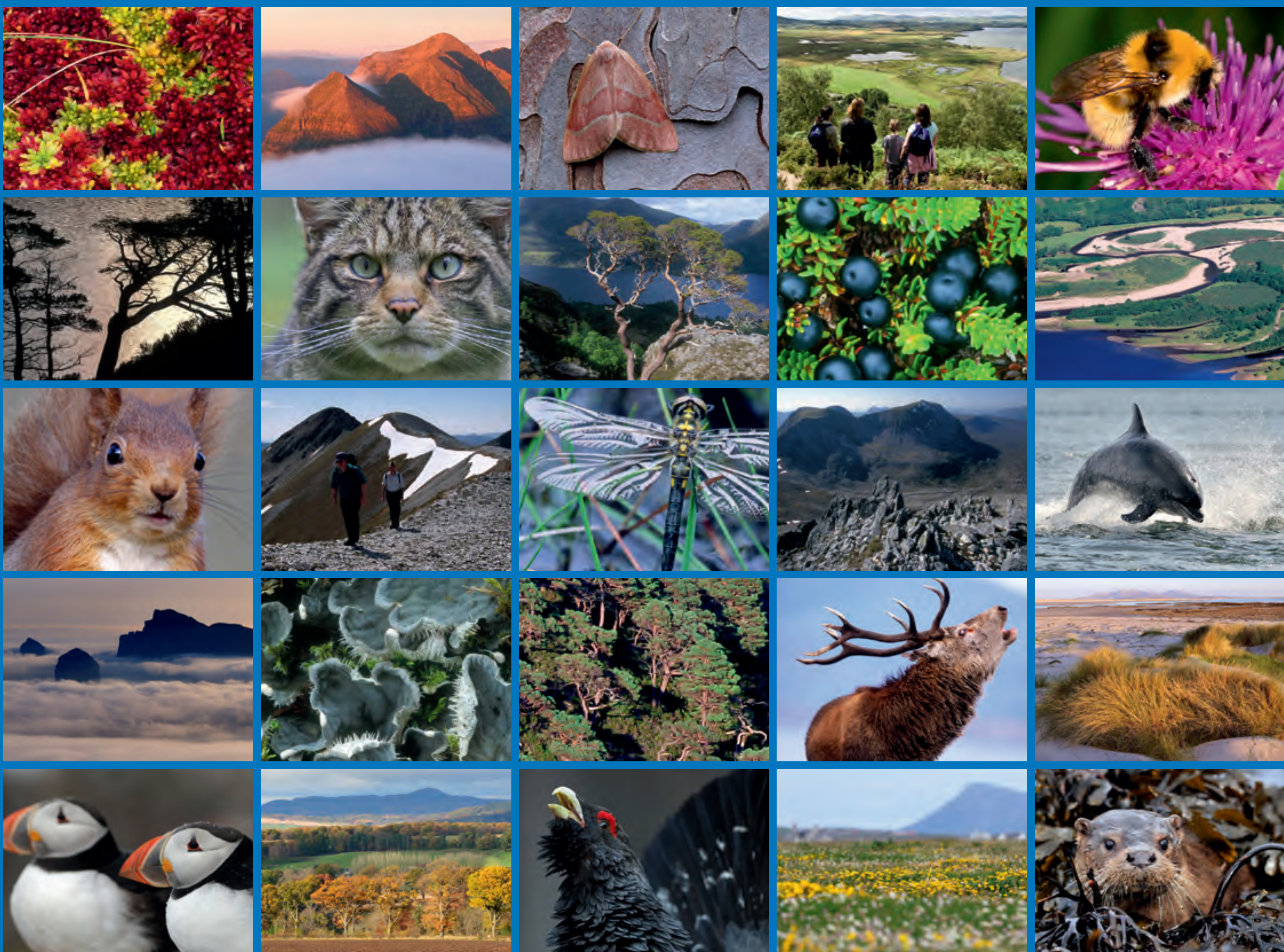
Scottish Natural Heritage
Commissioned Report No. 513

Guidelines for stocking of fish within designated natural heritage sites



Scottish Natural Heritage
Dualchas Nàdair na h-Alba
All of nature for all of Scotland
Nàdar air fad airson Alba air fad

COMMISSIONED REPORT

Commissioned Report No. 513

Guidelines for stocking of fish within designated natural heritage sites

For further information on this report please contact:

Julia Stubbs Partridge
Scottish Natural Heritage
Great Glen House
INVERNESS
IV3 8NW
Telephone: 01463 725000
E-mail:
Julia.Stubbs.Partridge@snh.gov.uk

This report should be quoted as:

Cowx I.G., Nunn A.D., Harvey J.P. & Noble R.A.A. (2012). Guidelines for stocking of fish within designated natural heritage sites. *Scottish Natural Heritage Commissioned Report No. 513.*

COMMISSIONED REPORT

Summary

Guidelines for stocking of fish within designated natural heritage sites

Commissioned Report No. 513 (iBids No. 10969)
Contractor: Cowx I.G., Nunn A.D., Harvey J.P. & Noble R.A.A (University of Hull International Fisheries Institute)
Year of publication: 2012

Background

Stocking of fish is a common practice throughout the UK, aimed at supporting wild populations and/or enhancing stocks for angler activities. There are concerns, however, that stocking may cause undue risks to the ecological functioning of water bodies, potentially leading to a loss of biodiversity and altered ecological status. Of particular concern are shifts in food web structure and trophic status that may occur following stocking of carnivorous species, and the impacts that this will have upon indigenous flora and fauna. This has serious implications for water bodies that are designated natural heritage sites. This report examines the potential impacts of stocking and introducing fish into open waters, with particular reference to designated natural heritage sites in Scotland. In addition, significant hazards are identified for which further research is required, and management guidelines to deal with potential problems are provided.

The aim of this project was to review the science that underpins guidelines already in place in the UK and Europe, with a view to developing guidance to safeguard designated natural heritage sites in Scotland. This project was also intended to contribute towards on-going discussions on 'A Code of Good Practice' for fishery managers and research elements of the Scottish Freshwater Fisheries Framework. The overall objective of the project was to undertake a hazard analysis and develop an advice support tool for Scottish Natural Heritage (SNH) to respond to consultations on fish stocking applications.

Main findings

- The potential risks associated with fish stock enhancement to native salmon, trout, charr, whitefish and lamprey populations, invertebrate and macrophyte species, and ecosystem functioning as a whole, are numerous. The main hazards are predation by stocked or introduced fishes, competition with stocked or introduced fishes, increased prevalance of disease and spread of non-native organisms, disruption of the native gene pool and eutrophication.
- Stocked or introduced fishes pose a direct threat through predation. Stocking of trout has been implicated in the decline or disappearance of many native fish species through direct predation and increased densities of stocked fishes may encourage larger numbers of piscivorous birds feeding on the stocked and wild fishes.
- Predation by stocked or introduced fishes may also have an influence on both nutrient budgets and ecosystem functioning in general. It can potentially raise nutrient levels (especially phosphorus) due to increased standing crop of fishes and by virtue of the

higher numbers of fishes feeding upon terrestrial organisms. Stocking benthic feeders such as carp can increase turbidity and mobilise nutrients and cause eutrophication.

- Predation on native fishes or preferential feeding of stocked fish on certain taxa can result in a switch in trophic states through disruption of food chains/webs, e.g. by grazing pressure on zooplankton.
- Stocked or introduced fishes pose a direct risk to native salmon, trout, charr and whitefish populations through a variety of competive interactions. This can be through displacement of native fishes through aggressive behaviour, competition for food resources and habitat with stocked fish resulting in reduced growth, survival, reproductive potential or energetic performance of native fishes. Overstocking can also lead to reduction in fishery performance through competitive bottlenecks.
- Stocking may result in genetic drift and dilution of gene pool, loss of genetic diversity and hybridisation. This is particularly pertinent where native stocks exhibit adaptation towards particular environments and stocking could lead to loss of fitness manifest as differences in growth potential, age at maturity, fecundity, and can have implications for coevolution and adaptation processes.
- Stocked fish also carry a risk of increased disease prevalence through tramission of pathogens. Care must be taken to ensure that other non-native, non target organisms are not introduced as part of the stock enhancement programme - this includes due diligence when exchanging water during transportation and when the fish are stocked into the receiving water body.

Recommendations

- It is recommended that where designated species are restricted to a small number of sites, introducing and, to a lesser extent, stocking fish, including trout, should be prohibited or restricted to safeguard populations.
- Where it is possible to remove or minimise the causes of declines in fisheries, habitat improvement is the most desirable option because it should lead to long-term sustainable improvements with minimal deleterious ecological impacts.
- It is recommended that generic codes of practice (as provided in Appendix 1) are followed meticulously, to ensure the risks associated with stocking or introducing fish or other aquatic organisms are evaluated and the correct decisions made to native biota.
- Much of the current information about stock enhancement remains fragmented and poorly documented, and fully elucidating the direct impacts of fish stock enhancement on designated natural heritage sites remains a matter of judgement. To improve this scenario, further research is required, including:
    - What is the relative frequency of piscivory and zooplanktivory in stocked or introduced trout, and the impacts that these feeding strategies (and zooplanktivory by coarse fishes) have upon nutrient cycling, trophic cascade and ecosystem functioning?
    - What are the relative risks of stocked or introduced fish competing with wild fish, and affecting the viability of wild populations as well as the indirect risks to invertebrates, nutrient status and ecosystem functioning?
    - Regarding ecosystem functioning, the key area that requires further research is the impacts of gradual, chronic increases in nutrient availability on the relative productivity of macrophytes and phytoplankton, together with concurrent shifts in invertebrate community structure and biodiversity in general. This area of research is required to address the possibility of alterations in ecosystem functioning via competition-induced shifts in food-web structure caused by stock enhancement activities.

For further information on this project contact:
Julia Stubbs Partridge
Scottish Natural Heritage, Great Glen House, INVERNESS IV3 8NW
Tel: 01463 725000

For further information on the SNH Research & Technical Support Programme contact:
DSU (Policy & Advice Directorate), Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.
Tel: 01463 725000 or research@snh.gov.uk

Table of Contents **Page**

1. BACKGROUND 1

2. METHODOLOGY 4
2.1 Definitions 4
2.1.1 Objectives of stock enhancement 4
2.2 Hazard identification and assessment 7
2.3 Preparation of guidelines and assessment of knowledge gaps 8

3. IDENTIFICATION AND ASSESSMENT OF HAZARDS AND RISKS RELATED TO THE STOCKING AND INTRODUCTION OF FRESHWATER FISHES TO DESIGNATED NATURAL HERITAGE SITES 9
3.1 Changes to ecosystem functioning 9
3.2 Impacts of nutrient import 13
3.3 Risks to native salmon, trout, charr, whitefish and lamprey populations 14
3.3.1 Predation (direct impacts of stocking) 15
3.3.2 Predation (indirect impacts of stocking) 17
3.3.3 Competition 18
3.3.4 Spawning and post-spawning recovery and survival 20
3.3.5 Genetic impacts 21
3.3.6 Parasites and diseases 26
3.3.7 Spread of non-native organisms with stocked fishes 28
3.3.8 Fishing pressure and mortality 29
3.4 Threats to protected invertebrates and macrophytes in designated sites 30
3.4.1 Invertebrates 30
3.4.2 Macrophytes 31
3.5 Case studies 32
3.5.1 Loch Lomond 33
3.5.2 Loch Leven 34
3.5.3 Lake of Menteith 34
3.5.4 Butterstone Loch 35
3.5.5 Lindores Loch 35
3.5.6 Esthwaite Water 36
3.5.7 Chew Valley Lake 36

4. GUIDANCE FOR STOCKING AND INTRODUCING FISHES TO DESIGNATED NATURAL HERITAGE SITES 38
4.1 Strategy and management of fish stock enhancement 38
4.2 Protocol for stocking of fish within designated natural heritage sites 40
4.2.1 Context 40
4.2.2 Legislative context 40
4.2.3 Proposed decision framework for stocking of fish within designated natural heritage sites 41

5. CONCLUSIONS AND RECOMMENDATIONS 47
5.1 Stock enhancement programmes 47
5.2 Priority hazards and future R&D 48
5.2.1 Predation 49
5.2.2 Competition 50

5.2.3 Eutrophication 51

6 REFERENCES 53

APPENDIX 1. GENERIC PROTOCOL FOR ASSESSMENT OF STOCK ENHANCEMENT PROGRAMMES 88

APPENDIX 2. STOCKING DENSITIES/BIOMASSES REPORTED FOR A NUMBER OF FISH SPECIES AND LIFE STAGES IN VARIOUS WATERBODIES. 117

1. Background

The stocking, translocation and introduction of fishes are frequently used by fisheries owners, managers and scientists throughout the world in an attempt to improve the quantity or quality of catches and provide long-term beneficial effects on fish stocks (Cowx, 1994a, b 1998a; Welcomme, 1988). Many thousands of stocking events, involving millions of individual fishes, take place annually in managed fisheries (Hickley, 1994). In some cases, stocking programmes may be justified; for example, to compensate for losses caused by pollution (Aprahamian *et al*., 2003). Recently, however, there have been concerns about the potential risks associated with stocking and introducing fishes, particularly with respect to ecosystem functioning, changes in community structure and losses of genetic integrity (Cowx, 1997; McGinnity *et al*., 1997, 2003; Cowx & Gerdeaux, 2004; Casal, 2006; Eby *et al*., 2006). Of particular concern are shifts in food-web structure and trophic status that may occur, and the impacts that these could have on indigenous flora and fauna. In addition, stocking or introductions may lead to competition with or predation on indigenous biota (Hickley & Chare, 2004; van Zyll de Jong *et al*., 2004): this can have serious implications for water bodies that are part of designated sites or support protected plant or animal species.

There is, therefore, a need for fisheries managers to be aware of the possible impacts of stock enhancement programmes, both in terms of the effects on ecosystem functioning and the likelihood of improvements in stocks. Unfortunately, information on the impacts of stock enhancement programmes is sparse, largely because of a lack of systematic monitoring and dissemination of information on the outcomes. Furthermore, although large sums of money have been invested in stocking activities, relatively few programmes have been properly evaluated and there is little evidence to suggest that stock enhancement exercises lead to tangible long-term benefits (Cowx, 1998a; Arlinghaus *et al*., 2002). Weaknesses in success of many programmes appear to result from indiscriminate stocking without well-defined objectives or prior appraisal of the likelihood of success. Notwithstanding, if stocking programmes are designed to achieve defined objectives and to be implemented following best-practice guidance, it should be possible to improve success rates and minimise or mitigate any detrimental effects. It should also be possible to identify situations when, because of risks to the wider ecosystem, it is inappropriate to undertake stock enhancement programmes. In the industrialised world, the most successful stock enhancement programmes have generally been associated with put-and-take and intensively stocked stillwater fisheries; stocking river fisheries has been less successful except perhaps where stocking has been used to establish populations or accelerate recovery (e.g. Bolland *et al*., 2009). The most successful enhancement programmes in developing countries have mostly been associated with reservoir fisheries that have been heavily stocked to increase yield.

Legislation came into force on 1 August 2008 that regulates the introduction (i.e. stocking) of all species of freshwater fish, native and non-native, within Scotland. Specifically, Section 35 of the Aquaculture and Fisheries (Scotland) Act (2007) inserted a new section, 33A, into the Salmon and Freshwater Fisheries (Consolidation) (Scotland) Act (2003). This makes it an offence for any person to intentionally introduce any live fish or the spawn of any fish into inland waters, or possess such with the intention of introduction without previous written agreement from the appropriate authority. The principal aim of the new provisions was to protect native biodiversity from the potential consequences of introducing non-native fishes into Scottish fresh waters. The provisions apply to all freshwater species, including Atlantic salmon (*Salmo salar* L.), sea trout (*Salmo trutta* L.), and coarse (i.e. non-salmonid) fishes.

The provisions are implemented through the relevant District Salmon Fishery Board (DSFB) when the fish to be introduced are Atlantic salmon or sea trout (*Salmo trutta* L.), and through Marine Scotland Science (MSS) where a DSFB does not operate or where the fish being introduced are not Atlantic salmon or sea trout (e.g. brown trout (*S. trutta*) from hatcheries or coarse fishes to still waters). Any licence application for an introduction that may adversely affect a Site of Special Scientific Interest (SSSI), Special Protection Area (SPA) or Special Area of Conservation (SAC) requires full consultation with Scottish Natural Heritage (SNH).

In this context, SNH has no specific policy in place for the introduction of freshwater fish to inland waters. However, in their response to Marine Scotland consultation document entitled: *A Strategic Framework for Scottish Freshwater Fisheries*, SNH outlined its broad objectives for supporting the conservation of wild fish and their habitats, along with the highest standards in their management, operation and promotion of the associated fisheries. These objectives are:

- To maintain self-sustaining populations of native fish, within their natural distribution, and to protect their natural genetic diversity by safeguarding local populations;
- To maintain fisheries which operate in a manner consistent with natural heritage objectives, and which accommodate other recreational users of aquatic ecosystems;
- To manage fish stocks on an individual catchment basis, working as far as possible with natural processes;
- To secure the survival of Scottish wild fish populations and communities of particular natural heritage importance by protecting key water bodies;
- To re-establish populations of native fish species in certain key areas where severe declines or extinctions have occurred;
- To prevent introduction or translocation to the wild of fish of species or types not native to that water body, except where the water body is isolated from the natural heritage network and not of natural heritage importance;
- To encourage anglers, fishery managers and owners, and those using water for other recreational purposes, to recognise each others' rights and responsibilities and to co-operate in developing national and local strategies for protection and management;
- To recognise the relationship between fish, fisheries and the ecosystems upon which they depend, and promote an ecosystem-based approach to fish and fishery management; and
- To promote the development of evidence-based stocking practices. This is particularly important given that recent evidence suggests that many current stocking practices can be damaging to native fish stocks.

SNH's primary concerns relating to the introduction of freshwater fish are:

a) The risk of modifying the *genetic diversity* that exists within wild populations, in a way that will make *local populations* less suited to their host environment in the long-term through cross breeding (*introgression*);
b) The potential to increase competition between introduced and wild fish populations if the numbers of fish added to the waterbody exceed its *carrying capacity*;
c) The potential to Increase the risk of predation by introduced fish on either native fish or other aquatic biota of conservation value;
d) Possible impacts on water quality by, for example, the selective removal of zooplankton by introduced fish, the disturbance of river or loch sediments;
e) Risks to biosecurity and the spread of disease, parasites and invasive non-native species;
f) The possibility of increasing the rate of exploitation of natural populations of fish, by man and other predators.

Unfortunately, information on the impacts of stocking or introducing fishes into water bodies designated for conservation purposes is disparate and needs collating to support the development of guidelines to aid decisions on whether or not stockings should take place. **The aim of this project is to review the science that underpins guidelines already in place in the UK and Europe, with a view to developing guidance to safeguard sites**

designated for nature conservation in Scotland. This is also intended to contribute towards on-going discussions on A Code of Good Practice for fishery managers and research elements of the Scottish Freshwater Fisheries Framework. The overall objective of the project is to undertake a hazard analysis and develop a decision-support tool for SNH to respond to consultations on fish stocking applications that may impact upon designated natural heritage sites. The guidance is, whenever possible, based on robust science and considers:

- the types of consultation that SNH may receive related to the introduction or stocking of freshwater fishes to designated natural heritage sites, including:
  - proposals to introduce/stock brown trout and rainbow trout (*Oncorhynchus mykiss* (Walbaum)) into stillwaters;
  - proposals to introduce/stock brown trout into running waters; and
  - proposals to introduce/stock coarse fishes into stillwaters.

- natural heritage issues related to fish introductions, at generic and specific scales, including:
  - still waters designated as SSSIs or SACs for their habitats, including macrophyte assemblages; and
  - running waters designated as SSSIs or SACs for interests including habitat, Atlantic salmon and other fishes, freshwater pearl mussel and macrophytes.

- prevention/mitigation of adverse impacts in relation to the above.

2. METHODOLOGY

2.1 Definitions

Inland fisheries are underpinned by a complex interaction of physical, chemical and biological conditions which need to be regulated in such a manner as to enhance the fishery output and maintain quality fishing. There are three broad types of approaches to fisheries management (Blankenship & Leber, 1995; Cowx & Gerdeaux, 2004).

- *Traditional management control tools* commonly applied to regulate fishing including gear and size restrictions, seasonal closures, quotas and bag limits, and limitations on entry, taxes, levies and property rights.

- *Habitat rehabilitation tools* to increase or recover available habitat and/or access to key habitat for at least some life stages of a target species (see Cowx & Welcomme, 1998). Such an approach may range from increased connectivity along a river (fish passage facilities), through habitat restoration to the installation of artificial habitats (such as low weirs or groynes).

- *Stock enhancement* referred to as the manipulation of the fish stocks by addition of a particular species to improve the fishery productivity (catch rates) or diversity of the fishery. Stock enhancement is widely popular due to its perceived simplicity (Welcomme & Bartley, 1998a, b). This aspect of stock enhancement is the focus of the remainder of this report, but it is important to recognise that **stock enhancement is not the sole mechanism for improvement of stocks**, as indicated above, this will be considered further in Section 4.

2.1.1 Objectives of stock enhancement

Stock enhancement is a commonly used measure for management of inland fisheries in use today. Most countries apply stock enhancement measures to some degree, in recognition that more conventional approaches to management have failed to control fisheries exploitation. It is also used to respond to depleted natural fish populations as a result of habitat degradation, or simply just to increase the fish stocks. In this context, stocking is an attempt to fix a problem, either real or perceived, or simply to increase stock abundance. Depending on the problem, stocking can be considered to be either a permanent or temporary solution. It can generally be divided into six main categories that match the Environment Agency's (EA) '*Work Instructions on Fish Stocking and Removal and on Administering and Determining Fish Movement Consents*', although a number of terminologies are applied throughout the fisheries sector (Table 1; modified from Molony *et al.*, 2003).

Mitigation or compensation

This encompasses stocking with fish carried out as a voluntary exercise or statutory function to compensate for a disturbance caused by human activities against lost production. A loss in fish production may be attributed to a scheme or activity that cannot be prevented or removed, such as construction of reservoirs, power stations, barrages and impassable barriers, land drainage works or similar habitat perturbations. However, where stocked fish are released into unaffected parts of the river catchment or lake, the impact on the wild stocks in these areas must be considered.

Table 1. Range of terms used to describe stock enhancement activities (modified from Molony et al., 2003).

Term	Definition	Source(s)
Restocking		
Mitigation	Production and release of fish to restore stock to original levels	Radtke & Davis (2000)
Stock enhancement	Supplementing natural recruitment with injection of external material	Ziemann (2001)
Ocean ranching	Releasing of fish to the ocean to be subsequently commercially harvested	Arnason (2001)
Marine ranching	Production of early life-stages of species in a hatchery for eventual release into natural or modified habitats	Bartley (1999)
Stock enhancement recovery	Production and release of fish for inter-generational benefit	Harada & Matsumiya (1992)
Augmentation		
Augmentation	Production and release of fish to complement natural recruitment where available habitat is below carrying capacity	Cowx (1994b), Bartley (1999)
Habitat enhancement	Production and release of fish to (re)colonise new/artificial habitats	Young (1999)
Mitigation	Stocking of fish into new/modified habitat to compensate for a decrease in a fishery	Cowx (1994b), Bartley (1999)
Addition		
Community change	Production and release of exotic fish to create new fisheries	Cowx (1994b), Bartley (1999)
Addition	Stocking of a new species into an area outside of its natural range	Rowland (1994)
Enhance	Production and release of fish to create new fisheries	Petr (1998)
Other terms		
Stock enhancement	Production and release of fish for public good	Drawbridge (2002)
Sea ranching	Production and release of fish for common good	Drawbridge (2002)
Stock enhancement continuous	Production and release of fish for intra-generational benefit	Harada & Matsumiya (1992)
Enhance	Production and release of fish to increase stocks above original levels	Radtke & Davis (2000)

Many of the traditional, long-term stocking programmes are carried out for mitigation or compensation. In such cases, stocking is often viewed as a long term solution (i.e. it must be done on a continual, usually annual, basis) and is unlikely to lead to the establishment of a self-sustaining natural population because the underlying reason for the stocking has not been addressed. The degree to which the fishery is dependent on stocking depends on the extent of ecosystem modification and can range from 'total', where the native stock would disappear without support, to 'partial', where the stock would be reduced to a proportion of that which might be expected if the system was unimpacted.

Restoration

Stocking for restoration is carried out after a limiting factor on stock recovery or improvement has been removed or reduced. An example may be a long-term improvement in water quality, habitat improvements, the easing of passage for migratory fish or a reduction in fishing pressure. All restoration stocking must be based on reliable evidence that such populations existed in that catchment, or waterbody, in the past.

Restoration stocking should generally not take place until defined limiting factors have been removed or ameliorated. However, situations may exist where it is necessary to initiate stocking in parallel with other habitat or fisheries management actions. Used in parallel, this can accelerate the stock recovery and/or to secure continued support for the restoration. Stocking programmes of this type should be a temporary measure and require a more active management strategy for the aquatic ecosystem and its fish populations. The ultimate objective is to create a fish stock and aquatic ecosystem that is self-sustaining.

Enhancement

Enhancement stocking is the principal method used to maintain or improve stocks where production is actually, or perceived to be, less than the water body could potentially sustain. Often, the reasons for the poor stocks cannot be identified and/or removed, or there is a desire to increase populations (usually for exploitation) to levels greater than those that can be achieved naturally. Typically, this type of stocking is used where those exploiting the fishery have expressed dissatisfaction with the quality of fishing, or to enhance stocks in sections of river where access is restricted by in-stream barriers. It also includes activities carried out to strengthen the quality and quantity of the spawning stock of a given species so as to improve natural reproduction potential. This can be for improvement of yield from a fishery or for conservation purposes where the natural breeding component is considered inadequate to maintain the stock at sustainable levels (see below).

The majority of stocking activity within the UK probably falls into this category and it is, in the main, driven by concerns about the quality of angling experienced in a given location. However, in many cases the assessment of the state of the stock has been unduly pessimistic. This often results from a poor understanding of natural fluctuations in fish abundance or unrealistically high expectations as to what levels of natural production can actually be achieved. As natural production can often already be limited or driven by natural population cycles, or if fish are already resource limited for food or space, then it is unlikely that stocking will have a beneficial long-term effect.

When stocking for enhancement is considered a long term, on-going solution, it can be defined as ‘ranching’ (supplementing natural juvenile recruitment through the growth of stocked fish) or, in the case of sport fishing, ‘put-and-take’ (stocking of fish into a water body for the express purposes of catching and removing for consumption). As a permanent solution, the strategy requires continuous application to maintain the desired fishery. Enhancement stocking is particularly favoured where there is a desire to introduce a species to a fishery and the species is unable to sustain itself naturally. Typical of this in the UK are the stocking of rainbow trout.

Conservation

Many fish species are under considerable threat from extinction, and stocking can be used to maintain these species. This is generally confined to those fish species or populations that are considered rare or threatened, including salmon, coregonids (*Coregonus albula* (L.), *C. lavaretus* (L.)), Arctic charr (*Salvelinus alpinus* (L.)) and European eel (*Anguilla anguilla* (L.)). This can be similar to mitigation stocking, but is usually more preservationist in its intent. Conservation stocking best takes place in areas where the threat of extinction no longer exists. Conservation stocking is also used to enhance populations of other fauna that depend on fish stocks (e.g. otters and bitterns). Stocking for conservation purposes should ideally be conducted in accordance with IUCN Translocation Guidelines, which protect the fauna of any receiving waterbody. The guidelines used by UK Conservation Agencies, which based on the IUCN guidelines, are provided by the Joint Nature Conservation Committee (JNCC) (http://jncc.defra.gov.uk/pdf/species_policy.pdf).

Creation of new fisheries

Where introductions are made as a management tool for commercial and recreational fisheries, the aim is to introduce new fish species for one of the following reasons:

- *Establish new fisheries* that are more resistant to fishing pressure or have greater market value than fisheries for native species. In recreational fisheries, new species are introduced to improve the variety available to anglers, or to include a species of particular trophy or sporting value into an area. Stocking fish into a newly created water, e.g. a redundant gravel pit, also falls into this category.
- *Fill a vacant niche* where existing fish species do not fully utilise the trophic and spatial resources available. In some natural waters evolutionary isolation has resulted in there being few native species, as in the UK and Ireland where faunas have been eliminated through glaciation. More commonly, the desire for introductions arises as a consequence of human activities. For example, many new reservoirs lack native species capable of fully colonising lentic waters. It is important to note, however, that naturally fishless water bodies can have an important conservation value because of their diversity of non-fish species and foodwebs.

Accidental introductions (of non-native species) into natural waters may also be important, either through escapes from captivity (e.g. aquaculture facilities) or colonisation by wild populations of non-native species.

Scientific Investigations

Stocking can, on some occasions, be used as a tool to investigate fisheries management issues. Stocking can be a useful method of investigating the carrying capacity of the habitat, fish migration and behaviour.

2.2 Hazard identification and assessment

The first step was to review the literature and other materials related to interactions between stocked and wild fishes and other aquatic organisms, and the impacts of stocking and introductions on ecosystem functioning. Many of these characteristics were reviewed in two Environment Agency (EA) and two Natural England (NE) projects (Noble *et al*., 2004; Cowx *et al*., 2006, 2007; Nunn *et al*., 2006), but were updated to incorporate new developments and specifically to consider the impacts of stocking in designated natural heritage sites. The review used the following resources:

- electronic search engines such as Web of Science (WoS) and Aquatic Sciences & Fisheries Abstracts (ASFA) to search for literature published since 1970;
- electronic search engines such as JStor and Scopus to search for literature published prior to 1970;
- information, particularly in the grey literature, from the extensive network of experts involved in the European Inland Fisheries Advisory Commission (EIFAC) and counterparts and colleagues in Europe and North America;
- local operational investigations and national projects undertaken by government agencies that were pertinent to delivering the outputs of this project; and
- the extensive collection of material held by the authors as part of on-going activities associated with inland fisheries and fish conservation.

The information was collated and consolidated into a review that summarises pertinent information relevant to the project. The review specifically:

- investigated the links between fish stocking/introductions to various types of water bodies and possible changes to ecological and trophic status that may arise;
- assessed if stocking/introductions pose a threat to fish and invertebrate assemblages or species, or macrophyte communities, and which species might be at risk and under which circumstances;
- assessed the likely impacts from stocking/introducing fishes into waters containing native salmon, trout, charr, whitefish or lamprey populations;
- determined how stocking/introductions might cause impacts on fish species or habitats of high conservation value and water body trophic level through shifts in ecosystem functioning.

This information was used to assess the potential magnitude of the problems posed by stock enhancement for the protection of designated natural heritage sites, their biodiversity and ecosystem functioning. Economic and societal risks did not form part of this review, although the implications on these factors have been commented on within this report. In particular, the work for this review built on the EU "Environmental Impacts of Alien Species in Aquaculture" (IMPASSE) project that undertook a comprehensive review of the impacts of non-native fish species in aquaculture and developed full risk-assessment and decision-support frameworks to underpin EC Council Regulation 708/2007 concerning the "Use of alien and locally-absent species in aquaculture".

2.3 Preparation of guidelines and assessment of knowledge gaps

The first part of the assessment identifies the priority hazards and risks that arise from stock enhancement programmes. This is used as input to drafting potential guidelines for use by SNH in their decision-making framework for consideration of stocking proposals on designated natural heritage sites. The output of the first part of the assessment identified gaps in knowledge and risks that require further assessment or research and development.

3. IDENTIFICATION AND ASSESSMENT OF HAZARDS AND RISKS RELATED TO THE STOCKING AND INTRODUCTION OF FRESHWATER FISHES TO DESIGNATED NATURAL HERITAGE SITES

Stocking and introduction of fishes are widespread and have been undertaken for many decades. Despite this, the long-term ecological effects of these activities are not well understood, and the potential risks associated with stocking and introducing fishes into freshwater bodies are numerous (Table 2). Indeed, the Global Invasive Species Programme listed eight fish species, including two salmonids, among the 'World's Worst Invasive Alien Species' (Cambray, 2003). Evidence suggests that, where natural recruitment is not limiting, stocking can have negative effects on the growth and survival of resident fish populations (Nielson *et al*., 1957; Harcup *et al*., 1984; Berg & Jorgensen, 1991; Naslund, 1992). This review focuses on salmonids, since this is the group of fishes most widely and heavily stocked in Scotland (and elsewhere), although other species are included where information exists as there is a growing demand for stocking of coarse fish in Scottish waters. It should be recognised, however, that **without precise information of stocking activities, it is not possible to predict the likely impacts of particular management techniques on specific water bodies.** Indeed, *"The impact of alien invasive sport fish is for the most part unpredictable in time and space, with the introduction of relatively few species having resulted in many extirpations of indigenous fish species worldwide"* (Cambray, 2003, p. 217). Moreover, **the impacts of particular management techniques will be site-specific, due to the inherent differences in ecosystem dynamics between water bodies.** Thus, selected case studies are presented to demonstrate the specific management techniques most likely to impact upon designated natural heritage sites.

3.1 Changes to ecosystem functioning

Ecosystems function through the complex interactions of primary and secondary producers with each other and their physical and chemical environment. Stocking and introduction of fishes have the potential to alter the functioning of ecosystems through a number of mechanisms.

Contemporary theory suggests that stillwaters, especially shallow lakes, exist in one of two opposing stable states; either a clear-water state dominated by submerged aquatic macrophytes or a turbid-water state dominated by phytoplankton (Scheffer *et al*., 1993). Under certain circumstances, it is possible for a lake to switch from one state to the other. Whenever this happens, there is necessarily a mechanism that has upset the equilibrium of the ecosystem (see Moss *et al*., 1996). Potential mechanisms are numerous, but include fish. It has long been recognised, for example, that planktivorous fishes are a major factor influencing the species and size composition of zooplankton in freshwater ecosystems (e.g. Hrbáček *et al*., 1961). Phenomena frequently attributed to heavy fish predation include suppressed zooplankton biomass, small individual size of zooplankton, and reduced representation of vulnerable (typically larger) species (Cryer *et al*., 1986). Large-bodied zooplankters are more efficient grazers of phytoplankton than smaller-bodied species and, in the absence of severe predation, dominate the zooplankton (Brooks & Dodson, 1965). Stocking planktivorous fishes may, therefore, cause changes in ecosystem functioning by increasing predation on zooplankton. This can, in turn, reduce grazing of phytoplankton and cause a reduction in water clarity (Hembre & Megard, 2005). For example, *Daphnia* spp. densities in Loch Leven, Scotland, decreased after the introduction of rainbow trout, and there was size-selective predation on *Daphnia* spp. longer than 1.4 mm (Yang *et al*., 1999). Similarly, selection of large zooplankton species and individuals by cyprinids and percids,

Table 2. Hazards associated with stock enhancement activities (modified from Molony et al., 2003). The risk and certainty of hazards occurring are scored as high (H), medium (M) or low (L) to indicate those of most concern.

Hazard	Risk	Certainty	Source(s)
Increased intra-specific competition: due to increased abundance of the species by the addition of hatchery-reared fishes.	M	M	Ackefors *et al.* (1991); Rowland (1994); Su & Liao (1999)
Shifts in prey abundance: changes in the abundance of prey species due to increases in fish abundance as a result of stocking.	L	M	Blaxter (2000)
Prey-switching by wild predators: changes in the targeted prey of wild predatory species, usually to focus on hatchery reared (naïve) fishes due to large numbers released.	L	L	Warburton *et al.* (1998); Wilhelm *et al.* (1999); Willette *et al.* (2001)
Starvation/ food limitation: due to overstocking.	L	M	Dushkina (1991); Ackefors *et al.* (1991)
Exceeding the carrying capacity of an ecosystem: due to continued stocking after recovery.	M	M	L'Abee-Lund (1991); Leber *et al.* (1998); FAO (1999); Blaxter (2000)
Inter-specific competition: competition between hatchery reared fish and other species with similar ecological requirements. May lead to a reduction in abundances of competing species and prey species.	H	M	Rowland (1994); Wiley (1995); FAO (1999)
Displacement of wild stock: by hatchery-reared conspecifics, although there are no well documented examples.	M	L	Blaxter (2000); L'Abee-Lund (1991); Leber *et al.* (1995, 1998); Bannister & Addison (1998); Butcher *et al.* (2000)
Introduction of diseases and parasites: especially due to poor hatchery management and husbandry.	H	H	Fjälling & Fürst (1987); Heggberget *et al.* (1993); Loneragan *et al.* (1998); Wootten (1998); FAO (1999); Burton & Tegner (2000); Lee *et al.* (2001)
Genetic bottleneck: due to lack of genetic management of broodstock within the production system.	H	H	Rowland (1994); Busack & Currens (1995); Compton (1995); Loneragan *et al.* (1998); Penman & McAndrew (1998); Utter (1998); Wootten (1998); Cross (1999); FAO (1999); Hershberger (2002); Lester (2002)
Loss of genetic diversity and fitness: certain alleles of wild fish may become rare due to the release of hatchery-reared fish with a low genetic diversity. This is of higher risk where the wild stock is reduced to very low levels prior to stock enhancement.	M/H	H	Leary *et al.* (1995); Penman & McAndrew (1998); Skibinski (1998); Utter (1998); FAO (1999); Burton & Tegner (2000); Lee *et al.* (2001); Lester (2002); Aprahamian *et al.* (2003)
Extinctions: the loss of species due to increases in the abundance of released fish and ecosystem shifts.	M	L	L'Abee-Lund (1991); Utter (1998); McDowell (2002)
Ecosystem shifts: shifts in the distribution of biomasses or other species, possibly resulting in the loss of other ecosystem values.	M	M	White *et al.* (1995); Crowe *et al.* (1997); Fielder *et al.* (1999); Arnason (2001); Lee *et al.* (2001)
Physical environmental damage: due to stocking operations.	H	H	Lee *et al.* (2001)
Hindrance of difficult management decisions: (e.g. reduction of effort) due to the perception that stock enhancement will allow fishing activities to continue unabated.	H	H	Burton & Tegner (2000)
Diversion of management resources from other activities: for example, other management strategies.	M	H	Burton & Tegner (2000)

such as roach (*Rutilus rutilus* (L.)), bream (*Abramis brama* (L.)), bleak (*Alburnus alburnus* (L.)) and perch (*Perca fluviatilis* L.), can cause shifts in the species composition of cladoceran communities, as well as reductions in the mean size of individuals of large species and in the assemblage as a whole (Perrow & Irvine, 1992; Mehner *et al*., 1995, 1996, 1997; Jeppesen *et al*., 1996; Mehner, 1996, 2000; Korponai *et al*., 1997; Węgleńska *et al*., 1997; Kubečka *et al*., 1998; Moss *et al*., 1998; Chappaz *et al*., 1999; Romare & Bergman, 1999; Romare *et al*., 1999, 2003; Vašek *et al*., 2003; Mátyás *et al*., 2004; Vašek & Kubečka, 2004). In particular, larval and juvenile fishes have the potential to suppress populations of large cladocerans and copepods (Mehner & Thiel, 1999; Hansson *et al*., 2007; Nunn *et al*., 2012). Cryer *et al*. (1986), for example, observed that in summers when 0+ roach were abundant, zooplankton was sparse and dominated by copepods and rotifers, with cladocerans present in only low densities. Stocked and introduced fishes, including trout, may also have significant impacts on populations of aquatic insects (Pope *et al*., 2009), although some studies detected only minor impacts (e.g. Wissinger *et al*., 2006).

Apart from the direct effects of fish predation on zooplankton demography, indirect impacts may also occur through shifts in their life history (e.g. changes in birth rates, fecundity, size and age at maturity, a switch from parthenogenetic to sexual reproduction, diapause), morphology (e.g. cyclomorphosis) or behaviour (e.g. diel vertical and horizontal migration) (Stibor & Luning, 1994; Pijanowska & Stolpe, 1996; Ślusarczyk, 1997; Hanazato *et al*., 2001; Hülsmann *et al*., 2004). All of these effects can have implications for ecosystem functioning. Such phenomena may mask the impacts of fish predation, however. Gliwicz (2001), for example, found that the species-specific density levels of particular zooplankton did not depend upon reproduction rate, since neither increased birth rates nor reproductive effort coincided with an increase in population density; a clear indication that larger numbers of prey were being consumed by fishes at the time of increased reproduction.

Some fish are piscivorous, and may predate upon native fishes. This may be of particular importance where native stocks include rare species or strains (see Section 3.3). Grey *et al*. (2002), for example, found that brown trout in Loch Ness preyed upon Arctic charr (*Salvelinus alpinus* (L.)) and smaller trout. Similarly, in the Cowichan River in British Columbia, the primary food items of large, non-native brown trout were native salmon and trout, and their eggs (Krueger & May, 1991). By contrast, Barnard (2006) found that fish comprised only a relatively small proportion of food eaten by large, stocked diploid brown trout. In other situations, piscivorous fishes may consume zooplanktivorous fishes (such as Arctic charr in Loch Ness; Winfield *et al*., 2002a), thereby reducing predation pressure on zooplankton. This, in turn, may cascade through the trophic levels and cause an increase in grazing pressure on phytoplankton by zooplankton, and an improvement in water clarity. Indeed, both brown and rainbow trout, as well as pike (*Esox lucius* L.), zander (*Sander lucioperca* (L.)) and eel, have been successfully used in biomanipulation experiments to control zooplanktivorous fishes in eutrophic lakes (Geist *et al*., 1993; Frankiewicz *et al*., 1996, 1999; Berg, 1998; Dörner *et al*., 1999; Dörner & Benndorf, 2003; Radke *et al*., 2003; Skov *et al*., 2003; Skov & Nilsson, 2007).

It is also possible that stocked fishes may preferentially feed upon certain invertebrate taxa, which may have important ramifications regarding ecosystem functioning. This may be of particular significance should the preferred prey be of conservation interest (see Section 3.4). Pelagic invertebrate predators of zooplankton, such as *Chaoborus* spp. larvae, *Leptodora kindtii* (Focke) and *Bythotrephes* spp., may limit populations of grazing zooplankton in a similar way to fishes (Hoffman *et al*., 2001; Riccardi *et al*., 2002; Liljendahl-Nurminen *et al*., 2003; Wojtal *et al*., 2004). Indeed, Wissel *et al*. (2000) found that moderate densities of zooplanktivorous fishes were beneficial to the long-term success of biomanipulation because of their feeding upon zooplanktivorous invertebrates. Preferential feeding of fishes upon such taxa may, thus, have similar effects as piscivory upon zooplanktivorous fish. In some situations, therefore, it may be possible for the introduction of fishes to cause a shift from a clear-water ecosystem dominated by macrophytes to a turbid-water ecosystem dominated by phytoplankton, while in others the opposite may occur (Demers *et al*., 2001; Donald *et al*., 2001; McQueen *et al*., 2001; Lathrop *et al*., 2002; Skov *et al*., 2003).

Another potential impact that should be considered is the higher nutrient levels (especially phosphorus) that may arise due to there being an increased standing crop of fishes (see Section 3.2). An increase in nutrient availability may cause a shift in trophic status of a water body, with inherent shifts in ecosystem functioning. Depending upon species, stocked fish may also have the capacity to alter ecosystem functioning via habitat modification. Carp (*Cyprinus carpio* L.), for example, are known to damage aquatic macrophytes through bioturbation (Cahn, 1929; Crivelli, 1983; Fletcher *et al*., 1985; Bruton & van As, 1986; Moyle *et al*., 1986; Welcomme, 1988; Khan *et al*., 2003; Pinto *et al*., 2005; Kloskowski, 2011), and detrimental effects have also been found for other benthivorous and herbivorous fish species, such as bream, tench (*Tinca tinca* (L.)) and grass carp (Cross, 1969; Stott, 1977; Williams *et al*., 2002; Dugdale *et al*., 2006). Similarly, the North American signal crayfish (*Pacifastacus leniusculus* (Dana)), which can be introduced with fish consignments (Section 3.3.7), can have significant impacts on habitat structure and ecosystem functioning. The North American signal crayfish is an opportunistic, polytrophic feeder that can exert significant predation pressure on macrophytes (Guan & Wiles, 1998; Nyström, 1999, 2002; Lewis, 2002). Furthermore, North American signal crayfish have a habit of burrowing, which can weaken bank sides and increase their susceptibility to erosion, potentially leading to increased sedimentation of spawning gravels. Other non-native crayfish species, and the Chinese mitten crab (*Eeriocheir sinensis* H. Milne Edwards), may have similar impacts on habitat structure, including aquatic macrophytes. This is of particular relevance because aquatic macrophytes are integral to ecosystem functioning through their provision of habitat for phytophilic zooplankton (Northcott, 1979; Whiteside *et al*., 1985; Garner *et al*., 1996; Bass *et al*., 1997; Scheffer, 1999; Nurminen *et al*., 2001; Balayla & Moss, 2003, 2004) and refuge for planktonic species from fish predation (Schriver *et al*., 1995; Stansfield *et al*., 1997; Bertolo *et al*., 1999; Perrow *et al*., 1999; Burks *et al*., 2001), and are important for oxygenation of water. In addition, aquatic macrophytes are important food sources for many species of waterfowl, including coot (*Fulica atra* L.), moorhen (*Gallinula chloropus* L.), Canada goose (*Branta canadensis* L.) and mute swan (*Cygnus olor* L.) (Schmieder *et al*., 2006). Although trout can alter very localised physical habitat (e.g. through their spawning activities), under normal circumstances, introduction of trout *per se* is unlikely to change ecosystem functioning in this manner.

A further issue is the potential for competition between stocked and native fishes, or for increased predator presence and predation following stocking (Section 3.3). Moreover, stocking of fishes may cause shifts in habitat use or feeding behaviour by native fish, which could have implications for ecosystem functioning. For example, in the absence of roach, young perch feed mainly upon planktonic cladocerans, whereas in the presence of roach they consume copepods and macroinvertebrates (Persson, 1987; Persson & Greenberg, 1990). Similar interactions have been observed between juvenile Atlantic salmon and Alpine bullhead (*Cottus poecilopus* Heckel) (Amundsen & Gabler, 2008), roach and bream larvae (Nunn *et al*., 2011), brown trout and Arctic charr (Langeland *et al*., 1991), and brown trout and Atlantic salmon (Heggenes & Saltveit, 1990). The interactions, therefore, are complex, and the impacts of stocking are likely to be site-specific.

Summary of potential changes to ecosystem functioning

- Switch of trophic states through grazing pressure on zooplankton.
- Disruption of food chains/webs, e.g. by predation of native fishes or preferential feeding of stocked fish on certain taxa.
- Higher nutrient levels (especially phosphorus) due to an increased standing crop of fishes.

3.2 Impacts of nutrient import

The trophic status of a water body generally describes its degree of fertility or productivity (Moss *et al*., 1996). Water bodies of low fertility or productivity are widely referred to as 'oligotrophic', while successively more fertile and productive systems are termed 'mesotrophic', 'eutrophic' and, ultimately, 'hypertrophic'. This section discusses the possible impacts of nutrient import to designated natural heritage sites in Scotland, either as fish biomass or through supplementary feeding.

The impacts of increased nutrient loading on fresh waters are numerous and well documented (see Moss, 1988, 1996; Moss *et al*., 1996). An increase in nutrient loading may cause a shift in trophic status of a water body, with inherent shifts in ecosystem functioning (Section 3.1). The increased standing crop of fishes following stocking may initially increase nutrient loading by conversion of animal and plant matter to nitrates and phosphates through the processes of digestion and excretion/defaecation, and later via their death, whereupon decomposition of their carcases releases nutrients into the water column or sediment. Increased availability of nutrients allows an increase in productivity of primary producers which, in turn, allows higher production of animals such as zooplankton and fishes. In addition, growth rates of native fishes may increase as a result of the overall increase in productivity of the water body and their feeding upon excess food (see below). The symptoms of nutrient loading are most frequently observed in aquatic plant communities (Carvalho & Moss, 1995), especially charophytes, which invariably require nutrient-poor conditions (see Section 3.4.2), but invertebrate and other animal groups are also affected. Regarding fishes, there is often a sequence of shifts in species composition from salmonids-to-percids-to-cyprinids with increasing trophic status (Persson *et al*., 1991; Sandström & Karås, 2002; Tammi *et al*., 2003). This is of particular significance to native brown trout, Arctic charr and coregonid populations that inhabit oligotrophic or mesotrophic stillwaters (Section 3.3; Mills *et al*., 1990; Winfield, 1992), especially as nutrient recycling by fishes is thought to be greatest in low-productivity systems (Griffiths, 2006). For example, rainbow trout are more tolerant of eutrophic conditions than are brown trout (Taylor, 1978) and, thus, eutrophication could increase the success of the former over the latter species (Phillips *et al*., 1985). Similarly, eutrophication is likely to increase the success of percids and cyprinids.

In addition, Carvalho & Moss (1995) found that mobilisation of nutrients and increased turbidity by carp, and to lesser extent bream and tench, was an important cause of eutrophication in a sample of stillwater SSSIs, and other workers obtained similar results (e.g. Williams *et al*., 2002; Zambrano *et al*., 2001; Miller & Crowl, 2006). Although trout can alter localised physical habitat (e.g. through their spawning activities), under normal circumstances, it appears unlikely that introduction of trout *per se* would significantly increase nutrient loading in this way. However, cyprinids or artificially high densities of trout, especially in shallow water bodies, may cause suspension of sediment, which could release nutrients into the water column.

It is also possible that stocking may increase nutrient loading by virtue of the higher numbers of fishes feeding upon terrestrial organisms. Indeed, Mehner *et al*. (2005, 2007) suggested that feeding of fishes on terrestrial insects may subsidise the nutrient pool of lakes, especially in small, oligotrophic water bodies in forested areas that have a high perimeter-to-area ratios. Similarly, whereas the food of Arctic charr in Loch Ness is derived primarily from autochthonous sources (Grey *et al*., 2002), the food of brown trout is believed to be derived largely from allochthonous inputs (Jones *et al*., 1998). Furthermore, fish stocking may increase nutrient loading via elevated numbers of birds visiting the water body (guanotrophication). The issues relating to this are discussed in Section 3.3.2.

