation in most cases.

Juvenile data are thus able to provide an overview of the status of juvenile stocks within a particular catchment once every five, or now six, years, and this information may be compared with estimates of carrying capacity for particular sites to provide an index of performance. The question is, is this frequency of assessment sufficient for the purposes of stock management? It may be that biological reference points based on juvenile data would need to be combined with some other form of annual assessment.

The current monitoring programme provides reasonable spatial coverage of rivers in England and Wales. However, surveys are currently targeted at rivers with a significant salmon or sea trout fishery. Whilst these rivers will undoubtedly contribute to a large portion of the overall trout production, it is important to bear in mind the potential contribution of smaller streams that are not currently monitored routinely, in terms of both trout production and biodiversity.

Existing survey sites are chosen to represent areas of juvenile habitat, typically incorporating a riffle sequence, and sites are normally confined to tributaries and rivers wider than 10 metres (Environment Agency AMS) due to the practicalities of sampling wider sections of river. Trout typically use smaller streams to spawn, so these sites may be considered representative of juvenile trout production. However, it is important to consider the timing of surveys since movement of trout parr in the autumn could influence results.

The level of detail of the data collected during routine juvenile surveys varies around the country. All areas currently record the overall abundance of 0+ and >0+ trout at each survey site, and these estimates may be compared with measures of site-specific carrying capacity. However, some areas record more detailed information on the age structures of populations at each survey site.

The most widely used survey methodology involves semi-quantitative monitoring techniques. Semi-quantitative population estimates must be converted to equivalent quantitative estimates in order to allow comparison with estimates of freshwater production. This process introduces an additional element of uncertainty in the assessment, and this must be taken into account when considering the robustness of the assessment method.

3.2.2 Freshwater production data

HabScore

HabScore combines habitat data with population data to assess the biological performance of a stream section by comparing observed numbers of fish with expected numbers, based on the quality of the available habitat.

However, there are number of issues with the HabScore models that must be considered if they are to be used in determining biological reference points for trout.

Firstly, the HabScore models can only account for the spatial component of the variance of trout abundance, although for English and Welsh streams this can be up to 73 per cent of the overall variance of which the HabScore models explain up to 63 per cent. Proportions of spatial and temporal variation vary substantially with the scale of analysis, from tributary to catchment level (Milner *et al.* 1995). Nevertheless, HabScore does provide a mechanism by which to predict the potential average maximum abundance of juvenile trout for riverine parts of a catchment, against which measured densities may be compared (Walker *et al*. 2006).

Secondly, the HabScore models are calibrated on a total of 602 sites that were surveyed throughout England and Wales during the early 1990s. Whilst HabScore was originally designed to model upland streams in Wales, it has more recently been extended to include all of England and Wales. This has resulted in the loss of sensitivity of the model and the reference sites used are heavily biased to Wales and the North East. It would be useful to have multiple versions of the model, with specific versions for different stream types.

Recommendation 3.1: Investigate the possibility of developing regional versions of the HabScore models based on region-specific reference sites. Note – this issue is already addressed by the RFHI models which assume that habitat quality varies from catchment to catchment, and that catchments that are geographically close to each other will be more similar than those that are further apart.

The survey data used in calibrating the HabScore models was chosen to represent pristine conditions. Whilst survey sites were selected on the basis that they weren't under pressure from factors affecting either recruitment or water quality, it is difficult to be certain that these populations were indeed 'at carrying capacity' at the time the surveys were carried out. It is therefore possible that the HabScore models underestimate expected abundances in some cases. One way of assessing the suitability of the reference sites used in the original calibration exercise is to examine time series of data for these sites, since the original surveys were carried out in the early 1990s. This was carried out for a number of the original SW Wales and N Wales reference sites (those for which data was easily accessible). Trout densities were found to have increased at some sites in recent years, suggesting that not all sites were at 'carrying capacity' at the time of the original surveys.

Recommendation 3.2: Investigate historic time series of data for calibration sites used in the original HabScore models to determine whether these are truly representative of 'pristine' conditions.

HabScore's underlying assumption that recruitment is not limiting is a complex one, particularly for older age classes since >0+ densities of trout may be limited by habitat bottlenecks acting earlier in the lifecycle (Wyatt and Barnard, 1997).

Finally, if HabScore is to form the basis of a method for determining BRPs for trout, it will be necessary to consider the potential implications of the variation in mean smolt age seen around the country on any reference point for older age classes of trout.

Although direct analysis of population smolt age structure is available for some stocks, in most cases information on smolt age distribution and mean length of smolt age classes is derived from scale-reading of adult returns. Several authors have collated data on smolt age structure for rivers where this type of information is available. Solomon (1995), provided a summary of smolt age composition of stocks for a total of 32 rivers in England and Wales, and Harris (2002) provided further information based on scale readings from 16 rivers in England and Wales. The data provided by these studies is summarised in Appendix I.

It is clear that smolt age distribution in England and Wales is dominated by S2 and S3 smolts, with a few S1s and S4s recorded in many stocks. On many rivers there appears to have been a shift in recent years towards increased production of one-year old smolts. It is not known if this trend is a symptom of reduced stock abundance and widespread under-recruitment into the juvenile population leading to decreased competition, faster parr growth and earlier smolt migration as a direct consequence, or if it is more directly linked to climate change (Harris 2002). Similar declines in mean smolt age of salmon have also recently been reported for the River Dee (Davidson *et al.* 2006).

These observed differences in mean smolt age around the country and the suggested trend towards increased production of younger smolts will need to be taken into account when considering reference points based on estimates of freshwater production taken from HabScore, although the resident trout component may make interpretation problematical.

> **Recommendation 3.3:** Investigate differences in mean smolt age around the country and suggested trend towards increased production of younger smolts.

Fisheries Classification Scheme (FCS) & River Fisheries Habitat Inventory (RFHI)

Whilst the Fisheries Classification Scheme (FCS) provides a useful tool for comparing fishery data on a national basis, its use in setting BRPs for trout presents some issues.

Firstly, because the FCS is calibrated on a random selection of survey sites, the outputs reflect the expected abundance of fish, based on contemporary sites, rather than trying to predict carrying capacity. This means that it is only possible to use the FCS to assess performance in relation to current levels of freshwater production.

The FCS uses very basic habitat data (width and gradient) as part of the evaluation process, which limits the sensitivity of the system.

The existing Fisheries Classification Scheme is currently being updated by the RFHI Phase 3 project, which aims to describe the expected abundance of fish in a water body based on geographic location and environmental gradients. The statistical modelling techniques used to develop the RFHI are more advanced than those used to develop the HabScore models ten years ago, and this will be reflected in improved performance and functionality of the models. However, the emphasis of the RFHI project to date has been on catchment-level assessment.

3.3 Reference point options (target or limit)

Having considered the various options for estimating potential freshwater production of juvenile trout, and the availability and quality of juvenile trout data, the next consideration is how to use this data to set and measure compliance with BRPs.

The models outlined in Section 3.1 all provide a means of estimating the expected abundance of fish within a given site or reach, be that based on pristine conditions (so reflecting carrying capacity) or on existing conditions (based on sites of a similar width/gradient). This in itself represents a potential target – a level of production which management action should aim to achieve. Alternatively, we may wish to set a limit reference point based on some proportion of carrying capacity, representing a level of production which management action should seek to avoid falling below.

In the case of juvenile trout, a limit reference point might be defined as some proportion of carrying capacity, or in the case of HabScore, as some proportion of the Habitat Quality Score (HQS). For instance, Walker *et al*. (2006), point out that recruitment at the Maximum Sustainable Yield (S_{MSY}) is generally between about 80 and 90 per cent of maximum recruitment for typical salmonid curves (Healey 1982; Potter *et al*. 2003 cited in Walker *et al*. 2006), and that a suitable limit reference point for juvenile trout production might therefore be set at a similar proportion of theoretical carrying capacity for trout. In the case of HabScore, this would mean approximately 80-90 per cent of the HQS. However, the HUI (representing the difference between the observed density and that which would be expected under pristine conditions) will vary for the same HQS between migratory and non-migratory populations for different age classes.

A suitable target reference point might then be set at a level of juvenile abundance designed to ensure that the limit reference point is met with the desired frequency. In order to set a target reference point, it is necessary to have a measure of the uncertainty involved in our assessments. In the case of HabScore, one possibility is to use the upper confidence limit of the HQS as a target reference point, thereby taking into account the uncertainty in our assessment of carrying capacity.

Assessing compliance against these possible reference points will involve comparing observed trout abundance derived from juvenile electric fishing surveys with the expected abundance equivalent to the appropriate reference point. HabScore already provides such an assessment in the form of the HUI which is a measure of the extent to which the habitat is used by salmonids. HabScore generates an estimate of HUI based on the difference between observed density and that which would be expected under pristine conditions (HQS). Confidence limits of this estimate are provided.

3.4 Link to management objectives

This chapter considers the feasibility of adopting a juvenile-based reference point for trout based on comparing observed juvenile trout abundance with potential freshwater production or ‘carrying capacity’.

This approach allows an assessment of the performance of juvenile trout populations to be made on a site-by-site or reach-by-reach basis, enabling management efforts to be targeted at areas where juvenile production is falling below potential. The approach allows us to consider the quality of available freshwater habitat, allowing management efforts to be targeted at areas where potential production may be improved.

This approach therefore permits us to address our first management objective outlined in Chapter 1: optimising freshwater production of trout. By focusing on the life stage that encompasses both migratory and non-migratory stock components, we are able to maximise the potential for a healthy adult population of both FR trout and sea trout.

4 Catch-based targets

Rod and net sea trout catch data have been collected since 1952 and became widely available after 1974, following the establishment of the Water Authorities with statutory duties to collect catch data; the data have been consistently collected since 1993. Rod catches are made almost exclusively in-river, unlike net fisheries, which are likely to exploit mixed coastal stocks, so this discussion refers to rod fisheries only. Rod catch data serve two broad purposes:

- through their putative representation of sea trout runs in-river (and in-season) they act as indices of stock and so are true BRPs;
- as indices of fishery performance they give a direct measurement of fishery quality, and thus are important in their own right for fishery management.

Two main problem areas with rod catch as BRPs are that catch data refer to sea trout only, as very few FR trout catch data are routinely available, and that the relationship between catch and stock is variable and uncertain. This chapter examines these and other issues surrounding the interpretation and use of rod catches as trout BRPs.

4.1 Data availability

Sea trout rod catch data are reported through the national licence return system, the return rate for which has varied from approximately 20-30 per cent in 1993 to 71-76 per cent for the period 1994 to 2004 inclusive (Milner *et al.* 2002), and approximately 80 per cent in recent years (Rob Evans, personal communication). Declaration rates for salmon have been estimated at 53 per cent in 1993 and 91 per cent for the period 1994 to 2004 (Environment Agency 2003). No specific declaration rate has been calculated for sea trout, but it is likely to be similar to that calculated for salmon.

Fishing effort data, expressed as number of days spent fishing, have been collated as part of the national licence return system since 1994, and this data can be used to obtain a catch per unit effort figure for sea trout. However, fishing effort data are currently not recorded separately for salmon and sea trout.

Catches of FR trout are not consistently reported, since there is currently no formal licence return system in place for non-migratory species. The only data available come from very limited records held by local angling associations and angler logbook schemes. For this reason, this type of catch-based reference point is not currently applicable to FR trout. This represents a significant gap in our knowledge and the feasibility of introducing a national licence return system to record rod catches of FR trout requires some consideration.

Recommendation 4.1: Investigate feasibility of introducing a national licence return system for recording rod catches of FR trout.

4.2 National, regional and river trends in rod catches of sea trout

Figure 4.1 shows the total declared sea trout rod catch for England and Wales for the period 1978 to 2005 (reproduced and updated from Evans and Greest 2006), as well as the number of sea trout caught and released, and the number of days declared fish by salmon and sea trout anglers since 1994.

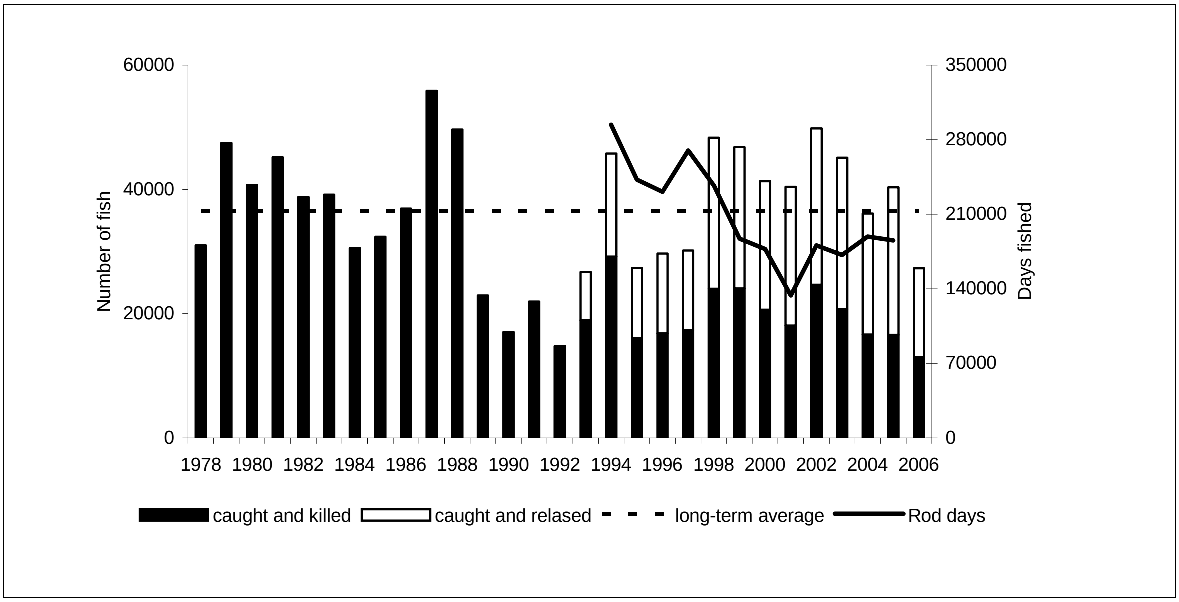

Figure 4.1 National sea trout rod catch 1978-2006 (reproduced and updated from Evans and Greest 2006)

The average declared annual rod catch of sea trout is 36,520 fish, ranging from 14,742 (1992) to 55,863 (1987), though there is no obvious trend in the declared sea trout rod catch for this period. Rod effort has declined from 294,000 days fished in 1994 to 185,000 in 2005, a reduction of nearly 40 per cent in just over 10 years (Evans and Greest 2006), although it is not known how this change in effort has been apportioned between salmon and sea trout fishing.

Effort data recorded since 1994 can be used to generate a measure of catch per unit effort (CPUE) which is considered a more accurate index of stock size than the catch itself, although interpretation of these data is still confounded by the unknown split between the fishing effort for salmon and that for sea trout. Information on species-specific rod effort data has recently been collated through questionnaires sent out with the first catch return reminder in 2006. Approximately 8,000 completed questionnaires have been received to date, but data has not yet been collated (Rob Evans, personal communication).

Recommendation 4.2: Investigate feasibility of routinely collecting data on rod effort for salmon and sea trout individually.

Figure 4.2 (reproduced and updated from Evans and Greest 2006) shows that sea trout CPUE has increased since the mid-1990s, but has decreased slightly over the last six years. This is in contrast to a relatively stable CPUE in salmon rod fisheries (Evans and Greest 2006).

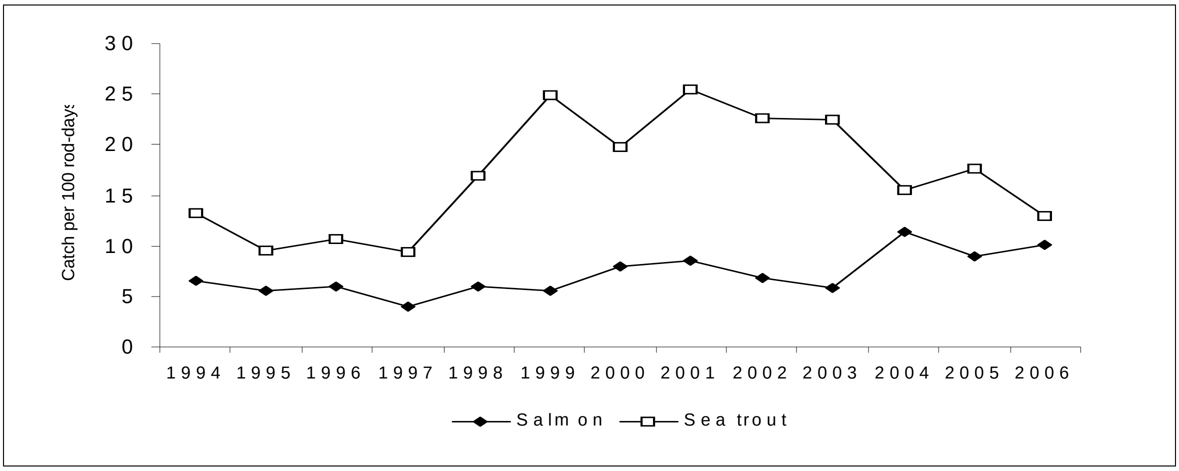

Figure 4.2 National salmon and sea trout catch per 100 rod-licence days (1994-2006) (reproduced and updated from Evans and Greest 2006)

Figure 4.3 shows a map of trends in rod catches in rivers in England and Wales for the period 1974-2005 which is based on data from Evans and Greest (2006). Figure 4.4 (reproduced and updated from Evans and Greest 2006) shows the river-by-river variation of trends in rod catches in England and Wales for the period 1974-2005 (rod catches corrected for under-reporting and $\log_{10}$ transformed). Trends in sea trout rod catches exhibit considerable variation around the country. Catches decreased from 1974 to 2005 in 45 per cent of rivers examined, 17 per cent significantly so (P<0.05). Increases in catches were recorded in 55 per cent of rivers, 37 per cent significantly so (P<0.05). Increased catches were especially notable in rivers in the North East of England and in some of the South Wales rivers that have been recovering from pollution. In other areas, sea trout stocks continue to be affected by water quality problems and barriers to migration (Evans and Greest 2006).

