ICES Mariculture Committee
ICES CM 2004/F:02 Ref. ACME

Report of the Working Group on Environmental Interactions of Mariculture (WGEIM)

5–9 April 2004
Galway, Ireland

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Contents

1 Opening of the meeting

Dr Edward Black (Chair) opened the 2004 meeting of the Working Group on the Environmental Interactions of Mariculture (WGEIM), at the Marine Institute, on 5 April 2004 in Galway, Ireland. This year's meeting was attended by ten members from seven countries and included an observer from the Irish Aquaculture Industry (see Annex 1). The membership constituted a range of expertise able to cover the all terms of reference for this meeting.

The group was welcomed to the Marine Institute on behalf of the Director, Michael Ơ'Cinnéide, by Dr Terry McMahon. The Chair expressed the Working Group's appreciation for the excellent staff and technical support that had been arranged to facilitate the efforts of the Working Group.

2 Adoption of Agenda

The proposed agenda was presented and adopted with only minor modifications of the Agenda. The adopted agenda is presented in Annex 2.

3 Terms of Reference for the meeting

The **Working Group on Environmental Interactions of Mariculture** [WGEIM] (Chair: E. Black, Canada) will meet in Galway, Ireland from 5–9 April 2004 to:

a) comment on the report of a Workshop to be organised jointly by the Xunta de Galicia and the Instituto Espanol de Oceanografia, Spain in 2003 on stock enhancement in the Galician rias;
b) update developments in the implementation of the Water Framework Directive, and activities arising from the European Commission policy on sustainable aquaculture;
c) prepare for possible publication a report on the "state of knowledge" of the potential impacts of escaped aquaculture marine (non-salmonid) finfish species on local native wild stocks (e.g., sea bass, sea bream, cod, turbot, halibut);
d) discuss risk assessment methods in relation to mariculture in a joint session with GESAMP WG 31;
e) conduct an analysis of the literature and research on the current bath treatments and in-feed additives (treatments) used to treat salmon for sea-lice, and produce a synthesis (state of knowledge) on their fate in the near and far field environment and their effects on non-target organisms (e.g., crustaceans and invertebrates).

WGEIM will report by 15 April 2004 for the attention of the Mariculture Committee and ACME.

Scientific Justification:	a) The rias of Galicia are the most important area for the production of farmed shellfish in western Europe. However, the very heavy reliance on mollusc (mussel, oyster, etc) cultivation has resulted in the large numbers of small businesses which comprise the bulk of the industry being vulnerable to external factors such as harmful algal blooms, climate change, market forces, etc, which are outside their control. There is therefore considerable interest in Galicia in both diversification and expansion of the industry. Similar pressures applying elsewhere in European aquaculture, for example the heavy reliance on salmon in Scotland and Norway, have similarly led to moves towards diversification. The purpose of the review is to assess the approach taken to resource allocation and prioritisation of species/techniques for development, which must balance

	environmental, technological, social, and economic factors. The workshop report will also contribute to the continuing WGEIM task to report on the potential impact of escaped (stocked) organisms on localized native stocks. b) The EC policy on Sustainable Aquaculture sets a new context for the aquaculture industry in the EU. It holds out the possibility, among other things, that Integrated Coastal Zone Management will become the normal approach to the management of the aquaculture development, and that new tools and processes will arise from the new policy. The Water Framework Directive will determine the direction of water quality regulation and improvement in the EU over the next 10–20 years. The coincidence of major new policy initiatives in both industrial development strategy and environmental quality presents European aquaculture with a unique set of opportunities and risks. c) In order to foster a sustainable development of coastal and marine aquaculture, there is a need to diversify production and to cultivate new species. A pro-active approach is required to avoid mistakes made previously when salmonid farming was developing. Mitigation strategies based on sound scientific criteria in relation to the species under consideration need to be prepared at an early stage of development. Studies would have to consider the status of the natural stocks in the area, the potential genetic, trophic and behavioural interactions, and, foremost and specifically, the development of methods for recovery of escaped fish in the event of large-scale escapements. This subject seems to be of particular importance for non-migratory fish stocks with small, localised populations (e.g., sea bass and seabream), or migratory species with different migratory patterns than salmonids (e.g., cod, halibut, turbot, and wolffish and other species). The WGAGFM will be asked for genetics interaction advice. The report will include an overall risk assessment and recommended mitigative strategy. d) The ICES WGEIM would greatly benefit from inputs of the GESAMP WG31 because risk assessment methodologies have not yet been addressed in its previous meetings. A critical factor in the evaluation of risks and definition of risk management option for member states to control the potential interactions between wild and cultured aquatic organisms is an understanding to the structure of population units of evolutionary significance. With the development of culture activities for these species there now a is a need to invest in studies on stock discrimination for sea bass and sea bream in coastal habitats of ICES member countries. This will enable better management of existing resources and allow integration of aquaculture into the existing mix of coastal resource users for member states. e) Documentation is available on the use of many different treatments and chemicals used in salmon aquaculture to treat sea-lice infestations. The trend has been to reduce the use of broadcast chemicals and baths and concentrate on those that have a direct effect on sea-lice in contact with the fish such as in-feed treatments. There is a need to conduct an analysis and synthesis of the recent literature and research conducted in countries producing salmon and provide a state of knowledge report on the fate of these baths and in-feed additives/treatments in the environment and their effects on non-target organisms such as crustaceans (e.g., lobsters) and invertebrates (e.g., bivalve molluscs, sea urchins) that are commonly found around salmon farms.

3.1 Term of reference (a): comment on the report of a Workshop to be organised jointly by the Xunta de Galicia and the Instituto Espanol de Oceanografia, Spain in 2003 on stock enhancement in the Galician rias

Broadly speaking, stock enhancement is often initiated when a fishery reduces a stock to the point where the commercial viability of a sustained fishery is in question (commonly a result of limited fishing opportunities in a managed fishery). The reduced supply of product to the marketplace is usually accompanied by elevated value per kg of fish. Two ways to meet the market demand are enhancement and/or aquaculture.

These two activities can have similar effects on endemic conspecific populations. Both strategies, when done without adequate safeguards, can result in a further reduction of the abundance of the endemic population and, in the case of very low numbers of the wild population, the potential loss of adaptive traits from the genome. Two key factors promoting these potential effects are the proportion of cultured/enhanced fish entering the breeding population of the endemic stock and the degree to which there are differences between the endemic and cultured/enhanced individuals.

The numbers issue is usually a more proximate concern for enhancement activities where the intent is to introduce many new individuals to the wild population in order to gain a rapid increase in return to the capture fishery. Commercial culture activities in contrast demand as few fish as possible escape to join the wild population. Every fish that escapes represents reduced profits critical to a successful culture business. Over time, if the amount of fish cultured grows, the number of escapes from the culture activities may rise and contribute a significant component to the wild population, but in the early stages of development, the relative number is usually small. Similarly, the effects of the introduced fish on the local predator-prey relationships and on local carrying capacity are a more proximate concern for enhancement activities.

To the extent that numbers alone are responsible for some of the effects on the wild population, studies of the effects of enhancement activities provide valuable information on the possible risks associated with escaped cultured organisms. Thus, the importance of monitoring the effects of enhancement not only addresses the need for information to improve the effectiveness of stock conservation and wild fishery enhancement, but can also supply valuable information for assessing the level of risk associated with potential culture activities.

Initial culture activities are commonly undertaken with individuals harvested from wild stocks, thus initially there is little genetic difference between wild and cultured stocks. However, as the amount of cultured product entering the market place grows, economic pressures associated with a competitive market place require commercial aquaculturists to select their brood stock for a genome that performs best under culture conditions. This will cause rapid differentiation between the wild and cultured stock. This can increase the impact of the escaped fish on the wild population out of proportion to that which might be expected by the relative number of cultured fish in the wild population alone.

In contrast, the pressures to differentiate the genome of wild and enhancement fish are less immediate. For commercial fisheries, enhancement sometimes focuses on producing larger healthier fish (relative to the wild population) to improve survival and increase returns to the fishery. It is important, however, to recognize that even where no selection programme exists, if enhancement continues over many generations, some differentiation between wild and enhanced fish is inevitable. The repeated removal of individuals from the pressures of natural selection for that potion of their life during which their survival is enhanced will ultimately affect the frequency of alleles in the genome. The time required for this to happen is not known and may be expected to vary between species and environments.

Because of the limited number of studies in the area, behavioural differences are not often addressed in discussion of the effects of mixing of wild and cultured/enhanced fishes. Experience with salmonids has demonstrated that learned behaviours such as predator avoidance and diet selection often differ in wild and cultured/enhanced stocks. Too few studies of the ecological significance of the effect of these behaviours have been conducted to properly evaluate the consequences for the wild population. There is reason, however, to believe that learned behaviour may be important in defining the fidelity of individuals to spawning areas. This fidelity is critical to maintaining the differences between stocks. It is easy to imagine that disruption this aspect of spawning behaviour could have a significant impact on the ability of a species to adapt to local environments.

Our Spanish colleagues have submitted a report on the workshop on stock enhancement in the Galician rias. The report is included as Annex 3. Many of the features discussed above are evident in the report. The report emphasises the importance of sustained monitoring in understanding the effects of enhancement and modifying activities to create the best outcome possible. The report also points out that enhancement is not solely an activity and responsibility of the government. The fishing sector is also a participant and should be involved in the design and implementation of enhancement activities.

3.2 Term of reference (b): update developments in the implementation of the Water Framework Directive

3.2.1 Summary of update on the implementation of the Water Framework Directive, and implications for aquaculture activities

The implementation of the Water Framework Directive (2000/60/EC) has progressed with the publication of all the guidance documents and the restructuring of the common implementation process from individual strategy drafting groups to a central EU-wide group (Ecological Status 2A working group) whose remit is to advise on the implementation and execution of intercalibration, classification, monitoring and eutrophication for all water body categories. The designation of water bodies (Typology process) has been completed in many member states as has the development of a list of EU-wide water body types (Eurotypes) for intercalibration purposes. As most countries have defined their water bodies and these tend to be large, on the scale of kilometres to low tens of kilometres, it is likely that the majority of mariculture activities will be considered as one of the pressures acting on the overall quality of the water body and not as a separate water body.

However, uncertainty still remains around some of the implications of monitoring for the Water Framework Directive and how the impact of aquaculture activities in a water body will be regarded. A number of questions present themselves:

- How will temporal and spatial averaging of ecological quality of water bodies be dealt with?

- Will the lowest quality assessment direct the classification of the water body or will it be carried out by averaging out the quality at a number of locations within a water body?
- From a temporal perspective, will measurements taken during periods of disturbance (e.g., elevated chemical use to treat sea-lice) not introduce a certain amount of sampling bias?
- How will large-scale aquaculture activities (e.g., bottom culture of mussels), which may constitute large proportion of the seabed in a water body, be dealt with in the context of the directive?
- What reference conditions will be utilised to classify these large-scale aquaculture areas, especially if the activity was originally carried out on habitat different from that created by the activity?
- What programmes might be introduced to improve ecological quality with a water body as a consequence of an aquaculture activity?

Many of these questions are elaborated upon in Annex 4 of this report. It is hoped that these issues can be clarified as member states continue to develop the classification tools and refine the monitoring programmes.

3.3 Term of reference (b): update developments in the implementation of, and activities arising from the European Commission policy on sustainable aquaculture

In September 2002 the European Commission published its strategy for the sustainable development of European aquaculture (http://europa.eu.int/eur-lex/en/com/cnc/2002/com2002_0511en01.pdf). The overarching aims of the strategy are the creation of long term employment, ensuring the availability of high quality, safe and healthy products while at the same time ensuring high animal health and welfare as well as high environmental standards. The document sets out those elements of the strategy that should be taken forward at European Commission level and those that should be taken forward at Member State level.

The review focused on those elements of the strategy currently being taken forward at Commission level including harmonisation of standards for organic production, rules concerning introductions, transfers and containment of aquatic organisms in aquaculture, updating and revising the legislation in relation to animal health and the re-focusing of priorities for public aid through the Financial Instrument for Fisheries Guidance (FIFG).

Currently there are no internationally binding organic aquaculture regulations. Council Regulation (EEC) 2092/91sets up a framework of Community rules on production, labelling and inspection for organic farming. In the interests of producers and consumers, the Commission wants to create specific common definitions and norms for organic aquaculture, and include norms for organic aquaculture in the Regulation. Consequently, an initial meeting of a Working Group on organic aquaculture was held in Toulouse in November 2003, and a further meeting was held in February 2004. Certification organisations met with other interest groups, and compared details of the conditions in the various certification schemes available to producers. The WG was able to establish that there were considerable differences among the various schemes, and meetings are continuing to investigate routes towards harmonisation.

There are currently no comprehensive rules at EU level regarding introductions, transfers and containment of aquatic organisms in aquaculture. In its Strategy document, the European Commission announced its intention to propose management rules for introductions, transfers and containment in aquaculture. These rules would be consistent with the provisions of the ICES Code of Practice on the Introductions and Transfer of Marine Organisms. As an initial step towards addressing this undertaking, DG Fisheries held a consultation meeting in Brussels on 2 December 2003 and has set up a Working Group. This WG has started to review the current legislation on the introduction of alien species, and also on containment of fish/shellfish at aquaculture facilities. It is not yet clear whether DG Fisheries will subsequently handle the issues of alien species independently of those of containment. The timetable proposed was for drafting of proposals, further consultations and adoption by the Commission during the first semester of 2004 followed by consideration by Council and European Parliament during the second semester of 2004.

The Strategy states that "there is a continuous need for the Commission to regularly review, update and simplify the animal health Community legislation for aquatic animals and products with regard to ever-changing developments, particularly in the diversity of aquaculture production and in international experience and scientific knowledge". In order to develop and progress this element of the Strategy, DG SANCO established an Expert Group whose task was to lay down what could be considered as the scientific basis of new legislation on fish diseases. In addition, several sub-groups were set up to present proposals on specific subjects such as disease control, imports, etc. Based on the proposals of the expert group, a proposal for a new EU Directive has been drafted and the Commission is currently seeking comments from Member States. It is envisaged that the new legislative will be in place in 2005 and that the existing legislation, e.g., Directive 91/67; 93/53: 95/70 will be repealed.

The Commission proposes that the intervention by public authorities in favour of aquaculture be re-directed towards favouring modernisation of the existing farms and diversification, rather than increasing production capacity for species where the market is close to saturation. Action should be taken on measures such as training, monitoring, research and development, and clean farming technologies. The improvement of traditional aquaculture activities such as mollusc farming, that are important in maintaining the social and environmental tissue of specific areas, should be encouraged. A proposal has been put forward for a Council Regulation to amend the existing detailed rules and arrangements regarding Community structural assistance in the fisheries sector. In relation to aquaculture, proposals include funding of data collection and assessment as part of the EIA process, funding for participation in eco-management and audit schemes, compensation for shellfish farmers where harvesting of shellfish is prohibited for six months or more due to algal toxins and funding of small-scale, applied research initiatives not exceeding Eur 150,000 over three years.

The full text of the update may be found in Annex 5.

3.4 Term of reference (c): prepare for possible publication a report on the "state of knowledge" of the potential impacts of escaped aquaculture marine (non-salmonid) finfish species on local native wild stocks (e.g., sea bass, sea bream, cod, turbot, halibut)

Work has been initiated on five documents dealing with the potential impacts of escaped aquaculture marine non-salmonid finfish species. Documents pertaining to four species (cod, halibut, sea bass, and sea bream) and an overview document now exist as early drafts (see Annex 6). A document on a fifth species, turbot, will be drafted in the intersessional period. In order to create an analysis that is transparent and clearly separates supposition from experience, the analyses follow the format of risk analysis.

Discussions have been initiated to publish these documents and the results of the analysis for a number of shellfish species (see recommendations for next year's meeting) in a single volume.

3.5 Term of reference (d): discuss risk assessment methods in relation to mariculture in a joint session with GESAMP WG 31

Since the 2003 meeting of WGEIM, GESAMP had started an initiative regarding risk assessment of aquaculture developments. The project was being coordinated by Uwe Barg at FAO Rome through GESAMP Working Group 31 on Environmental Impacts of Coastal Aquaculture, with the title "Environmental risk assessment and communication in coastal aquaculture". FAO had recognised that this topic had close links to subjects discussed at previous meetings of WGEIM. Therefore, it had been agreed that ICES would be linked to the project through common membership of the GESAMP Steering Group and ICES WGEIM.

The Steering Group for the GESAMP project included Edward Black, Harald Rosenthal, and Ian Davies and had met for three days in Rome in December 2003, under the chairmanship of Harald Rosenthal. The main purpose of the meeting was to orient the group to the task, and specifically to consider a background discussion paper (Hambrey, J. and T. Southall, 2002. Environmental risk assessment and communication in coastal aquaculture: A background and discussion paper for GESAMP Working Group 31 on Environmental Impacts of Coastal Aquaculture. Unpublished working paper. FAO Technical Secretariat for GESAMP) that had been prepared for GESAMP. This document included a broad review of risk assessment procedures, and particularly emphasised the need to take account of the uncertainties inherent in many of the underlying data and data interpretations.

The background paper noted that aquaculture, and in particular coastal aquaculture, continues to grow rapidly throughout the world. Actual and possible environmental impacts from coastal aquaculture include nutrient enrichment; chemical pollution; habitat loss and change; impacts on wild fish and shellfish populations; and upstream effects related to the production of fishmeal used in farmed fish feeds. A recent GESAMP report (GESAMP, 2001) includes adherence to the precautionary approach amongst a set of guiding principles for improved planning and management of coastal aquaculture development. Application of the precautionary approach implies more thorough assessments of risks related to any new or expanding activity.

Environmental risk assessment (ERA) is used widely to address the risks associated with industrial processes, and may serve as a useful tool to support an informed precautionary approach for coastal aquaculture development. It is a formal process consisting of four main steps: (i) hazard identification, (ii) hazard characterisation, (iii) exposure assessment, and (iv) risk characterisation.

Uncertainty is a particularly important issue for coastal aquaculture development. While some of the impacts (such as deposition of organic matter) can be predicted with reasonable confidence limits, impacts on the wider coastal environment, and in particular on wild fish populations, are highly uncertain, and this uncertainty is unlikely to be reduced significantly even with detailed long-term research. It is crucial that the nature and degree of uncertainty associated with the impacts is clearly characterised in any ERA and effectively communicated to decision-makers and interested stakeholders. There is little guidance available as to how this can best be achieved.

WGEIM noted many parallels between the above concepts and the ideas and procedures described in a background risk analysis paper presented to WGEIM 2004 under agenda item c), and subsequently applied by WGEIM to a range of fish species. As an initial contribution to the GESAMP process, WGEIM recommends that the papers presented and developed under agenda item c) should be forwarded to GESAMP as input to their project.

Reference

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). 2001. Planning and management for sustainable coastal aquaculture development. Rep.Stud.GESAMP, (68): 90p.

3.6 Term of reference (e): conduct an analysis of the literature and research on the current bath treatments and in-feed additives (treatments) used to treat salmon for sea-lice, and produce a synthesis (state of knowledge) on their fate in the near and far field environment and their effects on non-target organisms (e.g., crustaceans and invertebrates)

3.6.1 Summary of risk assessment of sea lice therapeutants

Sea lice, *Lepeophtheirus salmonis* and *Caligus elongates*, are ectoparasites of many species of fish and are a serious problem for salmon aquaculture industries. Sea lice are natural parasites of wild Atlantic salmon, and infestations have occurred routinely in European aquaculture and Atlantic Canada. Sea lice reproduce year round and the aim of a successful lice control strategy must be to pre-empt an infestation cycle becoming established on a farm by exerting a reliable control on juvenile and preadult stages, thus preventing the appearance of gravid females. Effective mitigation, management and control of sea lice infestations requires good husbandry, linked to the use of natural predators such as wrasse and effective anti-parasitic chemicals. Chemicals used in the treatment of sea lice infestations are normally subsequently released to the aquatic environment and may have impact on other aquatic organisms and their habitat. The chemical therapeutants available to control sea lice were reviewed and their risks to the aquatic ecosystem were assessed.

The review was limited to those chemicals that are currently authorized for use by the salmon aquaculture industry in Europe and North America. These are the organophosphate, azamethiphos, the pyrethroids, cypermethrin and deltamethrin, and hydrogen peroxide that are administered by bath techniques, the avermectin, emamectin benzoate and chitin synthesis inhibitors, telflubenzuron and diflubenzuron that are administered as additives in medicated feed. The number of chemicals authorized for use is limited because of the high cost of development and licensing for a small market relative to other markets for pesticides and medicinals.

The ecological risk of the sea lice therapeutants were assessed by review of the information on their distribution and persistence in the marine environment, their biological effects observed on marine organisms in laboratory and field studies, and the likelihood that these biological effects would occur during the use of these chemicals to treat sea lice infestations of cultured salmon.

The organophosphate, azamethiphos, was found to be a moderate risk to individuals of sensitive species but a low risk to populations. However, azamethiphos was not considered the treatment of choice because of the development of resistance to organohosphates by sea lice. Evidence suggested that the pyrethroids, cypermethrin and deltamethrin, have a risk of adverse effects to individuals of sensitive species but there is insufficient knowledge to extrapolate to populations. There is sufficient evidence of the development of resistance to advise against routine use of pyrethroids as the only means of control. Pyrethroids are not authorized for use in North America for treatment of sea lice infestations. The limited database on the in-feed medicines, the avermectins and the chintin synthesis inhibitors indicate that they are of relatively low risk to the marine ecosystem.

The nature and severity of the environmental risks presented by the use of the various chemicals available to control sea lice in farmed salmon varied considerably between treatment compounds. Current regulatory practices, particularly those leading to approvals/authorizations (including the need for veterinary prescription) for the use of products for sea lice control, include elements of assessment of the risk to the environment. This is the primary process by which the environmental risk is managed, that is through the decision on whether or not grant approval/authorization, and under what conditions.

However, there are considerable differences between the environmental characteristics of fish farm sites and their ability to accept discharges of sea lice treatments without giving rise to unacceptable environmental impacts, for example, differences in tidal currents and other hydrographic factors to dilute and/or disperse chemicals. Such site-specific risks can be managed through the application of appropriate Environmental Quality Standards for the chemicals concerned, and site-specific assessment of the maximum acceptable rate of use of the treatments.

The full body of the report may be found in Annex 7 of this document. This will be published as a chapter in The Handbook of Environmental Chemistry (Editor in Chief: O. Hutzinger), Volume 5 Water Pollution and Environmental Effects of Marine Finfish Aquaculture, Volume editor: Barry Hargrave, Springer-Verlag, Berlin. Deadline for final manuscript for printing is November 2004.

4 Other business

Succession of the Chair was discussed and the participants unanimously recommended the group put forward Mr Francis O'Beirn (Ireland) to be the next Chair.

4.1 Adoption of the report and recommendation for next meeting

The WGEIM approved the draft report and the recommendations resulting from the meeting, subject to final editorial work by the Chair. The Working Group recommended that its next meeting be held in Ottawa, Canada, from 11 April to 15 April 2005.

4.2 Closing of the meeting

This meeting in Galway, Ireland, was formally closed 9 April 2004.