Changes in trophic status may also occur in water bodies due to stocked fish (or fish growing-on in cages) being provided supplementary food. Numerous examples exist of the impacts of fish farms upon the sediment and biotic characteristics in the vicinity of fish cages, and trophic status in general (e.g. Penczak *et al*., 1982; Merican & Phillips, 1985; Phillips *et al*., 1985; Enell, 1995; Honkanen & Helminen, 2000; Yokoyama *et al*., 2006; Fernandez-Jover *et al*., 2007; Jusup *et al*., 2007; Azevedo *et al*., 2011). Impacts are invariably similar to those attributed to excessive 'groundbaiting' at cyprinid fisheries (e.g. Cryer & Edwards, 1987; Niesar *et al*., 2004; Arlinghaus & Niesar, 2005). A common symptom is an increase in trophic status, together with a deterioration in water quality (e.g. increased total-P, PO_4-P, NH_4-N, organic-N and total-C, and decreased dissolved oxygen concentrations) and associated changes in flora and fauna (Phillips *et al*., 1985; Jones, 1990; Fozzard *et al*., 1999; Marsden & Mackay, 2001; Schindler *et al*., 2001; Zambrano *et al*., 2001). In Loch Fad, western Scotland, for example, inorganic nitrogen, ortho-phosphate and suspended solids were significantly higher near fish cages than elsewhere, and the phytoplankton was dominated by the toxic cyanobacterium *Microcystis aeruginosa* (Kützing) (Stirling & Dey, 1990). Similarly, point sources of sewage and effluents from fish-rearing ponds are major contributors to phosphorus loading in Loch Leven (Bailey-Watts & Kirika, 1999; May *et al*., 2001, 2012; Carvalho *et al*., 2012; Spears *et al*., 2012), and approximately 50% of the summer total phosphorus load in Lake of Menteith is derived from fish-cage discharge (Fozzard *et al*., 1999; SEPA, 2002). The effects of cage farming systems on freshwater ecosystems are compounded by the relatively high water-retention times in lake basins compared with the more regular flushing of estuarine or coastal systems (Grey *et al*., 2004). Released and wild fishes often congregate around fish cages (e.g. Dempster *et al*., 2002, 2004), and may feed upon excess food (Phillips *et al*., 1985). The risks of large aggregations of fishes near fish cages are beyond the scope of this report, but include increased transmission of parasites between wild and farmed fishes (Phillips *et al*., 1985). It is also possible that fish feeding on supplementary food may grow to a size large enough to become piscivorous, with the inherent implications discussed in Section 3.3.1.

Summary of potential impacts of nutrient import

- Higher nutrient levels (especially phosphorus) due to increased standing crop of fishes.
- Increased nutrient loading by virtue of the higher numbers of fishes feeding upon terrestrial organisms.
- Mobilisation of nutrients and increased turbidity by carp, and to lesser extent other cyprinids, can cause eutrophication.
- Stocked fish (or fish growing-on in cages) may be provided with supplementary food, which increases nutrient inputs.

3.3 Risks to native salmon, trout, charr, whitefish and lamprey populations

In the UK, numerous water bodies support unique strains of salmon, brown trout and charr (Mills *et al*., 1990; Maitland, 2004; Wheeler *et al*., 2004; Adams & Maitland, 2007; Maitland *et al*., 2007), and a small number contain whitefish. Three species of whitefish occur in the

UK, namely vendace (*C. albula* (L.)), powan (syn. schelly/gwyniad, *C. lavaretus* (L.)) and pollan (*C. autumnalis* (Pallas)). Vendace is found only in Derwent Water in the English Lake District and a translocated population in Loch Skeen in Scotland, with the populations in Bassenthwaite Lake (English Lake District) and Castle and Mill Lochs (Scotland) now believed to be extinct (Maitland, 1990; C.W. Bean, *pers. comm.*). Powan is found in lochs Lomond and Eck (Scotland), Haweswater, Ullswater, Red Tarn and Brotherswater (England), and Llyn Tegid (Wales). Pollan is found in loughs Neagh, Erne, Derg and Ree in Ireland. All three species are listed in Appendix III (protected species) of the Bern Convention on the Conservation of European Wildlife and Natural Habitats (1979). A number of these populations are already under threat (Maitland & Lyle, 1990; Maitland, 1995), including from eutrophication and introductions of non-native fish species, such as ruffe (*Gymnocephalus cernuus* (L.)) (Winfield, 1992; Winfield *et al*., 1996, 1998, 2002b, c, 2007, 2010, 2011; Winfield & Durie, 2004). Consequently, attempts have been made to establish refuge populations for vendace (Loch Skeen, Loch Valley and Daer Reservoir, Scotland; I. Sime, *pers. comm.*) and powan (Loch Sloy and Carron Valley Reservoir, Scotland; Etheridge *et al*., 2010). Four new sites have been identified as potential refuges for powan (Allt na Lairige, Lochan Shira, Loch Glashan and Loch Tarsan) and translocations from Loch Lomond and Loch Eck have already taken place.

Sea lamprey (*Petromyzon marinus* L.), river lamprey (*Lampetra fluviatilis* (L.)) and brook lamprey (*Lampetra planeri* (Bloch)) are widely distributed throughout the British Isles, although in Scotland the sea and river lamprey are mainly, but not exclusively, distributed to the south of the Great Glen (Maitland & Campbell, 1992; Ecological Research Associates, 2005). All three lamprey species are protected under the EC Habitats Directive, the Bern Convention and UK Biodiversity Action Plans (Harvey & Cowx, 2003; Maitland, 2003; Nunn *et al*., 2008a; Harvey *et al*., 2010). Of particular note in Scotland is the population of dwarf river lamprey found only in Loch Lomond and the Endrick Water (Maitland *et al*., 1994).

This section reviews the likely impacts from stocking or introducing fishes into water bodies containing native salmon, brown trout, charr, whitefish or lamprey populations. When assessing the risks of stocking or introduction, it is important to consider the possibility of dispersal of fishes from their site of release into connected watercourses, and also the size of any residual stocks in water bodies that are stocked on a regular basis.

3.3.1 Predation (direct impacts of stocking)

Species interactions involving predation are probably the most evident and widely documented impact of stocked or introduced species, and can result in the complete elimination of indigenous species in parts of their range (see Holcík, 1991; Cowx, 1997; Cambray, 2003). Globally, introduced salmonids and piscivorous species, such as the largemouth bass (*Micropterus salmoides* (Lacépède)) and Nile perch (*Lates niloticus* (L.)), are particularly notorious. For example, bass (*Micropterus* spp.) and trout (*Salmo* spp., *Oncorhynchus* spp.) have been implicated in the decline or local extinction of eight cyprinid species, the Cape kurper (*Sandelia capensis* (Cuvier)) and the airbreathing shellear (*Kneria auriculata* (Pellegrin)) in South Africa (Skelton, 1993), with largemouth bass also one of the main causes of a decline in the endangered species *Anaecypris hispanica* (Steindachner) in Spain and Portugal (Collares-Pereira *et al.*, 1998). Similarly, populations of many species of galaxiid have declined or become extinct following introductions of salmonids (McDowall, 1990, 2003, 2006).

Some large trout are piscivorous, and may predate upon native fishes (Welton *et al*., 1997; Hyvärinen & Huusko, 2006; Pink *et al*., 2007; Arismendi *et al*., 2009; Nasmith *et al*., 2010). This may be of particular importance where native stocks include rare species or strains; as mentioned previously, brown trout in Loch Ness prey upon Arctic charr and smaller trout (Grey *et al*., 2002), and wild brown trout in the River North Esk, Scotland, sometimes consume salmon smolts (Shearer, 1992). Introductions of trout have been implicated in the decline or disappearance of many native fish species, through either predation or competition, with a paucity of indigenous fish species being reported in many areas where non-native trout occur (Nijjsen & de Groot, 1974; McDowall, 1990, 2003, 2006; Arthington & Blühdorn, 1996). There are also numerous reports of rainbow trout escaping from farms and decimating indigenous fish stocks, particularly through predation of juveniles. In parts of South Africa, rainbow trout prey on, and compete for food with, the rare indigenous Maluti minnow (*Oreodaimon quathlambae* (Barnard)) (Bruton & van As, 1986). In addition, rainbow trout have been shown to prey on the endangered barred galaxias (*Galaxias fuscus* Mack), yarra pygmy perch (*Nannoperca obscura* (Klunzinger)) and golden pygmy perch (*Nannoperca variegata* Kuiter & Allen) in Australia (Wager & Jackson, 1993). The distributions of rainbow trout and mountain galaxias (*Galaxias olidus* Günther) in the Australian Capital Territory appear to be mutually exclusive, presumably because of predation (Lintermans, 1991), and there have been similar impacts on the distribution of the common river galaxias (*Galaxias vulgaris* Stokell) in New Zealand (McDowall, 1990). Similarly, brown trout have been implicated in declines in populations of a number of fish species in Australia (Wager & Jackson, 1993). Lampreys are consumed by a range of predators, including brown and rainbow trout, with shoals of migrating or spawning lampreys being especially vulnerable to predation (Cochran *et al*., 1992; Cochran, 2009). Of the fish species that occur regularly in UK fresh waters, there is evidence that brown trout, rainbow trout, pike, eel, zander and European catfish (*Silurus glanis* L.) can exert significant predation pressure on fish populations (Linfield & Rickards, 1979; Fickling & Lee, 1983; Linfield, 1984; Hickley, 1986; Geist *et al*., 1993; Frankiewicz *et al*., 1996, 1999; Berg, 1998; Smith *et al*., 1998; Dörner *et al*., 1999; Dörner & Benndorf, 2003; Radke *et al*., 2003; Skov *et al*., 2003; Wysujack & Mehner, 2005; Skov & Nilsson, 2007; Copp *et al*., 2009b).

In some situations, piscivory by stocked trout on native fishes may potentially be more significant because of the similarity in habitat occupied by wild trout, charr and whitefish, and the artificially high numbers of trout following stocking. The degree of piscivory may depend upon the species of trout stocked, with brown trout seemingly more prone to exhibit piscivory than rainbow trout (Phillips *et al*., 1985). In the Cowichan River in British Columbia, the primary food items of large, non-native brown trout were native salmon and trout, and their eggs (Krueger & May, 1991). In spite of this, colonisation of the Cowichan River by brown trout apparently had little or no impact on native salmonid abundance (Wightman *et al*., 1998; Marsh, 2000). Barnard (2006) suggested that brown trout up to the 1+ age class were potentially at risk from predatory trout. However, fish comprised only a relatively small proportion of the food eaten by large, stocked diploid brown trout, although the data were not considered sufficient for firm conclusions to be drawn on the potential impacts of piscivory by stocked trout on wild fish communities (Barnard, 2006).

There appears to be little evidence that stocked trout predate on fish eggs or induce significant egg mortality. Predation may occur on eggs that have not been buried and are, therefore, unlikely to survive, but there is little evidence of trout predating upon eggs that would otherwise have been viable. An exception to this is in the Cowichan River in British Columbia, where the eggs of native salmonids were found to be important in the diet of non-native brown trout (Krueger & May, 1991). There may be a greater risk of predation on charr and whitefish eggs, however, because they are not buried to the same extent as salmon or trout eggs. Indeed, declines in a number of populations of whitefish species, including the powan in Loch Lomond, are thought to have been partly due to the spread of ruffe, which may feed on their eggs (Section 3.6; Adams & Tippett, 1991; Ogle, 1998; Winfield *et al*., 1998; Etheridge *et al*., 2011). A number of other fish species, particularly pumpkinseed (*Lepomis gibbosus* (L.)) and mosquitofishes (*Gambusia* spp.), feed on the eggs and larvae of fishes and amphibians (Gamradt & Kats, 1996; Ivantsoff, 1999; García-Berthou & Moreno-Amich, 2000; Zeiber *et al*., 2008; Reynolds, 2009), and signal crayfish, which can be introduced with fish consignments (Section 3.3.7), also pose a threat as they sometimes prey upon fishes or their eggs (Guan & Wiles, 1997; Nyström, 1999, 2002; Lewis, 2002;

Peay *et al*., 2009). No specific studies of the impacts of predation on lamprey eggs appear to have been conducted, although a number of fish species, including minnow (*Phoxinus phoxinus* (L.)), stone loach (*Barbatula barbatula* (L.)) and rainbow trout, have been observed to consume lamprey eggs (Close *et al*., 1995, cited in Cochran, 2009; Jang & Lucas, 2005). Indeed, large shoals of fishes, such as minnow, dace (*Leuciscus leuciscus* (L.)) and gudgeon (*Gobio gobio* (L.)), sometimes congregate immediately downstream of spawning lampreys (*pers. obs.*) and may consume drifting eggs. Although the impacts of predation on fish eggs by stocked or introduced fishes do not appear to have been quantified, concerns have been expressed by fishery managers as to possible damage arising from this behaviour.

3.3.2 Predation (indirect impacts of stocking)

Increased densities of fishes may encourage larger numbers of feeding cormorants (*Phalacrocorax carbo carbo* (L.), *P. carbo sinesis* (Blumenbach)) and, in some places, other piscivorous birds such as goosander (*Mergus merganser* L.), red-breasted merganser (*Mergus serrator* L.), grey heron (*Ardea cinerea* L.), osprey (*Pandion haliaetus* (L.)) and various divers (Gaviidae) and grebes (Podicipedidae), as well as otter (*Lutra lutra* (L.)) and American mink (*Neovison vison* (Schreber)). For example, it is thought that fish stocking led to an increase in cormorant numbers and predation at Loch Leven, Scotland (Stewart *et al*., 2005). The impacts of predation by piscivorous birds on inland fish populations can be acute (see Feltham *et al*., 1999; Cowx, 2003; Russell *et al*., 2003, 2008; Orpwood *et al*., 2010), and may impact directly upon fish species of high conservation value (Winfield *et al*., 2003). Cormorants are capable of removing considerable numbers and biomass of fishes through predation, with inherent implications for fish standing crop and community structure. There may also be ramifications for fish growth rates and fecundity. Britton *et al*. (2002), for example, observed that heavy predation by cormorants at Holme Pierrepont Rowing Course (Nottingham) caused an increase in growth rates and fecundity, and lower ages at maturity of the remaining fishes. Alternatively, a greater availability of (potentially naïve) fishes may reduce predation pressure on species of conservation interest because of the relatively larger size and ease of capture of stocked fish compared with native fishes, However, there was no significant difference in the ratios of wild and stocked brown trout in Loch Leven and the diets of cormorants feeding at the site (Stewart *et al*., 2005). It should be noted that increased densities of fishes could benefit other animals of conservation interest, such as osprey and otter. Indeed, predation by otters is being increasingly reported at intensively stocked recreational fisheries, as well as on newly established salmon populations (Kloskowski, 2000, 2005; Britton *et al*., 2005; Kortan *et al*., 2007, 2010).

Time of year is an important factor determining the impacts of predation by birds on fishes. At most sites, cormorant numbers are highest during the winter (Bearhop *et al*., 1999) and, therefore, predation on fish populations is usually highest at that time. Similarly, goosander, red-breasted merganser and most grebes and divers are usually present in the largest numbers at inland water bodies during winter. Exceptions are invariably where large breeding populations of birds occur in areas where they are not ordinarily found. Such cases are likely to be site-specific, as overwintering populations of cormorants and many of the other bird species greatly exceed those at other times of the year. Related to predation, an increased presence of piscivores may also be detrimental because of damage inflicted on fish during failed attacks (Russell *et al*., 2003, 2008; Orpwood *et al*., 2010; Kortan & Adámek, 2011). Fish that survive attacks by piscivorous birds and mammals frequently have characteristic wounds, which are potential sites of infection for pathogens such as fungi and bacteria. In addition, harassment of fish or even the mere presence of piscivorous birds and mammals may potentially impact upon fish health by increasing stress, which could suppress appetite, growth and immune responses. Moreover, avoidance of predation in refuge areas is likely to reduce the growth and reproduction potential of fish due to reduced feeding activity and energy intake (Kortan & Adámek, 2011).

Kirk *et al*. (2003) stated that cormorants may play an important role in transmission of parasites. Thus, there is a possibility that an increased presence of cormorants and other piscivorous birds may increase the incidence of parasitism in fishes (e.g. Bean & Winfield 1989, 1992). This may be of particular importance should the water body contain fish species of conservation interest. Some parasites with simple, direct life cycles are capable of infecting a variety of host types and may, thus, pass directly from birds to fishes. By contrast, parasites with complex life cycles exploit a variety of intermediate and definitive hosts, including fishes and birds. Visiting birds may potentially introduce parasites or increase their abundance through expulsion of eggs or infective stages with their faeces. In addition, in situations where the number of definitive hosts (i.e. piscivorous birds) is limited, increased bird numbers may allow an increase in the numbers of parasites that attain maturity, thereby increasing parasite reproduction potential.

Finally, it is also possible that piscivorous birds may impact upon the trophic status of water bodies. This may be of particular concern for water bodies that support fish species, such as charr (Mills *et al*., 1990) and whitefish (Winfield, 1992), that are dependent upon oligotrophic or mesotrophic conditions. Elevated numbers of piscivorous birds feeding at a given water body may facilitate nutrient cycling through their consumption of fishes and excretion/defaecation of waste products. The higher availability of nutrients allows an increase in primary and, accordingly, secondary productivity. In addition, increases in the numbers of 'loafing' or roosting birds may increase trophic status through a process known as guanotrophication (Leah *et al*., 1978; Moss & Leah, 1982; Ellis *et al*., 2006; Chaichana *et al*., 2010, 2011). Such situations are likely to be site-specific, and arise when loafing/roosting birds transfer nutrients from elsewhere into a water body with their excreta/faeces. It is also possible, however, that the opposite could occur, with large numbers of piscivorous birds feeding at given water body but roosting elsewhere, thereby removing nutrients from the catchment.

Summary of potential risks to native fish populations - Predation

- Stocking of trout has been implicated in the decline or disappearance of many native fish species - *risk medium/high*.
- Piscivory by stocked fish, which depends upon the species of trout stocked: brown trout more prone to exhibit piscivory than rainbow trout - *risk low*.
- Predation on eggs - *risk low.*
- Increased densities of fishes may encourage larger numbers of feeding piscivorous birds - *risk low/medium.*

3.3.3 Competition

The ecological regulation of salmonid stocks is generally considered to be density-dependent, linked to the productivity, suitability and carrying capacity of the environment (Armstrong *et al*., 2003; Klemetsen *et al*., 2003; Milner *et al*., 2003; Holmlund & Hammer, 2004). The capacity to support a stock of spawning-sized fish is finite, with occupation of the available habitat determined by density-dependent competition. It is inevitable, therefore, with the exception of populations operating well below their carrying capacity, that unless increases in stocked trout populations are compensated by sufficient increases in food availability, overall production of native trout will decrease (Phillips *et al*., 1985). This may equally apply to interactions between stocked trout and native charr and whitefish, because of overlap in habitat occupied by trout, charr and whitefish in oligotrophic lakes. Notwithstanding, Duncan (1991, cited in Welton *et al*., 1997) detected no impacts on charr from the presence of introduced rainbow trout in Loch Awe, Scotland.

Competition with stocked fishes may affect wild fish stocks through either: (1) displacement from habitat through aggressive behaviour (Altukhov, 1981; Garcia-Marin *et al*., 1991, 1998); or (2) reduced energetic performance (e.g. growth, reproductive capacity) because of increased energetic cost of territory defence and competition for food. These issues are important because selection and culture conditions in hatcheries generally, although not universally (see Fleming *et al*., 2000), lead to hatchery-reared fish being more aggressive and faster growing than wild fish (Bachman, 1984; Ferguson, 2003, 2004; Ferguson *et al*., 2007). However, the higher aggression of hatchery-reared salmonids does not necessarily confer a competitive advantage to stocked fish, as hatchery-reared salmonids are reportedly: (1) inefficient feeders in natural conditions; (2) less energetically efficient than wild fish; and (3) unable to capitalise on success during aggressive interactions (e.g. occupation of optimal foraging sites) (Bachman, 1984; Deverill *et al*., 1999a, b). Indeed, Weber & Fausch (2003) concluded that, whilst most evidence implied competitive differences between hatchery-reared and wild fish, the ecological consequences of these differences had not been quantified in most cases. Furthermore, competitive effects are not inevitable because of the interactions between genetic background, environment, life stage, size and other factors. Nevertheless, Fleming *et al*. (2000) found that resource competition and competitive displacement from introduced fish depressed productivity of native salmon populations by more than 30%, and concluded that continuous interventions could impact upon population productivity, disrupt local adaptations and reduce the genetic diversity of wild populations.

Competition for food resources and habitat with stocked fish may result in reduced growth, survival and reproductive potential of native fishes (Waples & Drake, 2004; Britton *et al*., 2007, 2011a). Competitive effects occur when behavioural interactions cause an unequal distribution of a resource that is directly or indirectly related to growth, survival or recruitment (Wootton, 1990; Ward *et al*., 2006). For example, fishes may alter their diets, and have lower growth rates, in the presence of competing species. Persson & Greenberg (1990) demonstrated that roach had a negative impact on the growth of juvenile perch, with individual growth rates of perch decreasing with increasing roach density, which was related to competition for food resources. In the absence of roach, perch fed mainly upon planktonic cladocerans, whereas in the presence of roach they consumed copepods and macroinvertebrates. Similarly, Amundsen & Gabler (2008) found empirical evidence for food limitation and competition between juvenile Atlantic salmon and Alpine bullhead, resulting in reduced food acquisition and growth rates in Atlantic salmon. Similar interactions have been observed between brown trout and Atlantic salmon (Heggenes & Saltveit, 1990), brown trout and Arctic charr (Langeland *et al*., 1991), roach and dace (Cowx, 1989), and roach and bream (Nunn *et al*., 2011). If stocked fish successfully occupy habitat and use resources that would otherwise be used by native fishes then, over time, the characteristics and contribution of the wild spawning stocks may change. This could potentially be realised in terms of overall size of the spawning stocks, or the size or age at maturity of the wild fish. Indeed, Nilsson (1955, 1961, 1965, 1967, cited in Klemetsen *et al*., 2003) demonstrated interactive segregation between lacustrine charr and trout, with charr inhabiting the littoral zone in allopatry and shifting to the pelagial or profundal in sympatry. The repeated injection of farmed fish into fisheries negates the effects of mortality (natural and fishery), and could potentially minimise the chances of wild fish maturing and occupying these niches. Ultimately, this has the potential to impact negatively on spawning stocks of native trout, charr and whitefish.

Competition for food, and space, both with conspecifics and other species may be particularly strong when unnaturally high densities of fishes are released in restricted areas, potentially leading to stunting of stocked/introduced and/or wild fishes. Stunting is a process whereby populations of species expand rapidly, producing large numbers of individuals that mature and breed at much-reduced sizes, and considerably diminishes the usefulness of the populations for angling or commercial purposes. Furthermore, stunted populations invariably subject food resources to a higher pressure than do size-structured populations, as small fishes require more prey per unit body weight than do larger individuals (e.g. Smith *et al*., 1996). Therefore they may exacerbate any impacts on food-web structure, nutrient dynamics and ecosystem functioning. A number of fish species have been reported as producing stunted populations and have been introduced into Scotland (e.g. crucian carp in lochs Lomond and Rannoch), whereas others occur there naturally (e.g. brown trout, Arctic charr, roach) (Adams, 1994; Fraser & Adams, 1997; Winfield *et al*., 2011).

Exploitation competition is often invoked as the mechanism underlying the decline of indigenous fish species in areas where non-native and stocked species become established and abundant. Exploitation competition occurs as a result of a shortage of a critical resource required by competing organisms. The resource is usually food or space (i.e. the physical habitat required for spawning, foraging and other activities). Competition may alternatively involve a collection of effects termed 'interference', including territoriality, injury or death by encounter and inhibition of reproduction (Schoener, 1986). These two types of competition are often imperfectly distinguished in the description of interactions between stocked/introduced and wild fishes. Exploitation competition is notoriously difficult to demonstrate in the field, and most of the examples of impacts attributed to competition have no experimental basis. Notwithstanding, there is a strong belief that stocked or introduced fishes may out-compete indigenous species to the point of causing considerable reductions in abundance or even the disappearance of species. For example, brown trout are reported to have competed with and displaced indigenous salmonids in North America (Clugston, 1990), and the decline and fragmentation of galaxiid and Macquarie perch (*Macquaria australasica* Cuvier & Valenciennes) populations in Australia has been attributed to competition with brown trout for food (Fletcher, 1979; Jackson & Williams, 1980; Wager & Jackson, 1993). Similarly, several native species have been out-competed by introduced tilapiine cichlids in the southern USA (Noble, 1980, cited in Welcomme, 1988), and in parts of South Africa, a rare species of kurper (*Sandelia bainsi* Castelnau) is reportedly threatened by competition with introduced rainbow trout, largemouth bass and translocated African sharptooth catfish (*Clarias gariepinus* (Burchell)) (Bruton & Van As, 1986; Cambray, 2003).

Summary of potential risks to native fish populations - Competition

- Displacement of native fishes through aggressive behaviour – *risk low/medium.*
- Competition reduces energetic performance of native fishes– *risk low.*
- Competition for food resources and habitat with stocked fish may result in reduced growth, survival and reproductive potential of native fishes – *risk medium.*
- Reduction in stocks of subordinate species or age groups – *risk medium*.
- Overstocking can lead to reduction in fishery performance through competitive bottlenecks – *risk variable*.

3.3.4 Spawning and post-spawning recovery and survival

The interactions of fertile stocked fish with spawning wild individuals is the fundamental source of genetic introgression between hatchery and wild strains of fishes (Fleming *et al*., 2000; Verspoor *et al*., 2005). However, the levels of introgression are variable, and not always as high as expected from stocking rates in relation to wild fish densities (Section 3.3.5). In some situations, stocked or escaped fish spawn earlier or later than wild fish, resulting in temporal segregation of spawning activity (Webb *et al*., 1991; Shields *et al*., 2005). In addition, in some southern English chalk streams, spawning of stocked brown trout occurs in the main river stem, whereas wild fish spawn in tributaries and carrier streams (Shields *et al*., 2005). Similarly, differences in the spawning distribution of wild and escaped

Atlantic salmon were observed in the River Polla, Scotland, with wild fish tending to spawn further upstream than escapees (Webb *et al*., 1991). Such differences in behaviour may reduce the risk of genetic introgression between stocked and wild fish, although it is not known how widespread the phenomenon is. Where trout occur in stillwaters, spawning usually occurs in tributary streams, and stocked trout may compete with native trout for spawning habitat. Stocking with triploid trout may eliminate this problem, although the wider consequences of stocking triploids require further study (see Noble *et al*., 2004). In the UK, charr and whitefish generally spawn in lakes (but see Walker, 2007), and so are less likely than brown trout and Atlantic salmon to compete directly with stocked trout for spawning habitat. Moreover, whitefish spawn at a different time of year to trout (and coarse fish), while charr populations can have autumn- and spring-spawning components.

Another factor that may affect the interactions between stocked and wild fishes is differential mortality because of the energetic costs of spawning. A potential hazard is that stocked fish may affect the post-spawning recovery and survival of wild fishes. The risk is probably dependent upon the relative condition of stocked and wild fishes at the end of the spawning period, and their relative competitive abilities. For example, it is believed that in some southern English chalk streams, stocked brown trout spawn earlier than wild fish (Shields *et al*., 2005) and, thus, presumably recover earlier than their wild conspecifics. Therefore, stocked fish may pose a hazard to the post-spawning recovery of wild fish, especially as they are generally more aggressive than wild fish (Bachman, 1984; Weber & Fausch, 2003; Ferguson, 2004). Stocking in the spring may minimise the impacts, as wild trout should have recovered from spawning and stocked fish will not have acclimatised to their new environment. There is also evidence from eastern Canada that a shift to earlier spawning times may increase the success of rainbow trout competing with brown trout (Dodge, 1983, cited in Phillips *et al*., 1985). Although self-sustaining populations of rainbow trout are relatively rare in the UK (Welton *et al*., 1997; Fausch, 2007), where they do occur with potential competition for spawning habitat, and earlier post-spawning recovery may be detrimental to native brown trout stocks.

Summary of potential risks to native fish populations - Spawning and post-spawning recovery and survival

- Stocked trout may compete with native trout for spawning habitat – *risk low.*
- Differential mortality may occur between stocked and wild fishes because of energetic costs of spawning – *risk low/medium.*

3.3.5 Genetic impacts

The potential genetic impacts of stocking and introducing fishes are well known (Carvalho & Cross, 1998; Ferguson, 2003, 2007; Simon & Townsend, 2003; Hänfling *et al*., 2005; Madeira *et al*., 2005; Izquierdo *et al*., 2006; Hänfling, 2007; Griffiths *et al*., 2009; Mehner *et al*., 2009; Hansen *et al*., 2010; Wollebaek *et al*., 2010; Nock *et al*., 2011; Winkler *et al*., 2011). Issues related to the release of strains or varieties can be similar to those associated with the release of non-native species (Coates, 1998). Stocked fishes often interbreed with their wild conspecifics, which may lead to disruption of genetic stocks (Ryman, 1981; Campton & Johnston, 1985; Taggart & Ferguson, 1986; Guyomard, 1989; Reisenbichler & Phelps, 1989; Utter *et al*., 1989; Harada *et al*., 1998; McGinnity *et al*., 2003). However, the contribution of stocked fishes to recruitment is variable. Hansen (2002), for example, calculated that the contribution of stocked brown trout to the gene pool of wild populations varied from 5% to as much as 88%. Differences in the timing of spawning (a high heritability trait) of stocked and wild brown trout are a major factor in reducing introgression (Ferguson, 2007). In some southern English chalk streams, for example, it is believed that fertile stocked brown trout spawn earlier than wild fish, thereby reducing (but not excluding) introgression of genetic material from hatchery fish into the wild populations (Shields *et al*., 2005). Genetic introgression is of particular concern where fish stocks are geographically isolated and may be genetically distinct (see Mills *et al*., 1990; Maitland, 2004; Adams & Maitland, 2007; Maitland *et al*., 2007). For example, ferox brown trout are genetically distinct from co-occurring brown trout in lochs Awe and Laggan in Scotland (Duguid *et al*., 2006). The use of all-female, triploid fishes has been suggested as a possible solution (see Noble *et al*., 2004 for a review) as they are considered infertile, although the processes used to induce triploidy may not be 100% effective (Pawson, 2003). The issues related specifically to the genetic impacts of stocking on indigenous brown trout populations and interbreeding between stocked brown trout and wild Atlantic salmon are reviewed in detail by Ferguson (2007) and Cowx *et al*. (2006), respectively.

The reproductive capabilities of farmed fishes are often lower than those of wild conspecifics (Chilcote *et al*., 1986; Leider *et al*., 1990; Campton *et al*., 1991; Jonsson *et al*., 1991; Fleming & Gross, 1993). For example, controlled experiments with Atlantic salmon from the River Imsa, Norway, demonstrated that spawning success was higher for wild than for hatchery-reared fish, even when the potential for genetic differences between the two groups was very limited (i.e. parents were of wild local stock, with only half a generation in captivity) (Jonsson & Fleming, 1993), and other studies have reported similar results. In particular, males of hatchery origin often appear to have a lower reproductive success than females of hatchery origin. Jonsson *et al.* (1990, 1991) observed that the proportion of unspawned individuals, particularly males, was higher among ocean-ranched Atlantic salmon than among wild fish. Similarly, Fleming & Gross (1993) calculated that sea-ranched coho salmon (*Oncorhynchus kisutch* (Walbaum)) averaged 72% of the breeding success of wild fish. Leider *et al.* (1990) reported the lifetime reproductive success of sea-ranched steelhead (anadromous rainbow trout) to be only 11-13% that of wild fish. In addition, sea-ranched fish stayed in the spawning area for shorter time periods, strayed more during the spawning period, and were injured more often than wild fish. Although the reproductive capabilities of farmed fishes are often lower than those of wild conspecifics, it is not always the case. For example, the reproduction of escaped fishes did not appear to differ from that of wild individuals in either the River Oselven, Norway, or the River Polla, Scotland (Webb *et al*., 1991; Fleming, 1995).

Stocked or introduced fishes (including escapees from aquaculture facilities) may interact with wild populations by breeding with either conspecifics or closely related species (Munday *et al*., 1992; Beveridge & Phillips, 1993; Ferguson, 2007). Indeed, there is now a substantial body of evidence of interbreeding between escapees from fish farms and wild populations, including between native and non-native *Oncorhynchus* and *Salmo* species or sub-species (Allendorf & Leary, 1988; Verspoor, 1988; Garcia de Leániz *et al*., 1989; see Cowx *et al*., 2006, 2010). In southern Norwegian rivers, for example, up to 28% of spawning Atlantic salmon may originate from fish farms (Munday *et al*., 1992). Similarly, numerous studies, including in Sweden, France, Spain, Ireland, Canada, Australia and the USA, have reported interbreeding of escapee brown trout and rainbow trout with indigenous populations, with introgression rates of up to 80% being recorded in France (Munday *et al*., 1992; Fletcher, 1986). The effects of interbreeding vary from no measurable impacts on the genetic structure of local stocks (Borgstrøm *et al*., 2002) to partial or complete displacement of genetically distinct indigenous populations by homogeneous hatchery fish (Munday *et al*., 1992). Thus, the threat of hybridization – whether intraspecific (between races, strains or sub-species of a species), interspecific (between species) or intergeneric (between genera) – with farmed fishes to the genetic integrity of wild populations must be considered a major concern for some species, especially salmonids and those of conservation importance.

Atlantic salmon in particular vary in their morphology from river to river, with wild populations often being adapted to the environments in specific rivers (Munday *et al*., 1992). Such adaptation is maintained by natal-stream homing of the adult fish. Many traits in Atlantic salmon have a heritable genetic basis, including growth rates, ages at maturity and smolting, egg sizes, the timing of sea migration and migratory behaviour at sea (Institute of Aquaculture, 1990). Similarly, farm-reared brown trout result in greater introgression in the freshwater component than in the anadromous component because farm-reared brown trout that became anadromous experience high mortality at sea. Given that anadromy is a threshold quantitative trait (i.e heritable), stocking farm-reared brown trout is likely to increase the freshwater component and reduce sea trout runs (Ferguson, 2007). Thus, there are concerns that the adaptive traits and reproductive fitness of genetically distinct wild stocks may be significantly affected by interbreeding with stocked or escapee fish (Beveridge & Phillips, 1993; Cowx *et al*., 2010). This issue was demonstrated by the conservation programme for Arctic charr in Lake Saimaa (Prammer *et al*., 1999). Prior to the early 1980s, only one self-sustaining population of Arctic charr was known to remain in the Lake Saimaa system, namely in Lake Kuolimo, and a restocking programme was initiated based on that stock. There were only a small number of individuals remaining and the hatchery-reared fish suffered from high egg and alevin mortality and disease susceptibility, which was suspected to be because of a lack of genetic variation in the broodstock (Prammer *et al*., 1999).

Stocking environmentally or genetically altered fishes thus represents a serious threat to the genetic integrity of wild populations (reviewed by Hindar *et al.*, 1991), notwithstanding that fish farmers now generally select broodstock to minimise genetic ‘pollution’ of wild stocks (Utter, 1998; Doyle *et al.*, 2001). In terms of genetics, the most critical period of interaction between stocked and wild populations is probably during reproduction, as reproductive success determines the extent of gene flow. The reproductive success of stocked fishes will also influence the potential for competition between wild, stocked and hybrid offspring. In addition, the aggregation of stocked and wild fishes during the spawning period could increase the risks of transmitting parasites and diseases. All of these interactions can influence the genetic structure of wild populations, either directly through gene flow or indirectly through altered selection or reduced population sizes (reviewed by Waples, 1991). Reproductive interactions are thus key to understanding the genetic threats posed by artificially cultured fishes to wild populations.

The three major threats to genetic diversity within species are extinction, hybridization and loss of local genetic adaptations. Extinction results in the complete loss of genes or gene combinations. The extinction of species or populations following stock enhancement programmes is primarily caused by competition with, or predation on, wild stocks. Such effects can occur through reductions in population size or alterations in the selective regime (e.g. through competition or predation) experienced by local populations (Billington & Hebert, 1991).

Hybridization represents a substantial threat to the genetic integrity of populations of several fish species. For example, Atlantic salmon and brown trout can produce viable hybrids and F2 and back-cross progeny (Alm, 1955; Piggins, 1965, 1966; Nygren *et al*., 1975), as can many other congeneric species (Chevassus, 1979; Verspoor & Hammar, 1991), although it should be noted that most F1 Atlantic salmon × brown trout hybrids are sterile. Intraspecific hybridization does not necessarily result in losses of individual genes but can rearrange gene combinations, which may lead to a loss of phenotypic adaptations to local environments (*cf*. Hindar *et al.*, 1991). This is a particular threat to salmonid populations because of the large number of releases associated with intentional stocking, ranching and escapes from aquaculture facilities (Ryman & Utter, 1987; Allendorf & Leary, 1988; Hindar *et al.*, 1991; Waples, 1991; Youngson *et al*., 1991; Carvalho & Cross, 1998; Fleming *et al*., 2000; Ferguson, 2003; Youngson *et al*., 2003). As a consequence of accidental releases, farmed Atlantic salmon have been recorded in substantial numbers in high-seas and coastal fisheries throughout the eastern Atlantic (Webb & Youngson, 1992; Hansen *et al*., 1993). In addition, escaped Atlantic salmon may enter rivers to spawn, sometimes in large numbers (e.g. in the River Polla, Scotland; Webb *et al*., 1991), and their progeny have been recorded in high frequencies (Webb *et al*., 1991, 1993; Lura & Økland, 1994; Clifford *et al*., 1997), with high frequencies of hybrids being detected in some rivers (Mathews *et al*., 2000). If genetic differences are too large, negative effects on fitness, known as outbreeding depression, may occur (Templeton *et al*., 1986; Lynch, 1991). Outbreeding depression resulting from loss of local adaptations can occur whenever there are genetic differences between populations (see below).

Adverse genetic and ecological impacts on wild Atlantic salmon populations resulting from releases or escapes of artificially propagated stocks have been reported in a number of countries, including Norway, Scotland, Ireland and Canada (Hearn & Kynard, 1986; Beall *et al*., 1989; Heggberget *et al*., 1993; Jones & Stanfield, 1993; Gross, 1998). Impacts on wild fish have included reductions in genetic diversity and capacity to evolve, introductions of genetic maladaptations as a result of interbreeding with artificially propagated individuals, and competition for food and space with hatchery stocks (Einum & Fleming, 1997; Gross, 1998). Ultimately, stocked or escaped Atlantic salmon are considered to have the potential to impact on the productivity, local adaptations and genetic diversity of wild populations (McGinnity *et al*., 1997, 2003; Fleming *et al*., 2000). By contrast, the magnitude of genetic introgression that results from stocking farm-reared brown trout is highly variable, unpredictable and apparently unrelated to the scale of the stocking (Ferguson, 2007). Where the relative numbers of stocked and native brown trout were estimated, the genetic impacts were much less than expected from equivalent survival levels. For example, Hansen (2002) recorded only 6% introgression in a population where the expected genetic contribution by farm-reared brown trout, based on the number of stocked fish and assuming equal survival and reproduction of native and stocked fish, was 64%. However, introgression has generally been greater in resident brown trout than in sea trout when rivers are stocked with farm-reared brown trout (Hansen *et al.*, 2000; Ruzzante *et al.*, 2004), and similar results have been obtained for other salmonid populations (LeClair *et al.*, 1999; Utter, 2001; Englbrecht *et al.*, 2002; Small *et al.*, 2004; Piller *et al.*, 2005).

Interspecific hybridization is also of great concern (Youngson *et al.*, 1993; Fleming *et al.*, 2000; Hansen, 2002; Verspoor *et al*., 2005). For example, evidence from rivers in western and northern Scotland suggests that farmed female Atlantic salmon spawn with brown trout more frequently than do their wild conspecifics, with the progeny of farmed female salmon containing up to 10% F1 hybrids (Youngson *et al*., 1993). The incidence of F1 hybrids in Norwegian rivers situated close to salmon farms increased three-fold following expansion of the aquaculture industry (Hindar & Balstad, 1994). This may be indicative of a breakdown in reproductive isolation between Atlantic salmon and brown trout, which could ultimately lead to gene introgression (Verspoor, 1988; Garcia de Leániz & Verspoor, 1989). Similarly, in Ireland, the third largest producer of farmed salmon (10 000 t in 2008) in Europe, the incidence of hybridization has increased in rivers in close proximity to farms (Matthews *et al*., 2000). It should be noted, however, that most F1 Atlantic salmon × brown trout hybrids are sterile.

Welcomme (1988) suggested that the stresses associated with stock enhancement may lead to a breakdown in normal behaviour and the formation of hybrids between species and even genera that do not normally hybridise in the wild. The potential for this could be further increased by degradation or loss of spawning areas. Habitat loss could reduce the areas suitable for spawning and induce a breakdown of normal reproductive isolating mechanisms. Irrespective, the problem is serious because of a potential loss of genetic fitness in wild populations (Hindar *et al.*, 1991; McDowell, 2002; McGinnity *et al.*, 2003). Repeated interactions between stocked and wild fishes result in lowered fitness, causing cumulative fitness depression and potentially an extinction vortex in vulnerable populations (McGinnity *et al.*, 2003).

The preservation of genetic variations within wild populations is particularly important for the maintenance of ecological fitness and function, and the ability to adapt to environmental changes (Jørstad *et al.*, 1999), as the fitness and adaptability of organisms are largely determined by genetic factors (O'Connell & Wright, 1997; Taniguchi, 2003). However, stock enhancement has the potential to replace local, adapted stocks with more homogeneous stocks from hatcheries, thereby limiting the sustainability of the species in the wild (Waples & Drake, 2004). Even though fishes have been artificially reared and released into the wild for over a century, relatively few experiments addressing the genetic effects of such releases have been carried out. The genetic effects following releases of farmed fishes may be positive, neutral or negative. If wild populations are highly inbred, positive genetic effects could occur because of limited gene flow from stocked fish. Neutral genetic effects would be expected where stocked fish are not genetically different from wild fish. No direct genetic effects would occur where hatchery-reared fish do not reproduce. By contrast, negative genetic effects would be expected after introgression of farmed fish into locally adapted, wild populations. Current knowledge suggests that negative genetic effects would be the most likely to occur in salmonids, as wild populations are typically genetically distinct (Allendorf & Utter, 1979; Ryman & Utter, 1987) and adapted to their local environment (Ricker, 1972; Taylor, 1991). Indeed, Reisenbichler (1988) presented strong evidence for local adaptation in coho salmon, demonstrating a negative association between both the recapture rates of transplanted fish relative to local stocks, and the geographical distance the transplanted fish had been transferred from their natal stream. Similar results have been obtained for Atlantic salmon (e.g. Ritter, 1975; Hansen *et al*., 1989). For many traits, hybrids between native and non-native stocks are intermediate between the parental groups (Bams, 1976; Brannon, 1982; Hemmingsen *et al*., 1986), indirectly indicating local adaptations.

Studies of juvenile salmonids in fresh water have shown that survival is often higher for wild fish than for released farmed fish (Schuck, 1948; Leider *et al.*, 1990). A direct genetic basis for reduced juvenile survival was indicated in a study of rainbow trout, where survival was higher for native fish than for non-native farmed fish and native × farmed hybrids (Reisenbichler & McIntyre, 1977). Other studies have shown that genetic changes occur in hatchery-propagated salmonids that would be expected to reduce their performance in the wild. For example, changes that affect swimming stamina (Green, 1964) and territorial (Norman, 1987) and concealment behaviour (Vincent, 1960). There have been fewer studies of post-smolts and sub-adults at sea, but the return rates of hatchery-produced fish are typically lower than those of wild fish, and the return rates of transplanted and crossbred stocks are generally lower than those of native populations (Ricker, 1972; Brannon, 1982; Bailey, 1987; Garcia de Leániz *et al.*, 1989). This could be related to either a lower ocean survival or increased straying rate of non-native fish, or both, and could have both genetic and phenotypic causes. Furthermore, disease resistance often differs between wild and stocked fish, invariably being greater in wild individuals, unless the disease itself was introduced (Hemmingsen *et al.*, 1986; Johnsen & Jensen, 1991).

As mentioned previously, the ever-expanding aquaculture industry is likely to lead to increases in the numbers of fishes released (Nash & Kensler, 1990; Riggs, 1990; Hindar *et al.*, 1991; Milner & Evans, 2004; Cowx *et al*., 2008; Bostock *et al*., 2010). Thus, the threat of hybridization (and introgression) with farmed fishes to the genetic integrity of wild populations is a major concern. It follows, therefore, that programmes involving the release of fishes should aim to minimise any genetic changes and conserve genetic resources (Carvalho, 1993; Ryman *et al*., 1995). Considering the limited knowledge of the selective significance of specific genes or gene combinations in natural populations, stocking practices must be essentially non-specific, although with an emphasis on maximising allelic diversity and the associated variance in ecologically-significant traits. Busack & Currens

(1995) and Lynch (1996) emphasised that although genotypic traits that do not closely relate to obvious fitness characters (i.e. molecular variation) can be measured with relative ease, it is much more difficult to assess genetic differences within or among populations in terms of quantitative traits. It is the latter, however, that determines fitness variation in physiological, morphological or behavioural characters, but understanding their control typically requires elaborate breeding experiments in controlled environments. Indeed, it is the poor understanding of the link between molecular variation and fitness parameters that is the obstacle in assessing genetic risks in stocking and introduction practices.

There is a theoretical, positive relationship between genetic variability and the ability to adapt to natural and anthropogenic changes in the environment. Similarly, a crucial factor determining the ecological persistence of stocked or introduced fishes is the association between genetic variability and population sizes, especially in founder stocks (introductions) or broodstocks. It is a lack of distinction between the direct effects of fish releases on population sizes and associated effects on genetic structure that has led to inadvertent, and sometimes irreversible, losses of genetic (and biological) diversity. There is thus a need to develop strategies that will minimise the genetic effects of fish stocking and introductions on wild populations; this is a fundamental issue that has been ignored in the past, but which must now be considered integral to the formulation of all stocking programmes. Actions to minimise potential genetic impacts are discussed in Section 4.3.4.

Summary of potential risks to fish populations – Genetic issues

- Stocks exhibit genetic variation that is manifest as differences in growth potential, age at maturity, fecundity, and can have implications for coevolution and adaptation processes – *risk medium /high depending on species*.
- Stocks exhibit adaptation towards particular environments and stocking could lead to loss of fitness – *risk medium /high*.
- Stocking may result in genetic drift and dilution of gene pool – *risk medium /high;* loss of genetic diversity – *risk medium /high*; and hybridisation – *risk medium/high*.

3.3.6 Parasites and diseases

A major risk of introducing or stocking fishes is the accidental introduction of non-native parasites and diseases, or an increased prevalence or intensity of native parasites and diseases caused by stocking fish with elevated pathogen loadings (Boxshall & Frear, 1990; Bauer, 1991; Kennedy, 1993; Cowx, 1994 a, b and c, 1998b; Gozlan *et al*., 2005; Thrush & Peeler, 2006; El-Rashidy & Boxshall, 2009; Peeler & Thrush, 2009; Hershberger *et al*., 2010; Taylor *et al*., 2010a; Peeler & Feist, 2011). As such, considerable efforts are currently focusing on reducing the possibility of species introductions, and mitigating any negative impacts when invasions occur (Peeler & Thrush, 2004; Peeler *et al*., 2004, 2007, 2009; Copp *et al*., 2005, 2009a; Stentiford *et al*., 2010; Taylor *et al*., 2010b, 2011; Tricarico *et al*., 2010). Although some parasites are host-specific, many are capable of infecting a wide range of host species. An example includes *Myxobolus* (syn. *Myxosoma*) *cerebralis* Hofer, the cause of whirling disease in rainbow trout, which is normally a harmless parasite of brown trout. Thus, it is possible that stocking or introducing fishes could infect native salmon, trout, charr, whitefish or lampreys with previously absent parasites and diseases. For example, important protozoan pathogens in wild fishes in Loch Fad, western Scotland, were probably introduced with stocked fish from a cage culture facility (McGuigan & Sommerville, 1985). The risks are dependent upon the relative disease status of the stocks and the condition of the fish farm supplying the stocking material. It is essential, therefore, that strict health-check regulations, such as those adopted by the Marine Scotland Fish Health

Inspectorate (MSFHI) or the EA under Section 30 of the Salmon and Freshwater Fish Act 1975, are enforced to minimise any undue health risks to native salmon, trout, charr, whitefish and lampreys. The role of the MSFHI in checking and monitoring fish health and the registration of hatchery facilities is described on the Scottish government webpage (http://www.scotland.gov.uk/Topics/marine/Fish-Shellfish/FHI).

The introduction and translocation of parasites and diseases can occur via fishes not intended for release or through indiscriminate or planned stocking. For example, the nematode *Anguillicoloides* (syn. *Anguillicola*) *crassus* Kuwahara, Niimi & Itagaki was introduced into Europe with oriental eels (*Anguilla* spp.) intended for human consumption, and has spread rapidly across Northern Europe (Kennedy & Fitch, 1990; Evans & Matthews, 1999; Costa-Dias *et al*., 2010; Kangur *et al*., 2010; Lefebvre *et al*., 2011). Similarly, the cestodes *Khawia sinensis* Hsü and *Bothriocephalus acheilognathi* Yamaguti were introduced to the UK with consignments of carp, despite strict health regulations on fish movements (Andrews *et al*., 1981; Chubb & Yeomans, 1995; Yeomans *et al*., 1997). Infectious dropsy of cyprinids was also spread throughout continental Europe with carp (from the former Yugoslavia). *Yersinia ruckeri* Ewing, Ross, Brenner & Fanning, the causative agent of enteric redmouth disease in parts of northern Europe, was introduced with uncontrolled shipments of fathead minnow *(Pimephelas promelas* (Rafinesque)) from North America (Michel *et al*., 1986). Furthermore, the pathogens of several fish diseases, including infectious pancreatic necrosis (IPN), infectious haematopoietic necrosis (IHN) and bacterial kidney disease (BKD), can be transmitted via gametes, so unfertilised eggs, sperm and embryos, as well as adult fishes, are all potential vectors. In excess of 100 parasites and diseases are now known to have been introduced to Europe with consignments of fishes (see Cowx *et al*., 2007).

A major impact of the North American signal crayfish, which can be introduced with fish consignments (Section 3.3.7), in Europe has been as a vector of the crayfish plague fungus (*Aphanomyces astaci* Schikora), which has caused large-scale mortalities amongst indigenous crayfish populations, particularly in England (Holdich & Reeve, 1991; Alderman 1997; Holdich *et al*., 2009). European crayfish species have no resistance against crayfish plague and, therefore, experience total mortality. By contrast, North American crayfish species have co-evolved with the disease and developed defence systems, making them a natural host and vector of the fungus (Evans & Edgerton, 2002). A large proportion of North American signal crayfish are carriers of the crayfish plague fungus. Thus, where North American signal crayfish populations become established, crayfish plague is also likely to become established. Consequently, the spread of North American signal crayfish is a serious threat to indigenous crayfishes in Europe. Indeed, the Global Invasive Species Programme listed the crayfish plague fungus among the ‘World’s Worst Invasive Alien Species’. Once present, fungal spores infest susceptible individuals and can ultimately infect entire populations. Other vectors of crayfish plague include birds or mammals moving between infected and uninfected water bodies, spores being transported by boats (e.g. on hulls or in bilges) or fishing gear (e.g. nets and waders), and transfers of contaminated water. In addition, North American signal crayfish may be introduced with consignments of fish (Section 3.3.7). Indeed, there are several locations where this could have occurred in Scotland, including the River Nairn, the River Shee and a number of stillwater fisheries (i.e. in the East Lothian Tyne catchment), and at least one of the farms that have been used to widely stock fishes in Scotland is now infested with North American signal crayfish (although it is unknown whether it was infested when stocking was occurring; C.W. Bean, *pers. comm*.). It should be noted that no crayfish species naturally occur in Scotland, although there are two populations of white-clawed crayfish (*Austropotamobius pallipes* (Lereboullet)) that were introduced more than 50 years ago (Bean *et al*., 2004).