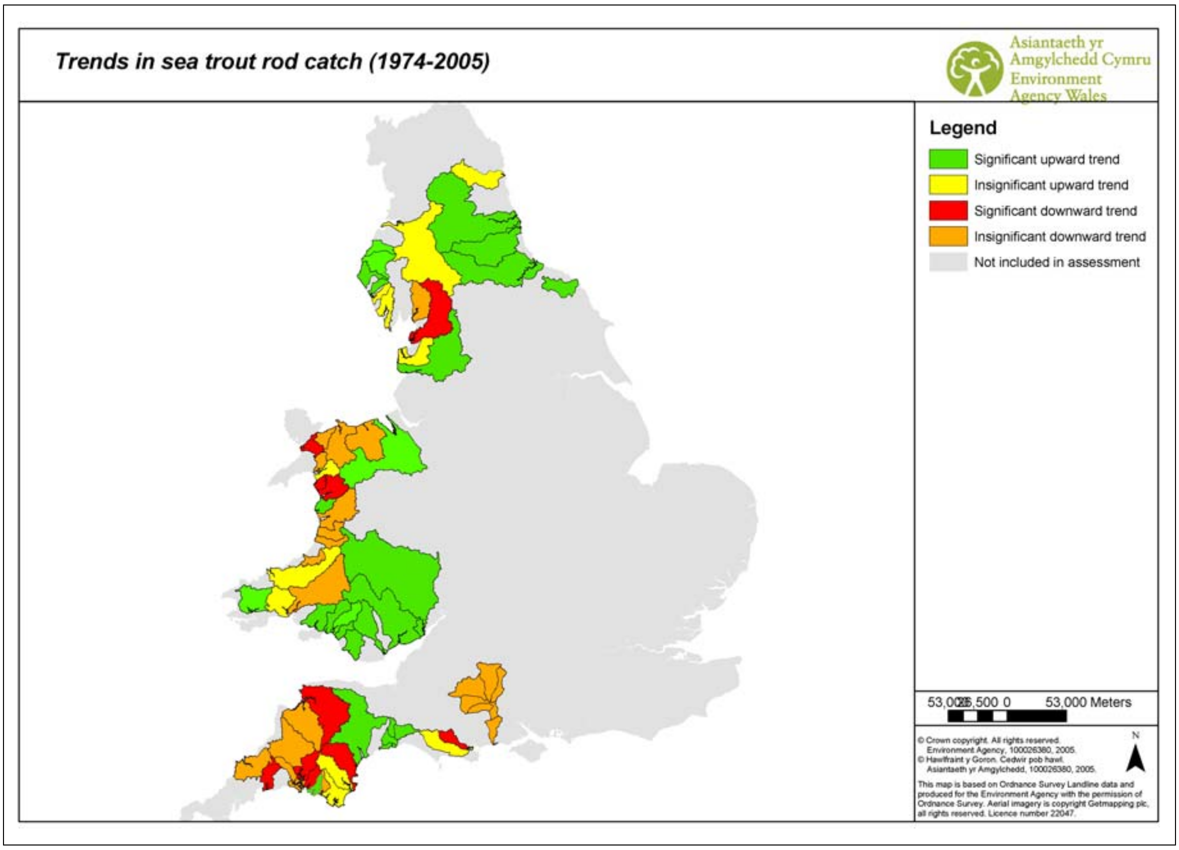

Figure 4.3 Map of rod catch trends in England and Wales 1974-2005

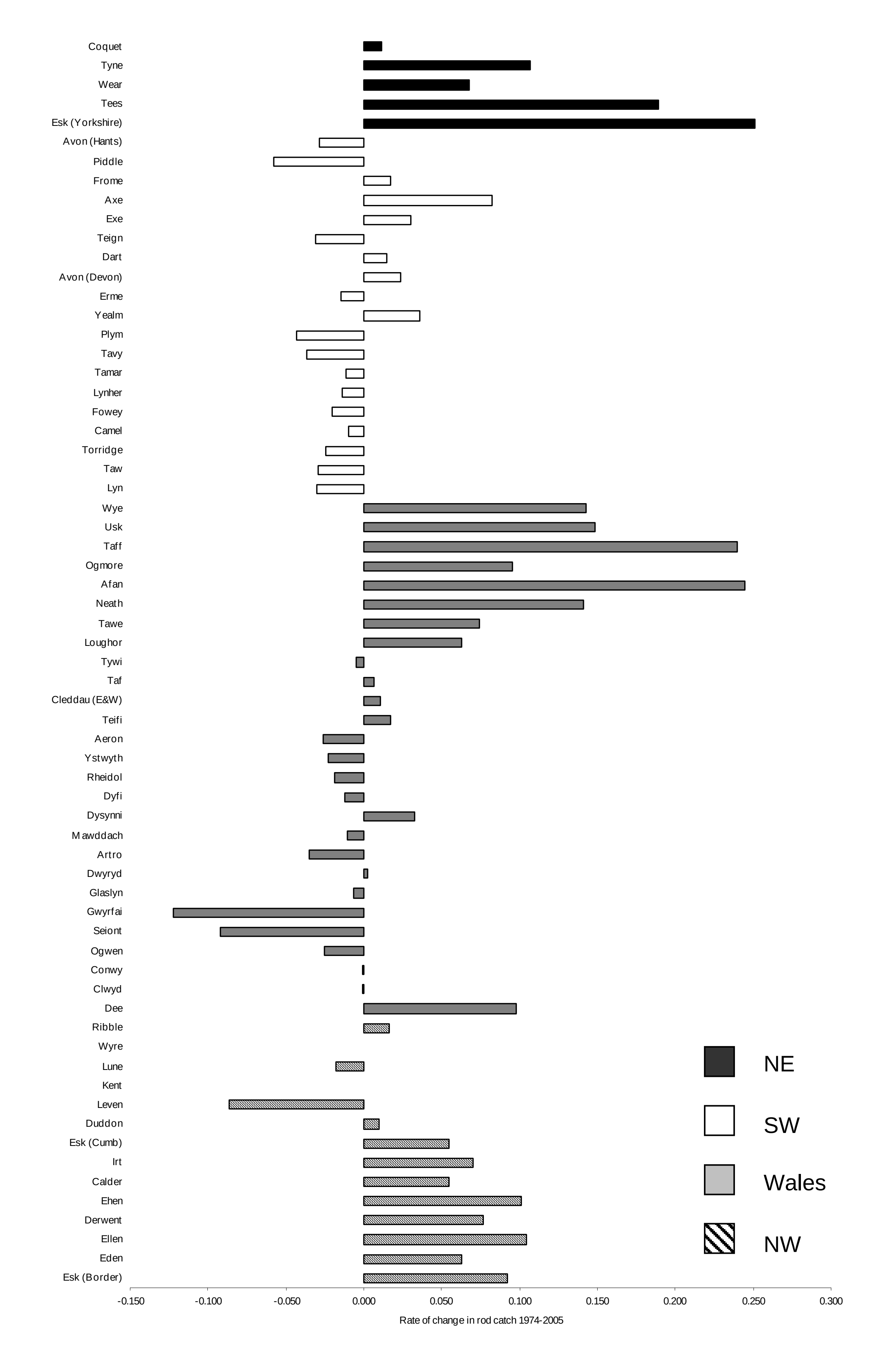

Figure 4.4 River-by-river variation of trends in sea trout rod catches in England Wales (reproduced and updated from Evans and Greest 2006)

It is possible to split rod catches by weight in order to estimate the approximate proportions of whitling (0-sea winter fish identified as .0+) and larger sea trout (one or more sea winters identified as >.0+) in regional rod catches in England and Wales (Rob Evans, personal communication). The weight splits used are based on work carried out by Harris (2002), which suggests that whitling are relatively rare and much bigger on first return in the North East and in the South of England. Thus, slightly different weight splits are applied for these regions. Examining the data in this manner can reveal whether significant increases or decreases in rod catch are attributable to increases or decreases in these particular broad age classes of sea trout. However, it has only been possible to examine rod catch trends for the period 1994-2005 in this manner, since weight split data are not available nationally prior to this time.

On a regional level, significant upward trends were evident in the numbers of >.0+ sea trout recorded in the North West and in the South of England. Conversely, no significant regional trends were evident in the numbers of whitling reported.

On a river-by-river basis, significant trends in the numbers of whitling and >.0+ sea trout reported were evident on a number of rivers for the period 1994-2005. These are summarised in Table 4.1.

Table 4.1 Trends in reported rod catches 1994-2005

Region	River[1]	Total reported rod catch	Rod catch of whitling	Rod catch of >.0+ sea trout
North West	Cumbr. Esk	/	+	/
	Duddon	+	/	/
	Ellen	+	+	/
South West	Dart	-	/	-
	Frome	+	/	+
	Lynher	-	-	/
	Plym	-	-	-
Wales	Clwyd	/	+	/
	Conwy	/	+	/
	Dee	+	/	+
	Dwyryd	/	/	-
	Gwyrfai	/	/	-
	Loughor	/	+	/
	Mawddach	+	/	/
	Neath	/	/	+
	Ogmore	+	/	/
	Neath	/	/	+
	Ogmore	+	/	/
	Ogwen	/	+	/
	Rheidol	+	/	/
	Seiont	/	/	+
	Teifi	+	/	+
	Usk	/	/	+
North East	Tees	+	+	+
	Tyne	+	/	+
	Wear	+	/	+
South	Test	/	/	+
	Itchen	/	/	+

Notes: [1]Only includes rivers for which significant rod catch trends were observed
‘+’ denotes statistically significant upward trend
‘-‘ denotes statistically significant downward trend
‘/ ‘denotes no significant trend

The national catch trend hides a complex mosaic of regional and smaller scale variations in rod catches, reflecting local circumstances. This is borne out through examination of trends in rod catches for individual regions and rivers, and for different age classes of fish within individual rivers.

A number of authors have demonstrated synchronous variation in both salmon and sea trout rod catches of geographically separate rivers, indicating that catches are responding to common influences (Elliott 1992; Milner *et al*. 2001). However, this type of synchronous variation is only evident when examining data from a relatively small number of rivers of the same geographical region. Such strong coherence might not be expected for wider, national groupings of rivers, since these will be subject to a wide variety of factors determined by location and geography that may, for instance, influence marine migrations and survival of different run groups (Milner *et al.* 2001).

The results of a number of studies suggest that catch data do not vary randomly between and within rivers, but rather show quite specific spatial and temporal patterns which may be explicable in terms of the effects of river size, and coherence of national trends in catch, but moderated by the effects of local influences (Milner *et al.* 2001). Such influences on rod catch are examined in more detail in Section 4.3.2.

It is important to consider the influences of run, effort and catchability on stock. These are considered in more detail in Section 4.3.1.

4.3 Factors affecting rod catches

4.3.1 Catch as a predictor of stock

Rod and net catches fluctuate considerably over time, due to factors such as stock size, fishing effort, recording accuracy and environmental, social and economic factors that influence the accessibility, availability and vulnerability of stock to fishermen. Thus, catch data alone may not be a reliable measure of stock abundance, and the relationship between catch and stock size is not always straightforward, regardless of the accuracy of the data.

In most rivers, however, the rod catch represents the only available measure of the size of the stock, so it is important to consider whether general relationships between rod catch and the number of adult sea trout in freshwater can be established.

In classical fishing theory, the catch of fish by any one method at any one time can be derived from the number of fish in the available stock (N), the catchability of those fish (q) and the fishing effort exerted (f). Thus, if values of f and q are known, it is possible to derive an estimate of the stock from catch data (Shelton 2001).

Rod fisheries for homing migratory species such as the sea trout present a slightly more complex scenario, since rod and line fisheries are essentially interceptory and are based upon groups of fish temporarily resident or passing through fishing zones (Shelton 2001).

Effort (f) and catchability (q) are likely to vary considerably, both between rivers and between years, since these will be influenced by a range of factors.

A consistent measure of effort may be difficult to obtain since there may be changes in the anglers fishing from year to year, and their skill, or changes in the methods that anglers are allowed to use (Gardiner 2001). Exploitation rates for any given river may thus vary considerably between years, or even within season. Regulation changes also occur frequently in migratory salmonid fisheries, to alter exploitation rates (Milner *et al.* 2001). Finally, exploitation rate is likely to vary with stock abundance (Hansen 2001).

Variation in catchability is likely to occur both between and within rivers, since catchability is likely to be influenced by the behaviour of both fish and angler, and by environmental conditions (Shelton 2001; Milner *et al.* 2001).

Given these sources of variation, the use of catch data as an index of stock involves considerable error and uncertainty. This area has been relatively under-researched, particularly for sea trout, with few published studies. It will be important to consider how this uncertainty may be reduced and whether it may be reduced sufficiently to consider using catch data as a basis for biological reference points.

In some cases, rod catch has been considered a reliable indicator of a known adult stock size for salmon – for example, the River Bush in Northern Ireland (Crozier and Kennedy 2001), and the River Coquet in North East England (Solomon and Potter 1992). However, in other systems, the opposite has been reported – such as the River Frome and the River Tamar in SW England (Beaumont *et al.* 1991; Hendry *et al.* 2007).

For sea trout, a significant relationship between rod catch and stock size has been demonstrated on the Rivers Tamar and Dee. Conversely, data examined for the Rivers Kent, Lune and Fowey showed no significant relationship between rod catch and stock size (Shields *et al.* 2006).

Whilst it may, therefore, be possible to use catch data in comparisons of relative fish abundance between rivers and between years, rod catch is unlikely to provide a means for determining absolute stock abundance, unless reliable independent measures of effort and 'catchability' are available.

Fishing effort, expressed as number of days spent fishing, has been systematically recorded for all rivers in England and Wales since 1994, although inconsistently reported for the rivers Test and Itchen where owner returns are the principal reporting method (Milner *et al.* 2001). This data can be used to derive annual catch-per-effort figures for sea trout, which are considered a more reliable index of stock size than the rod catch itself, although fishing effort is currently only reported for salmon and sea trout combined. Independent measures of sea trout run size are currently available for a limited number of rivers in England and Wales (those with traps or counters).

Figure 4.5 gives a comparison of rod catch, catch-per-licence and approximate run size derived from trap or counter data for five rivers in England and Wales for the period 1994-2005. It is evident that rod catches are significantly positively correlated with catch per licence day ($P<0.05$) for three of these rivers, indicating that effort is likely to be fairly consistent between years. However, on the Lune and the Kent, rod catch and catch per licence day deviate considerably in certain years. However, declared effort is currently only recorded for salmon and sea trout combined. Conversely, with the exception of the Tamar, independent measures of stock size do not correlate well with rod catches, indicating that rod catches alone may not be representative of stock size for these rivers. The variation in exploitation rates of sea trout between rivers is considerable (Table 4.2).

Table 4.2 Estimates of sea trout exploitation rates for eight rivers in England and Wales

River	Estimated exploitation rate %	Years	Source
Tawe	30.0	1992	Solomon, 1995
Lune	20.5	1993-2004	Shields *et al.* 2006
Fowey	17.9	1995-2004	Shields *et al.* 2006
Kent	12.6	1993-2004	Shields *et al.* 2006
Tamar	4.6	1994-2004	Shields *et al.* 2006
Coquet	2.8	1959-1982	Solomon, 1995
Dee	2.7	1991-1992	Shields *et al.* 2006
Axe	2.5	1960-1976	Solomon, 1995

If rod catches are to form the basis of a catch-based biological reference point for sea trout, it will be important to examine and assess annual exploitation levels in individual rivers. In the case of salmon, work is currently underway to develop a model providing annual estimates of exploitation in response to changing fishing effort for individual rivers (Wyatt 2003). The main input variable to this model is the total salmon fishing effort, measured in terms of licence days, and adjusted for the split in salmon/sea trout effort. It may therefore be possible to adapt such a model in the future to provide similar estimates of annual river-specific exploitation directed at sea trout.

Figure 4.6 provides an overview of the annual variation in fishing effort (measured in terms of rod days) and exploitation rates (derived from rod catch and trap or counter data – see Table 4.2) for five rivers in England and Wales. These two variables appear to be reasonably well correlated, particularly on the Kent, and on the Lune in more recent years. However, no significant correlation between fishing effort and exploitation rate was evident for these five rivers. This issue requires further consideration if an exploitation model such as that detailed above is to be applied to sea trout in the future.

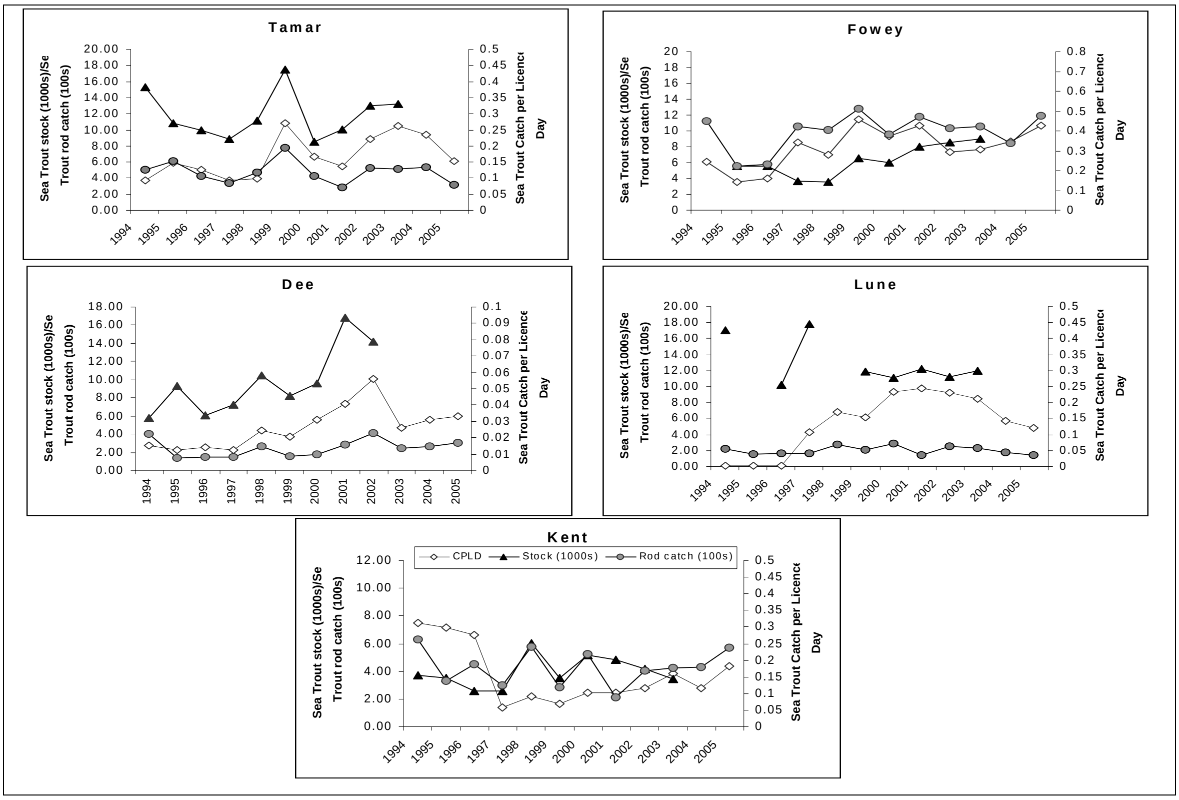

Figure 4.5 Declared rod catch and catch per licence day for eight rivers in England and Wales, 1994-2005.