5 Annexes

Annex 1 Agenda

Monday, 5 April

09:30	Chair's Welcome Participants to the meeting Self Introduction of Participants
10:00	Formal Welcome by Host Chair's response House keeping and support arrangements
10:15	Review of and Comments on TOR
10:30	Tabling of inter-sessional work and comments.
11:00	*Health Break*
11:30	Plenary Session Identification of subgroups Designation of Rapporteurs
12:00	***LUNCH***
13:00	Break out to drafting groups
15:00	*Health Break*
15:30	Return to Drafting Groups
17:00	Plenary Session Report on progress

Tuesday, 6 April

09:00	Plenary Session – Adjustment to work distribution and identification of additional resources that may be required.
09:15	Drafting groups reconvene
10:00	*Health Break*
10:30	Drafting groups reconvene
12:00	*LUNCH*
13:00	Drafting groups reconvene
15:00	*Health Break*
15:30	Drafting groups reconvene
16:00	Presenation by Fiona Geoghegan (Marine Institute, Fish Health Unit, Dublin) on Proposed EU COUNCIL DIRECTIVE on health condition for the placing on the market and import of fish and their products and on minimum measures for the prevention, control and eradication of certain fish diseases.
16:45	Days progress distributed and read
17:00	Presentation of Progress and discussion
18:00	End of session

Wednesday, 7 April

09:00	Plenary session – As needed to discuss nocturnal discussions and thoughts.
09:15	Drafting groups reconvene
10:00	*Health Break*
10:30	Drafting groups reconvene
12:00	*LUNCH*
13:00	Field Trip to farm sites

Thursday, 8 April

09:00	Progress distributed and read
09:15	Presentation of Progress and discussion
10:00	*Health Break*
10:30	Drafting groups reconvene
12:00	*LUNCH*
13:00	Drafting groups reconvene
15:00	*Health Break*
15:30	Drafting groups reconvene
16:45	Days progress distributed and read
17:00	Presentation of Progress and discussion
18:00	End of session
19:30	Dinner Hosted by the Marine Institute

Friday, 9 April

09:00	Rapporteurs pass draft recommendations and 2005 ToR proposals to the chair Drafting of final document - groups reconvene
10:00	*Health Break*
10:30	Drafting groups reconvene
11:00	Discussion of proposed recommendations and 2005 Tor
12:00	*LUNCH*
13:00	Discussions of draft final document and proposals for 2005
15:00	*Health Break*
15:30	Final modifications of draft
17:00	End of 2004 meeting

Annex 2 List of participants

Members of the Working Group on Environmental Interactions of Mariculture (WGEIM)

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Annex 3 Comments on the Workshop on Stock Enhancement in the Galician Rias

Workshop on Stock Enhancement in the Galician Rías
IGAFA, Illa de Arousa - Galicia, Spain – 6–7 May, 2003

Organised by:
Xunta de Galicia (Galicia Regional Government)
and
Instituto Español de Oceanografía. Centro Oceanográfico de Vigo

1. General conditions for a stock enhancement programme

The following aspects must be taken into consideration when implementing a stock enhancement programme:

1.1 Broodstock characteristics

- Broodstock must be autochthonous, preferably captured in the area where they will be released thereafter. It is important that the quality of the individuals is certified by a centre that would also perform the necessary sanitary and genetic controls.
- Broodstock must be composed of a sufficient number of individuals so as to ensure that genetic diversity is maintained.

1.2 Release size

- A minimum size for each species and the capability to incorporate them into fisheries must be established in order to obtain significative recaptures.
- It is advisable to take into account the following: recapture rates obtained with that specific size, growing potential, production costs and patterns of migratory behaviour among the designated species.

1.3 Adaptation to environment prior to release

- This is an important aspect, since it is during the first weeks immediately after release that the highest mortality rates are recorded among the released individuals. Therefore, it is advisable to carry out adaptation experiments on these species, either in laboratories or in the natural environment, since they register diverse behaviours (e.g., flatfishes require sandy bottoms and must become familiarised with the presence of predators; lobsters need habitats offering them a variety of shelters,…)

1.4 Characteristics of the habitat

- A complete study of the potential release areas must be carried out, taking into account, among others, the following factors: predator density, competitors, food availability, as well as the reaction faced by the ecosystem with the release of new individuals.

1.5 Tagging

- For an effective monitoring of the recaptures, a tagging exercise prior to release must be performed on all the individuals.
- When using external tags, it is of great concern the high rate of tags lost, becoming entangled with seaweed, or swallowed by other individuals.
- Magnetic tagging has proved to be both useful and effective in detecting marks. In order to implement this method, fish markets should be equipped accordingly.

1.6 Monitoring released individuals

- It is fundamental the presence of a monitoring team in the area of release carrying out studies on growth, mortality, migrations, depth distribution, sanitary controls and recapture rates. In order to perform this, research and/or commercial surveys must be carried out so as to establish a relationship between recaptured individuals both tagged and wild.

1.7 Legal measures for fisheries protection and/or regulation

- It is necessary to point out that a stock enhancement programme will lose effectiveness unless strict accompanying measures are adopted, related to fisheries legal protection and regulation. Commitment on the side of the extractive sector in this kind of activities is also essential.

1.8 Socio-economic study of the stocking programme

- Socio-economic studies must be carried out, following the example of other countries that are already implementing stocking programmes, in order to assess the cost-benefit analysis involved in this type of projects. The indicators traditionally used relate the economic benefice provided to fisheries to production and release costs.

2 Assessment of species type in Galicia

Once identified the general characteristics that a stocking programme must fulfil, an assessment of three target species in Galicia (lobster, turbot and sea bream) follows, on the basis of previous culture and enhancement experiences.

2.1 Crustaceans

2.1.1 Target species:

- From the point of view of the feasibility of the released individuals, spiny spider crab (*Maja squinado*) is a species that should be taken into consideration, since it presents high growth rates, a not too demanding behaviour regarding the required substratum or shelters and both lower number of competitors and intensity of competitiveness than European lobster (*Homarus gammarus*). Nevertheless, production techniques of hatchery-reared larval individuals are not ready yet. Besides, it should be considered whether, according to the present status of the natural stocks, it is advisable the implementation of a stocking programme or the management of the catches is preferred instead.
- As for lobster, weight data belonging to individuals sold in Galician fish markets for the last 10 years show a steep decrease and very low annual selling figures. Although data of catch per unit of effort would be more conclusive, according to the present situation of the commercial fishery, the need for an enhancement of the natural populations through stocking with hatchery-reared juveniles should be considered. Besides, hatchery production techniques are sufficiently developed.

2.1.2 Areas of release:

As an initial measure, areas with the following characteristics should be preferred:

- Those where good rates of catches are, or used to be, obtained
- Areas with hydrodynamic retention in order to avoid the dispersion of released individuals
- Those whose substratas offer shelter and food
- Selected spots should be studied as for their number of predators, competitors and food availability for a minimum period of one year.

2.1.3 Broodstock:

- Local breeding females are required, keeping their health under control and rejecting ill or injured individuals
- The minimum effective number of breeders must be determined in order to ensure the conservation of genetic variability
- Females preferably with a carapace length between 28 and 35 cm should be selected so as to ensure spawning quality
- Female maintenance conditions in the facilities must be cared for, in order to reduce losses of valuable individuals extracted from the natural environment

- It is desirable to carry out a study on natural genetic variability so as to determine the feasibility of employing captured breeders in spots away from the potential release areas, without interfering with natural genetic diversity.

2.1.4 Larval rearing

Intensive production techniques of juveniles are well developed, however survival results are still variable and unpredictable. Therefore:

- It is advisable to carry on with larval rearing studies in order to achieve better results, more predictable, and less variable, which could mean important savings in production costs.

2.1.5 Quality of the juveniles produced

In order to maximise the results, the aim must be set to produce competitive juveniles, adapted to the environment in which they will be released, with a normal morphology with regard to pigmentation and claw dimension. Therefore:

- It would be appropriate to stimulate natural escape behaviour in the presence of predators, search and use of shelters, interaction with other individuals,… prior to their release.

2.1.6 Release methods

- A total length of 6–7 cm is considered to be the minimum release size to assure positive results
- The best season for spawning will fundamentally depend on the predator activity that has been detected in the studies mentioned in point 2. It must also be taken into account the development level of annual recruitment in the environment.
- Individuals must be released in shallow waters (short time in the water column subject to predation) having been previously acclimated to the phisicochemical conditions of the environment.
- Predator activity must be controlled in the first two hours after the first releases are made, in order to determine the losses.
- It is advisable not to carry out too many releases, nor to release a great number of individuals in the same spot so as to avoid that natural recruitment is negatively affected by stocking, or that the carrying capacity of the environment is exceeded.

2.1.7Control of recaptures:

- It is necessary to perform a tagging exercise on all released individuals
- At the moment, the best tagging method for lobster is the use of coded-wire microtags containing a binary code, which implies the setting out of detectors, both in those spots designed for monitoring the catches and where they are first sold.
- In order to achieve this, it is necessary to obtain collaboration from the extractive sector so as to ensure the control of the catches.

2.1.8 All these measures should be accompanied by strict regulations for the management and protection of resources if pre-existing stocks are to be recovered.

2.2 Flatfishes

2.2.1 Target species

Turbot (*Psetta maxima*) is chosen as a species type, given its commercial interest and the low annual catch rates observed (around 100 tons). Besides, another positive element presented by this species is the fact that culture techniques, biology and is bathymetric distribution in the rias are well-known.

2.2.2 Broodstock

Broodstock must be autoctonous and captured in the wild. In order to guarantee their genetic variability, the availability of a minimum effective number of breeders must be assured. It is important the existence of a centre that certifies the quality of the breeders, carrying out the necessary sanitary and genetic controls on the stock.

The artificial fertilisation process must be carried out on an individual basis (one male and one female per fertilisation) with tagged individuals providing a well-identified history. They should be included in genetic selection programmes so as to enhance broodstock.

2.2.3 Larval rearing

Semi-intensive and intensive larval production techniques are already developed; nevertheless, further research must be conducted on larval culture in this species, with the aim of enhancing the survival rates that are currently obtained.

2.2.4 Larval quality

In order to avoid genotypical alterations, it is important not to release individuals presenting malformations or morphologic anomalies (lack of the operculum, albinism, etc.)

Pathologic and genetic studies on the wild population must be conducted prior to release, aimed at assuring the non-interference of the released individuals in the essential characteristics of the wild population.

2.2.5 Release areas and seasons

In reference to the ecological study of the release area, it is advisable to release 0-group individuals, between 6 and 10 cm, on beaches with a sandy substratum and great wave exposure, located on the outer areas of the rias. Once these areas are spotted, a study on the carrying capacity and the presence of predators and competitors must be conducted.

The most appropriate season for release is from September to December, when individuals from the same wild population have reached the same size.

2.2.6 Release methods

With regard to the adequate size for release, taking into account previous experiences, related to growing capacity, movement, recapture rates and cost, the most adequate size to perform stock enhancement exercises is between 6 and 10 cm, but it is advisable to carry out previous experiments with these sizes before conducting any massive releases, since there are no records in Galicia with such sizes.

As for the conditioning prior to release, one of the main problems observed in previous experiences of turbot stocking is the high mortality rate of this species immediately after being released, so it is considered as a priority to reduce such mortality subjecting individuals to a period of previous adaptation, taking into account the following:

- Identification and defense mechanisms in the presence of predators;
- Development of colour adaptation and burying capacities.

Thus, it is suggested to carry out experiments with individuals in both wild and reared environments, in closed areas, with the aim at improving their capacity of adaptation to the natural environment.

Despite being the tagging method most widely used, “T”-bar anchor tags present high rates of loss, which are necessary to reduce in order to increase recapture rates. Thus, it is recommended to essay other tags, of magnetic type, aiming at improving the final effectiveness of the stock enhancement programme.

Regarding transport and release techniques of individuals, it is essential to carry out previous studies to determine which is the most adequate and less stressing method for this species.

2.2.7 Control of recaptures

Every stock enhancement programme calls for a continuous release process, for five years at least, so it can have any effects on the fishery. After this period, its possible positive effect on fishery recruitment can be assessed.

As an informative reference, it should be pointed out that in previous experiences with plaice carried out in Japan, between 100,000 and 400,000 individuals were released on an annual basis, for a five-year period, subsequently observing a possitive effect on the commercial catches.

It is essential the existence of a monitoring team in the natural environment conducting studies on growth, mortality, migrations, depth distribution, sanitary controls and recapture rates. Thus, it is necessary to carry out research and/or commercial surveys so as to establish a relationship between the characteristics of tagged recaptured individuals and those belonging to autochthonous wild stocks.

Finally, with regard to the impact on fisheries, it is fundamental to involve of the extractive sector in the stocking programme, to carry out socio-economic studies of the programme and to analyse its contribution to the fishery’s total catches. A final analysis of the stocking programme, regarding its costs in relation with the benefits provided to the fishery, will finally allow for the assessment on the effectiveness of the selected stock enhancement method.

2.3 Roundfishes

2.3.1 Target species

The following species were considered, red seabream (*Pagellus bogaraveo*), pollack (*Pollachius pollachius*), white seabream (*Diplodus sargus*) and European seabass (*Dicentrarchus labrax*), as potential candidates for stocking, according to their commercial importance, the state of development of their culture and the state of natural stocks in Galicia.

Among the main requirements to carry out a stocking programme, have been analysed those shown in the tables below, where an evaluation of these species for future stocking programmes is made.

According to the above-mentioned criteria, Red seabream (*Pagellus bogaraveo*) was selected as a target species for the first phase of a stocking programme in Galicia.

2.3.2 Broodstock

Broodstock must be autochthonous, captured in the wild environment. A minimum effective number of broodstock must be assured in order to guarantee genetic variability.

At present, red seabream wild broodstock are available with information on their genetic variability and sanitary controlled. Spontaneous spawning is obtained between January and May.

It is essential the existence of a centre certifying broodstock quality and conducting sanitary and genetic controls.

2.3.3 Larval rearing

Techniques for the intensive production of juveniles are already developed; however, it is advisable to carry out further studies on larval rearing enhancement to obtain an improvement on survival and a greater predictability of the results.

2.3.4Quality of the juveniles produced

It is important that none of the released individuals present morphologic or pathologic anomalies (lack of operculum, lordosis, …)

To optimise the results it is important to release juveniles adapted to the environment and competitive within the natural stock.

2.3.5 Release areas and seasons

With regard to the release area, even though no previous records of red seabream larvae releases can be found, we have received information from fishermen about certain nursery areas, where it would be adequate to carry out the first releases.

The best stocking season observed for other sparids was Spring–Summer, thus this seems the most advisable season to us, although we do not hold sufficient information on red seabream. It would be desirable to perform adaptation tests to the natural environment prior to release.

2.3.6 Release methods

According to previous stocking experiences with other sparids in the Gulf of Cadiz, an initial release size could be determined between 30 and 100 g. In any case, it is regarded as essential to conduct experiences aimed at determining the most suitable release size.

Juveniles transport and release techniques are well-known, since they have been carried out successfully for other purposes.

2.3.7 Control of recaptures

In order to perform these stocking programmes, it is necessary to complete tagging systems for this species. “T”-tags are habitual in stocking programmes, successfully employed on wild stocks of this same species.

Once the stocking programme has been accomplished, growth in the natural environment, recapture rates, movements, length of time in which recaptures appear, relationship between wild and released recaptures in experimental fishings, and cost/benefit analysis must be subject to control by a monitoring team.

With regard to the other three proposed species, pollack, white seabream and European seabass, the following evaluation tables are enclosed for reference in future stocking programmes.

	Broodstock			**Culture**				**Stocking**			
Species	Existing in Galicia	Availability	Genetic and pathologic studies	Breeding control	Spontaneous spawning	Larval rearing	Growth	Release season	Tagging techniques	Transport techniques	Release techniques
Red seabream	YES	YES	DONE	NO	YES	YES	YES	Spring–Summer	Microchip "T"-tags	YES	YES (cages)
Pollack	NO	YES	Non-existing	NO	YES	YES	YES	Unknown	NO	YES	NO
White seabream	NO	YES	Non-existing	NO	YES	YES	YES	Unknown	"T"-tags	YES	YES
European seabass	NO	YES	Non-existing	YES	YES	YES	YES	Unknown	NO	YES	NO

	Biology and Ecology		
Species	**Migrations**	**Knowledge of hatchery habitat**	**Ichthyoplankton surveys**
Red seabream	Migratory	YES	¿?
Pollack	Migratory	NO	¿?
White seabream	Non-migratory	YES	¿?
European seabass	Migratory	YES	¿?

3 Workshop conclusions

1) Stocking programmes must be addressed within a long-term project, and they need to have a purpose of annual continuity.
2) Few species fulfil the necessary requirements to be included in a stocking programme. The most important are:
   - culture techniques well-known
   - commercial catches showing a significant decline in evolution
   - being of high commercial interest
   - released juvenile must remain within the release area
3) The following aspects must be taken into account in a stocking programme:
   - Breedstock characteristics
   - Release size
   - Knowledge of the potential habitats for release
   - Adaptation to environment prior to release
   - Carrying capacity
   - Tagging systems
   - Monitoring of released individuals
   - Area and resource protection measures after the release
   - Socio-economic study on the programme. Cost-benefit analysis
4) Stocking possibilites concerning three species have been assessed: two fishes (turbot and red seabream) and one crustacean (lobster), taking into account previous release experiences and known culture techniques.
5) Multidisciplinary studies in collaboration with other teams and institutions in order to develop stocking programmes adequately.
6) Once the species suitable for stocking are identified, contacts with other world stocking centres should be established and meetings with experts working on these species could be held.
7) It is essential the involvement of the concerned fishing sector in the stocking design and its actions.

Annex 4 An update on the implementation of the Water Framework Directive, and implications for aquaculture activities

1 Introduction

WGEIM 2003 (Working Group on Environmental Interactions of Mariculture. ICES CM 2003/F:04, Section 5: Annex 4) presented a comprehensive introduction and overview of the Water Framework Directive (WFD). The overview detailed the goals of the directive (one of which is to achieve good ecological status for all surface waters by 2015), and a mechanism to implement the directive across the EU while detailing a series of deadlines within which various tasks must be implemented. In addition, the mechanisms for measuring ecological change (i.e., classification tools) and other monitoring tools (chemical, hydromorphology) were presented. WGEIM 2003 also discussed the ways in which processes under the WFD could interact with mariculture, and the constraints and opportunites that could follow from this. However, WGEIM 2003 also pointed out several areas of uncertainty which would be important in determining the degree to which future mariculture activities might be influenced by the WFD.

This section of the report provides an update on the implementation of the Water Framework Directive. It identifies some relevant deadlines and provides further commentary on the implications the directive may have on mariculture activities.

1.1 Deadlines

The implementation of the Water Framework Directive is driven by a number of deadlines set out in the directive. Some of the more relevant deadlines to marine waters are:

- To characterise river basin districts in terms of pressures, impacts and economics of water uses, including a register of protected areas lying within the river basin district, by 2004 (Article 5, Article 6, Annex II, Annex III);
- To carry out intercalibration of the surface water ecological quality status assessment systems by 2006 (Annex V)
- To make operational the monitoring networks by 2006 (Article 8)
- Based on sound monitoring and the analysis of the characteristics of the river basin, to identify by 2009 a programme of measures for achieving the environmental objectives of the Water Framework Directive cost-effectively (Article 11, Annex III);
- To make the measures of the programme operational by 2012 (Article 11)
- To implement the programmes of measures and achieve the environmental objectives by 2015 (Article 4)

The bodies responsible for each of the deadlines varies from Member State to Member State. These bodies are identified in the national regulations that were transposed in December 2003. Typically, the responsibility rests with the River Basin District projects (which are commonly overseen by local authorities) which are the administrative arm of the directive with support from a number of state and semi-state agencies.

1.2 Typology

The fundamental management unit established under the WFD is the water body. Water bodies are assigned to appropriate types, and therefore typology is the mechanism for defining and grouping water bodies of similar general nature under the WFD. The health of each water body will be classified by any one of the three potential monitoring mechanisms (surveillance, operational and investigative see WGEIM 2003 for descriptions). These management units also form the basic units upon which the Pressures and Impacts analysis (risk assessment) would be carried out. A typology has now been completed and agreed on an EU-wide basis, with a view to identifying common coastal and transitional water types that can be utilised in the intercalibration exercise. As an example of the use of typology, the output from the United Kingdom and Republic of Ireland (UK-ROI) typology exercise is described below.

1.2.1 A Summary of the Typology for Coastal & Transitional Waters of the UK and Ireland

The Directive requires Member States to differentiate relevant surface water bodies according to type using either “System A” or “System B” (see Annex II of the Directive). The UK-ROI decided to use System B in coastal and transitional waters. The UK-ROI has closely followed the Guidance document produced by the EU CIS Working Group 2.4 (COAST) in deriving its final typology. The Guidance document describes how both the obligatory and optional factors within System B could be used.

The obligatory factors for differentiation of types and water bodies in both coastal and transitional waters are latitude and longitude, salinity and tidal range. Latitude and longitude are accounted for by the location of the coastal areas of member states in one of a series of trans-national ecoregions (the UK and ROI lie within the Atlantic and North Sea ecoregions)., All UK and ROI coastal waters are euhaline, i.e., > 30, so no further discrimination using this factor was necessary in deciding the typology. Transitional waters (estuaries) have been defined as either polyhaline or mesohaline, or just predominately polyhaline. It was not considered necessary to subdivide UK-ROI transitional waters further. This is in line with the CIS 2.4 Guidance that Member States may aggregate descriptors within ranges if there is no biological difference. Types have been differentiated by tidal range using the agreed definitions in the CIS 2.4 Guidance, i.e., micro-tidal < 1m, meso-tidal 1–5 m, and macro-tidal > 5m.

Certain optional factors were also used in the typology of transitional waters. These include mixing characteristics, mean substratum composition and extent of intertidal area. In coastal waters, the biologically relevant optional factor was wave exposure and this was used in combination with the obligatory factors to define the final types.

The use of the approach described above has resulted in the identification of 12 coastal water body types and 6 transitional water body types for the UK-ROI. These water bodies are presented in the following figures (Figures A4.1–3) with descriptions of the types in the following tables (Tables A4.1–2).

Legend for Figures A4.1–3:

In order to differentiate types on the following maps the colour coding as shown below has been adopted.