Many diseases that infect hatchery-reared salmonids, and which now occur in the wild, were originally imported (Peeler *et al*., 2011). For example, rainbow trout from western North

America are thought to have been the vector for the introduction of furunculosis to Europe and South America (Snieszko, 1973). Similarly, furunculosis was probably introduced into the UK with brown trout from Denmark, spread throughout the country following movements of farmed trout (Pillay, 1992), and subsequently imported to Norway with Atlantic salmon smolts from Scotland (Egidius, 1987). In addition, Atlantic salmon populations in Norway have suffered massive mortalities and, in some areas, total eradication caused by the monogenean trematode *Gyrodactylus salaris* Malmberg, which was introduced with infected Atlantic salmon from Sweden (Johnsen & Jensen, 1991; Mo, 1992; Munday *et al*., 1992; Pillay, 1992; Bakke *et al*., 2002). Although there is no evidence for the presence of *G. salaris* in the UK, efforts should be made to reduce the possibility of it being imported with consignments of fishes (Shinn *et al*., 2001; Peeler *et al*., 2004).

These examples illustrate the risks of introducing and dispersing parasites and diseases as a result of stocking and introducing fishes. It should also be recognised that the high stocking densities in many aquaculture facilities may increase fish stress, potentially suppressing immunity and leading to outbreaks of disease, which may then become a source of transmission to wild stocks in the vicinity (Liao *et al*., 2003; Bartley *et al*., 2006). The high concentrations of wild animals in the vicinity of many aquaculture facilities may further increase the possibility of transmitting diseases, or facilitate the life cycles of parasites by providing intermediate hosts. The spread of parasites and diseases is of relevance to environmental protection and may exact high ecological and economic costs (Munday *et al*., 1992; Pillay, 1992), although no information exists concerning the latter. The challenge that faces regulators is that of minimising the introduction and spread of parasites and diseases, but this is fraught with difficulties. As described previously, *K. sinensis* has spread throughout the UK since it was introduced, with its known range coinciding almost exactly with that of its host, the common carp (Chubb & Yeomans, 1995). The rapid spread of *K. sinensis* with movements of carp from the Far East, through Russia to Western Europe demonstrates the limitations of regulatory mechanisms across Europe, including the UK, in preventing its dispersal, and the challenges facing regulators is minimising the introduction and spread of parasites and diseases.

Summary of potential risks to fish populations - Parasites and diseases

- High stocking densities in many aquaculture facilities may increase fish stress, potentially suppressing immunity and leading to outbreaks of disease, which may then become a source of transmission to wild stocks – *risk variable*

3.3.7 Spread of non-native organisms with stocked fishes

Whilst the movement of parasites, pathogens and diseases can occur via fishes released or through indiscriminate or planned stocking, there is also the risk of other biota 'piggy-backing' in the consignment for stocking. For example, movements of contaminated batches of fishes for fisheries purposes allowed topmouth gudgeon (*Pseudorasbora parva* (Temminck & Schlegel)) to become the most invasive fish species in Europe, and which now has a pan-European distribution (Gozlan *et al*., 2010). Similarly, North American signal crayfish may also be moved to water bodies with consignments of fish. There are several locations where this could have occurred in Scotland, including the River Nairn, the River Shee and still water fisheries (i.e. in the East Lothian Tyne catchment) (C.W. Bean, *pers. comm*.).

This transfer of passengers with the target species for stocking can occur when the stocked fish are introduced into their intended water body. However, unintended introductions can occur when water is exchanged during transportation of the stock, and eggs and juveniles of the unintended biota escape with the water change.

Summary of potential risks from spread of non-native organisms with stocked fishes

- Care must be taken to ensure that other non-native, non target organisms are not introduced as part of the stock enhancement programme - this includes due diligence when exchanging water during transportation and when the fish are stocked into the receiving water body – *risk medium*.

3.3.8 Fishing pressure and mortality

Fishery enhancement through stocking has the purpose of either sustaining fisheries at their current levels of exploitation or expanding the fisheries to support greater levels of fishing pressure. A potential hazard with this is that increased fishing pressure may result in higher mortality of wild fish, either through higher direct fishing mortality or through increased mortality caused by catch-and-release stress (Cooke & Schramm, 2007; Arlinghaus *et al*., 2009). If fishing mortality exceeds the natural productivity of the system, impacts on the viability of wild populations through a reduction in the size of the spawning stocks may occur. Furthermore, there is a risk that increased fishing pressure on stocked fish may lead to an increase in the bycatch of charr and/or whitefish (such as observed with the introduced whitefish population in Carron Valley Reservoir; C.W. Bean, *pers. comm*), with concomitant increases in fishing mortality. Policies to balance appropriate stocking strategies to the conditions in receiving water bodies provide scope to mitigate these types of hazards, although to what extent this occurs is uncertain. **Where vulnerable or rare species are restricted to a small number of designated sites, as are the whitefish, the precautionary principle should apply, such that stocking or introduction of fishes, including trout, should be prohibited or restricted to safeguard populations.** It is recognised, however, that this could be difficult where vulnerable or rare species were introduced to an existing fishery that was maintained by stocking (e.g. whitefish in Carron Valley Reservoir). In addition, care should be taken to ensure that increases in the presence of anglers do not disturb populations of breeding birds and other fauna.

There are several indirect pressures exerted by enhancing fisheries. These include:

- increased presence of avian predators such as cormorants associated with the timing of stocking to take advantage of the elevated food resources;
- increased angling pressure around the time of stocking, which may coincide with sensitive breeding times of birds and other wildlife, or more generally disturb wildlife through, for example, trampling of vegetation.

Summary of potential risks to fish populations - fishing pressure and mortality

- Increased fishing pressure may result in higher mortality of wild fish, either through higher direct fishing mortality or through increased mortality caused by catch-and-release stress - *risk low/medium.*
- Stocking may encourage increased fishing pressure that can indirectly lead to disturbance of wildlife - *risk low/medium*.

3.4 Threats to protected invertebrates and macrophytes in designated sites

It is well documented that fishes may preferentially prey upon certain invertebrate taxa over others (e.g. Maitland, 1965; Mann & Orr, 1969; Pedley & Jones, 1978; Nunn *et al*., 2007b, 2012). In addition, the North American signal crayfish can exert significant predation pressure on aquatic invertebrates and macrophytes (Guan & Wiles, 1997; Nyström, 1999, 2002; Lewis, 2002; Peay *et al*., 2009). This may be problematic should the prey be designated species or assemblages. Alternatively, the presence of fishes could cause a shift in the diets of other species to feed upon designated animals or plants. The various direct and indirect impacts of fish predation upon ecosystem functioning as a whole have been discussed in Section 3.1. This section discusses whether stocking or introducing fishes poses a threat to designated invertebrate assemblages or species, or macrophyte communities, and identifies which species might be at risk, and under which circumstances.

3.4.1 Invertebrates

Invertebrate species that are protected under Schedule 5 of the Wildlife and Countryside Act (1981 and as amended) and that occur in Scottish fresh waters are the white-clawed crayfish, freshwater pearl mussel (*Margaritifera margaritifera* (L.)) and medicinal leech (*Hirudo medicinalis* L.). The freshwater pearl mussel and white-clawed crayfish are also listed on Annex 2 of the Habitats Directive. There are 19 Special Areas of Conservation for pearl mussel in Scotland. Unless stated otherwise, the information used in this section was obtained from the JNCC and UK BAP websites (http://www.jncc.gov.uk/ and http://www.ukbap.org.uk/).

The favoured habitats of the medicinal leech are unlikely to be targeted for fish introductions, although the possibility of migration of stocked fishes from their site of release elsewhere should be acknowledged. By contrast, the white-clawed crayfish and freshwater pearl mussel favour clean, well-oxygenated streams, rivers and lakes, and often coexist with native brown trout and salmon (Hastie *et al*., 2003; Holdich, 2003; Skinner *et al*., 2003). Indeed, the freshwater pearl mussel requires salmonid hosts for its glochidia larvae (Hastie & Cosgrove, 2001; Hastie & Young, 2003; Geist *et al*., 2006; Taeubert *et al*., 2010). Although the white-clawed crayfish is widespread in England and Wales, many populations have been lost since the 1970s as a result of crayfish plague, competition from exotic crayfish species, habitat modification and pollution (Holdich & Reeve, 1991; Holdich & Rogers, 1997; Bubb *et al*., 2005, 2008; Holdich *et al*., 2009). Similarly, the freshwater pearl mussel has suffered from poor water quality, habitat degradation, flow regulation, fisheries management and over-exploitation (Cosgrove *et al*., 2000; Hastie *et al*., 2000; Bolland *et al*., 2010). Considering the current distributions of each of the above invertebrate species, the freshwater pearl mussel is the most likely to be at risk from fish stocking and introductions in Scotland. The white-clawed crayfish does not naturally occur in Scotland, although two introduced populations exist (Holdich, 2003; Bean *et al*., 2004). Large fishes could potentially predate upon mussels, while a range of indirect effects may also occur (Sections 3.1 & 3.2). **Stocking of fishes, particularly non-native species, should therefore be restricted or prohibited where key populations of freshwater pearl mussel exist.** Consideration should also be given to the possibility of migration of stocked fishes from their site of release into tributaries that may be important for the freshwater pearl mussel.

Lists of conservation priority species can be found on the UK Biodiversity Action Plan website (http://www.ukbap.org.uk/). Excluding those included in the Wildlife and Countryside Act, priority invertebrates that occur in Scottish fresh waters are one species of diving beetle (*Hydroporus rufifrons* (Müller)), one species of reed beetle (*Donacia aquatica* (L.)), one species of stonefly (*Brachyptera putata* (Newman)) and one species of cranefly (*Rhabdomastix laeta* (Loew)). However, these species occupy microhabitats not generally exploited by fish. For example, *H. rufifrons* occurs in extremely shallow, temporary pools in unimproved pasture, while the larvae of *D. aquatica* inhabit aquatic weedbeds. *Brachyptera* *putata* and *R. laeta* are mostly confined to running water, and so could be affected by fish stocking and introductions.

3.4.2 Macrophytes

Macrophyte species that are protected under Schedule 8 of the Wildlife and Countryside Act (1981 as amended) and macrophyte species that occur in Scottish fresh waters are pigmyweed (*Crassula aquatica* (L.)), floating water plantain (*Luronium natans* (L.)), slender naiad (*Najas flexilis* (Willd.)) and bearded stonewort (*Chara canescens* Desv. & Lois.). Floating water plantain and slender naiad are also listed on Annex 2 of the Habitats Directive.

Pigmyweed is known only from the River Shiel, where it occurs in shallow waters or in wetlands and vernal pools, including bare mud when water levels recede. Floating water plantain occurs in a range of freshwater situations, but thrives best in areas where growth of emergent vegetation is restricted (Willby & Eaton, 1993; Lansdown & Wade, 2003; Bazydło, 2004). Its distribution is localised in the UK, with recent records from Wales, the West Midlands and northern England, but it also occurs as an introduction to ditches in the Norfolk Broads and a few localities in Scotland. Slender naiad occurs in deep, often coloured or turbid water in mesotrophic lakes (Wingfield *et al*., 2005). It is seldom found in water less than 1 m in depth. In the UK, this species is found exclusively in Scotland, with many sites on islands off the west coast, as well as important populations elsewhere in Argyll, Perthshire and Highland. Bearded stonewort is a species of clear, brackish water up to 2.5 m deep in lagoons, lakes and pools by the coast. It usually prefers sites in the 4-20 ‰ salinity range, although its English sites are unusual in being inland and very low salinity (<1 ‰). In the UK, bearded stonewort is restricted to three sites near Peterborough, Cambridgeshire, and one site in the Outer Hebrides (Loch Mor, Baleshare).

In addition to those listed in the Wildlife and Countryside Act, priority macrophyte and charophyte species that occur in Scottish fresh waters are Baltic stonewort (*Chara baltica* Bruz.), lesser bearded stonewort (*Chara curta* Braun), mossy stonewort (*Chara muscosa* Groves & Bullock-Webster), slender stonewort (*Nitella gracilis* (Smith)), tassel stonewort (*Tolypella intricata* (Trent.)), bird's nest stonewort (*Tolypella nidifica* (Müller)), great tassel stonewort (*Tolypella prolifera* (Ziz)), marsh clubmoss (*Lycopodiella inundata* (L.)), pillwort (*Pilularia globulifera* L.), grass-wrack pondweed (*Potamogeton compressus* L.) and Shetland pondweed (*Potamogeton rutilus* Wolfg.)

Trout rarely consume aquatic plant material, so herbivory is an unlikely impact of trout stocking, although other fish species (e.g. carp, grass carp, tench, rudd (*Scardinius erythrophthalmus* (L.))) do consume vegetation (Cross, 1969; Stott, 1977; Williams *et al*., 2002; Tomec *et al*., 2003). Indeed, grass carp have been used to control excessive growths of aquatic vegetation (Cross, 1969; Stott, 1977). As mentioned previously (Section 3.1), trout are not known to alter physical habitat and so, under normal circumstances, it appears unlikely that introduction of trout *per se* would significantly damage aquatic vegetation although, again, other species (especially carp) can cause substantial damage. Threats common to the majority of the above aquatic plant species include habitat loss, land drainage, changes in management regimes, pollution and afforestation. However, the most prevalent threat appears to be nutrient enrichment. Thus, the most likely impacts of stocking fishes, especially trout, on aquatic plants are indirect, such as increases in nutrient loading and/or shifts in ecosystem functioning. Elevated fish biomass, together with mobilisation of nutrients and increased turbidity, could potentially contribute towards eutrophication, the symptoms of which are most frequently observed in aquatic plant communities (Carvalho & Moss, 1995). The potential consequences of shifts in ecosystem functioning or increases in nutrient loading are discussed in Sections 3.1 & 3.2, respectively, and so are not addressed further here.

The species known to occur in Scotland that are most likely to suffer from the stocking or introduction of fishes are those that inhabit areas potentially suitable for fish stocking and introduction, namely floating water plantain, slender naiad, bearded stonewort, Baltic stonewort, lesser bearded stonewort, mossy stonewort, slender stonewort, bird's nest stonewort, grass-wrack pondweed and Shetland pondweed. The habitats favoured by tassel stonewort and great tassel stonewort are unlikely to be targeted for fish stocking or introduction (e.g. shallow, overgrown habitats such as drainage ditches, marshes and ephemeral pools). In addition, a number of the species (e.g. pigmyweed, marsh clubmoss, pillwort) inhabit the drawdown zones of ponds and lakes, and so are likely to coexist with fishes only periodically. Detailed information on each of the plant species can be obtained from the JNCC and UK BAP websites (http://www.jncc.gov.uk/ and http://www.ukbap.org.uk/).

Another issue that should be considered is the indirect impact of fish stocking and introduction upon vegetation via increased angling pressure. Similar to trampling and grazing by livestock, angling has the potential to impact upon aquatic vegetation as a result of trampling by anglers and clearance of fishable areas (Linton & Goulder, 2000; Goulder, 2001). Ironically, in the absence of appropriate management, the effects of angling may be beneficial as there is often an increase in aquatic vascular plant (macrophyte) species richness in ponds that are used for angling, at least partly caused by disturbance. **Where protected plant species are restricted to a small number of sites, introducing and, to a lesser extent, stocking fishes, especially herbivorous species but including salmonids, should be prohibited or restricted to safeguard populations.**

Summary of potential risks to designated invertebrates and macrophytes

- Most invertebrate species indentified do not occupy habitats stocked with trout.
- Habitats favoured by protected macrophytes unlikely to be targeted for fish stocking, although this possibility may be higher for stocking of coarse fish – *risk low*.
- Trout rarely consume macrophytes but coarse fish consume soft-fleshed macrophytes and carp displace rooted vegetation – *risk low*.

3.5 Case studies

A number of designated natural heritage sites are managed as trout fisheries, including Loch Leven (National Nature Reserve [NNR], Ramsar site, SPA and SSSI), Lake of Menteith (SAC and SSSI), Butterstone Loch (SPA and SSSI) and Lindores Loch (SSSI) in Scotland, and Esthwaite Water (Ramsar site and SSSI) and Chew Valley Lake (SPA and SSSI) in England. Loch Leven and Chew Valley Lake were selected as case study sites because of their long-term management as trout fisheries and status as SPAs. Esthwaite Water was selected as it hosts an established salmonid farm, and has been extensively studied by the Freshwater Biological Association (FBA). Less published information appears to exist for Lake of Menteith, but it was nonetheless chosen as a case study site because of its status as a SAC and its management as a trout fishery. In addition, Loch Lomond (NNR, Ramsar site and SPA) was selected because of its importance as a natural heritage site. As mentioned above, however, **without precise information on stocking activities and other management techniques, it is not possible to predict the likely impacts on ecosystem functioning and trophic status of specific water bodies.** Moreover, **the impacts of particular management techniques will be site-specific, because of the inherent differences in ecosystem dynamics between water bodies.** Similarly, it is not possible to determine nutrient budgets without site-specific data (see Johnes *et al*., 1996; Moss *et al*., 1996). Thus, before nutrient fluxes can be modelled, appropriate data should be sought from the relevant authorities (e.g. Scottish Environment Protection Agency (SEPA). **For designated natural heritage sites, the precautionary principle should apply, such that introduction and, to a lesser extent, stocking of fishes, including trout, should be prohibited or restricted to safeguard the conservation interests of such water bodies.** Unless stated otherwise, the information on Loch Lomond, Butterstone Loch, Lindores Loch, Esthwaite Water and Chew Valley Lake was obtained from the JNCC website (http://www.jncc.gov.uk/).

3.5.1 Loch Lomond

Loch Lomond is a large (7027 ha), deep (max. depth 191 m) lake located near Glasgow, west-central Scotland. The loch is the largest area of fresh water in Britain, the second largest in volume (2652 × 10^6 m^2), and the third longest (36.3 km) and deepest. It comprises an oligotrophic north basin (2280 ha surface area, 140 m mean depth), a mesotrophic middle basin (1410 ha surface area, 60 m mean depth) and a eutrophic south basin (3350 ha surface area, 27 m mean depth) (Winfield *et al*., 2011).

Loch Lomond is designated as a NNR, Ramsar site and SPA, and the surrounding woodland is designated as an SAC and SSSI. The Ramsar site and SPA consist of a marshy area around the lower reaches of the River Endrick and four wooded islands in the loch, with the loch shore comprising low-lying, regularly flooded wetlands, woodland fringes and rough pasture. The floodplain mire consists mainly of eutrophic-mesotrophic swamp communities, dominated by reed-canary grass (*Phalaris arundinacea* L.), with sharp-flowered rush (*Juncus acutiflorus* Ehrh. ex Hoffm.), bladder-sedge (*Carex vesicaria* L.), water sedge (*Carex aquatilis* Wahlenb.) and common sedge (*Carex nigra* (L.)) also present. The shore zone of the islands is species rich and supports a variety of plants, including globeflower (*Trollius europaeus* L.), columbine (*Aquilegia vulgaris* L.) and goldilocks (*Ranunculus auricomus* L.). The site is also rich in invertebrates and supports a Red Data Book moth, the bulrush wainscot (*Nonagria typhae* (Thunberg)), and *Holopedium gibberum* Zaddach is present in the zooplankton (Pomeroy, 1994). In addition, the Endrick confluence supports internationally important numbers of Greenland white-fronted goose (*Anser albifrons flavirostris* (Scopoli)) and the islands are used by breeding capercaillie (*Tetrao urogallus* L.). Indeed, the site qualifies under Article 4.1 of the Directive (79/409/EEC) by supporting up to 1.5% of the breeding population of capercaillie and up to 1.1% of the wintering population of Greenland white-fronted goose in Britain.

The fish community of Loch Lomond is also of conservation importance. Indeed, a survey of 235 NNRs throughout Britain identified Loch Lomond as being of outstanding importance for fishes (Lyle & Maitland, 1992). The loch is one of only two sites in Scotland that supports natural populations of powan, and is the only site in Scotland to support a race of river lamprey that completes its entire life cycle to fresh water (Brown & Scott, 1994; Maitland *et al*., 1994); note a non-migratory population of river lamprey also exists in Lough Neagh (Goodwin *et al.,* 2006). Populations of brook and sea lamprey are also of particular note, and the loch has important salmon, sea trout and pike fisheries (Adams, 1994; Maitland *et al*., 1994). Although Loch Lomond has not been the target of stocking, in the last two decades a number of fish species have been introduced illegally (e.g. ruffe, bream, dace, gudgeon, crucian carp, chub [*Leuciscus cephalus* (L.)]), some of which have established self-sustaining populations (Adams & Maitland, 1991; Adams, 1994; Winfield *et al*., 2007, 2011). Of all of the introductions, that of the ruffe has been the most dramatic and concerning (Winfield *et al*., 2011). Declines in the endemic population of powan are thought to be partly due to the spread of ruffe, which may feed on their eggs (Ogle, 1998; Winfield *et al*., 1998; Etheridge *et al*., 2011). Indeed, Adams & Tippett (1991) calculated that ruffe accounted for 64% of powan egg predation.

3.5.2 Loch Leven

Loch Leven lies midway between the Forth and Tay estuaries in east-central Scotland. It is the largest naturally eutrophic loch in Britain and Ireland, with a surface area of 1330 ha and mean and maximum depths of 3.9 m and 25.5 m, respectively (Wright, 2003). The loch has suffered from eutrophication for more than a century, with blooms of cyanobacteria occurring every year, although there have been signs of improvement (e.g. reduced algal biomass, increased macrophyte biomass, increased invertebrate diversity) in the last three decades (Fozzard *et al*., 1999; Carvalho *et al*., 2012; Dudley *et al*., 2012; Gunn *et al*., 2012; May *et al*., 2012; Spears *et al*., 2012).

Loch Leven supports internationally important wintering populations of waterfowl (Carrs *et al*., 2012). The site qualifies under Article 4.1 of the Directive (79/409/EEC) by supporting populations of European importance of whooper swan (*Cygnus cygnus* L.) (up to 1.8% of the wintering population in Great Britain). The site also qualifies under Article 4.2 of the Directive (79/409/EEC) by supporting populations of European importance of pink-footed goose (*Anser brachyrhynchus* Baillon) (up to 8.1% of the wintering Eastern Greenland/Iceland/UK population) and shoveler (*Anas clypeata* L.) (up to 1.3% of the wintering Northwestern/Central Europe population). In addition, the area qualifies under Article 4.2 of the Directive (79/409/EEC) by regularly supporting at least 20 000 waterfowl, including goldeneye (*Bucephala clangula* (L.)), tufted duck (*Aythya fuligula* (L.)), pochard (*Aythya ferina* (L.)), teal (*Anas crecca* L.), gadwall (*Anas strepera* L.), cormorant, shoveler, pink-footed goose and whooper swan. During the breeding season, the area regularly supports black-headed gull (*Larus ridibundus* L.) (4% of the Great Britain population on average). As well as birds, Loch Leven supports the nationally rare invertebrates *Macroplea appendiculata* (Panzer), *Thanatophilus dispar* (Herbst) and *Saldula fucicola* (Sahlberg), and the nationally important plant species *Juncus filiformis* L. and *Hierochloe odorata* (L.).

Loch Leven supports a natural brown trout fishery (Thorpe, 1974a, b; Fozzard *et al*., 1999), as well as populations of perch, pike, minnow (*Phoxinus phoxinus* (L.)), three-spined stickleback (*Gasterosteus aculeatus* L.), stone loach (*Barbatula barbatula* (L.)), brook lamprey and eel (Stewart *et al*., 2005; Winfield *et al*., 2012). In addition, supplementary stocking of brown trout commenced in 1983 and of rainbow trout in 1993 (Wright, 2003). From 1998 to 2001, between 100 000 and 200 000 brown trout were stocked into the loch as fingerlings in spring, with approximately 30 000 rainbow trout stocked annually from March to August (Stewart *et al*., 2005). However, stocking no longer occurs in Loch Leven (Winfield *et al*., 2012).

The loch has a relatively high phosphorus load (~8 t yr^{-1}), with run-off from the land and waste from over-wintering waterfowl being the most important contributors (Bailey-Watts & Kirika, 1999; May *et al*., 2001, 2012). This, combined with its shallow depth and high water-retention time, makes the loch prone to algal blooms (May *et al*., 2001; Carvalho *et al*., 2004). Other consequences of eutrophication include a long-term decline in submerged macrophytes and invertebrate species diversity (Morgan, 1970; Jupp & Spence, 1977). Although stocking no longer takes place, it is likely that the long-term problem of eutrophication and stocking large numbers of fishes in the past impacted upon the ecosystem functioning and trophic status of the water body, as described in Section 3.3. In addition, the large numbers of waterfowl that congregate on the loch, especially during winter, may increase nutrient loading through guanotrophication.

3.5.3 Lake of Menteith

Lake of Menteith is a large (263 ha), mesotrophic kettle hole, located close to Loch Lomond and the Trossachs, west-central Scotland. The lake is designated for its aquatic macrophytes, which include a population of slender naiad (Fozzard *et al*., 1999). A number of fish species inhabit the lake, including roach, pike and brown trout, and a rainbow trout fishery has operated on the lake since 1967 (SEPA, 2002).

One of the main threats to slender naiad, as with many other plant species and ecosystem functioning in general, is eutrophication. In Lake of Menteith, approximately 50% of the summer total phosphorus load is derived from consented fish-cage discharge, and the lake suffers from severe blue-green algal blooms in the autumn (Fozzard *et al*., 1999; SEPA, 2002). Whereas many macrophyte and algae species are capable of utilising bicarbonate (often the predominant form of carbon in eutrophic waters) when carbon dioxide is scare, slender naiad is not. In such situations, the photosynthetic capacity of slender naiad is reduced, and the plant is outcompeted by other species (Wingfield *et al*., 2005). Furthermore, increasing production by the fish farm is continuing to increase nutrient loading in the lake (SEPA, 2002). Despite this, published information regarding the current ecology or management of Lake of Menteith is apparently sparse, although data presumably exist in the grey literature. It is likely, however, that the activities of the fish farm and stocking large numbers of fishes (~1000 per week) impact upon the ecosystem functioning and trophic status of the water body, as described in Sections 3.1 & 3.2. In addition, the site experienced a major fish kill in 2009 (C.W. Bean, *pers. comm.;* http://www.timesonline.co.uk/tol/news/uk/scotland/article6182383.ece).

3.5.4 Butterstone Loch

Butterstone Loch is a small (43.5 ha), shallow (max. depth 9 m) meso-oligotrophic lake located north of Perth, east-central Scotland. The loch has a diverse aquatic flora and an extensive area of fen, and formerly supported *Nitella* spp., *Chara* spp., *Isoetes* spp. and slender naiad, a benthic-dominated diatom community and diverse zooplankton (Bennion *et al*., 2010). In addition, it is one of a small number of sites in the UK to have been recolonised by ospreys, and there is a range of other breeding, and wintering, birds.

Butterstone Loch is designated as an SPA and SSSI. The loch has undergone major ecological changes in the last century, including increases in plankton production and the relative importance of diatoms associated with eutrophic conditions, and reductions in the diversity and changes in the composition of the zooplankton and aquatic plant communities, with nutrient enrichment most likely a major factor (Bennion *et al*., 2010). Nutrient enrichment dates back to *c*. 1900, with a further increase *c*. 1970; the most likely cause was intensification of agriculture in the catchment, with effluent and waste food from a fish farm being another source (Bennion *et al*., 2010). Indeed, between 1981 and 2004 there were six cages in the loch, stocked with 5.1 t of rainbow trout, with the input of phosphate from fish feed being ~70 kg yr^{-1} (Bennion *et al*., 2010). The cages were then removed in 2004 and the current consent allows 150-600 trout (brown and rainbow) to be stocked per week. The contribution of fish to the degradation of the loch has not been quantified, but the impacts of the fish cages, exacerbated by stocking, may have been substantial (Bennion *et al*., 2010).

3.5.5 Lindores Loch

Lindores Loch is a small (40.5 ha), shallow (max. depth 3.5 m) mesotrophic lake located near Perth, east-central Scotland. The loch has extensive charophyte beds and a number of *Potamogeton* spp., and formerly supported *Nitella* spp., *Chara* spp., *Ranunculus* spp., Nymphaeaceae, alternate water-milfoil (*Myriophyllum alterniflorum* DC.) and slender naiad, a non-planktonic diatom community and a diverse zooplankton community dominated by plant-associated taxa (Bennion *et al*., 2010). In addition, the freshwater transition mire is one of the most extensive and least disturbed in the area and, together with adjoining rich-fen and alder-willow carr, supports a diverse flora that includes a number of local rarities (Bennion *et al*., 2010). There is also a diverse breeding bird community that includes several regionally uncommon waterfowl species and one national rarity.

Lindores Loch is designated as a SSSI for its transition mire and breeding bird assemblages. The loch has undergone major ecological changes in the last century, including increases in plankton production and the relative importance of diatoms associated with eutrophic conditions, and reductions in the diversity and changes in the composition of the zooplankton and aquatic plant communities, with nutrient enrichment most likely a major factor (Bennion *et al*., 2010). The first major change occurred pre-1900, with another *c*. 1945, but the most significant change occurred *c*. 1970 when 48 000 rainbow trout, 4000 brown trout and 1500 brook trout (*Salvelinus fontinalis* (Mitchill)) were introduced (Bennion *et al*., 2010). It is believed that the fishery had consent to stock 500 brown trout per year and 200 rainbow trout per week (Bennion *et al*., 2010), although the loch has not been stocked for several years (R. Gardiner, *pers. comm.*). The fishery also had consent to control coarse fish (eel, perch, pike and carp) populations, with 600-1000 pike being removed annually (Bennion *et al*., 2010), although this no longer occurrs (R. Gardiner, *pers. comm.*). Although the contribution of past fish stocking to the current status of the loch has not been quantified, there is evidence from the zooplankton record that increased fish predation may have played a role (Bennion *et al*., 2010).

3.5.6 Esthwaite Water

Esthwaite Water is located between Windermere and Coniston Water in the Lake District, north-west England. The lake is 2.5 km long, approximately 0.5 km wide, with a mean depth of 6.4 m, and is one of the best examples of a mesotrophic lake in England and Wales. The complex of open water, fen and grassland communities supports a characteristic flora that includes examples of nationally rare and local species. Of particular note are the slender naiad and elongated sedge (*Carex elongata* L.), with Esthwaite Water the only known location of slender naiad in England and Wales, although it is now feared to be extinct at this site (Wade, 1994, cited in Wingfield *et al*., 2005). Over 120 invertebrate species have been recorded, including the uncommon water boatman *Sigara semistriata* (Fieber), the local caddis fly species *Cyrnus flavidus* (McLachlan), *Oecetis furva* (Rambur) and *Polycentropus kingi* (McLachlan), the rare cladoceran *Alonella exigua* (Lilljeborg) and, notably, the triclad *Bdellocephala punctata* (Pallas). In addition, the lake is of local importance for breeding birds, including great crested grebe (*Podiceps cristatus* (L.)), teal, tufted duck, red-breasted merganser, pochard and sedge warbler (*Acrocephalus schoenobaenus* (L.)).

Esthwaite Water's current condition is considered unfavourable due to eutrophication, which has resulted in a significant deterioration of the aquatic macrophyte flora. In spite of this, recent published information regarding the current ecology or management of the lake is apparently sparse, although data presumably exist in the grey literature. The single most important source of nutrients, especially phosphorus, to the lake is a fish farm. The farm produces approximately 100 tonnes (wet weight) of rainbow trout per annum (Hall *et al*., 1993, cited in Grey *et al*., 2004), with an estimated 150-300 kg of waste food and 250-300 kg (dry weight) of faeces introduced into the aquatic environment for every tonne of fish produced (Phillips *et al*., 1985, cited in Grey *et al*., 2004). Grey *et al*. (2004) investigated the fate of waste food from the fish farm at Esthwaite Water using stable isotope analyses, and demonstrated incorporation of pellet-derived material into the diets of planktonic and benthic communities. Moreover, after allowing for a number of trophic transfers, it was demonstrated that the predatory cladoceran *L. kindtii* also utilised pellet material, while roach were probably short-circuiting the food chain by directly consuming particulate pellet material, as well as through ingestion of zooplankton. Indeed, a simple two-source mixing model revealed that approximately 65% of *Daphnia* spp. and >80% of roach body carbon may be derived from pellet material in the plankton, and that chironomid larvae may incorporate >50% in the sediment environs.

3.5.7 Chew Valley Lake

Chew Valley Lake is located to the south of Bristol, south-west England. The lake is a shallow, productive, hard-water reservoir, with mean and maximum depths of 4.3 m and 11.5 m, respectively (Ibbotson & Klee, 2002). The reservoir was created in the early 1950s by damming the River Chew, and developed as a trout fishery (Wilson, 1971). It has a surface area of approximately 500 ha when full, but water levels can fluctuate widely, and a relatively small draw-down can result in large areas of the littoral zone becoming exposed (Ibbotson & Klee, 2002). The sparse submerged vegetation is composed largely of fennel pondweed (*Potamogeton pectinatus* L.), lesser pondweed (*Potamogeton pusillus* L.), opposite-leaved pondweed (*Groenlandia densa* (L.)) and water crowfoots (*Ranunculus* spp.).

Chew Valley Lake is designated as an SPA on the basis of its waterfowl populations, qualifying under Article 4.2 of the Directive (79/409/EEC) by supporting populations of European importance of shoveler (up to 1.3% of the wintering Northwestern/Central Europe population). The reservoir also supports nationally important numbers of teal, gadwall and tufted duck. Species over-wintering include goldeneye, wigeon (*Anas penelope* L.), snipe (*Gallinago gallinago* (L.)), lapwing (*Vanellus vanellus* (L.)), redshank (*Tringa totanus* (L.)) and Slavonian grebe (*Podiceps auritus* (L.)). Up to 50 broods of great crested grebe and 28 of little grebe (*Tachybaptus ruficollis* (Pallas)) are raised annually, with autumn numbers of the former species being the highest in Britain. Duck species breeding regularly include gadwall, mallard (*Anas platyrhynchos* L.), shoveler, pochard, tufted duck, ruddy duck (*Oxyura jamaicensis* (Gmelin)) and shelduck (*Tadorna tadorna* (L.)). The lake also supports up to 42 000 roosting black-headed gull, common gull (*Larus canus* L.) and lesser black-backed gull (*Larus fuscus* L.).

Chew Valley Lake has been managed as a trout fishery since its creation in the 1950s (Wilson, 1971). As well as trout, Wilson *et al*. (1975) noted that roach, perch, eel and three-spined stickleback were present, with pike introduced illegally in 1990 (Ibbotson & Klee, 2002). Concerns about the threat of pike to the trout fishery resulted in the instigation of an annual pike removal programme. In 1996, for example, gill netting removed 37% of pike large enough to consume trout (Ibbotson & Klee, 2002). Although less than 4% of pike were found to have consumed trout, they contributed >50% to the total biomass of food consumed because of their large size, and this increased to 97% during the trout fishing season (as an outcome of increased stocking). Despite this, of the 35 tonnes of trout stocked in 1996, the predicted loss through pike predation was less than 7% (Ibbotson & Klee, 2002).

Information regarding the current ecology or management of Chew Valley Lake is apparently sparse, despite a doubling of trout stocking rates since 1985 (Ibbotson & Klee, 2002). In total, Chew Valley Lake received 53 500 trout in 2002, 57 500 in 2003, and 50 000 in 2004, and it is inevitable that stocking such large numbers of fishes impacts upon the ecosystem functioning and trophic status of the water body, as described in Section 3.3. In addition, the large numbers of waterfowl and gulls (Laridae) that congregate on the reservoir are likely to increase nutrient loading through guanotrophication. Notwithstanding, the current condition of the reservoir is considered favourable.

4. GUIDANCE FOR STOCKING AND INTRODUCING FISHES TO DESIGNATED NATURAL HERITAGE SITES

4.1 Strategy and management of fish stock enhancement

The previous sections have outlined the range of activities that constitute stock enhancement and the threats and issues posed by stock enhancement activities, especially when uncontrolled. A number of issues arise that need to be addressed if stock enhancement programmes are to be carried out in an ecologically acceptable, socially responsible, technically feasible and cost-effective manner. In particular, the threats posed by fish stock enhancement programmes are especially insidious because few management measures exist to overcome any adverse effects. Furthermore, many stock enhancement programmes appear to have been unsuccessful because of poor project planning and poorly defined objectives (Cowx 1994a, b, 1999). Consequently, there is a need to adopt a strategic approach to fish stock enhancement activities to both improve overall success and minimise impacts on the ecological functioning of recipient water bodies.

To this end, a number of guidelines or codes of practice are available in the public domain. Unfortunately, these are voluntary in nature and mostly focus in the risks associated with stocking and introductions (see IUCN, 1987, 1995; EIFAC, 1988; ICES, 2005 for examples). There is a requirement for more information on the ecological, genetic and pathological impacts of stocking and introduction linked to the economic and social aspects of fisheries enhancement programmes, to aid decision-making. In this respect, the relative merits and cost effectiveness of stocking of different life stages, and at different times of the year, could be useful in determining whether stocking contributes to improved stock status (Aprahamian *et al*., 2003). Integral within any decision-support tool is the inclusion of protocols to ensure that stocking and introductions are conducted in the most effective manner to maximise the success of the activity.

Cowx (1998a, b) presented a strategy for the management of stocking proposals that identified the different levels of data collection and processing required, and presented critical decision levels with some relevant queries (Fig. 1). Cowx (1998c) stated that stocking and introductions should be rejected if the answers to any of the queries in the strategy are negative. This protocol is essentially divided into six components, namely: (1) identification of the objectives of the stocking programme; (2) identification and (3) evaluation of the management options, including assessments of potential ecological and environmental risks; (4) choice of management options; and (5) implementation and (6) monitoring of the activities. The principle behind the strategy is a logical review and decision process for the holistic evaluation of stocking exercises, integrating ecological, fishery, socio-economic and implementation considerations. At each stage of the process, decisions have to be made about the acceptability of potential impacts of stocking or introduction. This decision framework forms the basis of the EA's *'Fish Stocking Work Instruction'*, and could form the basis for supporting decision-making for stocking of fish within designated natural heritage sites in Scotland by the apprpropiate authority. Details of each step in the decision framework are provided in Appendix 1 and referred to in the simplified procedure proposed for stocking of fish within designated natural heritage sites in Scotland (Section 4.2). This protocol is recommended because of the complexity of undertaking a full risk assessment for each stocking project and the need to streamline the process. This should, however, not circumvent any difficult decisions if the potential risks are deemed unacceptable or if there is uncertainty about the outcome of any stocking programme. Under these circumstances a full risk assessment, as outlined in Appendix 1, should be undertaken.

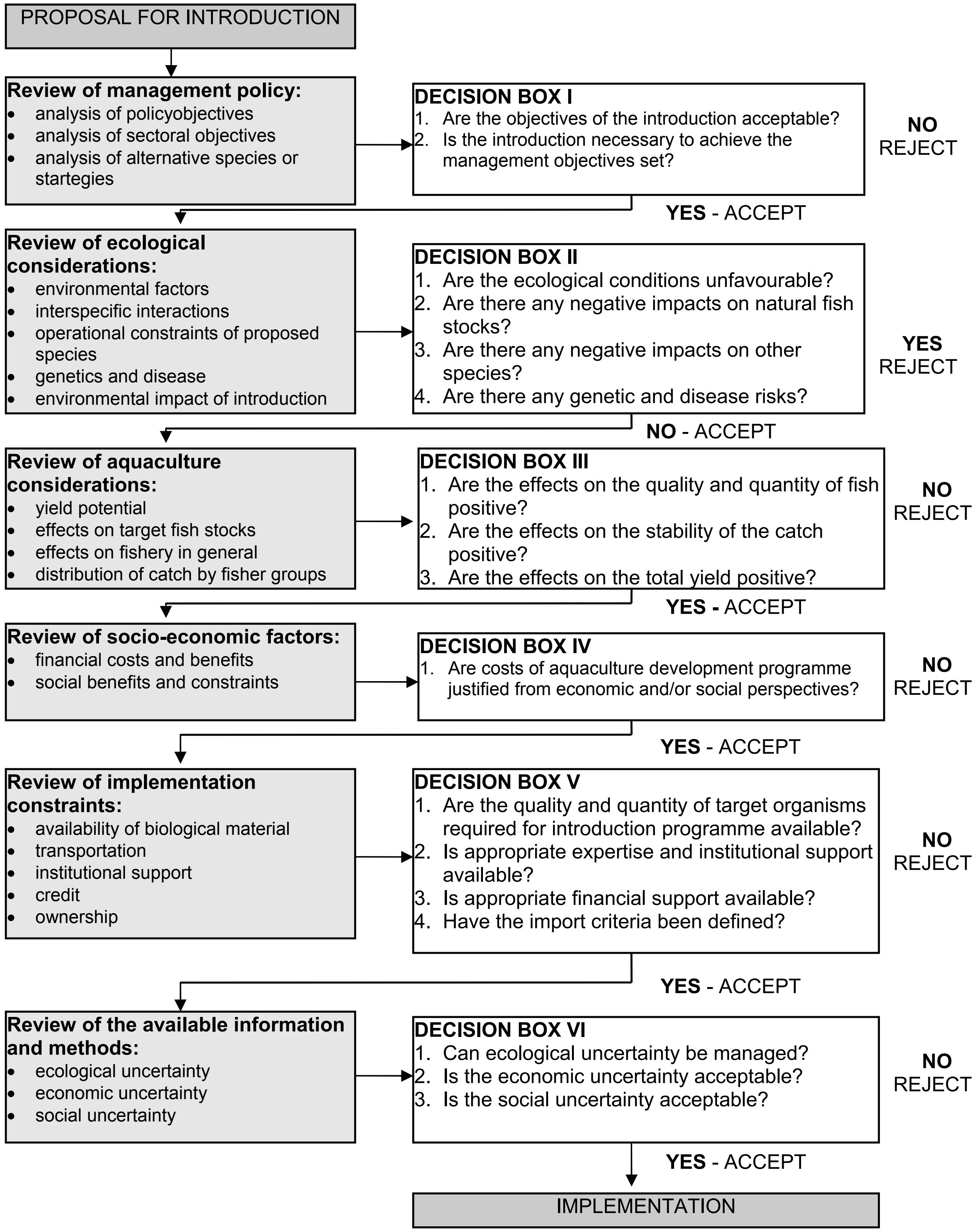


Figure 1. Outline protocol to support evaluating a stocking programme to minimise the potential risk, maximise the potential benefit and monitor the success of the project. Review boxes on the left illustrate the different levels of data collection and processing, whilst decision boxes on the right provide the respective decision levels with some relevant questions. The stock enhancement activity should be rejected if answers to any of the questions are unacceptable (modified from Cowx, 1998a).

4.2 Protocol for stocking of fish within designated natural heritage sites

4.2.1 Context

The previous section provided a detailed analysis of the issues and mechanisms that need to be considered when planning a fish stocking programme. The guidelines highlight the various impacts that could potentially arise in the context of a decision-support tool to inform managers and policy makers. The procedures and decisions taken are framed around a risk assessment procedure that accounts for scale of the likely environmental, ecological, genetic, disease and socio-economic consequences (impacts or benefits) of the stocking event and the degree of uncertainty about whether the impacts will arise. Guidance is also provided to inform decisions on what species should be stocked, sourcing of stocking material, when and how to stock to maximise success. As such, the guidance describes the minimum standards of environmentally friendly, ethically appropriate and - depending on local situations - socially acceptable stocking programmes, and provides a protocol to make strategic decisions on whether to stock a particular water body. This section offers guidance on the information that should be made available to the regulatory authorities in Scotland to aid decision-making of proposed fish stocking events, with particular reference to designated natural heritage sites. It draws on the provisions of the generic guidance outlined in Appendix 1.

4.2.2 Legislative context

Section 35 of the Aquaculture and Fisheries (Scotland) Act 2007, which inserts a new section 33A into the Salmon and Freshwater Fisheries (Consolidation) (Scotland) Act 2003, makes it an offence for any person to intentionally introduce any live fish or spawn of any fish into inland waters, or possess such with the intention of introduction without previous written agreement of the competent authority. The principal aim of these provisions is to protect native biodiversity from the consequences of introductions of non-native fish into Scottish freshwaters. The provisions apply to all introductions of freshwater fish including, salmon, trout and coarse fish to any inland Scottish water system. The legislation became active on 1 August 2008.

Marine Scotland Science (MSS) is the competent authority that deals with most applications to introduce fish into lochs, rivers or reservoirs. Upon receipt of an application, MSS checks whether the site is within a designated natural heritage site. In such cases, MSS consults with SNH. Where the application refers to a location outwith a designated natural heritage site, MSS will not consult SNH unless there is good reason to do so – such as a proposal to introduce a species of high conservation risk, or a proposal that would adversely impact on a designated natural heritage site.

Where a District Salmon Fishery Board (DSFB) operates, and the fish to be introduced are salmon or sea trout, then the relevant DSFB will administer any application from individuals or other bodies who wish to introduce fish. The DSFB will issue written agreement or refusal to the applicant. Currently, there is no obligation for DSFBs to consult SNH. However, the Association of Salmon Fishery Board (ASFB) stocking policy indicates this mechanism should take place where the stocking is within a designated natural heritage site. Where the stocking of salmon is proposed within or connected to a site designated as an SAC for that species, or freshwater pearl mussel, the DSFB must adhere to the requirements of the Habitats Directive and undertake a full appraisal of that activity against the conservation objectives of that site. The collection of salmon broodstock, an activity licensed by Marine Scotland Policy, must also undertake a similar appraisal prior to the issuing of any licence for that purpose.

4.2.3 Proposed decision framework for stocking of fish within designated natural heritage sites

As previously indicated, the generic workflow model provided in Figure 1 outlines the key issues that should be considered when any stock improvement activity is proposed (i.e. ecological, genetic, disease, social and economic factors; see Appendix 1 for details). In many cases, however, stocking has been taking places for decades and the protocol needs to be tuned to local circumstances. In the context of this document, the relevant circumstances are stocking within designated natural heritage sites in Scotland. The overriding issue surrounding stocking in designated natural heritage sites is that there should be no damage to the site, particularly with respect to the designated features. This requires not only an assessment of the status of existing fish stocks and ecological functioning, but an appraisal of the condition of the water body, and the natural and artificial factors that influence the condition of the site, together with an assessment of the likely impacts of any proposed stocking activity. A mechanism for achieving this objective is outlined in Figure 2. This leads the developer through the process and outlines the information that is required by the regulator, in this case MSS or the DSFB - both in consultation with SNH, to make an informed decision on whether the stocking should be permitted. The flowchart draws on information outlined in the generic framework but also cross references to queries that will be raised by SNH (Box 1) and will require answers/resolution with reference to possible adverse effects on the designated site. Within the framework, decisions on whether a stocking event should be permitted should be based on scientific evidence not anecdotal information, and the precautionary approach should be adopted if information is lacking and the risks are considered unacceptable by the regulatory authority.

In addition, it is desirable that only species appropriate to the receiving water body are considered for any stock enhancement activity. The Scottish Government has developed a matrix that outlines those species acceptable for particular water body types in Scotland (Table 3) and this should be consulted to avoid any unnecessary applications that will not be approved by the regulatory authorities.

One of the biggest problems encountered when consenting a stocking operation is determining the correct stocking density. If too many fishes are present, increased mortality rates, through predation and starvation, reduced growth rates and increased dispersal, generally follow. In worst-case scenarios, overstocking can lead to habitat deterioration and a reduction in the performance of the fishery. For fisheries already subjected to stocking activities, lower stocking densities should reduce the potential for competitive interactions between native and stocked fishes, as pressure for finite resources is reduced. This is of particular importance for water bodies that support unique strains of brown trout, charr and whitefish. Reduced stocking densities should also minimise any detrimental impacts on the ecosystem as a whole.

A review of typical stocking densities, mostly accessed from the grey literature, is provided in Appendix 2 and summarised in Table 4 for the main species stocked in differing types of UK waters. The range of densities/biomass stocked varies considerably depending on the species or water body type. What is evident from the review is that stocking densities are not explicitly driven by the objective of the enhancement programme but often by the financial resources of the fishery owner (ability to purchase stock) and/or economic objectives of the fishery. The latter is particularly true for improved and intensively stocked coarse fisheries (Figure 3) where the sole objective is to improve fishery performance (angler satisfaction) in small water bodies (North, 2002). These drivers are not necessarily commensurate with providing an ecologically balanced and socially and environmentally acceptable stocking density/biomass. For example, stocking in improved and intensive fisheries is well beyond the standing crop (biomass) found in natural fisheries (usually < 250 kg ha^{-1}; Figure 3) and thus typically require supplementary feeding and other management

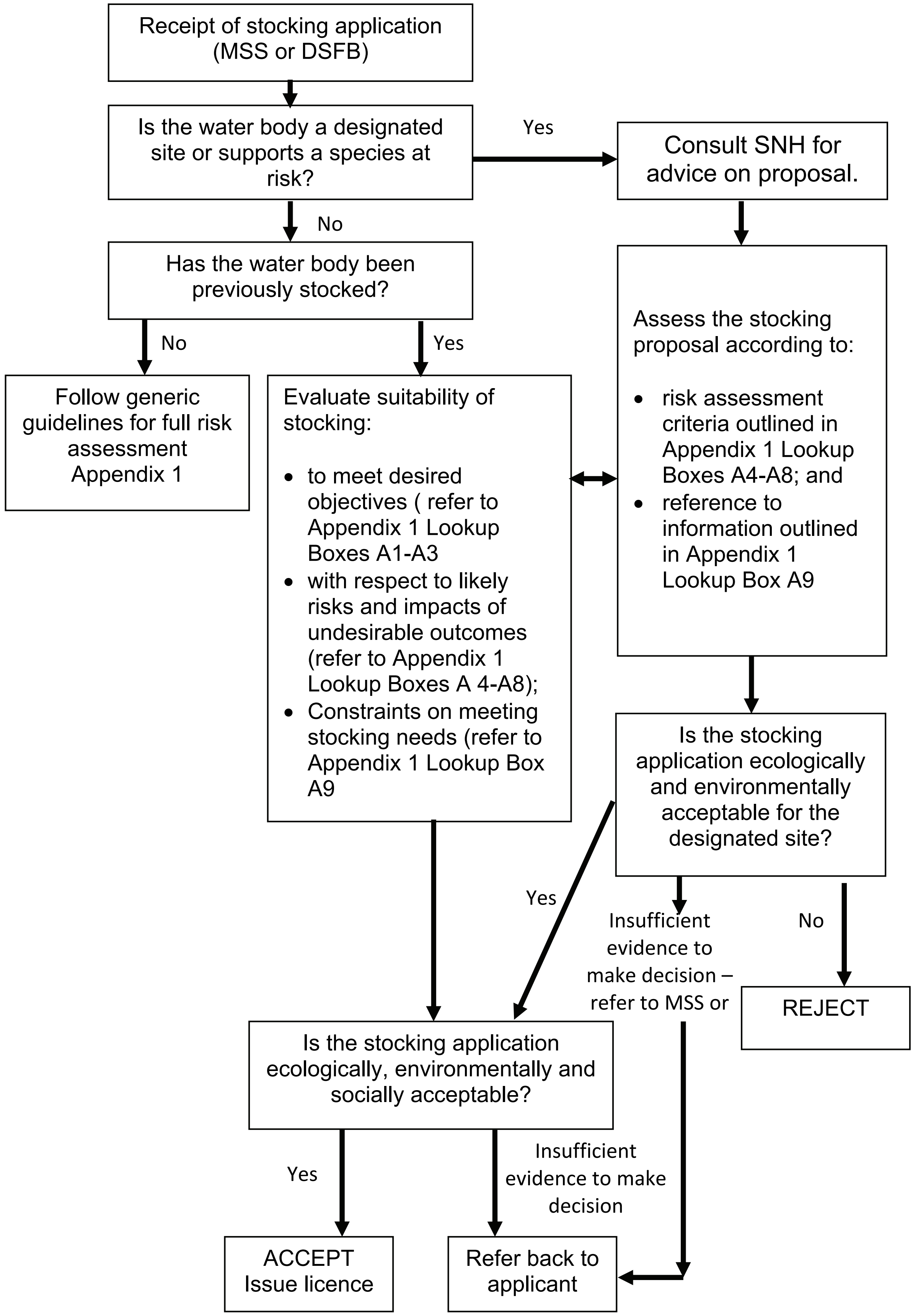


Figure 2. Proposed decision framework for stocking of fish within designated natural heritage sites.