Recommendation 4.3: Investigate the feasibility of developing a model to provide annual estimates of exploitation for sea trout.

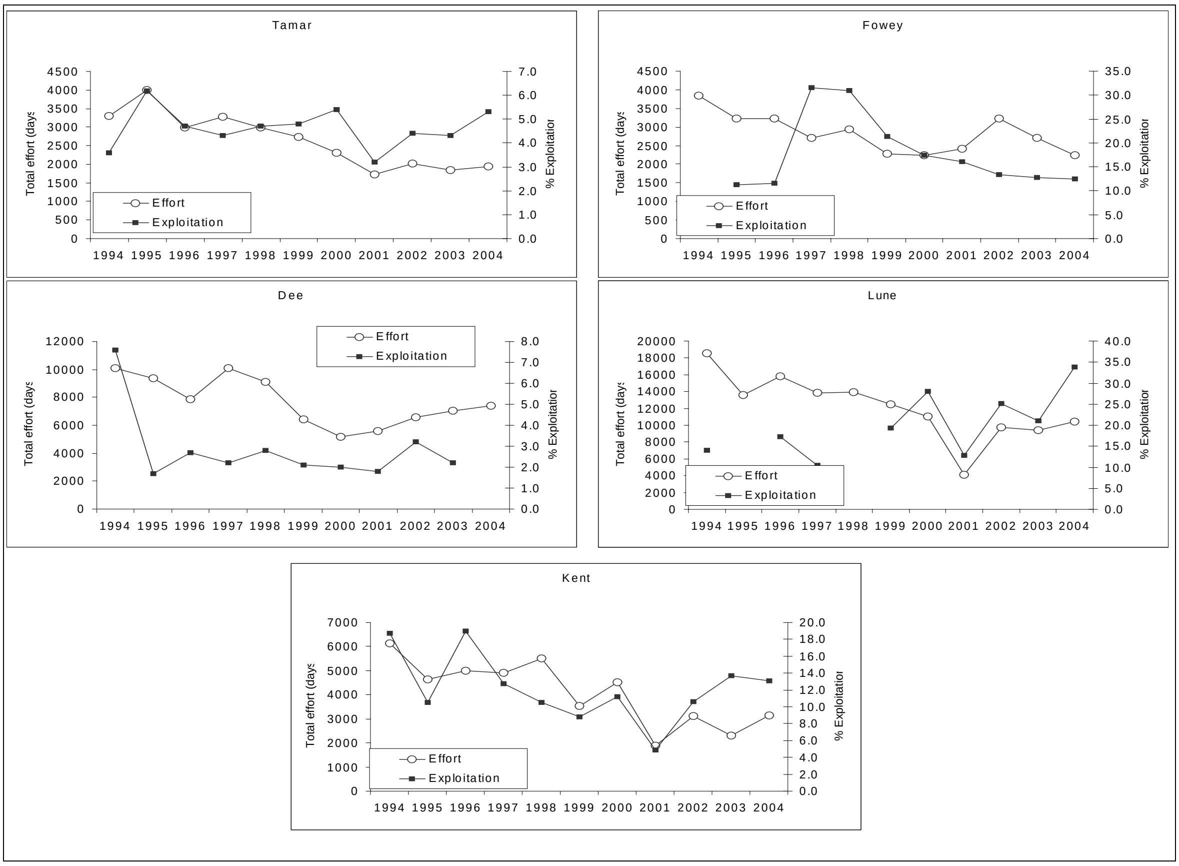

Figure 4.6 Annual variation in observed sea trout exploitation rates and total angling effort on monitored rivers 1994-2005.

4.3.2 Physical catchment features

It is possible to identify physical attributes of river systems associated with higher salmon or sea trout catches. For example, the popular conception is that large swiftly flowing rivers with long main river sections make good salmon rivers, and smaller rivers comprised of many tributaries and short main stems make good sea trout rivers (Champion *et al.* 1998).

Analysis carried out by Champion *et al.* (1998) on data from Welsh rivers supports this view, and suggests that rivers may be categorised by comparing their catch of salmon and sea trout with their physical attributes. Such an approach converges on the HabScore methodology, in that the geographical attributes of a river will dictate its suitability for salmon and sea trout. Examination of the temporal variance in catch may then allow determination of reasons for the variability in different river types.

For salmon, smolt production is believed to be proportional to the available spawning area and juvenile carrying capacity. Since these features are likely to increase with wetted area, it is reasonable to assume that the run of salmon is proportional to the rivers size or some surrogate of size. If rod catch is any index of annual run, significant correlation between river size and mean annual rod catch should be expected (Milner *et al.* 2001).

Figure 4.7 shows that average salmon catches (C), effort (f) and catch per effort (C/f) (period 1994-2005) for England and Wales are all significantly positively correlated with catchment area. For sea trout, a similar correlation might be expected, although the results may be confounded by the presence of FR trout. Correlations for sea trout for the period 1994-2000 (summed values, period 1994-2000) are shown in Figure 4.8.

It is clear from these figures that a significant relationship exists between catchment area and the numbers of salmon and sea trout taken by rod and line. These findings are consistent with work carried out by Milner *et al.* (2006) who found that mean rod catches of both salmon and sea trout increased significantly ($p<0.01$) with river size, but the relationship for sea trout was weaker ($R^2=0.20$) than for salmon ($R^2=0.36$). They proposed a mechanism for this by which trout, which are more abundant than juvenile salmon in smaller streams, would be proportionately more abundant in smaller catchments in which small channels contribute a greater proportion of the wetted area.

However, the relationship between catchment area and catch per unit effort is different for salmon and sea trout, with a positive correlation for salmon and a negative one for sea trout. This suggests that catch per unit effort increases with catchment size for salmon, but decreases with catchment size for sea trout. This may be a reflection of the sea trout's preference for smaller streams, which will contribute more to the total wetted area in smaller catchments. This may equally be an artefact of the data, but removing the outlying rivers (Tees, Exe, Wye and Severn) still produces a highly significant result. However, rod effort is based on the number of days reported fished each year, and this figure does not distinguish between effort for salmon and effort for sea trout.

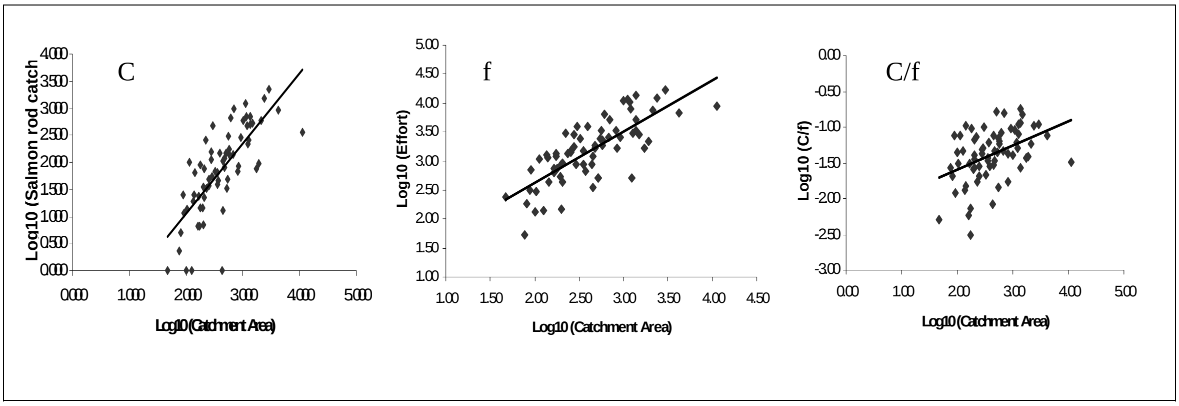

Figure 4.7 Relationship between salmon rod catch (C), effort (f) and (C/f) and catchment size, England and Wales log transformed data (period 1994-2005) (df=68; r^2=56.6%, p<0.001; r^2=57.3%, p<0.001 and r^2=17.6%, p<0.001 respectively).

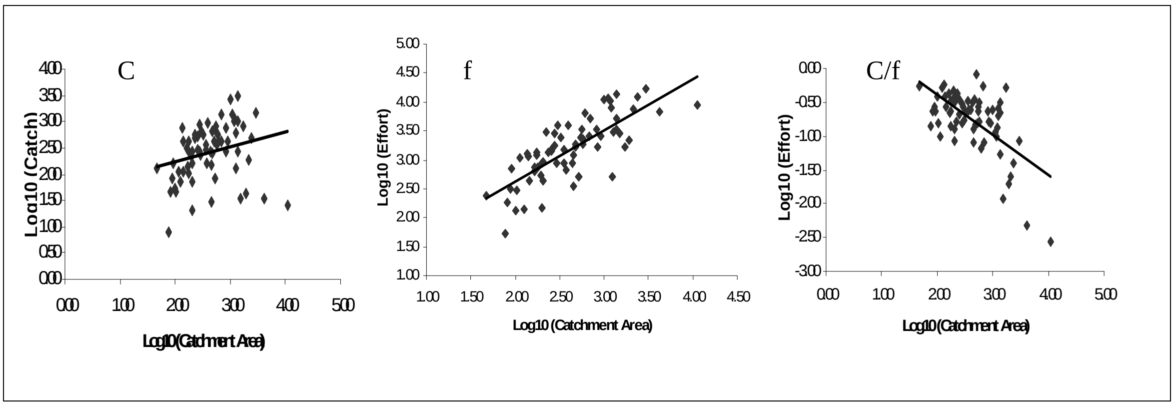

Figure 4.8 Relationship between sea trout rod catch (C), effort (f) and (C/f) and catchment size, England and Wales log transformed data (period 1994-2005) (df=68, r^2=5.9%, p<0.05; r^2=57.3%, p<0.001 and r^2=36.7%, p<0.001 respectively).

Catchment size alone explains only a small proportion of the variation in sea trout rod catches, and it is therefore interesting to consider what other variables might improve the predictive power of this simple model.

Champion *et al.* (1998), for instance, examined a number of physical river attributes believed to influence rod catches of salmon and sea trout; these included the number of tributaries within a given catchment, the length of main river and the average length of tributaries. The results of these analyses suggest that rivers providing a substantial salmon catch are large, with a high ratio of main river to number of tributaries, whilst rivers providing a substantial sea trout catch may also be large, but with a low ratio of main river to number of tributaries.

Other variables likely to influence rod catches of sea trout may be related to the amount of available sea trout producing area, or factors influencing the degree of anadromy expressed by trout populations within a particular catchment.

A number of factors are likely to influence the amount of sea trout producing area available within a catchment.

Barriers to migration, both manmade and natural, can in some catchments vastly reduce the available spawning and rearing area available for sea trout.

Stream size may have an influence on the numbers of sea trout produced within a given catchment. For instance, Milner *et al.* (2006) found that juvenile trout were dominant over juvenile salmon in streams less than six metres wide in all rivers studied within the North East Atlantic Commission (NEAC) area. Because such small channels contribute a greater proportion of total wetted area in small catchments, they argued that sea trout would also be proportionately more abundant in these catchments.

Borgstrom and Heggenes (1988) hypothesised that water levels in the smaller trout streams may be so low that feeding opportunities for larger pre-smolts may be reduced. In this situation, one might expect more trout to smolt young and leave the stream at a smaller body size. Since the decision to migrate is believed to be linked to growth rate and food availability (Jonsson and Jonsson 1993), one might also expect trout residing in these smaller streams to favour a migratory lifestyle, provided that is commensurate with greater overall lifetime fitness.

A number of factors may influence the degree to which anadromy is expressed within a particular catchment.

In brown trout, anadromy is believed to be partly environmentally and partly genetically determined; thus, individuals may be migratory or resident, depending on external influences (Jonsson and Jonsson 1993). It has been suggested that for a migratory life-history pattern to exist, the gain in fitness from moving to a new habitat, minus the cost of moving, must be higher than remaining in just one habitat (Gross 1987).

The advantages of migration are clear. Anadromous fish are able to achieve higher growth rates and larger adult sizes by moving into more productive marine feeding areas, thereby counteracting growth constraints in the juvenile rearing area (Jonsson 1985). Mature migrants benefit from higher age-specific fecundity (Gross 1987). However, migration by anadromous fishes also has a cost. Fish migration involves a cycle of movement and, although the downstream migration in rivers and streams may be energetically ‘cheap’, the upstream return rarely is. The costs of migration include not only physically moving to new habitats, but also physiological adaptations, establishing new territories and dealing with predators and diseases (Northcote 1992).

In many situations only a portion of the population migrates, and the decision to migrate is believed to involve a cost-benefit trade-off, balancing the growth, reproductive and mortality potential of the two habitats (Morinville and Rasmussen

2003). Migration has been shown to be stimulated by low growth efficiency in the nursery area (Forseth *et al.* 1999; Morinville and Rasmussen 2003). However, the tendency to migrate is likely to be offset if migratory costs are too large or feeding opportunities at sea are poor (Jonsson and Jonsson 2006) or if predation risks in lower rivers and estuaries are greater.

A number of studies have examined migratory costs in anadromous salmonids in relation to features such as water discharge and altitude of the spawning site (Bohlin *et al.* 2001; Jonsson and Jonsson 2006). The general conclusions of these studies suggest that the proportion of trout stocks migrating to sea might be expected to decrease with increasing distance between the river mouth and the spawning site, or with increasing altitude, since increasing migratory costs reduce the fitness benefits of anadromy. A smaller proportion of the total trout population might thus be expected to adopt a migratory life strategy in river catchments where a large proportion of suitable spawning area occurs at high altitudes or large distances from the sea. If this is the case, one might expect to see smaller sea trout catches for such catchments.

4.4 Building a simple model to predict sea trout abundance

Having considered which variables may best explain the variation in rod catches of sea trout around the country, it is interesting to consider the feasibility of building a simple model to predict sea trout abundance for individual catchments.

It is clear from data presented in Section 4.3.2 that a significant relationship exists between reported sea trout catch and catchment area.

The next step is to introduce other variables to this model, to improve its predictive power. The following variables were considered for inclusion in the model:

- catchment area adjusted to reflect the area accessible to migratory fish;
- accessible wetted area provided by streams less than six metres wide (based on outputs of a GIS model predicting wetted stream width – outputs provide rough estimate of wetted area only);
- wetted area provided by streams lower than 150 m in altitude;
- number of tributaries within each catchment;
- overall catchment gradient;

The influence of each of these variables on sea trout catch was considered both in isolation and in combination with other variables.

The strongest relationships were achieved using simple linear regressions of the catchment area adjusted to reflect the area accessible to migratory fish, and the accessible wetted area provided by streams of less than six metres in width, against average sea trout rod catch for the period 1994-2005. These regressions are shown in Figure 4.9 and account for 11 and 13 per cent of the variance in rod catch respectively.

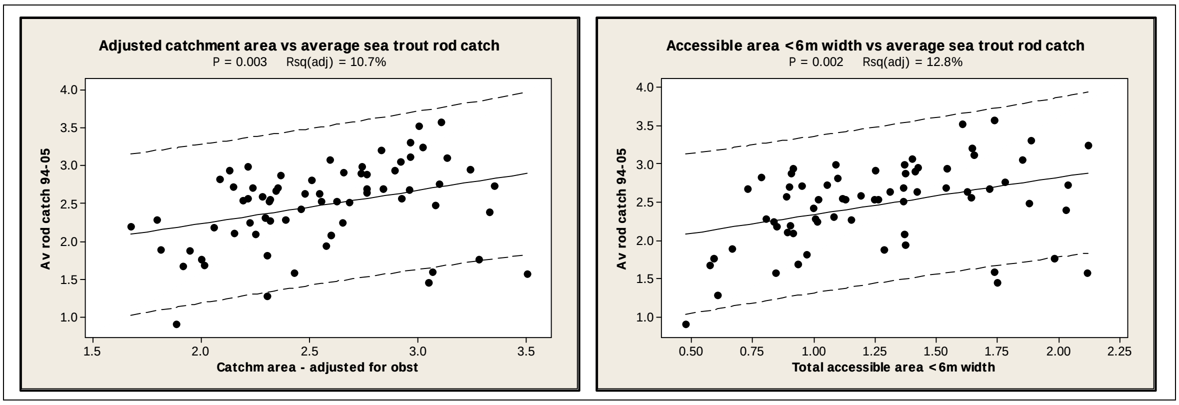

Figure 4.9 Linear regressions of adjusted catchment area and accessible wetted area < 6 m wide against average sea trout rod catch for the period 1994-2005.

Adding other variables to this model did not improve its predictive power, presumably because the variables are essentially surrogates for catchment size. There is clearly still a lot of variability to account for in the relationships illustrated above and this merits further exploration. We may not currently have measures of the main factors influencing variability of rod catches, if those factors are indeed measurable.

Recommendation 4.4: Carry out further investigation of a wider set of river characteristics, with the object of defining a subset of variables accounting for the greatest between-rivers variance of sea trout catch.

One aspect of such relationships which warrants further investigation is the influence of outliers, and the possible reasons for these outliers. It is evident in the above figures that a numbers of rivers with large catchment areas produce only small catches of sea trout – namely the rivers Wye, Tees, Exe and Severn. This may be a result of the physical nature of these catchments, with factors such as extreme tidal ranges and long distance to spawning tributaries making them less suitable for sea trout.

There is thus a case for removing these outliers from the relationship. Doing so improves the predictive power of the model considerably, with adjusted catchment area and accessible wetted area <6 m width then accounting for over 35 and over 38 per cent of the variance in rod catch respectively (Figure 4.10).