CW1	Exposed, Macro-tidal
CW2	Exposed, Meso-tidal
CW3	Exposed, Micro-tidal
CW4	Moderately exposed, Macro-tidal
CW5	Moderately exposed, Meso-tidal
CW6	Moderately exposed, Micro-tidal
CW7	Sheltered, Macro-tidal
CW8	Sheltered, Meso-tidal
TW1	
TW2	
TW3	
TW4	

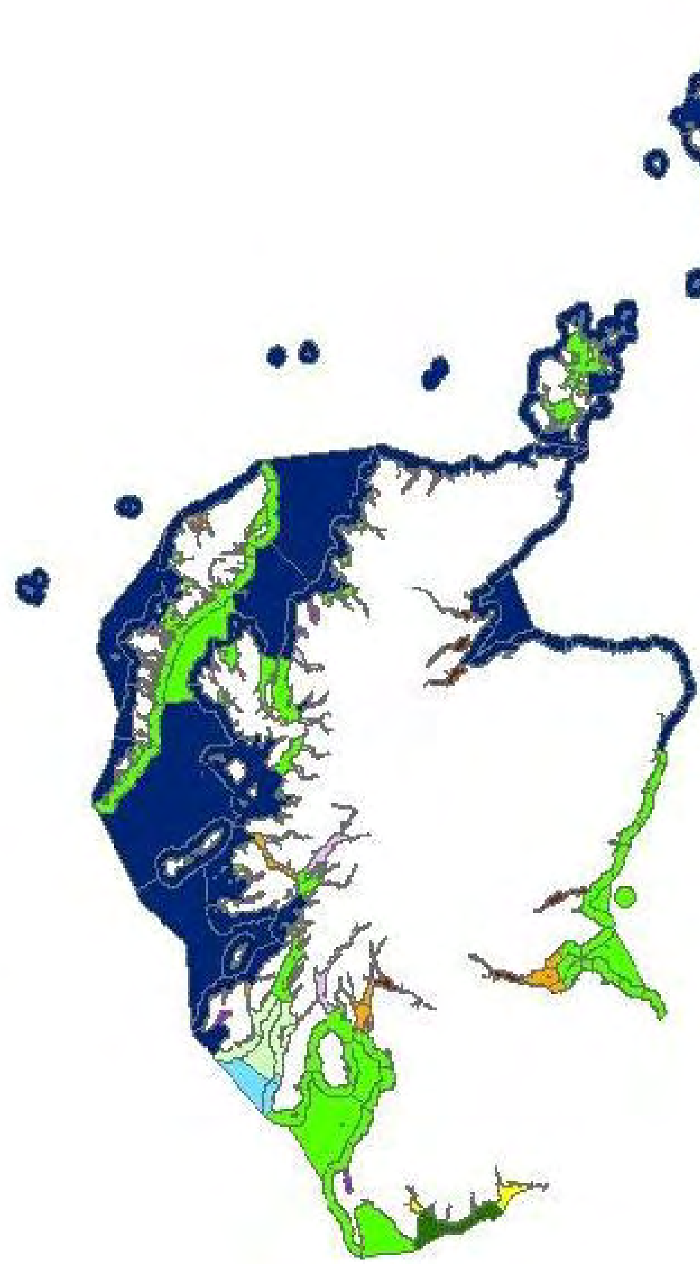


Figure A4.1. Coastal and transitional water bodies in Scotland.

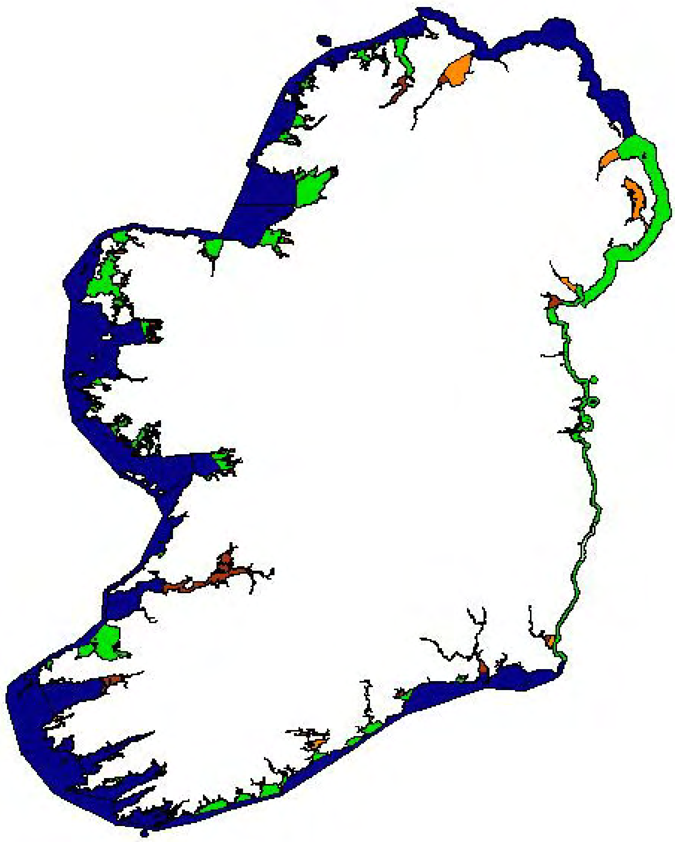
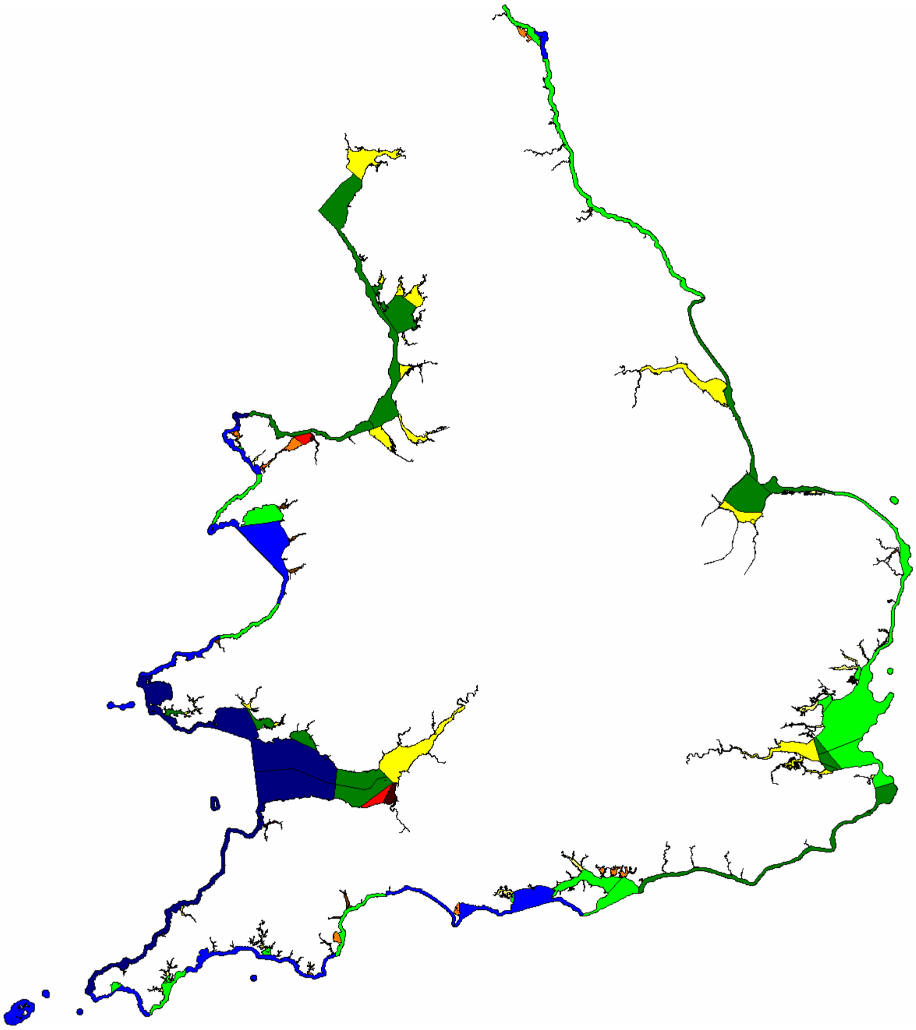


Figure A4.2. Coastal and transitional water bodies in Ireland.

Source (ROI): EPA, Copyright Government of Ireland.



FigureA4.3. Coastal and Transitional water bodies in England and Wales.

Table A4.1. Predominant typology characteristics of main transitional water types in UK-ROI.

Type	Name	Mixing Characteristics	Salinity	Mean Tidal Range	Exposure	Depth	Substratum	Example
TW1		Partly mixed/ stratified	Mesohaline/ polyhaline	Strongly macrotidal	Sheltered	Intertidal/ shallow sub-tidal	Sand and mud	
TW2		Partly mixed/ stratified	Mesohaline/ polyhaline	Strongly mesotidal	Sheltered	Intertidal/ shallow sub-tidal	Sand and mud	Tees Estuary Dart Estuary
TW3		Fully mixed	Polyhaline	Macrotidal	Sheltered	Extensive intertidal	Sand or mud	Mersey Estuary Humber Estuary
TW4		Fully mixed	Polyhaline	Mesotidal	Sheltered	Extensive intertidal	Sand or mud	
TW5	Transitional Sea Lochs		Polyhaline	Mesotidal	Sheltered			Gare Loch Loch Eil Loch Linnhe Loch Etive
TW6	Transitional lagoons	Partly mixed/ stratified	Oligohaline - polyhaline	N/A	Sheltered	Shallow	Mud	

Table A4.2. Predominant typology characteristics of main coastal water types in UK-ROI

Type	Name	Salinity	Mean Tidal Range	Exposure	Example
CW1		Euhaline	Macrotidal	Exposed	South Wales North coast Cornwall, Devon
CW2		Euhaline	Mesotidal	Exposed	North West Scotland West coast of Ireland Cardigan Bay
CW3		Euhaline	Microtidal	Exposed	North coast Northern Ireland Islay to Mull of Kintyre
CW4		Euhaline	Macrotidal	Moderately Exposed	North West England Kent and Sussex coast
CW5		Euhaline	Mesotidal	Moderately Exposed	Northumberland coast North Channel Scotland
CW6		Euhaline	Microtidal	Moderately Exposed	Sound of Jura
CW7		Euhaline	Macrotidal	Sheltered	Bridgwater Bay Outer Wash (Embayment)
CW8		Euhaline	Mesotidal	Sheltered	Firth of Forth Firth of Clyde Hampshire Harbours (Embayment)
CW9		Euhaline	Microtidal	Sheltered	
CW10	Coastal lagoon	Euhaline	N/A	Sheltered	
CW11	Sea Lochs (Shallow)	Euhaline	Mesotidal	Sheltered	Busta Voe Loch Ryan Loch Indaal Loch Skipport
CW12	Sea Lochs (Deep)	Euhaline	Mesotidal	Sheltered	Loch Long Loch Torridon Firth of Clyde Loch Fyne Loch Nevis

1.3 CIS and EcoStat

The Common Implementation Strategy (CIS) for the Water Framework Directive was agreed by the European Commission, Member States and Norway in May 2001. Although implementing the Directive remains the responsibility of individual Member States, a common strategy was necessary to:

- Develop a common understanding of approaches;
- Elaborate informal technical guidance including best practice examples;
- Share experiences and resources;
- Avoid duplication of efforts; and
- Limit the risk of bad application.

Experts from the member states and candidate countries, as well as stakeholders and non-governmental organisations were all involved in the CIS process. This common implementation process was unique to this directive.

The primary output of the CIS was the publication of numerous Guidance Documents, with a view to providing a harmonised approach to the implementation of the directive.

The CIS working groups have now formally completed their task and the joint implementation process will be assumed by single EU-wide working group entitled the Ecological Status 2A working group. The remit of the new EcoStat WG covers intercalibration, classification, monitoring and eutrophication for all water body categories. Drafting groups will be established within each of the working groups to deal with specific issues.

1.4 Characterisation

The Water Framework Directive utilises the river basin as the natural unit for water management. Each river basin within a Member State must be assigned to a River Basin District (RBD) and the Member State must arrange for co-ordination of administrative arrangements for water management in relation to each RBD lying within its territory. A River Basin District must include coastal/marine waters up to one nautical mile beyond the baseline from which territorial waters are measured. A river basin is the area of land from which all surface run-off flows through a sequence of streams, rivers and possibly lakes into the sea at a single river mouth, estuary or delta. A RBD is an area of land and sea made up of one or more neighbouring river basins together with their associated groundwater, transitional and coastal waters and identified as the main area for co-ordinated water management.

In Ireland (as an example), Local Authorities will have the primary role in promoting, establishing and implementing these river basin projects. For example, River Basin Districts in Ireland will be determined by the natural grouping of hydrometric areas into water resource regions already familiar to Local Authorities and other public bodies.

Article 5 of the WFD requires that Member States undertake an analysis of the characteristics of each River Basin District by December 2004. This characterisation must identify all surface and groundwater water bodies that are currently at risk of failing to meet specific WFD objectives. The output of the reports must include:

- Identification of Pressures

To deliver the characterisation of River Basin Districts requires the identification of human activities or pressures that have the potential, on their own or in conjunction with other activities, to jeopardise the achievement of the Directive's environmental objectives.

- Risk Assessment Process

River Basin characterisation is at the heart of a risk-based approach to environmental protection and enhancement that is integral to the Directive. Characterisation will not end in December 2004, it is an ongoing process for regulatory authorities and stakeholders, and will support the development of River Basin Management Plans.

- List of protected areas

The characterisation report must contain a list of all designated protected areas including, *inter alia*, Natura 2000 (Conservation) sites, shellfish growing waters and harvesting areas, bathing waters and drinking water sites.
The outputs will most likely be presented in a GIS format. The outputs from characterisation will input to the design of Programmes of Measures and help define surveillance and operational monitoring plans. This risk-based approach will ensure that resources are targeted at real environmental problems.

1.5 Intercalibration

Since the implementation process for the WFD began, countries have been working, largely independently, to develop classification tools for the various ecological quality elements (benthic fauna, fish, phytoplankton communites, etc). Descriptions of classification tools from Spain, UK and Ireland, Greece, Sweden and Estonia are currently in circulation. Countries are presently defining values for their assessment tools which they consider equivalent to the boundaries between the status classes (high, good, moderate, etc).

To ensure that these independent national processes lead to consistent classification of water bodies throughout the EU, a requirement for intercalibration of assessment tools was built into the Directive. Intercalibration is therefore an EU-wide exercise, divided by eco-region, to assess the behaviour of the classification tools around the boundaries of ecological quality within the water bodies. The purpose of the intercalibration exercise is to ensure comparable ecological quality assessment systems and harmonised ecological quality criteria for surface waters in the Member States. This ensures a harmonised approach to define one of the main environmental objectives of the WFD, the "good ecological status", by establishing:

- Consistency between the class (good/high and good moderate) boundaries and the normative definitions (i.e., definitions of quality elements for each level of water quality).
- Comparability with classification systems in other Member States.

The main outcome required of the intercalibration exercise is confirmation that protocols being implemented in each state for identifying good status boundaries are consistent with the normative definitions of the water quality status, and therefore consistent among countries.

For the purposes of Intercalibration exercise, a Europe-wide list of water body types has been generated. These eurotypes (as they are known) represent a list of types that are commonly found among some or all of the member states.

For intercalibration purposes, the marine area is divided into four different Geographical Intercalibration Groups (GIGs), according to the WFD Annex XI ecoregions and ecoregion complexes, based on salinity and tidal range:

- Mediterranean Sea (microtidal – euhaline) – (Tables 3, 4);
- Baltic Sea (microtidal, oligo – polyhaline) – (Table A4.5);
- NE Atlantic complex (NE Atlantic, North Sea, Barents Sea, Norwegian Sea)- (Tables A4.6, 7);
- Black Sea (microtidal, oligo – polyhaline).

The rationale for the selection of common types to be used in the intercalibration exercise is that a common type should be shared at least by two or more Member States/Candidate Countries (see Table A4.8 as an example from the NEA area). For the Mediterranean, two common transitional water types are now proposed for the intercalibration network. In the Baltic Sea, no common transitional water types have been identified, and therefore intercalibration will be only focus on coastal types. In the NE Atlantic GIG, common types were distinguished both for transitional and coastal waters. To date (April 2004), no common types have been selected for transitional and coastal waters in the Black Sea.

Table A4.3. Proposed coastal water body types for Intercalibration in the Mediterranean Sea.

Type	**Name of Type**	**Substratum**	**Depth**
CW – M1	Rocky shallow coast	Rocky	shallow
CW – M2	Rocky deep coast	Rocky	deep
CW – M3	Sedimentary shallow coast	Sedimentary	shallow
CW – M4	Sedimentary deep coast	Sedimentary	deep

Table A4.4. Preliminary proposal for the definition of the common intercalibration types for the transitional waters in the Mediterranean Sea.

Type	Description	Tidal range
TW-M5	Running transitional waters: deltas and river mouths	
TW-M6	Lentic transitional waters: Lagoons	>0.5 m
TW-M7	Lentic transitional waters: Coastal ponds	<0.5 m

Table A4.5. Common coastal types identified within the Baltic Sea for finalisation of the register for the intercalibration network.

New Type ID	Types[1] merged/ added	Description	Countries having type
CW-B0	New type[2]	Low oligohaline (salinity 0.5–3) sheltered, shallow, > 150 ice days	Sweden, Finland. No sites submitted. Countries will try to find new sites.
CW-B2	CW-B2	High oligohaline (salinity 3–6), sheltered, shallow, >150 ice days	Sweden, Finland. Insufficient sites at present. Countries will try to find additional sites.
CW-B3	CW-B3	High oligohaline (salinity 3–6), sheltered, shallow 90–150 ice days	Sweden, Finland, (possibly also Estonia). Insufficient sites at present. Countries should try to find additional sites.
CW-B12	Former CW-B5 and –B9	Mesohaline (salinity 6–22), sheltered, shallow	Poland, Denmark, Sweden. At present only sites at good/moderate boundary been submitted but more to be found.
CW-B13	Former CW-B6, and –B4 and –B10	Mesohaline, exposed, shallow	Sweden, Estonia (possibly also Lithuania, Latvia and Poland).
CW-B14	Former CW-B7 and –B8	Mesohaline, sheltered, shallow lagoons	Germany, Denmark, Poland.

Table A4.6. Common coastal types identified within the NE Atlantic ecoregion complex for intercalibration.

New Type ID	**Type**	**Name**	**Salinity**	**Tidal range**	**Depth**	**Current velocity**	**Exposure**	**Mixing**	**Residence time**
CW –NEA1	CW –NEA1	Exposed, euhaline, shallow	Fully saline (>30)	Mesotidal (1–5m)	Shallow (<30m)	Medium (1–3 knots)	Exposed	Fully mixed	Days
CWNEA26	Former CW – NEA2 and – NEA6	Sheltered, euhaline, shallow	Fully saline (>30)	Mesotidal (1–5m)	Shallow (<30m)	low – Medium (<1–3 knots)	Sheltered	Fully mixed	Days
CW – NEA3	CW – NEA3	Polyhaline, exposed (Wadden Sea type)	Poly-haline (18–30)	Mesotidal (1–5m)	Shallow (<30m)	Medium (1–3 knots)	Exposed	Fully mixed	Days
CW – NEA4	CW – NEA4	Polyhaline, mesotidal, moderately exposed (Wadden Sea type)	Poly-haline (18–30)	Mesotidal (1–5m)	Shallow (<30m)	Medium (1–3 knots)	Moderately exposed	Fully mixed	Days
CW – NEA7	CW – NEA7	Deep, low current, sheltered	Fully saline (>30)	Mesotidal (1–5m)	Deep (>30m)	low (<1 knot)	Sheltered	Fully mixed	Days
CW – NEA9	CW – NEA9	Fjord with a shallow sill at the mouth with a very deep maximum depth in the central basin with poor deepwater exchange.	Poly-haline (18–30)	Microtidal (<1)	Deep (>30m)	low (<1 knot)	Sheltered	Permanetly Stratified	Weeks
CW – NEA10	CW – NEA10	Polyhaline, microtidal exposed, deep (Skaggerak outer arc type)	Poly-haline (18–30)	Microtidal (<1)	Deep (>30m)	low (<1 knot)	Exposed	Permanently Stratified	Days

Table A4.7. Common transitional water type identified within the NE Atlantic ecoregion complex for intercalibration.

Type	**Name**	**Salinity**	**Tidal range**	**Depth**	**Current velocity**	**Exposure**	**Mixing**	**Residence time**
TW-NEA11	NE Atlantic Transitional waters	Oligo-Euhaline (0–35)	Mesotidal	Shallow (<30m)	Medium	Sheltered or moderately Exposed	Partially- or Permanently Stratified	Days-Weeks

Table A4.8. Countries sharing the common coastal types identified for intercalibration in the NE Atlantic ecoregion complex.

Type	**Name**	**BE**	**DK**	**FR**	**DE**	**IE**	**NL**	**NO**	**PT**	**ES**	**SE**	**UK**
CW – NEA1	Exposed	X		X	x	x	x	x	x	X		x
CW – NEA26	Sheltered		x	x	x	x	x	x	X	x		x
CW – NEA3	Polyhaline, exposed				x		x					
CW – NEA4	Polyhaline, moderately exposed				x		x					
CW – NEA7	Deep, low current, sheltered							x				x
CW – NEA9	Fjord with a shallow sill at the mouth.							X			x	
CW – NEA10	Skagerrak outer arc type							x			x	

1.6 Discussion, and implications of WFD for mariculture

1.6.1 Definition of Water Bodies and Reference Conditions

The WGEIM 2003 report discussed the potential significance to aquaculture of the policies adopted by member states in defining water bodies, particularly regarding the geographical scale of water bodies. Two scenarios present themselves. If water bodies are large, the localised impact of mariculture can be viewed in the context of the wider environment of the water body. Mariculture sites would be considered to be located within larger water bodies, and to act as pressures on the water quality within the wider water body. However, the definition of water bodies has to take account of the particular pressures that may impact the ecological quality of surface waters in the area. Pollution control authorities will use the water body as the primary unit for pollution assessment, control and regulation in the future. From that point of view, arguments can be expressed calling for a one-to-one relationship between water bodies and potentially significant sources of pollution. Such arguments could lead to the definition of rather small water bodies, perhaps on the scale of single areas leased for mariculture.

This debate is not confined to pressures arising from mariculture, but arises in the same way in relation to other activities operating within, and impacting on the coastal zone, such as domestic waste disposal, industrial effluent discharge, farming, forestry, etc.

The consequences for mariculture of the debate between the above two points of view are considerable. In the former case, in which larger water bodies are defined, the localised impact of mariculture can be viewed in the context of the wider environment of the water body. It would then be appropriate to assess the pressure from mariculture on the overall ecological quality of the water body. In the latter case, attention is closely focused on the effect of the pressures on the ecological quality immediately surrounding the mariculture unit. Such close focus would increase the likelihood of these small water bodies failing to meet the target of good ecological quality. For example, the well-established localised enrichment of sea bed sediments arising from fish farming (and to a lesser degree shellfish cultivation) is known to commonly result in alterations to the benthic infaunal community. The WGEIM report noted that the former approach appeared more consistent with the general philosophy underlying the WFD.

The typology exercise carried out in the UK and Ireland has defined relatively large waters bodies. A similar output has been presented from other member states. The example below (Figure A4.4) from the northwest coast of Scotland demonstrates that water bodies are typically large and can encompass entire sea lochs, extensive areas of coastline and large tracts of open waters.