BOX 1: ISSUES TO BE CONSIDERED BY PROPONENT AND EVALUATOR WHEN CONSIDERING STOCKING OF FISH WITHIN DESIGNATED NATURAL HERITAGE SITES. ***NOTE a negative response or proposal not supported by adequate scientific evidence should elicit a rejection.***

- Does the proposed stocking event threaten the conservation status of protected species or the conservation objectives of a designated site? (see NOTES below).
- Has sufficient ecological justification been provided by the applicant for introducing fish to a designated site? (refer to Appendix 1 Look-up Boxes A6)
- Have alternative fisheries management options been considered in full (refer to Appendix 1 Look-up Box A2)
- Is sufficient scientific evidence provided to demonstrate that the fish (number/biomass and species: NOTE 2) to be stocked will not negatively impact species or habitats of conservation concern through:
  - ✓ competition (between stocked and wild fish populations if the numbers of fish added to the waterbody exceed its carrying capacity) (refer to Appendix 1 Look-up Box A6),
  - ✓ predation on either native fish or other aquatic biota of conservation value (refer to Look-up Boxes 6),
  - ✓ disruption of ecosystem functioning and water quality by, for example, the selective removal of zooplankton by introduced fish, the disturbance of river or loch sediments;
  - ✓ the spread of disease, parasites and invasive non-native species (sourced from a supplier who can guarantee stock is will not result in the importation of disease, parasites or invasive non-native species (such as North American signal crayfish) (refer to Appendix 1 Look-up Box A8);
  - ✓ genetic introgression (refer to Appendix 1 Look-up Box A7);
  - ✓ increasing the rate of exploitation of natural populations of fish, by man and other predators;
  - ✓ has reference been made to accceptablity of proposed species and the receiving environment (Table 3).
- Has the level and timing of fish introduction (stocking) has been carefully considered and is the number and biomass of fish to be introduced appropriate for the target water body. Relevant details should be provided (e.g. are the fish to be stocked diploid or triploid (sterile)). NOTE: maximum recommended levels of stocking are such that the final densities do not exceed <100 kg ha^{-1} for brown or rainbow trout and < 300 kg ha^{-1} for coarse fish (refer to Figure A3).
- Is the water body isolated and not contiguous with open waters (i.e. are fish secure in the stocked water body and unlikely to escape into the wild).
- Is a contingency plan to monitor any potential impact and respond to any adverse scenario been provided? (see section A3.9)

NOTE 1: An appraisal of the consequences for natural heritage features (including notified features) should be included.

NOTE 2: Species that are not appropriate for introduction are listed on Schedule 9 of the Wildlife & Countryside Act 1981 or Schedule 1 of the Conservation of Native Freshwater Fish Stocks: The Prohibition of Keeping or Release of Live Fish (Specified Species) (Scotland) Order 2003). These include Arctic charr, vendace or powan. It is also recommended that any species not locally native to either that catchment, or native to Scotland, (e.g. barbel, bleak, bream, chub, dace, gudgeon, tench, orfe, silver bream, ruffe and grayling) are included under the schedule.

NOTE 3: SNH consider it inappropriate to stock any species into any open water habitats within a catchment where it does not currently exist, or where there is no historical evidence to support its presence there in the past;

NOTE 4: SNH generally consider it inappropriate to introduce fish of any species into naturally fishless lochs or lochans;

NOTE 5: SNH consider it inappropriate to introduce Atlantic salmon that originates from a catchment other than that proposed for introduction. In some cases, this may also apply to the movement of salmon at finer geographical scales;

NOTE 6: SNH consider it inappropriate to stock salmonids above naturally impassable waterfalls, unless a satisfactory case has been made for doing so.

Table 3. List of species and list of criteria that must be satisfied before a species can be considered acceptable by the Scottish Government for stocking into a particular water body in Scotland.

	Atlantic salmon	Brown/ sea trout		Rainbow trout	Vendace / Powan	ARCTIC CHARR	PIKE	PERCH	ROACH	COMMON CARP	CRUCIAN CARP	BARBEL	CHUB	BREAM	TENCH	DACE	GUDGEON	RUDD	GRAYLING	ORFE	RUFFE	SILVER BREAM
Type of Area		Dip	Trip																			
Open water within a catchment where there is no evidence species present locally or of local historical stocking																						
Fishless lochans																						
Water has nature conservation designation																						
Introduction could affect nature conservation site																						
Open water where species has been present in the past (either by stocking or native) but species not currently present												?	?	?	?	?	?	?	?			
Closed ARTIFICIAL water within a catchment where species is absent																						
Open water within a catchment previously stocked with species and with extant local wild or native population		A																				
Open water with screens within catchment where species is locally native or naturalised		A																				
Closed water within a catchment where species is already native or naturalised																						
Other types of waterbody																						

Notes:
Red - presumption against INTRODUCTION except in exceptional cases in support of conservation.
Amber - application will be considered further.
Green - applications will in general be acceptable
1. For each species involved, questions in left-hand column of above matrix are to be considered in order from top to bottom.
2. Localness of source stock will be key factor in assessing amber applications for brown trout and salmon and some of the "Other species"
3. Presumption against ESTABLISHMENT of NON-NATIVE fish species (highlighted in red and capitals) to new catchments OR DISRUPTION OF ENVIRNONMENT/BIODIVERSITY
? = Species which have limited distribution as native or stocked in the past and should be given greater consideration before decision of impact is made.
A = brown trout of fish farm origin to open still waters that had been regularly stocked, and brown trout from breeding programmes with wild local broodstock to open still waters to be treated as Green for the present.

interventions such as aeration. The abundance of fish in these intensively stocked fisheries also raises questions about fish welfare and health, with concerns expressed by both regulatory authorities and animal welfare lobby groups (Taylor *et al.*, 2004; Cowx *et al.*, 2007). In recent years, there has been a shift in demand from natural to high performance fisheries. Intensively stocked fisheries are concentrated around urban areas. It is likely there will be considerable pressure to establish more of these fisheries in the future. Given the concerns outlined above, due care must be taken to ensure these fisheries are not overstocked or stocked with inappropriate species to the detriment of fish welfare and wider ecosystem functioning.

Determination of optimal stocking densities should be based on assessment of the carrying capacity of the receiving water body, and be commensurate with the risk and scale of the stocking programmes. For lakes, the optimal density can be determined from relationships between environmental parameters such as shore-line development, water depth and fish biomass (Leopold & Bninska, 1984). This has been further developed by Medley & Lorenzen (2006) to estimate optimal stocking density for culture-based fisheries. Unfortunately this model relates to fisheries where stocks are exploited, and not necessarily to recreational put-and-take or catch-and-release fisheries, where densities are often kept artificially high to

Table 4. Ranges of stocking densities/biomasses reported for a number of fish species and life stages in various water bodies. Specific studies (with references) are detailed in Appendix 2.

Species	**Life stage**	**Water body type**	**Densities / Biomasses**	**Comments**
Atlantic salmon	Green ova	Rivers	0.4-59.0 m^{-2}	
Atlantic salmon	Fry	Rivers	0.60-2.11 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality
Atlantic salmon	0+ parr	Rivers	0.15-0.40 m^{-2}	
Atlantic salmon	1+ parr	Rivers	0.05-0.20 m^{-2}	
Brown trout	0+ and >0+	Rivers	0-478 kg ha^{-1}	Densities and biomasses derived for 20 streams containing freshwater pearl mussels (see Geist *et al.* 2006)
Rainbow, brown and brook trout	Adult	Lakes	2-733 ha^{-1}	
Sea trout	Fry	Rivers	0.40-1.50 m^{-2}	
Sea trout	1+ parr	Rivers	1.0-2.0 m^{-2}	
Coarse fish	n/s	Still waters	10-500 kg ha^{-1}	Approximate natural density
Coarse fish	n/s	Stillwaters	10-126 000 ha^{-1}	Improved and intensive fisheries
Barbel	Adult	Stillwaters	>500 kg ha^{-1}	
Common carp	Adult	Shallow pond mesocosms	174->200 kg ha^{-1}	

increase angler satisfaction. No definitive relationships are available for calculating stocking densities of different species in rivers; these are generally based on the experience of the fishery managers. However, when calculating stocking densities, consideration must be given to the existing stock biomass, the residual stock remaining from previous stocking events, and allowances should be given for migration/dispersal, predation and predicted survival of the stocked fishes. Values of between 10 and 80% annual mortality are given in the literature (EIFAC, 1984), so compensatory densities will be difficult to determine. It is most important to ensure that overstocking is avoided.

Given the lack of definitive data and/or procedures for calculating stock densities, where there is firm evidence that stocking will not damage a designated natural heritage site, then **recommended stock levels within designated natural heritage sites or elsewhere in Scotland should be such that the final densities in the receiving water body do not exceed 100 kg ha^{-1} for brown or rainbow trout, or 300 kg ha^{-1} for coarse fish.** These stock levels will maintain quality fisheries and prevent overstocking of the water body and thus avoid disruption of ecosystem functioning and loss of biodiversity.

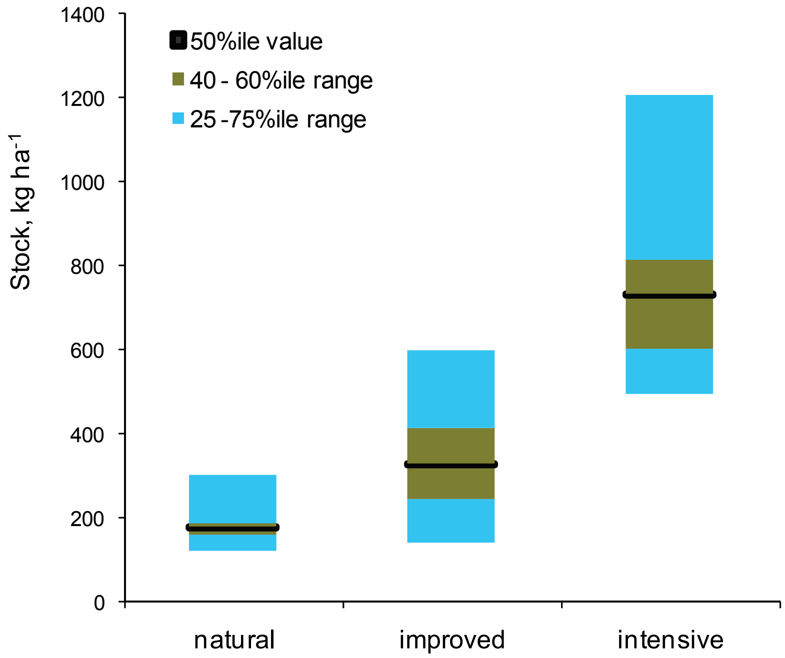


Figure 3. Relative abundance by biomass (kg ha^{-1}) of fish species in three categories of stillwater coarse fishery in England and Wales (adapted from North, 2002).

To improve understanding of the most appropriate stocking density all projects should have in place the methodology to enable adequate monitoring of progress and, ultimately, success or failure.

Finally, before stocking programmes are undertaken a thorough evaluation of the reasons for the action should be examined and alternative approaches to enhancement (e.g. habitat improvements or better fisheries management) should be considered.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Stock enhancement programmes

Stocking is an important tool in the management of fisheries, whether for commercial, recreational or conservation purposes. However, the threats posed by fish stock enhancement programmes, especially introductions, are particularly insidious because few management tools to overcome any adverse effects are available (see Britton *et al*., 2011b). **It is recommended, therefore, that the precautionary approach should be adopted with regard to the stocking and introduction of species, particularly in designated natural heritage sites or in the case of non-native fishes. It is also recommended that a strategic planning approach to stocking, similar to that which is expounded through these guidelines, is adopted.** This draws the attention of the fishery managers and owners to the many problems that must be resolved within a wider fisheries sector context before stocking programmes are likely to achieve their objectives. As part of this approach, a number of aspects should be considered at an early stage (expanded from Li & Moyle, 1994 and Cowx & Godkin, 1999):

- whenever stocking or introduction of fishes is being considered, the aims and specific objectives of the exercise must be clearly defined and adhered to. Also, the potential economic and environmental advantages should be demonstrated, although it is recognised that in some situations (e.g. applications to stock or introduce fishes for conservation purposes) there may be no economic imperative. These should be matched against the disadvantages or problems that may ensue.

- before stocking programmes are undertaken a thorough evaluation of the reasons for the action should be examined and alternative approaches to enhancement (e.g. habitat improvements or better fisheries management) should be considered/discounted.

- if it is possible to remove or minimise the causes of declines in fisheries, this course of action should be taken, and the fisheries may then recover without stocking. Habitat improvement is the most desirable alternative because it should lead to long-term sustainable improvements with minimal deleterious ecological impacts.

- the wider issues and constraints that are likely to affect the long-term success of stocking programmes should be reviewed and considered in the design of enhancement projects.

- stock enhancement activities should be considered mainly for systems that have been so altered by human activity that original fish communities have been disrupted or eliminated and there is no possibility for restoration of the habitats and enhancement of the community based on residual or relict stocks.

- when evaluating stocking as a possible management tool, the relative benefits and costs of all options should be considered. The "do nothing" option should not be disregarded but should be considered as fully as any of the other options under discussion, despite possible public pressure to stock.

- regulators must consider the potential long-term implications of stock enhancement activity on the ecosystem, and should not be guided solely by short-term economic gains. The entire catchment and any adjacent water bodies must be taken into account when considering the proposals.

- the potential for proposed stocking programmes to introduce new parasites or diseases into recipient systems should be assessed through risk assessment protocols.

- the strategy for any programme of stocking, translocation or introduction should be carefully tailored to suit the species in question, taking into account its entire suite of ecological prerequisites, so as to maximize the chances of success.

- the potential adverse impacts of stocking in terms of environmental, genetic and ecological interactions should be considered fully. The 'precautionary principle' should be adopted where foreseen adverse impacts cannot be mitigated, particularly in the case of designated natural heritage sites. Species that might be sensitive to the proposed introductions should be identified in the receiving rivers and lochs. Special consideration should be given to rare species or those most ecologically similar to the species proposed for introduction.

- introductions should be considered mainly for water bodies that are sufficiently isolated to prevent the uncontrolled spread of introduced species. Since most problem waters are not isolated, the best alternative is to evaluate the potential effects of introductions on all connected waters, no matter how distant. Nearby unconnected waters should also be evaluated, as they will be at increased risk of illegal fish transfers.

- significant new stockings or introductions should be evaluated by an independent review panel of scientists familiar with ecological principles and aquatic systems. It is important not to be hasty with introductions, as most effects are irreversible.

- all projects should have in place the methodology to enable adequate monitoring of progress and, ultimately, success or failure. This should include a mechanism of disseminating the outcomes to minimize the risks of any unforeseen adverse effects in future exercises.

- a series of guidelines should be produced for all species that are stocked or introduced, clearly defining the most effective protocol for deciding whether or not stocking should take place, how it should be implemented and the potential impacts of such activities.

When assessing the viability of stocking programmes an evaluation of the most cost-effective options in relation to expected benefits should be undertaken. All too often the strategy is to make do with existing circumstances, whereas a little forward planning may improve the outcome considerably.

It is recommended that all stock enhancement programmes are properly formulated and planned before implementation to avoid indiscriminate and often futile stocking activities. The expected outcome for particular stocking exercises should be compared with wider fisheries sector objectives, and constraints that are likely to prevent a successful outcome should be considered in all appraisals. To this end, practical guidelines for stocking various fish species in a range of water-body types to meet specific objectives should be made available through government agencies and international advisory bodies. Finally, it is recommended that stock enhancement programmes, existing as well as proposed, should be independently assessed to ensure that the wider environmental, ecological and socio-economic issues have been thoroughly reviewed.

5.2 Priority hazards and future R&D

Section 3 identified the potential risks associated with stocking or introducing fishes into designated natural heritage sites. Given the current paucity of knowledge on the impacts of stocking and introduction of fishes, particularly on ecosystem functioning, significant gaps in knowledge were identified, limiting the efficacy of stocking policy for natural heritage sites. This section highlights priority hazards (i.e. those considered to be high risk or for which understanding of the potential risks is limited), and develops a strategic approach to the R&D required to evaluate the hazards of stocking or introducing fishes into designated natural heritage sites.

Overall, research needs to address the following question, “Does stocking or introducing fishes pose a threat to the conservation status of designated natural heritage sites?” Within this overarching question, the key areas for which information is lacking are:

- possible changes to ecosystem functioning;
- impacts of nutrient import to receiving systems;
- risks to native trout, charr, whitefish and lamprey populations; and
- threats to designated invertebrate and macrophyte species.

The three main issues underlying these priority hazards are:

- predation;
- competition; and
- eutrophication.

The relative significance of each of these potential impacts depends partly upon the conservation interests of particular natural heritage sites. Moreover, the impacts of fish stocking will vary between water bodies, depending upon the relative characteristics of their respective ecosystems and stocking programmes. For example, eutrophication may be of considerable concern for water bodies designated for their aquatic macrophytes, but of less *direct* importance for those designated for their waterfowl populations. Similarly, predation by stocked fishes may be of particular concern for water bodies containing designated invertebrate or fish species, but (generally) of less importance in water bodies designated for their macrophytes. Ultimately, however, it is the combination of each of these potential impacts that requires research, to assess whether stocking or introducing fishes poses a threat to the conservation status of the receiving water bodies. This section provides suggestions for R&D modules to address the dearth of knowledge in these areas.

5.2.1 Predation

Stocked or introduced fishes pose a direct predation threat to populations of native (or translocated) trout, charr and whitefish, and designated invertebrates (Sections 3.3 & 3.4). For example, it is thought that one of the translocations of vendace may have been negatively impacted by stocking activities, through either predation of juvenile vendace or competition for food (C.W. Bean, *pers. comm.*). To address this issue, fish diets could be assessed using a combination of three approaches. Firstly, the gut contents of fishes caught by anglers could be retained for analysis. This would require a field scientist to visit the fishery on a number of occasions over the season to remove alimentary tracts from fishes caught by anglers. Additionally, diet information could be obtained using non-destructive methods (e.g. stomach pump) during fisheries surveys. Finally, fishes caught during fisheries surveys could be retained for analysis in the laboratory, although this could create problems with the fishery owners and may not be a viable option.

Although predation on fish eggs is unlikely to be a high-risk hazard, the issue may be raised by anglers. As such, the full risk assessment may need to provide scientific evidence to allay such concerns, including for charr and whitefish. However, assessment of the consumption of eggs by fishes is extremely difficult. Firstly, it is unlikely that angling clubs will give permission for fish surveys during the spawning period. In addition, it is not likely to be possible to determine whether any eggs found in the diet were consumed during active foraging within redds or through ingestion of loose eggs. Furthermore, unless sampling is intensive, the probability of catching fishes with eggs in their guts is low, due to the rapid digestion of eggs. Dietary analysis, therefore, may not be feasible, so proof that egg predation is not a concern may only be possible by inference from spatial ecological studies.

Predation by stocked or introduced fishes may also have an influence both on nutrient budgets and ecosystem functioning in general (Sections 3.1 & 3.2). The key area that requires further research is the relative frequency of piscivory and zooplanktivory in stocked trout, and the impacts that these feeding strategies (and zooplanktivory by coarse fishes) have upon nutrient cycling, trophic cascade and ecosystem function. It will also be important to consider that there may be indirect impacts on plankton demography caused by fish stocking. Whether trout are piscivorous or planktivorous/benthivorous is partly determined by fish size. This has implications for the sizes at which fishes should be stocked, and is likely to differ between trout species, and according to the relative availability of different food groups.

The relative frequency of piscivory and zooplanktivory in stocked trout can be determined from diet analyses, as described above. A range of statistical techniques could be employed to investigate the impacts of fish predation on nutrient budgets and ecosystem functioning. For example, if appropriate data can be obtained, the impacts of various scenarios (e.g. fish densities and piscivore:planktivore ratios) upon nutrient fluxes and ecosystem functioning could be modelled. Alternatively, food intake and the impacts on zooplankton demography could be determined from bioenergetics models (Mehner, 1996; Karjalainen *et al*., 1997; Penczak *et al*., 2002). Stable isotope analysis may also be useful to quantify the transfer of energy between trophic levels (Grey *et al*., 2004; Cunjak *et al*., 2005; Stenroth *et al*., 2006; Yokoyama *et al*., 2006).

5.2.2 Competition

Competition with stocked or introduced fishes poses a direct risk to native salmon, trout, charr and whitefish populations (Section 3.3). Research to investigate competition is required to better understand the risks of stocked or introduced fishes competing with wild individuals, and affecting the viability of wild populations. Such research should be undertaken in tandem with assessments of displacement (see below) to enable the effects of competition to be elucidated. Population parameters, such as abundance, growth and mortality of wild and stocked fishes, should be assessed from regular surveys and analysis of fishery catch statistics. Any changes in these parameters should be evaluated with respect to habitat characteristics and historical data. Competition for food resources between stocked or introduced fishes and native trout, charr and whitefish populations could be investigated through diet analyses, as described above. In addition, RNA/DNA ratio analysis could be used to investigate the impacts of fish stocking and introduction on the nutrition and condition of native fishes (Buckley *et al*., 1999; Caldarone *et al*., 2006).

Given the difficulties associated with the direct observation of fish behaviour in the wild (see Bachman, 1984), studies of aggression and competition are most frequently undertaken in laboratory-based or artificial stream facilities. It is recognised, however, that such trials may not be feasible because of cost restrictions. Moreover, laboratory-based observations are not necessarily a reliable predictor of behaviour in the wild. Thus, it is proposed that evidence for differences in aggression and competitive behaviour are inferred from field studies of stocked systems. Population parameters, catch statistics (where ecological R&D is linked to assessment of fishery performance), and retention and displacement of wild fishes can be used to assess aggressive and competitive behaviour. Additionally, the spatial ecology of stocked/introduced and wild fishes could be assessed using telemetry techniques (radio tracking). Telemetry can provide detailed information on fish movements and distributions, but imposes higher project costs than other methods.

Research into the potential for stocked or introduced fishes to interfere with the spawning of wild stocks requires an assessment of the spatial ecology and behaviour of stocked individuals during the spawning period. It is anticipated that it will be difficult (and too expensive in terms of man-power) to make accurate, direct observations of stocked fishes in spawning areas. However, telemetry could be used to assess the association of stocked fishes with native spawning stocks. The difficulty associated with tracking fishes in large areas of open water minimises the value of this method in stillwater fisheries. Thus, fishes should be tagged and released into the still water of capture, but monitoring should occur around spawning habitats in accessible tributary streams, to determine whether stocked fishes are associated with spawning behaviour. Ideally, fishes should be captured and tagged during end-of-season surveys, thereby allowing them ample time to recuperate. Monitoring of radio-tagged fishes should be undertaken during the winter, focusing on the reproductive period. A number of stocked and wild fishes will need to be tagged to determine any differences in spatial ecology between the two groups.

The potential influences of stocked or introduced fishes on the post-spawning recovery of native fishes are likely to be difficult to determine. The most suitable way to assess this is probably to study the age structure of native fish populations to identify the survival of fishes past maturity, and the capacity for individuals to spawn the following year. Additionally, the relative condition of fishes post-spawning could be assessed prior to the fishing season. In practical terms, it is probable that this could be assessed only once. To obtain valid, meaningful results from both of these approaches, control fisheries (both stocked and un-stocked) must be assessed for post-spawning recovery to enable the relative influence of stocked fishes to be elucidated. This element of research should make full use of existing studies into the spawning behaviour of stocked brown trout (e.g. Shields *et al*., 2005).

Competition with stocked or introduced fishes also poses an indirect risk to designated invertebrates, nutrient status and ecosystem functioning (Sections 3.1, 3.2 & 3.4). This element of research is required to address the possibility of shifts in habitat use or diets of native fishes following stocking or introduction of fishes. Shifts in habitat use could be identified through assessments of the spatial ecology of stocked and wild fishes, as described above, while the diets of native fishes could be compared before and after stocking of fishes. However, care is needed to account for natural variations in habitat use and diet, for example because of seasonal fluctuations in prey availability.

5.2.3 Eutrophication

Eutrophication poses a direct risk to native trout, charr and whitefish populations, invertebrate and macrophyte species, and ecosystem functioning as a whole (Sections 3.1, 3.3 & 3.4). There are a multitude of processes by which this can occur, many of which are complex and poorly understood. Before the effects of eutrophication can be modelled, a nutrient budget should be calculated (see Johnes *et al*., 1996; Moss *et al*., 1996). Included in this should be the contributions of fish-farm effluent, supplementary feed and bird faeces to the nutrient budget. A range of statistical techniques can then be employed to investigate the impacts of eutrophication on ecosystem functioning. For example, if appropriate data can be obtained, the impacts of various nutrient loading scenarios upon productivity and food-web structure could be modelled. In addition, stable isotope analysis may be useful to quantify the transfer of energy between trophic levels, and the inputs of various sources of nutrients (e.g. sewage treatment works, agricultural run-off, fish farms).

Regarding fishes, it is often the associated changes in water quality, rather than the nutrients themselves, that are limiting. Thus, the tolerable limits of various water quality parameters (e.g. dissolved oxygen, temperature) should be investigated for all life stages of trout, charr and whitefish, where these are not available. Included in this should be the comparative limits of fish species likely to compete with native fishes with increasing productivity. It is unknown exactly how eutrophication impacts upon the range of designated invertebrate species. Thus, studies should be initiated to determine the likely responses of key species or assemblages to increases in trophic state, either directly or indirectly (e.g. through their food and habitats). Similarly, the nutrient tolerances for the various species of designated macrophytes should be determined where this information is not available.

Regarding ecosystem functioning, the key area that requires further research is the impacts of gradual, chronic increases in nutrient availability on the relative productivity of macrophytes and phytoplankton, together with concurrent shifts in invertebrate community structure and biodiversity in general. Useful tools are already available to examine and model responses to nutrient increases (e.g. Bennion et al., 2010) but further research is required to understand more fully the implications of alterations in ecosystem functioning through competition-induced shifts in food-web structure.

6 REFERENCES

Aass, P. (1993). Stocking strategy for the rehabilitation of a regulated brown trout (*Salmo trutta*) river. *Regulated Rivers: Research and Management*, 8, 135-144.

Ackefors, H., Johansson, N. & Wahlberg, B. (1991). The Swedish compensatory programme for salmon in the Baltic: an action plan with biological and economic implications. *ICES Marine Science Symposium*, 192, 109-119.

Adams, C.E. (1994). The fish community of Loch Lomond, Scotland: its history and rapidly changing status. *Hydrobiologia*, 290, 91-102.

Adams, C.E. & Maitland, P.S. (1991). Evidence of further invasions of Loch Lomond by non-native fish species with the discovery of a roach × bream, *Rutilus rutilus* (L.) × *Abramis brama* (L.), hybrid. *Journal of Fish Biology*, 38, 961-963.

Adams, C.E. & Maitland, P.S. (2007). Arctic charr in Britain and Ireland – 15 species or one? *Ecology of Freshwater Fish*, 16, 20-28.

Adams, C.E. & Tippett, R. (1991). Powan, *Coregonus lavaretus* (L.), ova predation by newly introduced ruffe, *Gymnocephalus cernuus* (L.), in Loch Lomond, Scotland. *Aquaculture and Fisheries Management*, 22, 239-246.

Alderman, D.J. (1997). History of the spread of crayfish plague in Europe, in crustaceans: bacterial and fungal diseasaes. *QIE Scientific and Technical Review*, 15, 15-23.

Allendorf, F.W. (1991). Ecological and genetic effects of fish introductions: synthesis and recommendations. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 178-181.

Allendorf, F.W. & Leary, R.F. (1988). Conservation and distribution of genetic variation in a polytypic species, the cutthroat trout. *Conservation Biology*, 2, 170-184.

Allendorf, F.W. & Utter, F.M. (1979). Population genetics. In *Fish Physiology*, vol. VII (eds W.S. Hoar, D.J. Randall & J.R. Brett), pp. 407-454. Academic Press, New York.

Alm, G. (1955). Artificial hybridization between different species of the salmon family. *Report of the Institute for Freshwater Research*, *Drottingholm*, 36, 13-56.

Altukhov, Y.P. (1981). The stock concept from the viewpoint of population genetics. *Canadian Journal of Fisheries and Aquatic Sciences*, 38, 1523-1538.

Amundsen, P.A. & Gabler, H.M. (2008). Food consumption and growth of Atlantic salmon *Salmo salar* parr in sub-arctic rivers: empirical support for food limitation and competition. *Journal of Fish Biology*, 73, 250-261.

Andrews, C., Chubb, J.C., Coles, T. & Dearsley, A. (1981). The occurrence of *Bothriocephalus acheilognathi* Yamaguti, 1934 (*B. gowkongensis*) (Cestoda, Pseudophyllidea) in the British Isles. *Journal of Fish Diseases*, 4, 89-93.

Aprahamian, M.W., Martin Smith, K., McGinnity, P., McKelvey, J. & Taylor, J. (2003). Restocking of salmonids – opportunities and limitations. *Fisheries Research*, 62, 211-227.

Arismendi, I., Soto, D., Penaluna, B., Jara, C., Leal, C. & León-Muñoz, J. (2009). Aquaculture, non-native salmonid invasions and associated declines of native fishes in Northern Patagonian lakes. *Freshwater Biology*, 54, 1135-1147.

Arlinghaus, R. & Niesar, M. (2005). Nutrient digestibility of angling baits for carp, *Cyprinus carpio*, with implications for groundbait formulation and eutrophication control. *Fisheries Management and Ecology*, 12, 91-97.

Arlinghaus, R., Mehner T. & Cowx I.G. (2002). Reconciling traditional inland fisheries management and sustainability in industrialised countries, with emphasis on Europe. *Fish and Fisheries*, 3, 261-316.

Arlinghaus, R., Klefoth, T., Cooke, S.J., Gingerich, A. & Suski, C. (2009). Physiological and behavioural consequences of catch-and-release angling on northern pike (*Esox lucius* L.). *Fisheries Research*, 97, 223-233.

Armstrong, J.D., Kemp, P.S., Kennedy, G.J.A., Ladle, M. & Milner, N.J. (2003). Habitat requirements of Atlantic salmon and brown trout in rivers and streams. *Fisheries Research*, 62, 143-170.

Arnason, R. (2001). The economics of ocean ranching: experiences, outlook and theory. *FAO Fisheries Technical Paper*, 413, 45 pp.

Arthington, A.H. & Blühdorn, D.R. (1996). The effects of species interactions resulting from aquaculture operations. In *Aquaculture and Water Resources Management* (eds D.J. Baird, M. Beveridge, L. Kelly & J. Muir), pp. 114-139. Fishing News Books, Blackwell Scientific Publications, Oxford.

Azevedo, P.A., Podemski, C.L., Hesslein, R.H., Kasian, S.E.M., Findlay, D.L. & Bureau, D.P. (2011). Estimation of waste outputs by a rainbow trout cage farm using a nutritional approach and monitoring of lake water quality. *Aquaculture*, 311, 175-186.

Bachman, R.A. (1984). Foraging behaviour of free-ranging wild and hatchery brown trout in a stream. *Transactions of the American Fisheries Society*, 113, 1-32.

Bailey, J.K. (1987). Canadian sea ranching programme (east coast). In *World Symposium on Selection*, *Hybridisation and Genetic Engineering in Aquaculture*, vol. II (ed. K. Tiews), pp. 443-448. Heinemann, Berlin.

Bailey-Watts, A.E. & Kirika, A. (1999). Poor water quality in Loch Leven (Scotland) in 1995 in spite of reduced phosphorus loadings since 1985: the influences of catchment management and inter-annual weather variation. *Hydrobiologia*, 403, 135-151.

Bakke, T.A., Harris, P.D. & Cable, J. (2002). Host specificity dynamics: observations on gyrodactylid monogeneans. *International Journal for Parasitology*, 32, 281-308.

Balayla, D.J. & Moss, B. (2003). Spatial patterns and population dynamics of plant-associated microcrustacea (Cladocera) in an English shallow lake (Little Mere, Cheshire). *Aquatic Ecology*, 37, 417-435.

Balayla, D.J. & Moss, B. (2004). Relative importance of grazing on algae by plant-associated and open-water microcrustacea (Cladocera). *Archiv für Hydrobiologie*, 161, 199-224.

Bams, R.A. (1976). Survival and propensity for homing as affected by presence or absence of locally adapted paternal genes in two transplanted populations of pink salmon (*Oncorhynchus gorbuscha*). *Journal of the Fisheries Research Board of Canada*, 33, 2716-2725.

Bannister, R.C.A. & Addison, J.T. (1998). Enhancing lobster stocks: a review of recent European methods, results and future prospects. *Bulletin of Marine Sciences*, 62, 369-387.

Barnard, S. (2006). The effects of stocked brown trout on the survival of wild fish populations. *Environment Agency R&D Note 490*. EA, Bristol.

Bartley, D.M. (1999). Marine ranching: a global perspective. In *Stock Enhancement and Sea Ranching*. (eds B.R. Howell, E. Moksness & T. Svasand) pp. 79–90. Blackwell Science, Oxford.

Bartley, D.M., Bindad-Reantaso, M.G. & Subasinghe, R.P. (2006). A risk analysis framework for animal health management in marine stock enhancement programs. *Fisheries Research*, 80, 28-36.

Bass, J.A.B., Leach, D.V. & Pinder, L.C.V. (1997). The invertebrate community of submerged *Nuphar lutea* (L.) leaves in the River Great Ouse. *Regulated Rivers: Research and Management*, 13, 259-266.

Bauer, O.N. (1991). Spread of parasites and diseases of aquatic organisms by acclimatization: a short review. *Journal of Fish Biology*, 39, 679-686.

Bazydło, E. (2004). Effect of environmental conditions on the populations of *Luronium natans* (L.) Raf. *Polish Journal of Ecology*, 52, 181-189.

Beall, E., Heland, M. & Marty, C. (1989). Interspecific relationships between emerging Atlantic salmon, *Salmo salar*, and coho salmon, *Oncorhynchus kisutch*, juveniles. *Journal of Fish Biology*, 35 (Suppl. A), 285-293.

Bean, C.W. & Winfield, I.J. (1989). Biological and ecological effects of a *Ligula intestinalis* (L.) infestation of the gudgeon (*Gobio gobio* L.) in Lough Neagh, Northern Ireland. *Journal of Fish Biology*, 34, 135-147.

Bean, C.W. & Winfield, I.J. (1992). The influences of *Ligula intestinalis* (L.) on the distribution of juvenile roach *Rutilus rutilus* (L.) and gudgeon *Gobio gobio* (L.) in Lough Neagh, Northern Ireland. *Netherlands Journal of Zoology*, 42, 416-429.

Bean, C.W., Maitland, P.S. & Collen, P. (2004). Crayfish in Scotland: a review of current status and legislative control. *Freshwater Crayfish*, 15, 220-228.

Bearhop, S., Thompson, D.R., Waldron, S., Russell, I.C., Alexander, G. & Furness, R.W. (1999). Stable isotopes indicate the extent of freshwater feeding by cormorants *Phalacrocorax carbo* shot at inland fisheries in England. *Journal of Applied Ecology*, 36, 75-84.

Bennion, H., Rawcliffe, R., Goldsmith, B., Davidson, T., Sayer, C., Yang, H. & Rose, N. (2010). Palaeoecological study of two lochs: Butterstone Loch and Lindores Loch. *Scottish Natural Heritage Commissioned Report No. 355.* SNH, Battleby.

Berg, S. (1998). Diet of rainbow trout, *Oncorhynchus mykiss*, stocked as a biomanipulation tool in a eutrophic brackish lake. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 88-98. Fishing News Books, Blackwell Science, Oxford.

Berg, S. & Jorgensen, J. (1991). Stocking experiments with 0+ and 1+ trout parr, *Salmo trutta* L., of wild and hatchery origin: 1. Post-stocking mortality and smolt yield. *Journal of Fish Biology*, 39, 151-169.

Bergan, P.I., Gausen, D. & Hansen, L.P. (1991). Attempts to reduce the impact of reared Atlantic salmon on wild in Norway. *Aquaculture*, 98, 319-324.

Berka, R. (1986). The transport of live fish. *EIFAC Technical Paper*, 48, 52 pp.

Bertolo, A., Lacroix, G., Lescher-Moutoué, F. & Sala, S. (1999). Effects of physical refuges on fish-plankton interactions. *Freshwater Biology*, 41, 795-808.

Beveridge, M.C.M. & Phillips, M.J. (1993). Environmental impact of tropical aquaculture. In *Environment and Aquaculture in Developing Countries* (eds R.S.V. Pullin & J.L. MacLean), pp. 213-236. *ICLARM Conference Proceedings*, 31, Manila.

Billington, N. & Hebert, P. (eds) (1991). Proceedings of the ecological and genetic implications of fish introductions symposium. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 181 pp.

Blaxter, J.H.S. (2000). The enhancement of marine fish stocks. *Advances in Marine Biology*, 38, 2-54.

Blankenship, H.L. & Leber, K.M. (1995). A responsible approach to marine stock enhancement. *American Fisheries Society Symposium*, 15, 167-175.

Bolland, J.D., Cowx, I.G. & Lucas, M.C. (2009). Dispersal and survival of stocked cyprinids in a small English river: comparison with wild fishes using a multi-method approach. *Journal of Fish Biology*, 74, 2313-2328.

Bolland, J.D. Bracken, L.J., Martin, R. & Lucas, M.C. (2010). A protocol for stocking hatchery reared freshwater pearl mussel *Margaritifera margaritifera. Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 695-704.

Borgstrøm R., Skaala Ø. & Aastveit A.H. (2002) High mortality in introduced brown trout depressed potential gene flow to a wild population. *Journal of Fish Biology* **64**, 1085-1097.

Bostock, J., McAndrew, B., Richards, R., Jauncey, K., Telfer, T., Lorenzen, K., Little, D., Ross, L., Handisyde, N., Gatward, I. & Corner, R. (2010). Aquaculture: global status and trends. *Philosophical Transactions of the Royal Society B*, 365, 2897-2912.

Boxshall, G.A. & Frear, P.A. (1990). *Tracheliastes maculatus* (Kollar, 1836) (Crustacea: Lernaeopodidae) a new species for Britain. *Journal of Fish Biology*, 37, 489-491.

Brannon, E.L. (1982). Orientation mechanisms of homing salmonids. In *Proceedings of the Salmon and Trout Migratory Behavior Symposium* (eds E.L. Brannon & E.O. Salo). University of Washington, Seattle.

Britton, J.R., Harvey, J.P., Cowx, I.G., Holden, T., Feltham, M.J., Wilson, B.R. & Davies, J.M. (2002). Compensatory responses of fish populations in a shallow eutrophic lake to heavy depredation pressure by cormorants and the implications for management. In *Management and Ecology of Lake and Reservoir Fisheries* (ed. I.G. Cowx), pp. 171-183. Fishing News Books, Blackwell Science, Oxford.

Britton, J.R., Shepherd, J.S., Toms, S. & Simpson, V. (2005). Presence of carp, *Cyprinus carpio*, in the diet of the otter, *Lutra lutra*. *Fisheries Management and Ecology*, 12, 221-223.

Britton, J.R., Boar, R.R., Grey, J., Foster, J., Lugonzo, J. & Harper, D.M. (2007). From introduction to fishery dominance: the initial impacts of the invasive carp *Cyprinus carpio* in Lake Naivasha, Kenya, 1999 to 2006. *Journal of Fish Biology*, 71 (Suppl. D), 239-257.

Britton, J.R., Cucherousset, J., Grey, J. & Gozlan, R.E. (2011a). Determining the strength of exploitative competition from an introduced fish: roles of density, biomass and body size. *Ecology of Freshwater Fish*, 20, 74-79.

Britton, J.R., Gozlan, R.E. & Copp, G.H. (2011b). Managing non-native fish in the environment. *Fish and Fisheries*, in press.

Broughton, N.M., Goldspink, G. & Jones, N.V. (1981). Histological differences in the lateral musculature of 0-group roach, *Rutilus rutilus* (L.) from different habitats. *Journal of Fish Biology*, 18, 117-122.

Brown, E.A.R. & Scott, D.B.C. (1994). Life histories of the powan, *Coregonus lavaretus* (L.) (Salmonidae, Coregoninae) of Loch Lomond and Loch Eck. *Hydrobiologia*, 290, 121-133.

Bruton, M.N. & van As, J.G. (1986). Faunal invasions of aquatic ecosystems in southern Africa, with suggestions for their management. In *The Ecology and Management of Biological Invasions in Southern Africa* (eds I.A.W. MacDonald, F.J. Kruge & A.A. Ferrar), pp. 199-212. Oxford University Press, Cape Town.

Bubb, D.H., Thom, T.J. & Lucas, M.C. (2005). The within-catchment invasion of the non-indigenous signal crayfish *Pacifastacus leniusculus* (Dana), in upland rivers. *Bulletin Français de la Pêche et de la Pisciculture*, 376-37, 665-673.

Bubb, D.H., Thom, T.J. & Lucas, M.C. (2008). Spatial ecology of the white-clawed crayfish in an upland stream and implications for the conservation of this endangered species. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 647-657.

Buckley, L., Caldarone, E. & Ong, T.L. (1999). RNA-DNA ratio and other nucleic acid-based indicators for growth and condition of marine fishes. *Hydrobiologia*, 401, 265-277.

Burks, R.L., Jeppesen, E. & Lodge, D.M. (2001). Littoral zone structures as *Daphnia* refugia against fish predators. *Limnology and Oceanography*, 46, 230-237.

Burton, M.J. & Tegner, R.S. (2000). Enhancement of red abalone *Haliotis rufescens* stocks at San Miguel Island: reassessing a success story. *Marine Ecology Progress Series*, 202, 303-308.

Busack, C.A. & Currens, K.P. (1995). Genetic risks and hazards in hatchery operations: fundamental concepts and issues. In: *Uses and Effects of Cultured Fishes in Aquatic Ecosystems*. (eds H.L. Schramm & R.G. Piper). pp. 71-80. American Fisheries Society Bethesda.

Butcher, A., Burke, J. & Brown, I. (eds) (2000). The Maroochy Estuary fish stocking program, 1995-1999. *Department of Primary Industries*. DPI, Queensland.

Cahn, A.R. (1929). The effect of carp on a small lake: the carp as a dominant. *Ecology*, 10, 271-274.

Caldarone, E.M., Clemmesen, C.M., Berdalet, E., Miller, T.J., Folkvord, A., Holt, G.J., Olivar, M.P. & Suthers, I.M. (2006). Intercalibration of four spectrofluorometric protocols for measuring RNA/DNA ratios in larval and juvenile fish. *Limnology and Oceanography Methods*, 4, 153-163.

Cambray, J.A. (2003). Impact on indigenous species biodiversity caused by the globalisation of alien recreational freshwater fisheries. *Hydrobiologia*, 500, 217-230.

Campbell, M.L. (2006). Risk assessment (modified organism impact assessment) to update information on *Undaria pinnatifida*. *All Oceans Ecology Client Report AOE2006-02*.

Campton, D.E. & Johnston, J.M. (1985). Electrophoretic evidence for a genetic admixture of native and nonnative rainbow trout in the Yakima River, Washington. *Transactions of the American Fisheries Society*, 114, 782-793.

Campton, D.E., Akllendorf, F.W., Behnke, R.J., Utter, F.M., Chilcote, M.W., Leider, S.A. & Loch, J.J. (1991). Reproductive success of hatchery and wild steelhead: a reanalysis. *Transactions of the American Fisheries Society*, 120, 816-827.

Carrs, D., Spears, B.M., Quinn, L. & Cooper, R. (2012). Long-term variations in waterfowl populations in Loch Leven: identifying discontinuities between local and national trends. *Hydrobiologia*, in press.

Carvalho, G.R. (1993). Evolutionary aspects of fish distributions: genetic variability and adaptation. *Journal of Fish Biology*, 43 (Suppl. A), 53-73.

Carvalho, G.R. & Cross, T.F. (1998). Enhancing fish production through introductions and stocking: genetic perspectives. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 329-337. Fishing News Books, Blackwell Scientific Publications, Oxford.

Carvalho, L. & Moss, B. (1995). The current status of a sample of English Sites of Special Scientific Interest subject to eutrophication. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 5, 191-204.

Carvalho, L., Kirika, A. & Gunn, I. (2004). Long-term patterns of change in physical, chemical and biological aspects of water quality at Loch Leven. *Scottish Natural Heritage Commissioned Report No. 031 (ROAME No. F01LH03C)*. SNH, Battleby.

Carvalho, L., Miller, C., Spears, B.M., Gunn, I.D.M., Bennion, H., Kirika, A. & May, L. (2012). Water quality of Loch Leven: responses to enrichment, restoration and climate change. *Hydrobiologia*, in press.

Casal, C.M.V. (2006). Global documentation of fish introductions: the growing crisis and recommendations for action. *Biological Invasions*, 8, 3-11.

Celikkale, M.S. (1990). Inland fisheries of Turkey. In *Management of Freshwater Fisheries* (eds W.L.T. van Densen, B. Steinmetz & R.H. Hughes), pp. 493-504. Pudoc, Wageningen.

Chaichana, R., Leah, R. & Moss, B. (2010). Birds as eutrophicating agents: a nutrient budget for a small lake in a protected area. *Hydrobiologia*, 646, 111-121.

Chaichana, R., Leah, R. & Moss, B. (2011). Conservation of pond systems: a case study of intractability, Brown Moss, UK. *Hydrobiologia*, 664, 17-33.

Chappaz, R., Doucende, D. & Barthelemy, R. (1999). Patterns of change in zooplankton community structures and the selective feeding of bleak, *Alburnus alburnus* (L.) in the Serre Ponçon dam between 1980 and 1996. *Hydrobiologia*, 391, 127-134.

Chevassus, B. (1979). Hybridization in salmonids: results and perspectives. *Aquaculture*, 17, 113-128.

Chilcote, M.W., Leider, S.A. & Loch, J.J. (1986). Differential reproductive success of hatchery and wild summer-run steelhead under natural conditions. *Transactions of the American Fisheries Society*, 115, 726-735.

Chubb, J.C. & Yeomans, W.E. (1995). *Khawia sinensis* Hsü 1935 (Cestoda: Carophyllidae) a tapeworm new to the British Isles: a threat to carp fisheries? *Fisheries Management and Ecology*, 2, 263-278.

Clifford, S.L., McGinnity, P. & Ferguson, A. (1997). Genetic changes in Atlantic salmon (*Salmo salar* L.) populations of NW Irish rivers resulting from escapes of adult farm salmon. *Canadian Journal of Fisheries and Aquatic Sciences*, 55, 358 363.

Clugston, J.P. (1990). Exotic animals and plants in aquaculture. *Reviews in Aquatic Sciences*, 2, 481-489.

Coates, D. (1998). Codes of practice for the stocking and introduction of fish. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 383-396. Fishing News Books, Blackwell Scientific Publications, Oxford.

Cochran, P.A. (2009). Predation on lampreys. *American Fisheries Society Symposium*, 72, 139-151.

Cochran, P.A., Leisten, A.A. & Sneen, M.E. (1992). Cases of predation and parasitism on lampreys in Wisconsin. *Journal of Freshwater Ecology*, 7, 435-436.

Collares-Pereira, M.J., Pires, A.M., Coelho, M.M. & Cowx, I.G. (1998). Towards a conservation strategy for *Anaecypris hispanica*, the most endangered non-migratory fish in Portuguese streams. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 437-449. Fishing News Books, Blackwell Scientific Publications, Oxford.

Compton, D.E. (1995). Genetic effects of hatchery fish on wild populations of Pacific salmon and steelhead: what do we really know? *American Fisheries Society Symposium*, 15, 337-353.

Cooke, S.J. & Schramm, H.L. (2007). Catch-and-release science and its application to conservation and management of recreational fisheries. *Fisheries Management and Ecology*, 14, 73-79.

Copp, G.H., Garthwaite, R. & Gozlan, R.E. (2005). Risk identification and assessment of non-native freshwater fishes: a summary of concepts and perspectives on protocols for the UK. *Journal of Applied Ichthyology*, 21, 371-373.

Copp, G.H., Vilizzi, L., Mumford, J., Fenwick, G.V., Godard, M.J. & Gozlan, R.E. (2009a). Calibration of FISK, an invasiveness screening tool for non-native freshwater fishes. *Risk Analysis*, 29, 457-467.

Copp, G.H., Britton, J.R., Cucherousset, J., García-Berthou, E., Kirk, R., Peeler, E. & Stakėnas, S. (2009a). Voracious invader or benign feline? A review of the environmental biology of European catfish *Silurus glanis* in its native and introduced ranges. *Fish and Fisheries*, 10, 252-282.

Cosgrove, P.J., Young, M.R., Hastie, L.C., Gaywood, M. & Boon, P.J. (2000). The status of the freshwater pearl mussel *Margaritifera margaritifera* Linn. in Scotland. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 10, 197-208.

Costa-Dias, S., Dias, E., Lobón-Cerviá, J., Antunes, C. & Coimbra, J. (2010). Infection by *Anguillicoloides crassus* in a riverine stock of European eel, *Anguilla anguilla*. *Fisheries Management and Ecology*, 17, 485-492.

Cowx, I.G. (1989). Interaction between the roach, *Rutilus rutilus*, and dace, *Leuciscus leuciscus*, populations in a river catchment in south-west England. *Journal of Fish Biology*, 35 (Suppl. A), 279-284.