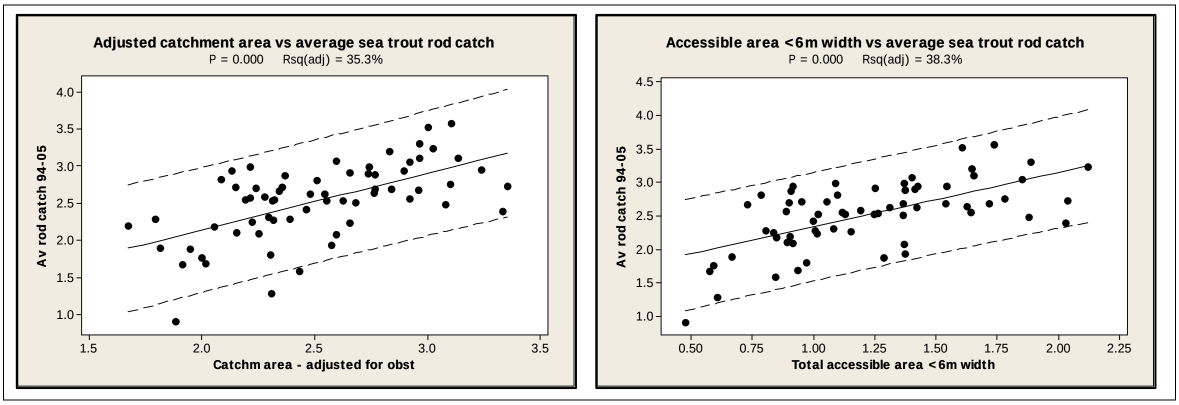

Figure 4.10 Linear regressions of adjusted catchment area and accessible wetted area < 6 m wide against average sea trout rod catch for the period 1994-2005 with outliers (Wye, Tees, Exe and Severn) removed.

4.5 Reference point options

It is clear that rod catch data can potentially serve two main purposes for BRPs:

- Firstly, rod catches may be used as indices of stock, representing true reference points. However, there are a number of sources of variation in catches and the use of catch data as an index of stock therefore involves considerable error and uncertainty.

- Alternatively, rod catches may be used use in comparisons of relative fish abundance between rivers and between years, allowing an assessment to be made of how a particular river is performing in a national context.

Whilst further work is clearly required before catch-based BRPs can be established, it is nevertheless possible to use the information outlined here to understand how a river is performing nationally in its sea trout rod catch. From this, it is possible to examine how a river's performance changes over time.

A decision-tree might therefore include the following questions:

- Is the sea trout catch for the river falling below national expectation based on the catchment size?
- If so, what are the likely causes? Is the catch truly representative of stock, or do the natural features of that river simply make it less suitable for sea trout?
- What do trends in rod catches tell us? Is the river improving or deteriorating?
- Finally, it is important to consider the 'baseline' that has been used to represent 'national expectation' and what implications this may have for the outcome of the assessment.

The performance of the River Tamar is examined in this context in Chapter 6.

4.6 Link to management objectives

This section considers the feasibility of adopting a catch-based reference point for trout based on reported rod catches of sea trout as indices of stock abundance, or in comparisons of relative fish abundance to provide information on performance in a national context. Rod catches may also be used as indices of fishery performance, giving a direct measurement of fishery quality.

This approach ties in with management objectives to optimise recruitment to homewater fisheries and maximise sustainable catch potential. It can, however, only apply to sea trout, since catches of FR trout are not currently consistently reported.

5 BRPs based on stock recruitment relationships

5.1 Salmon management

The current approach to managing salmon stocks in England and Wales, and indeed throughout the North Atlantic Region, follows advice from the North Atlantic Salmon Conservation Organisation (NASCO) that salmon stocks should be conserved by ensuring that an adequate number of spawners enter each river to optimise national production (Environment Agency 2003).

The derivation of an 'adequate' spawning stock size is based on the assumption that the number of fish produced in the next generation (recruits) is related to the number of adult fish in the previous generation (stock) (Walker *et al.* 2006). For salmonids, this premise is supported by a large number of studies on population dynamics (Prevost and Chaput 2001). Salmonid recruitment is strongly influenced by both intrinsic and extrinsic factors, but is believed to be largely determined by density-dependent regulation in the early life stages due to limited resources (Walker *et al.* 2006).

The relationship between spawners and recruits can be summarised in a density-dependent stock-recruitment relationship, and several model types can be applied, corresponding to various theoretical and empirical models. The two main varieties are the 'Ricker' model – represented by a dome-shaped curve where recruits are maximised at some intermediate stock level; and the 'Beverton-Holt' model – represented by an asymptotic curve where recruit level remains constant above some level of spawning stock (Environment Agency 2003). Such curves define freshwater survival, whilst marine survival is described by a 'replacement line', representing the density-independent survival of smolts to returning adults (Figure 5.1).

Stock recruitment models can be used to derive various categories of spawning reference point for use in management. Some are shown diagrammatically in Figure 5.1 and include 'maximum recruitment' (Sm), defined as the level of spawning stock which maximises recruitment (only definable on a dome-shaped curve); 'replacement stock' (Sr), defined as the point where recruits exactly replace the spawners which generated them; and 'maximum gain' (Sg), defined as the escapement level that maximises potential catch levels – the point at which the number of surplus recruits is maximised (Environment Agency 2003).

This third spawning reference point (Sg), representing the stock size at which yield should be maximised, currently forms the basis of the conservation limits used for salmon in England and Wales. In this case, benchmark values for marine and freshwater survival are defined; these equate to an assumed pristine freshwater state for the latter, but correspond to recent marine survival rates in the case of the former to reflect the fall in sea survival rates across the North Atlantic in the last 10-20 years. The number of spawners that would equate to the maximum catch is calculated based on these values and is typically expressed in terms of egg numbers. This value – the 'conservation limit' (CL) - is treated as a lower limit, and is linked to a more precautionary management objective which requires egg deposition to exceed the CL four years out of five to ensure a river stock formally complies with its CL.

Because stock-recruitment (SR) relationships are not available for most rivers, the procedures used to set conservation limits for salmon in England and Wales are based, in part, on parameters derived from an estimated SR relationship for the River Bush in

Northern Ireland (such as the slope of the ascending limb of the SR curve). In addition, a habitat model predicts the height of the SR curve (or carrying capacity) based on river-specific data. The replacement line is also adjusted according to river-specific estimates of sea age composition and sex ratio (Environment Agency 2003).

Stock status is assessed annually in relation to the conservation limit, with spawner numbers usually derived from rod catches and assumed exploitation rates (in the absence of trap or counter-based run estimates). Further details of the methods employed are given in the Environment Agency's Salmon Acton Plan Guidelines (Environment Agency 2003).

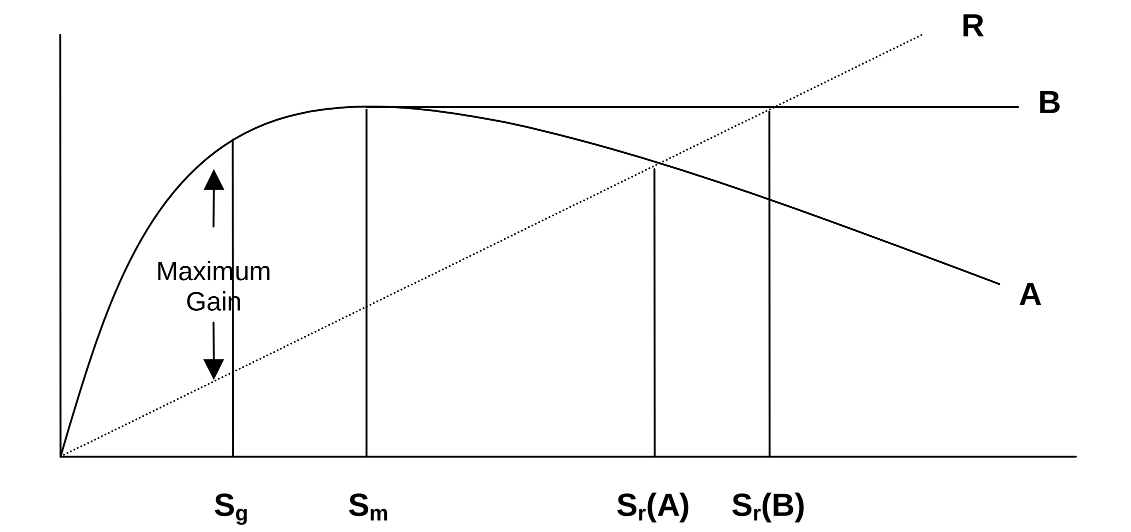

Figure 5.1 Diagrammatic stock recruitment curve (reproduced from Environment Agency's Salmon Action Plan Guidelines).

Whilst this method is relatively well established and widely applied, there remain some areas of uncertainty, particularly in using rod catch to estimate run size. Failure of the conservation limit can be triggered by a number of factors, including low freshwater production or marine survival, or high exploitation rates. It can therefore be difficult to ascertain the reasons for failure in some instances. Secondly, a problem at one stage of the lifecycle may be masked by good performance at another (Wyatt *et al.* 2007).

5.2 Sea trout management

Given that an established method exists for setting river-specific conservation limits for salmon, it would seem logical to apply a similar method to sea trout management, especially since the sea trout's lifecycle is similar in many respects to that of salmon.

The methodology developed for salmon, however, involves the transport of a known stock-recruitment relationship from the River Bush to other rivers, and this process is based on the underlying assumption that the population dynamics of each stock are similar and that differences in river-specific production occur as a result of differences in carrying capacity. Applying such a method to sea trout would therefore require a stock-recruitment relationship for at least one 'typical' sea trout river and a method by which to transport this relationship to other rivers (Walker *et al.* 2006).

However, few studies have described the relationship between stock and recruitment for sea trout, and SR relationships for brown trout populations that include the anadromous form remain largely unexamined. The complex life history of the brown trout, its aptitude for multiple spawning and lack of understanding of the relationships between resident and anadromous populations within the same catchment have generally hampered attempts to establish realistic SR relationships (Poole *et al.* 2006).

Published SR relationships for sea trout are currently available for three river systems: Black Brows Beck in Cumbria, the Bresle in Upper Normandy, and the Burrishoole system in western Ireland. These published SR relationships for sea trout are reviewed in Section 5.3.1.

The River Dee in North Wales has an extensive data set for adult sea trout, and it may therefore be possible to derive a further (adult to adult) stock recruitment relationship for this river system. This is explored further in Section 5.3.2.

Finally, the potential difficulties in determining a biological reference point for trout based on stock recruitment relationships are explored in Sections 5.3.3.

5.3 Review of data – existing SR relationships for sea trout

5.3.1 Black Brows Beck stock recruitment relationship

Black Brows Beck is a small, shallow stream (length 512 m, mean width 0.8 m) located in the North West of England, serving as a nursery for progeny of sea trout. It is one of several tributaries of Dale Park Beck which is a major tributary of Rusland Pool. The total length of the main stem and all tributaries is 27 km, and it discharges into the estuary of the River Leven.

Black Brows Beck was the subject of extensive monitoring over a period of 35 years from 1966 to 2000 (Elliott 1994), and detailed information is available on many stock characteristics. Spawning typically occurs during November and December with eggs hatching in February/early March. The parr stage lasts approximately two years, and the majority of trout migrate to sea in May of their third year. There is little emigration from the stream prior to this time. The majority of spawners are males and females in their fourth year, who have spent two years in freshwater and over a year in the sea. Repeat female spawners are very rare, and all female spawners are anadromous.

An annual census of redd numbers has been carried out in a 120 m^2 study section of the Black Brows Beck since November 1966, allowing information to be derived on the number of female sea trout returning to spawn annually. Excavation of redds has provided information on the mean number of eggs per redd and this has been related back to female body size. Rod catch data indicate that early spawners were the largest females. It is therefore possible to estimate egg density in each year based on the number of redds observed and the time of spawning (Elliott and Elliott 2006).

It has been possible to relate these egg densities to equivalent trout densities at five different life stages: parr aged 0+ years sampled in late May or early June (R_1) and late August or early September (R_2); parr aged 1+ years sampled in late May or early June (R_3) and late August or early September (R_4) and spawning females (R_5).

Various density-dependent SR models were tested on these data, and the Ricker model was found to provide a good fit for all life stages. A significant relationship was evident between total egg production (progeny as R eggs per 60 m^2) and egg density at the start of each year class, and the Ricker model again provided a good fit.

This model was used to determine various possible targets for the sea trout population in Black Brows Beck. Values for the equilibrium density at replacement (S*), density at maximum recruitment (R_{MAX}), parent stock density providing the density at maximum recruitment (S_{MR}), maximum surplus yield of recruits (R_{MSY}) and parent stock density providing the maximum surplus yield of recruits (S_{MSY}) are shown in Table 5.1.

Table 5.1 Black Brows Beck reference point options derived from Ricker model fitted to stock-recruitment data for the period 1966-2000 (Elliott and Elliott 2006)

Target description	Target value
S* – equilibrium density at replacement	4,656 (eggs per 60 m^2)
R_{MAX} – density of progeny at maximum recruitment	5,362 (eggs per 60 m^2)
S_{MR} – parent stock density providing maximum recruitment	2,857 (eggs per 60 m^2)
R_{MSY} – maximum surplus yield of recruits	3,090 (eggs per 60 m^2)
S_{MSY} – parent stock density providing the maximum surplus yield of recruits	1,804 (eggs per 60 m^2)

The targets shown in Table 5.1 are based on the spawning area of the stream where the original estimates were made. These targets may be adjusted to consider the whole of the Black Brows Beck and not just the spawning area. This adjustment results in targets decreased to 37 per cent of the original values (Table 5.2).

Table 5.2 Black Brows Beck reference point options adjusted for whole stream area (Elliott and Elliott 2006)

Target description	Target area	Target value
S* – equilibrium density at replacement	Spawning area	7,760 (eggs per 100 m^2)
S* – equilibrium density at replacement	Whole stream	2,839 (eggs per 100 m^2)
S_{MR} – parent stock density providing maximum recruitment	Spawning area	4,762 (eggs per 100 m^2)
S_{MR} – parent stock density providing maximum recruitment	Whole stream	1,742 (eggs per 100 m^2)
S_{MSY} – parent stock density providing the maximum surplus yield of recruits	Spawning area	3,007 (eggs per 100 m^2)
S_{MSY} – parent stock density providing the maximum surplus yield of recruits	Whole stream	1,100 (eggs per 100 m^2)

The results of Elliott's work on Black Brows Beck clearly illustrate that survivor density at different stages in the lifecycle is dependent on egg density at the start of each year class, with 47 per cent of the variability in egg production between year classes explained by variation in initial egg density. This provides evidence for density-dependent population regulation in Black Brows Beck (Elliott and Elliott 2006).

A Ricker stock recruitment model was found to provide the best fit for the Black Brows Beck data, and fitting this model enabled estimation of various reference point options. However, Elliott and Elliott (2006) advise caution before applying these results to other sea trout populations, pointing out that although the data are for a relatively stable population in a relatively benign environment, negative density-dependent relationships were still only found to account for about half of the inter-generational variability in recruitment. Elliott suggests that density-dependent factors may be less important in harsher environments, and advises against using stock-recruitment models to predict spawning targets for inherently unstable populations.

5.3.2 Burrishoole stock-recruitment relationship

The Burrishoole system is a spate river catchment in western Ireland drained by approximately 45 km of shallow streams which discharge to the north-east corner of Clew Bay, on the mid-west coast of Ireland. The Burrishoole system is characterised by a chain of three main lakes: one brackish water lake, Lough Furnace, and two freshwater lakes, Lough Feeagh and Bunaveela Lough, where the majority of freshwater trout production occurs (Walker *et al.* 2006).

Both resident and anadromous trout populations occur in the Burrishoole system, with sea trout smolts typically running to sea at two or three years old between March and June of each year. The majority of sea trout will return to freshwater as whitling (or finnock) (Poole *et al.* 2006). The Burrishoole system was stocked with around 50,000 reared trout between 1993 and 1998 as part of a sea trout enhancement programme.

Fish trapping has taken place in the Burrishoole since 1958, with a full census of all trout movements both up and downstream from 1971 onwards. These data were used to estimate the annual spawning escapement of sea trout classified into three sea age categories: finnock - also known as whitling (.0+), one sea winter (.1+) and older fish (>.1+). Sex ratios and mean fecundities were estimated from historical trap and rod catch data (fecundity determined using ovaries removed from 102 rod-caught females), from which an estimate of the number of eggs deposited each year was made. The relationships between spawning stock (as estimated egg deposition) and recruitment as (i) spring smolts, (ii) smolts and 1+ juvenile autumn trout and (iii) total recruitment (0+ & 1+ autumn trout and 2- and 3-year smolts) were then described by fitting both Beverton-Holt and Ricker models to the Burrishoole dataset (Poole *et al.* 2006).

The asymptotic Beverton-Holt model was found to fit the Burrishoole data better than the Ricker model for all levels of recruitment. This is likely to be related to the fact that the Burrishoole system encompasses the entire catchment, including considerable lake area. The Burrishoole suffered a collapse in sea trout stocks in the late 1980s, in common with many other rivers in the mid-western region of Ireland. It is therefore likely that the stock may have been below the system's carrying capacity for much of the time series examined (Poole *et al.* 2006).

The study by Poole *et al*. (2006) of trout population dynamics in the Burrishoole is one of the first studies to consider relationships between spawning stock and subsequent recruitment in a population of stream and lake cohabiting migratory and resident trout. The SR model suggests that the production of smolts, or juvenile recruits, is closely related to the level of egg deposition by migratory trout. This supports the view that the contribution of eggs spawned by resident trout is probably low, and that the propensity for marine migration is probably under strong genetic control (Poole *et al.* 2006).

Poole *et al.* (2006) use the outputs of the fitted model to derive a BRP representing the egg deposition limit (stock) below which recruitment and adult return are most strongly reduced (the inflection point). Thus, for the Burrishoole system, an annual deposition of at least 476,000 eggs is necessary to maintain total sea trout recruitment above this point. This equates to approximately 800-1,008 eggs per hectare (in 472 hectare of productive habitat) (Poole *et al.* 2006). These values are summarised in Table 5.3.

Table 5.3 Burrishoole reference point options derived from Beverton-Holt model fitted to stock-recruitment data for the period 1971-2003 (Poole *et al.* 2006)

Target description	Recruitment based on	Target value	Target per unit area (eggs per Ha)
1/b (inflection point) – representing the egg deposition limit (stock) below which recruitment and adult return are most strongly reduced	Smolts + 1+ juvenile autumn trout	376,000	-
1/b (inflection point) – representing the egg deposition limit (stock) below which recruitment and adult return are most strongly reduced	Total trout recruitment	476,000	800-1,008

5.3.3 Bresle stock-recruitment relationship

The River Bresle is a chalk stream which flows in a north-westerly direction through the Normandy-Picardy plateau and drains into the English Channel at Le Tréport. The main channel is 72 km in length, 40 km of which is accessible to migratory salmonids.