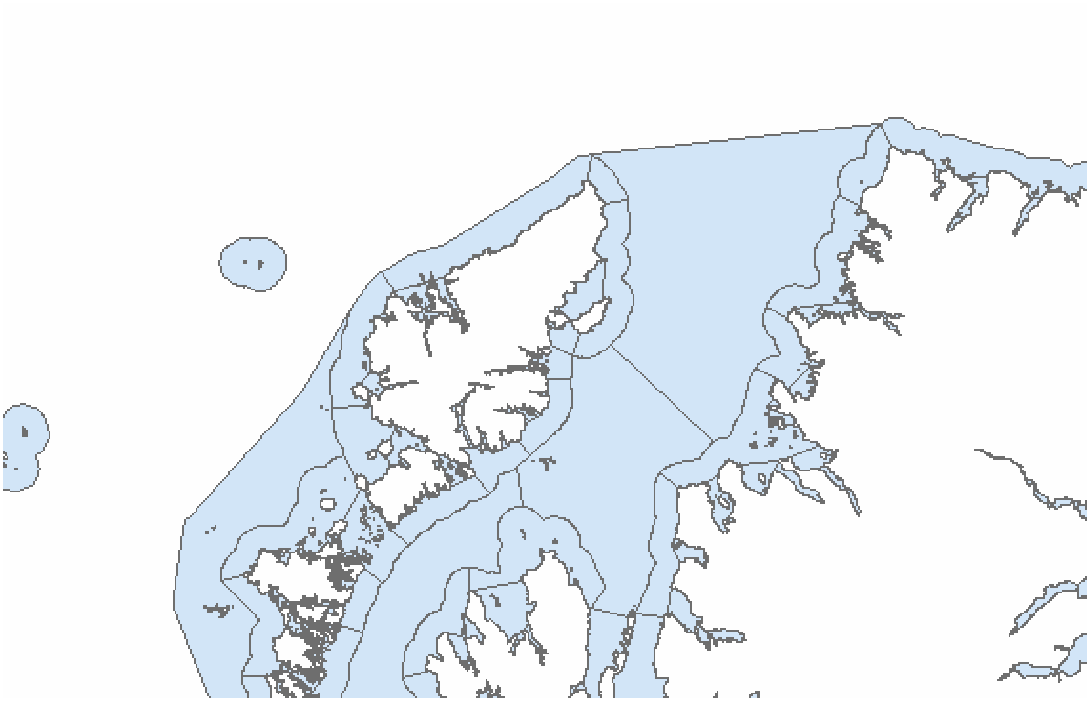


Figure A4.4. Northwest coast of Scotland showing definitions of water bodies in inshore and more open coastal waters

One apparent drawback from the definition of the water bodies at such relatively large scales is that it has proven very difficult to attach an ecological significance to the proposed physical types. However, it has long been known that it is not possible to relate the diverse array of habitats within an estuary or sea area to one physical type, because of the complex mosaic of habitats. Because typology is the basis of defining reference conditions and an anchor for high status and classification, the consequence of adopting this or any other set of simple physical types is that reference conditions must cover a wide range of habitats within each type.

In consequence, the concept of a mix of habitat specific reference conditions has been agreed as the way forward. Deriving habitat specific reference conditions allows the development of appropriate habitat specific reference conditions for each quality element. For example, the physical factors to which phytoplankton communities relate will be different to those of invertebrates and macroalgae – which relate more to substrate – and those of phytoplankton – which relate to the water column.

Through the development of habitat specific reference conditions it will be possible to allocate a mix of appropriate habitat specific reference conditions to each of the physical types. This freedom to apply a mix of the most appropriate habitat specific reference conditions to a water body recognises the complex mosaic of habitats within it. The most appropriate type specific reference conditions for each of the physical types will be thus chosen as an appropriate subset of the universal set of all habitat specific reference conditions.

While it is somewhat clearer what implications the WFD may have for intensive aquaculture activities that are confined to small spatial areas, there are certain activities e.g. bottom culture of mussels and intertidal culture of oysters, that may be impacted as a consequence of the Directive. Given that these activities may constitute large proportions of a water body the areas may be representative of the water body. These activities have defined impacts on the benthos over wide spatial scales (on the order of km^2) and consequently may put the water body at risk of failing to meet good ecological status. Initial risk assessment efforts carried out by England and Wales has determined that the shellfisheries (even if comprising up to 50% of a water body) may not be considered of having high pressure on a water body. However, this exercise considered managed wild-fisheries only and not true aquaculture operations. As yet, the implications on broad-scale aquaculture activities have not been fully assessed and discussed.

In summary, since the WGEIM 2003 report, most countries have defined their water bodies. These tend to be large, on the scale of kilometres to low tens of kilometres, and therefore the majority of mariculture activities will be considered as one of the pressures acting on the overall quality of the water body.

1.6.2 Chemicals used in mariculture

It is very likely that the chemicals used in fish farming activities will be considered as specific pollutants under Annex 8 of the Directive. This means that countries will be required to undertake chemical monitoring in water bodies where the risk assessment suggests that the quality may fail to attain overall good status, as a result of the discharge of these chemicals. The results from such chemical monitoring should be assessed against EQS values, which have been designed to protect the environment from unacceptable impacts from the chemicals concerned. Therefore, it is likely that EQSs will need to be developed for aquaculture chemicals, probably on a national basis. The Scottish Environmental Protection Agency has applied EQSs to aquaculture chemicals used in Scotland (see Table A4.9 below), and their approach may provide a useful lead for other countries with fish farming industries in coastal or transitional waters.

Table A4.9. Environmental Quality Standards for fish farm medicines applied by the Scottish Environmental Protection Agency

Active ingredient	Mode of application	Environmental Quality Standards
Azamethiphos	Bath	• Maximum allowable concentration (MAC), 3 hours after release, of 250 ng/l. • 24h MAC of 150 ng/l. • 72h MAC of 40 ng/l
Cypermethrin	Bath	• Short term (3 hour) EQS of 16 ng/l • Maximum allowable concentration (MAC) of 0.5ng/l, applied 24 hours after release • Annual average EQS is 0.05ng/l
Hydrogen peroxide	Bath	None – not considered to be a significant environmental risk
Teflubenzuron	In-feed	• 6.0 ng/l as an annual average in sea water and 30 ng/l as a MAC • 2.0 ug/kg dry wt/5 cm core depth as a general sediment quality standard to be applied as a MAC to surface sediment (cores 5cm depth) at more than 100m from the cages • 10.0 mg/kg dry wt/5cm core depth as a standard applied as an average value within the immediate under cage impact zone defined as surface area under and around cages to a distance of 25m from cage edges.
Emamectin benzoate	In-feed	• Concentrations in sediment should not exceed 0.763 ug kg^{-1} outside the AZE. • Concentrations in sediment should not exceed 7.63 ug kg^{-1} inside the AZE. • Concentrations in sea water should not exceed 2.2×10^{-4} ug l^{-1}. • Maximum number of treatments: • three treatments in any 12 calendar months, and five treatments in any two year growth cycle.

1.6.3 Classification of Water Bodies

The 2003 WGEIM noted, with reference to classification schemes, that "it was not yet clear how the national schemes, and subsequently the inter-compared schemes, will accommodate differences in the values of biological or hydro-chemical elements within water bodies. How this is to be done is clearly of importance to mariculture activities, as mariculture sites will present pressures on the environment and some of the elements of the assessment will be at less than reference status at these sites. Again, this question is not confined to mariculture. Many other anthropogenic activities that result in waste discharges are subject to the same uncertainties."

Some aspects of these issues have been developed, for example it now seems clear that quality status assessments will be made against habitat-specific reference conditions for each of the relevant quality elements, and that the final overall status assessment will default to the lowest of the component assessments. Therefore, a water assessed to be at high status for most quality elements, but at only moderate status for, say, benthic fauna, will be classified as of overall moderate status.

However, significant uncertainties remain unresolved in other aspects of classification. Examples include:

a) Assessment of chemical data against EQS values. Chemical monitoring will be required on several occasions during the year, and the primary assessment tool will be the calculation of an annual average for comparison with the EQS. Additional complexities arise in the case of non-continuous inputs of chemicals, such as will occur in the case of periodic use of sea lice treatment chemicals at fish farms. Current guidance suggests that sampling programmes should be designed so that periods of high use (and potentially increased concentrations) are covered. However, how such temporally biased sampling should be used to calculate an annual average (e.g., by time-weighting each sample in some way) has not been defined. The details of the final procedure will be an important factor with regard to fish farm chemicals. Similar issues of temporal averaging will also be relevant to other quality elements where more than one sampling event will take place each year.
b) Spatial averaging of monitoring data. In addition to the temporal averaging questions discussed above, uncertainties remain in how data from more than one sampling location within a water body should be combined to derive an overall assessment of the water body for that particular quality element. Defaulting to the worst case

may result in large water bodies receiving overall classifications dominated by results from single stations reflecting conditions in a small proportion of the whole water body, and might be viewed as giving a misleading impression of the water body as a whole. The influence of a small impacted area of sea bed below a fish farm on the overall classification of a larger water body is therefore not entirely clear.

c) Bottom cultivation of shellfish, predominantly mussels. It is well recognised that the character of the seabed and benthic fauna in the area of a dense mussel bed is very different from that in areas without mussel beds. In addition, commercial trawling or dredging for fish has been classified as a pressure on the morphology of the sea bed that can be (depending upon its intensity) a significant pressure on the quality on the benthic environment and associated fauna. It is therefore possible that risk assessments and subsequent monitoring may show that overall ecological status in some areas used for bottom cultivation of shellfish has been reduced. However, such assessments will be heavily dependent on the selection of reference conditions. Should the reference conditions reflect/accept the presence of the mussel beds, or should the mussel beds be considered as a pressure on the "normal" fauna of the area? The areas of sea bed in coastal waters utilised for bottom cultivation of mussels in some countries can be quite large in comparison to, say, the areas directly impacted by caged fish farming.

In addition, many waste discharges, including those from aquaculture, result in degradation of environmental quality in the immediate area of the discharge outlet (e.g., a few metres round the end of a piped discharge, or on the sea bed immediately under fish cages). Current regulatory practices recognise that such areas of impact, areas where EQS values may be exceeded, are an almost inevitable consequence of waste disposal and many other activities in coastal waters. The extent of such zones are an important element of the assessment of the acceptability of these activities. Such assessments will currently include the risk of impacts on the wider ecosystem in the receiving waters, which in many cases will be managed through the application of appropriate EQSs. While the application of EQSs is very much in keeping with the WFD, for both (priority) hazardous substances and specific pollutants, it is not yet clear how the mixing zone concept will be accommodated within WFD.

1.6.4 Measures to improve ecological quality (mitigation measures)

The overall aim of the Water Framework Directive is the achievement of good water status in all waters by 2015. It is probable that the initial classification will result in some water bodies being classified as a having an ecological status below the target level. In such cases, Member States will then be required to take steps to improve the status of these water bodies.

"Member states should adopt measures to eliminate pollution of surface waters by the priority substances and progressively to reduce pollution by other substances which would otherwise prevent Member States from achieving the objectives for the bodies of surface water."

"…… specific measures for the progressive reduction of discharges, emissions and losses of priority substances …."

The WGEIM 2003 report noted that, at that time, it was not clear what measures/actions Member States may choose to take. It seemed that rather little consideration had yet been given to this aspect of the Directive, but that it was anticipated that additional management and mitigative actions may be required of aquaculture operations in some areas where good ecological status has not been achieved.

There has only been limited development in this area over the last year. The UK-ROI have very recently established three new Working Groups to consider possible Programmes of Measures to respond to pressures on morphology, on water quality and on water resources. It is too early to make any assessment of the advice that these Groups may offer; the Directive does not require the identification of a programme of measures for achieving the environmental objectives of the Water Framework Directive until 2009. The suggestions of possible additional management and mitigative actions may be required of aquaculture operations in some areas where good ecological status has not been achieved, as discussed in the WGEIM 2003 report, remain to be confirmed.

Annex 5 Notes of EU level developments on the implementation of the EU Strategy for Aquaculture

The Strategy for the Sustainable Development of European Aquaculture (COM 2002 511 final) set out a wide range of policy principles on which the future development of aquaculture in the EU would be based. The Commission Strategy for the sustainable development of the European Aquaculture industry aims at:

- Creating long term secure employment, in particular in fishing-dependant areas;
- Assuring the availability to consumers of products that are healthy, safe and of good quality, as well as promoting high animal health and welfare standards;
- Ensuring an environmentally sound industry.

The scope and content of the more environmentally-oriented aspects of this document were reviewed in the 2003 WGEIM report.

The Strategy document also included an Annex which listed the proposed measures, and indicated those elements which should be taken forward by the EU centrally, and those which should be considered as the responsibility of individual member states to progress. Those elements of the strategy considered to be the responsibility of individual Member States include:

- Implementation of ICZM strategies and give priority to use of appropriate technologies;
- Increasing the use of existing opportunities for use of quality labelling;
- Considering aquaculture training needs when defining the European Social Fund programmes;
- Increasing stakeholder participation in aquaculture policy planning;
- Encouraging the use of mitigation measures and facilitating licensing of sites (for cage fallowing) and building permits (for sedimentation ponds);

The purpose of this paper is to summarise the main initiatives arising from the Strategy document that are being taken forward now (April 2004) by the EU.

1. Harmonisation of standards for organic aquaculture production

Organic production is based on 4 main principles:

- Consumers are entitled to know what they are eating, i.e., what the products contain and how they are produced.
- The welfare of animals should be taken into consideration in such a way that their natural needs are attended to.
- The production must be sustainable, i.e., there should be efficient use of resources and minimum pollution.
- The food must not contain chemical compounds that are potentially harmful to human beings or to the environment.

The EU, the United States and Canada all have certification schemes for production of organic food. Historically, regulations for organic production have been designed at the national level, with an independent certification body being responsible for the certification and monitoring of farms. However, there international standards also exist, the *IFOAM Basic Standards*, which specify minimum requirements for organic farming and which influence the design of national regulations. During the 1990s, EU regulations which cover organic production have come into effect. EC Directive 2078/92 covers agricultural production methods compatible with environmental conservation and maintenance of the countryside, and EC Regulation 2092/91 covers the certification of organic food labelling.

In order for a product to be certified as organic, the principles described above must be incorporated into detailed standards of production. In Europe, each country has one or several certification bodies which has specified detailed standards for organic agriculture. Recently, standards have also been introduced for aquaculture in the UK (*Soil Association*, *Food Certification*), Norway (*Debio*) and Sweden (*Krav*). Producers which are certified according to the standards can market their product with the label of the organic certification body.

As indicated in the Strategy, the organic logo is an important indicator of reliable organic quality, but there are no internationally binding organic aquaculture regulations. Council Regulation (EEC) 2092/91 sets up a framework of Community rules on production, labelling and inspection for organic farming. In the interests of producers and purchasers, the Commission wants to create specific common definitions and norms for organic aquaculture, and include norms for organic aquaculture in the Regulation.

Consequently, an initial meeting of a working group on organic aquaculture was held in Toulouse in November 2003, and further meeting was held in February 2004. Certification organisations met with other interest groups, and compared details of the conditions in the various certification schemes available to producers. The WG was able to establish that there were very considerable differences between the various schemes, and meetings are continuing to investigate routes to harmonisation.

2. Introductions, transfers and containment of aquatic organisms in aquaculture

There are currently no comprehensive rules at EU level regarding introductions, transfers and containment of aquatic organisms in aquaculture. In its Strategy document, the European Commission announced its intention to propose management rules for introductions, transfers and containment in aquaculture. It was stated that these rules would be consistent with the provisions of the ICES Code of Practice on the Introductions and Transfer of Marine Organisms. Two other sets of non-binding codes of practice and guidelines would also be considered in this context:

- 1998 EIFAC (FAO) Codes of Practice and manual of procedures for consideration of introductions and transfers of marine and freshwater organisms;
- 2001 NASCO Guidelines on Containment of Farm Salmon (CLN (01)53), as subsequently incorporated into NASCO's "Williamsburg Resolution" agreed at the NASCO Council meeting in June 2003.

The strategy on introductions, transfers and containment would also be informed by, and be consistent with, a number of other programmes and actions including:

- The Biodiversity action plan for fisheries (COM(2001)162final (27/3/2001) Volume IV), where Action IX, concerns *inter alia* a thorough evaluation of the potential impact of new non-indigenous species to aquaculture and promoting the application of the ICES/EIFAC Codes together with development of guidelines on containment of farmed fish. In the Council Conclusions on this action plan the need to minimise the genetic risk for wild fish stocks caused by escapement of farmed fish was acknowledged;
- Rio Article 15 and other subsequent international agreements recognise formally the need for a precautionary approach in relation to species introductions;
- FAO in its Code of Conduct for responsible fisheries (1995) calls at 9.3.1 for: 'efforts …to minimize the harmful effects of introducing non-native species or genetically altered stocks for aquaculture ….promote steps to minimize adverse genetic, disease and other effects of escaped farmed fish on wild stocks" and at 9.3.2 calls for "codes of practice and procedures for introductions and transfers of aquatic organisms";
- The Bergen Declaration, (action 33) agreed at the Fifth International Conference on the Protection of the North Sea (signed by B, Dk, F, D, NL, Nor, Sw, CH and the EC), in the context of environmental protection requirements, acknowledged the guidelines developed by NASCO in cooperation with the salmon farming industry in the North Atlantic on containment of farmed salmon and invited development and implementation of the FAO Code of Conduct.

Regarding introductions and transfers, a recent review (Minchin & Rosenthal, 2002) on species introduced for stocking and aquaculture describes problems which have occurred. This documents 69 cases of introductions of exotic species cultivated or used for re-stocking in Europe. The list includes ten cases involving algae, one flowering plant, three gastropod molluscs, 28 bivalve shellfish cases and 27 cases involving fish.

The review describes *inter alia*:

- Problems resulting from oyster movements including two protozoan parasites of the native European oyster which has resulted in the decimation of this species in Europe;
- Problems involving the introduction of exotic crayfish species to freshwaters for restocking purposes, including the early spread of the crayfish plague most likely with introductions from North America;
- The recent spread of king crab *Paralithoides camtschatica* along northern Norway following its introduction to Northern Russia in the 1960s from the Pacific Ocean;
- Problems with salmonids including the spread of the gill parasite *Gyrodactylus salaris* and the introduction of eggs of Pacific Coho salmon to France in 1971.

Regarding containment, the current scientific view is that escaped fish do have a negative effect on wild fish but that this is difficult to quantify due to the lack of a baseline. The Strategy document states that "escaped fish inter-breeding with native populations may induce long-term damage by the loss of genetic diversity. The introduction of foreign species may lead to biodiversity threats if the released or escaped exotics take root in their new environment. The potential deliberate release of transgenic fish without containment measures raises public concern in terms of risk to the environment. Introduction of new species may also lead to the introduction of diseases, both to farmed and wild stocks.

To minimise other potential environmental risks, the Commission will consider the development of rules on containment of farmed fish, the implementation of management rules on the introduction of non-indigenous aquatic species, as well as the need for specific legislation on transgenic fish."

The declared objectives of the Commission are to examine the possibility for rules on containment so as to minimise problems arising from escapees, to reduce the risks associated with non-indigenous species, propose rules for introductions, and to examine the need for specific legislation on transgenic fish. As an initial step towards addressing this undertaking, DG Fisheries held a consultation meeting in Brussels on 2nd December 2003 and has set up a Working Group. This WG has started to review the current legislation on the introduction of alien species, and also on containment of fish/shellfish at aquaculture facilities. It is not yet clear whether DG Fish will subsequently handle the issues of alien species independently of those of containment. The timetable proposed was for drafting of proposals, further consultations and adoption by the Commission during the first semester of 2004 followed by consideration by Council and European Parliamment during the second semester 2004.

3. Animal health issues

The Strategy noted that "the first Community legislation concerning animal health in aquaculture production was adopted in 1991. Today, detailed and harmonised legislation is in place covering animal health aspects of the aquaculture production. The primary legislation includes conditions governing the placing on the market of aquaculture animals and products (Directive 91/67/EEC as last amended by Directive 98/45/EC), measures for the control of certain fish diseases (Directive 93/53/EC, as last amended by Commission Decision 2001/228/EC) and of certain diseases affecting bivalve molluscs (Directive 95/70/EC as last amended by Commission Decision 2001/293/EC) . However, the legislation is specific to the situation of the sector in the late 1980s and early 1990s, so it needs to be updated and adapted to the present conditions of production and market."

The Strategy states that "there is a continuous need for the Commission to regularly review, update and simplify the animal health Community legislation for aquatic animals and products with regard to ever changing developments, particularly in the diversity of aquaculture production and in international experience and scientific knowledge"

In order to develop and progress this element of the Strategy, DG SANCO established an Expert Group whose task was to lay down what could be considered as the scientific basis of new legislation on fish diseases. In addition, several sub-groups were set up to present proposal on specific subjects such as disease control, imports, etc. The Federation of European Aquaculture Producers (FEAP) had an opportunity to comment on the different proposals from the experts during the process.

The main principles that have been proposed by the Expert Group include:

- That the Competent Authority in each Member State shall have access to qualified laboratory services and competence in risk analysis and epidemiology;
- That all farms must have a licence to carry out its activity;
- That a licence should not be granted if the activity in question would lead to an unacceptable risk of spreading diseases to other fish farms or to wild stocks;
- That all farms must apply a minimum of disease preventive management;
- That all farms must apply a risk based animal health surveillance scheme;
- That all farms must be categorised in one of three categories for each disease in the disease list;
- That all movements of animals must be accompanied by a movement document;
- That in relation to transport, minimum hygiene requirements must be applied;
- For a legal base requiring treatment / disinfection of effluent from slaughterhouses in relation to slaughter of diseased fish if the diseases are subject to control measures;
- The proposal for a legal base to take urgent measures in case of emerging diseases.

Based on the proposals of the expert Group, a proposal for a new EU Directive has been drafted and the Commission are currently seeking comments from Member States. It is envisaged that the new legislative will be in place in 2005 and that the existing legislation e.g. Directive 91/67; 93/53: 95/70 will be repealed.

4. Structural funds

Two of the aims expressed in the Strategy document are to re-focus priorities for public aid through the Financial Instrument for Fisheries Guidance (FIFG) and to promote research on new species and strains, as well as on alternative protein sources for fish feed

The Strategy expanded on these aims, stating that Regulation 2792/99 clearly states that increases in production that are likely to disrupt the market should not be encouraged. Therefore the Commission proposes that the intervention by public authorities in favour of aquaculture be re-directed towards favouring modernisation of the existing farms and diversification, rather than increasing production capacity for species where the market is close to saturation. Action should be taken on measures such as training, monitoring, research and development and clean farming technologies. The improvement of traditional aquaculture activities such as mollusc farming, that are important in maintaining the social and environmental tissue of specific areas, should be encouraged.

On new species, the Commission believes that research on species diversification is a top priority, for both fish and molluscs. Selected new species must necessarily respond to customers' preferences, in accordance with new market trends. Efforts should possibly be oriented to species such as seaweed, molluscs and herbivorous fish that are able to utilise the primary production more efficiently. Another priority is the introduction of effective genetic improvement programmes using selective breeding, as this will lead to considerable gains in productivity. Introduction of new species should be carried out in such a way to avoid the introduction of diseases.