Cowx, I.G. (ed.) (1994a). *Rehabilitation of Freshwater Fisheries.* Fishing News Books, Blackwell Scientific Publications, Oxford.

Cowx, I.G. (1994b). Stocking strategies. *Fisheries Management and Ecology*, 1, 15-31.

Cowx, I.G. (1994c). Strategic approach to fishery rehabilitation In *Rehabilitation of Freshwater Fisheries* (ed. I.G. Cowx), pp. 1-9. Fishing News Books, Blackwell Scientific Publications, Oxford.

Cowx, I.G. (1997). L'introduction d'espèces éstrangères de poissons dans les eaux douces européennes: succès économiques ou désastre écologique? *Bulletin Français de la Pêche et de la Pisciculture*, 344/345, 57-78.

Cowx, I.G. (1998a). An appraisal of stocking strategies in the light of developing country constraints. *Fisheries Management and Ecology*, 6, 21-34.

Cowx, I.G. (1998b). Stocking strategies: issues and options for future enhancement programmes. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 3-13. Fishing News Books, Blackwell Scientific Publications, Oxford.

Cowx, I.G. (ed.) (1998c). *Stocking and Introduction of Fish.* Fishing News Books, Blackwell Scientific Publications, Oxford.

Cowx, I.G. (1999). An appraisal of stocking strategies in the light of developing country constraints. *Fisheries Management and Ecology*, 6, 21-34.

Cowx, I.G. (ed.) (2003). *Interactions between Fish and Birds: Implications for Management*. Fishing News Books, Blackwell Science, Oxford.

Cowx, I.G. & Gerdeaux, D. (2004). The effects of fisheries management practices on freshwater ecosystems. *Fisheries Management and Ecology*, 11, 145-151.

Cowx, I.G. & Godkin, P.A. (1999). Analysis of the environmental and economic impact of operations to reinforce the aquatic fauna of fresh waters for fishery purposes. *Report to the DGXIV*. European Commission, Brussels.

Cowx, I.G. & Welcomme, R.L. (eds) (1998). *Rehabilitation of Rivers for Fish*. Fishing News Books, Blackwell Scientific Publications, Oxford.

Cowx, I.G., Noble, R.A.A., Nunn, A.D. & Harvey, J.P. (2006). Hybridisation between stocked brown trout and wild Atlantic salmon. *Report to English Nature*. EN, Peterborough.

Cowx, I.G., Noble, R.A.A., Nunn, A.D., Harvey, J.P. & Ibáñez-Aguirre, A.L. (2007). Fish stocking technical manual. *Report to Environment Agency*. EA, Bristol.

Cowx, I.G., Miossec, L., Lapègue, S. & Nunn, A.D. (2008). Analysis of changes in genetic integrity of indigenous populations resulting from alien species introduction and genetically-modified organisms. *IMPASSE Project Report to European Commission*. EC, Brussels.

Cowx, I.G., Bolland, J.D., Nunn, A.D., Kerins, G., Stein, J., Blackburn, J., Hart, A., Henry, C., Britton, J.R. & Copp, G.H. (2010). Defining environmental risk assessment criteria for genetically-modified fishes to be placed on the EU market. *Report to European Food Safety Authority*. EFSA, Parma.

Cresswell, R.C. (1981). Post-stocking movements and recapture of hatchery-reared trout released into flowing waters – a review. *Journal of Fish Biology*, 18, 429-442.

Crivelli, A.J. (1983). The destruction of aquatic vegetation by carp. *Hydrobiologia*, 106, 37-41.

Cross, D.G. (1969). Aquatic weed control using grass carp. *Journal of Fish Biology*, 1, 27-30.

Cross, T.F. (1999). Genetic considerations in enhancement and ranching of marine and anadromous species. In *Stock Enhancement and Sea Ranching* (eds B.R. Howell, E. Moksness & T. Svasand), pp. 37-48. Fishing News Books, Oxford.

Crowe, T.P., Amos, M.J. & Lee, C.L. (1997). The potential of reseeding with juveniles as a tool for the management of trochus fisheries. In *Trochus: Status*, *Hatchery Practice and Nutrition* (eds C.L. Lee & P.W. Lynch), pp. 170-177. ACIAR, Proceedings 79.

Cryer, M. & Edwards, R.W. (1987). The impact of angler groundbait on benthic invertebrates and sediment respiration in a shallow eutrophic reservoir. *Environmental Pollution*, 46, 137-150.

Cryer, M., Peirson, G. & Townsend, C.R. (1986). Reciprocal interactions between roach *Rutilus rutilus* (L.) and zooplankton in a small lake: Prey dynamics and fish growth and recruitment. *Limnology and Oceanography*, 31, 1022-1038.

Cunjak, R.A., Roussel, J.M., Gray, M.A., Dietrich, J.P., Cartwright, D.F., Munkittrick, K.R. & Jardine, T.D. (2005). Using stable isotope analysis with telemetry or mark-recapture data to identify fish movement and foraging. *Oecologia*, 144, 636-646.

De Kinkelin, P. & Hedrick, R.P. (1991). International veterinary guidelines for the transport of live fish or fish eggs. *Annual Review of Fish Diseases*, 27-40.

Demers, E. McQueen, D.J., Ramcharan, C.W. & Pérez-Fuentetaja, A. (2001). Did piscivores regulate changes in fish community structure? The Dorset food web piscivore manipulation project. *Archiv für Hydrobiologie Special Issues in Advanced Limnology*, 56, 49-80.

Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J.T., Gimenez-Casalduero, F. & Valle, C. (2002). Attraction of wild fish to sea-cage fish farms in the south-western Mediterranean Sea: spatial and short-term temporal variability. *Marine Ecology Progress Series*, 242, 237-252.

Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J. & Kingsford, M. (2004). Extensive aggregations of wild fish at coastal sea-cage fish farms. *Hydrobiologia*, 525, 245-248.

Deverill, J.I., Adams, C.E. & Bean, C.W. (1999a). Prior residence, aggression and territory acquisition in hatchery-reared and wild brown trout. *Journal of Fish Biology*, 55, 868-875.

Deverill, J.I., Bean, C.W. & Adams, C.A. (1999b). Last one to the freeze-dried chironomid is probably a stockie. *Salmo Trutta*, 2, 62-63.

Donald, D.B. Vinebrooke, R.D., Anderson, R.S., Syrgiannis, J. & Graham, M.D. (2001). Recovery of zooplankton assemblages in mountain lakes from the effects of introduced sport fish. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 1822-1830.

Dörner, H. & Benndorf, J. (2003). Piscivory by large eels on young-of-the-year fishes: its potential as a biomanipulation tool. *Journal of Fish Biology*, 62, 491-494.

Dörner, H., Wagner, A. & Benndorf, J. (1999). Predation by piscivorous fish on age-0 fish: spatial and temporal variability in a biomanipulated lake (Bautzen reservoir, Germany). *Hydrobiologia*, 409, 39-46.

Doyle, R.W., Shackel, N.L., Basiao, Z., Uraiwan, S., Matricia, T. & Talbot, A.J. (1991). Selective diversification of aquaculture stocks: a proposal for economically sustainable genetic conservation. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 148-154.

Doyle, R.W., Perez-Enriquez, R., Takagi, M. & Taniguchi, N. (2001). Selective recovery of founder genetic diversity in aquacultural broodstocks and captive, endangered fish populations. *Genetica*, 111, 291-304.

Drawbridge M.A. (2002) The role of aquaculture in the restoration of coastal fisheries. In: A.A. Costa-Pierce (ed.) *Ecological Aquaculture, The Evolution of the Blue Revolution*. Blackwell Science, Oxford, pp. 314-336.

Dudley, B., Gunn, I.D.M., Carvalho, L., Proctor, I., O'Hare, M.T., Murphy, K.J. & Milligan, A. (2012). Changes in aquatic macrophyte communities in Loch Leven: evidence of recovery from eutrophication? *Hydrobiologia*, in press.

Dugdale, T.M., Hicks, B.J., De Winton, M. & Taumoepeau, A. (2006). Fish exclosures versus intensive fishing to restore charophytes in a shallow New Zealand lake. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 193-202.

Duguid, R. A., Ferguson, A. & Prodöhl, P. (2006). Reproductive isolation and genetic differentiation of ferox trout from sympatric brown trout in Loch Awe and Loch Laggan, Scotland. *Journal of Fish Biology*, 69 (Suppl. A), 89-114.

Dushkina, L.A. (1991). Relationships between mariculture and natural populations in the seas of the USSR. *ICES Marine Science Symposium*, 192, 68.

EA (Environment Agency) (2006). Freshwater fisheries and wildlife conservation: a good practice guide. Environment Agency, Bristol.

Eby, L.A., Roach, W.A., Crowder, L.B. & Stanford, J.A. (2006). Effects of stocking-up freshwater food webs. *Trends in Ecology and Evolution*, 21, 576-584.

Ecological Research Associates (2005). National lamprey survey of Scotland. *Scottish Natural Heritage Commissioned Report (ROAME No. F02AC602)*. SNH, Battleby.

Egidius, E. (1987). Import of furunculosis to Norway with Atlantic salmon smolts from Scotland. *ICES CM*, 1987/F, 8.

EIFAC (European Inland Fisheries Advisory Commission) (1984). Documents presented at the symposium on stock enhancement in the management of freshwater fish, vol. 1: Stocking. *EIFAC Technical Paper, No. 42 (Suppl. 1).* EIFAC, Rome.

EIFAC (European Inland Fisheries Advisory Commission) (1988). Codes of practice and manual of procedures for consideration of introductions and transfers of marine and freshwater organisms. *FAO/EIFAC Occasional Paper, No. 23*. EIFAC, Rome.

Einum, S. & Fleming, I.A. (1997). Genetic divergence and interactions in the wild among native, farmed and hybrid Atlantic salmon. *Journal of Fish Biology*, 50, 634-651.

Ellis, J.C., Farina, J.M. & Witman, J.D. (2006). Nutrient transfer from sea to land: the case of gulls and cormorants in the Gulf of Maine. *Journal of Animal Ecology*, 75, 565-574.

El-Rashidy, H.H. & Boxshall, G.A. (2009). Parasites gained: alien parasites switching to native hosts. *Journal of Parasitology*, 95, 1326-1329.

Elser, J.J., Luecke, C., Brett, M.T. & Goldman, C.R. (1995). Effects of food web compensation after manipulation of rainbow trout in an oligotrophic lake. *Ecology*, 76, 52-69.

Enell, M. (1995). Environmental-impact of nutrients from Nordic fish farming. *Water Science and Technology*, 31, 61-71.

Englbrecht, C., Schliewen, U. & Tautz, D. (2002). The impact of stocking on the genetic integrity of Arctic charr (*Salvelinus*) populations from the Alpine region. *Molecular Ecology*, 11, 1017-1027.

Etheridge, E.C., Bean, C.W., Maitland, P.S. & Adams, C.E. (2010). Morphological and ecological responses to a conservation translocation of powan (*Coregonus lavaretus*) in Scotland. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 274-281.

Etheridge, E.C., Bean, C.W. & Adams, C.E. (2011). An experimental approach to estimating vulnerability of European whitefish (*Coregonus lavaretus*) ova to predation by invasive ruffe (*Gymnocephalus cernuus*). *Ecology of Freshwater Fish*.

Evans, D.W. & Matthews, M.A. (1999). *Anguillicola crassus* (Nematoda, Dracunculoidea); first documented record of this swimbladder parasite of eels in Ireland. *Journal of Fish Biology*, 55, 665-668.

Evans, L.H. & Edgerton, B.F. (2002). Pathogens, parasites and commensals. In *Biology of Freshwater Crayfish* (ed. D.M. Holdich), pp. 377-438. Blackwell Science, Oxford.

FAO (Food & Agricultural Organisation of the United Nations) (1999). Global characterisation of inland fishery enhancements and associated environmental impacts. *FAO Fisheries Circular 945, FAO Inland Water Resources & Aquaculture Service, Fishery Resource Division.* FAO, Rome.

Fausch, K.D. (2007). Introduction, establishment and effects of non-native salmonids: considering the risk of rainbow trout invasion in the United Kingdom. *Journal of Fish Biology*, 71 (Suppl. D), 1-32.

Feltham, M.J., Davies, J.M., Wilson, B.R., Holden, T., Cowx, I.G., Harvey, J.P. & Britton, J.R. (1999). Case studies of the impact of fish-eating birds on inland fisheries in England and Wales. *Ministry of Agriculture*, *Fisheries & Food Project VC0106*. MAFF, London.

Ferguson, A. (2003). Brown trout genetic diversity: origins, importance and the impacts of supplemental stocking. In *Proceedings of the Institute of Fisheries Management 34th Annual Study Course*, pp. 32-49. *IFM, Nottingham*.

Ferguson, A. (2004). Genepool. *Salmo trutta*, 7, 48-51.

Ferguson, A. (2007). Genetic impacts of stocking on indigenous brown trout populations. *Environment Agency Science Report SC040071/SR*. EA, Bristol.

Ferguson, A., Fleming, I., Hindar, K., Skaala, Ø., McGinnity, P., Cross, T. & Prodöhl, P. (2007). Farm escapes. In *The Atlantic Salmon: Genetics*, *Conservation and Management* (eds E. Verspoor, L. Stradmeyer and J. Nielsen), pp. 367-409. Blackwell Publishing, Oxford.

Fernandez-Jover, D., Sanchez-Jerez, P., Bayle-Sempere, J., Carratala, A. & Leon, V.M. (2007). Addition of dissolved nitrogen and dissolved organic carbon from wild fish faeces and food around Mediterranean fish farms: implications for waste-dispersal models. *Journal of Experimental Marine Biology and Ecology*, 340, 160-168.

Fickling, N.J. & Lee, R.L.G. (1983). A review of the ecological impact of the introduction of the zander (*Stizostedion lucioperca* L.) into waters of the Eurasian mainland. *Fisheries Management*, 14, 151-155.

Fielder, D.S., Bardsley, W.J. & Allan, G.L. (1999). Enhancement of mulloway (*Argyrosomus japonicus*) in intermittently opening lagoons. *Fisheries Research & Development Corporation Project No 95/148, New South Wales Fisheries Final Report Series 14*. FRDC, New South Wales.

Fisher, K.A.M. & Broughton, N.M. (1984). The effect of cyprinid introductions on angler success in the River Derwent, Derbyshire. *Fisheries Management*, 15, 35-40.

Fjälling, A. & Fürst, M. (1987). The development of a fishery for the crayfish *Pacifastacus leniusculus* in Sweden 1960-1986. *Freshwater Crayfish*, 7, 223-230.

Fjellheim, A., Raddum, G.G. & Saegrov, H. (1993). Stocking experiments with wild brown trout (*Salmo trutta*) in two mountain reservoirs. In *The Rehabilitation of Freshwater Fisheries* (ed. I.G. Cowx), pp. 268-279. Fishing News Books, Blackwell Scientific Publications, Oxford.

Fleming, I.A. (1995). Reproductive success and the genetic threat of cultured fish to wild populations. In *Protection of Aquatic Biodiversity* (eds D.P. Philipp, J.M. Epifanio, J.E. Marsden, J.E. Claussen & R.J. Wolotira Jr.), pp. 117-135. Proceedings of the World Fisheries Congress, Theme 3. Oxford & IBH Publishing Co. Pvt., New Delhi.

Fleming, I.A. & Gross, M.R. (1993). Breeding success of hatchery and wild coho salmon (*Oncorhynchus kisutch*) in competition. *Ecological Applications*, 3, 230-245.

Fleming, I.A., Hindar, K., Mjolnerod, I.B., Jonsson, B., Balstad, T. & Lamberg, A. (2000). Lifetime success and interactions of farm salmon invading a native population. *Proceedings of the Royal Society of London Series B*, 267, 1517-1523.

Fletcher, A.R. (1979). Effects of *Salmo trutta* on *Galaxias olidus* and macroinvertebrates in stream communities. *M.Sc. Thesis*, *Monash University*, *Victoria*.

Fletcher, A.R. (1986). Effects of introduced fish in Australia. In *Limnology in Australia* (eds P. DeDeckker & W.D. Williams), pp. 231-238. CSIRO, Melbourne and Dr W. Junk, Dordecht.

Fletcher, A.R., Morison, A.K. & Hulme, D.J. (1985). Effects of carp, *Cyprinus carpio* L. on communities of aquatic vegetation and turbidity of water bodies in the lower Goulburn river basin. *Australian Journal of Marine and Freshwater Research*, 36, 311-327.

Fozzard, I., Doughty, R., Ferrier, R.C., Leatherland, T. & Owen, R. (1999). A quality classification for management of Scottish standing waters. *Hydrobiologia*, 395/396, 433-453.

Frankiewicz, P., Dąbrowski, K. & Zalewski, M. (1996). Mechanism of establishing bimodality in a size distribution of age-0 pikeperch, *Stizostedion lucioperca* (L.) in the Sulejow reservoir, central Poland. *Annales Zoologici Fennici*, 33, 321-327.

Frankiewicz, P., Dąbrowski, K., Martyniak, A. & Zalewski, M. (1999). Cannibalism as a regulatory force of pikeperch, *Stizostedion lucioperca* (L.), population dynamics in the lowland Sulejow reservoir (Central Poland). *Hydrobiologia*, 408/409, 47-55.

Fraser, D. & Adams, C.E. (1997). A crucian carp *Carassius carassius* (L.) in Loch Rannoch, Scotland: further evidence of the threat posed to unique fish communities by introduction of alien fish species. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 7, 323-326.

Gamradt, S.C. & Kats, L.B. (1996). Effect of introduced crayfish and mosquitofish on California newts. *Conservation Biology*, 10, 1155-1162.

García-Berthou, E. (2002). Ontogenetic diet shifts and interrupted piscivory in introduced largemouth bass (*Micropterus salmoides*). *International Review of Hydrobiology*, 87, 353-363.

García-Berthou, E. & Moreno-Amich, R. (2000). Food of introduced pumpkinseed sunfish: ontogenetic diet shift and seasonal variation. *Journal of Fish Biology*, 57, 29-40.

García de Leániz, C. & Verspoor, E. (1989). Natural hybridization between Atlantic salmon, *Salmo salar*, and brown trout, *Salmo trutta*, in northern Spain. *Journal of Fish Biology*, 34, 41-46.

Garcia de Leániz, C., Verspoor, E. & Hawkins, A.D. (1989). Genetic determination of the contribution of stocked and wild Atlantic salmon *Salmo salar* L., to the angling fisheries in two Spanish rivers. *Journal of Fish Biology*, 35 (Suppl. A), 261-270.

Garcia-Marin, J.L., Jorde, P.E., Ryman, N., Utter, F. & Pla, C. (1991). Management implications of genetic differentiation between native and hatchery populations of brown trout (*Salmo trutta*) in Spain. *Aquaculture*, 95, 235-249.

Garcia-Marin, J.L., Sanz, N & Pla, C. (1998). Proportions of native and introduced brown trout in adjacent fished and unfished Spanish rivers. *Conservation Biology*, 12, 313-319.

Garner, P., Bass, J.A.B. & Collett, G.D. (1996). The effects of weed cutting upon the biota of a large regulated river. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 6, 21-29.

Geist, D.R., Scholz, A.T. & Soltero, R.A. (1993). Relationship between phytoplankton volume and rainbow trout-*Daphnia pulex* interactions after phosphorus inactivation, Medical Lake, Washington. *Journal of Freshwater Ecology*, 8, 341-353.

Geist, J., Porkka, M. & Kuehn, R. (2006). The status of host fish populations and fish species richness in European freshwater pearl mussel (*Margaritifera margaritifera*) streams. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 251-266.

Gliwicz, Z.M. (2001). Species-specific population-density thresholds in cladocerans? *Hydrobiologia*, 442, 291-300.

Goodwin, C.E., Griffiths, D., Dick, J.T.A. & Elwood, R.W. (2006). A freshwater-feeding *Lampetra fluviatilis L.* population in Lough Neagh, Northern Ireland. *Journal of Fish Biology*, 68, 628-633.

Goulder, R. (2001). Angling and species richness of aquatic macrophytes in ponds. *Freshwater Forum*, 15, 71-76.

Gozlan, R.E. (2008). Introduction of non-native freshwater fish: is it all bad? *Fish and Fisheries*, 8, 1-10.

Gozlan, R.E., St-Hilaire, S., Feist, S.W., Martin, P. & Kent, M.L. (2005). Disease threat to European fish. *Nature*, 435, 1046.

Gozlan, R.E., Andreou, D, Asaeda, T., Beyer, K., Bouhadad, R., Burnard, D., Caiola, N., Cakic, P., Djikanovic, V., Esmaeili, H.R., Falka, I., Golicher, D., Harka, A., Jeney, G., Kovac, V., Musil, J., Nocita, A., Povz, M., Poulet, N., Virbickas, T., Wolter, T., Tarkan, S., Tricarico, E., Verreycken, H., Witkowski, A., Zhang, C.G., Zweimuelle, I. & Britton, J.R. (2010). Pan-continental invasion of *Pseudorasbora parva*: towards a better understanding of freshwater fish invasions. *Fish and Fisheries*, 11, 315-340.

Green, D.M. Jr. (1964). A comparison of stamina of brook trout from wild and domestic parents. *Transactions of the American Fisheries Society*, 93, 96-100.

Grey, J., Thackeray, S.J., Jones, R.I. & Shine, A. (2002). Ferox trout (*Salmo trutta*) as ‘Russian dolls’: complementary gut content and stable isotope analyses of the Loch Ness foodweb. *Freshwater Biology*, 47, 1235-1243.

Grey, J., Waldron, S. & Hutchinson, R. (2004). The utility of carbon and nitrogen isotope analyses to trace contributions from fish farms to the receiving communities of freshwater lakes: a pilot study in Esthwaite Water, UK. *Hydrobiologia*, 524, 253-262.

Griffiths, D. (2006). The direct contribution of fish to lake phosphorus cycles. *Ecology of Freshwater Fish*, 15, 86-95.

Griffiths, A.M.; Bright, D. & Stevens, J.R. (2009) Comparison of patterns of genetic variability in wild and supportively bred stocks of brown trout, *Salmo trutta. Fisheries Management and Ecology*, **16,** 514-519.

Gross, M. (1998). One species with two biologies: Atlantic salmon (*Salmo salar*) in the wild and in aquaculture. *Canadian Journal of Fisheries and Aquatic Sciences*, 55 (Suppl. 1), 131-144.

Guan, R. & Wiles, P.R. (1997). Ecological impact of introduced crayfish on benthic fishes in a British lowland river. *Conservation Biology*, 11, 641-647.

Guan, R. & Wiles, P.R. (1998). Feeding ecology of the signal crayfish *Pacifastacus leniusculus* in a British lowland river. *Aquaculture*, 169, 177-193.

Gunn, I.D.M., O'Hare, M.T., Maitland, P.S. & May, L. (2012). Long-term trends in Loch Leven invertebrate communities. *Hydrobiologia*, in press.

Guyomard, R. (1989). Diversité génétique de la truite commune. *Bulletin Français de la Pêche et de la Pisciculture*, 93, 118-135.

Hanazato, T., Fueki, K. & Yoshimoto, M. (2001). Fish-induced life-history shifts in the cladocerans *Daphnia* and *Simocephalus*: are they positive or negative responses? *Journal of Plankton Research*, 23, 945-951.

Hänfling, B. (2007). Understanding the establishment success of non-indigenous fishes: lessons from population genetics. *Journal of Fish Biology*, 71 (Suppl. D), 115-135.

Hänfling, B., Bolton, P., Harley, M. & Carvalho, G.R. (2005). A molecular approach to detect hybridisation between crucian carp (*Carassius carassius*) and non indigenous carp species (*Carassius* spp. and *Cyprinus carpio*). *Freshwater Biology*, 50, 403-417.

Hansen, L.P., Jonsson, B. & Andersen, R. (1989). Salmon ranching experiments in the River Imsa: is homing dependent on sequential imprinting of the smolts? In *Proceedings of the Salmonid Migration and Distribution Symposium* (eds E. Brannon & B. Jonsson), pp. 19-29. School of Fisheries, University of Washington, Seattle.

Hansen, L.P., Jacobsen, J.A. & Lund, R.A. (1993). High numbers of farmed Atlantic salmon, *Salmo salar* L., observed in oceanic waters north of the Faroe Islands. *Aquaculture and Fisheries Management*, 24, 777-781.Hansen, M.M. (2002). Estimating the long-term effects of stocking domesticated trout into wild brown trout (*Salmo trutta*) populations: an approach using microsatellite DNA analysis of historical and contemporary samples. *Molecular Ecology*, 11, 1003-1015.

Hansen, M.M., Ruzzante, D.E., Nielson, E.E. & Mensberg, K.L.D. (2000). Microsatellite and mitochondrial DNA polymorphism reveals life-history dependent interbreeding between hatchery and wild brown trout (*Salmo trutta* L.). *Molecular Ecology*, 9, 583-594.

Hansen, M.M., Meier, K. & Mensberg, K.L.D. (2010). Identifying footprints of selection in stocked brown trout populations: a spatio-temporal approach. *Molecular Ecology*, 19, 1787-1800.

Hansson, L.A., Nicolle, A., Brodersen, J., Romare, P., Nilsson, P.A., Brönmark, C. & Skov, C. (2007). Consequences of fish predation, migration, and juvenile ontogeny on zooplankton spring dynamics. *Limnology and Oceanography*, 52, 696-706.

Harada Y. & Matsumiya Y. (1992) A theoretical study on resource enhancement by stocking, with special reference to its intergenerational effects. Nippon Suisan Gakkaishi 58, 1833–1842.

Harada, Y., Yokota, M. & Iizuka, M. (1998). Genetic risk of domestication in artificial fish stocking and its possible reduction. *Researches on Population Ecology*, 40, 311-324.

Harcup, M.F., Williams, R. & Ellis, D.M. (1984). Movements of brown trout, *Salmo trutta* L., in the River Gwyddon, South Wales. *Journal of Fish Biology*, 24, 415-426.

Harvey, J.P. & Cowx, I.G. (2003). Monitoring the river, brook and sea lamprey. *Conserving Natura 2000 Rivers Monitoring Series No. 5.* English Nature, Peterborough.

Harvey, J.P., Noble, R.A.A., Nunn, A.D., Taylor, R.J. & Cowx, I.G. (2010). Monitoring sea lamprey *Petromyzon marinus* ammocoetes in SAC rivers: a case study on the River Wye. In *Conservation Monitoring in Freshwater Habitats – a Practical Guide and Case Studies* (eds C. Hurford, M. Schneider & I.G. Cowx), pp. 195-208. Springer, London.

Hastie, L.C. & Cosgrove, P.J. (2001). The decline of migratory salmonid stocks: a new threat to pearl mussels in Scotland. *Freshwater Forum*, 15, 85-96.

Hastie, L.C. & Young, M.R. (2003). Conservation of the freshwater pearl mussel. 2. Relationship with salmonids. *Conserving Natura 2000 Rivers Conservation Techniques Series No. 3*. English Nature, Peterborough.

Hastie, L.C., Young, M.R., Boon, P.J., Cosgrove, P.J. & Henninger, B. (2000). Sizes, densities and age structures of Scottish *Margaritifera margaritifera* (L.) populations. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 10, 229-247.

Hastie, L.C., Cooksley, S.L., Scougall, F., Young, M.R., Boon, P.J. & Gaywood, M.J. (2003). Characterization of freshwater pearl mussel (*Margaritifera margaritifera*) riverine habitat using River Habitat Survey data. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 13, 213-224.

Hearn, W.E. & Kynard, B.E. (1986). Habitat utilization and behavioral interaction of juvenile Atlantic salmon (*Salmo salar*) and rainbow trout (*S. gairdneri*) in tributaries of the White River of Vermont. *Canadian Journal of Fisheries and Aquatic Sciences*, 43, 1988-1998.

Heggberget, T.G., Johnsen, B.O., Hindar, K., Jonsson, B., Hansen, L.P., Hvidsten, N.A. & Jensen, A.J. (1993). Interactions between wild and cultured Atlantic salmon: a review of the Norwegian experience. *Fisheries Research*, 18, 123-146.

Hegge, O., Hesthagen, T. & Skurdal, J. (1993). Juvenile competitive bottleneck in the production of brown trout in hydroelectric reservoirs due to intraspecific habitat segregation. *Regulated Rivers: Research and Management*, 8, 41-48.

Heggenes, J. & Saltveit, S.J. (1990). Seasonal and spatial microhabitat selection and segregation in young Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., in a Norwegian river. *Journal of Fish Biology*, 36, 707-720.

Hembre, L.K. & Megard, R.O. (2005). Timing of predation by rainbow trout controls *Daphnia* demography and the trophic status of a Minnesota lake. *Freshwater Biology*, 50, 1064-1080.

Hemmer, H. (1990). *Domestication: the Decline of Environmental Appreciation*. Cambridge University Press, Cambridge.

Hemmingsen, A.R., Holt, F.A., Ewing, R.D. & McIntyre, J.D. (1986). Susceptibility of progeny from crosses among three stocks of coho salmon to infection by *Ceratomyxa shasta*. *Transactions of the American Fisheries Society*, 115, 492-495.

Hershberger, P.K., van der Leeuw, B.K., Gregg, J.L., Grady, C.A., Lujan, K.M., Gutenberger, S.K., Purcell, M.K., Woodson, J.C., Winton, J.R. & Parsley, M.J. (2010). Amplification and transport of an endemic fish disease by an introduced species. *Biological Invasions*, 12, 3665-3675.

Hershberger, W.K. (2002). Genetic changes in marine aquaculture species and the potential for impacts on natural populations. In *Responsible Marine Aquaculture* (eds S.R.R. tickney & J.P. McVey), pp. 221-231. CABI Publishing, Wallingford.

Hickley, P. (1986). Invasion by zander and the management of fish stocks. *Philosophical Transactions of the Royal Society of London (Series B)*, 314, 571-582.

Hickley, P. (1994). Stocking and introduction of fish - a synthesis. In *Rehabilitation of Freshwater Fisheries* (ed. I.G. Cowx), pp. 247-254. Fishing News Books, Blackwell Scientific Publications, Oxford.

Hickley, P. & Chare, S. (2004). Fisheries for non-native species in England and Wales: angling or the environment? *Fisheries Management and Ecology*, 11, 203-212.

Hindar, K. (1992). Conservation and sustainable use of Atlantic salmon. In *Conservation of Biodiversity for Sustainable Development* (eds O.T. Sandlund, K. Hindar & A.H.D. Brown). Scandinavian University Press, Oslo.

Hindar, K. & Balstad, T. (1994). Salmonid culture and interspecific hybridization. *Conservation Biology*, 8, 881-882.

Hindar, K., Ryman, N. & Utter, F. (1991). Genetic effects of cultured fish on natural fish populations. *Canadian Journal of Fisheries and Aquatic Sciences*, 48, 945-957.

Hoffman, J.C., Smith, M.E. & Lehman, J.T. (2001). Perch or plankton: top-down control of *Daphnia* by yellow perch (*Perca flavescens*) or *Bythotrephes cederstroemi* in an inland lake? *Freshwater Biology*, 46, 759-775.

Holcík, J. (1991). Fish introductions in Europe with particular reference to its central and eastern part. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 13-23.

Holdich, D. (2003). Ecology of the white-clawed crayfish. *Conserving Natura 2000 Rivers Ecology Series No. 1.* English Nature, Peterborough.

Holdich, D.M. & Reeve, I.D. (1991). Distribution of freshwater crayfish in the British Isles, with particular reference to crayfish plague, alien introductions and water quality. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 1, 139-158.

Holdich, D.M. & Rogers, W.D. (1997). The white-clawed crayfish, *Austropotamobius pallipes*, in Great Britain and Ireland with particular reference to its conservation in Great Britain. *Bulletin Français de la Pêche et de la Pisciculture*, 347, 597-616.

Holdich, D.M., Reynolds, J.D., Souty-Grosset, C. & Sibley, P.J. (2009). A review of the ever increasing threat to European crayfish from non-indigenous crayfish species. *Knowledge and Management of Aquatic Ecosystems*, 394-395, 1-46.

Holmlund, C.M. & Hammer, M. (2004). Effects of fish stocking on ecosystem services: An overview and case study using the Stockholm archipelago. *Environmental Management*, 33, 799-820.

Honkanen, T. & Helminen, H. (2000). Impacts of fish farming on eutrophication: comparisons among different characteristics of ecosystem. *International Review of Hydrobiology*, 85, 673-686.

Hrbáček, J., Dvorakova, M., Korínek, V. & Prochàzkóva, L. (1961). Demonstration of the effect of fish stock on the species composition of zooplankton and the intensity of metabolism of the whole plankton association. *Verhandlungen der Internationalen Vereinigung für theoretische und angewandte Limnologie*, 14, 192-195.

Hülsmann, S., Vijverberg, J., Boersma, M. & Mooij, W. M. (2004). Effects of infochemicals released by gape-limited fish on life history traits of *Daphnia*: a maladaptive response? *Journal of Plankton Research*, 26, 535-543.

Hyvärinen, P. & Huusko, A. (2006). Diet of brown trout in relation to variation in abundance and size of pelagic fish prey. *Journal of Fish Biology*, 68, 87-98.

Ibbotson, A. & Klee, C. (2002). Impacts and subsequent control of an introduced predator: the case of pike, *Esox lucius*, in Chew Valley Lake. In *Management and Ecology of Lake and Reservoir Fisheries* (ed. I.G. Cowx), pp. 203-216. Fishing News Books, Blackwell Science, Oxford.

ICES (International Council for Exploration of the Sea) (2005). *ICES Code of Practice on the Introductions and Transfers of Marine Organisms*. ICES, Copenhagen. http://www.ices.dk/pubs/Miscellaneous/ICESCodeofPractice.pdf.

IFM (Institute of Fisheries Management) (1991). The creation and management of pond fisheries. Institute of Fisheries Management, Nottingham.

Institute of Aquaculture (1990) *Fish farming and the Scottish freshwater environment*. Report prepared for the Nature Conservancy Council by: Institute of Aquaculture, University of Stirling; Institute of Freshwater Ecology, Bush Estate, Penicuik; and Instituite of Terrestrial Ecology, Banchory. Nature Conservancy Council, Edinburgh.

IUCN (International Union for Conservation of Nature) (1987). *Position Statement on Translocation of Living Organisms*. IUCN, Switzerland. http://www.iucn-org.ac.psiweb.com/themes/ssc/pubs/policy/transe.htm.

IUCN/SSC (International Union for Conservation of Nature) (1995). *Guidelines for Reintroductions*. IUCN, Switzerland. http://www.iucn-org.ac.psiweb.com/themes/ssc/pubs/policy/reinte.htm.

Ivantsoff, W.A. (1999). Detection of predation on Australian native fishes by *Gambusia holbrooki*. *Marine and Freshwater Research*, 50, 467-468.

Izquierdoa, J.I., Castilloa, A.G.F., Ayllona, F., de la Hozb, J. & Garcia-Vazqueza, E. (2006). *Environmental Biology of Fishes*, 75,153-157.

Jackson, P.D. & Williams, W.D. (1980). Effects of brown trout *Salmo trutta* L. on the distribution of some native fishes in three areas of southern Victoria. *Australian Journal of Marine and Freshwater Research*, 31, 61-67.

Jang, M.H. & Lucas, M.C. (2005). Reproductive ecology of the river lamprey. *Journal of Fish Biology*, 66, 499-512.

Jeppesen, E., Søndergaard, M., Jensen, J., Mortensen, E. & Sortkjaer, O. (1996). Fish-induced changes in zooplankton grazing on phytoplankton and bacterioplankton: a long-term study in shallow hypertrophic Lake Sobygaard. *Journal of Plankton Research*, 18, 1605-1625.

Johnes, P., Moss, B. & Phillips, G. (1996). The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: testing of a model for use in conservation and water quality management. *Freshwater Biology*, 36, 451-473.

Johnsen, B.O. & Jensen, A.J. (1991). The *Gyrodactylus* story in Norway. *Aquaculture*, 98, 289-302.

Jokikokko, E. (1999). Density of brown trout, *Salmo trutta* L., and Atlantic salmon, *Salmo salar* L., parr after point and scatter stocking of fry. *Fisheries Management and Ecology*, 6, 475-486.

Jones, J.G. (1990). Pollution from fish farms. *Water and Environment Journal*, 4, 14-18.

Jones, M.L. & Stanfield, L.W. (1993). Effects of exotic juvenile salmonines on growth and survival of juvenile Atlantic salmon (*Salmo salar*) in a Lake Ontario tributary. *Canadian Special Publication in Fisheries and Aquatic Sciences*, 118, 71-79.

Jones, R.I., Grey, J., Quarmby, C. & Sleep, D. (1998). An assessment using stable isotopes of the importance of allochthonous organic carbon sources to the pelagic food web in Loch Ness. *Proceedings of the Royal Society of London B*, 265, 105-111.

Jonsson, B. & Fleming, I.A. (1993). Enhancement of wild salmon populations. In *Human Impact on Self-recruiting Populations* (ed. G. Sundnes), pp. 209-238. Symposium of the Royal Norwegian Society of Sciences & Letters. Tapir Press, Trondheim.

Jonsson, B., Jonsson, N. & Hansen, L.P. (1990). Does juvenile experience affect migration and spawning of adult Atlantic salmon? *Behavioural Ecology and Sociobiology*, 26, 225-230.

Jonsson, B., Jonsson, N. & Hansen, L.P. (1991). Differences in life history and migratory behaviour between wild and hatchery-reared Atlantic salmon in nature. *Aquaculture*, 98, 69-78.

Jørstad, K.E., Skaala, O & Naedal, G. (1999). Genetic diversity and the Norwegian sea ranching program: an evaluation. In *Stock Enhancement and Sea Ranching* (eds B.R. Howell, E. Maksness & T. Svasand), pp. 49-62. Blackwell Publishing, Oxford.

Jupp, B.P. & Spence, D.H.N. (1977). Limitations on macrophytes in a eutrophic lake, Loch Leven: I. Effects of phytoplankton. *Journal of Ecology*, 65, 175-186.

Jusup, M., Geček, S. & Legović, T. (2007). Impact of aquacultures on the marine ecosystem: modelling benthic carbon loading over variable depth. *Ecological Modelling*, 200, 459-466.

Kangur, A., Kangur, P., Kangur, K., Järvalt, A. & Haldna, M. (2010). *Anguillicoloides crassus* infection of European eel, *Anguilla anguilla* (L.), in inland waters of Estonia: history of introduction, prevalence and intensity. *Journal of Applied Ichthyology*, 26 (Suppl. 2), 74-80.

Karjalainen, J., Miserque, D. & Huuskonen, H. (1997). The estimation of food consumption in larval and juvenile fish: Experimental evaluation of bioenergetics models. *Journal of Fish Biology*, 51, 39-51.

Kelly-Quinn, M. & Bracken, J.J. (1989). Survival of stocked hatchery-reared brown trout, *Salmo trutta* L., fry in relation to carrying capacity of a trout nursery stream. *Aquaculture and Fisheries Management*, 20, 211-226.

Kennedy, C.R. (1993). Introductions, spread and colonization of new localities by fish helminth and crustacean parasites in the British Isles: a perspective and appraisal. *Journal of Fish Biology*, 43, 287-301.

Kennedy, C.R. & Fitch, D.J. (1990). Colonization, larval survival and epidemiology of the nematode *Anguillicola crassus*, parasitic in the eel, *Anguilla anguilla* in Britain and Ireland: an hypothesis. *Journal of Fish Biology*, 36, 117-131.

Khan, T.A., Wilson, M.E. & Khan, M.I. (2003). Evidence for invasive carp mediated trophic cascade in shallow lakes of western Victoria, Australia. *Hydrobiologia*, 506, 465-472.

King, A.J. (2005). Ontogenetic dietary shifts of fishes in an Australian floodplain river. *Marine and Freshwater Research*, 56, 215-225.

Kirk, R.S., Lewis, J.W. & Russell, I.C. (2003). The role of cormorants in transmission of parasitic diseases of fish. In *Proceedings of the Institute of Fisheries Management 34th Annual Study Course*, pp. 115-121. IFM, Nottingham.

Klemetsen, A., Amundsen, P.A., Dempson, J.B., Jonsson, B., Jonsson, N., O'Connell, M.F. & Mortensen, E. (2003). Atlantic salmon *Salmo salar* L., brown trout *Salmo trutta* L. and Arctic charr *Salvelinus alpinus* (L.): a review of aspects of their life histories. *Ecology of Freshwater Fish*, 12, 1-59.

Kloskowski, J. (2000). Selective predation by otters *Lutra lutra* on common carp *Cyprinus carpio* at farmed fisheries. *Mammalia*, 64, 287-294.

Kloskowski, J. (2005). Otter *Lutra lutra* damage at farmed fisheries in southeastern Poland, II: exploitation of common carp *Cyprinus carpio*. *Wildlife Biology*, 11, 257-261.

Kloskowski, J. (2011). Impact of common carp *Cyprinus carpio* on aquatic communities: direct trophic effects versus habitat deterioration. *Fundamental and Applied Limnology*, 178, 245-255.

Kohler, C.S. & Stanley, J.G. (1984). A suggested protocol for evaluating proposed exotic fish introductions in the United States. In *Distribution*, *Biology and Management of Exotic Fishes* (eds W.R. Courtenay Jr. & J.R. Stauffer Jr.), pp. 387-406. John Hopkins University Press, Baltimore.

Korponai, J., Mátyás, K., Paulovits, G., Tátrai, I. & Kovács, N. (1997). The effect of different fish communities on the cladoceran plankton assemblages of the Kis-Balaton Reservoir, Hungary. *Hydrobiologia*, 360, 211-221.

Kortan, J. & Adámek, Z. (2011). Behavioural response of carp (*Cyprinus carpio*, L.) pond stock upon occurrence of hunting great cormorant (*Phalacrocorax carbo sinensis*) flocks. *Aquaculture International*, 19,121-129.

Kortan, J., Adámek, Z. & Polakova, S. (2007). Winter predation by otter, *Lutra lutra* on carp pond systems in South Bohemia (Czech Republic). *Folia Zoologica*, 56, 416-428.

Kortan, J., Adámek, Z. & Vrana, P. (2010). Otter, *Lutra lutra*, feeding pattern in the Kamenice River (Czech Republic) with newly established Atlantic salmon, *Salmo salar*, population. *Folia Zoologica*, 59, 223-230.

Krueger, C.C. & May, B. (1991). Ecological and genetic effects of salmonid introductions in North America. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 66-77.

Kubečka, J., Seda, J. & Matěna, J. (1998). Fish-zooplankton interactions during spring in a deep reservoir. *International Review of Hydrobiology*, 83, 431-441.

L'Abee-Lund, J.H. (1991). Stocking of hatchery-reared fish: an enhancement method? *Fauna*, 44, 173-180.

Langeland, A., L'Abee-Lund, J. H., Jonsson, B. & Jonsson, N. (1991). Resource partitioning and niche shift in Arctic charr *Salvelinus alpinus* and brown trout *Salmo trutta*. *Journal of Animal Ecology*, 60, 895-912.

Lansdown, R.V. & Wade, P.M. (2003). Ecology of the floating water-plantain, *Luronium natans*. *Conserving Natura 2000 Rivers Ecology Series No. 9.* English Nature, Peterborough.

Lathrop, R.C., Johnson, B.M., Johnson, T.B., Vogelsang, M.T., Carpenter, S.R., Hrabik, T.R., Kitchell, J.F., Magnuson, J.J., Rudstam, L.G. & Stewart, R.S. (2002). Stocking piscivores to improve fishing and water clarity: a synthesis of the Lake Mendota biomanipulation project. *Freshwater Biology*, 47, 2410-2424.

Leah, R.T., Moss, B. & Forrest, D.E. (1978). Experiments with large enclosures in a fertile, shallow brackish lake, Hickling Broad, United Kingdom. *Internationale Revue der Gesamten Hydrobiologie*, 63, 291-310.

Leary, R.F., Allendorf, F.W. & Sage, O.K. (1995). Hybridization and introgression between introduced and native fish. *American Fisheries Society Symposium*, 15, 91-101.

Leber, K.M., Brennan, N.P. & Arce, S.M. (1995). Marine enhancement with striped mullet: are hatchery releases replenishing or displacing wild stocks? *American Fisheries Society Symposium*, 15, 376-387.

Leber, K.M., Brennan, N.P. & Arce, S.M. (1998). Recruitment patterns of cultured juvenile Pacific threadfin, *Polydactylus sexfasciatus* (Polynemidae), released along sandy marine shores in Hawaii. *Bulletin of Marine Sciences*, 62, 389-408.

LeClair, L.L., Phelps, S.R. & Tynan, T.J. (1999). Little gene flow from a hatchery strain of chum salmon to local wild populations. *North American Journal of Fisheries Management*, 19, 530-535.

Lee, C.L., Purcell, S.W. & Maguire, G.B. (2001). Farming Trochus. *Aquaculture WA*, 8, 1-8.

Lefebvre, F., Fazio, G., Palstra, A.P., Székely, C. & Crivelli, A.J. (2011). An evaluation of indices of gross pathology associated with the nematode *Anguillicoloides crassus* in eels. *Journal of Fish Diseases*, 34, 31-45.

Leider, S.A., Hulett, P.L., Loch, J.J. & Chilcote, M.W. (1990). Electrophoretic comparison of the reproductive success of naturally spawning transplanted and wild steelhead trout through the returning adult stage. *Aquaculture*, 88, 239-252.

Leopold, M. & Bninska, M. (1984). The effectiveness of eel stocking in Polish lakes. *EIFAC Technical Paper*, 42 (Suppl. 1), 41-45.

Lester, L.J. (2002). What role does genetics play in responsible aquaculture. In *Responsible Marine Aquaculture* (eds R.R. Stickney & J.P. McVey), pp. 233-261. CABI Publishing, Wallingford.

Lewis, S.D. (2002). *Pacifastacus*. In *Biology of Freshwater Crayfish* (ed. D.M. Holdich), pp. 511-540. Blackwell Science, Oxford.

Li, H.W. & Moyle, P.B. (1994). Management of introduced fishes. In *Inland Fisheries Management in North America* (eds C.C. Kohler & W.A. Hubert), pp. 287-308. American Fisheries Society, Bethesda.

Liao, I.C., Su, M.S. & Laeño, A. (2003). Status of research in stock enhancement and sea ranching. *Reviews in Fish Biology and Fisheries*, 13, 151-163.

Liljendahl-Nurminen, A., Horppila, J., Malinen, T., Eloranta, P., Vinni, M., Alajärvi, E. & Valtonen, S. (2003). The supremacy of invertebrate predators over fish – factors behind the unconventional seasonal dynamics of cladocerans in Lake Hiidenvesi. *Archiv für Hydrobiologie*, 158, 75-96.

Linfield, R.S.J. (1984). The impact of zander (*Stizostedion lucioperca* (L.)) in the United Kingdom and the future management of affected fisheries in the Anglian region. *EIFAC Technical Paper*, 42 (Suppl. 2), 353-362.

Linfield, R.S.J. & Rickards, R.B. (1979). The zander in perspective. *Fisheries Management*, 10, 1-16.

Lintermans, M. (1991). The decline of native fish in the Canberra Region: the impacts of introduced species. *Bogong,* **12**, 18-22.

Linton, S. & Goulder, R. (2000). Botanical conservation value related to origin and management of ponds. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 10, 77-91.

Loneragan, N., Die, D.J., Kailis, G.M., Watson, R. & Preston, N. (1998). Developing and assessing techniques for enhancing tropical Australian prawn fisheries and the feasibility of enhancing the brown tiger prawn (*Penaeus esculentus*) fishery in Exmouth

Gulf. *Fisheries Research & Development Corporation Project No. 1998/222.* FRDC, Canberra.

Lura, H. & Økland, F. (1994). Content of synthetic astaxanthin in escaped farmed Atlantic salmon, *Salmo salar* L., ascending Norwegian rivers. *Fisheries Management and Ecology*, 1, 205 216.

Lyle, A.A. & Maitland, P.S. (1992). Conservation of freshwater fish in the British Isles: the status of fish in National Nature Reserves. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 2, 19-34.

Lynch, M. (1991). The genetic interpretation of inbreeding depression and outbreeding depression. *Evolution*, 45, 622-629.

Lynch, M. (1996). A quantitative-genetic perspective on conservation issues. In *Conservation Genetics: Case Histories from Nature* (eds J.C. Avise & J.L. Hamrick), pp. 471-501. Chapman & Hall, London.

Madeira, M.J., Gómez-Moliner, B.J. & Barbé, A.M. (2005). Genetic introgression on freshwater fish populations caused by restocking programmes. *Biological Invasions*, 7, 117-125.

Maitland, P.S. (1965). The feeding relationships of salmon, trout, minnows, stone loach and three-spined sticklebacks in the River Endrick, Scotland. *Journal of Animal Ecology*, 34, 109-133.

Maitland, P.S. (1990). *The Conservation of Rare British Freshwater Fish.* Nature Conservancy Council, Edinburgh.

Maitland, P.S. (1995). The conservation of freshwater fish: past and present experience. *Biological Conservation*, 72, 259-270.

Maitland, P.S. (2003). Ecology of the river, brook and sea lamprey. *Conserving Natura 2000 Rivers Ecology Series No. 5.* English Nature, Peterborough.

Maitland, P.S. (2004). Keys to the freshwater fish of Britain and Ireland, with notes on their distribution and ecology. *Freshwater Biological Association Scientific Publication No. 62.* FBA, Ambleside.

Maitland, P.S. & Campbell, R.N. (1992). *Freshwater Fishes of the British Isles*. Harper Collins, London.

Maitland, P.S. & Lyle, A.A. (1990). Practical conservation of British fishes: current action on six declining species. *Journal of Fish Biology*, 37 (Suppl. A), 255-256.

Maitland, P.S., Morris, K.H. & East, K. (1994). The ecology of lampreys (Petromyzonidae) in the Loch Lomond area. *Hydrobiologia*, 290, 105-120.

Maitland, P.S., Winfield, I.J., McCarthy, I.D. & Igoe, F. (2007). The status of Arctic charr *Salvelinus alpinus* in Britain and Ireland. *Ecology of Freshwater Fish*, 16, 6-19.

Mann, R.H.K. & Orr, D.R.O. (1969). A preliminary study of the feeding relationships of fish in a hard-water and soft-water stream in southern England. *Journal of Fish Biology*, 1, 31-44.

Marsden, M.W. & Mackay, D.W. (2001). Water quality in Scotland: the view of the regulator. *Science of the Total Environment*, 265, 369-386.

Marsh, K. (2000). Cowichan River days. *Fly Rod & Reel Magazine*, March/April.

Matthews, M.A., Poole, W.R., Thompson, C.E., McKillen, J., Ferguson, A., Hindar, K. & Wheelan, K.F. (2000). Incidence of hybridization between Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., in Ireland. *Fisheries Management and Ecology*, 7, 337-347.

Mátyás, K., Korponai, J., Tátrai, I. & Paulovits, G. (2004). Long term effect of cyprinid fishes on phytoplankton and zooplankton communities in a shallow water protection reservoir. *International Review of Hydrobiology* 89, 68-78.