Sea trout runs on the Bresle are evaluated by double trapping, coupled with mark-recapture operations using three trapping facilities. Trapping commenced in 1982 for smolts and 1984 for adults and has run to the present day. Each adult fish is measured and weighed and sex is determined using external criteria for the autumn run only. Fecundity has been estimated based on 114 female fish caught by rods and nets. Ninety per cent of smolts are measured and length-weight relationships and age distributions are estimated from selected samples.

The results of trapping operations indicate that the majority of sea trout smolts go to sea at age one, after which adults return annually to spawn. Spawning runs are dominated by 1SW fish (.1+). Resident brown trout can contribute up to 12 per cent of the smolt run in the Bresle.

Euzenat *et al.* (2006) described the relationship between spawning stock and recruits in the River Bresle using both Beverton-Holt and Ricker SR models. Two reference points were derived from these SR relationships: S_M, which defines the level of exploitation that maximises returning spawners, and S_G, defined as the spawning level that maximises the potential catch. Neither model was well matched to the SR data, although their parameter estimates were similar. The Ricker model was nevertheless found to provide the best fit and was used to derive the reference points in Table 5.4.

Table 5.4 Bresle reference point options derived from Ricker model fitted to stock-recruitment data for the period 1984-2002 (Euzenat *et al.* 2006)

Target description	Target value
S_R – replacement stock	1,550 spawners laying 3.4 million eggs
R_{MAX} - maximum production	7,000 smolts equivalent to 2.6 smolts per 100 m^2
S_{MR} - spawning stock giving maximum recruitment	955 fish, equivalent to 2.4 million eggs or 875 eggs per 100 m^2
S_{MSY} – stock level providing maximum surplus production	605 spawners or 1.5 million eggs

Although both the Ricker and Beverton-Holt models give consistent parameter values, the variability of recruitment suggests that recruitment is independent of stock. Euzenat *et al.* (2006) warn that the stock-recruitment relationship cannot be considered robust. This presents obvious problems if this model is to be transported to other rivers.

5.3.4 Comparison of published stock-recruitment relationships

The biological reference points derived from stock-recruitment models fitted to trout population data for the three river systems described above are presented in Table 5.5. The reference points derived from these models are based on different measures of recruitment (eggs or smolts) and use different definitions of 'productive' area. It is thus difficult to compare the reference points put forward for sea trout in each system, particularly for the Burrishoole system where the derived reference point is based on a different definition to the other two systems.

It is clear that the three river systems differ considerably in their physical characteristics and the life-history characteristics of the sea trout they produce. Black Brows Beck, for instance, is a very small stream supporting adult sea trout of limited sea age diversity which typically spawn only once; competition from other species is limited. Work carried out on the Burrishoole and the Bresle, in contrast, has allowed the examination of catchment-scale stock-recruitment relationships – one for a lake-river system (the Burrishoole) and one for a river-only system (the Bresle). In both cases, the authors draw attention to the difficulties in considering the effects of non-anadromous trout and other fish species (principally salmon) (Milner *et al.* 2006).

Table 5.5 Reference points derived from stock-recruitment models fitted to trout data for the Black Brows Beck, the Burrishoole and the Bresle (taken from Elliott 2006; Poole *et al.* 2006 and Euzenat *et al.* 2006).

River	Model providing best fit	Stock measured as	Recruits measured as	Reference points based on	Area used	Reference points (per 100 m²)				
						S	R_{MAX}	S_{MR}	R_{MSY}	S_{MSY}
Black Brows Beck	Ricker	Eggs	Eggs	Spawning area only	150 m^2	7,760	8,937	4,762	5,150	3,007
Black Brows Beck	Ricker	Eggs	Eggs	Whole stream area	410 m^2	2,839	-	1,742	-	1,100
Black Brows Beck	Ricker	Eggs	Eggs	Whole river	20,000 m^2	1,164	-	714	-	451
Bresle	Ricker	Eggs	Smolts	Accessible juv habitat	2,700 units of 100 m^2	1,259	2.6	875	-	556

River	Model providing best fit	Stock measured as	Recruits measured as	Reference points based on	Area used	Reference points (per 100 m²)	
						1/b (eggs)	1/b (smolts)
Burrishoole	Beverton-Holt	Eggs	Smolts	'Productive habitat'	472 Ha (4.72 km^2)	5.06	0.034

5.4 Derivation of SR relationship for sea trout from Dee data

Long-term datasets such as those discussed in Section 5.3 are incredibly valuable in developing a better understanding of stock-recruitment processes in trout, though the availability of such datasets is currently limited. With the single exception of the growing time series of data now available from the River Dee Stock Assessment Programme (DSAP), no in-depth and long-term monitoring studies on sea trout stock abundance and composition are currently being undertaken in England and Wales. Good datasets are currently being developed for a number of other index rivers such as the Tamar and the Lune, although the monitoring programmes for these rivers have not been running as long as the DSAP programme.

Partial trapping and tagging of adult sea trout has been carried out on the River Dee at Chester weir since 1991, with approximately 20,000 sea trout sampled to date. Annual estimates of run size, for whitling and older fish separately, are provided by mark recapture data based on fish trapped and marked at Chester weir then recaptured at the weir in the year following tagging. Trapped fish are examined to collect data on age, length, weight and other biological characteristics.

Returning maiden adult fish (.0+, .1+ and .2+ fish) may be split into year class groups by adjusting for the proportion of different smolt ages (S1, S2, S3, S4) occurring in each year. This information can be used to provide an estimate of the total number of recruits per year class group, which can be plotted against the spawners that gave rise to them. In the example given here, spawners are expressed as eggs, based on applying a length fecundity relationship derived by Solomon (1994), along with estimates of the proportion of females and of maturing whitling to the Dee data.

Spawners (eggs) may be plotted against recruits (maiden adults) to explore the relationship between stock and recruits on the Dee. A Ricker model was fitted to the data using the method detailed by Hillborn and Walters (1992). The fitted curve is shown in Figure 5.2 and provides a reasonable fit for so few data points, though the fit is not significant (P = 0.21).

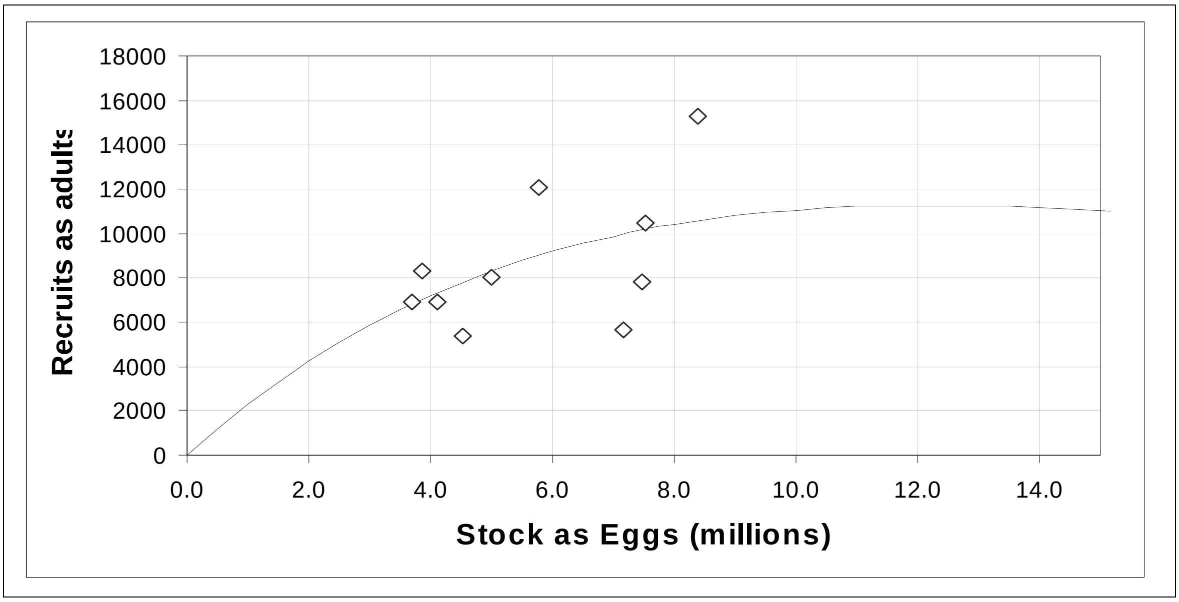

Figure 5.2 Ricker model fitted to stock-recruitment data for the River Dee 1991-2001.

The fitted Ricker model suggests that maximum recruitment on the Dee occurs at a stock size of approximately 11 million eggs, which equates to an approximate egg deposition of 178 eggs per 100 m^2, based on an accessible area of 6.17 km^2. This figure is considerably lower than those derived from stock-recruitment models fitted to the Black Brows Beck and Bresle data, though higher than that derived for the Burrishoole. However, the River Dee is a much larger river system than the Black Brows Beck, Burrishoole and Bresle, and it is likely that the main river contributes a high proportion of wetted area, whilst contributing little trout spawning. It is therefore interesting to consider the egg deposition rate at maximum recruitment with the main stem wetted areas removed. This gives an approximate accessible wetted area of 5.48 km^2 and an approximate egg deposition of 200 eggs per 100 m^2.

Recommendation 5.1: Carry out further analysis of Dee data. For example, examine fit of a Beverton-Holt model.

5.5 Reference point options – difficulties applying method to other rivers

Using stock-recruitment relationships to derive biological reference points for sea trout management presents a number of problems, not least in understanding the reproductive contribution of non-anadromous trout. The SR relationships outlined in these case studies are from very different types of catchment, and can therefore not currently be transported to sea trout populations in general. Further development of biological reference points based on SR relationships is therefore likely to be constrained, at least in the short term, by these information gaps, and it will be important to explore alternative or complementary methods (Milner *et al.* 2006).

The management of Atlantic salmon stocks in Eastern Canada was previously based on a generalised annual target egg deposition which had its origin in the work of Elson on the Pollett River in New Brunswick during the 1950s and 1970s (Elson 1957, 1975). A target egg deposition value of 240 eggs per 100 m^2 of fluvial habitat was considered as the egg deposition level that maximised smolt production and was applied equally to all rivers, regardless of the characteristics of the river or of the returning adults.

Since the transferability of stock-recruitment models between rivers is not possible, this generalised annual egg deposition target for salmon may represent an acceptable interim method of setting SR-based BRPs for sea trout. Such an approach would involve deriving an acceptable annual egg deposition target for sea trout which could be applied across all rivers. The published SR models for sea trout provide a basis for such an approach; indeed, the whole river targets for the Black Brows Beck and the Bresle are quite similar.

5.6 Link to management objectives

This section has examined the feasibility of deriving biological reference points for trout based on published stock-recruitment models for sea trout.

Whilst this approach is not presently feasible given current data limitations, with further advances in our understanding of sea trout, reference points based on salmon-type egg-deposition targets have the potential to address a number of the management objectives outlined in Chapter 1 – namely optimising freshwater production of trout and recruitment to homewater fisheries, and maximising sustainable catch potential.

6 Assessment of trout stocks and use of BRPs for trout; a case study of the River Tamar

The River Tamar rises at Kilkhampton, and flows in a southerly direction to the English Channel via Plymouth Sound. Land use within the catchment is predominantly agricultural. The total catchment area of the River Tamar is approximately 914 km^2, and the average annual run off is 22.5 m^3s^{-1} (Hendry *et al.* 2007).

The Tamar is considered a prime game fishing river, offering fishing for FR trout throughout the whole system and angling for migratory sea trout and salmon in the main river and lower reaches of the major tributaries.

Tamar sea trout stocks have remained reasonably stable over the past 30 years, as indicated by the rod catch (Figure 6.6), with no significant trend in evidence. No data are available for rod catches of FR trout. There has, however, been a substantial deterioration in water quality in the upper Tamar catchment over the same period, the primary causes of which are believed to be land use changes and poor farming practices (Hendry *et al.* 2006).

Fisheries data for the River Tamar have been collected consistently over a long period of time, and good quality datasets exist for both adult and juvenile trout. Available datasets are summarised in Table 6.1.

Table 6.1 Fisheries data available for the Tamar catchment

Data type	Time period	Frequency
Rod catches	1952-2005	Annual and monthly catch
Rod effort	1994-2005	Annual
Net catches	1952-2005	Annual catch
Fish counter	1994-2005	Annual and monthly counts
Juvenile surveys	1971-2005	1 & 5 years (Q & SQ)
HabScore surveys	2001-2005	Five-yearly

These sources of data are used in this chapter to explore possible approaches to setting reference points for trout, providing a comparison of stock assessment methods. The outcomes of each approach are compared and their management implications considered, along with the strengths and weaknesses of each approach.

6.1 Assessment based on juvenile reference points

6.1.1 Available data

HabScore survey data are available for a total of 48 survey sites in the Tamar catchment. These can be used in conjunction with data from electric fishing surveys to assess the performance of juvenile trout populations on a site-by-site basis.

Table 6.2 shows the numbers of fisheries sites surveyed annually for which HabScore survey data are also available. Data collected prior to the Environment Agency's monitoring review in 2001 are not included.

Table 6.2 Number of sites surveyed annually in the Tamar catchment

Year	Number of quantitative surveys	Number of semi-quantitative surveys
2002	8	3
2003	10	2
2004	9	36
2005	11	3

The Environment Agency's monitoring programme involves the monitoring of a small number of sites within the Tamar catchment on an annual basis, to assess temporal trends in fish populations. A wider network of sites has been monitored on a five-yearly basis to examine spatial variation in fish populations. The last full catchment survey for the Tamar was carried out in 2004. The programme has recently been reviewed and the frequency of monitoring has changed to a six-year cycle as of 2007. A proportion of the spatial survey sites are likely to be dropped form the programme in the future.

The spatial distribution of survey sites is shown in Figure 6.1. Those sites monitored on an annual basis are shown in green; these sites are surveyed quantitatively. Sites monitored once every five/six years are shown in blue; these sites are monitored using semi-quantitative techniques.

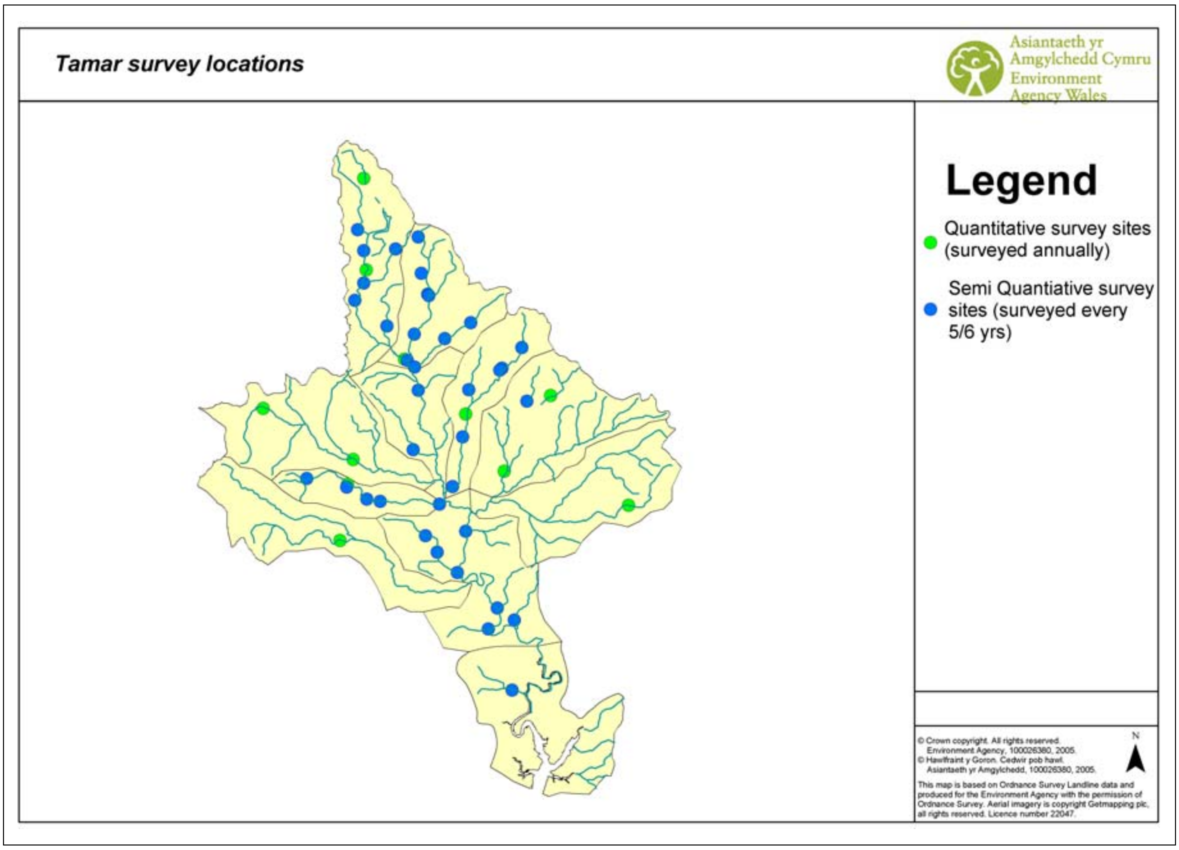

Figure 6.1 Spatial distribution of Tamar survey sites

HabScore provides an estimate of the long-term average fish densities expected at a site, termed the Habitat Quality Score (HQS). Since the estimate is based on notionally pristine reference sites, HQS may be interpreted as a measure of carrying capacity for these sites. Separate estimates are given for trout fry, trout parr and trout above 20 cm.

These estimates may then be compared with observed fish densities to provide an indication of performance. Separate assessments may be made for individual sites for whichever years fisheries survey data are available.

Recorded and predicted trout densities can be plotted on histograms, to compare distributions of observed trout densities and estimates carrying capacities. Figure 6.2 provides an overview of the distribution of expected trout densities (HQS) for all sites surveyed during 2004. Figure 6.3 shows the distribution of observed trout densities for all sites surveyed during 2004. These figures indicate that, for the most part, observed densities of trout are below expectation. Although there are a number of sites recording reasonably high densities of trout (fry and parr), for many sites, densities of all age classes of trout were below five fish per 100 m^2. Another feature of the survey data is the relatively high number of zero densities recorded, when all of the sites offered some habitat suitable for fry and parr.

It is useful to consider the observed fish densities as a proportion of the HQS. The HabScore model provides this type of information as a HUI for each site. Figure 6.4 provides an overview of HUI values obtained for sites surveyed in 2004. Figure 6.5 provides an overview of the geographical distribution of HUI values throughout the catchment. It is evident that fish densities at the majority of sites are currently below carrying capacity. For fry, it is evident that observed densities fall well short of expected, with the majority of sites achieving habitat utilisation indices of less than 0.1. For parr and adult trout, there is a more even spread of HUI values.