Priorities in these and other areas through modification of the FIFG regulation and modification of national FIFG programmes can be summarised as:

- Redirecting public aid towards favouring modernisation of the existing farms and diversification, rather than increasing production capacity for species where the market is close to saturation
- Support for measures such as training, monitoring, research and development and clean farming technologies
- Increasing the range of products and the stability of supply
- Developing new tools to gather statistical information on production and markets
- Improving public support for transnational marketing campaigns
- Strengthening the support to further develop Producers' organisations /associations, co-operatives and trade organisations
- Recognise and strengthen the positive impact of extensive culture

A proposal has been put forward (COM(2003)658 final, 2003/0261 (CNS)) for a Council Regulation to amend Regulation (EC) No 2792/1999 laying down the detailed rules and arrangements regarding Community structural assistance in the fisheries sector. The draft proposal particularly notes that the EU Aquaculture Strategy contains a series of actions aiming to:

- Create long term secure employment in the aquaculture sector, in particular in fisheries dependent areas,
- assure the availability to consumers of aquaculture products that are healthy, safe and of good quality, as well as promoting high animal health and welfare standards, and
- ensure an environmentally sound development of the aquaculture industry.
-

The proposal states that, within the measures related to aquaculture in the FIFG Programmes, priority shall be given to the following areas factors:

(i) the development of techniques that substantially reduce environmental impacts,
(ii) the improvement of traditional aquaculture activities such as mollusc farming, that are important in maintaining the social and environmental fabric of specific areas,
(iii) the modernisation of existing enterprises,
(iv) farmed species diversification;

A number of amendments to the regulations governing FIFG schemes related aquaculture and the environment are proposed and summarised below:

- The promoters of intensive fish farming projects may be granted public aid to cover the cost of collecting information on environmental impact and assessment costs as part of the EIA process.
- The initial costs incurred by aquaculture enterprises to join in the Community eco-management and audit schemes set up by Regulation (EC) No 761/2001, as well as investments in works concerning the installation or improvement of water circulation in aquaculture enterprises and on service vessels shall be eligible for support.
- The Member States may grant financial compensation to shellfish farmers where the contamination due to the growth of toxic algae makes it necessary, for the protection of human health, to suspend harvesting for more than six consecutive months.
- Small-scale, applied-research initiatives, not exceeding EUR 150 000 in total cost and three years in duration, carried out by an economic operator, a scientific or technical body or other competent body, shall be eligible for support as pilot projects, provided that they contribute to the objectives of sustainable development of the aquaculture industry in the Community.

5 Other relevant programmes

a) The EU CRAFT programme can support applied research in the aquaculture field. The primary assessment criteria are scientific, and it is necessary to demonstrate that the outcome of the research would have some breadth of application. This is garanteed in some way by the fact that participating enterprises must be from three different EU contries at least. However, the views and needs of the industry strongly steer the content of the programme.

b) By contrast, funds may become available under Framework Programme 6 (FP6) for projects that provide scientific support to policy, ie directly support the implementation and development of the Common Fisheries Policy. Projects for this funding mechanism are determined primarily by DGFish and assessed by DG Research.

c) Additionally, cohesion funds may be made available throuth the Interreg Programme relying on structural funds (FEDER/EFRED) under the responsibility of European Regions.

The above programmes have not been developed in direct response to the EU Aquaculture Strategy, but both will be influenced to a degree by the policies in the Strategy.

Reference

Minchin, D., and Rosenthal, H. 2002. Exotics for stocking and aquaculture: making correct decisions. pp. 206–216. *In* "Invasive aquatic species of Europe: Distribution, impacts and Management". Ed. by E. Leppäkoski, S. Gollasch, and S. Olenin. Kluwer Academic Publishers, Doordrecht, Boston, London. IX, 583 pp. (ISBN 1-4020-0837-6).

Annex 6 Preliminary drafts of "state of knowledge" of the potential impacts of escaped aquaculture marine non-salmonid finfish species on local native wild stocks (e.g., sea bream, cod, turbot, halibut)

Environmental Effects of New Aquaculture Species

Introduction

Today's fisheries management decisions are based on an array of factors that extend beyond considerations of the local social and technical aspects of the fishery, the biology of the organism harvested and environmental processes. International trade plays an important role in determining where and what type of fisheries oriented environmental regulations may be applied without risk of economic repercussions.

Zero risk decisions (including the decision to defer a decision) do not exist in modern fisheries resource management. Under World Trade Organization (WTO) rules any country found to impede trade with out adequate reason can be subject to penalties and counter-veiling duties that are not restricted to the economic sector in which the trade has been inappropriately limited. For example an unjustified trade restriction in fisheries could result in the affected country, with approval of the WTO, implementing trade restrictions in another economic sector such as the auto industry sector. These trade regulations are not designed to limit a countries ability to protect its natural resources they simply require that any regulations that affect international trade be justified in a fashion that is internationally acceptable.

An important tool in designing and justifying regulatory actions in the international market place is risk analysis. The Office International des Epizootic manual for disease control uses risk as the basis for justifying restriction on movement of aquatic animal in response to concerns about disease transfer and control. Their stated intent was to provide guidelines and principles for conducting transparent, objective and defensible risk analyses for international trade. ICES has embraced this approach in their latest (2003) Code of Practice for the Introduction and Transfer of Marine Organisms (hereafter referred to as the ICES Code). One part of the ICES Code is specifically designed to address the "ecological and environmental impacts of introduced and transferred species that may escape the confines of cultivation and become established in the receiving environment."

The following assessments of potential environmental effects of cultured species newly gaining prominence (newly cultured species) have been formulated following a risk analysis model. The intend in doing so is to create a clear and transparent basis upon which member countries can elaborate application of the code as it applies to their specific environmental conditions and the newly cultured species they will produce.

For clarity of process the entire analysis is broken down into 5 components: Hazard Identification, Risk analysis, Risk Management and Risk Communication. The process and its components are represented diagrammatically in Figure A6.1. The Risk Assessment component is further broken down into 4 subcomponent steps: Release Assessment, Exposure Assessment, Consequence Assessment and Risk Assessment.

In this analysis we do not discuss hazards of the culture of new exotic species. Local jurisdictions should subject new aquaculture species that are exotic to their proposed location of culture, to an evaluation under the ICES Code of Practice for the Introduction and Transfer of Marine Species prior to permitting their culture.

Similarly, we do not discuss potential disease interactions other than to encourage member states to apply the aforementioned ICES Code and the OIE protocols.

The following description of risk analysis for evaluation of aquaculture activities is an adaptation of the process used by the OIE to analyse risks associated with aquatic disease.

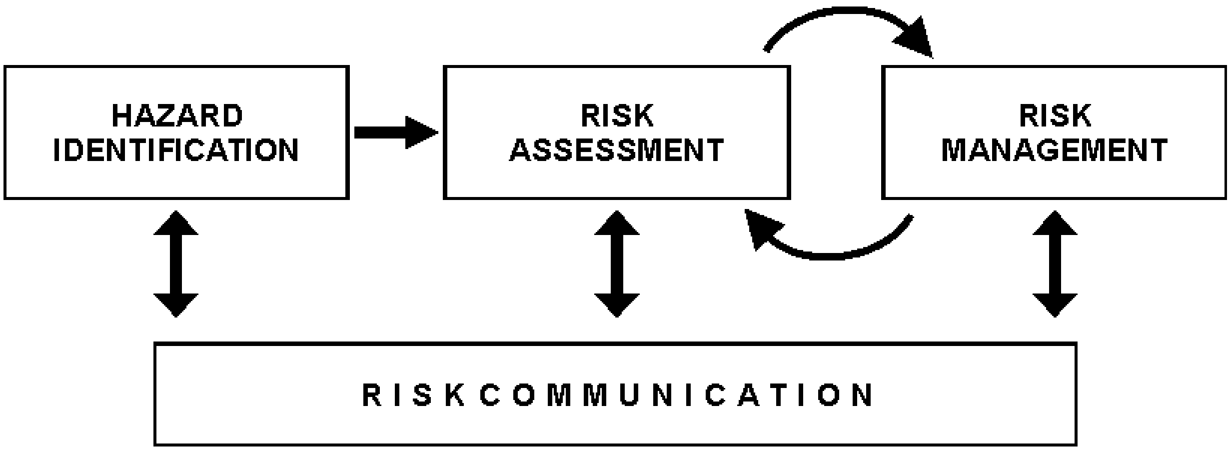


Figure A6.1. The four components of risk analysis (after OIE 2003).

Hazard identification

Hazard identification involves identifying the aspects of the new species for culture that could potentially produce adverse consequences for the local environment.

Hazard identification is a categorisation step, identifying effects dichotomously as hazards or not hazards. The risk assessment should be concluded if hazard identification fails to identify an increased risk of an effect associated with culture of the new species.

Risks identified should be those appropriate to the species being cultured and the stock from which the cultured population is derived. It is then necessary to identify whether the risk of each hazard effect is already present in the local environment, and whether it is subject to control or eradication, and to ensure that culture limitations applied to imported stock are not more trade restrictive than those culturing local conspecifics.

Enhancement of wild populations is an example of an alternate risk that might have a some what similar effect as escapes from commercial aquaculture facilities. While this may reduce the marginal effect of aquaculture escapes these past enhancement activities can also contribute significant information for identification of hazard effects.

An evaluation of the local authorities' surveillance and control programmes, and *zoning* and regionalisation systems are also important inputs for assessing the likelihood of a hazard effect.

Risk assessment steps

1. Release assessment

Release assessment consists of a description the pathway(s) necessary to 'release' (that is, introduce) a hazard into a particular environment, and estimating the likelihood of that complete process occurring. The release assessment describes the likelihood of the 'release' of each of the hazards under each specified set of conditions with respect to amounts and timing, and how these might change as a result of various actions, events or measures. Examples of the kind of inputs that may be required in the release assessment are:

a) **Biological factors**

– Species, strain or genotype, and age of animals,

b) **Area Specific factors**

– Density of culture facilities, numerical abundance in each containment unit

– Evaluation of surveillance and control programmes, and zoning systems of local authorities.

– Potential release sites due to transport, culture and treatment,

c) **Species specific factors**

- Schooling behaviour,,
- Exploratory behaviour,
- Jumping behaviour
- Rubbing or nibbling behaviour
- Effect of handling behaviour (e.g., jumping)
- Effect of starvation
- Effect of medication
- Effect of external predators or activity on or about containment structure
- Effect of genetic manipulation
- Effect of domestication on behavior

If the release assessment demonstrates no significant risk, the risk assessment need not continue.

2. *Exposure assessment*

Exposure assessment consists of describing the biological pathway(s) necessary for exposure of the local environment to the hazards and estimating the likelihood of these exposure(s) occurring, and of the spread or establishment of the hazard.

The likelihood of exposure to the hazards is estimated for specified exposure conditions with respect to amounts, timing, frequency, duration of exposure, routes of exposure, and the number, species and other characteristics of environment exposed. Examples of the kind of inputs that may be required in the exposure assessment are:

a) **Biological factors**

- Presence of species for potential hybridization/intergradation
- Genotype of conspecifics
- Properties of the cultured fish that would affect interbreeding (e.g., mate preference, timing of spawning, survival to spawning.
- Success as a predator
- Success at avoiding predation
- Success as a competitor for resources
- Migratory or dispersal habits,
- Ability to find spawning aggregations

b) **Area Specific factors**

- Aquatic animal demographics (e.g., presence and distribution of known con-specifics, competitors, predators and prey),
- Human and terrestrial animal demographics (e.g., possibility of scavengers, presence of piscivorous birds, sport and commercial fishing activity),
- Geographical and environmental characteristics (e.g., hydrographic data, temperature ranges, water courses).

c) **Species specific factors**

- Whether there has been significant genetic differentiation between wild and cultured conspecific strains,
- Abundance of conspecifics, predators, prey and competitors.
- Waste disposal practices.

If the exposure assessment demonstrates no significant risk, the risk assessment should conclude at this step.

3. ***Consequence assessment***

Consequence assessment consists of identifying the potential biological, environmental and economic consequences. A causal process must exist by which exposures to a hazard result in adverse health, environmental or socio-economic consequences. Examples of consequences include:

a) **Direct consequences**

- The scale and potential significance of interbreeding with local populations,
- Adverse, and possibly irreversible, consequences to the environment,

b) **Indirect consequences**

- Surveillance and control costs,
- Compensation costs,

- Potential trade losses,

- Adverse consumer reaction.

4. *Risk estimation*

Risk estimation consists of integrating the results of the release assessment, exposure assessment, and consequence assessment to produce overall measures of risks associated with the hazards identified at the outset. Thus risk estimation takes into account the whole of the risk pathway from hazard identified to unwanted outcome.

Qualitative assessments should always be performed and quantitative assessments should be used to further inform the outcome of the qualitative assessment. Because of its more precise nature quantitative analysis is necessarily more focused in nature and has the potential to be more precise but less accurate over all the potential aspects of a hazard.
For a quantitative assessment, the final outputs may include:

- The various populations of *aquatic animals* and/or estimated numbers of *aquaculture establishments* or people likely to experience health impacts of various degrees of severity over time;
- Probability distributions, confidence intervals, and other means for expressing the uncertainties in these estimates;
- Portrayal of the variance of all model inputs;
- A sensitivity analysis to rank the inputs as to their contribution to the variance of the risk estimation output;
- Analysis of the dependence and correlation between model inputs.

Risk management components

1. Risk evaluation – the process of comparing the risk estimated in the risk assessment with the appropriate level of protection.

2. Option evaluation – the process of identifying, evaluating the efficacy and feasibility of, and selecting measures to reduce the risk associated with culturing a new species in line with the appropriate level of protection. The efficacy is the degree to which an option reduces the likelihood and/or magnitude of adverse environmental and economic consequences. Evaluating the efficacy of the options selected is an iterative process that involves their incorporation into the risk assessment and then comparing the resulting level of risk with that considered acceptable. The evaluation for feasibility normally focuses on technical, operational and economic factors affecting the implementation of the risk management options.

3. Implementation – the process of following through with the risk management decision and ensuring that the risk management measures are in place.

4. Monitoring and review – the ongoing process by which the risk management measures are continuously audited to ensure that they are achieving the results intended.

Principles of risk communication

1. Risk communication is the process by which information and opinions regarding hazards and risks are gathered from potentially affected and interested parties during a risk analysis, and by which the results of the risk assessment and proposed risk management measures are communicated to the decision makers and interested. It is a multidimensional and iterative process and should ideally begin at the start of the risk analysis process and continue throughout.

2. A risk communication strategy should be put in place at the start of each risk analysis.

3. The communication of risk should be an open, interactive, iterative and transparent exchange of information that may continue after a decision is reached.
4. The principal participants in risk communication include the local authorities and other stakeholders such as recreational and commercial fishermen, conservation and wildlife groups, consumer groups, and domestic and foreign industry groups.

5. The assumptions and uncertainty in the model, model inputs and the risk estimates of the risk assessment should be communicated.

6. Peer review of risk analyses is an essential component of risk communication for obtaining a scientific critique aimed at ensuring that the data, information, methods and assumptions are the best available.

The limitations of advice provided

The above outline demonstrates that a proper risk analysis is the product of the extensive consultation and communication. That process can not be completed with the time and resources available to this working group. In no way should the following analysis be considered a substitute for completion of a full risk analysis prior to development of extensive industries based on the species identified below.

Instead, it is the intent of the working group to provide a substantive component of the historical data accumulation and organization of information that would be necessary for a proper risk analysis. Based on available information the group will provide what insights it can into the unique information requirement that might be required for the risk analysis for each of the species. Member countries should complete a full risk analysis for the conditions in their area before culturing the new species. The working group will identify those aspects to which attention should be given for special attention for each species and where possible will identify areas where knowledge development would prove most valuable at improving the accuracy or reducing uncertainty in the risk analysis.

Before undertaking any risk analysis it is very important that the country undertaking the analysis define *a priori* and explicitly what is their acceptable level of protection and the benefits they are willing to forego to achieve that level of protection. Failure to do so may compromise objectivity and markedly reduce the value of the analysis.

Sea bass – *Dicentrachus labrax*

1 Hazard Identification

1.1 Life history – Description of wild population

1.1.1 Distribution

The native sea bass range extends from the Mediterranean to the North Sea. It is a euryhaline and eurythermic (5 to 28°C) species. It can survive in freshwater for weeks (some attempts to rear it in freshwater have proved successful) and can be sometimes found in rivers. Its salinity preference is $15 g.l^{-1}$ (Saillant *et al.*, 2003) and it is not sensitive to flood conditions. Traditionally it was found in coastal lagoons as well at river mouths. When 12 mm in length, larvae actively swim to the nursery habitats. Up to the size of 20 mm, sea bass are pelagic, then demersal. Sea bass hunt for preys in zones of breaking waves.

Throughout their life, sea bass aggregate in schools ranging in size from some tens to thousands of individuals. In the Mediterranean, juveniles live in near-shore coastal areas, in lagoons and/or estuaries from spring to autumn. This distribution appears to be linked to optimal feeding requirements (these areas are very productive). Movements of year class 1 may be as much as 70 kilometres (Chauvet *et al.*, 1992). Juveniles stay in groups in shallow areas in open sea and into estuaries. When adults leave these areas they move to depths less than 50 m deep in autumn. Much of the stock then remains inshore, and appears to move little (only a few kilometres). Chauvet *et al.* (1992) were unable to detect any movement in fish more than 25 cm in length during a 300 day tagging experiment in the Gulf of Lion.

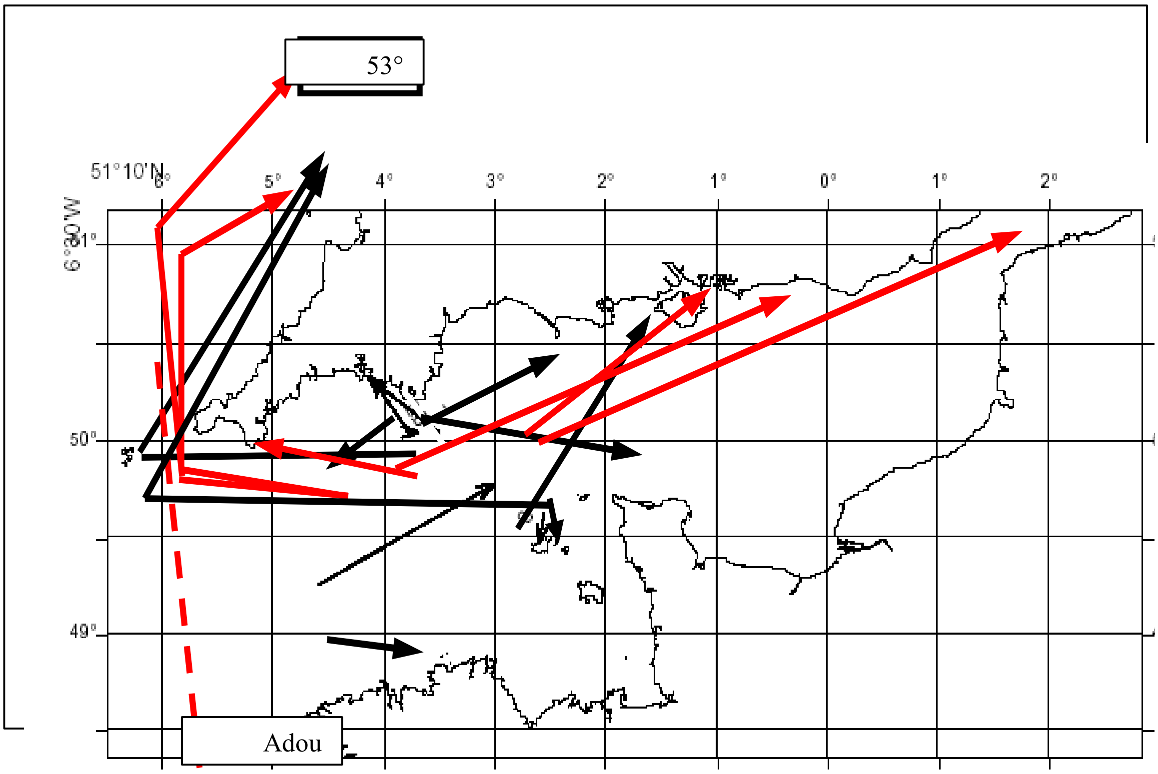


Figure A6.2. Adults displacements from releases in 2002 (red) and 2003 (black). From Morizur (2004).

In the Atlantic, juveniles behave as they do in the Mediterranean, where they also colonize tidal flats and salt marshes. In contrast to the Mediterranean area, the sea bass adults in the Atlantic appear to undertake large movements. Tagging experiments (Morizur, 2004) conducted in the winter demonstrated that large adults may migrate for more than 400 miles within a month (see Figure A6.2).

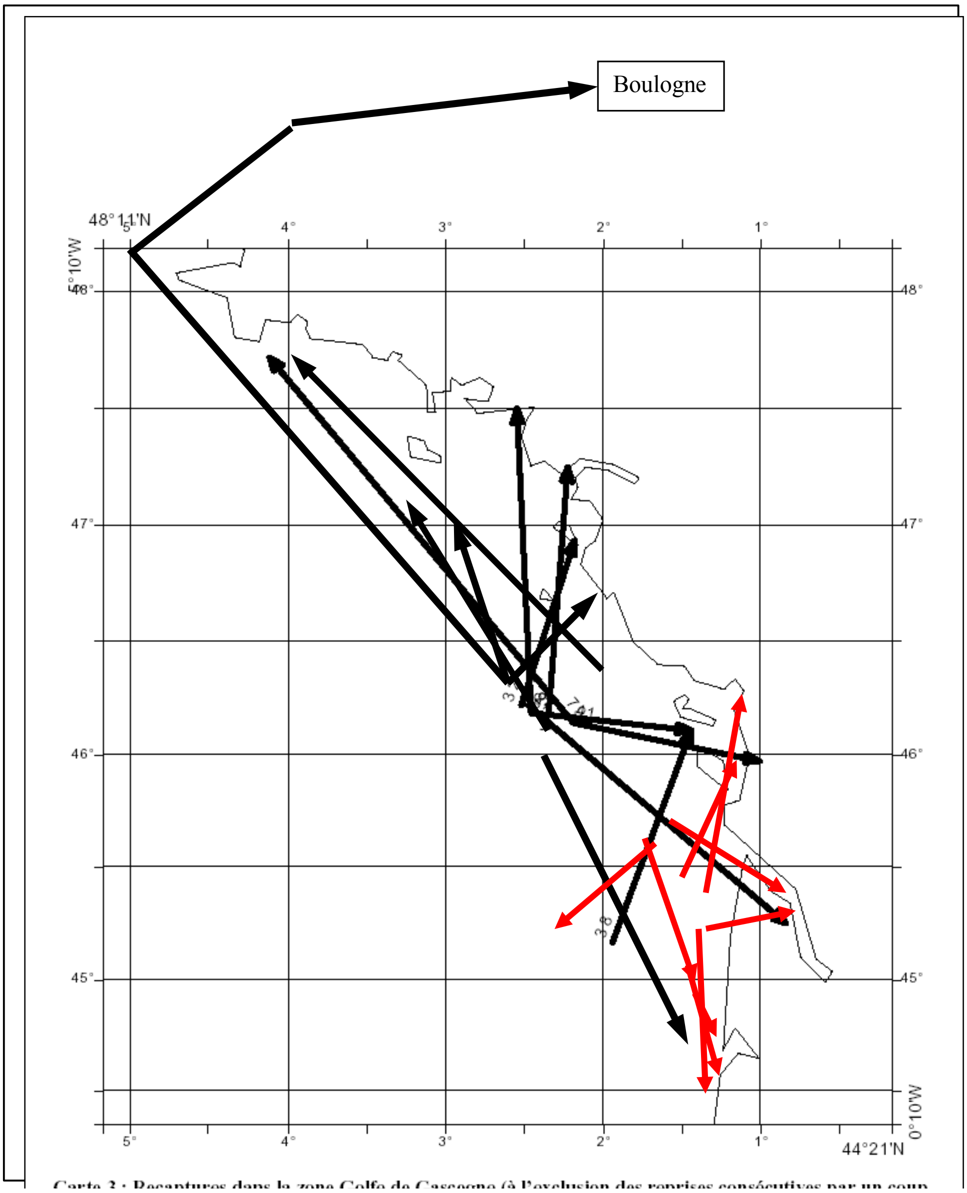


Figure A6.3. Adult displacements in 2002 (red) and 2003 (black). From Morizur (2004).