May, L., Bailey-Watts, A.E. & Kirika, A. (2001). The relationship between *Trichocerca pusilla* (Jennings), *Aulacoseira* spp. and water temperature in Loch Leven, Scotland, U.K. *Hydrobiologia*, 446/447, 29-34.

May, L., Defew, L.H., Bennion, H. & Kirika, A. (2012). Historical changes (1905-2005) in external phosphorus loads to Loch Leven, Scotland, UK. *Hydrobiologia*, in press.

McDowall, R.M. (1990). When galaxiid and salmonid fishes meet – a family reunion in New Zealand. *Journal of Fish Biology*, 37 (Suppl. A), 35-43.

McDowall, R.M. (2003). Impacts of introduced salmonids on native galaxiids in New Zealand upland streams: a new look at an old problem. *Transactions of the American Fisheries Society*, 132, 229-238.

McDowall, R.M. (2006). Crying wolf, crying foul, or crying shame: alien salmonids and a biodiversity crisis in the southern cool-temperate galaxioid fishes? *Reviews in Fish Biology and Fisheries*, 16, 233-422.

McDowell, N. (2002). Stream of escaped farm fish raises fears for wild salmon. *Nature*, 416, 571.

McGinnity, P., Stone, C., Taggart, J.B., Cooke, D., Cotter, D. Hynes, R., McCamley, C., Cross, T. & Ferguson, A. (1997). Genetic impact of escaped farmed Atlantic salmon (*Salmo salar* L.) on native populations: use of DNA profiling to assess freshwater performance of wild, farmed, and hybrid progeny in a natural river environment. *ICES Journal of Marine Sciences*, 54, 998-1008.

McGinnity, P., Prodöhl, P., Ferguson, A., Hynes, R., Ó Maoiléidigh, N., Baker, N., Cotter, D., O'Hea, B., Cooke, D., Rogan, G., Taggart, J. & Cross, T. (2003). Fitness reduction and potential extinction of wild populations of Atlantic salmon, *Salmo salar*, as a result of interactions with escaped farm salmon *Proceedings of the Royal Society of London Series B*, 270, 2443-2450.

McGuigan, J.B. & Sommerville, C. (1985). Studies of the effects of cage culture of fish on the parasite fauna in a lowland freshwater loch in the west of Scotland. *Parasitology Research*, 71, 673-682.

McQueen, D.J. Ramcharan, C.W., Yan, N.D., Demers, E., Perez-Fuentetaja, A & Dillon, P.J. (2001). The Dorset food web piscivore manipulation project. *Archiv für Hydrobiologie Special Issues Advances in Limnology*, 56, 257-288.

Medley, P.A.H. & Lorenzen, K. (2006). *EnhanceFish: a Decision Support Tool for Aquaculture-based Fisheries Enhancement.* Software Package. Imperial College, London. http://www.aquaticresources.org/enhancefish.

Mehner, T. (1996). Predation impact of age-0 fish on a copepod population in a Baltic Sea inlet as estimated by two bioenergetics models. *Journal of Plankton Research*, 18, 1323-1340.

Mehner, T. (2000). Influence of spring warming on the predation rate of underyearling fish on *Daphnia* – a deterministic simulation approach. *Freshwater Biology*, 45, 252-263.

Mehner, T. & Thiel, R. (1999). A review of predation impact by 0+ fish on zooplankton in fresh and brackish waters of the temperate northern hemisphere. *Environmental Biology of Fishes*, 56, 169-181.

Mehner, T., Schultz, H. & Herbst, R. (1995). Interaction of zooplankton dynamics and diet of 0+ perch (*Perca fluviatilis* L.) in the top-down manipulated Bautzen reservoir (Saxony, Germany) during summer. *Limnologica*, 25, 1-9.

Mehner, T., Schultz, H., Bauer, D., Herbst, R., Voigt, H., & Benndorf, J. (1996). Intraguild predation and cannibalism in age-0 perch (*Perca fluviatilis*) and age-0 zander (*Stizostedion lucioperca*): interactions with zooplankton succession, prey fish availability and temperature. *Annales Zoologici Fennici*, 33, 353-361.

Mehner, T., Plewa, M., Hülsmann, S., Voigt, H. & Benndorf, J. (1997). Age-0 fish predation on daphnids – spatial and temporal variability in the top-down manipulated Bautzen reservoir, Germany. *Archiv für Hydrobiologie Special Issue Advances in Limnolog*y, 49, 13-25.

Mehner, T., Ihlau, J., Dörner, H. & Hölker, F. (2005). Can feeding of fish on terrestrial insects subsidize the nutrient pool of lakes? *Limnology and Oceanography*, 50, 2022-2031.

Mehner, T., Ihlau, J., Dörner, H., Hupfer, M. & Hölker, F. (2007). The role of insectivorous fish in fostering the allochthonous subsidy of lakes. *Limnology and Oceanography*, 52, 2718-2821.

Mehner, T., Pohlmann, K., Elkin, C., Monaghan, M.T. & Freyhof, J. (2009). Genetic mixing from enhancement stocking in commercially exploited vendace populations. *Journal of Applied Ecology*, 46, 1340-1349.

Merican, Z.O. & Phillips, M.J. (1985). Solid waste production from rainbow trout, *Salmo gairdneri* Richardson, cage culture. *Aquaculture and Fisheries Management*, 16, 55-70.

Michel, C., Faivre, B. & De Kinkelin, P. (1986). A clinical case of enteric red-mouth in minnows (*Pimephales promelas*) imported into Europe as bait-fish. *Bulletin of the European Association of Fish Pathologists*, 6, 97-99.

Miller, S.A. & Crowl, T.A. (2006). Effects of common carp (*Cyprinus carpio*) on macrophytes and invertebrate communities in a shallow lake. *Freshwater Biology*, 51, 85-94.

Mills, C.A., Heaney, S.I., Butterwick, C., Corry, J.E. & Elliott, J.M. (1990). Lake enrichment and the status of Windermere charr, *Salvelinus alpinus* (L.). *Journal of Fish Biology*, 37 (Suppl. A), 167-174.

Milner, N.J., Elliott, J.M., Armstrong, J.D., Gardiner, R., Welton, J.S. & Ladle, M. (2003). The natural control of salmon and trout populations in streams. *Fisheries Research*, 62, 111-125.

Milner, N.J. & Evans, R. (2004). The incidence of escaped Irish farmed salmon in English and Welsh rivers. *Fisheries Management and Ecology*, 10, 403-406.

Mo, T.A. (1992). Seasonal variations in the prevalence and infestation intensity of *Gyrodactylus salaris* Malmberg, 1957 (Monogenea, Gyrodactylidae) on Atlantic salmon parr, *Salmo salar* L., in the River Batnfjordselva, Norway. *Journal of Fish Biology*, 41, 697-707.

Molony, B.W., Lenanton, R., Jackson, G. & Norriss, J. (2003). Stock enhancement as a fisheries management tool. *Reviews in Fish Biology and Fisheries*, 13, 409-432.

Morgan, N.C. (1970). Changes in the flora and fauna of a nutrient enriched lake. *Hydrobiologia*, 35, 545-553.

Moss, B. (1988). *Ecology of Fresh Waters: Man and Medium*. Blackwell Science, Oxford.

Moss, B. (1996). A land awash with nutrients – the problem of eutrophication. *Chemistry and Industry*, June, 407-411.

Moss, B. & Leah, R.T. (1982). Changes in the ecosystem of a guanotrophic and brackish shallow lake in eastern England: potential problems in its restoration. *Internationale Revue der Gesamten Hydrobiologie*, 67, 635-659.

Moss, B., Madgwick, J. & Phillips, G. (1996). *A Guide to the Restoration of Nutrient-Enriched Shallow Lakes*. Broads Authority, Norwich.

Moss, B., Kornijów, R. & Measey, G.J. (1998). The effects of nymphaeid (*Nuphar lutea*) density and predation by perch (*Perca fluviatilis*) on the zooplankton communities in a shallow lake. *Freshwater Biology*, 39, 689-697.

Moyle, P.B., Haram, W.L. & Barton, B.B. (1986). The Frankenstein effect: impact of introduced fishes on native fishes in North America. In *Fish Culture in Fisheries Management* (ed. R.H. Stroud), pp. 415-426. American Fisheries Society, Bethesda.

Munday, B.L., Eleftheriou, A., Kentori, M. & Divanach, P. (1992). The interactions of aquaculture and the environment – a bibliographical review. *Commission of the European Communities, Directorate General of Fisheries.* EC, Brussels.

Nash, C.E. & Kensler, C.B. (1990). A global overview of aquaculture production in 1987. *World Aquaculture*, 21, 104-112.

Naslund, I. (1992). Survival and distribution of pond- and hatchery-reared 0+ brown trout, *Salmo trutta* L., released in a Swedish stream. *Aquaculture and Fisheries Management*, 23, 477-488.

Nasmith, L.E., Tonn, W.M., Paszkowski, C.A. & Scrimgeour, G.J. (2010). Effects of stocked trout on native fish communities in boreal foothills lakes. *Ecology of Freshwater Fish*, 19, 279-289.

Nielson, R.S., Reimers, N. & Kennedy, H.D. (1957). A six-year study of the survival and vitality of hatchery-reared rainbow trout of catchable size, in Convict Creek, California. *California Fish and Game*, 43, 5-42.

Niesar, M., Arlinghaus, R., Rennert, B. & Mehner, T. (2004). Coupling insights from a carp, *Cyprinus carpio*, angler survey with feeding experiments to evaluate composition, quality and phosphorus input of groundbait in coarse fishing. *Fisheries Management and Ecology*, 11, 225-235.

Nijjsen, H. & de Groot, S.J. (1974). Catalogue of fish species in the Netherlands. *Beaufortia*, 21, 173-207.

Noble, R.A.A., Cowx, I.G. & Harvey, J.P. (2004). Triploid trout in native trout waters: phase 2. Hazard identification, preliminary risk assessment and strategy to achieve full risk assessment. *Environment Agency R&D Technical Report W2-078/TR2*. EA, Bristol.

Nock, C.J., Ovenden, J.R., Butler, G.L., Wooden, I., Moore, A. & Baverstock, P.R. (2011). Population structure, effective population size and adverse effects of stocking in the endangered Australian eastern freshwater cod *Maccullochella ikei*. *Journal of Fish Biology*, 78, 303-321.

Norman, L. (1987). Stream aquarium observations of territorial behaviour in young salmon (*Salmo salar* L.) of wild and hatchery origin. *Salmon Research Institute*, *Report 2* (in Swedish, English summary). SRI, Älvkarleby.

North, R. (2001). Stocking densities for stillwater coarse fish. *Environment Agency R&D Technical Report W269*. EA, Bristol.

North, R. (2002). Factors affecting the performance of stillwater coarse fisheries in England and Wales. In *Management and Ecology of Lake and Reservoir Fisheries* (ed. I.G. Cowx), pp. 284-298. Fishing News Books, Blackwell Science, Oxford.

Northcott, D. (1979). The importance of aquatic macrophytes in the provision of crustacean zooplankton food for young roach. In *Proceedings of the 1st British Freshwater Fish Conference*, pp. 123-134. University of Liverpool, Liverpool.

Nunn, A.D., Noble, R.A.A. & Cowx, I.G. (2006). Scoping and determining the risks from trout introductions to the conservation status of SAC/SPA stillwaters. *Report to English Nature*. EN, Peterborough.

Nunn, A.D., Harvey, J.P. & Cowx, I.G. (2007a). The food and feeding relationships of larval and 0+ year juvenile fishes in lowland rivers and connected waterbodies. I. Ontogenetic shifts and interspecific diet similarity. *Journal of Fish Biology*, 70, 726-742.

Nunn, A.D., Harvey, J.P. & Cowx, I.G. (2007b). The food and feeding relationships of larval and 0+ year juvenile fishes in lowland rivers and connected waterbodies. II. Prey selection and the influence of gape. *Journal of Fish Biology*, 70, 743-757.

Nunn, A.D., Harvey, J.P., Noble, R.A.A. & Cowx, I.G. (2008a). Condition assessment of lamprey populations in the Yorkshire Ouse catchment, north-east England, and the potential influence of physical migration barriers. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 175-189.

Nunn, A.D., Noble, R.A.A., Harvey, J.P. & Cowx, I.G. (2008b). The diets and parasites of larval and 0+ juvenile twaite shad in the lower reaches and estuaries of the rivers Wye, Usk and Towy, UK. *Hydrobiologia*, 614, 209-218.

Nunn, A.D., Tewson, L.H., Taylor, M.J. & Cowx, I.G. (2011). The role of food availability during critical life-stages in determination of the survival and growth of underyearling coarse fish. *Environment Agency Science Project No. SC060048*. EA, Bristol.

Nunn, A.D., Tewson, L.H. & Cowx, I.G. (2012). The foraging ecology of larval and juvenile fishes. *Reviews in Fish Biology and Fisheries*, in press.

Nurminen, L., Horppila, J. & Tallberg, P. (2001). Seasonal development of the cladoceran assemblage in a turbid lake: the role of emergent macrophytes. *Archiv für Hydrobiologie*, 151, 127-140.

Nygren, A., Nyman, L., Svensson, K. & Jahnke, G. (1975). Cytological and biochemical studies in back-crosses between the hybrid Atlantic salmon x sea trout and its parental species. *Hereditas*, 81, 55-62.

Nyström, P. (1999). Ecological impact of introduced and native crayfish on freshwater communities: European perspectives. In *Crayfish in Europe as Alien Species: How to Make the Best of a Bad Situation* (eds F. Gherardi & D.M. Holdich), pp. 63-85. A. A. Balkema, Rotterdam.

Nyström, P. (2002). Ecology. In *Biology of Freshwater Crayfish* (ed. D.M. Holdich), pp. 192-235. Blackwell Science, Oxford.

O'Connell, M. & Wright, J.M. (1997). Microsatelite DNA in fishes. *Reviews in Fish Biology and Fisheries*, 7, 331-363.

O'Grady, M.F. (1984). The importance of genotype, size on stocking and stocking date to the survival of brown trout released into Irish lakes. *EIFAC Technical Paper*, 42 (Suppl. 1), 178-191.

Ogle, D.H. (1998). A synopsis of the biology and life history of ruffe. *Journal of Great Lakes Research*, 24, 170-185.

Orpwood, J.E., Miles, M.S., Russell, I.C. & Armstrong, J.D. (2010). Efficacy of artificial shelters for roach, *Rutilus rutilus*, against predators in the presence of reeds. *Fisheries Management and Ecology*, 17, 356-365.

Parkos, J., Santucci, V. C. & Wahl, D. (2003). Effects of common carp (*Cyprinus carpio*) on multiple trophic levels in shallow mesocosms. *Canadian Journal of Fisheries and Aquatic Sciences*, 60, 182-192.

Pawson, M.G. (2003). Triploid trout in native trout waters, phase 1: literature review and recommendations for phase 2. *Environment Agency R&D Technical Report W2-078/TR1*. EA, Bristol.

Peay, S., Guthrie, N., Spees, J., Nilsson, E. & Bradley, P. (2009). The impact of signal crayfish (*Pacifastacus leniusculus*) on the recruitment of salmonid fish in a headwater stream in Yorkshire, England. *Knowledge and Management of Aquatic Ecosystems*, 394-395, 1-15.

Pedley, R.B. & Jones, J.W. (1978). The comparative feeding behaviour of brown trout and Atlantic salmon in the Llyn Dwythwch, Wales. *Journal of Fish Biology*, 12, 239-256.

Peeler, E.J. & Feist, S.W. (2011). Human intervention in freshwater ecosystems drives disease emergence. *Freshwater Biology*, 56, 705-716.

Peeler, E.J. & Thrush, M.A. (2004). Qualitative analysis of the risk of introducing *Gyrodactylus salaris* into the United Kingdom. *Diseases of Aquatic Organisms*, 62, 103-113.

Peeler, E.J. & Thrush, M.A. (2009). Assessment of exotic fish disease introduction and establishment in the United Kingdom via live fish transporters. *Diseases of Aquatic Organisms*, 83, 85-95.

Peeler, E.J., Gardiner, R. & Thrush, M.A. (2004). Qualitative risk assessment of routes of transmission of the exotic fish parasite *Gyrodactylus salaris* between river catchments in England and Wales. *Preventive Veterinary Medicine*, 64, 175-189.

Peeler, E.J., Murray, A.G., Thebault, A., Brun, E., Giovaninni, A. & Thrush, M.A. (2007). The application of risk analysis in aquatic animal health management. *Preventive Veterinary Medicine*, 81 (Special Issue), 3-20.

Peeler, E.J., Afonso, A., Berthe, F.C.J., Brun, E., Rodgers, C.J., Roque, A., Whittington, R.J. & Thrush, M.A. (2009). Epizootic haematopoietic necrosis virus: an assessment of the likelihood of introduction and establishment in England and Wales. *Preventive Veterinary Medicine*, 91, 241-253.

Peeler, E.J., Oidtmann, B.C., Midtlyng, P.J., Miossec, L. & Gozaln R.E. (2011). Non-native aquatic animals introductions have driven disease emergence in Europe. *Biological Invasions*, in press.

Penczak, T., Galicka, W., Molinski, M., Kusto, E. & Zalewski, M. (1982). The enrichment of a mesotrophic lake by carbon, phosphorus and nitrogen from the cage aquaculture of rainbow trout, *Salmo gairdneri*. *Journal of Applied Ecology*, 19, 371-393.

Penczak, T., O'Hara, K. & Kostrzewa, J. (2002). Fish bioenergetics model used for estimation of food consumption in a navigation canal with heavy traffic. *Hydrobiologia*, 479, 109-123.

Penman, D.J. & McAndrew, B.J. (1998). Practical aspects of selection and fitness of stocked fish. In *Inland Fisheries Enhancements* (ed. Petr, T.), pp. 223-234. *FAO Fisheries Technical Paper*, 374.

Perrow, M.R. & Irvine, K. (1992). The relationship between cladoceran body size and the growth of underyearling roach (*Rutilus rutilus* (L.)) in two shallow lowland lakes: a mechanism for density-dependent reductions in growth. *Hydrobiologia*, 241, 155-161.

Perrow, M.R., Jowitt, A.J.D., Stansfield, J.H. & Phillips, G.L. (1999). The practical importance of the interactions between fish, zooplankton and macrophytes in shallow lake restoration. *Hydrobiologia*, 396, 199-210.

Persson, L. (1987). Competition-induced switch in young of the year perch, *Perca fluviatilis*: an experimental test of resource limitation. *Environmental Biology of Fishes*, 19, 235-239.

Persson, L. & Greenberg, L.A. (1990). Juvenile competitive bottlenecks: The perch (*Perca fluviatilis*)-roach (*Rutilus rutilus*) interaction. *Ecology*, 71, 44-56.

Persson, L., Diehl, S., Johansson, L., Andersson, G. & Hamrin, S.F. (1991). Shifts in fish communities along the productivity gradient of temperate lakes – patterns and the importance of size-structured interactions. *Journal of Fish Biology*, 38, 281-293.

Petr T. (1998) Review of the administration and of benefits from fishery enhancements in Australia. In: Petr, T. (ed.), Inland Fisheries Enhancements. *FAO Fisheries Technical Paper* **374**, pp. 65–78.

Philippart, J.C. & Baras, E. (1988). The biology and management of the barbel, *Barbus barbus* (L.) in the Belgian River Meuse basin, with special reference to the reconstruction of populations using intensively-reared fish. In *Proceedings of the 19th (1988) Institute of Fisheries Management Annual Study Course*, pp. 61-82. IFM, Southampton.

Phillips, M.J., Beveridge, M.C.M. & Ross, L.G. (1985). The environmental impact of salmonid cage culture on inland fisheries: present status and future trends. *Journal of Fish Biology*, 27 (Suppl. A), 123-137.

Piggins, D.J. (1965). Salmon and sea trout hybrids. *Annual Report of Salmon Research Trust for Ireland.* SRTI, Dublin.

Piggins, D.J. (1966). Further study on the specific characteristics of brown trout and salmon-sea trout hybrids. *Annual Report of Salmon Research Trust for Ireland*. SRTI, Dublin.

Pijanowska, J. & Stolpe, G. (1996). Summer diapause in *Daphnia* as a reaction to the presence of fish. *Journal of Plankton Research*, 18, 1407-1412.

Pillay, T.V.R. (1992). *Aquaculture and the Environment*. Fishing News Books, Blackwell Scientific Publications, Oxford.

Piller, K.R., Wilson, C.C., Lee, C.E. & Lyons, J. (2005). Conservation genetics of inland lake trout in the upper Mississippi River basin: stocked or native ancestry? *Transactions of the American Fisheries Society*, 134, 789-802.

Pink, M., Fox, M.G. & Pratt, T.C. (2007). Numerical and behavioural response of cyprinids to the introduction of predatory brook trout in two oligotrophic lakes in northern Ontario. *Ecology of Freshwater Fish*, 16, 238-249.

Pinto, L., Chandrasena, N., Pera, J., Hawkins, P., Eccles, D. & Sim, R. (2005). Managing invasive carp (*Cyprinus carpio* L.) for habitat enhancement at Botany Wetlands, Australia. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 15, 447-462.

Pomeroy, P.P. (1994). Zooplankton in Loch Lomond: perspectives, predation and powan. *Hydrobiologia*, 290, 75-90.

Pope, K.L., Piovia-Scott, J. & Lawler, S.P. (2009). Changes in aquatic insect emergence in response to whole-lake experimental manipulations of introduced trout. *Freshwater Biology*, 54, 982-993.

Prammer, C.R., Aho, T., Püronen, J., Estoup, A., Cornuet, J.M. & Ranta, E. (1999). Microsatellite analysis of hatchery and natural populations of Arctic charr, *Salvelinus alpinus*, from the Nordic region: implications for conservation. Hereditas, in press.

Radke, R.J., Kahl, U. & Benndorf, J. (2003). Food-web manipulation of drinking water reservoirs with salmonids: vertical distribution of prey and predator. *Limnologica*, 33, 92-98.

Radtke H.A. & Davis S.W. (2000) Economic feasibility of salmon enhancement propagation programs. In: Knudson, E.E., Steward, C.R., MacDonald, D.D., Williams, J.E. and Reiser, D.W. (eds) *Sustainable Fisheries Management: Pacific Salmon*. CRC Press, Boca Raton, pp. 381–392.

Reisenbichler, R.R. (1988). Relation between distance transferred from natal stream and recovery rate for hatchery coho salmon. *North American Journal of Fisheries Management*, 8, 172-174.

Reisenbichler, R.R. & McIntyre, J.D. (1977). Genetic differences in growth and survival of juvenile hatchery and wild steelhead trout. *Journal of the Fisheries Research Board of Canada*, 34, 123-128.

Reisenbichler, R.R. & Phelps, S.R. (1989). Genetic variation in steelhead (*Salmo gairdneri*) form the north coast of Washington. *Canadian Journal of Fisheries and Aquatic Sciences*, 46, 66-73.

Reynolds, S.J. (2009). Impact of the introduced poeciliid *Gambusia holbrooki* on amphibians in southwestern Australia. *Copeia*, 2, 296-302.

Riccardi, N., Giussani, G. & Lagorio, L. (2002). Morphological variation and life history changes of a *Daphnia hyalina* population exposed to *Chaoborus flavicans* larvae predation (L. Candia, Northern Italy). *Journal of Limnology*, 61, 41-48.

Ricker, W.E. (1972). Hereditary and environmental factors affecting certain salmonid populations. In *The Stock Concept in Pacific Salmon* (eds R.C. Simon & P.A. Larkin), pp. 19-160. MacMillan Lectures in Fisheries, University of British Columbia, Vancouver.

Riggs, L. (1990). Principles for genetic conservation and production quality*. Report for the Northwest Power Planning Council (Contract No. C90-005).* Portland, Oregon.

Ritter, J.A. (1975). Lower ocean survival rates for hatchery-reared Atlantic salmon (*Salmo salar*) stocks released in rivers other than their native streams. *ICES Committee for Anadromous and Catadromous Fishes*, *C.M. 1975/M:26.* ICES, Copenhagen.

Romare, P. & Bergman, E. (1999). Juvenile fish expansion following biomanipulation and its effect on zooplankton. *Hydrobiologia*, 404, 89-97.

Romare, P., Bergman, E. & Hansson, L.A. (1999). The impact of larval and juvenile fish on zooplankton and algal dynamics. *Limnology and Oceanography*, 44, 1655-1666.

Romare, P., Berg, S., Lauridsen, T. & Jeppesen, E. (2003). Spatial and temporal distribution of fish and zooplankton in a shallow lake. *Freshwater Biology*, 48, 1353-1362.

Rowland, S.J. (1994). Stocking of freshwater fishes and policy in New South Wales. In *Translocation Issues in Western Australia* (ed. Prokop, F.), pp. 50-62. Fisheries Department of Western Australia, Fisheries Management Paper 83. FDWA, Perth.

Russell, I.C., Dare, P.J., McKay, H.V. & Ives, S.J. (2003). The potential for using fish refuges to reduce damage to inland fisheries by cormorants, *Phalacrocorax carbo*. In *Interactions between Fish and Birds: Implications for Management* (ed. I.G. Cowx), pp. 259-277*.* Fishing News Books, Blackwell Science, Oxford.

Russell, I., Parrott, D., Ives, M., Goldsmith, D., Fox, S., Clifton-Dey, D., Prickett, A. & Drew, T. (2008). Reducing fish losses to cormorants using artificial fish refuges: an experimental study. *Fisheries Management and Ecology*, 15, 189-198.

Ruzzante, D.E., Hansen, M.M., Meldrup, D. & Ebert, K.M. (2004). Stocking impact and migration pattern in an anadromous brown trout (*Salmo trutta*) complex: where have all the stocked spawning sea trout gone? *Molecular Ecology*, 13, 1433-1445.

Ryman, N. (1981). Conservation of genetic resources: experiences for brown trout (*Salmo trutta*). In: N. Ryman (ed.) Fish Gene Pools. *Ecological Bulletins*, 34, 61-74.

Ryman, N. & Laikre, L. (1991). Effects of supportive breeding on the genetically effective population size. *Conservation Biology*, 5, 325-329.

Ryman, N. & Utter, F. (eds) (1987). *Populations Genetics and Fishery Management.* Washington University Press, Seattle.

Ryman, N., Utter, F. & Laikre, L. (1995). Protection of intraspecific biodiversity in exploited fishes. *Reviews in Fish Biology and Fisheries*, 5, 417-446.

SAC (Salmon Advisory Committee) (1991). Assessment of stocking as a salmon management strategy. *Report to Ministry of Agriculture*, *Fisheries & Food*. MAFF, London.

Sandström, A. & Karås, P. (2002). Effects of eutrophication on young-of-the-year freshwater fish communities in coastal areas of the Baltic. *Environmental Biology of Fishes*, 63, 89-101.

Scheffer, M. (1999). The effect of aquatic vegetation on turbidity; how important are the filter feeders? *Hydrobiologia*, 408/409, 307-316.

Scheffer, M., Hosper, S.H., Meijer, M.L., Moss, B. & Jeppesen, E. (1993). Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution*, 8, 275-279.

Schindler, D.E., Knapp, R.A. & Leavitt, P.R. (2001). Alteration of nutrient cycles and algal production resulting from fish introductions into mountain lakes. *Ecosystems*, 4, 308-321.

Schmieder, K., Werner, S. & Bauer, H.G. (2006). Submersed macrophytes as a food source for wintering waterbirds at Lake Constance. *Aquatic Botany*, 84, 245-250.

Schoener, T.W. (1986). Resource partitioning. In *Community Ecology: Pattern and Process* (eds J. Kikkawa & D.J. Anderson). Fishing News Books, Blackwell Scientific Publications, Melbourne.

Schriver, P., Bøgestrand, J., Jeppesen, E. & Søndergaard, M. (1995). Impact of submerged macrophytes on fish-zooplankton-phytoplankton interactions: large-scale enclosure experiments in a shallow eutrophic lake. *Freshwater Biology*, 33, 255-270.

Schuck, H.A. (1948). Survival of hatchery trout in streams and possible methods of improving the quality of hatchery trout. *Progressive Fish Culturist*, 10, 3-14.

SEPA (Scottish Environment Protection Agency) (2000). Ponds, pools and lochans: guidance on good practice in the management and creation of small waterbodies in Scotland. Scottish Environment Protection Agency, Stirling.

SEPA (Scottish Environment Protection Sgency) (2002). Total phosphorus water quality standards for Scottish freshwater lochs. *Scottish Environment Protection Agency Policy No. 16*. SEPA, Stirling.

Shearer, W.M. (1992). *The Atlantic Salmon: Natural History*, *Exploitation and Future Management*. Fishing News Books, Oxford.

Shields, B.A., Stubbing, D.N., Summers, D.W. & Giles, N. (2005). Temporal and spatial segregation of spawning by wild and farm-reared brown trout, *Salmo trutta* L., in the River Avon, Wiltshire, UK. *Fisheries Management and Ecology*, 12, 77-79.

Shinn, A.P., Gibson, D.I. & Sommerville, C. (2001). Morphometric discrimination of *Gyrodactylus salaris* Malmberg (Monogenea) from species of *Gyrodactylus* parasitising British salmonids using novel parameters. *Journal of Fish Diseases*, 24, 83-97.

Simon, K.S. & Townsend, C.R. (2003). Impacts of freshwater invaders at different levels of ecological organisation, with emphasis on salmonids and ecosystem consequences. *Freshwater Biology*, 48, 982-994.

Skelton, P.H. (1993). *A Complete Guide to the Freshwater Fishes of Southern Africa*. Southern Book Publishers, Half Way House, South Africa.

Skibinski, D.O.F. (1998). Genetical aspects of fisheries enhancement. In *Inland Fisheries Enhancements* (ed. T. Petr), pp. 205-222. *FAO Fisheries Technical Paper*, 374.

Skinner, A., Young, M. & Hastie, L. (2003). Ecology of the freshwater pearl mussel. *Conserving Natura 2000 Rivers Ecology Series No. 2.* English Nature, Peterborough.

Skov, C. & Nilsson, P.A. (2007). Evaluating stocking of YOY pike *Esox lucius* as a tool in the restoration of shallow lakes. *Freshwater Biology*, 52, 1834-1845.

Skov, C., Lousdal, O., Johansen, P.H. & Berg, S. (2003). Piscivory of 0+ pike (*Esox lucius* L.) in a small eutrophic lake and its implication for biomanipulation. *Hydrobiologia*, 506, 481-487.

Ślusarczyk, M. (1997). Impact of fish predation on a small-bodied cladoceran: limitation or stimulation? *Hydrobiologia*, 342/343, 215-221.

Small, M.P., Pichahchy, A.E., Von Bargen, J.F. & Young, S.F. (2004). Have native coho salmon (*Oncorhynchus kisutch*) persisted in the Nooksack and Samish rivers despite continuous hatchery production throughout the past century? *Conservation Genetics*, 5, 367-379.

Smith, P.A., Leah, R.T. & Eaton, J.W. (1996). Removal of pikeperch (*Stizostedion lucioperca*) from a British Canal as a management technique to reduce impact on prey fish populations. *Annales Zoologici Fennici*, 33, 537-546.

Smith, P.A., Leah, R.T. & Eaton, J.W. (1998). A review of the current knowledge on the introduction, ecology and management of zander, *Stizostedion lucioperca*, in the UK. In *Stocking and Introduction of Fish* (ed. Cowx, I.G.), pp. 209-224. Fishing News Books, Blackwell Science, Oxford.

Snieszko, S.F. (1973). Furunculosis of Salmonidae. *EIFAC Technical Paper*, 17 (Suppl. 2), 157-162.

Spears, B.M., Carvalho, L., Perkins, R., Kirika, A. & Paterson, D.M. (2012). Long-term variation and regulation of internal phosphorus loading in Loch Leven. *Hydrobiologia*, in press.

Sproul, J.T. & Tominaga, O. (1992). An economic review of the Japanese flounder stock enhancement project in Ishikari Bay, Hokkaido. *Bulletin of Marine Science*, 50, 75-88.

Ståhl, G. & Hindar, K. (1988). Genetisk struktur hos norsk laks: status og perspektiver. *Rapport fra fiskeforskningen, Direktoratet for Naturforvaltning No. 1*. DFN, Trondheim (in Norwegian).

Stansfield, J.H., Perrow, M.R., Tench, L.D., Jowitt, A.J.D. & Taylor, A.A.L. (1997). Submerged macrophytes as refuges for grazing Cladocera against fish predation: observations on seasonal changes in relation to macrophyte cover and predation pressure. *Hydrobiologia*, 342/343, 229-240.

Stenroth, P., Holmqvist, N., Nyström, P., Berglund, O., Larsson, P. & Granéli, W. (2006). Stable isotopes as an indicator of diet in omnivorous crayfish (*Pacifastacus leniusculus*): the influence of tissue, sample treatment, and season. *Canadian Journal of Fisheries and Aquatic Sciences*, 63, 821-831.

Stentiford, G.D., Oidtmann, B., Scott, A. & Peeler, E.J. (2010). Crustacean diseases in European legislation: implications for importing and exporting nations. *Aquaculture*, 306, 27-34.

Stewart, D.C., Middlemas, S.J., Gardiner, W.R., Mackay, S. & Armstrong, J.D. (2005). Diet and prey selection of cormorants (*Phalacrocorax carbo*) at Loch Leven, a major stocked trout fishery. *Journal of Zoology*, 267, 191-201.

Stibor, H. & Luning, J. (1994). Predator-induced phenotypic variation in the pattern of growth and reproduction in *Daphnia hyalina* (Crustacea, Cladocera). *Functional Ecology*, 8, 97-101.

Stirling, H.P. & Dey, T. (1990). Impact of intensive cage fish farming on the phytoplankton and periphyton of a Scottish freshwater loch. *Hydrobiologia*, 190, 193-214.

Stott, B. (1977). On the question of the introduction of grass carp (*Ctenopharyngodon idella* Val.) into the United Kingdom. *Fisheries Management*, 8, 63-71.

Su, M.S. & Liao, I.C. (1999). Research and development of prawn stock enhancement in Taiwan. In *Stock Enhancement and Sea Ranching* (eds B.R. Howell, E. Moksness & T. Svåsand), pp. 379-392. Fishing News Books, Oxford.

Taeubert, J.E., Denic, M., Gum, B., Lange, M. & Geist, J. (2010). Suitability of different salmonid strains as hosts for the endangered freshwater pearl mussel (*Margaritifera margaritifera* L.). *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 728-734.

Taggart, J.B. & Ferguson, A. (1986). Electrophoretic evaluation of a supplemental stocking programme for brown trout (*Salmo trutta* L.). *Aquaculture and Fisheries Management*, 17, 155-162.

Tammi, J., Appelberg, M, Beier, U., Hesthagen, T., Lappalainen, A. & Rask, M. (2003). Fish status survey of Nordic lakes: effects of acidification, eutrophication and stocking activity on present fish species composition. *Ambio*, 32, 98-105.

Taniguchi, N. (2003). Genetic factors in broodstock management for seed production. *Reviews in Fish Biology and Fisheries*, 13, 177-185.

Taylor, A. A. L., Britton, J. R. & Cowx, I. G. (2004). Does the stock density of stillwater catch and release fisheries affect the growth performance of introduced cultured barbel? *Journal of Fish Biology*, 65 (Suppl. A), 308-313.

Taylor, A.H. (1978). An analysis of the trout fishing at Eye Brook – a eutrophic reservoir. *Journal of Animal Ecology*, 47, 407-423.

Taylor, E.B. (1991). A review of local adaptation in Salmonidae, with particular reference to Pacific and Atlantic salmon. *Aquaculture*, 98, 185-207.

Taylor, N.G.H., Way, K., Jeffery, K.R. & Peeler, E.J. (2010a). The role of live fish movements in spreading koi herpesvirus throughout England and Wales. *Journal of Fish Diseases*, 33, 1005-1007.

Taylor, N.G.H., Dixon, P.F., Jeffery, K.R., Peeler, E.J., Denham, K.L. & Way, K. (2010b). Koi herpesvirus: distribution and prospects for control in England and Wales. *Journal of Fish Diseases*, 33, 221-230.

Taylor, N.G.H., Norman, R.A., Way, K. & Peeler, E.J. (2011). Modelling the koi herpesvirus (KHV) epidemic highlights the importance of active surveillance within a national control policy. *Journal of Applied Ecology*, 48, 348-355.

Templeton, A.R., Hemmer, H., Mace, G., Seal, U.S., Shields, W.M. & Woodruff, D.S. (1986). Local adaptation, coadaptation, and population boundaries. *Zoo Biology*, 5, 115-125.

Templeton, R.G. (ed.) (1984). *Freshwater Fisheries Management.* Fishing News Books, Surrey, England.

Thorpe, J.E. (1974a). Estimation of the number of brown trout *Salmo trutta* (L.) in Loch Leven, Kinross, Scotland. *Journal of Fish Biology*, 6, 135-152.

Thorpe, J.E. (1974b). The movements of brown trout, *Salmo trutta* (L.) in Loch Leven, Kinross, Scotland. *Journal of Fish Biology*, 6, 153-180.

Thrush, M. & Peeler, E. (2006). Stochastic simulation of live salmonid movement in England and Wales to predict potential spread of exotic pathogens. *Diseases of Aquatic Organisms*, 72, 115-123.

Tomec, M., Teskeredžić, Z. & Teskeredžić, E. (2003). Food and nutritive value of gut contents of rudd (*Scardinius erythrophthalmus* L.) from Vrana Lake, Cres Island, Croatia. *Czech Journal of Animal Science*, 48, 28-34.

Tonkin, Z.D., Humphries, P. & Pridmore, P.A. (2006). Ontogeny of feeding in two native and one alien fish species from the Murray-Darling Basin, Australia. *Environmental Biology of Fishes*, 76, 303-315.

Tricarico, E., Vilizzi, L., Gherardi, F. & Copp, G.H. (2010). Calibration of FI-ISK, an invasiveness screening tool for non-native freshwater invertebrates. *Risk Analysis*, 30, 285-292.

Utter, F. (1998). Genetic problems of hatchery-reared progeny released into the wild, and how to cope with them. *Bulletin of Marine Sciences*, 62, 623-640.

Utter, F. (2001). Patterns of subspecific anthropogenic introgression in two salmonid genera. *Reviews in Fish Biology and Fisheries*, 10, 265-279.

Utter, F., Milner, G., Ståhl, G. & Teel, D. (1989). Genetic population structure of chinook salmon in the Pacific Northwest. *Fishery Bulletin*, 85, 13-23.

van Zyll de Jong, M.C., Gibson, R.J. & Cowx, I.G. (2004). Impacts of stocking and introductions on freshwater fisheries of Newfoundland and Labrador, Canada. *Fisheries Management and Ecology*, 11, 183-193.

Vašek, M. & Kubečka, J. (2004). *In situ* diel patterns of zooplankton consumption by subadult/adult roach *Rutilus rutilus*, bream *Abramis brama*, and bleak *Alburnus alburnus*. *Folia Zoologica*, 53, 203-214.

Vašek, M., Kubečka, J. & Sed'a, J. (2003). Cyprinid predation on zooplankton along the longitudinal profile of a canyon-shaped reservoir. *Archiv für Hydrobiologie*, 156, 535-550.

Verspoor, E. (1988). Widespread hybridization between native Atlantic salmon, *Salmo salar*, and introduced brown trout, *S. trutta*, in eastern Newfoundland. *Journal of Fish Biology*, 32, 327-334.

Verspoor, E. & Hammar, J. (1991). Introgressive hybridization in fishes: the biochemical evidence. *Journal of Fish Biology*, 39 (Suppl. A), 309-334.

Verspoor, E., Beardmore, J.A., Consuegra, S., Garciá de Leániz, C., Hindar, K., Jordan, W.C., Koljonen, M.-L., Mahkrov, A.A., Paaver, T., Sánchez, J.A, Skaala, Ø., Titov, S. & Cross, T.F. (2005). Population structure in the Atlantic salmon: insights from 40 years of research into genetic protein variation. *Journal of Fish Biology*, 67 (Suppl. A), 3-54.

Vilizzi, L. (1998). Observations on ontogenetic shifts in the diet of 0+ carp, *Cyprinus carpio* L., from the River Murray, Australia. *Folia Zoologica*, 47, 225-229.

Vincent, R.E. (1960). Some influences of domestication upon three stocks of brook trout (*Salvelinus fontinalis* Mitchell). *Transactions of the American Fisheries Society*, 89, 35-52.

Wager, R. & Jackson, P. (1993). *The Action Plan for Australian Freshwater Fishes*. Australian Nature Conservation Agency, Canberra.

Walker, A.F. (2007). Stream spawning of Arctic charr in Scotland. *Ecology of Freshwater Fish*, 16, 47-53.

Waples, R.S. (1991). Genetic interactions between hatchery and wild salmonids: lessons for the Pacific NorthWest. *Canadian Journal of Fisheries and Aquatic Sciences*, 48 (Suppl. 1), 124-133.

Waples, R.S. & Drake, J. (2004). Risk/benefit considerations for marine stock enhancement: a Pacific salmon perspective. In *Stock Enhancement and Sea Ranching. Developments*

Pitfalls and Opportunities (eds K.M. Leber, Blakenship H.L. & Svasand T.), pp. 260-306. Blackwell Publishing, Oxford.

Warburton, K., Retif, S. & Hume, D. (1998). Generalists as sequential specialists: diets and prey-switching in juvenile silver perch. *Environmental Biology of Fishes*, 51, 445-454.

Ward, A.J.W., Webster, M.M. & Hart, P.J.B. (2006). Intraspecific food competition in fishes. *Fish and Fisheries*, 7, 231-261.

Webb, J.H. & Youngson, A.F. (1992). Reared Atlantic salmon, *Salmo salar* L., in the catches of a salmon fishery on the western coast of Scotland. *Aquaculture and Fisheries Management*, 23, 217-221.

Webb, J.H., Hay, D.W. Cunningham, P.D. & Youngson, A.F. (1991). The spawning behaviour of escaped farmed and wild adult Atlantic salmon (*Salmo salar* L.) in a northern Scottish river. *Aquaculture*, 98, 97-110.

Webb, J.H., Youngson, A.F., Thompson, C.E., Hay, D.W., Donaldson, M.J. & McLaren, I.S. (1993). Spawning of escaped farmed salmon (*Salmo salar* L.) in western and northern Scottish rivers; egg deposition by females. *Aquaculture and Fisheries Management*, 24, 663-670.Weber, E.D. & Fausch, K. (2003). Interactions between hatchery and wild salmonids in streams: differences in biology and evidence for competition. *Canadian Journal of Fisheries and Aquatic Science*, 60, 1018-1036.

Węgleńska, T., Ejsmont-Karabin, J. & Rybak, J.I. (1997). Biotic interactions of the zooplankton community of a shallow, humic lake. *Hydrobiologia*, 342/343, 185-195.

Welcomme, R.L. (1988). International introductions of inland aquatic species. *FAO Fisheries Technical Paper*, 294.

Welcomme, R.L. & Bartley, D.M. (1998a). An evaluation of present techniques for the enhancement of fisheries. *FAO Fisheries Technical Paper*, 374.

Welcomme, R.L. & Bartley, D.M. (1998b). Current approaches to the enhancement of fisheries. *Fisheries Management and Ecology*, 5, 351-382.

Wells, R.D.S., Bannon, H.J. & Hicks, B.J. (2003). Control of macrophytes by grass carp (*Ctenopharyngodon idella*) in a Waikato drain, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 37, 85-93.

Welton, J.S., Ibbotson, A.T., Ladle, M. & Brookes, A.M. (1997). Impact of stocked rainbow trout on resident salmonid populations, phase 1: scoping study. *Environment Agency R&D Technical Report W61.* EA, Bristol.

Werner, E.E. & Gilliam, J.F. (1984). The ontogenetic niche and species interactions in size-structured populations. *Annual Review of Ecology and Systematics*, 15, 393-425.

Wheeler, A.C., Merrett, N.R. & Quigley, D.T.G. (2004). Additional records and notes for Wheeler's (1992) *List of the Common and Scientific Names of Fishes of the British Isles*. *Journal of Fish Biology*, 65 (Suppl. B), 1-40.

White, R.J., Karr, J.R. & Nehlsen, W. (1995). Better roles for fish stocking in aquatic resource management. *American Fisheries Society Symposium*, 15, 527-547.

Whiteside, M.C., Doolittle, W.L. & Swindoll, C.M. (1985). Zooplankton as food resources for larval fish. *Verhandlungen der Internationalen Vereinigung für theoretische und angewandte Limnologie*, 22, 2523-2526.

Wightman, J.C., Ward B.R., Ptolemy R.A. & Axford F.N. (1998). A recovery plan for east coast Vancouver Island steelhead trout. *Report to Ministry of Environment, Lands & Parks*. MELP, Nanaimo, British Columbia.

Wiley, R.W. (1995). A common sense protocol for the use of hatchery-reared trout. *American Fisheries Society Symposium*, 15, 465-471.

Wilhelm, P.M., Parker, B.R., Schindler, D.W. & Donald, D.B. (1999). Seasonal food habits of bull trout from a small alpine lake in the Canadian Rocky Mountains. *Transactions of the American Fisheries Society*, 128, 1176-1192.

Willby, N.J. & Eaton, J.W. (1993). The distribution, ecology and conservation of *Luronium natans* (L.) Raf. in Britain. *Journal of Aquatic Plant Management*, 31, 70-76.

Willette, T.M., Cooney, R.T., Patrick, V., Mason, D.M., Thomas, G.L. & Scheel, D. (2001). Ecological processes influencing mortality of juvenile pink salmon (*Oncorhynchus gorbuscha*) in Prince William Sound, Alaska. *Fisheries Oceanography*, 10 (Suppl. 1), 14-41.

Williams, A.E., Moss, B. & Eaton, J. (2002). Fish induced macrophyte loss in shallow lakes: top-down and bottom-up processes in mesocosm experiments. *Freshwater Biology*, 47, 2216-2232.

Wilson, R.S. (1971). The decline of a roach *Rutilus rutilus* (L.) population in Chew Valley Lake. *Journal of Fish Biology*, 3, 129-137.

Wilson, R.S., Maxwell, T.R.A., Mance, G., Sleigh, M.A. & Milne, R.A. (1975). Biological aspects of Chew Valley and Blagdon Lakes, England. *Freshwater Biology*, 5, 379-393.

Winfield, I.J. (1992). Threats to the lake fish communities of the UK arising from eutrophication and species introductions. *Netherlands Journal of Zoology*, 42, 233-242.

Winfield, I.J. & Durie, N.C. (2004). Fish introductions and their management in the English Lake District. *Fisheries Management and Ecology*, 11, 195-201.

Winfield, I.J., Adams, C.E. & Fletcher, J.M. (1996). Recent introductions of the ruffe (*Gymnocephalus cernuus*) to three United Kingdom lakes containing *Coregonus* species. *Annales Zoologici Fennici*, 33, 459-466.

Winfield, I.J., Dodge, D.P. & Rösch, R. (1998). Introductions of the ruffe, *Gymnocephalus cernuus*, to new areas of Europe and to North America: history, the present situation and management implications. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 191-200. Fishing News Books, Blackwell Science, Oxford.

Winfield, I.J., Bean, C.W. & Hewitt, D.P. (2002a). The relationship between spatial distribution and diet of Arctic charr, *Salvelinus alpinus*, in Loch Ness, UK. *Environmental Biology of Fishes*, 64, 63-73.

Winfield, I.J., Fletcher, J.M. & James, J.B. (2002b). Species introductions to two English lake fish communities of high conservation value: a 10-year study. In *Conservation of Freshwater Fishes: Options for the Future* (eds M.J. Collares-Pereira, I.G. Cowx & M.M. Coelho), pp. 271-281. Fishing News Books, Blackwell Science, Oxford.

Winfield, I.J., Fletcher, J.M. & Winfield, D.K. (2002c). Conservation of the endangered whitefish, *Coregonus lavaretus*, population of Haweswater, UK. In *Management and Ecology of Lake and Reservoir Fisheries* (ed. I.G. Cowx), pp. 232-241. Fishing News Books, Blackwell Science, Oxford.

Winfield, I.J., Crawshaw, D.H. & Durie, N.C. (2003). Management of the cormorant, *Phalacrocorax carbo*, and endangered whitefish, *Coregonus lavaretus*, populations of Haweswater, UK. In *Interactions between Fish and Birds: Implications for Management* (ed. I.G. Cowx), pp. 335-344. Fishing News Books, Blackwell Science, Oxford.

Winfield, I.J., Fletcher, J.M., James, J.B., Duigan, C.A., Bean, C.W. & Durie, N.C. (2007). Long-term case histories of ruffe (*Gymnocephalus cernuus*) introductions to four U.K. lakes containing native vendace (*Coregonus albula*) or whitefish (*C. lavaretus*) populations. *Advances in Limnology*, 60, 301-309.

Winfield, I.J., Fletcher, J.M. & James, J.B. (2010). An overview of fish species introductions to the English Lake District, UK, an area of outstanding conservation and fisheries importance. *Journal Applied Ichthyology*, 26 (Suppl. 2), 60-65.

Winfield, I.J., Fletcher, J.M. & James, J.B. (2011). Invasive fish species in the largest lakes of Scotland, Northern Ireland, Wales and England: the collective UK experience. *Hydrobiologia*, 660, 93-103.

Winfield, I.J., Adams, C.E., Armstrong, J.D., Gardiner, R., Kirika, A., Montgomery, J., Spears, B.M., Stewart, D.C. & Thorpe, J.E. (2012). Changes in the fish community of Loch Leven: untangling anthropogenic pressures. *Hydrobiologia*, in press.

Wingfield, R., Murphy, K. & Gaywood, M. (2005). Lake habitat suitability for the rare European macrophyte *Najas flexilis* (Willd.) Rostk. & Schmidt. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 15, 227-241.

Winkler, K.A., Pamminger-Lahnsteiner, B., Wanzenböck, J. & Weiss, S. (2011). Hybridization and restricted gene flow between native and introduced stocks of Alpine whitefish (*Coregonus* sp.) across multiple environments. *Molecular Ecology*, 20, 456-472.

Wissel, B., Freier, K., Muller, B., Koop, J. & Benndorf, J. (2000). Moderate planktivorous fish biomass stabilizes biomanipulation by suppressing large invertebrate predators of *Daphnia*. *Archiv für Hydrobiologie,* 149, 177-192.

Wissinger, S.A., McIntosh, A.R. & Greig, H.A. (2006). Impacts of introduced brown and rainbow trout on benthic invertebrate communities in shallow New Zealand lakes. *Freshwater Biology*, 51, 2009-2028.

Wojtal, A., Frankiewicz, P., Wagner-Łotkowska, I. & Zalewski, M. (2004). The evaluation of the role of pelagic invertebrate versus vertebrate predators on the seasonal dynamics of filtering Cladocera in a shallow, eutrophic reservoir. *Hydrobiologia*, 515, 123-135.