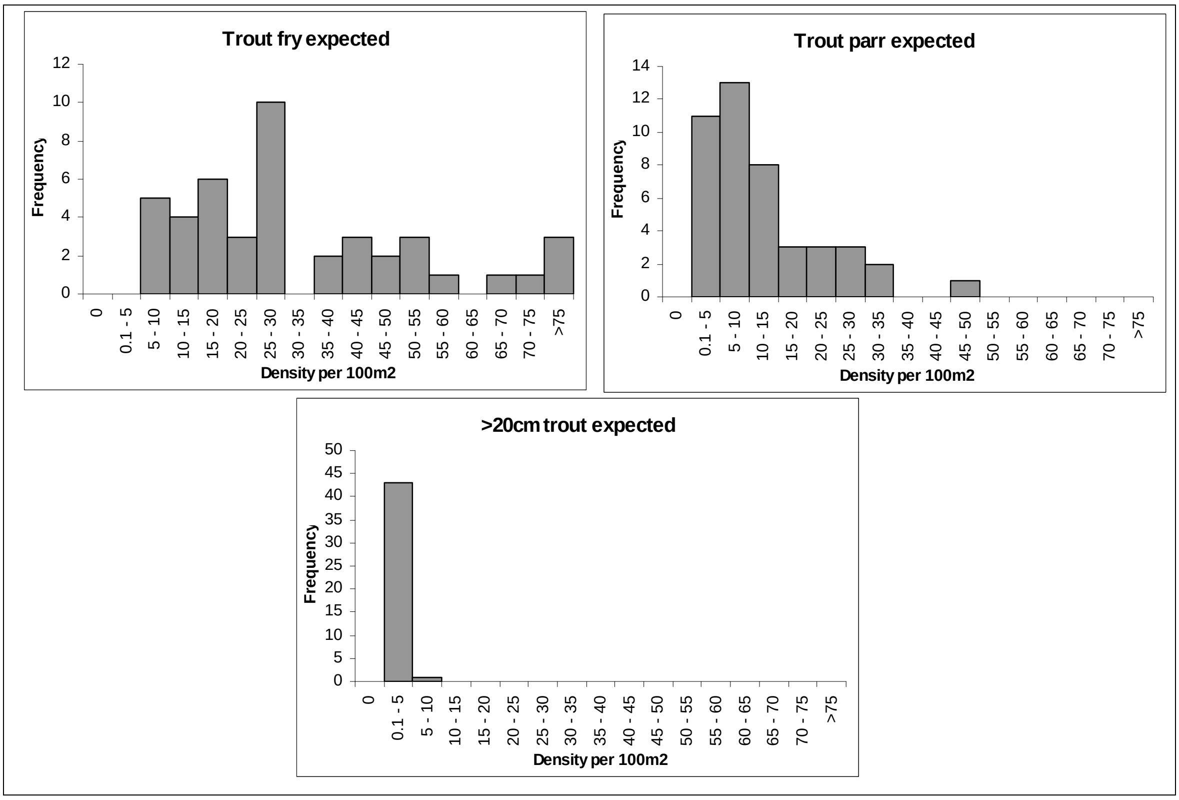

Figure 6.2 Histogram of Habitat Quality Scores for trout, Tamar 2004

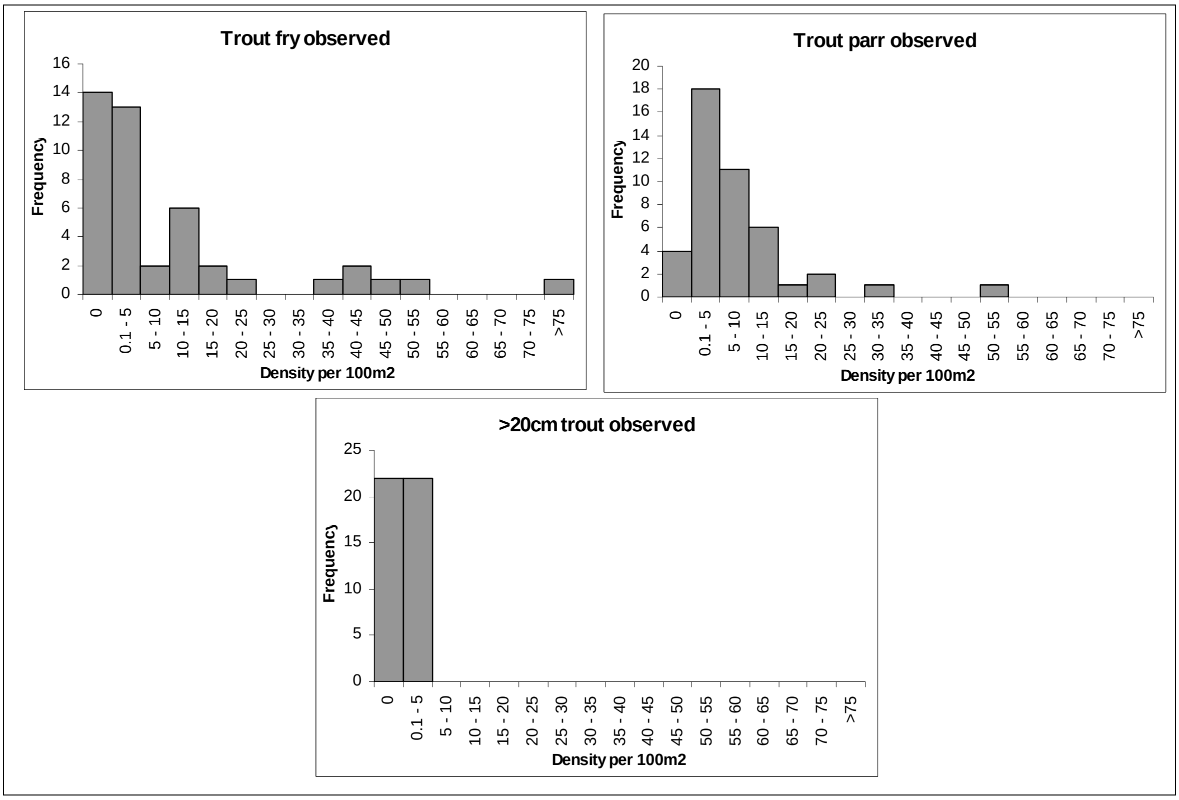

Figure 6.3 Histogram of observed trout densities, Tamar 2004

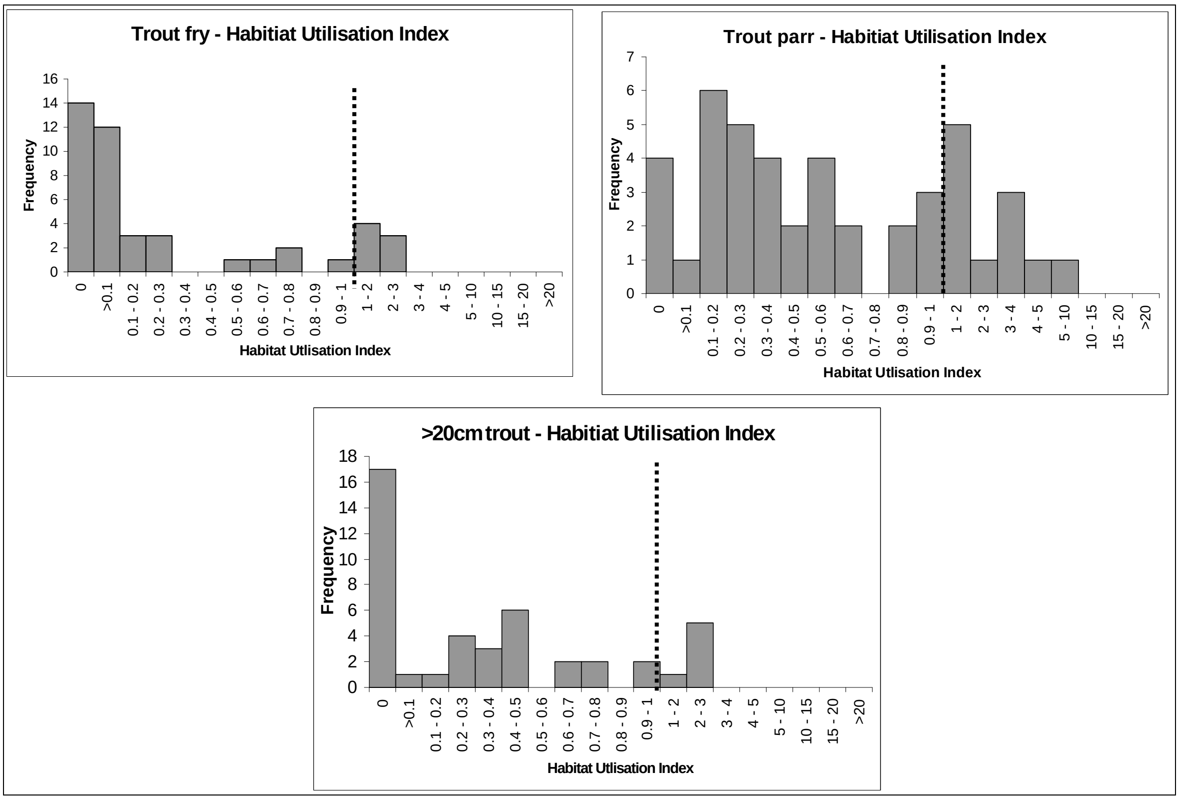

Figure 6.4 Histogram of Habitat Utilisation Indices, Tamar 2004

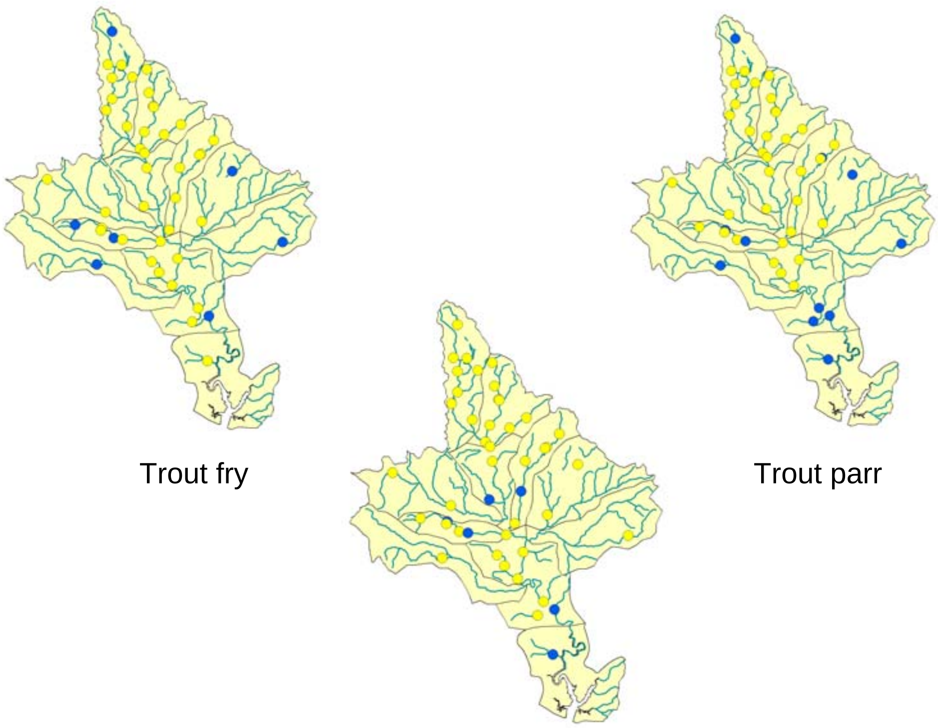

Figure 6.5 Geographical distribution of Habitat Utilisation Indices, Tamar 2004 (HUI <1 shown in yellow; HUI >=1 shown in blue)

6.1.2 Target options

Various reference point options are discussed in Section 3.1.3 and compliance against some of these reference options is explored here using 2004 data for the River Tamar. The reference point options under consideration are as follows:

- The HQS itself is taken as a 'target' reference point representing carrying capacity. This target would imply a management aim of ensuring that all juvenile rearing habitat in freshwater is maximally used. This reference point may therefore be comparable to the 'maximum recruitment' point on a stock-recruitment curve.
- A 'limit' reference point may be taken as some proportion of the HQS. For instance, Walker *et al*. (2006) point out that that recruitment at S_{MSY} is generally between about 80 and 90 per cent of maximum recruitment for typical salmonid curves (Healey 1982; Potter *et al*. 2003 in Walker *et al*. 2006), and that a reference point for juvenile trout production might therefore be set at a similar proportion of theoretical carrying capacity for trout.
- Alternatively, we may choose to adopt a less conservative limit reference point based on some lower proportion of the HQS. A limit reference point based on an arbitrary value of 50 per cent of carrying capacity is proposed in this example. However, recruitment falls off rapidly at spawning stock levels below S_{MSY}.

In all cases, the assessment measure will be the observed fish abundance taken as a proportion of the target. HabScore automatically generates this statistic for the observed fish density as a proportion of HQS, referred to as the HUI. However, it will be important to take account of the uncertainty in assessment methods.

HabScore provides 90 per cent confidence limits around its estimate of HQS, and this data may be factored into any assessment of performance. One way of doing this may be to generate a HUI based on the upper 90 per cent confidence limit of HQS. Alternatively, it may be possible to use the lower 90 per cent confidence limit of the HUI as an assessment measure.

The various target options are summarised in Table 6.3.

Table 6.3 Juvenile reference point options based on HabScore

Reference point	Type	Assessment measure	Comments
Habitat Quality Score (HQS)	Target	Habitat Utilisation Index (1.0)	HQS taken as measure of carrying capacity
Upper confidence limit of HQS	Target	Habitat Utilisation Index based on upper confidence limit of HQS	This option factors in uncertainty in the estimate of HQS
80 per cent of HQS	Limit	Habitat Utilisation Index (0.8)	80% of HQS is taken as equivalent to recruitment at MSY
50 per cent of HQS	Limit	Habitat Utilisation Index (0.5)	50% of HQS is taken to represent lowest acceptable level

6.1.3 Assessment of juvenile trout performance in the Tamar catchment using different reference point options

HabScore survey data and juvenile fisheries survey data collected during 2004 were used to assess juvenile trout performance in the Tamar catchment, using the reference point options outlined above. Table 6.4 shows the number of sites ‘passing’ in each scenario, with the percentage shown in brackets. A total of 41 sites were surveyed.

Table 6.4 Assessment of juvenile trout performance in the Tamar catchment 2004 using different reference point options (HUI as assessment measure)

	Number of sites (%) achieving target[1]			
Age category	**Upper cl of HQS[2]**	**HQS[3]**	**80% of HQS[3]**	**50% of HQS[3]**
Trout fry	0 (0)	7 (16)	9 (20)	12 (27)
Trout parr	2 (5)	11 (24)	15 (33)	21 (47)
Trout adults	1 (3)	6 (13)	8 (18)	12 (27)

[1]Percentage of sites passing shown in brackets
[2]Percentages based on total of 38 sites for which data were available (confidence limits were not provided for estimates of HQS from seven sites)
[3]Percentages based on total of 45 sites for which data were available

Examining the assessment of performance against the various ‘target’ or ‘limit’ reference point options outlined above provides a range of results. These range from virtually all sites failing when using the upper confidence limit of the HQS as a reference point, to between 18 and 33 per cent of sites passing when using 80 per cent of the HQS as a reference point.

Using the ‘limit’ reference point of 50 per cent of the HQS provides a good overview of which sites should be considered at particular risk. Between 53 per cent (parr) and 73 per cent (fry/adults) of sites currently fail to reach even 50 per cent of carrying capacity, depending on what age class is assessed. This suggests that these sites are likely to be impacted in some way if the assumptions of HabScore hold true – namely, that recruitment and water quality are not limiting. However, using a reference point based on 50 per cent of 'carrying capacity' could be considered too high risk.

The question remains as to what the management aims and response should be. For instance, should freshwater production at each site for each age class be maximised, or is this unrealistic, given the central assumption of the HabScore model that recruitment and water quality are not limiting. Ultimately, this decision will depend on the level of risk that is considered acceptable.

Recommendation 6.1: Establish what level of risk is acceptable to fisheries managers in setting biological reference points for juvenile trout.

This type of site- or reach-scale juvenile assessment allows areas where juvenile production is limited to be identified, allowing management efforts to be targeted at these areas. Such an approach takes into account the quality of the available juvenile habitat and allows management efforts to be targeted at areas where potential production may be improved. It may be more prudent to examine HabScore assessments for trout alongside those for salmon, where the two species co-occur, to better inform judgements about performance of sites and potential impacts.

6.1.4 Reach/catchment-scale assessment

HabScore software can only analyse data from a single site. However, the most powerful analysis of impacts comes from an assessment of fishery and habitat data from a number of sites within a river reach. A method is available to compare observed trout populations to HabScore predictions for a number of sites, though this method assumes that the habitats at sites within the reach are not very different (Wyatt *et al.* 1995). If this assumption is unlikely to be correct, but the model variances for the sites are still small compared to the unexplained variance, the method will still be approximately correct. If neither of the two conditions is met, the method will tend to be conservative, tending to underestimate the significance of any difference.

The main difficulty in implementing this method is that many of the sites for which data are available have been surveyed semi-quantitatively, and variance estimates for the converted population estimates are therefore not available. HabScore is unable to generate figures for the unexplained and model variances, making it impossible to estimate the variances of the reach-based HUI estimates.

Recommendation 6.2: Review calibration methods for converting semi-quantitative data to quantitative data. Establish consistent approach across E&W and consider use of converted data within HabScore model (variance estimates are required).

Other catchment-scale assessment options such as the Fisheries Classification Scheme and the River Fisheries Habitat Inventory are available, and the merits of these are discussed in Section 3.1.2.

In conclusion, this HabScore-based assessment suggests that the juvenile trout populations on the Tamar are underperforming.

6.2 Assessment based on catch-based reference points

This section examines the River Tamar's performance in relation to catch-based targets. Long-term trends in rod catches are considered alongside the Tamar's performance compared to other rivers nationally.