Very few sea bass appear to move between Channel areas and the Gulf of Biscay. In contrast, tagging experiments in the Gulf of Lyon indicated that adults more than 3 year old were sedentary (Chauvet *et al*, 1992). Movements seemed bigger in winter and are probably linked to environmental and feeding conditions. On the western coast of England and Wales the migration pattern appears to be southward in winter and northward in spring (Pickett and Pawson, 1994).

Some cases of large escapes have been reported. These cultured sea bass have been found in significant numbers near these farms for some months, demonstrating a relatively high degree of site fidelity.

The stock is said "not under threat" in the Atlantic ICES sub-areas. The status in the Mediterranean is not known.

1.1.2 Growth and Survival

Females are 20% bigger than males of the same age (Saillant *et al*, 2002). In the Mediterranean sea bass are generally 40% bigger than in the Atlantic sea bass of the same age. Growth almost ceases in colder (below 10°C) winter waters.

Table A6.1 summarizes the growth (length in cm/weight in grams) for wild females in these two areas (from Barnabé, 1989), and for cultured females (Dosdat, personal communication.).

Table A6.1.

Age (year)	1	2	3	4	5	6	7
Wild Atlantic	8/10	16/45	23/130	29/260	35/450	39/600	44/900
Wild Mediterranean	17/55	28/230	39/600	47/1100	54/1600	59/2100	
Cultured Mediterranean	21/100	32/350	42/800				

1.1.3 Diet

In both the Atlantic and Mediterranean, young (first year) sea bass diet is largely crustaceans (mysids, amphipods, decapods). Adult sea bass eat fish (sardines), crustaceans (shrimps) and cephalopods. The change in diet occurs when the sea bass are around 40 cm long. At all ages sea bass are cannibalistic (Pickett and Pawson, 1994). Sea bass can eat other sea bass up to half its size by weight. In cultivation, it is reported to be cannibalistic even during the very young stages (less than 6 months). This behaviour has may affect the ability for escapees to survive in the wild. Different year classes of wild sea bass generally do not mix.

1.1.4 Abundance

The abundance of wild sea bass is not precisely known. In the Mediterranean, the total landings from capture fisheries is estimated at 13 000 tonnes per year. Based on a minimum fishing mortality of 0.2 (ICES, 2002), the total wild stock could be estimated at a maximum of 65 000 tonnes. Based on an assumptions of an average weight of 1kg of the landings, and a mortality rate of 0.9 (Pickett and Pawson, 1994), the number of wild fish in the Mediterranean should be evaluated at $210 * 10^6$ individuals. In the Atlantic, sea bass abundance is very variable and appears to be in response to climatic conditions rather than feed availability. The present level of exploitation in the Atlantic is considered sustainable by the ICES (2002). This species is not subjected to TAC and quotas.

1.1.5 Reproduction and spawning

In the Mediterranean, age at first maturity is 2 years for the males, and 3 years for females. Age of maturity seems to be delayed by one year in the Atlantic. First spawning occurs even later, particularly in the Northern areas (6 years in the Channel).

Spawning areas have been determined in the Channel, Celtic Sea, west of Brittany and Gulf of Biscay. Spawning has been observed (Barnabé, 1976) inshore on rocky bottom in the Mediterranean (by 5–6m depth). Spawning occurs in December to February in the Mediterranean and in March to June in the Atlantic. In spawning areas, adults are known to concentrate period above the spawning grounds. In the Atlantic and Channel, adults do not show fidelity to a precise spawning area (ICES, 2002). At maturity, in the Atlantic region, adults move outside their feeding areas, not necessarily recruiting to their parent spawning stocks. Unlike salmonids, sea bass continues to feed during maturation and spawning.

Spawning occurs in the middle of the water column. Consequently spawning areas are not precisely defined. Males need to be very close to the female (fertilisation off the eggs has to occur within seconds), but it is not known if fish forms pairs. It is most likely that they do not. Eggs are emitted once a year over a period of a few hours and fertilised eggs are planktonic. Reproductive success appears to be linked to temperature in Ireland (Pawson, 1992; Fahy *et al*., 2000).

Cultivated fish are reported to mature in the sea cages, at the same time that conspecific wild populations in the area of the cages do. Maturation and reproduction is under the control of temperature and photoperiod.

1.1.6 Genetic structure of the populations

There is evidence that there are three endemic populations of sea bass: one covering the area of the Atlantic and Sea of Alboran, one in the western and one in the eastern Mediterranean (Patarnello *et al*., 1993; Cesaroni *et al*., 1997). This differentiation was described using the allele frequency of six microsatellite loci. Microsatellite analysis is very sensitive in detecting genetic variability, but it is not always clear what is intra and inter population genetic variability. It is suspected that the passive retention of larvae on either side Gibraltar Strait is not a sufficient explanation for the persistence of the pattern that has been detected. Castilho and Mc Andrew (1998) reported possible population structuring along the coast of Portugal using allozymes. However, allozyme work on wild populations suggests a different interpretation from other markers. There seems to be significant genetic divergence between the eastern and western Mediterranean (Bahri-Sfar *et al*., 2000), as well as differentiation within the eastern population. In the Gulf of Lyon, differentiation also occurs in the eastern stock between “groups” of fish that grow in lagoon environments and those that live in the open sea, although both groups appear to share the same breeding areas (Allegrucci *et al*., 1997;

Lemaire *et al*., 2000). Thirteen enzymatic loci exhibited moderate to high values compared with microsatellites. This was interpreted as evidence that these allozymes are non-neutral, and then submitted to environmental pressure. However, only six loci seemed to be implicated in differentiation between marine and lagoon samples. The cause of differentiation for the other allozymes is unclear. A possible explanation for the pattern of marine and lagoon population has been suggested by Lemaire *et al*. (2000). In the Atlantic the mixing of recruiting bass in the Channel and Celtic sea populations is inferred by the very low genetic structuring of these stocks and the very limited genetic differentiation between spawning stocks in that region. This suggests that mixing between generations is sufficient to homogenise the genetic make-up of the bass population in Northwest Europe. In contrast Mediterranean sea bass are known to migrate between coastal and off-shore grounds, and homing behaviour is suspected due to local genetic differentiation over small areas (Allegrucci *et al*., 1997).

These genetic variations have yet to be correlated with phenotypic variations.

Another closely related species *(Dicentrarchus punctatus)* living in the same ecological and geographic areas is thought not to interbreed naturally with sea bass, but artificial breeding has been reported and hybrids have been produced (Ky, IFREMER, personal communication). The fertility of these hybrids has not yet been confirmed. A genetic distance tree inferred from the polymorphism at six microsatellite loci shows a distinct pattern for the two species. *D. labrax* samples appear to be genetically more homogeneous than *D. punctatus,* indicating a lesser level of gene flow in the latter species (Bonhomme *et al*., 2002). While appearing more differentiated, *D. punctatus* presents no clear geographical organization of its genetic variability in contrast to *D. labrax* samples.

For cultured sea bass, from the early 70s to the mid 80s, breeders were originating from the wild. Then cultured breeders began being utilised, with a growing risk of in-breeding. From the 90s, genetic improvement through selective breeding occurred in France, Greece and Italy, increasing the distance between wild and farmed population. In fish farms, males are generally outnumbering females. The exact nature of the mechanism controlling this which is behind is not demonstrated yet, but environmental effects on sex determination have been proved (Saillant *et al*., 2002).

From various surveys, it appears that the majority of the breeders utilised by the producers are originating from the Western stock. Transport of non local stocks has already occurred in the Eastern Mediterranean and the Channel. This is due to the fact that the very first developments occurred in Mediterranean France, Spain and Italy, while fry and eggs have been exported to other countries. Even when local stocks are utilised a drift may occur.

1.2 Known effects of cultured populations

The only data available on escapes indicates that when sea bass from cultured Eastern Mediterranean populations escaped in the western Mediterranean, they established and maintained distinct populations of the eastern Mediterranean phenotype without intergradating with the local population. . No stock enhancement or voluntary release operations have been reported.

While there is limited experience with the effects of escape of sea bass raised in culture, there is probable cause to believe that in some locations, some effects might occur. Differentiation within the wild stocks and between wild and cultured stocks suggests the potential for disruption through introgression, particularly where small stocks may be involved. However it is clear from the example in the Eastern Mediterranean that this will not always happen.

Sea bass have been escaping captivity in the Mediterranean for the last 2 decades. The lack of data on sea bass survival over from before and after the advent of sea bass culture and on the degree of genetic differentiation between wild and cultured sea bass (which appears lows at present, see Bahri-Sfar *et al*, 2004) precludes determining if there has been any genetic based effect of sea bass escapes on the wild population. Opportunities for interbreeding are highly probable, even if no interbreeding has been reported. (Bahri-Sfar *et al*., 2004).

There is inadequate data with which to comment on whether predation is a significant controlling factor for wild populations. The seasonal migration of the northern population could be indicative of nutritional resource limitation or simply a response to local water temperatures. During the growth period (April to October), surveys on condition index, feeding status and fat contents of year 0 and year 1 classes did not show any shortfall in feeding resources, even in the years of high recruitment. Thus there is no reasonable cause, *a priori,* to believe that significant changes in predator or prey abundances might occur as the result of escapes from the existing level of sea bass culture.

A similar lack of correlative data between the ecology of the Mediterranean in the region of sea bass culture before and after the culturing began also precludes our ability to comment on whether ecological shifts might result from sea bass culture activities as now practiced.

Based on presumptive evidence for potential impacts due to genetic introgression the following risk assessment has been constructed.

2 Risk Assessment

2.1 Release Assessment

The majority of the sea bass reared in Europe are maintained in floating sea cages, into near shore locations. One big production unit (1500 tonnes a year) is based on land in Northern France (on the Channel) using heated effluent from a power plant. Some farms are using salt marches to produce fish in ponds in the Gulf of Biscaye, in Spain and in Italy. The bulk of the production is based in Greece, using the sea cage technology, but all the Mediterranean countries are

producing sea bass. Farms using the recirculating technology are expending slowly. Usually, 10 to 20 gram fish issuing from hatcheries are entering these farms, where they are reared at a maximum density of 20 kg/m^3. Sorting operations are frequent all along the production cycle, because of both cannibalism and the high heterogeneity in growth rates. It necessitates anaesthetising the fish. The typical production cycle duration is 2 years to produce 250–400 gram fish. It requires one more year to produce 600–800 gram ones for specific markets. The market for sea bass is changing and is asking for bigger fish (three year age) than the traditional pan size, age at which bass are mature males or immature females. Consequently there is an increasing incidence of mature fish in the sea cages.
Recently, a movement of sea cage locations far from the coast occurred in Spain and Greece, less protected areas, using basically the same technology. It is due to the environmental pressure on the near shore sites. The main forces limiting this run to off shore locations are economical, then technical. The production is likely to expand in the near future, probably by moving fish cages towards these new locations. Closing the breeder stocks for selective breeding purposes began in the late 90s in some big companies.

The aquaculture production was about 62000 metric tonnes in 2002, essentially from the Mediterranean Sea, representing a standing stock of about 100 000 tonnes. The average weight can be estimated at 200 g, which mean some 500.10^6 individuals being under cultivation, to be compared to the evaluation of the number of wild individuals (see section 114).

No information is available on the actual number of escaped fish from the farms. This information is not compulsory to the fish farmers where it is produced. It is clear that the sea cage technology is more likely to be concerned with escapees than land based farms, particularly those utilising recirculating systems. It can be assume that the range of 0.1 to 3% of the reared stock escaping from the cages in the salmonid industry is applicable to the sea bass industry. The primary cause for releases is containment failures, cage broken by bad weather conditions or net opened by external predators (including the human) or lack of maintenance. Sorting operations, since they are done on the cages, and transport may also induce un-intentional releases.

In addition, farmed sea bass are reported to produce viable eggs and sperm in the cages. One 800 g female may produce 300 000 eggs during the spawning season. This brings two additional consequences: wild males fertilizing cultured females and cultured males fertilizing wild females (very unlikely) or cultured stocks dispersing embryos in the environment, the survival of which will be possible but highly dependant on the sites. The farming sites appeared more favourable to this last issue in the Mediterranean than in the Atlantic, where farms are generally far from the spawning areas and weather condition more unfavourable.

Survival time of cultured fish in the wild at any age has not been investigated.

2.2 Exposure assessment

The global number of cultured fish is in the proportion of 2.5 to 1 with wild populations. But it is probably more than that in locations where fish farm are, even when considering a potential spreading of escapees by 50 kilometres apart the fish farm. A rough calculation leads to around 50 000 wild fish into such an area in average, where an average fish farm rears 1.0 millions individuals. The maximum escapees would be 30 000 in this condition, within the same order of magnitude. This could have effects on displacement of wild population in some specific sites.

Sea bass are cannibalistic, so aggregation of small fish with big ones is unlikely. Escapees will not be subjected to direct predation by wild congeners differently from the wild bass. Being globally bigger than wild bass at the same age, escapees could feed on wild conspecifics more easily, particularly in cases where fish farms are close to nurseries where wild juveniles are regrouping.

Dempster *et al.* (2002) described the aggregation of wild fish around sea cages off the eastern coast of Spain during autumn. It appears that wild sea bass were rare around the cages, being outnumbered by *Sardinella* (round sardinella) and *Boops* (bogue) which are preys for them. Only in case where escapes occurred some months before significant number of sea bass were encountered. This acted towards a low occurrence of escapees mating with wild sea bass around the cages.

Young sea bass are attracted by food and low salinities near the coast, so that they will be close to the farms. There is a high probability for escapees of the same class of age to mix with the wild individuals, but the degree of mixing of subpopulation is not known. If the persistence of the two populations reported by Lemaire *et al.* (2000) is confirmed, the genetic mixing could be low when these fish would maturate. These movements may be equally important in the Atlantic and in the Mediterranean where coastal lagoons may act as feeding reservoirs in the same way than salt marshes do. The major issue would consider the available food of these areas to support increased sea bass population. They will probably migrate in winter to off shore zones.

Hunting behaviour of reared fish is probably not degraded when in the wild at present, the majority of these fish being from the first or second generation from wild breeders. The response to predator is known to be a phenotypic trait under genetic control that may be degraded as a result of selection and domestication on farms, thus decreasing their fitness. Counterbalancing this is the observation that sea bass move in large shoals in sea cages. This behaviour acts in favour of a better survival in the wild, provided that the feed would not be limiting. The fact that reared fish can support overcrowding more easily, with lower stress, acts in their favour when confronted to wild predators.

2.3 Consequence Assessment

The high proportion of males in the cages is acting to decrease the risk for fish in the cage to fertilise the other sex in the wild because the sperm viability in sea water is some 20 seconds while it is some 2–3 minutes for eggs. But to produce bigger fish, only females that grow faster are selected in the farms. So in the same location fish cages supporting maturing females would be isolated from the rest of the males and non maturing females. A solution to avoid those eggs being fertilised by captive males should be to separate them in a sufficient manner.

Increased sea bass fishing have been reported in Sicily in the Regions where sea cages were in activity, whereas it was not in others (Cannizzaro *et al*., 1999). Even if the origin of the fish (wild or cultured) was not investigated, it indicated that these cages could enhance the productivity of these oligotrophic domains and/or act as artificial reefs. This is not contradictory with the observations by Dempster *et al.* (2002). It means that only fish with specific feeding regime adapted to new food resources concentrated around the cages, being possibly future preys for sea bass, presumably wild.

Escaped fish are likely to have a lower survival rates than their conspecific when living in the natural environment because of the unpaired quality of fish coming from hatcheries (lateral line deficiency, sensorial organs, lordosis, loss of fitness, increase fragility of triploids) even if the global quality of hatchery juveniles improves those recent years (Felip *et al*, 1999; Peruzzi and Chatain, 2000; Dosdat *et al*., 2001; Koumoudouros *et al*., 2002).

It appears a small risk of reproduction success between wild and cultured populations in the areas of cage culture, either because wild individuals come to sea cages in few numbers or because cultured individuals remain near the cages for some time after escaping. Nevertheless, this might depend upon the location of these farms with regards to migration patterns. In the absence of long term surveys, the stability of the equilibrium is not demonstrated.

Bahri-Sfar *et al*. (2004) indicated the very great difficulty for Greek fish farmers in the isle of Leros to catch individuals from the eastern population in the environment, even far from fish farms, when individuals from the western population seemed more frequent. The same observation was found in the Bardawill Lagoon (Egypt), where escaped fish originating from the western Mediterranean are suspected having bred with their escaped congeners for generations since 1982. The authors make the hypothesis that some behavioural issues could explain this very low level of introgression of western populations into eastern ones and the maintaining of a western genotype. It could also signify the displacement of local population to the profit of imported ones.

3 Risk Management

3.1 Regulation

Because sea bass stocks are divided into highly localized populations, the use of the local strain for culture purposes is to be recommended until more robust containment technologies dramatically reduce the probability of escapes occurring.

The present approach has a further risk of reduced genetic variability unless a sufficient level of heterozygosity is maintained in the cultured broodstocks used for selective breeding. It is noted that used of sea bass broodstock that have been selected for culture is progressively expanding throughout Europe.

Economic performance of cultured stocks will strongly influence broodstock selection for the mariculture industry. Regulatory based risk management tools must include consideration of the commercial cost-effectiveness and its affects on the ability of a local sea bass culture industry to compete internationally. The EU funded project "Heritabolum" is intending to evaluate the relative performances of these various populations under various production constraints. It would also lead to recommendation for the use of these populations with regards to the environmental interactions.

A more effective long-term solution would involve development of more robust containment technologies. To encourage implementation authorities might consider economic incentives such as reduced annual site licencing fees. Ultimately, taxes on escapees could be also an incentive to make farmers invest additionally in security.

At present, the declaration of sea bass escapees is not compulsory in any country. To reduce uncertainty, the need for regulatory enforcement, and improved mandatory reporting should be introduced. Since there is no additional cost inferred to it, would be profitable to both the industry and the environment.

Pump ashore systems (closed, recirculating, integrated systems) greatly reduces the risk of escapees. Cost effective development of these systems must be encouraged.

The sitting of new sea cages farms should take account of areas important (e.g., feeding and breeding) to local sea bass populations. Implementation of such a policy however, requires fisheries managers invest in getting a better knowledge on these areas, particularly in the eastern Mediterranean where they have apparently been poorly studied.

The use sterile fish to limit both gamete emission and possible interbreeding with the wild is a frequently recommended measure and would significantly reduce the risk of interactions with wild stocks. Sea bass triploids have been produced at pilot scale in France, targeting the increased productivity of sterile fish. It showed a decrease occurrence of breeding, but not totally (98% of success). Sea bass triploids demonstrated they were more fragile than the diploids. This technique for supplying triploid was not economically viable, so the industry did not incorporate them in the production process (in contrast to trout farming where producing of big triploid trout is less costly than producing diploids). Hybridisation with the related species (*D. punctatus*) is another way to investigate. Hybrids have been reported to have a high survival rate, to produce in some case spontaneous triploids, but the sterility of diploid hybrids is not demonstrated yet.

At present, it appears that government based financial incentives might be necessary to introduce sterile sea bass technology. Even that approach will fail if the product is not acceptable to the consumer.

Particular attention should be paid to the movement of fish farms to offshore locations, where weather conditions could induce a more frequent occurrence of containment failure. Effective development requires improved designs for mooring systems and containment technologies.

3.2 Code of practice – Certification

In all cases, the training of the operators should an essential preoccupation by the fish farmer.

The maintenance, replacement and monitoring of nets is of paramount importance to limit accidental escapes. Periodic inspection should be compulsory, and particular attention devoted to net replacement either for cleaning or increasing the mesh size. Particularly, the way the sorting and treatment operations are conducted need improvement and more attention by the producer.

Other aspects of fish farming that might be enhanced in a code of practice for a local industry include:

- Practice for sorting and bath treatments.
- Practice for transport : apply to juvenile supply to the farm as well as to extracting adults at commercial age
- Improved methodologies for net replacements
- Mooring and anchoring particularly when going to more exposed areas
- In the particular case of 3 year age fish, the female should be separated from the rest of the production, mainly composed with males or immature females to avoid any fertilization of their eggs.
- Training of people

3.3 Research

Some research initiatives strongly suggested for their ability to improve our ability to create effective risk management schemes and reduce uncertainty in predicted outcomes include:

- Studies needed to determine the survival of escapees, their migration pattern in relation to their location (e.g. inshore or offshore) and the season they are released (The impact of releases in summer may be different from winter, when sea bass are not feeding intensively but are reproducing).
- Development of tools to distinguish wild fish from escapees
- Better information on the structure and habitat use of wild populations
- Development of offshore systems to reduce interactions with inshore wild populations
- Monitor behaviour of adults and juveniles being released in off shore locations (It would be especially wise to invest in this type of research now, before the cultured stock used by industry has had time to further genetically differentiate from local stocks)
- Photoperiod effect in delaying maturation
- Another possible hypothesis, not specific to sea bass, could be to produce fish that genetically do not synthesized an essential component, they can only find in artificial feed, which make fish not surviving in the wild. However, this solution is highly hypothetic and need substantial theoretical developments (animal welfare, technical feasibility, …), but could also be applied to GMO fish if ever.
- On way to decrease the impact of releases in a given environment is to maximise the wild stocks in areas where farming activities reside, particularly where wild stocks are scarce. This imposes to have the tools to evaluate these stocks available.
- Tools to enable recognising wild from escapees are not really available, and new development are necessary to implement their monitoring.

Contingency Planning

Recovering escaped sea bass within some days/weeks around the cages seems possible and efficient, particularly for adults. Increasing the fishing activities (e.g., by fishermen) after major releases in the vicinity of the farm has proved efficient. Fishing technique could also be adapted (use of specific or illegal devices specifically authorized for these issues). It also rises the problem of the property rights provided that the fish become “res nulla” when they are out of the cage, even if the ground where the fish are caught is rented to the fish farmer. Here a law modification could ease the process.