Wollebaek, J., Heggenes, J. & Røed, K.H. (2010). Disentangling stocking introgression and natural migration in brown trout: survival success and recruitment failure in populations with semi-supportive breeding. *Freshwater Biology*, 55, 2626-2638.

Wootten, R. (1998). Health management in stocked fisheries. In *Inland Fisheries Enhancements* (ed. T. Petr), pp. 235-244. *FAO Fisheries Technical Paper*, 374.

Wootton, R.J. (1990). *Ecology of Teleost Fishes*. Chapman & Hall, London.

Wright, G.A. (2003). Impact of cormorants on the Loch Leven trout fishery and the effectiveness of shooting as mitigation. In *Interactions between Fish and Birds: Implications for Management* (ed. I.G. Cowx), pp. 288-297. Fishing News Books, Blackwell Science, Oxford.

Wysujack, K. & Mehner, T. (2005). Can feeding of European catfish prevent cyprinids from reaching a size refuge? *Ecology of Freshwater Fish*, 14, 87-95.

Yang, Y.F., May, L., Gunn, I., Huang, X.F. & Liu, J.K. (1999). Comparative studies on effects of predation by fish on *Daphnia* in Lake Donghu (China) and Loch Leven (Scotland). *Ecosystem Approaches for Fisheries Management*, 16, 265-281.

Yeomans, W.E., Chubb, J.C. & Sweeting, R.A. (1997). *Khawia sinensis* (Cestoda: Caryophyllidea): an indicator of legislative failure to protect freshwater habitats in the British Isles? *Journal of Fish Biology*, 51, 880-885.

Yokoyama, H., Abo, K. & Ishihi, Y. (2006). Quantifying aquaculture-derived organic matter in the sediment in and around a coastal fish farm using stable carbon and nitrogen isotope ratios. *Aquaculture*, 254, 411-425.

Young K.A. (1999) Managing the decline of Pacific salmon: metapopulation theory and artificial recolonisation as ecological mitigation. *Canadian Journal of Fisheries and Aquatic Sciences* **56**, 1700–1706.

Youngson, A.F., Martin, S.A.M., Jordan, W.C. & Verspoor, E. (1991). Genetic protein variation in Atlantic salmon in Scotland: comparison of wild and farmed fish. *Aquaculture*, 98, 231-242.

Youngson, A.F., Webb, J.H., Thompson, C.E. & Knox, D. (1993). Spawning of escaped farmed Atlantic salmon (*Salmo salar*): hybridization of females with brown trout (*Salmo trutta*). *Canadian Journal of Fisheries and Aquatic Sciences*, 50 1986-1990.Youngson, A.F., Jordon, W.C., Verspoor, E., McGinnity, P., Cross, T. & Ferguson, A. (2003). Management of salmonid fisheries in the British Isles: towards a practical approach based on population genetics. *Fisheries Research*, 62, 193-209.

Zalewski, M., Frankiewicz, P. & Brewinska, B. (1985). The factors limiting growth and survival of brown trout, *Salmo trutta* m. *fario* L. introduced to different types of stream. *Journal of Fish Biology*, 27 (Suppl. A), 59-73.

Zambrano, L., Scheffer, M. & Martinez-Ramos, M. (2001). Catastrophic response of lakes to benthivorous fish introduction. *Oikos*, 94, 344-350.

Zeiber, R.A., Sutton, T.M. & Fisher, B.E. (2008). Western mosquitofish predation on native amphibian eggs and larvae. *Journal of Freshwater Ecology*, 23, 663-671.

Ziemann, D.A. (2001) The potential for the restoration of marine ornamental fish populations through hatchery releases. *Aquarium Sciences and Conservation* **3**, 107–117.

Appendix 1. Generic protocol for assessment of stock enhancement programmes

A.1 Context

Cowx (1994b) proposed a decision support tool for managing whether stocking should be permitted or not. The strategy stipulates that stocking for specific objectives should only be carried out following a detailed assessment, and only under certain conditions. The underlying principle for the management of stocking stipulates that once the objectives for stocking have been set, through a thorough assessment of the status and limitations of the fishery, a specific stocking strategy is developed to achieve the desired objective. This is equivalent to identifying the bottlenecks constraining the potential performance of the fishery. Once the fishery has been confirmed as potentially requiring enhancement, scenario overviews must assess the critical bottlenecks to the fish population or fishery performance and, through this, determine whether stocking of the species is a viable option for enhancement. It should then be evaluated against ecological and environmental risk criteria, and a cost-benefit analysis should be carried out. Finally, the overall feasibility of the action assessed in terms of environmental and ecological risk, bio-economic gain and practicality should be evaluated. If at any stage of these assessments the risks, costs, feasibility or potential benefits are deemed unacceptable, the programme should be rejected and alternative strategies considered.

Whilst this framework offers generic guidelines for appraisal of stocking activities and support for implementation, it is limited in its assessment of risks and uncertainty about stocking on fisheries and the environment, and does not necessarily provide support for the decision-making process at each stage of evaluation. At each stage of the process, decisions have to be made about the acceptability of the risks and uncertainty of certain impacts of stocking or introductions, and the ability to manage those risks. This appendix develops the decision tree of Cowx (1994b) to accommodate risk and uncertainty as well as enhancing protocols associated with other aspects that require decision making, and aims to provide a framework for guidelines for stocking fish in conservation areas. It is based on the project management approach and describes the sequence of processes that should be undertaken when any stock enhancement programme is proposed through to its final implementation. Essentially, the activities can be divided into the following phases: identification, preparation, appraisal, implementation and evaluation. Each step has a series of look-up boxes associated with it that must be adhered to before progressing to the next step. Where decisions are required, advancement is only possible where a beneficial response is forthcoming. It should be recognised that the protocol outlined is based on the premise that no stocking has taken place in the past and that a full evaluation is required to ensure that any action is carried out in an ecologically acceptable, environmentally friendly, socially responsible and cost-effective manner (although it is recognised that social and economic issues are of low priority with applications to stock or introduce fishes to designated natural heritage sites). Where stocking has taken place in the recent past it will be possible to circumvent several of the steps to utilise knowledge about the outcomes of previous stocking events (Go to Lookup Box A5). If the stocking programme is accepted, implementation should only be allowed once a suitable post-stocking evaluation scheme is incorporated into the programme. The lack of suitable monitoring programmes of historical activities has led to uncertainty over the success or failure of schemes, and the inability to detect impacts or attribute improvements to the stocking *per se* (Cowx, 1994b, 1998a, b).

A.2 Generic framework for stocking enhancement

A.2.1 *Identification of management objectives (see Look-up Box A1)*

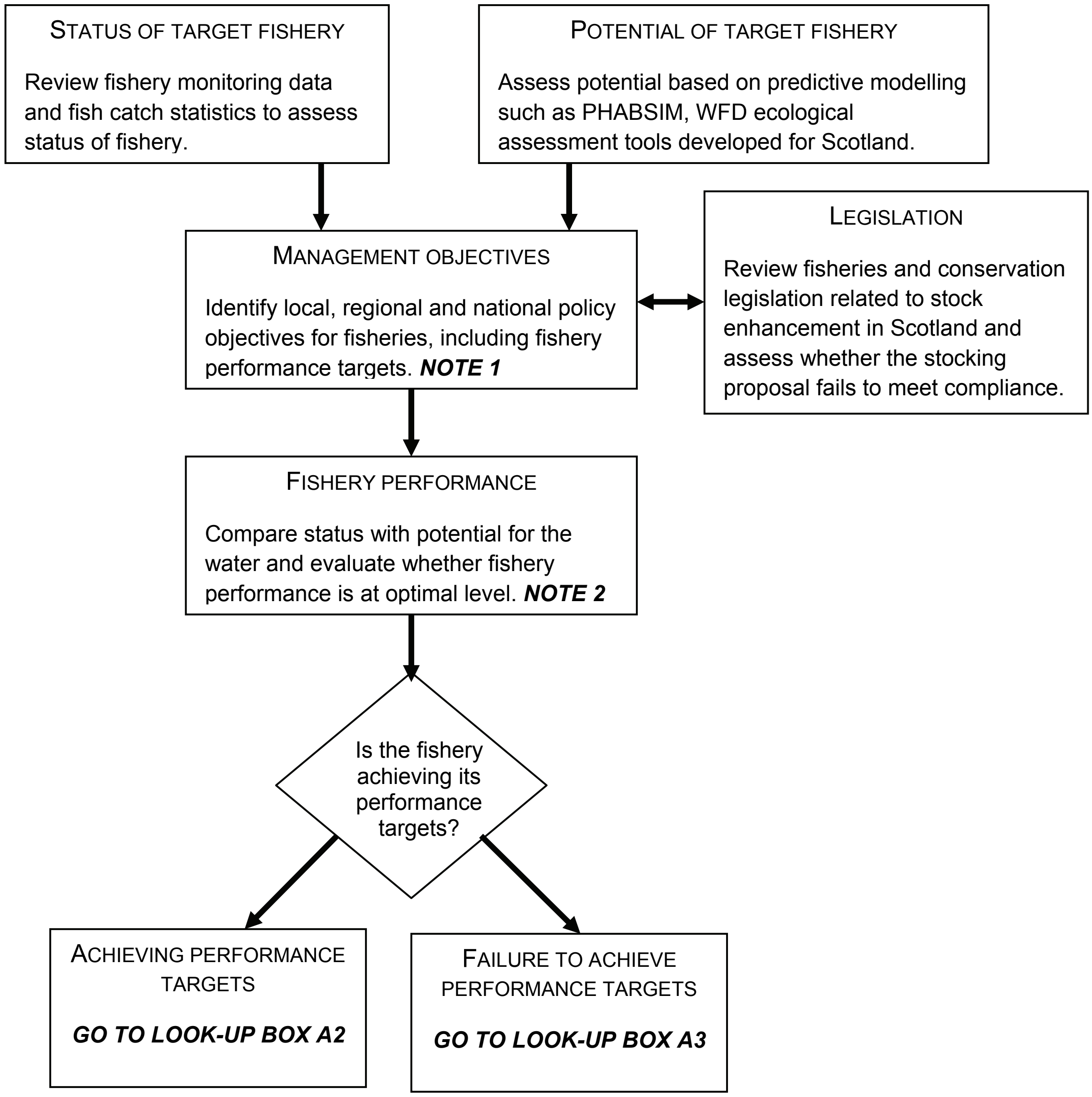
The first step when considering any stock improvement activity must be to ensure proper clarification of the management policy and objectives. It is only then that the project proposal can be properly formulated to achieve the desired effects. Part of this exercise includes establishing whether the stock is below optimum production level or whether the quality of the stock (e.g. in terms of age or size distribution) could be improved. This requires not only an assessment of the status of existing stocks, but an appraisal of the condition of the water body, and the natural and artificial factors that may limit production. Where the recruitment of wild fishes has been reduced by anthropogenic disturbance or the fishery is under performing, the requirement to protect the residual stocks from genetic impacts of non-native fishes remains, and it is unlikely that such species will be a favoured tool for mitigation. In these scenarios, mitigation through habitat rehabilitation and associated short-term assisted breeding programmes of extant, indigenous stocks is likely to be the preferred option. Indeed, the use of non-native fish species in this case may have negative effects on the mitigation activities. Therefore, the use of non-native fish species should be restricted to fishery enhancement scenarios with the objective of maintaining or enhancing the stocks of takeable-sized fish in a put-and-take or catch-and-release fishery. Assessments must be based on firm evidence from scientific studies (preferably of a long-term nature to overcome shorter-term fluctuations) and not on hearsay or unsubstantiated complaints. This process is outlined in Look-up Box A1.

A.2.2 *Alternative stock enhancement strategies (see Look-up Boxes A2 and A3)*

The first criterion is to establish if the fishery is of the desired quality to meet current or anticipated future demands, and thus whether there is a need for any stock improvement. Where enhancement of the stocks is considered necessary, a number of approaches are available, in addition to enhancement stocking, and these should be explored in the first instance or the “do nothing” (precautionary) approach adopted (Look-up Box A2). There are a number of options available to improve fisheries that do not (negatively) impact on the environment or fisheries. One is to alter the ecosystem, to improve both the fisheries and the conditions for exploitation (see Templeton, 1984; Cowx & Welcomme, 1998). The second option is to adopt traditional management measures that regulate catches and access to the fisheries, to manage exploitation pressure. The third option is to promote sustainable angling in the general area and at specific fisheries, possibly by developing and promoting ‘wild’ fisheries, which are maintained by natural recruitment and are not stocked. Many anglers are prepared to pay a premium for quality fishing based on wild stocks, which could offset lost revenues that may otherwise have been derived from intensively stocked fisheries. Where there are economic or practical constraints preventing alternative strategies, enhancement stocking may be desirable to boost performance.

If production is considered to be below the potential of the system (Look-up Box A1), it is important to try and identify the constraints and resolve them before stocking is carried out (Look-up Box A3). If no apparent cause can be identified, or if the cause cannot be removed or removal is not cost effective, enhancement stocking could be considered, but there is a risk that the stocking could fail if the water body is not capable of supporting a sustainable population. In such cases, alternative improvement strategies should be considered or the “do nothing” approach adopted, with resources concentrated on water bodies that possibly could be improved. This does not, however, exclude put-and-take fisheries that are stocked to provide catchable-sized fishes for rapid exploitation by anglers and that do not consider sustainability through natural recruitment.

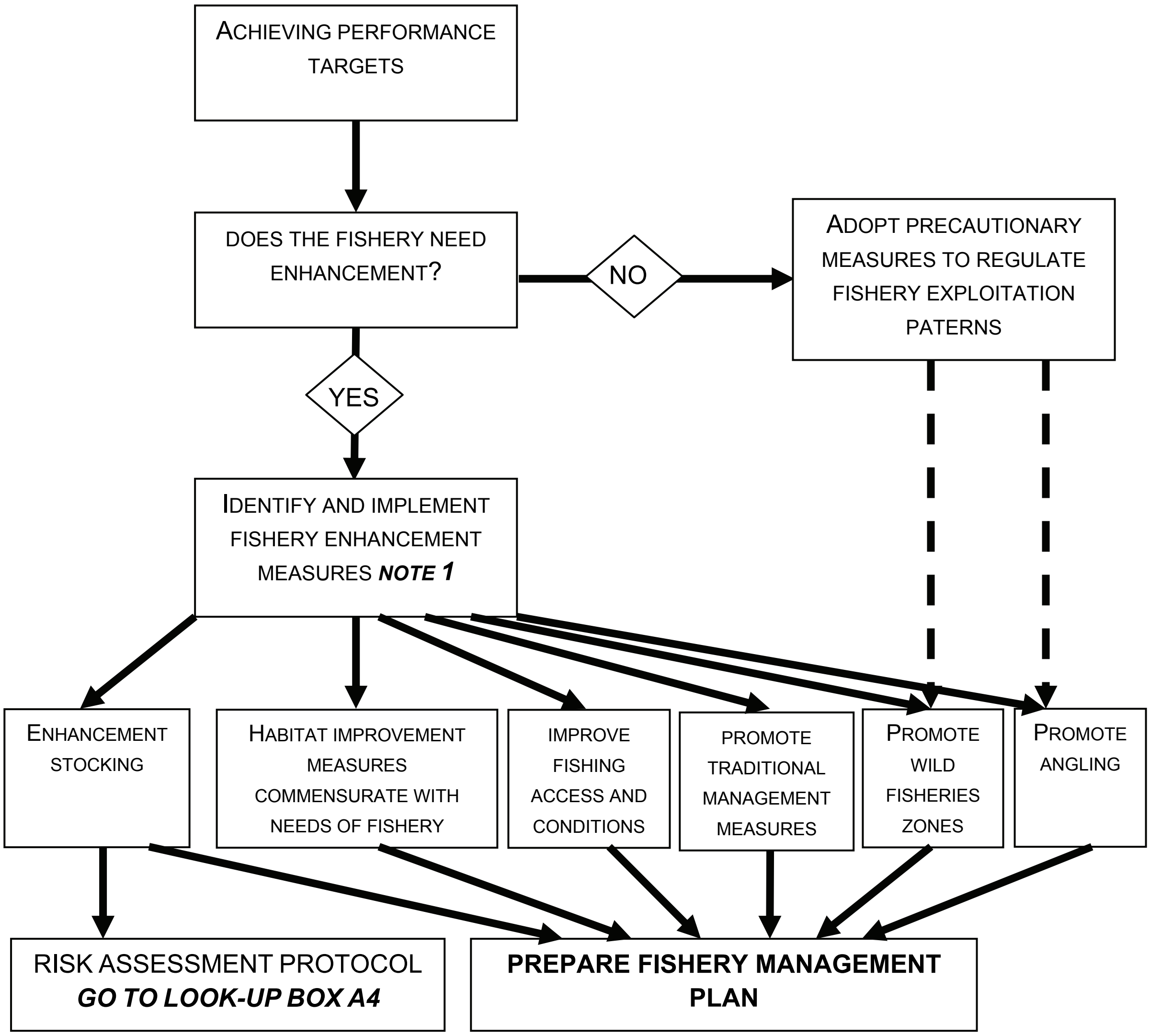
LOOK-UP BOX A1: OBJECTIVES AND PERFORMANCE



NOTE 1: Fishery performance targets refer to conservation objectives and/or ecologically sustainable development, as well as wider socio-economic benefits that are derived from recreational angling and net fisheries. This may be problematic if stakeholder (angler) defined targets are not known or unrealistic. Anglers may want a high CPUE or a 'high-quality' fishing experience, while fishery owners may want sustainable, profitable catches. Attention must also be paid to non-extractive stakeholders who tend to have different values for water bodies.

NOTE 2: Where quality information is not available, expert judgement (independent of the fishery owner) will have to be used, or the water body compared with nearby water bodies of similar character.

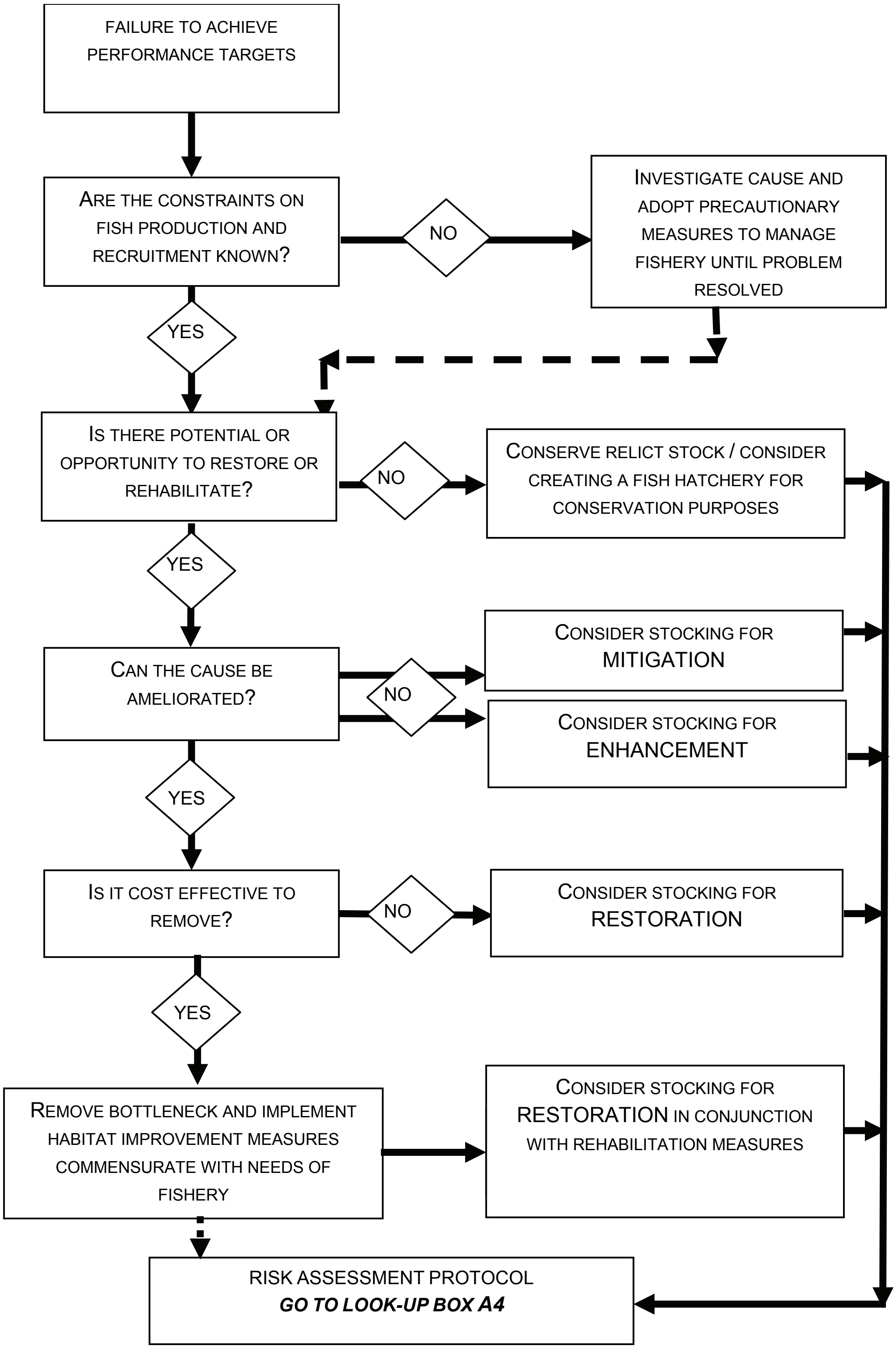
LOOK-UP BOX A2: ALTERNATIVE FISHERY ENHANCEMENT STRATEGIES



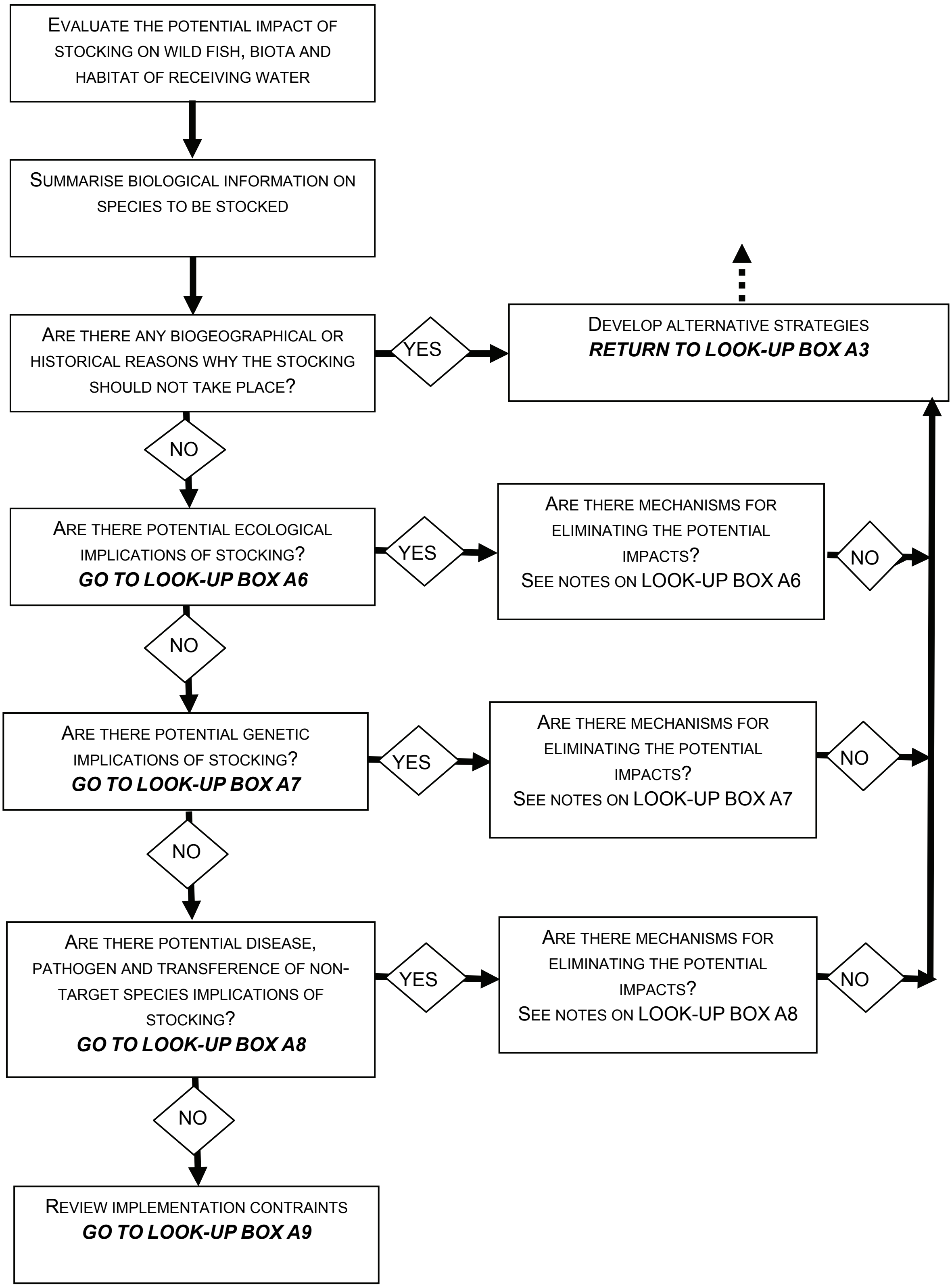
NOTE 1: The various fishery enhancement measures proposed are not mutually exclusive and in many cases should be considered as complimentary.

Where the limiting factor(s) can be isolated, efforts should be made to resolve the problems before resorting to stocking. If remedial action cannot be taken, because it is either impractical or not cost effective, then mitigation stocking could be considered. This is unlikely to lead to a sustainable population, however, and fish may have to be stocked on a regular basis and appropriate risk assessment should be undertaken (see Look-up Box A4).

LOOK-UP BOX A3: ALTERNATIVE FISHERY ENHANCEMENT STRATEGIES



LOOK-UP BOX A4: RISK ASSESSMENT



Finally, **if it is possible to remove or minimise the causes of declines in fisheries, this course of action should be taken, and the fisheries may then recover without stocking. Habitat improvement is the most desirable option because it should lead to long-term sustainable improvements with minimal deleterious ecological impacts.** It is also an efficient use of resources because it may have greater long-term benefits than enhancement stocking and also other conservation and ecological benefits (e.g. improved primary or secondary production). In cases where natural recovery may be ineffective because, for example, spawning stocks have been reduced to an apparently critically low level, restoration stocking may be appropriate to promote stock recruitment. Restoration stocking may also be desirable (either once rehabilitation is complete or simultaneously with it) to accelerate fishery recovery or to maintain local interest and momentum in fishery rehabilitation.

To aid the decision-making process, a technique commonly employed in development project formulation, the logical framework (Table A1), can be used. This approach is useful in setting out the design of stocking programmes in a clear and logical way so that any weaknesses that exist can be addressed at an early stage, or if these are insurmountable, the programmes can be aborted. The logical project framework approach emphasizes the value of choosing measurable indicators that can be assessed through the life of the project, and instructs the managers to assess carefully the risks and assumptions on which the project is based.

Indicators are used to determine the extent to which the objectives are being achieved and can be measured at different times, notabably in the **monitoring of project performance (stocking success), appraisal and evaluation phases.** Where possible, the indicators should define the target groups, quantities, quality, time and location. The section devoted to risks and assumptions of the logical framework is concerned with establishing realistic parameters of the environment in which the project is to function. Table A1 illustrates a project framework format that might be adopted at the onset of a project.

Starting with the aim of the project, a series of objectives, outputs and inputs are developed down the first column at the left-hand side of the page. The second column addresses the indicators that have been determined at the outset of the project and how they can be verified as the project is developed further through the various phases of the project approach. The final column assesses the risks and assumptions that underpin the elements described in the first two columns. As the project develops so the logical framework will be modified to account for new information likely to affect the project elements.

In the theoretical example (Table A1), embodied in the overall development aim is a familiar theme associated with recreational fisheries. A certain stretch of river is deemed to have deteriorated in productivity as a recreational fishery. The concerns of the local angling community have been transmitted to the river manager. If the aim and objectives of the logical framework are considered (Table A1), the recommended course of action is restocking. However, the chances of success are limited if the original development aim is to be pursued. Thus, to commit scarce resources to this project aim will probably result in only short-term and easily dissipated benefits. In essence, the anglers would welcome the restocking but this action would very likely not contribute to any lasting improvement in the 'quality' of the fishery. In the assumptions/risks (Table A1), attention is drawn to the perceived nature of the problem and the question of the value of restocking as a corrective measure. At this stage the project planners might reject this option, re-examine it from a different perspective or re-orientate the project to address the problem of the perceived poor quality of the recreational fishery.

Table A1. Logical framework example

Project Structure	Indicators/Means of verification	Assumptions/Risks
Overall Development Aim:		
Improve by stocking the angling quality of a 10 km stretch of River X.	More fish sightings. Better angling results. Wider variety of species caught.	That the problem is 'real' and not merely perceived. That stocking is a viable method of improving the angling productivity of the river.
Specific Objectives:		
Assess the current status of fish populations. Determine the biomass of fishes that might be stocked. Determine the species mix of fishes that should be stocked. Assess other factors that might be affecting the productivity of the fishery.	Biomass per m^2 of indigenous fishes assessed pre- and post-stocking. Determination of biomass of key 'sports fish' species. Assessment of predator/prey, etc., relationships between fish species. Assessment of water quality parameters, history of fish diseases, etc.	That the rivers can be effectively sampled. That the methods of population assessment are appropriate and results are reliable. That the predator/prey, water quality, etc., relationships can be elucidated to determine their effects on the fisheries.
Outputs:		
Improved knowledge of biology/distribution of fishes. Increased and / or maintained biomass of fishes available to anglers. Improved relationship with local angling clubs.	Pre/post-stocking assessments to determine density of indigenous/stocked fishes. Monitoring of (match) angling results. Monitoring of level of public (angler) complaints about fishing. Monitoring of changes in numbers/frequency of anglers fishing 10 km stretch of river. Monitoring of changes in the level of legal infringements by anglers.	That the stocked fishes survive in the river. That the stocked fish population does not dissipate. That other (angling) species do not suffer as a result of restocking. That there is a tangible relationship between anglers' complaints and the quality of the river fishery.
Inputs:		
3 man-month survey of 10-km stretch of river prior to stocking. 10-km stretch stocked with y tonnes of fish species a, b, c. 3 man-month survey of 10-km stretch of river post-stocking.		That the resources are available to undertake the survey work properly. That cost-effective and disease-free fishes are available for stocking. That pre/post-stocking surveys are compatible.

4.2.3 *Risks and uncertainty (see Look-up Boxes A4 and A5)*

Before any stock enhancement programme is implemented, a thorough assessment of the risks associated with the exercise must be undertaken (Look-up Box A4). Where stocking has taken place in the recent past it is possible to circumvent the main risk assessment and utilise knowledge about the outcomes of previous stocking events (Lookup Box A5). Risk assessment is used to determine the likelihood that an event may occur and what the consequences of such an event will be. A risk management framework operates by establishing the context (i.e. stocking event), identifying the risks on the existing situation (consequence and likelihood), assessing the risks and treating the risks. A measure of risk is typically derived by multiplying *likelihood* by *consequence*. A risk matrix, based on the International Council for the Exploration of the Sea (ICES) Code of Conduct for Species Introduction (after Campbell, 2006), is used to determine the level of risk (Table A2). The ratings refer to the probability (*likelihood*) of the impact (*consequence*) occurring if a species is stocked in a water based on attributes about the ecology of the species and the environment into which the species is being released. The *likelihood* of an event occurring according to the ratings in Table A2 is defined in Table A3. The *consequence* refers to the scale of the potential impacts based on knowledge of ecological interactions between the species to be stocked and those in the receiving water. The ratings are, where possible, based on scientific evidence, otherwise expert judgment is required. The latter introduces a level of uncertainty into the assessment procedure that must be accounted for. As a consequence, there is a need to introduce a further layer to the matrix that accounts for uncertainty in the knowledge base or processes in nature (Table A4). Where knowledge is deficient or uncertainty high, the precautionary principle should apply to prevent unforeseen impacts.

Table A2. Risk matrix. N = negligible; L = low, M = moderate; H = high; E = extreme.

	Consequence				
Likelihood	Insignificant	Minor	Moderate	Major	Significant
Rare	N	L	L	M	M
Unlikely	N	L	M	H	H
Possible	N	L	H	H	E
Likely	N	M	H	E	E
Almost certain	N	M	E	E	E

One further element associated with risk is the degree of isolation of the receiving water. For example, an internal, fully recirculatory aquaculture unit will carry minimal risk compared with a stocking directly into a river or lake with no form of containment. Consequently, as part of the assessment procedure a weighting factor can be applied to the scoring system to reflect the degree of isolation (Table A5).

Table A3. Likelihood rating.

Likelihood	**Description**	**%**
Rare	Event will only occur in exceptional circumstances	<5
Unlikely	Event could occur but not expected	25
Possible	Event could occur	50
Likely	Event will probably occur in most circumstances	75
Almost certain	Event is expected to occur in most circumstances	>95

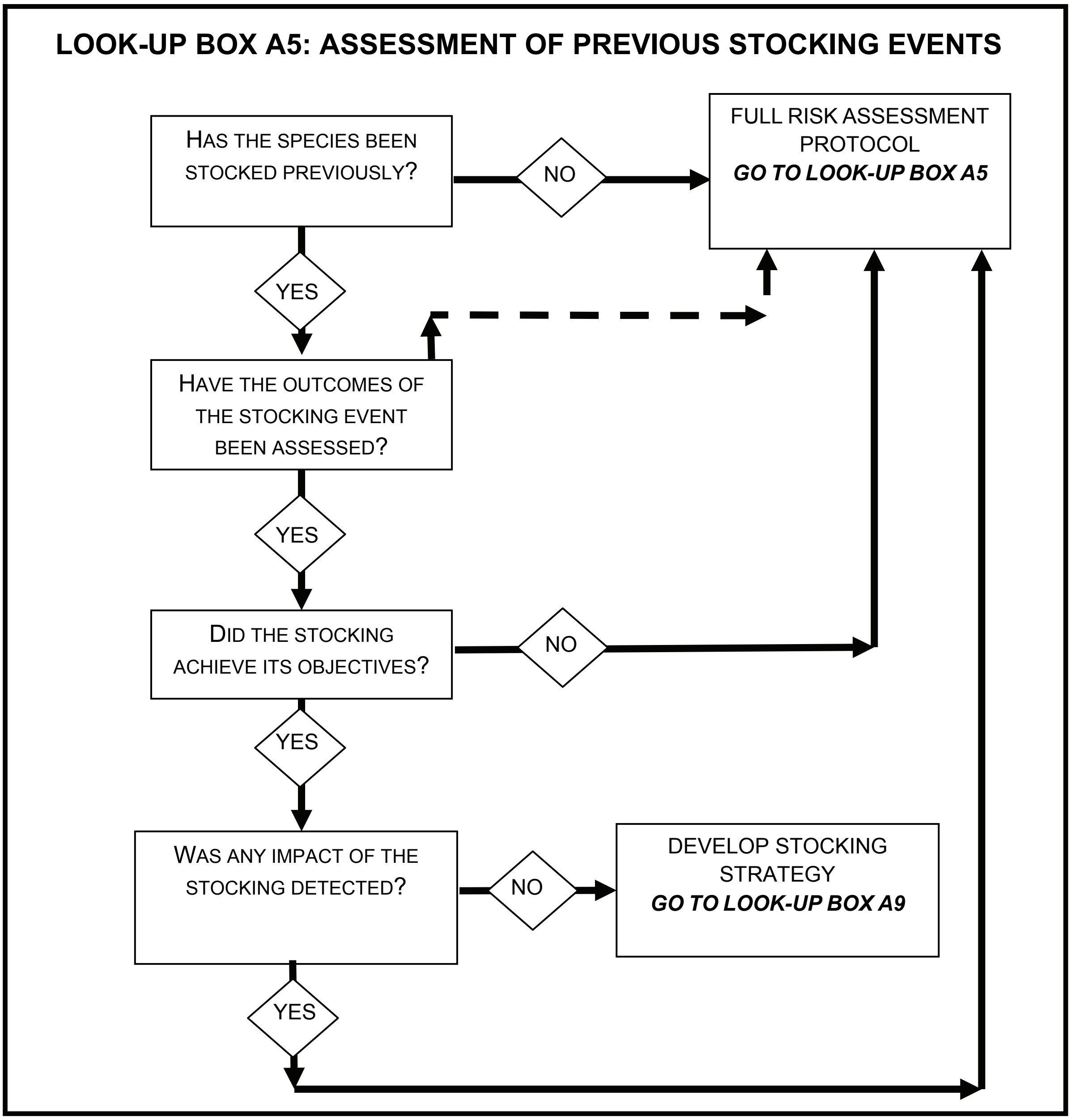


Table A4. Weighting to account for uncertainty about potential risks from stocking. (Note weightings are arbitrarily defined in this example and should be set to reflect the scale risk likely to accrue from the event.

Degree of certainty	**Description**	**Weighting**
High	Well-established knowledge from existing stock enhancement programmes	0.5
Medium	Knowledge from limited stock enhancement programmes supported by documented ecological and environmental studies	1.0
Low	Little or no previous knowledge from stock enhancement programmes and little or no supporting ecological and environmental studies	3.0

Table A5. Suggested weighting to account for degree of isolation of receiving water body.

Degree of isolation of receiving water body	Weighting
Internal, fully recirculatory aquaculture unit	0.5
Isolated still water or garden pond not prone to flooding	1.0
Isolated still water or garden pond prone to flooding	1.5
Still water linked to river or lake by temporary or permanent stream, but with screening of outlet	2.0
Still water linked to river or lake by temporary or permanent stream – no screening	4.0
Open water	5.0

It should also be recognised that the risks associated with stock enhancement can be reduced by mitigatory actions such as quarantining or stocking with reproductively sterile fishes (e.g. triploids). If applied, these procedures should be weighted into the overall assessment.

The risk assessment process addresses the major biological, environmental, and if necessary benefits to region economies (Look-up Box A4). With stocking, it should provide a standardised approach for evaluating the risk of genetic and ecological impacts as well as the potential for introducing non-target species, especially pathogens, which might impact on the native flora and fauna of the proposed receiving water. The following section provides generic guidelines on the procedures and decisions that must be taken when evaluating whether a stocking or introduction event should go ahead. These should be taken prior to applying for a licence to stock to ensure the actions are compatible with the ecological objectives of the receiving water body.

In the first instance, an evaluation of previous stocking events should be carried out as this could potentially avoid in-depth risk assessments (Look-up Box A4). As part of the evaluation, the outcomes of previous events should be assessed and used to decide on the risk assessment procedures and derivation of stocking strategies.

Should a stocking programme be considered, an evaluation of the potential impacts, proportionate to the size/level of risk of the programme, should be undertaken. This takes the form of a series of steps to review the possible ecological, genetic and disease interactions that may arise from the stocking or introduction. In the first instance, the biology of the species to be stocked or introduced, especially relative to the donor and receiving water bodies, should be collated as a baseline for comparison with the faunal status of the receiving water body. The native range and range changes caused by translocation or introduction events should be described to assess what factors limit the species in its native range and if the species is likely to breed naturally in the receiving water. The physiological tolerances (e.g. water quality, temperature, flow requirements, oxygen, salinity) at each life stage (early life stages, adult and reproductive stages) should be collated to assess whether the species will establish and thrive, survive or fail to survive. Other information to be collected relates to the species' habitat, trophic position, reproductive preferences and limits, behavioural traits, migratory behaviour, growth characteristics and longevity, plus potential diseases and parasites. Much of this information may be available on FishBase (www.fishbase.org), but it should be supplemented to ensure the characteristics of the species relative to the donor and receiving waters are evaluated. These baseline inventory data are used to query the ecological, genetic and pathological risks from stocking or introduction on the native biota and receiving ecosystem.

PREDICT ECOLOGICAL RISKS (***LOOK-UP BOX A6***)

Before a stocking (or introduction) is undertaken, the suitability of the recipient habitat should be assessed. Details of physico-chemical factors and environmental tolerances of the proposed species to be released should be included in the evaluation. Although this refers primarily to introducing a new species, it is also relevant to species translocated within its natural distribution range or into water bodies that are not commensurate with its expected ecological guild. Unsuitability of the receiving water habitats may be grounds for rejecting the proposal. If the proposal is to be implemented, the risks of ecological disruption must be assessed, together with levels of uncertainty (see Look-up Box 6). Issues to be examined include interactions through predation and competition, disruption of habitats, whether there will there be niche overlaps with native species and whether there will there be negative impacts on species of high conservation value. Food webs should be constructed using whatever information is available, and the potential effects of the stock enhancement activities on trophic structure evaluated. This would essentially provide an overview of the possible interactions among native and stocked/introduced species. If major gaps of understanding emerge from the above exercise, further research should be conducted on the system. This assessment is largely redundant where the species is being stocked to supplement existing stocks but may be of relevance where a species is introduced into a habitat that is not typical of its natural preferences, but within its natural distribution range. In this case there is the distinct possibility of disruption of food webs and predation as the species widens its niche breadth to accommodate the new habitat conditions.

PREDICT GENETIC IMPACTS (***LOOK-UP BOX A7***)

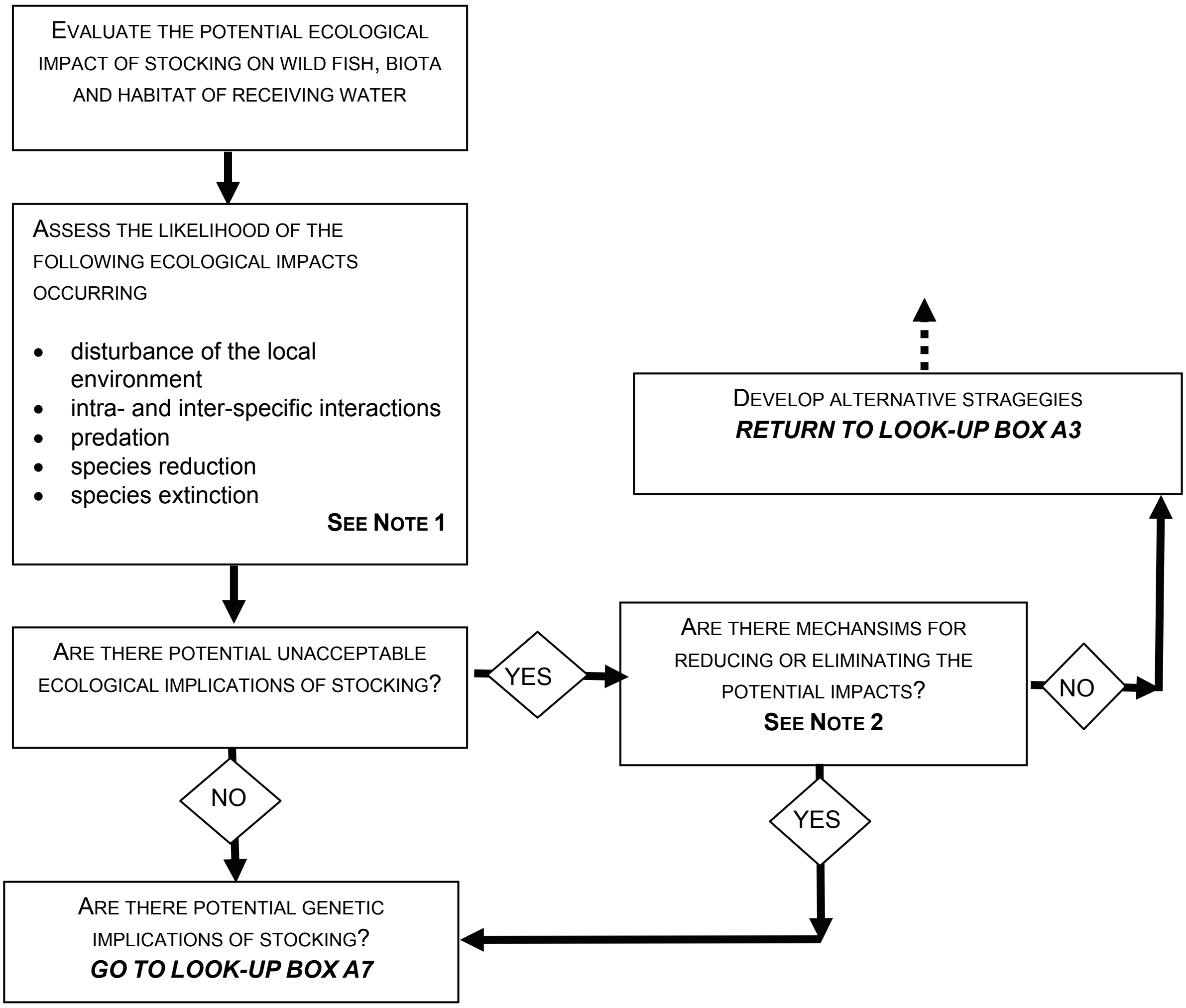
Genetic impacts through hybridisation, inbreeding and loss of genetic integrity can hamper the outcome of stock enhancement programmes. Evidence suggests that stocking, especially of farm-reared fishes, is a threat to the genetic integrity of wild populations through reproductive interactions (Section 3.3.5). Carvalho (1993) and Ryman *et al.* (1995) suggested that the release of fishes should aim to minimise genetically-based changes and to conserve genetic resources. Therefore, if there is a possibility of inbreeding, hybridisation or loss of genetic integrity the programme should be rejected or protocols, such as stocking with triploids, should be adopted to minimise the risk.

PREDICT RISK OF DISEASES AND PARASITES (***LOOK-UP BOX A8***)

There is currently a major concern over the spread and impacts of diseases and parasites related to stocking and introduction events, and there is a need to protect natural environments from unwanted pathogens. Minimising the risks of disease and parasite transference is one of the main criteria that must be achieved to maximise benefits. The MSFHI has an established protocol to assess the health risk posed by stocked fishes to wild stocks. Their role in checking and monitoring fish health and the registration of hatchery facilities is described on the Scottish Government webpage (http://www.scotland.gov.uk/Topics/marine/Fish-Shellfish/FHI) and detailed in the Section on Health Checks below. All fishes stocked into open waters must first be checked for a range of parasites and symptoms of clinical disease (the protocol operates at a set level of confidence of detection). The presence of any one of these pathogens or significant evidence of clinical disease is grounds for rejecting a proposed stocking operation. Fish movements to fully enclosed waters, where the risk of transfer to the wider environment is reduced, do not need a mandatory health check. However, agencies usually advise fishery managers to obtain a health check and insist on health checks for all movements.

The acts of stocking and introduction, irrespective of whether they involve the transfer of pathogens, can elevate the risks of fish disease. Hence, it is equally important to identify the disease potential of stocks in the receiving water and whether and how this might change as a result of stocking (see Look-up box A8).

LOOK-UP BOX A6: ECOLOGICAL RISK ASSESSMENT



NOTE 1: Use baseline inventory data to score the potential ecological impacts according to the table below. The assessment is made using the protocol described in Section A2.2. If the score in any column exceeds 8 either seek mitigatory measures or reject the proposal.
NOTE 2: If the score is high consider possible mechanisms for eliminating or reducing the risk to an acceptable level as described in Section A2.3.

	Likelihood	Consequence **Note 3**	Certainty	Score
	Column 1	Column 2	Column 3	= [(1 + 2) * 3]
Disturbance of the local environment				
Intra- and inter-specific interactions				
Predation				
Species reduction				
Species extinction				
	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = very certain 2 = reasonably certain 3 = reasonably uncertain 4 = uncertain	**Total score =**

NOTE 3: Where the stocking is into a designated (e.g. SAC, SSSI) water body the scoring for consequences can be weighted to reflect the conservation sensitivity of the receiving water body.

LOOK-UP BOX A7: GENETIC RISK ASSESSMENT

EVALUATE THE POTENTIAL GENETIC IMPACTS OF STOCKING ON WILD FISHES AND BIOTA OF RECEIVING

ASSESS THE LIKELIHOOD OF THE FOLLOWING GENETIC IMPACTS OCCURRING

- hybridization
- disruption of local genetic adaptations
- loss of genetic integrity
- impacts resulting from use of polyploidy animals

SEE NOTE 1

ARE THERE POTENTIAL UNACCEPTABLE GENETIC IMPLICATIONS OF STOCKING?

YES

ARE THERE MECHANISMS FOR ELIMINATING OR REDUCING THE POTENTIAL IMPACTS?
SEE NOTE 2

NO

DEVELOP ALTERNATIVE STRATEGIES
RETURN TO LOOK-UP BOX A3

NO

YES

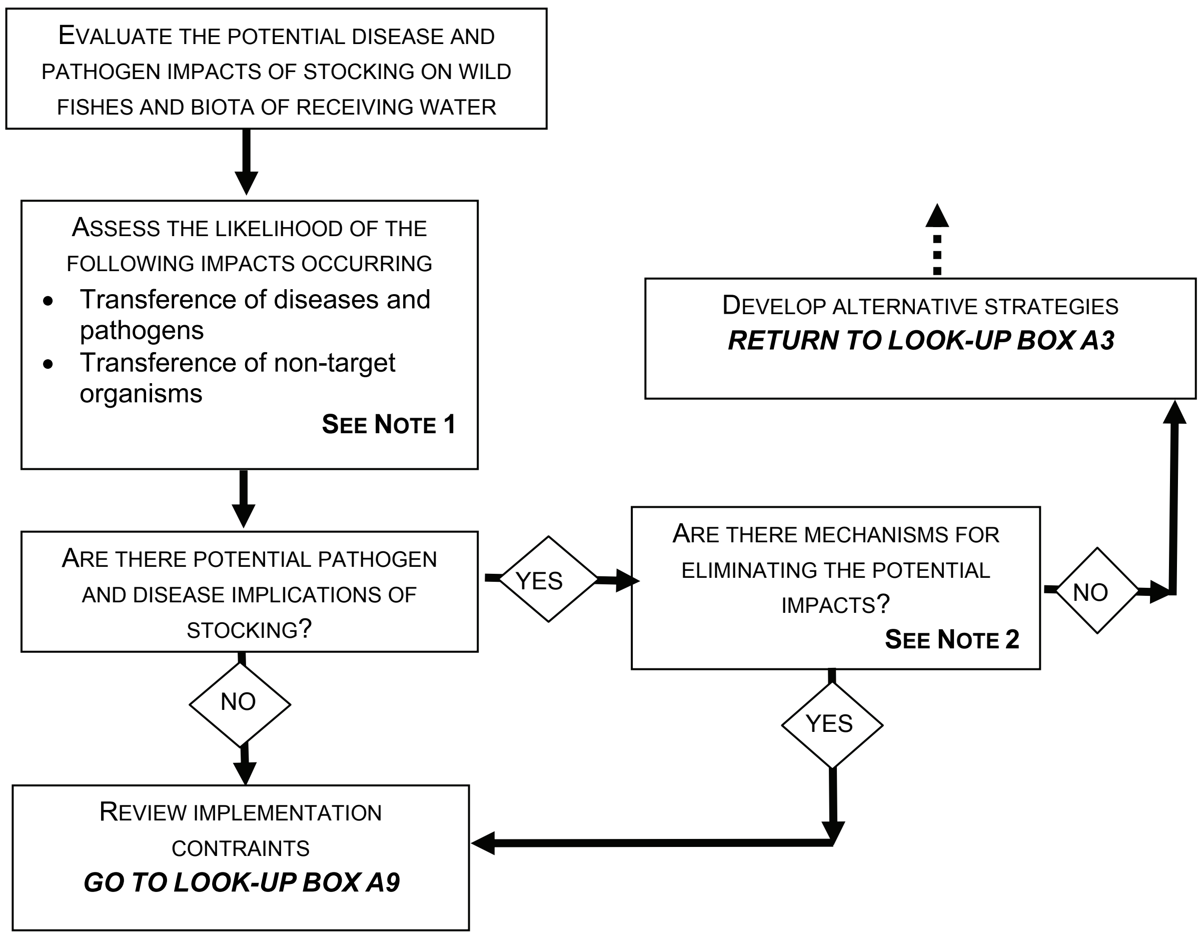
ARE THERE POTENTIAL PATHOGEN, DISEASE AND TRANSFERENCE OF NON-TARGET SPECIES IMPLICATIONS OF STOCKING?
GO TO LOOK-UP BOX A8

NOTE 1: Use baseline inventory data to score the potential genetic impacts according to the table below. The assessment is made using the protocol described in Section A2.2. If the score in any column exceeds 8 either seek mitigatory measures or reject the proposal.