Rod catch records for the River Tamar extend back to the early 1950s, although rod catch data have been more consistently recorded since 1974. Rod catches of sea trout in the River Tamar have fluctuated since 1974, though no trend is evident for this period (Figure 6.6). Catches increased through the late 1970s, reaching an all-time high in 1981 and then decreasing through the 1980s to an all-time low in 1990. Catches increased again in the early 1990s and have since remained reasonably stable, with a mean catch of 508 fish recorded for the period 1996-2005. Catches of sea trout in the Tamar are made up predominantly of whitling. Estimated proportions of whitling and older sea trout taken by the rods during the period 1994-2005 are shown in Figure 6.7.

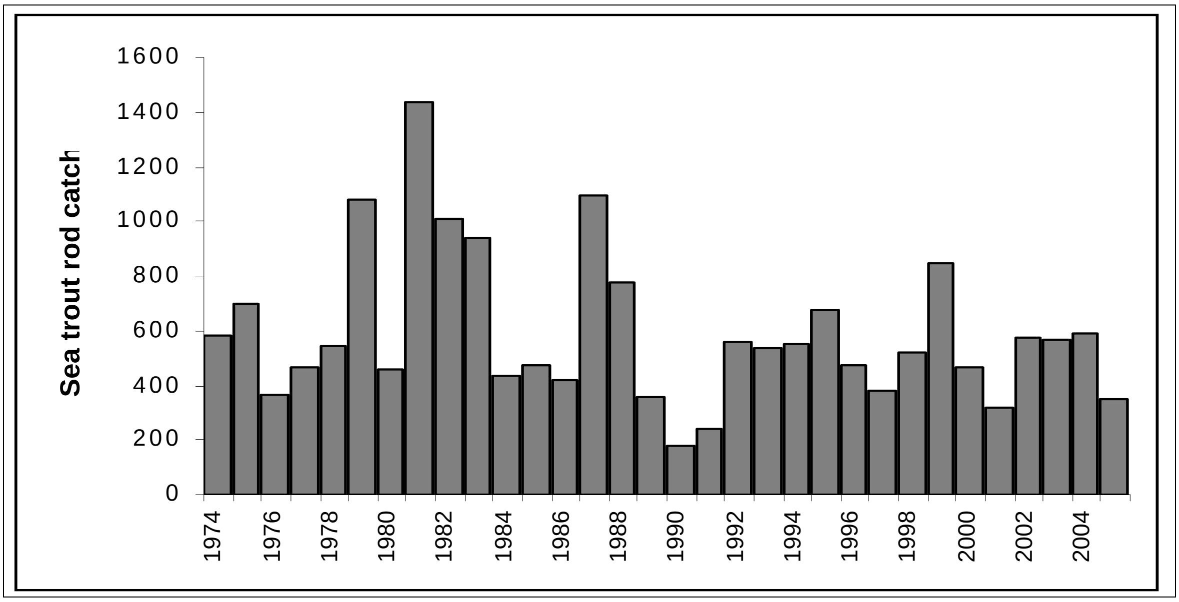

Figure 6.6 Sea trout rod catches River Tamar 1975-2005

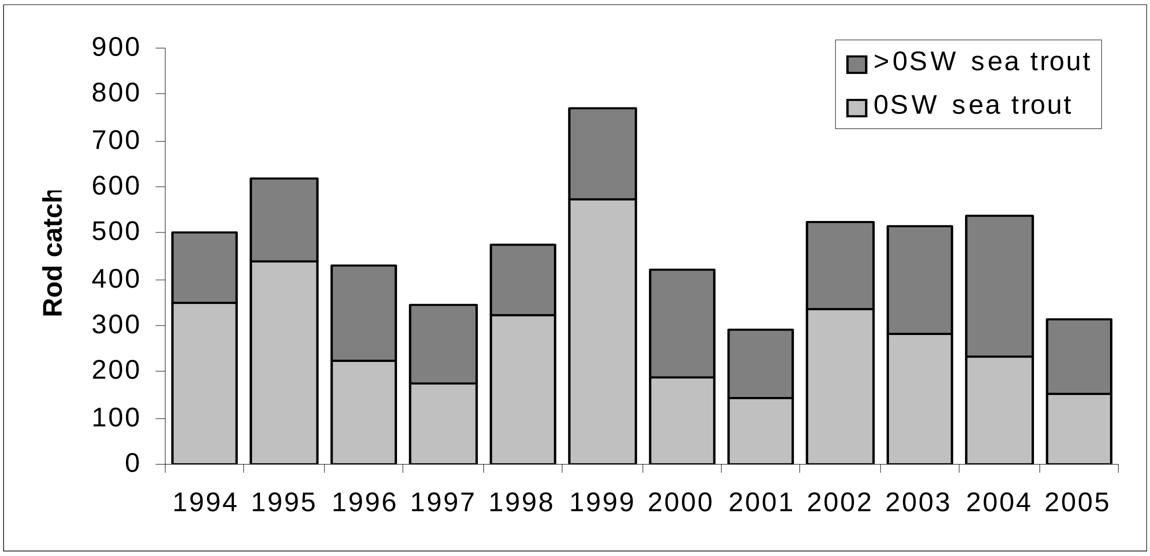

Figure 6.7 Estimated proportions of whitling (.0+SW) and older sea trout (>.0+SW) in Tamar rod catches

Upstream migrating sea trout in the Tamar have been monitored since 1994 with a validated fish counter located at the head of tide at Gunnislake Weir. This provides an independent measure of stock size which may be compared against rod catches. Counts are partitioned into salmon and sea trout based on signal size, since there is little overlap in the size ranges of the two species in this river. Figure 6.8 shows the recorded counts of sea trout for 1994-2006 adjusted for fish pass efficiency which is estimated to be 75 per cent. As with rod catch, there is no trend evident for this period.

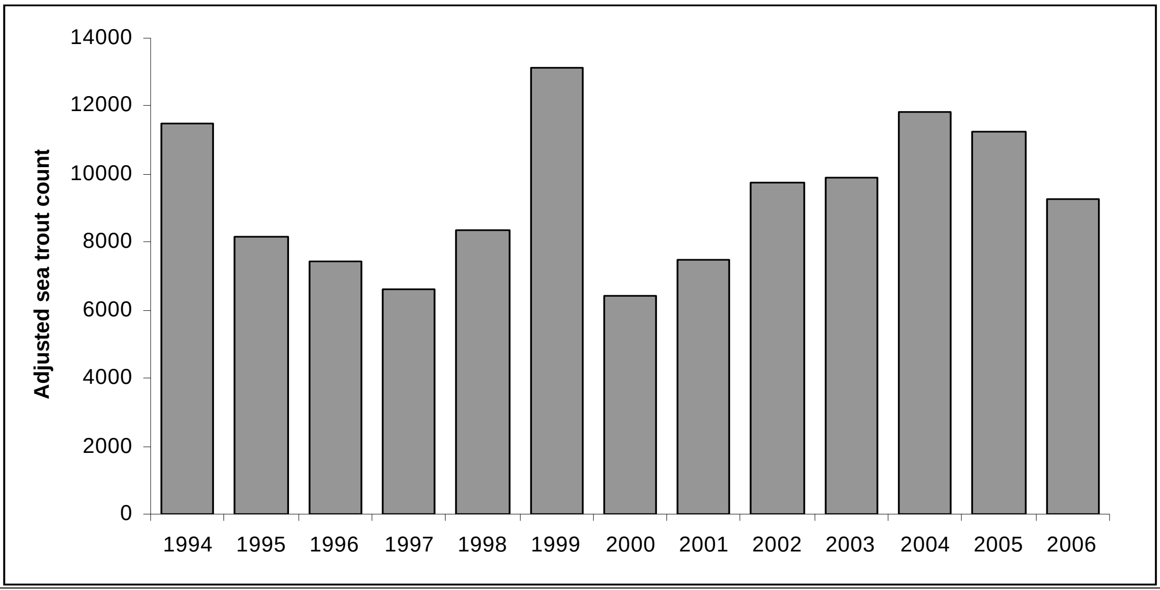

Figure 6.8 Adjusted counts of sea trout at Gunnislake Weir, River Tamar

Comparison of the declared rod catch and recorded sea trout counts at Gunnislake enables estimation of the annual exploitation rate for sea trout on the Tamar. The estimated level of exploitation has remained relatively stable at a low level throughout the study period, ranging from 3.2 per cent in 2001 (when exploitation is likely to have been affected by access restrictions imposed as a result of the foot-and-mouth disease outbreak) to 6.2 per cent in 1995.

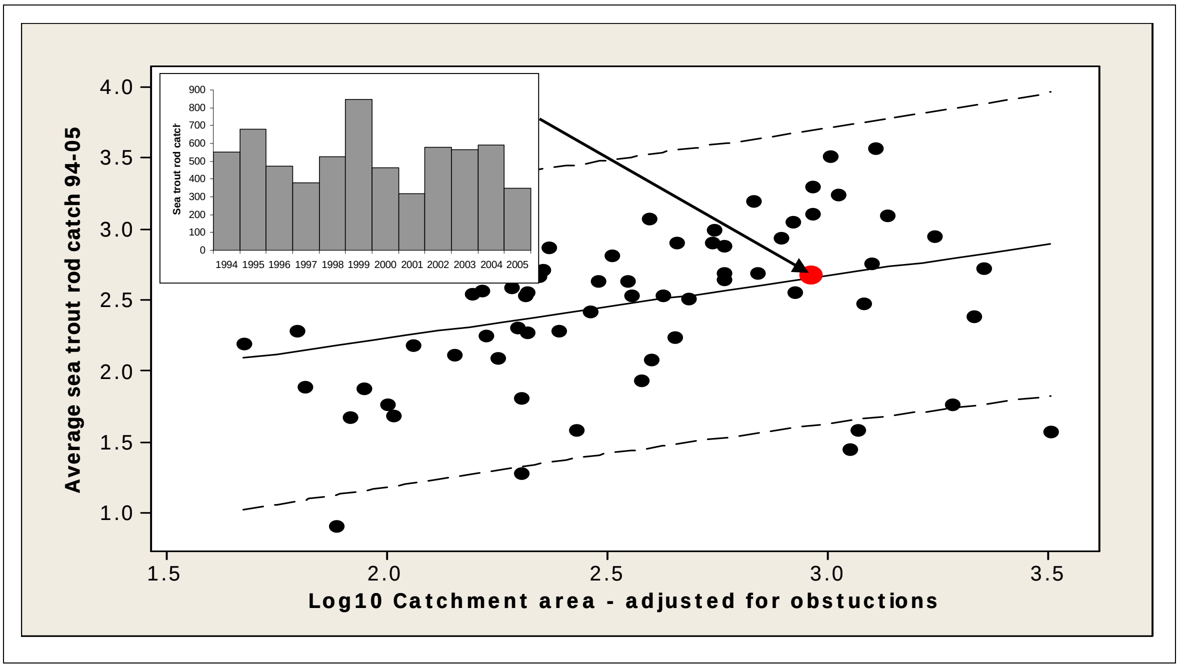

Figure 6.9 National relationship between catchment area and sea trout rod catch; River Tamar highlighted in red.

Figure 6.9 shows the relationship between catchment size (adjusted for obstructions) and declared sea trout rod catch for all sea trout rivers nationally. The position of the River Tamar in this relationship is highlighted. The Tamar is clearly performing in line with national expectation, and shows no decreasing trend, which would suggest that there is no cause for concern.

Trends in rod catch on the River Tamar were compared against various rod catch data sets at a range of spatial levels (national, regional and local) in order to confirm the above conclusion. Regional and local levels are represented by data from the South West region, and a river adjacent to the Tamar respectively.

The technique used to compare trends is based on one described by Hendry *et al.* (2007) whereby logged annual figures in the selected comparison set are deducted from the logged annual figures for the same year in the Tamar set. The technique provides an indication of the relative performance of the Tamar rod fishery, by smoothing out fluctuations between years and facilitating statistical analysis of trends.

These analyses reveal that between 1974 and 2005, rod catch in the Tamar performed better in relative terms than on the adjacent River Taw (Figure 6.10). In contrast, the Tamar rod catch appeared to perform less well compared to the national rod catch over the same period, although this trend was not found to be significant (Figure 6.11). It is possible that this relationship is very much driven by the rapidly improving performance of the sea trout fisheries on recovering rivers such as the Tyne. No relative trends were evident in relation to average regional catch or in more recent years (1994-2005).

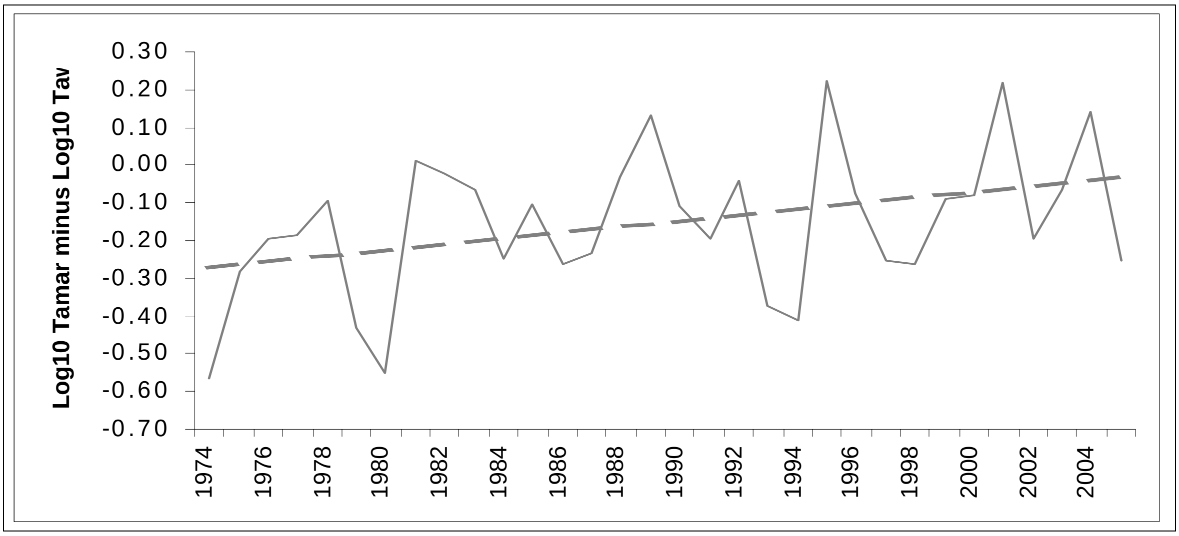

Figure 6.10 Comparison of sea trout rod catches from 1974 to 2005 – River Tamar versus River Taw (P=0.033)

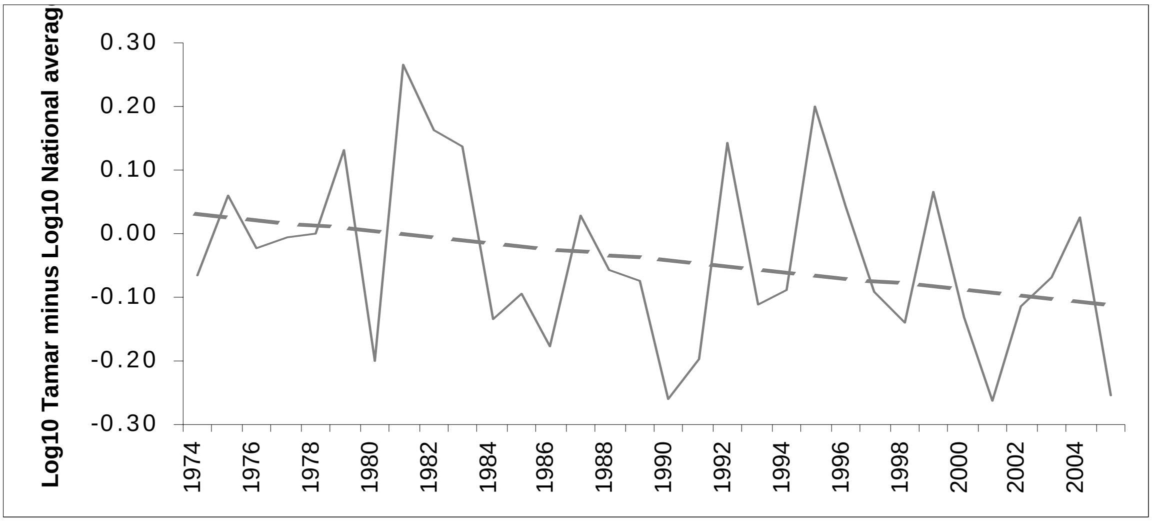

Figure 6.11 Comparison of sea trout rod catches from 1974 to 2005 – River Tamar versus national average catch (P=0.085)

In conclusion, the assessment of the Tamar sea trout stock based on catch data appears to show that the stock is performing as expected from a national viewpoint.

6.3 Assessment based on stock-recruitment reference points

Chapter 5 examined three published stock-recruitment relationships for sea trout, and also examined available data from the River Dee. It is clear that such relationships cannot currently be transported between rivers, and it is therefore not possible at this stage to derive an egg-deposition based reference point for the River Tamar. It is nevertheless interesting to consider the current level of egg deposition for the Tamar and to look at how this relates to reference points derived for these other rivers.

Egg deposition may be estimated from either rod catches or trap data as long as estimates of various parameters such as rod catch declaration rate, exploitation rate, post rod-fishery survival, proportion of maturing whitling, sex ratios and fecundity are available. Table 6.5 provides a summary of the parameters used in estimating annual egg deposition rates for the Tamar, and their derivation.

Table 6.5 Parameters used to estimate annual sea trout egg deposition in Tamar

Parameter	Value	Derivation
Reported sea trout rod catch	Varies annually	National rod licence returns
Proportion of catch declared	0.9	Environment Agency SAP guidelines
Proportion .0+SW fish in undeclared catch	0.5	Default value used in absence of better information
Proportion >.0+SW fish in undeclared catch	0.5	Default value used in absence of better information
Exploitation rate	Varies annually	Taken from Shields *et al.* 2006; based on Gunnislake trap data
Proportion maturing whitling	0.82	Based on data from the River Dee
Post rod fishery survival	0.9	Average figure based on radio tracking on Tywi (sea trout) and Dee (salmon)
Proportion female .0+SW fish	0.7	Based on data from the River Dee
Proportion female >.0+SW fish	0.9	Based on data from the River Dee
Fecundity	Based on weight	Weight fecundity relationship from Harris (1970) on River Dyfi study of stripped fish

Estimated annual egg deposition, split into eggs contributed by .0+SW and >.0+SW fish, is given in Table 6.6, and the results are displayed in Figure 6.12. Whilst whitling tend to dominate rod catches on the Tamar, the majority of eggs are actually typically contributed by >.0+SW fish.