The degree at which farm should be monitored must be a function of the degree of risk in relation to the farming system. In this respect, there is a decreasing need for intensity from sea ranching to offshore sea cages, inshore sea cages, flow through land based systems, closed systems and integrated systems.

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Sea bream – *Sparus auratus*

1. Hazard Identification

1.1 Life history – Description of wild population

1.1.1 Distribution

Sea bream is present from the Mediterranean Sea to the south of England. It is a marine species than can support desalination, but not as much as the sea bass, and hypersalination. Its thermal preferendum is higher than for sea bass by two degrees. It used to live in coastal lagoons during summer (from 1 to 3 years) and moves to the open sea when temperature decreases in autumn. They usually stay in near-shore areas. Up to the size of 20 mm, they are pelagic, then demersal. Aggregative distribution near the estuaries have been reported in the Mediterranean (Chauvet *et al*, 1992; Sanchez-Lamadrid, 2002) in 0+ fish, when their behaviour is gregarious, but fresh water flooding appeared to impact heavily. It very sensible to oxygen depletion and more generally to water quality.

In the Mediterranean, 1+ juveniles live in near-shore coastal areas (< 30m depth) and in lagoons, moving along the coast for feeding purposes. A tagging experiment by Chauvet *et al*. (1992) in the Gulf of Lion demonstrated that 200 g fish may swim along the coast for 130 kilometres within 130 days. These movements of first year classes present very low offshore amplitude and juveniles stay in shallow areas in open sea. Bauchot *et al*. (1986) stated that they had never been founded by depth more than 30 m. They are not erratic, but oriented northwards. In those movements, all the different populations mix.

Bigger fish appear to winter in rocky areas inshore, but their locations are not well described. Their return to lagoons is decreasing with age (Lasserre, 1976). Fish of more than 4 years are almost absent of these migrations. The adults may be encountered into large shoals close to the coast. In the French Mediterranean, they have been reported to feed on mussel ropes and to date any attempts to investigate about their behaviour failed. Some individuals have been cached by 150 m depth.

Little information is available for Atlantic stocks. From an experiment in the Bay of Cadiz (Sanchez-Lamadrid, 2004), it appears that one year old wild fishes stays in 5–15 m depth areas. Bigger individuals concentrate in deeper areas during winter, and may be caught by more than 100 meter depth.

Overfishing of this species is reported in southern Spain (Sanchez-Lamadrid, 2002). In other locations, no trends during the last ten years may prove that the stock is under threat.

1.1.2 Growth

In the Mediterranean, growth is 20 to 30% higher than in sea bass during the first 2 years. The thermal preferendum for growth is 22–24°, and the growth stops under 13°C. The bay of Cadix and coastal lagoons in the Gulf of Lion have been reported to be nursery areas.

Table A6.2 summarizes the growth (length in cm/weight in g) for females in these two areas (from Lasserre, 1976).

Table A6.2.

Age (year)	**1**	**2**	**3**	**4**	**5**
Wild Atlantic	17/100	26/250	32/400	36/650	42/1000
Wild Mediterranean	19/120	28/310	35/550	41/900	45/1200
Cultured Mediterranean	21/150	32/400	37/700		

1.1.3 Diet

Contrarily to sea bass, sea bream is omnivorous, eating preferably small arthropods and polychaetes in the young stages, then molluscs (bivalves and gastropods), crabs and algae on rocky or sandy bottoms when bigger than 30 cm. Instead of ingesting individuals, it masticates and breaks the prey into small parts. It does not present any hunting behaviour. Juveniles mainly feed in estuaries, *Posidonia* beds and coastal lagoons. Trophic migrations are one of the major driving factors for population mixing.

1.1.4 Abundance

Abundance of this species is not well known, particularly because many landing statistics are mixing all sparid species into one category. An overall landing of 5000 tonnes per year seems to be realistic for the Mediterranean sea (Le Corre, personal communication). Based on the assumption that the fishing pressure on this species is the same than for sea bass, which is far to be demonstrated, the standing stocks should be 25 000 tonnes in this area. This species is catched at any size, from 100 g to 5 kg, and the average weight of the landing is not known.

1.1.5 Reproduction and spawning

Sea bream is a multi-spawning species, which releases eggs during 2 to 3 months. Reproduction period occurs at the end of the autumn (October to December) in the Mediterranean Sea, and at the beginning of Summer in the Northern area of repartition. Sea bream is a protandric hermaphrodite species, that means that during the first 3 or 4 years of his life the individuals are males, and then become females. The first maturation occurs at the age of 2. In the Mediterranean, spawning areas are on shore, by 50 m depth at maximum. In the Gulf of Lion, sea bream are all migrating during the autumn for hundred kilometres to the Rhone's delta to reproduce in zones of 5–25 m depth, at temperature under 19°C (Lasserre? 1976) . In the southern Atlantic (Spain), breeding areas appeared to be by 50 to 100m depth. Eggs are pelagic, but are supposed to be under the halocline in winter (Divanach, 1985) where they are protected from the UV rays.

1.1.6 Genetic structure of the population

Allozyme and microsatellite variations and variation in mitochondrial DNA have been examined in sea bream. Fish from six different origins from Portugal to Greece were analysed. Sea bream from the study by Alarcon *et al* (2004) presented a high degree of genetic variability among wild populations. The reason why such a high variability (2 to 10 times higher than in other sparid species) could be maintained remains unknown. A combination of molecular, demographic and evolutionary factors is suggested by the authors. The partition of this variability using both allozymes and microsatellites showed that most of the genetic variation was within population. This could indicate substantial gene flow from the Eastern Mediterranean to the Azores (Zouros *et al*, 1998) and that structuring pattern is probably not associated with geographic and/or oceanic factors (Palma *et al*, 2001; Alarcon *et al*, 2004). However, the sampling protocols of these studies was probably not comprehensive enough, and additional studies are required to confirm this genetic status.

1.2 Known effect of culture populations

Only one case of intentional release of cultured sea bream is actually reported. It occurred in the southern Atlantic coast of Spain, and in the bay of Cadix (Sanchez-Lamadrid, 2002; 2004). These studies suggest that released fishes stay in the same areas provided that feeding resources under the form of molluscs are available and the salinity remains constant. Survival was better for 100g fish than for 15g ones, due to their higher capacity to adapt. Good growth rates and condition indexes suggested that the behaviour of released fish was adapted to life in the wild. Only in good water quality sites (high oxygen, high salinity, low organic load) and in sites where feed was very abundant fish were recaptured. This suggests that carrying capacity for wild fish could also be altered by released fish. This effect has been reported in Japan (Yamada *et al*, 1992) in a related species (*Pagrus major*). In the Bay of Cadix displacements of one year released fishes were less than 12 km within few months. This scale of dispersal is probably dependant on the areas of release. In these studies, predation by birds is reported to occur intensively on small fish.

Breeding of intentionally released fish with wild populations has been reported (Sanchez-Lamadrid, 2004) one year after they have been released near the coast, 15 km far from there. Released fish were mature and were caught in shoals where they were mixed with wild conspecifics. They presented the same spawning behaviour than wild specimen after one year in the natural environment, proving that gene flows would probably occur between cultured and wild populations.

2 Risk Assessment

2.1 Release assessment

Sea bream are usually reared in the same structure and the same farms than sea bass. So the description from the sea bass section is available, except for the bay of Biscaye where temperature is not high enough for semi intensive culture in earthen ponds. It is currently produced in the Gulf of Aquaba (Israel). Being not cannibalistic, and having a lower dispersion of individual weights within a batch, sorting operation are less frequent than in sea bass. 81 000 tons of sea bream have been produced in 2002, almost exclusively in the Mediterranean countries. To produce this, the standing stock is about 100 000 tons, which means some 450.10^6 individuals being cultivated.

The comparison of wild stock genetic structure with aquaculture stocks demonstrated a low but significant loss of variability among stocks. Effects of domestication, determined by a measure of the heterozygosity, was apparent in some aquaculture stocks. Genetic drift, probably caused by propagation practices, is most likely responsible for the decrease in genetic variation (Palma *et al*, 2001). Mass selection programmes have been developed in France and Israel since that could increase the tendency (Gorshkov *et al*, 2002).

In the case of sea bream, the risk of inbreeding and loss genetic variability in cultured stocks could be enhanced by the hermaphrodite status of the species. Inter-generic breeding have been attempted to produce sterile hybrids between P*agrus* and *Sparus* (Paspatis *et al.,* 1999). Survival rates are poor and the technique has not been diffused to the industry.

No information is available on the number of escapees, and the evaluation made for sea bass is applicable in the same way. Sorting operations being less frequent, the risk for escapees may be lesser in this case. On the contrary, sea bream has a feeding behaviour that make it to crunch its feed using its powerful jaws. Sea bream are thus nibbling the nets to feed on the epifauna, and the occurrence of holes in the cages is greater, imposing a higher maintenance level.

Sea bream are usually sold before the age of 3. It means that all the fish in the farms are males, and the production of fertilised eggs in sea cages is very unlikely, contrarily to sea bass.

2.2 Exposure assessment

There have been no studies carried out on the interaction between wild and cultured sea bream. In the study by Dempster *et al*. (2002), very few sea bream have been reported to be near the sea cages where both sea bream and bass were reared, contrarily to other sparids species (Boops sp., Oblada sp.). This will not facilitate aggregation of wild and cultured fish in these areas. However, based on the known ecology of the species, given the location of fish farms and the aggregative behaviour of year class 1 and year class 2 for feeding, it is highly probable that escapees would mix with their wild conspecific. The experiments by Sanchez-Lamadrid (2001, 2004) in the Atlantic demonstrates this can occur. The risk of displacement of wild populations by escapees, and the competition for feed should be a consequence. Aggressive behaviour have been deported in reared sea bream, in the absence of feed limitation. This could be also the case in the wild, thus acting in the sense of territoriality aggressiveness.

The same author observed escapees mating with wild sea bream in the Atlantic. This is likely to occur in the Mediterranean where aggregative behaviour during the reproduction season has been reported.

2.3 Consequence assessment

All life stages may be impacted by accidental releases of cultured sea bream, either through feed competition or genetic introgression.

Sea bream is not an endangered species and stocks appear robust despite the relatively high fishing pressure on the juveniles. Sea bream is migratory, and a natural genetic partitioning is not demonstrated. A case of high growth rate strain has been reported but not confirmed, which could be due to a genetic and environment interaction. The intra-population genetic diversity appears high. Since breeder stocks are from wild origins, the apparent lack of geographically linked genetic structuring within the species would decrease the risk for adverse genetic interactions from escapees or intentional population displacements.

The level at which the escapees could displace natural populations or shorten the feed supply is not known and cannot be derived from existing studies. It only can be said that it more likely to occur during the period of high feeding intensity, i.e., spring and summer. Released sea bream presenting the same feeding behaviour than the wild ones, it is quite clear that it will essentially depend upon the relative numerical abundance in a given area of wild and released fish. The degree at which the fitness of cultured fish in the wild is questionable, but there are some evidence that it could be lees than wild congeners due to some physiological deficiencies (dorsal deformations, olfactory abnormalities, etc…) that have been currently reported (Mana and Kawamura, 2002).

3 Risk Management

3.1 Regulation

Because sea bream stocks are not differentiated into localized populations, the use of the local strain for culture purposes may be not recommended. Nevertheless, the risk coming from of reduced genetic variability when a sufficient level of heterozygosity is not maintained in the cultured broodstocks used for selective breeding has to be considered. The level of impairment induced by consanguinity is not known in this species. It is noted that used of sea bream broodstock that have been selected for culture is progressively expanding throughout Europe. Caution is to be taken to avoid dispatching of strains genetically far from their parents.

An effective long-term solution would involve development of more robust containment technologies. To encourage implementation authorities might consider economic incentives such as reduced annual site licensing fees. Ultimately, taxes on escapees could be also an incentive to make farmers invest additionally in security.

At present, the declaration of sea bream escapees is not compulsory in any country. To reduce uncertainty, the need for regulatory enforcement, and improved mandatory reporting should be introduced. Since there is no additional cost inferred to it, would be profitable to both the industry and the environment.

Pump ashore systems (closed, recirculating, integrated systems) greatly reduces the risk of escapees. Cost effective development of these systems must be encouraged.

The sitting of new sea cages farms should take account of areas important (e.g. feeding and breeding) to local sea bream populations. Implementation of such a policy however, requires fisheries managers invest in getting a better knowledge on these areas, particularly in the eastern Mediterranean and the Atlantic where they have apparently been poorly studied.

The use sterile fish to limit both gamete emission and possible interbreeding with the wild is a frequently recommended measure and would significantly reduce the risk of interactions with wild stocks. Sea bream triploids have been produced by some fish farm, targeting the locking of the genetic progress gain through selective breeding. The results are not public. Hybridisation with the related species or genus (*Pagrus pagrus, Pagrus major, Dentex dentex*) is another way to investigate. Hybrids have been reported to have lower survival rates, but the sterility of diploid hybrids is not demonstrated yet. See Colombo works on hybrids.

At present, it appears that government based financial incentives might be necessary to introduce sterile technology. Even that approach will fail if the product is not acceptable to the consumer.

Particular attention should be paid to the movement of fish farms to offshore locations, where weather conditions could induce a more frequent occurrence of containment failure. Effective development requires improved designs for mooring systems and containment technologies.

3.2 Code of practice – Certification

In all cases, the training of the operators should an essential preoccupation by the fish farmer.

The maintenance, replacement and monitoring of nets is of paramount importance to limit accidental escapes, particularly in case of a nibbling species. Periodic inspection should be compulsory, and particular attention devoted to net replacement either for cleaning or increasing the mesh size. Being not cannibalistic, the sorting frequency is not high. This species is not very susceptible to diseases, so the risks associated with fish manipulation is lower than in sea bass. Being omnivorous, sea bream are cleaning the net by eating spontaneously the bio-fouling.

Other aspects of fish farming that might be enhanced in a code of practice for a local industry include:

- Practice for sorting and bath treatments;
- Practice for transport : apply to juvenile supply to the farm as well as to extracting adults at commercial age;
- Improved methodologies for net replacements;
- Mooring and anchoring particularly when going to more exposed areas;
- Training of people.

3.3 Research

Some research initiatives strongly suggested for their ability to improve our ability to create effective risk management schemes and reduce uncertainty in predicted outcomes include:

- Studies needed to determine the survival of escapees, their migration pattern in relation to their location (e.g. inshore or offshore) and the season they are released (The impact of releases in summer may be different from winter, when sea bass are not feeding intensively but are reproducing).
- Development of tools to distinguish wild fish from escapees
- Better information on the structure and habitat use of wild populations
- Development of offshore systems to reduce interactions with inshore wild populations
- Monitor behaviour of adults and juveniles being released in off shore locations (It would be especially wise to invest in this type of research now, before the cultured stock used by industry has had time to further genetically differentiate from local stocks)
- Photoperiod effect in delaying maturation
- Another possible hypothesis, not specific to sea bream, could be to produce fish that genetically do not synthesized an essential component, they can only find in artificial feed, which make fish not surviving in the wild. However, this solution is highly hypothetic and need substantial theoretical developments (animal welfare, technical feasibility, …) but could also be applied to GMO fish if ever.
- On way to decrease the impact of releases in a given environment is to maximise the wild stocks in areas where farming activities reside, particularly where wild stocks are scarce. This imposes to have the tools to evaluate these stocks available.
- Tools to enable recognising wild from escapees are not really available, and new development are necessary to implement their monitoring.

3.4 Contingency Planning

Recovering escaped sea bream within some days/weeks around the cages seems hard to implement in this species, particularly when adults are released. Fishing technique could be developed (use of specific or illegal devices specifically authorized for these issues).

The degree at which farm should be monitored must be a function of the degree of risk in relation to the farming system. In this respect there is a decreasing need for intensity from sea ranching to offshore sea cages, inshore sea cages, flow through land based systems, closed systems and integrated systems.

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Cod (*Gadus morhua*)

1 Hazard Identification

1.1 Life History of Wild Populations

1.1.1 Distribution and movements

Currently, the main areas where the aquaculture industry is actively engaged in seeking to develop cod farming are Scotland, Norway and Ireland.

In the waters to the west of Scotland, there is a continuous distribution of cod eggs and larvae around the west and north coasts during the spawning season. Juvenile cod during their first year are found close inshore or around the mouths of sea lochs. Recent recruits to the adult cod population are widely distributed on the west coast of Scotland, mainly in offshore areas where they can occur in large shoals.

To the east of the UK, after hatching at a length of about 0.4 cm length, the young fish grow to between 2 and 8 cm by June, and are concentrated mainly in the eastern and northern parts of the North Sea. By the following winter, the young fish are between 13 cm and 26 cm in length and are concentrated in the shallow coastal waters of the eastern

North Sea. One and 2 year old cod can be found all over the North Sea, although by age 3 they are distributed mainly towards the northern part of the North Sea (CEFAS).

At the moment there is very little conclusive information on cod nursery areas. The general feeling at the moment is that juvenile cod prefer rocky inshore areas. However, they have also been found on offshore gravel banks (CEFAS) in the southern North Sea and sand banks off West Coast Scotland (METACOD and METAGADOID projects). These projects are on going and should have more to report in 1–2 years. The Clyde has also been identified as a preferred area for juvenile cod.

There is now evidence that there are least three reproductively isolated populations of cod in the North Sea and that there may be further isolated populations in the northwest Atlantic and Norway. The amount of information is limited and but the FP5 project (METACOD) will investigate this issue. From evidence of NW Atlantic stocks we might expect that the different reproductive units might intermix to some extent during the summer. In Icelandic waters, most mature cod are found in the north and east in waters between 100–300m deep.

There is some understanding of the movements of cod to the west of Scotland. In late summer, cod move from west of Hebrides to North coast. In late winter and early spring they reverse this movement. There is little movement between the Hebrides area and the North Sea. There is information to indicate that cod migrate along clines of preferred ambient temperatures (Rose, 1993)

Eggs and larvae are dispersed by currents until the young cod move onshore in the spring where they feed and grow in shallow waters for the first year. Cod reach maturity as 2 year old fish but do not begin to spawn until they are at least 4 years old. All are spawning by 6 years of age. Spawning individuals migrate to a number of distinct spawning areas within each of the metapopulation ranges. Non-spawning adult populations can be either migratory or resident. Results from tagging experiments show that there is a little interchange of cod between the North Sea and West of Scotland, but that there is much more exchange between the Eastern Channel (VIId) and the Skagerrak (IIIa). Tagging studies carried out over several decades have also shown that the maximum distance travelled from the release point is about 200 miles; but a few long-distance migrations have been recorded. In one experiment in June 1957, when cod were released in the central North Sea, two fish were recaptured off the Faroe Islands in September 1957 and one fish was recaptured off Newfoundland in December 1961.

FRS tagging data show that there is little exchange between Firth of Clyde cod and those in the Minch, particularly in the North Minch, north of Skye. Cod from the Minch have been caught north of Scotland but there is little apparent exchange between Minch cod and cod in the Moray Firth (NW North Sea).

NEED INFO ON NORWAY STOCKS

There are currently two international projects (METACOD and CODYSSEY) looking at cod migration patterns, as well as research projects at the Norwegian Institute of Marine Research. CEFAS have collaborated with eight other research laboratories from Norway, Sweden, Scotland, Germany, Denmark, Iceland and the Faeroe Islands with the CODYSSEY project. CODYSSEY is an EU-funded R&D project into the movements and behaviour of cod in the NE Atlantic. This will attempt to define characteristic behaviours and migrations of cod in four different ecosystems: the North Sea, the Baltic Sea, the Barents Sea and the Icelandic/ Faroe plateau.

Whether there will be an impact from escapes from fish farms will depend on the exact nature of the population structure in the wild stock and the genetic nature of the farmed stock of the same species (WGEIM 2003 Report).

1.1.2 Growth and Mortality

Under typical growth rates in Scottish waters, wild cod will reach 20 cm after 1 year, 50 cm after 2 years, and 80 cm after 4 years. Data on growth rates of farmed cod transferred to net pens in Scotland at an average weight of 5 g in July are summarised below:

Date	Average weight (g)
July – 1st year	5
October – 1st year	40
December – 1st year	120
February – 2nd year	230
April – 2nd year	350
December – 2nd year	2000
December – 3rd year	3500

A growth trial in net pens carried out on wild cod captured from Bay Bulls in Newfoundland, showed that when cod fed on either capelin or two different types of formulated wet diets, fish grew on average between 33–34% over a three-month period of the trial (Clark, 1995).

Predation mortality of cod eggs is predominantly from sprat and herring, as well as juvenile and adult cod cannibalism. The survivability of settling larvae has been linked in many studies to the complexity of the seabed, and is one of the targets of the METACOD project.

Most mortality occurs during the juvenile stages A significant portion of the mortality can be due to starvation and cannibalism by older cod, as well as predation by other piscivores. Not surprisingly therefore, different age classes of cod do not aggregate together. After about one year's growth, young cod (in Scotland, at ~20cm length) generally move offshore to feed where they become susceptible to increased fishing pressure prior to recruiting to the spawning stock.

Most cod stocks in the North Atlantic are below the ICES precautionary level, and in some ICES areas there is a moratorium on cod fisheries. Many of these populations have been in decline for more than a decade, and as the metapopulation shrinks and can no longer support all its sub-populations, fisheries have witnessed the disappearance of some local cod populations.

ADD TEXT ON FISHING MORTALITY

1.1.3 Diet

In a study off the west coast of Sweden, Stefan (1990) reported that cod ranging in size from 6 to 97 cm fed at 40–90 m depths. Diets consisted mostly of benthic and epibenthic species (Stefan, 1990), with 75% crustaceans and fish. At larger sizes, the proportions of benthic species to copepods increases with size. Young cod up to 1–3 cm size feed exclusively in the water column on copepods, then at 4–6 cm sizes add benthic prey species such as mysids and amphipods, but copepods remain an important food item. Large cod also consume molluscs, worms and smaller fish.

Juvenile cod are preyed upon by larger piscivorous fish, including larger cod, seals and cetaceans and birds. The proportion of each of the prey types has been shown to vary from year to year. Cannibalism is a large part of predator-prey relations, with larger 0-group cod and older cod consuming smaller ones. Stomach content surveys seem to be most comprehensive in the Baltic Sea. Studies from Newfoundland corroborate these findings. Seals are a significant predator of adult cod; 82% of seal diet in Northern Scotland made up of fish, with 50% sandeels and cod also important prey items. A Canadian study also found that grey seal predation caused 10–20% of mortality in cod stocks.

NEEDS NORTH SEA AND WEST COAST OF SCOTLAND DATA

1.1.4 Abundance

Cod stocks around Scotland are under severe fishing pressure. Spawning stock levels for both the North Sea and West Coast stocks are below safe biological limits. Stocks have been below ICES precautionary levels since 1988. ICES advised the European Commission and national governments that all fisheries which target cod, even as a bycatch, in the North Sea, Skagerrak, Irish Sea and waters west of Scotland should be closed (ICES Advisory Committee on Fishery Management [ACFM] 2002). Around Iceland, there has been low spawning stock biomass and weak recruitment since the mid-1980s.