NOTE 2: If the score is high consider possible mechanisms for eliminating or reducing the risk to an acceptable level as described in Section A2.3.

	Likelihood	Consequence	Certainty	Score
	Column 1	Column 2	Column 3	= [(1 + 2) * 3]
Hybridization				
Disruption of local genetic adaptations				
Loss of genetic integrity				
Impacts resulting from use of polyploidy animals				
	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = very certain 2 = reasonably certain 3 = reasonably uncertain 4 - uncertain	**Total score =**

LOOK-UP BOX A8: TRANSFERENCE OF DISEASES, PATHOGENS AND NON-TARGET SPECIES RISK ASSESSMENT



NOTE 1: Use baseline inventory data to score the potential impacts of introducing diseases, pathogens or non-target organisms according to the table below. The assessment is made using the protocol described in Section A2.2. If the score in any column exceeds 8 either seek mitigatory measures or reject the proposal.

NOTE 2: If the score is high consider possible mechanisms for eliminating or reducing the risk to an acceptable level as described in Section A2.3.

	Likelihood	Consequence	Certainty	Score
	Column 1	Column 2	Column 3	= [(1 + 2) * 3]
Transference of diseases and pathogens				
Transference of non-target organisms				
	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = negligible 2 = low 3 = moderate 4 = high 5 = extreme	1 = very certain 2 = reasonably certain 3 = reasonably uncertain 4 - uncertain	**Total score =**

A2.4 *Risk minimisation practices*

Stock enhancement programmes are widespread and can be substantial. Indeed, without stock enhancement programmes it is likely that many fisheries would collapse, which could also have social and economic repercussions. Consequently, fisheries and stocking practices cannot be curtailed, but need to be managed on a more strategic basis. However, there is evidence to suggest that many of these practices are carried out indiscriminately. Furthermore, stocking is an expensive method for managing fish stocks, and it also involves risks that should be assessed before the stocking is approved. Thus, there is a need for a mechanism or protocol to improve the efficacy of stocking operations. There are a number of flow charts that aid this process (Section A3.2), but these need to integrate a greater understanding of the undesirable effects of stocking and introductions to aid the decision of whether or not to stock. The decision trees in Look-up Boxes A4 to A8 provide for this, and the process outlined should be used as a template, in conjunction with other issues on genetics and disease control discussed later, to minimise the harmful effects of stocking (and introductions) and maximise the success of the ventures.

MINIMISING ECOLOGICAL RISKS

Stocking or introducing any species is accompanied by the risk of corrupting native fish communities and the fisheries they support through predation, competition, disease, hybridisation and adverse environmental impacts. The protocol that underlies this report effectively guides the proposer through the information that must be acquired and the issues that must be addressed if a proposal to stock or introduce is to be considered. The most effective mechanism to minimise the ecological risks of stocking or introductions is to produce an impact statement based on known ecological impacts from elsewhere or predicted impacts based on ecological knowledge of the species and biodiversity of the recipient water body. The degree of scrutiny required for the impact statement will depend on whether the water has been stocked in the recent (last five years) past, the scale of the stocking event and the presence of any conservation species or features of ecological interest. A series of generic impact statements for the most commonly stocked species can be used in many cases but tuned to the specific stocking programme being assessed.

It should be noted that it is highly unlikely that any proposal for stocking or introducing a species into new environments can be evaluated to the full. There is no definitive rule of how a species that occupies a certain niche in an unmanaged system or how a previously unstocked system will react when stocked to a higher biomass or numerical abundance in the same or new environments. Consequently, the risks are potentially high, and the only certainty is that the stocking or introduction could affect the ecosystem in some, unknown, manner through predation, competition, habitat degradation or impacts on other biota. Consequently, every effort should be made to prevent the introduction of fish species into areas beyond their natural range, or indeed the translocation of fish species to uncolonized waters within their natural range. Legislation should be put into place and all government agencies should be advised to encourage the use of endemic species for stock enhancement programmes. In the case of England and Wales, the EA's operational instructions ensure that non-native species (except for carp and rainbow trout) are only stocked into enclosed waters, where potential impacts can be contained and more easily managed than in open waters. A similar protocol exists for Scotland that identifies criteria that must be satisfied before a species can be considered acceptable by the Scottish Government for stocking into a particular water body in Scotland. The decision matrix is provided in Table 3 of the review.

MECHANISMS FOR MINIMISING GENETIC DIFFERENCES

There is a need to develop strategies that will minimise the genetic effects of cultured fishes and introduced strains on wild stocks in recipient water bodies. A number of such measures have been identified in relation to hatchery practice and fish releases (Ryman, 1981; Ståhl & Hindar, 1988; Allendorf, 1991; Bergan *et al.*, 1991; Hindar *et al.*, 1991; Waples, 1991; Hindar, 1992; Cowx, 1999). The following provides some mechanisms that should be adopted in relation to hatchery-reared fishes:

- closed culture – this provides secure containment of farmed fishes, for example in land-based operations.
- avoid stocking with species or strains genetically close to those in the receiving water.
- avoid species that have the potential to breed.
- sterilisation – this is an easily induced way of avoiding direct genetic effects, for example through the use of triploid fishes.
- location – locating fish farms far from wild populations, and choosing locations for ranching that minimise straying and may reduce gene flow to wild populations.
- reduced or selective fishing – native populations can be protected by reducing fishing pressure or by directing that pressure towards cultured fishes.
- restrictions on transport – the spread of non-native genes and diseases is reduced by restricting transport of live fishes and eggs..
- gene banks – extinction of local populations can be counteracted by the establishment of gene banks.

Artificial propagation of fishes has proceeded with good intentions, including rehabilitation or supplementation of wild populations and production of food fisheries. The consequences, however, have often been unforeseen and at times overlooked. Since aquaculture for both stocking/introduction and domestic supply is likely to continue, mechanisms to minimise any negative effects are necessary. These include the following.

Minimise genetic difference. Minimising the genetic difference between cultured and wild populations will not stem gene flow, but is a potentially effective means of reducing its negative effects. Many of the methods for accomplishing this, however, are not without problems. For instance, establishment of hatchery broodstocks from local rather than foreign populations will not prevent the apparently inevitable divergence of the cultured population from its wild genotype. Each generation of cultured fish will undergo unintentional artificial selection. Incorporation of wild fishes from local populations into the hatchery broodstock each year may help solve this problem, but the hatchery may become a sink for wild populations, with wild fish being constantly removed to supply the hatchery. Moreover, it will not prevent the harmful effects of mixing gene pools (see Hindar *et al.*, 1991; Waples, 1991). Supportive breeding or supplementation of wild populations, where a fraction of the wild adults are brought into the hatchery for artificial reproduction and their offspring released back into the natal stream, is also problematic as it can lead to reductions in the genetically effective population size, and depletion of genetic variability (Ryman & Laikre, 1991).

Maximise divergence from the wild phenotype. Domesticating fishes to the point where they are unable to breed successfully in the wild, or for farmed fishes even to the point where they are unable to survive in the wild, are likely to be effective means of reducing or eliminating genetic threats. The majority of animals that have been thoroughly domesticated are unable to survive in the wild or successfully breed with wild populations (Hemmer, 1990). This strategy could also be effective in reducing or eliminating ecological interactions when cultured fishes are unable to breed successfully among themselves in the wild or establish feral populations that could ecologically threaten wild populations. While there could be practical problems when trying to implement this option, particularly for ranching, it would allow breeding programmes to concentrate on developing fish specially adapted to local aquaculture environments and thus potentially increase economic benefits to farmers (e.g. Doyle *et al.*, 1991).

Minimising the spread of parasites and diseases

Minimising the inadvertent introduction or transfer of parasites and diseases is an important aspect of any movement of fishes. Many examples of the translocation of parasites and diseases have been identified, although the impacts of these are rarely evaluated in economic or ecological terms. Irrespective, **the risks associated with the transfer of parasites and diseases are high, and measures that minimise or eliminate these problems should be introduced.** Three possible strategies exist (Kohler & Stanley, 1984; EIFAC, 1988; DeKinkelin & Hedrick, 1991):

- improved control over fish movements through legislation;
- veterinary inspections and health checks;
- quarantining.

LEGISLATION

Improved control over fish movements is essential to stem the continuing dispersal of pathogens and the accidental introduction of fishes with consignments of target species. This can only be achieved by improving the understanding of the consequences of introducing and translocating fishes and other aquatic organisms. The EU is likely to play an active role in this respect **under the EU regulation No 708/2007** *'concerning the use of alien and locally absent species in aquaculture* (Council of Europe 10922/5/06 rev 5), Council Directive 2006/88/EC of 24 October 2006 on animal health requirements for aquaculture animals and products thereof, and on the prevention and control of certain diseases in aquatic animals, and the Fish Health Directive (Council of Europe 14117/2/05). Also Directive 93/53/EEC introduces minimum EU measures for the control of List I and II diseases (defined below) (see Diseases of Fish (Control) Regulations 1994 (SI 1994/1447)).

The main provisions are:

- a list of diseases and susceptible aquaculture species;
- approval of farming zones on the basis of this list;
- a principle of freedom of trade between approved zones;
- the obligation to monitor zones and record species introduced;
- a possible protective clause;
- equivalent rules for aquaculture animals or products imported from third countries to be introduced into Community waters.

Generally, three criteria are relevant to the control of diseases: the species to be moved, and the places of origin and destination of the fishes or fish products. The regulation centres on the following: establishment of health status zones according to the presence or absence of specified fish diseases; protection from contamination of approved zones of the EU free of the more serious fish diseases; and the free movement of live fishes and shellfish between farms and zones of equivalent health status.

Directive 91/67/EEC categorises fish diseases into three groups: List I encompasses potentially serious diseases not presently found in the EU; List II, highly infectious or contagious diseases with major economic impacts found in certain areas of the EU; and List III, infectious diseases with less severe economic impacts. Directive 91/67/EEC focuses on preventing both the introduction of List I diseases into the EU, and the spread of List II diseases beyond those areas presently affected.

Ornamental tropical fishes destined to remain permanently in aquaria may be moved freely around the EU without the need for movement documents. Otherwise, species with appropriate documentation that are or are not susceptible either to List I or II diseases may be moved providing specified criteria are met.

Farmed or wild fishes susceptible to infectious haematopoietic necrosis (IHN) or viral haemorrhagic septicaemia (VHS), as well as wild fishes, eggs and gametes of non-susceptible species, may only be imported from approved zones. The eggs or gametes of farmed or wild fishes susceptible to IHN or VHS may only be imported from other approved zones or approved farms in non-approved zones. Farmed fishes, eggs and gametes of non-susceptible species may only be imported from other approved zones, approved farms in non-approved zones or from farms in non-approved zones that do not hold susceptible species and that are not connected to any watercourse, coastal or estuarial waters.

HEALTH CHECKS

Veterinary inspections/health checks form an essential element of all fish movements. Such checks should be mandatory for all movements to reduce the disease risk associated with stocking and introductions, including where fishes are being stocked into fully enclosed stillwaters. The latter is important because fishes may later be moved from fully enclosed stillwaters to other water bodies or escape during extreme floods, and could become a pathogen source. It is recommended that all health checks are carried out by competent and approved authorities. The checks should be based on the source stocks and water bodies (including testing of sympatric species) and not at the point of release. The role of the MSFHI in checking and monitoring fish health and the registration of hatchery facilities is described on the Scottish government webpage (http://www.scotland.gov.uk/Topics/marine/Fish-Shellfish/FHI).

The fish health criteria forming the basis of a refusal to consent to stock should include:

- introductions from sites subject to movement restrictions under the Diseases of Fish Act (1937) (and 1983, as amended) or the EC Fish Health Directive.
- where a health check indicates that either:
    - a Category 1 disease is present;
    - a Category 2 disease is present in the source water; or
    - fishes in the source water or consignment to be released are either clinically diseased or show signs of clinical disease.
- where there is a significant risk of infecting farmed stocks of Atlantic salmon within the immediate vicinity of the receiving site with IPN.

QUARANTINING

Quarantine is usually defined as the placing of organisms under observation, in isolation, whereby their disease and/or parasite status can be assessed and controlled prior to release. This is not the only method of minimising risks, however. Many other procedures can greatly enhance safety, regardless of whether the organisms are kept in isolation. A somewhat broader interpretation of ‘quarantine’ should therefore be applied to imply a process by which risks are minimised, whether or not the organisms are actually kept in isolation for any period (i.e. kept “in quarantine”).

Quarantine procedures should be applied if and when there is a risk of introducing or transferring non-native diseases and/or parasites, whether identifiable or not, together with the organisms in question. The actual procedures recommended depend very much on the disease history of the stocks in question, the expertise available at the point of origin of the stocks, the degree of confidence in the abilities of the exporting agency, and the expertise and facilities available at the destination of the stock to be moved. The safest quarantine strategy is not to move any organisms at all; everything else involves risks. The purpose of an effective quarantine strategy is to minimise the risks as far as is possible, whilst still enabling the movement, and eventual release, of the organisms. In the UK, quarantining is an option rarely chosen when the origin of stocking material is within the country. Instead, restrictions on movement are usually affected if a health check on the source population proves positive for a pathogen. Nevertheless, quarantining may come into effect if the fishes destined for stocking or introduction are imported. In such cases, health checks are the primary strategy for preventing the spread of diseases, but quarantining in secure units may be necessary if there is doubt over the origin and health status of the material.

A2.5 *Cost and revenue considerations (Look-up Boxes A9a and Ab)*

In any proposal, the overall costs and benefits of the stocking programme should be evaluated to ensure that the outcomes are justified in terms of benefits to the locality or region. Benefits relate to the outcomes of stocking events and include:

- harvested yields;
- opportunity of employment;
- changes in fishery status;
- recreational benefits (income, employment); and
- benefits to conservation of endangered species.

Analyses of this type are critical to ensure benefits accrue to the local economies commensurate with the risks to the environment. A simple assessment of this nature should also highlight stocking programmes that have little tangible benefit and reduce the number of unnecessary stocking events.

A.3 Generic strategies for improving stocking success

Guidelines for stocking and introducing fishes are available in many countries (e.g. EIFAC, 1988; ICES, 2005). These are often species-specific or relate to particular types of water bodies. The main issues and options covered in these guidelines have been summarized by Cowx (1994b) and are illustrated in Figure 1. Cowx (1994b) further identified a series of resource problems in project planning that should be assessed, plus other critical conditions that need to be met to ensure successful stocking (Fig. A1). This step-wise approach to planning and implementing stocking programmes ensures that the main ecological and practical aspects are addressed at an early stage.

Essentially, the strategy identifies the mechanisms by which the objectives for stocking programmes will be achieved. Appropriate implementation strategies are essential if stocking programmes are to be a success. The issues that must be considered include:

- source of fishes;
- size of fishes;
- stocking densities;
- species of fishes;
- mechanisms and timing of release;
- pre-conditioning and acclimatisation; and
- handling and transportation.

All these aspects must be taken into account and documented at the planning stage of the exercise to maximise the benefits and minimise any potential risks.

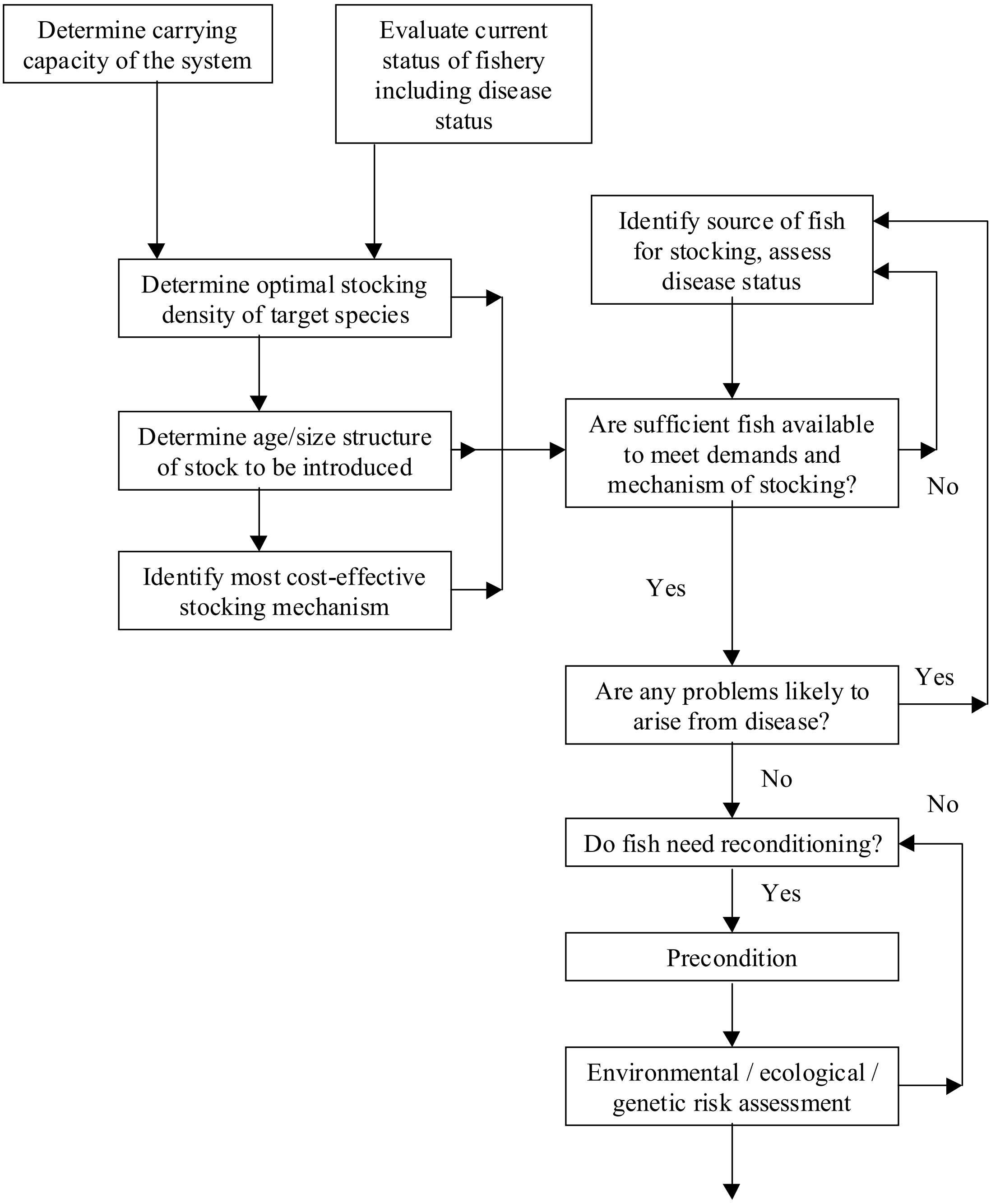


Figure A1. Flow chart illustrating the resource problems that must be considered when planning a stocking exercise (from Cowx, 1994b).

A3.1 *Source of fishes*

There is an increasing awareness of the importance of maintaining genetic integrity of fish stocks. Consideration should therefore be given to minimising the dilution of genetic variation by indiscriminate stocking policies with fishes of unknown origin. Before implementing a stocking programme, a number of options relating to the source of the fishes should be considered.

OPTIONS FOR SYSTEMS WHERE THE SPECIES IS EXTINCT

- donor stock from a system with the same biological characteristics as the recipient system.
- receiving water should be as close to the original site as possible, and preferably within the same catchment
- genetic variability is assured by using a large number of broodstock (see FRS/MS guidance on stocking: Hatchery Work in Support of Salmon Fisheries *Scottish Fisheries Research Report No..65 2007* and Salmon and Sea Trout: To Stock or Not? *Scottish Fisheries Information Pamphlet No. 22 2003).*
- stock chosen from a lake or part of a river with a similar environment (e.g. size of stream, gradient, water temperature, flow regime, altitude, profile).
- artificial propagation based on stock from i) or ii) (sufficient fishes should be used as broodstock to avoid reducing genetic variability of the species).

Whenever possible, stock enhancment programmes for systems where the species is extinct should follow the IUCN and JNCC guidelines.

OPTIONS FOR DEPLETED OR RELICT STOCKS

- build up of stock by hatchery production based entirely on local stock and return brood stock to home system.
- redistribution of adults from elsewhere in the catchment (may be unsuited for introduction to other parts with different prevailing conditions).
- choose stock from a system with a similar environment.

OPTIONS FOR WATER BODIES WHERE NEW SPECIES TO BE INTRODUCED

- farm-reared fishes, certified disease-free.
- stocks from lakes or parts of rivers with similar environments that have been quarantined and certified clear of parasites and diseases alien to recipient system.
- no obvious ecological problems likely to be caused – as with introductions of predators.

Stocked fishes should not have been reared in captivity for more than one generation, to limit the possible effects of selection within hatcheries, thus particular care must be taken when obtaining fishes from hatcheries.

A3.2 *Size of fishes*

Selection of the size of fishes to be stocked or introduced requires knowledge of their likely impacts on native fishes and the ecosystem in general, together with a cost-benefit analysis. The significance of the size or age of fishes released is most apparent for species that undergo size-related or ontogenetic shifts in feeding behaviour or habitat use. For instance, many fish species are initially planktivorous, but switch to piscivory or benthivory with development (Werner & Gilliam, 1984; Vilizzi, 1998; García-Berthou, 2002; King, 2005; Tonkin *et al*., 2006; Nunn *et al*., 2007a, 2008b). The size or age of fish released therefore determines the position they occupy in the food web and, hence, their impacts upon ecosystem functioning and trophic status. Many trout species consume zooplankton and benthic macroinvertebrates when young, but may become increasingly piscivorous as they grow, which may have implications for ecosystem functioning. Releasing fishes at small sizes should reduce the incidence of piscivory and aggressive behaviour towards wild fishes.

In terms of cost-benefit, two main factors influence the size chosen for stocking material: cost and survival. The release of fishes at smaller sizes risks higher mortality, but the cost of stocking material increases exponentially with fish size, especially in slow-growing species, because fewer fishes are needed to obtain the same amount of additional catch from stocking when the size of released fishes is increased. However, this must be balanced against the uncertainty in fishery yield and, hence, economic yield from stocking, which decreases as a function of fish size. It is generally thought that there is a transition size (juvenile bottleneck) after which the yield from stocking is changed from unpredictable to predictable and the uncertainty is lowered considerably. The actual size chosen depends on an empirically determined balance between these two factors, unless there is some feature of the biology of the species that determines the size at which the fishes have to be stocked.

In principle, the size that optimizes the yield from stocking (benefits) in relation to cost of the activity should be preferred. The optimum size to give the maximum benefit should be determined for all stocking programmes. In fisheries where exploitation is well managed, and the fishes allowed to achieve a reasonable size before being exploited, the optimum size is probably somewhere in the early juvenile period. However, if fisheries are poorly managed and the exploitation of young fishes is intense, this point is probably in the larval period because the production costs of the stocking material are much lower.

With the exception of salmon (Table A6), few data are available on the success of stocking different sizes or life stages of fishes. However, the general trend is that migratory and anadromous fishes, such as salmonids, are usually stocked at young life stages (fry) to allow them to acclimate to the natal river and prepare for migration as their size increases. Cyprinids and other non-migratory forms are generally stocked at a larger stage (fingerlings ~ 12 cm) as they are often supplementing a failure in natural recruitment. These fishes are expected to grow on to a large size based on the natural productivity of the stocked water body.

Table A6. Performance of stocking salmon into rivers at different densities and life stages (data from Cowx, 1994b and references therein).

Stage	Density m^{-2}	% survival to end of growing season	Estimated smolt production 100 m^{-2}
Green ova	6.2-59.0	1.7-4.0	4.3-10.0
Eyed ova	0.4-11.0	3.5-19.4	8.8-48.5
Unfed fry	0.3-29.3	1.3-38.6	3.3-96.5
Fed fry	0.1-1.8	6.7-22.7	2.5-56.8

Recreational fisheries are increasingly tending to rely on even larger fishes of catchable size and less on growth in the natural environment. Specialist fisheries stock large-sized individuals to attract anglers, who are willing to pay high prices to capture specimen-sized fishes, particularly carp. Indeed, many fisheries in France and the UK are deliberately stocked with carp greater than 10 kg in weight. Put-and-take trout fisheries also stock with table-sized fish as these individuals are given little opportunity to increase in size. It is estimated that more than 80% of captures occur in the first 40 days after stocking, with overwinter survival of uncaught fishes being low.

A3.3 *Stocking densities*

One of the greatest concerns with stock enhancement programmes is that they rarely consider the capacity of the recipient system to support the enhanced stocks (Kelly-Quinn & Bracken, 1989). If too many fishes are present, increased mortality rates, through predation and starvation, reduced growth rates and increased dispersion, generally follow. Thus, whilst stocking and introduction may produce large increases in fish numbers at certain times or in localised areas, no more fish will survive than the resources will allow. Evidence for such a competitive bottleneck has been provided by Hegge *et al*. (1993), where the capacity for enhancing trout stocks in a stream was limited by benthic feeding conditions. In worst-case scenarios, overstocking can lead to reductions in the performance of fisheries, below that prior to the introductions. For example, when the spawning stock of salmon exceeds an optimal level, the number of smolts produced may decrease (SAC, 1991). For fisheries already subjected to stocking activities, reducing stocking densities should reduce the potential for competitive interactions between native and stocked fishes, as pressure for finite resources is reduced. This is of particular importance for water bodies that support unique strains of brown trout, charr and whitefish. Reducing stocking densities should also minimise any detrimental impacts on the ecosystem as a whole.

Determination of optimal stocking densities should be based of assessment of the carrying capacity of the receiving water body, and be commensurate with the risk and scale of the stocking programmes. For lakes, the optimal density can be determined from relationships between environmental parameters such as shore-line development and water depth and fish biomass (Leopold & Bninska, 1984). This has been further developed by Medley & Lorenzen (2006) to estimate optimal stocking density for culture-based fisheries. Unfortunately this model relates to fisheries where stocks are exploited, and not necessarily to recreational put-and-take or catch-and-release fisheries, where densities are often kept artificially high to increase angler satisfaction. No definitive relationships are available for calculating stocking densities of different species in rivers; these are generally based on the experience of the fishery managers. Hickley (1994) suggested that a database could be produced to provide guidance on the stocking densities that maximise the benefits in terms of improving stocks. Such a database should be based on the measured success of stocking at different densities. Thus, effort is required to construct tables to indicate the success of stocking of all species at different densities. This can only be achieved if the outcomes of stocking programmes are evaluated and reported. When calculating stocking densities, consideration must be given to the existing stock biomass, the residual stock remaining from previous stocking events, and allowances should be given for migration/dispersal, predation and predicted survival of the stocked fishes. Values of between 10 and 80% annual mortality are given in the literature (EIFAC, 1984), so compensatory densities will be difficult to determine. The most important issue is that overstocking is avoided.

A3.4 *Species of fishes*

The impacts of stock enhancement programmes on the recipient water bodies depend partly upon the species of fish released (see Cambray, 2003; Gozlan, 2008). For example, there is evidence that zander can have significant impacts on fish populations (Linfield & Rickards, 1976; Fickling & Lee, 1983; Linfield, 1984; Hickley, 1986; Smith *et al*., 1998). Indeed, the introduction of zander into Lake Egridir, Turkey, resulted in the worldwide extinction of two endemic *Phoxinellus* spp. and considerable declines in the biomass of other cyprinid populations (Celikkale, 1990). Similarly, declines in a number of populations of whitefish species, including the powan in Loch Lomond, are thought to have been partly due to the spread of ruffe, which may feed on their eggs (Adams & Tippett, 1991; Ogle, 1998; Winfield *et al*., 1998; Etheridge *et al*., 2011), and trout (or trout farming) are believed to have had negative impacts on a number of Scottish natural heritage sites (e.g. Lake of Menteith, Lindores Loch, Butterstone Loch; Section 3.6). Stocking may also lead to undesirable changes in habitat that may impact on the populations of indigenous species the programme is designed to enhance. For example, the introduction of grass carp may greatly reduce the growth of aquatic macrophytes (Cross, 1969; Stott, 1977), which may be reflected in the productivity of other species that use the vegetation either directly or indirectly. Moreover, by selectively feeding on soft-leaved species, grass carp can lead to an increase in the biomass of tougher (ligninous) species, which may be more of a nuisance than the macrophytes originally targeted for control (Wells *et al*., 2003).

The stocking or introduction of piscivorous fishes can initiate trophic cascades that decrease phytoplankton biomass and increase water clarity (Geist *et al*., 1993; Frankiewicz *et al*., 1996, 1999; Berg, 1998; Dörner *et al*., 1999; Dörner & Benndorf, 2003; Radke *et al*., 2003; Skov *et al*., 2003; Skov & Nilsson, 2007). However, if stocked or introduced fishes are zooplanktivorous, increased zooplanktivory may decrease the abundance of large-bodied zooplankton (e.g. *Daphnia* spp.), and result in an increased biomass of algae and lower water transparency. Elser *et al*. (1995) examined the effects of discontinuing the stocking of rainbow trout on food-web interactions and ecosystem properties (e.g. light penetration, primary productivity) in an oligotrophic lake (Castle Lake, USA). Contrary to their expectations, the reduction of rainbow trout abundance resulted in compensatory increases in the abundance of other zooplanktivorous fishes, with consequent increases in planktivory on daphnids, increased algal biomass and decreased water clarity.

The selection of fish species to stock or introduce should therefore be based upon knowledge of their likely impacts on native fishes and the ecosystem in general. Species that are ecologically similar to native fishes are most likely to compete for resources, whereas dissimilar species may potentially alter ecosystem functioning through occupation of vacant niches. Stocking triploids has the potential to avoid inter-breeding between stocked and native fishes. This is of particular importance for water bodies that support unique strains of species. However, triploids may interfere with the post-spawning recovery of wild fishes.

A3.5 *Mechanism and timing of release*

There is a considerable volume of literature on the most appropriate time to stock or introduce salmonids in terms of maximising stocking success. The general conclusion is that stocking in the spring is more efficient (4-12 times) than in the winter (Cresswell, 1981; Aass, 1993; O'Grady, 1984). Ideally, fishes should be released when flow rates and water temperatures are low, to minimise fish displacement and stress. In addition, releases should preferably take place when the productivity of the receiving water body is high, but not during the spawning period as the released fishes may interfere with natural reproduction processes. Stocking in early summer, after the coarse fish spawning season and when natural food availability is high, is preferable to allow the fishes to acclimatise to conditions in the receiving water body before overwintering.

There is less information on the effects of time of release on the dynamics of receiving water bodies. An exception is Hembre & Megard (2005), who investigated how the timing (autumn versus spring) of rainbow trout stocking affected ecosystem functioning in a Minnesota lake. *Daphnia* spp. were almost eliminated from the lake during winters after trout were stocked in the autumn. Stocking in spring alleviated predation over the winter, but increased predation on *Daphnia* spp. during the spring, summer and autumn. However, the high mortality caused by spring-stocked trout was offset by higher rates of reproduction by the relatively large populations of fecund *Daphnia* spp. that survived the winter. Grazing by dense populations of *Daphnia* spp. increased water clarity during May and June that were inhibited in autumn-stocking years.

Three mechanisms for releasing fishes are used, namely spot planting (releasing all the fish in a single batch), scatter planting (simultaneously releasing batches of fish at several locations) and trickle planting (releasing batches of fish over an extended time period) (Cowx, 1994a). Spot planting can lead to competition amongst released fishes and with native stocks, and in rivers is often associated with downstream displacement of fishes (Cresswell, 1981). Scatter planting minimises the potential for competitive interactions by reducing overdispersion of released fishes. Similarly, trickle planting minimises the potential for competition, but is often constrained by lack of manpower, finance and available stock. Evidence suggests that, in terms of stocking success, scatter and trickle planting should be preferred over spot planting (Berg & Jorgensen, 1991; Fjellheim *et al*., 1993). There appears to be little information regarding the effects of these mechanisms of release on the receiving water body, but it could perhaps be assumed that impacts would be greatest following spot planting, due to the artificially high and localised fish densities associated with this practice.

The favoured mechanism of release should be trickle planting, as it has the potential to reduce both competition with native fishes and impacts on the ecosystem in general by releasing batches of fish over an extended time period. However, if manpower, finance or available stock are limited, scatter planting could be employed, with batches of fish released simultaneously at several locations. This technique reduces the potential for competitive interactions by minimising overdispersion of the released fish (Berg & Jorgensen, 1991; Fjellheim *et al*., 1993).

A3.6 *Pre-conditioning and acclimatisation*

As previously discussed, pre-conditioning fishes to prevailing conditions in the receiving water body potentially improves their survival. For example, fishes that are farm-reared or are to be transferred from still to running water should be exposed to running-water conditions for an extended period before their release. This exercises the red-muscle tissue in the fish, increasing their ability for sustained swimming (Broughton *et al*., 1981; Fisher & Broughton, 1984). Acclimatisation to temperature is also thought to be important (Philippart & Baras, 1988). More importantly, pre-conditioning exposes the fishes to the natural habitat characteristics and instils life skills to help avoid predation and improve feeding success. It is recognised, however, that pre-conditioning and acclimatisation may not be possible in many situations.

A3.7 *Handling and transportation*

Handling and transportation causes stress and possibly damage to fishes, which can subsequently affect post-stocking survival. As a result, procedures that minimise handling time or frequency should be adopted from the time of capture of the donor fishes to planting into the recipient system. Berka (1986) provided an overview of the procedures for transportation of fishes. These procedures should be applied each time fishes are moved, especially over long distances or for extended periods of time.

- The techniques employed to capture the fishes in the first instance should cause minimal damage; seine netting and electric fishing are the preferred techniques. During collection and transportation, handling should be avoided where possible. Fishes should be transported in low densities and provided with an ample supply of oxygenated water.
- All fishes should be starved for at least 24 hours prior to transportation to reduce oxygen demand, due to increased respiration rates during digestion, and minimise ammonia production. If the fishes are to be transported long distances consideration should be given to reducing the effective toxicity of unionised ammonia by lowering the temperature and pH.
- The use of suitable anaesthetics should be considered to reduce physical activity and hence both the risk of damage and rate of respiration.

- There is no point in stocking fishes that are in poor condition or health, as this will affect the success of the stocking programme.

A3.8 *Assess feasibility and acceptability of stocking programme (Look-up Box A9)*

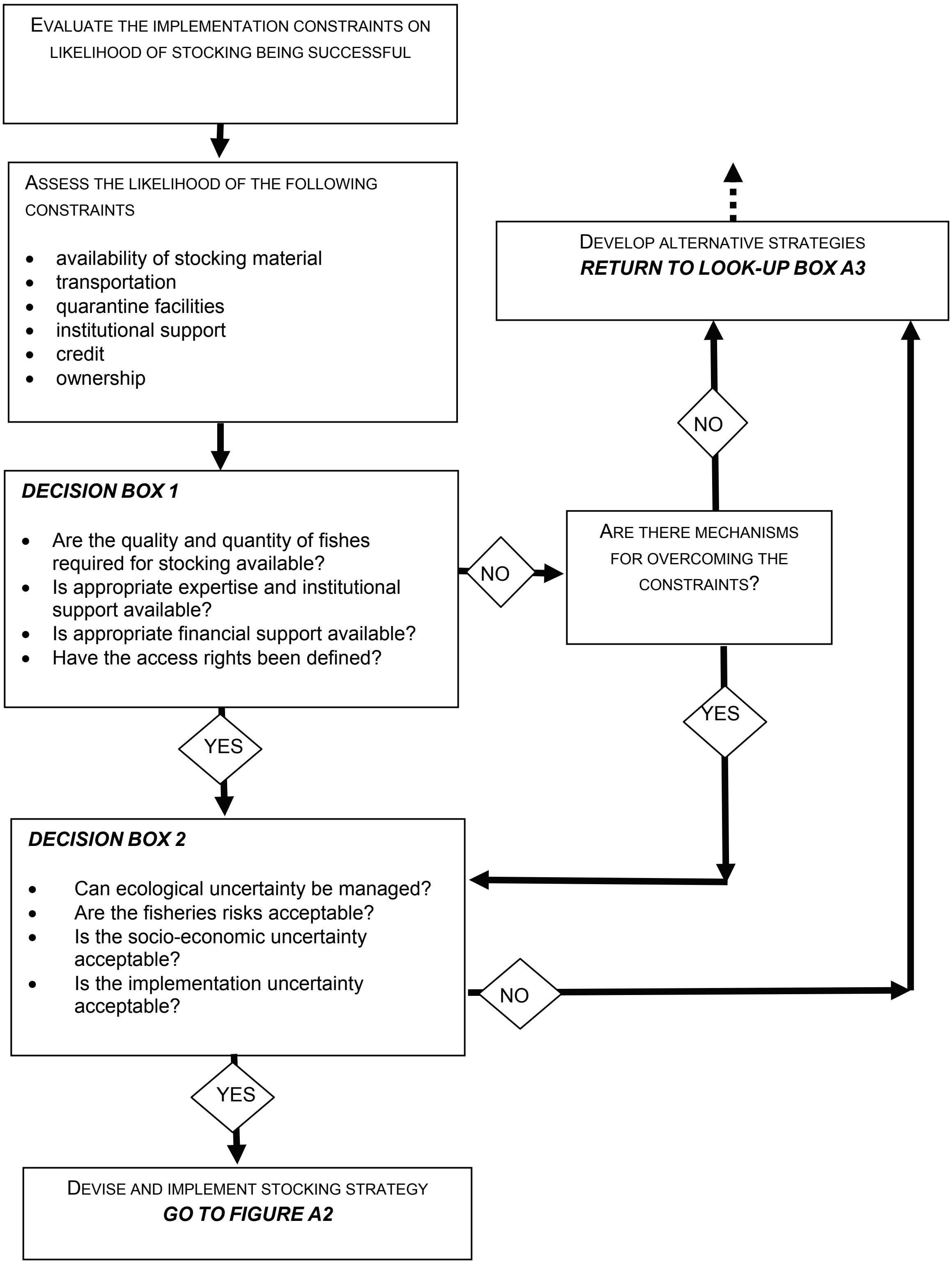
Stocking is an important tool in the management of fisheries, but the feasibility or practicability of proposed stocking programmes must be assessed, commensurate with the size of the stock enhancement programmes and/or the associated risks, before they are allowed to go ahead. Assessments should be based on a brief study that examines whether the objectives and defined outcomes of the stock enhancement programmes are achievable within socially acceptable environmental, genetic and ecological levels of risk. In essence, the proponents of stock enhancement programmes should provide a summary report that includes basic information about the recipient water body, physico-chemical information about flash and normal flooding time, duration, water level, water and soil conditions, water-body management system, inlet and outlet channels, fishermen access trends, fish community structure and abundance, fish migration routes and any conservation-related issues. This information should be used to appraise the potential benefits and impacts and associated risks from stocking on the receiving ecosystem and associated biota. The various issues that need attention are highlighted in Look-up Box A9. In cases where the potential risks and uncertainty about the impacts and benefits are high, an independent appraisal should be conducted. Much of the data can be provided in a generic form once the initial stocking or similar event has been appraised. When evaluating stocking proposals, the "do nothing" option should not be disregarded but should be considered as fully as any of the other options under discussion, despite possible public pressure to stock. Generally, it would be expected that all stocking programmes should be economically viable and contribute to the well-being of the stocks. Unfortunately, financially-driven enhancement programmes are rarely successful because the returns in terms of increased yield (revenue) do not usually cover the costs of the stocking programmes. The 'precautionary principle' should be adopted if any adverse impacts are foreseen.

If the decision-making authority approves the proposal, an executive plan (working plan) should be produced to implement the enhancement programme. All projects should have in place the methodology to enable adequate monitoring of progress and, ultimately, success or failure. This post-stocking appraisal should include a mechanism of disseminating the outcome to minimise the risks any unforeseen adverse effects in future exercises.

A3.9 *Post-stocking monitoring*

Integral with this process should be a post-stocking monitoring programme to measure the outcome of the intervention. Any measure of the success of a stocking programme will depend on the extent to which its objectives are realised. These may vary: for instance, when stocking commercial (capture) fisheries the usual measure is the extent to which the financial value of the catches is improved, whereas in recreational fisheries the criterion is a more elusive one of angler satisfaction. Whichever criterion is used, data on the stocking programme, including economic costs and benefits, are needed. Post-stocking monitoring programmes should also include fish health monitoring when the fishes are captured and species-specific harvesting data recorded by number and weight (Cowx, 1998c).

LOOK-UP BOX A9: REVIEW OF IMPLEMENTATION CONSTRAINTS



After implementation, it is desirable to evaluate stocking programmes on the basis of ecological, economic, genetic, disease and parasite risks and social aspects. In this context, an evaluation plan, proportionate to the scale and potential impacts of the stocking programme, should be prepared and executed. This should run over at least a 3-5-year period, preferably longer where intensive stocking or predatory species are concerned, and include technical, ecological, genetic and social considerations. The long-term holistic approach will assist in identifying and solving:

- impacts on the habitats (e.g. loss of aquatic vegetation, changes in the composition of aquatic vegetation, increases in dissolved solids and turbidity) of recipient ecosystems;
- impacts on the trophic dynamics of recipient ecosystems (e.g. changes in the quality and quantity of plankton communities, increases in single age groups of particular fish species, changes in the quality and quantity of benthic organisms);
- changes in the genetic integrity of stocked/resident fish species (e.g. the presence of hybrids, deformed fishes, fish maturing earlier or later than conspecifics in similar water bodies, egg quality, survival of larvae and juveniles);
- impacts of latent disease and parasites, which were not detected during quarantine;
- changes in species and catch composition;
- changes in growth performance of stocked/resident fish species;
- changes in production trends of stocked/resident fish species;
- changes in the socio-economic conditions of people related to the fisheries.

This post-stocking appraisal should include a mechanism of disseminating the outcomes to highlight the risks of any unforeseen adverse effects in similar exercises.

Appendix 2. Stocking densities/biomasses reported for a number of fish species and life stages in various waterbodies.

Species	Life stage	Waterbody	Densities/Biomasses	Comments	References
Atlantic salmon	Green ova	Various rivers	6.2-59.0 m^{-2}	Estimated 4.3-10.0 smolts 100 m^{-2}	Cowx (1994c)
Atlantic salmon	Eyed ova	Various rivers	0.4-11.0 m^{-2}	Estimated 8.8-48.5 smolts 100 m^{-2}	Cowx (1994c)
Atlantic salmon	Eyed ova	Various rivers, England and Wales	1.70-4.50 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Harris (1994), cited in Aprahamian *et al*. (2003)
Atlantic salmon	Ova and fry	Various rivers, Scotland	2-5 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Egglishaw *et al*. (1984) , cited in Aprahamian *et al*. (2003)
Atlantic salmon	Fry (unfed)	Various rivers	0.3-29.3 m^{-2}	Estimated 3.3-96.5 smolts 100 m^{-2}	Cowx (1994c)
Atlantic salmon	Fry (unfed)	Various rivers, England and Wales	1.70-4.0 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Harris (1994), cited in Aprahamian *et al*. (2003)
Atlantic salmon	Fry	Various rivers	0.1-1.8 m^{-2}	Estimated 2.5-56.8 smolts 100 m^{-2}	Cowx (1994c)
Atlantic salmon	Fry	River Viantienjoki, Finland	0.63-2.11 m^{-2}	Point and scatter stocking in early June both appear to be suitable in small rivers	Jokikokko (1999)
Atlantic salmon	Fry	Various rivers, England and Wales	0.60-1.80 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Harris (1994), cited in Aprahamian *et al*. (2003)
Atlantic salmon	0+ parr	Various rivers, England and Wales	0.15-0.40 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Harris (1994), cited in Aprahamian *et al*. (2003)
Atlantic salmon	1+ parr	Various rivers, England and Wales	0.05-0.20 m^{-2}	Recommended densities for stocking to maximise smolt output, depending upon habitat quality	Harris (1994), cited in Aprahamian *et al*. (2003)

Barbel	Adult	Four stillwaters, England (0.3-2.0 ha)	>500 kg ha^{-1}	Growth depressed when common carp present. Growth similar to natural habitats when density <200 kg ha^{-1} and carp absent	Taylor *et al*. (2004)
Brook trout	Adult	Two oligotrophic lakes in northern Ontario, Canada	2 kg ha^{-1}	Significant increase in mean shoal sizes of potential prey species following stocking, no statistical changes in abundance or habitat use	Pink *et al*. (2007)
Brown trout	Fry	River Viantienjoki, Finland	0.63-2.14 m^{-2}	Point and scatter stocking in early June both appear to be suitable in small rivers	Jokikokko (1999)
Brown trout	1+ parr	Various rivers, Poland	5 m^{-2}	–	Zalewski *et al*. (1985)
Brown trout	0+ and >0+	Various rivers, Germany, Czech Republic, France and Finland	Mean 2861 ha^{-1} (range 0-8710 ha^{-1}) and 119 kg ha^{-1} (range 0-478 kg ha^{-1})	Status of fish populations in 20 freshwater pearl mussel streams	Geist *et al*. (2006)
Coarse fish	n/s	Mature acid/neutral upland lakes/reservoirs	100 kg ha^{-1}	Approximate natural density. In waterbodies of high conservation value it may be necessary to restrict stocking to ensure that densities do not affect designated features	EA (2006)
Coarse fish	n/s	Recently created lakes/gravel pits	150 kg ha^{-1}	Approximate natural density. In waterbodies of high conservation value it may be necessary to restrict stocking to ensure that densities do not affect designated features	EA (2006)
Coarse fish	n/s	Mature gravel pits	250 kg ha^{-1}	Approximate natural density. In waterbodies of high conservation value it may be necessary to restrict stocking to ensure that densities do not affect designated features	EA (2006)
Coarse fish	n/s	Mature lowland lakes	350 kg ha^{-1}	Approximate natural density. In waterbodies of high conservation value it may be necessary to restrict stocking to ensure that densities do not affect designated features	EA (2006)

Coarse fish	n/s	Eutrophic ponds	500 kg ha^{-1}	Approximate natural density. In waterbodies of high conservation value it may be necessary to restrict stocking to ensure that densities do not affect designated features	EA (2006)
Coarse fish	n/s	New fisheries	150 kg ha^{-1}	–	IFM (1991)
Coarse fish	n/s	Established fisheries	300-400 kg ha^{-1}	–	IFM (1991)
Coarse fish	n/s	Nutrient-enriched shallow lakes	<10-40 kg ha^{-1}	>10-40 kg ha^{-1} likely to suppress *Daphnia* spp. populations and lead to increases in phytoplankton biomass	Moss *et al*. (1996)
Coarse fish	n/s	280 stillwaters, England and Wales	10 to 126 000 ha^{-1} and <5 to >14 000 kg ha^{-1}	50% of fisheries contained <5000 ha^{-1} and <370 kg ha^{-1}	North (2001, 2002)
Common carp	Adult	Mesocosms in four shallow 0.4 ha ponds, USA	>174 kg ha^{-1}	Increased turbidity, suspended solids and total phosphorus, reduced macrophyte and macroinvertebrate abundance	Parkos *et al*. (2003)
Common carp	Adult	Mesocosms in Little Mere, Cheshire, England (eutrophic, 1.5 m max. depth)	>200 kg ha^{-1}	Adverse impacts on macrophytes	Williams *et al*. (2002)
n/s	n/s	n/s	<100 kg ha^{-1}	>100 kg ha^{-1} has strongly detrimental impacts (reduced biodiversity, loss of submerged plants, increased turbidity)	SEPA (2000)
Rainbow trout	Adult	Long Lake, Minnesota, USA (dimictic, 66.5 ha, 7.63 × 10^{6} m^{3}, 2.4 km long, 24 m max. depth, 13 m mean depth)	~75 ha^{-1}	*Daphnia* spp. nearly eliminated during winters after trout stocked in autumn. Impacts less severe for spring stocking	Hembre & Megard (2005)
Rainbow trout /brown trout /brook trout	Adult	Five meso-eutrophic lakes in Alberta, Canada (mean 19.5 [range 3.3-28.8] ha, 10.3 [range 6.0-12.5] m max. depth)	Mean 495.4 (151-733) ha^{-1}	Forage fish largest in stocked lakes, consistent with size-limited predation. No demonstrable effects on abundance	Nasmith *et al*. (2010)

Sea trout	0+ fry	Afon Iwrch, Wales	Trickle stocking at 0.75 m^{-2} recommended in September	Estimated ~0.03 smolts m^{-2}	Scott *et al*. (1997) , cited in Aprahamian *et al*. (2003)
Sea trout	Fry	Various rivers, Wales	0.40-1.50 m^{-2}	–	Wyatt (1989); Hoggarth (1992); Hoggarth *et al*. (1992) , all cited in Aprahamian *et al*. (2003)
Sea trout	1+ parr	The Møbæk, Øster Velling bæk and Hjorthede bæk tributaries of the River Gudenå, Denmark	1.0-2.0 m^{-2}	Wild trout negatively affected by introduction of domestic trout and wild trout from another stream. Smolt yields at 2+ were 3.2% (for 0+ trout stocked in autumn) and 7.0% (for 1 + trout stocked in spring) of the domestic trout stocked	Berg &Jorgensen (1991)
Trout	n/s	n/s	75-100 kg ha^{-1}	–	IFM (1991)

n/s = not stated

www.snh.gov.uk

ISBN: 978-1-85397-865-4

Policy and Advice Directorate, Great Glen House,
Leachkin Road, Inverness IV3 8NW
T: 01463 725000

You can download a copy of this publication from the SNH website.

recycle

Scottish Natural Heritage
Dualchas Nàdair na h-Alba
All of nature for all of Scotland
Nàdar air fad airson Alba air fad