The total estimated egg deposition per 100 m^2 of accessible area for the Tamar may be compared with the S_{MR} egg deposition targets derived for the Black Brows Beck, Bresle and Dee systems – that is to say the spawning stock giving rise to maximum recruitment. No S_{MR} target was derived for the Burrishoole since a Beverton model provided the best fit for these data. The average estimated egg deposition per 100 m^2 for the Tamar (1994-2005) was 1,025 eggs/100 m^2. This compares to S_{MR} target of 714 eggs/100 m^2 for the Black Brows Beck and 875 eggs/100 m^2 for the Bresle.

Table 6.6 Estimated egg deposition by sea trout on the River Tamar 1994-2005

Year	0SW fish egg deposition	>0SW fish egg deposition	Total egg deposition	Total egg deposition per 100 m^2 accessible area
1994	16,439,911	20,459,669	36,899,580	1261.24
1995	11,381,543	13,487,297	24,868,840	850.03
1996	9,598,565	18,364,070	27,962,636	955.77
1997	8,312,453	17,109,392	25,421,846	868.93
1998	11,468,970	15,711,860	27,180,831	929.05
1999	19,337,988	19,299,173	38,637,160	1320.63
2000	7,356,342	19,448,635	26,804,976	916.20
2001	10,066,325	16,810,096	26,876,420	918.65
2002	12,683,342	22,424,914	35,108,255	1200.01
2003	12,541,387	24,871,247	37,412,634	1278.78
2004	8,946,290	22,629,269	31,575,559	1079.26
2005	7,181,618	14,109,590	21,291,208	727.74

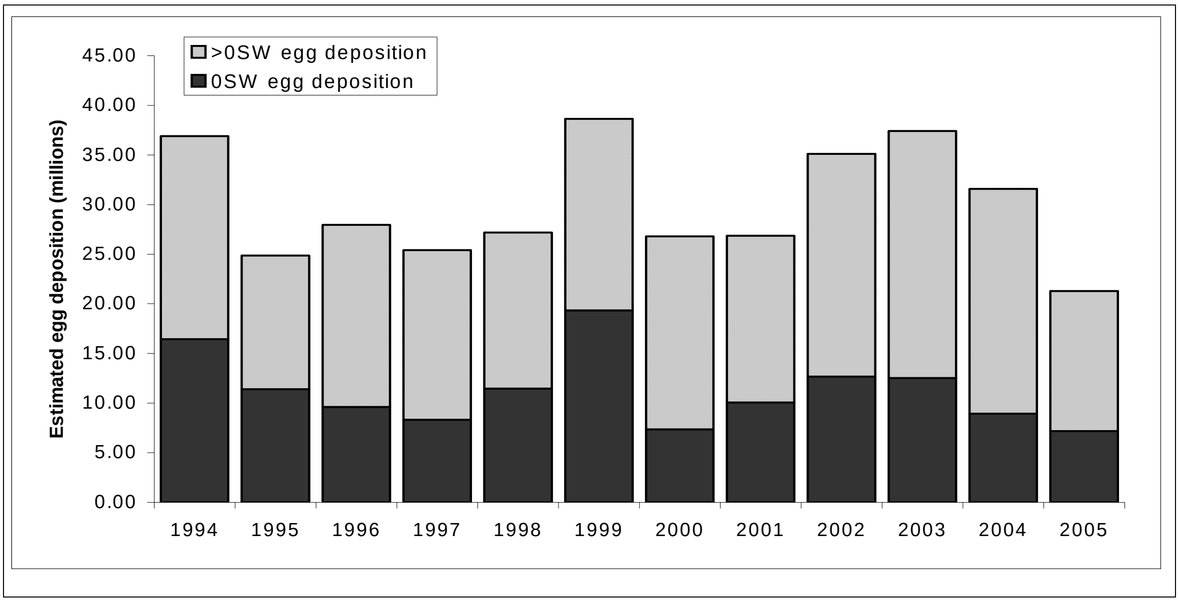

Figure 6.12 Estimated egg deposition by sea trout on the River Tamar 1994-2005

6.4 Summary of outcomes and management implications

Analysis of the available historic fish population data for the River Tamar allows an appraisal of the components that may be used to assess stock size, and stock performance in relation to possible biological reference points.

The assessment based on juvenile data suggests that juvenile trout populations at more than half of the sites surveyed fail to reach even 50 per cent of predicted carrying capacity. Certain age classes of trout were found to be entirely absent at a relatively large number of sites, despite all sites apparently offering some suitable habitat. Estimates of carrying capacity for different age classes of trout suggest that the quality of fry habitat is generally good throughout the catchment, whilst available parr habitat is of lower quality.

Overall, these results indicate that there may be issues with both the quality of the available habitat (particularly parr habitat) and underutilisation of that habitat (particularly for fry), suggesting that the system is underperforming in terms of its juvenile trout stocks. However, these results are based on juvenile data for just one year (1994). Also, it is possible that the assumptions of HabScore are not valid, and recruitment and water quality are limiting the juvenile population. Indeed, it is recognised that water quality is poor in the upper Tamar catchment (Hendry *et al.* 2006), although it is not clear if it is actually impacting on the fish populations.

Trends in historical juvenile trout data are difficult to assess since different numbers of sites have been surveyed in different years. However, the general indication is that there has been an overall increase in mean fry densities and an overall decrease in mean parr densities recorded over the period 1970 to present.

Conversely, sea trout catch statistics suggest that River Tamar sea trout stocks are performing in line with national expectation (based on the expected sea trout rod catch for the catchment area). Rod catches have remained reasonably stable over the period 1974-2005, suggesting there is no cause for concern over adult stocks. Estimated egg deposition figures suggest that egg deposition by sea trout has remained relatively stable over the period 1994-2005, with an average of 30 million eggs deposited each year, equating to approximately 1,025 eggs per 100 m^2 of available habitat.

It is, however, worth bearing in mind the potential weaknesses of each approach.

The juvenile assessment, for instance, is dependent on the performance of the HabScore models in predicting carrying capacity at sites within the Tamar catchment. Many of the potential problems with the HabScore models are discussed in detail in Section 3.2.2. Area staff have raised concerns over the performance of the HabScore models in predicting juvenile salmon densities in the River Tamar (Simon Toms, personal communication), suggesting that HabScore's expected density estimates tend to considerably underestimate the actual potential of the sites. If this is the case, then it may be equally likely that HabScore provides poor estimates of carrying capacity for juvenile trout. The juvenile assessment may actually be worse than is indicated here.

Catch-based targets are likely to be influenced by a number of factors affecting both the accuracy of catch statistics and how representative rod catches are likely to be of stocks. These issues are covered in more detail in Section 4.3. The catchment area/rod catch relationships which have been used to determine the performance of Tamar sea trout stocks nationally can only account for a proportion of the overall variance in rod catches. There are clearly a number of other factors influencing the variability of rod catches which have not been captured here; it is likely that we simply don't know what these factors are or if they are indeed measurable.

Whilst it has been possible to estimate the number of eggs deposited each year by sea trout in the River Tamar, many of the variables used in calculating these estimates are themselves estimates with varying degrees of uncertainty.

Finally, whilst the juvenile-based approach will take into account both anadromous and non-anadromous stock components (indistinguishable from one another at this stage), assessments based on rod catches or estimates of egg deposition will only take into account the contribution of anadromous sea trout. This may not present a problem in catchments where the majority of egg deposition is contributed by migratory trout, but a large number of other catchments, the Tamar included, support a significant brown trout population.

It is apparent that each of the approaches outlined has its weaknesses when used as a single interpretation of the Tamar trout stocks. Conflicting views on the status of Tamar trout stocks may be obtained using these different assessment methods. Resolution of such conflicts in the data will be essential in ensuring that a reliable indicator of the performance of stocks is established. More significantly, it is likely that reconciling these different methods of stock assessment into a single "lifecycle model" approach will be more robust and informative.

7 Conclusions and recommendations

7.1 Conclusions

This report has explored the feasibility of setting biological reference points (BRPs) for brown trout (*Salmo trutta* L.), highlighting the main difficulties involved in terms of the biology of the species, monitoring issues and current management regimes.

7.1.1 Biological issues

The principal biological issue stems from the species having two broad life-history types – the anadromous sea trout and the freshwater resident trout. In fact, many authors would argue that life-history tactics cannot be classified solely as anadromous or freshwater resident, but rather as a continuum of life-history tactics in space and time. This idea is based on the principle that migration is likely to be a trade-off of costs and benefits of the environment, regardless of the distance and environment travelled (Cucherousset *et al.* 2005). Thus, Cucherousset *et al.* (2005) put forward the following argument: because (i) the brown trout is a highly polymorphic and ecologically variable species, (ii) life-history traits are phenotypically plastic in response to environment and genetic parameters, and (iii) aquatic ecosystems present a continuous gradient of physical conditions, brown trout can exhibit a continuum in time and space of life-history tactics to optimise individual fitness and population persistence. This results in considerable variability in life-history tactics among individuals and populations of trout, with a number of different migration patterns. Differences between the sexes occur, with males typically having a higher tendency to remain in the natal river, and females more likely to migrate to sea (Cucherousset *et al.* 2005).

Investigations carried out during the course of this project suggest that a single biological reference point for trout is unlikely to meet all proposed management objectives, primarily due to the complexity of the trout's lifecycle, but also because of the way in which monitoring data is currently collected.

A more suitable approach would likely involve a hierarchy of diagnostic stages, with different reference points defined for different life stages, perhaps taking a similar form to the existing Salmon Lifecycle Model that incorporates all forms of monitoring into one assessment model. A "Trout Lifecycle Model" would theoretically be possible, and would capitalise on the River Fisheries Habitat Inventory which was developed in support of the Salmon Lifecycle Model. However, more research is needed to make such an approach possible. Recommendations with respect to research needs are detailed in Section 7.2.1 below.

7.1.2 Monitoring issues

The data required to develop and use BRPs may not be matched by the current monitoring programmes. This project has highlighted a number of data shortfalls:

- Data on rod catches of freshwater resident trout are largely lacking due to the current lack of a national catch-return system for non-migratory trout.
- Rod effort data are currently only available as ‘rod licence’ days for salmon and sea trout combined.
- Age data collected during routine juvenile surveys are limited in some areas. Freshwater age structure of trout is likely to be a key factor in understanding the causes of anadromy and distribution of anadromy within individual catchments.

Recommendations for monitoring are detailed in Section 7.2.2 below.

7.1.3 Management issues

Certain management issues arise as a result of the different licensing, catch recording and other regulatory practices applied to sea trout and FR trout. For instance, regulatory controls for sea trout and freshwater resident trout can differ considerably, due mainly to differences in the locations and methods used by fisheries for the two forms. Whilst such an approach is inconsistent with the taxonomy of the brown trout, it does make sense in practical terms (Milner *et al.* 2006).

In setting BRPs for trout, it will be necessary to understand the contributions of both anadromous and FR forms to total trout production.

7.2 Recommendations

7.2.1 Research needs

- Establish an objective baseline of trout distribution (both FR trout and sea trout morphs) in England and Wales.
- Continue to promote collaborative research into the causes and relative incidence of anadromy in England and Wales.
- Further understand genetic diversity of trout stocks.
- Investigate the possibility of developing regional versions of the HabScore models based on region-specific reference sites. Note: this issue is already addressed by the RFHI models which assume that habitat quality varies from catchment to catchment, and that catchments that are geographically close to each other will be more similar than those that are further apart.
- Investigate historic time series of data for calibration sites used in the original HabScore models to determine whether these are truly representative of ‘pristine’ conditions.
- Investigate differences in mean smolt age around the country and the suggested trend towards increased production of younger smolts.
- Investigate feasibility of developing a model to provide annual estimates of exploitation for sea trout.

- Carry out further investigation of a wider set of river characteristics with the aim of defining a subset of variables accounting for the greatest between-rivers variance of sea trout catch.
- Carry out further analysis of Dee data. For example, examine the fit of a Beverton-Holt model.
- Review calibration methods for converting semi-quantitative data to quantitative data. Establish a consistent approach across England and Wales and consider the use of converted data within the HabScore model where population variance estimates are required.

7.2.2 Data collection

- Investigate feasibility of introducing a national licence return system for recording rod catches of FR trout.
- Investigate feasibility of collecting data on rod effort for salmon and sea trout individually.
- Improve quality of age data collected during routine juvenile fisheries surveys.

7.2.3 Policy/management decisions

- Establish what level of risk is acceptable to fisheries managers in setting biological reference points for trout.

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Appendix I

Smolt age composition of stocks (from Solomon 1995 and Harris 2002)

River	Years	Mean smolt age	Sample size	% smolt age composition				Source
				S1	S2	S3	S4	
Tweed (A)	~ 1930	2.22	3197	0.1	79.1	19.6	1.2	Nall 1930b
Coquet (A)	~ 1930	2.11	46	0.0	89.0	11.0	0.0	Nall 1930a
Coquet (A)	1996-98	2.13	238	1.3	84.0	14.7	0	Harris 2002
Tyne (S)	~ 1925	2.14		0.0	86.0	14.0	0.0	Pentelow *et al* 1993
Wear (A)	1992-93	2.20	223	12.6	60.1	22.4	4.9	IC Russell (pers comm)
Wear (A)	1996-98	2.05	231	2.6	89.6	7.8	0.0	Harris 2002
Tees (S)	1930-31	2.26	453	0.0	74.2	25.8	0.0	Pentelow *et al* 1993
Yorks Esk (A)	1980-85	2.09	793	1.4	87.9	10.7	0.0	Dr S Axford (pers comm)
Yorks Esk (A)	1993	2.06	1251	4.8	84.4	10.8	0.0	Dr S Axford (pers comm)
Ouse (S)	1951-86	2.35	92	0.0	66.3	32.6	1.1	Fetter, undated
Beaulieu (S)	~ 1930	2.47	100	0.0	57.0	39.0	4.0	Nall 1930a
Teign (A)	1996-98	2.38	556	0.0	62.4	37.6	0.0	Harris 2002
Axe (S)	1964	2.21	1199	0.4	78.3	21.3	0.0	Allan *et al* 1965
Tamar (A)	1989-90	2.32	216	1.4	64.8	33.8	0.0	Dr K Broad (pers comm.)
Tamar (A)	1996-98	2.25	373	0.0	75.9	23.6	0.5	Harris, 2002
Fowey (A)	1979-81	2.79	712	0.0	25.0	71.5	3.5	Sambrook undated
Camel (A)	1996-98	2.27	584	1.0	71.6	26.7	0.7	Harris 2002
Taw (A)	1996-98	2.05	191	1.6	92.2	6.2	0.0	Harris 2002
Ogmore (A)	1982	2.10	39	0.0	83.0	17.0	0.0	Welsh WA 1982
Ogmore (A)	1992	1.98	83	2.4	97.6	0.0	0.0	A Winstone (pers comm.)
Afan (A)	1980-82	2.18	55	0.0	81.8	18.2	0.0	A Winstone (pers comm.)
Tawe (A)	1991-92	2.02	554	6.1	86.1	7.8	0.0	D Mee (pers comm)
Loughor (A)	1982-83	2.08	168	0.0	91.7	8.3	0.0	Welsh WA
Tywi (A)	1967-69	2.31	347	0.3	68.6	31.1	0.0	Harris 1970
Tywi (A)	1988-92	2.24	1847	2.9	70.6	26.1	0.4	D Evans (pers comm.)
Tywi (A)	1996-98	2.05	528	4.6	85.5	9.7	0.2	Harris 2002
Teifi (A)	1967-69	2.29	252	0.8	69.4	29.4	0.4	Harris 1970
Teifi (A)	1996-98	2.06	290	1.0	92.1	6.9	0.0	Harris 2002
Rheidol (A)	1967-69	2.03	200	3.5	90.5	6.0	0.0	Harris 1970
Dyfi (A)	1915-32	2.30	651	0.2	70.8	28.1	0.9	Nall 1933
Dyfi (A)	1967-69	2.37	1523	0.4	62.5	36.4	0.7	Harris 1970
Dyfi (S)	1968-69	2.42	911	0.8	58.0	40.3	1.0	Harris 1970
Dyfi (A)	1996-98	1.99	612	6.1	88.5	5.4	0.0	Harris 2002
Dysinni (A)	1967-69	2.12	211	4.0	80.5	15.0	0.5	Harris 1970

River	Years	Mean smolt age	Sample size	% smolt age composition				Source
				S1	S2	S3	S4	
Dysinni (S)	1968-69	2.18	142	0.0	81.7	18.3	0.0	Harris 1970
Eden (Mawddach) (S)	1987	1.89	609	11.0	89.0	0.0	0.0	Bareham 1987
Glaslyn (A)	1976	2.00	83	9.6	80.7	9.6	0.0	Thomas 1976
Dwyfor (A)	1996-98	2.01	217	3.2	92.6	4.1	0.0	Harris 2002
Gwyrfai (A)	1975-77	2.02	69	1.4	95.7	2.9	0.0	Brassington 1979
Conwy (A)	1976-91	2.06	286	2.5	88.6	8.9	0.0	M Scott (pers comm)
Clwyd (A)	1996-98	2.01	167	4.2	91.0	4.8	0.0	Harris 2002
Dee (A)	1991-93	2.12	286	1.2	85.4	13.3	0.1	I Davidson (pers comm.)
Dee (A)	1996-98	2.06	1340	2.6	88.8	8.4	0.2	Harris 2002
Ribble (A)	1935	2.26	173	1.1	71.7	27.2	0.0	Nall 1938
Ribble (A)	1996-98	2.09	313	0.3	90.4	9.3	0.0	Harris 2002
Wyre (A)	1935	2.45	107	1.9	57.9	33.7	6.5	Nall 1938
Lune A)	1935	2.24	222	0.0	77.0	22.1	0.9	Nall 1938
Lune (A)	1993	2.14	459	2.6	81.5	15.5	0.4	D McCubbing (pers comm.)
Lune (A)	1996-98	2.10	233	0.9	88.4	10.3	0.4	Harris 2002
Kent (A)	1935	2.36	194	3.1	60.3	34.5	2.1	Nall 1938
Kent (A)	1996-98	2.09	196	2.6	86.2	11.2	0.0	Harris 2002
Leven (A)	1933-34	2.32	742	1.1	67.0	30.8	1.1	Nall & Fell 1935
Duddon (A)	1934	2.31	169	0.0	69.8	29.6	0.6	Nall & Fell 1935
Border Esk (A)	1930-31	2.13	1482	1.2	84.2	14.6	0.0	Nall 1932
Border Esk (A)	1996-98	2.05	509	3.9	87.5	8.6	0.0	Harris 2002

(S) data from smolt scale readings
(A) data from adult scale readings

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