The combined average landings of wild cod in the waters off Ireland and UK have plummeted from 75,000 tons per annum to less than 25,000 tons since the mid-1990s (Marine Institute, Stock Book 2001). The ICES ACFM report for 2003 estimates that the spawning stock biomass of cod to the west of Scotland in 2002 was 2230 tons, with 3,000,000 individuals recruiting at age 1. The spawning stock biomass in the North Sea, English Channel and Skagerrak combined was 54400 tons, with 168,000,000 recruits at age 1.

NEED NORWEGIAN COASTAL STOCK DATA/STATUS

1.1.5 Reproduction and spawning

Adult male and female cod form pair bonds, but egg fertilization is external. Females are batch spawners often producing 15 egg batches over a period of six weeks. Around Scotland, cod may reach maturity at 2 years of age, but do not spawn until 4 years old. At age 6, all fish are mature. However, most fish are caught in the fishery by the time they are age 2.

Data taken from the ICES International Bottom Trawl Surveys, two EU funded projects (STEREO 1999; METACOD 2002, 2003), and ichthyoplankton surveys and responses to questionnaires taken from fishermen have found that cod spawn throughout much of the North Sea, although some spawning aggregations do occur. The main spawning areas in the North Sea are in the central North Sea around the Dogger Bank, the southern North Sea, and the German Bight. There is also a center of spawning in the NW North Sea in the Moray Firth (CEFAS). The EU projects are producing much useful information, and the FRS has produced a report on North Sea spawning grounds.

The timing of spawning is well documented as being between January and April, with the more northern areas spawning later than the more southern areas. Eggs, which are about 1.4 mm in diameter, are found floating in the surface layers over large areas of the North Sea. They typically hatch over a period of 11–30 days, depending on water temperature. Cod juveniles live in upper water column until around August before settling down to a demersal life style, driven mainly by changes in food requirements from predominantly copepods to benthic species. *C. finmarchus* are the staple prey of first feeding larvae of Atlantic cod.

Spawning aggregations also appear to occur in the Irish Sea and off the NW coast of Scotland. Spawning on the West Coast takes place between January and April, mainly in offshore areas. One adult female can produce around 4 million eggs (depending on size) per season. Development time for cod eggs is 11–30 days in NE Atlantic depending on water temperature. Larvae hatch in the early spring and the live in the upper water column till August when they take up a demersal live style.

In Iceland, mature cod in the spawning period were typically found in waters over 300 m in depth, indicating that spawning normally occurs offshore (Begg and Marteinsdottir, 2002).

NEED NORWAY DATA

A Canadian study on variation in size-specific fecundity of cod sampled from the Gulf of St. Lawrence and the Georges Bank indicated significant variation that could not be attributed to physiological conditions (McIntyre and Hutchings, 2003).

1.1.6 Genetic structure of wild populations

To evaluate the potential effects of cultured cod on wild populations, the structure and variability of wild cod populations must be understood. Smedbol and Wroblewski (2002) have described cod population genetics as "metapopulations". A metapopulation is a set of distinct, local populations within some larger area where movement from one population to another is possible. There is an ongoing debate about the large scale and small scale structure of cod populations. The metapopulation structure incorporates concepts of discrete local breeding populations connected by immigration and emigration. Depending on factors such as the distance between areas occupied, geographic or oceanic barriers, and the dispersive ability of the species, the degree of segregation between subpopulations can range from slight to almost complete isolation. However, exchange between subpopulations of the metapopulation prevents the development of separate autonomous populations. Begg and Marteindottir (2002) typify a cod metapopulation as a composite of local populations (i.e., spawning components) between which individuals move, and where 'source' populations provide immigrants to less productive 'sink' populations.

Recent analyses assume that there are the equivalent of at least 3 metapopulations of cod; one in the NE Atlantic, one in the NW Atlantic, and one between the other two (Imsland and Jónsdóttir 2003). The METACOD project will be looking at genetic differentiation between cod stocks by the analysis of microsatellite DNA and S*yp*I. and others. Studies from the Danish Institute for Fisheries Research have also looked at genetic differentiation in cod stocks. Preliminary information from Ireland suggests that sub-populations of cod can be identified in coastal waters. Begg and Marteindottir (2002) identified a metapopulation of cod in Iceland, with regional spawning populations located around the country being interconnected by dispersal and migration. The main spawning component of the metapopulation was found off the SW coast of Iceland.

NEED NORWAY DATA

1.2 Known Effects of Cultured Populations

The most recent review on enhancement of marine stocks, including pelagic and bottom-dwelling finfish and crustacean species, has been prepared by Blaxter (2000). Problems discussed by Blaxter include (a) the viability of released fry (quality of seed), (b) survival after release and releasing strategy, (c) carrying capacity in relation to the size of released stock and interactions with the receiving ecosystem, and (d) the impact on wild stocks.

Historically, cod stock enhancement occurred in Norway, Sweden, Denmark, Faroe Islands and North America. Svåsand *et al.* (2000) have reviewed the effects of these attempts to supplement wild stocks with cultured cod. Releases have involved fish between 8 and 41 cm in length. (wild cod in Scotland are ~20 cm long at year 1 and 50 cm at year 2). The numbers of fish released are relatively small, and have varied between 500 and approximately 400,000 fish.

From intentional release studies, survivability of released cod is highly dependent on the age and size at release. The average rate of mortality of released yolk-sac larvae in Norway was 23% per day during the first 10 days, with only 0.15 % surviving the first 40 days after release. The optimal timing for release is generally after they have reached the size at which they settle to the benthos.

Studies from Norway suggest that released reared cod have a variable fidelity to an area. Fish from one resident, southern coastal population were fairly stationary when released, with more than 80% of fish recaptured within 5 km of the release site, and no more than 5% dispersing more than 10 km. Reared fish from another northern population had only 45% recaptured within 10 km of the release site. In Denmark, 72% of recaptures were taken within 40 km of the site of release. In the Faroes more than 50% of the recaptures occurred within 10 km of the release site. On this scale of dispersal (within 50 km of release), Svåsand *et al.* (2000) stressed that results obtained in one area cannot be generalized to other area.

To see an impact on environmental carrying capacity for wild stocks, the addition of escaped reared fish would have to reduce the amount of resources available for the wild stocks. Given the low abundance of stocks over more than a decade and the nature of the metapopulation structure of cod populations, it is likely to be very difficult to detect carrying capacity effects at the metapopulation level. With the potential movement of individuals between sub-populations, it may also be difficult to detect carrying capacity constraints at the sub-population level; but if it were detectable it is most likely to be evident at the subpopulation level. Differential growth and mortality may be indicative of this type of effect. It is possible that genetic or other markers of subpopulations may become available that would allow an analytical approach to determination of the effects of escapes on sub-populations. It has been suggested (FRS, pers. comm.) that it may be possible to trace escaped fish back to the farm of origin by either using molecular markers or the analysis of otolith morphology, as it is very likely that these will show farm-specific patterns (FRS, pers comm.).

Jørstad and Naevdal (1992) and Jørstad (1994) reported on an extensive investigation of the effects of mass rearing and release of 0-group cod in fjords and coastal areas of Norway. Each year since 1987, pond produced cod have been liberated in Masfjorden, a small fjord north of Bergen. The released cod as well as the wild fish and those recaptured in the fjord system have been genetically characterized by electrophoretic analyses of haemoglobin and several enzymes. In 1990 and 1991 about half of the released cod consisted of offspring of broodstock homozygous for a rare allele (Pgi-1(30). This broodstock was produced by crossing pre-selected heterozygotes for this allele, the homozygotes among the offspring were sorted out on the basis of biopsy sampling of muscle tissue, and when matured, used as parents. Genetic tagging seems to be a useful method both for studying factors controlling survival, growth rate and dispersal of released cod (Jørstad, 1994), and also for more long-term studies on hybridization and gene introgression between natural and reared populations.

Svasand (1993) looked at behavioural differences between reared, released and wild juvenile cod, using Floy anchor tags and oxytretracycline markers. While differences in individual behavior patterns occur, no differences in migration patterns between wild and reared specimens have been demonstrated.

Nordeide and Salvanes (1991) compared the stomach contents and liver weights of reared, newly released cod and wild cod; the stomach contents and abundance of potential predators were also described. During the first three days after release, the reared cod fed mainly on non-evasive prey of Gastropoda, Bivalves, and Actinaria. This is in contrast to wild juvenile cod, which mainly fed on Gobidae, Brachyura, and Mysidacea. Large cod, pollock, and ling preyed upon the released cod immediately after their release whereas during the months following release the stomach contents of large predators were dominated by Labridae and Salmonidae, which are also the typical prey of wild cod. The abundance of predators did not seem to increase within the area of release. However, a study of Svåsand and Kristiansen (1985) found no difference in dietary composition of cod after five months post-release. This suggests that although the foraging behaviour of newly released cod is poorer than wild conspecifics, they adopt similar feeding behaviour to wild fish within 5 months after release.

Svåsand *et al.* (2000) reviewed studies on the ecosystem level effects of large scale releases of reared cod in the Masfjorden and Troms areas of Norway. The Masfjorden studies involved a control fjord and an experimental fjord into which large numbers of reared cod were released. Both sites were monitored before and after the release to detect potential interactions between released cod, its predators (large cod, pollack) and competitors (poor cod), and population characteristics (abundance, growth, condition factor, liver index). The abundance of selected prey species was also monitored. Only minor effects could be ascribed to the releases of cod (Fosså *et al*., 1994). Recent unpublished data on the poor cod suggests a reduction in size in the experimental area, but not in the control area. For wild cod however, there was a slight reduction in condition factor and liver index. Higher densities in the experimental fjord became undetectable within 1.5 years. Data suggest that reared cod suffered higher mortality than the wild cod.

In the Troms area experiment, releases did not increase the biomass of cod in the fjord, nor did they reduce prey abundance. A strong year class at that time was believed to have lowered growth rates and may have has an effect on the ecosystem similar to an average year class enhanced by released fish.

Extensive genetic studies and monitoring were carried out as part of studies in Masenfjorden and Øygarden for both the released and wild cod. Except for the enzyme GPI, fish did not differ. Patterns of change associated with the GPI frequencies were attributed to genetic drift rather than local adaptation.

Otterlind (1985) has reviewed the literature and reported on the occurrence and migratory habits of Baltic cod based on experiences since the 1950s, with results from extensive tagging trials combined with information on changes in allele frequency for haemoglobin types, meristic characters and otolith types. About 15 transplantation experiments with tagged cod assessed the potential homing ability of the fish. Waters west of Bomholm constitute an area of hydrographic instability with varying cod migrations and passive transport by currents of fry. Migration east of Bornholm refers – except for local stocks and a varying contribution from the west, mainly to fish raised in the central Baltic and northern areas. Fish in the latter group migrate primarily southward for spawning; as adults they usually stay east and north of Bornholm. Results of the transplantation experiments support a strong linkage between cod migration and hydrographic factors. Cod tagged and transplanted to a new area behaved and moved in the same way as the local stock. Indications of "homing" can be found in areas with suitable hydrographic gradients, such as changes in salinity, for example, in Oresund. This information is useful to assess potential risks of impacts of escapes within each of the identified separate Baltic cod stock components.

An overview of stocking and enhancement programs performed along the coasts of North America has been compiled by Richards and Edwards (1986). No considerations were given in this review to the potential impact of these releases on natural ecosystems. Further references relating to cultured and wild cod interactions include Jørstad et al (1994a, 1994b) and Kitada et al (1992). The latter studied the effectiveness of fish stock enhancement programs using a two-stage random sampling survey of commercial landings for cod and flounder.

2. Risk Assessment

2.1 Release Assessment

The inability to reliably produce cod fry for aquaculture has been a significant constraint on the development of the industry. In 2002, a breakthrough in the production of cod fry occurred in Norway when ~3 million fry were produced. In addition, survival rates of 87% from hatching to 0.2 g were reported in one hatchery in Scotland. These recent success stories are due to improved knowledge and an increased number of enterprises. A production target of 10 million fry is expected in the next few years, which will be followed by a subsequent substantial increase in production. As can be seen from Figure A6.4, intensive fry production is the dominant production method.

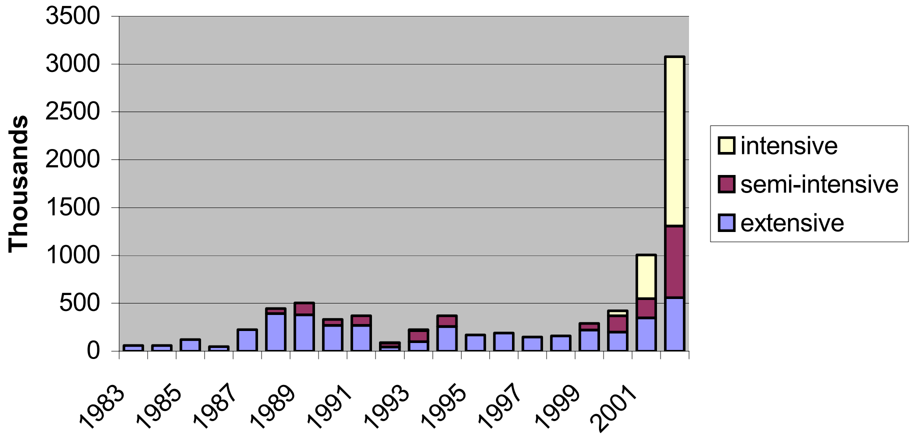

Figure A6.4. Total production of cod fry in Norway 1983–2002 (Karlsen and Adoff, 2003).

Fry production in other countries is less developed. In Scotland, around 50,000 juveniles were stocked in 2002, with 15 tons of cod produced in 2000 and 2001. More than 100 tons of production is predicted in 2003. In Ireland, a research fellowship is in place to identify and harness potentially exploitable research and technology so as to enable the establishment of a commercially viable cod hatchery in Ireland. The primary deliverable is a detailed methodology, developed in collaboration with industry, for the hatchery production of juvenile cod in commercial conditions. The work is being carried out at MRI Carna Laboratories, National University of Ireland, Co Galway.

Many farms currently growing cod utilize the same cage types used for salmon farming, namely, circular PVC cages ("polar circles") and galvanised steel cages. Based on concerns that cod feed on epifauna growing on net enclosures, double netting is commonly employed. There is a need for additional research with regards to rearing technology, feed developments, and the prevention of maturation.

Cod culture in sea cages is currently confined to relatively sheltered inshore areas, compared to salmon culture. The siting, distribution and position of farms "licensed" to hold cod will be held by the regulatory bodies in each ICES member country (e.g., FRS/SEPA in Scotland and The Ministry of Fisheries in Norway). From the FRS (Scotland) database, 21 out of 483 registered farms have multi-species licenses and therefore have the potential to stock cod. However, it is unlikely that all sites will use this option. At present (2004), only 12 sea sites (4 in Shetland, 3 in Southwest Scotland, 3 in northwest Scotland, 1 in Orkney, 1 in Skye), and about 11 pump-ashore sites, mostly hatcheries (6 in Shetland, 1 in Orkney, 4 in Southwest Scotland) are actively cultivating cod. No aquaculture licenses have yet been issued in Ireland, although several applications are being evaluated. Cod reared in pump ashore facilities, particularly those employing discharge water treatment (filtration and sterilization) pose a negligible risk in terms of fish escapes.

FAO data show that the production of farmed cod in 2001 occurred in Norway (608 tons), UK (15 tons), and Iceland (140 tons). Recently, it was predicted (John Goodlad, Buckland Lecture) that cod production may increase from 6000 tons in 2003, to 200,000 tons in 2010, and 400,000 tons in 2020, mostly in Norway. Predictions for Scotland suggest 25,000 tons will be produced by 2012–2014. This dramatic increase in cod farming will inevitably lead to an increased risk of escapes.

Rearing trials suggest that sites with water currents in excess of one metre per second are unsuitable for growing cod. Consequently, cod farms will tend to be located in less exposed locations, in terms of both tidal currents and wave action, and thus the risks associated with storm damage will be less than those for salmon (assuming engineering comparability of equipment).

There have been no reported escapes of farmed cod in Scotland to date; however, extensive information is available on rate of escapes from Scottish salmon farms, due to compulsory notification of escapes (Registration of Shellfish and Fish farming business and Registration Order 1985). This is also compulsory in Ireland and Norway. Over the last 5 years, there have been 20–25 escapes per year from Scottish fish farms, mostly from Atlantic salmon farms.

Rates of escapes of salmon in Scotland have been between 94,000 and 500,000 adult fish (1–4 kg) per year. The main causes of escapes from salmon farms have been: human error, equipment failure, bad weather and predator attacks. These factors could also be considered the main areas of risk with regard to cod farming with some modifications:

- The generally sheltered location of the cod farms at present would lessen the risks of storm damage, but shelter could increase the risk of predator (e.g. seal) attacks.
- Human error and equipment failure could probably be regarded has having similar levels of risk as salmon farming.
- "Nibbling" of nets does not appear to be a significant factor with cod (Scottish Executive Working Group on Escapes).
- Unlike salmon, cod shoal rather than school, so the motivation for a contained cod to follow an escaping cod is less than it would be for salmon in similar circumstances.
- Cod can be transferred to sea pens at weights above 5 g, whereas the minimum weight at transfer of salmon smolts to sea is typically 35 g. The risk of escape through minor holes in the net is consequently greater for juvenile cod

2.2 Exposure Assessment

Genetic interactions between farmed and wild salmonids are dependent on escapes of fish from holding facilities. Cod pose the additional risk of continuous spawning at sea (usually January to June, depending on area, and if photoperiod manipulation is employed). This could be exacerbated by the fact that to remain competitive with wild fisheries, farms specializing in the high value niche market for larger cod would have large, mature fish in their systems. Studies have shown that cage reared cod will spawn concurrently with wild cod in the same region.

Most adult cod stocks do not frequent shallow, coastal waters typical of the technologies presently used in the salmon industry, and are likely to form the basis for the cod farming industry. As such, direct interaction between caged and wild adults will be limited.

Wild, juvenile cod are known to occupy areas where cobbles and kelp can be used for predator evasion and have diverse feeding opportunities. Eelgrass beds are also known to be important nursery areas. Escapes in these areas would therefore have a high probability of interacting with wild juvenile cod.

In the event of escapes, the age of the released cod may determine how they fit into the marine ecosystem. For instance, wild juveniles typically establish schools in inshore, shallow water areas, while adults are found in deeper, more oceanic areas. Escaped cod may follow this migration pattern. Escaped juvenile fish may therefore join conspecifics of similar size in inshore waters. Where mature fish escape at the appropriate time of year, they may migrate and breed with wild populations.

Conversely, juvenile wild cod may enter cages and be exposed to predation during their first year when they have a pelagic life style, but after that it is unlikely that they will be exposed to predation by caged fish. However, the numbers of juveniles lost in this way may not be significant, as juveniles are known not to inhabit the same area as older cod (perhaps to avoid cannibalism) and may therefore actively avoid older cod in cages.

It is not known if adaptation to local environments exists in marine fishes like cod, but if it does, such adaptation will depend on the degree of isolation from other conspecifics. The Danish Institute of Fisheries Research will study the abilities for local adaptations in marine fishes, which could give more information on escapee cod.

Identification of areas where cod culture is likely to occur should be straightforward, utilizing records from regulatory bodies in member countries. At least in the first instance, cod farming is likely to occupy the same general areas of coastal waters as salmon farming. Some competition for space may occur, but as farmers seek to grow cod experimentally at established salmon farms, little additional capital will be required to establish cod farming initially.

2.3 Consequence Assessment

Blaxter (2000) concluded in a review that "unless a small wild population is swamped by large-scale releases (or stocking) of reared fish, it seems unlikely that the reared fish will out-compete the wild fish". However, the accidental release of fish from culture sites will potentially impact local populations by affecting all life-cycle stages and exposing wild fish to feed competition and behavioral stress. In particular, this interaction would occur when territorial behaviour is a key component controlling population density in a given habitat.

It could be envisaged that the impact of escapes would be minimal if farming involved the rearing of wild-caught juveniles from a local stock which was widespread and abundant, and showed a regional rather than local population structure. On the other hand, a significant impact could occur if farmed fish were of non-local origin with a narrow genetic base (i.e., a high degree of inbreeding), and fish escaped and mixed with a highly structured stock with a restricted local population. Since cod are now very scarce in many inshore waters (such as Scotland), the impacts on wild cod populations may be highest if large numbers of non-native farmed stocks escape to depleted local stocks.

Whether cultured for all of their life cycle or only part of it, cultured fish face different selective pressures and a different "learning environment" when compared with wild populations. Consequently, cultured cod will ultimately express different genetic, phenotypic and behavioral traits than wild cod. The critical question is how significant what will these differences be, and to what degree will they impact wild populations when cultured and wild populations interact. Though experience with cod culture (as an enhancement activity) dates back to the middle of 1800s, actual investigations into the differences between wild and cultured cod are derived from more recent studies in the 1980s 90s. Our knowledge is also limited by a number of factors including the short time cod have been under continuous selection for culture; the incomplete knowledge of the genetic structure of wild and cultured cod populations; and the fact that, both in culture and in the wild, the selective pressures on the cod genome are constantly changing.

The potential for inter-species hybridisation involving escaped farmed cod is not thought to be a problem (FRS, unpub.). An extensive e-journals literature search found no reference to any reference on cod hybridisation. However, experiences with salmon suggest that further research may be required. Youngson *et al.* (1993) have identified what is likely a behavioral deficiency in escaped farmed salmon that has led to increased levels of hybridization with brown trout. Such hybridization was found to be ten times more frequent among escaped farmed than wild Atlantic salmon females (WGEIM 2003). Effects of any interbreeding will depend on the genetic (and numerical differences) between cultured and wild stocks. The significance of this will be small, regardless of the numbers involved, as long as the genetic differences between wild and cultured are small. Selection over time inevitably creates larger genetic differences, so impacts will depend on the ratio between the numbers of wild to escaped fish in a population. In addition, the possibility of these escaped fish contributing to the recovery of wild cod stocks will need to be investigated more seriously. More recent studies on NE Atlantic cod have been conducted by Dr T Svåsand from the Institute of Marine Research in Norway, including comparisons of wild and cultured cod in regards to behaviour, migration patterns, stomach contents, and growth. Methods and efficiency of feeding methods have been shown to be different in wild and reared cod, with the wild cod generally out-competing reared cod. Therefore, escaped fish are likely to have lower survival rates than wild conspecifics.

Cod milt and eggs are known to survive for a relatively long time after release, and fertilization of eggs can occur upwards of 60 minutes after release. Gametes from wild cod outside net pens could therefore potentially interact with gametes produced by farmed fish inside the cages.

The current trend among start-up cod hatcheries in EU countries is to source either eggs or broodstock from established farms that are certified disease free, minimizing risks associated with introducing wild cod of indeterminate health status. Consequently, the practice of introducing non-indigenous cod may increase and accelerate the rate of genetic divergence between farmed and wild cod stocks.

3 Risk Management

Whether there will be an impact from escapes from fish farms will depend on the exact nature of the population structure in the wild stock and the genetic nature of the farmed stock. For instance, it could be envisaged that the impact of escapes would be minimal if farming involved the rearing of wild-caught juveniles from a local stock which was widespread and abundant, and showed a regional rather than local population structure. This would be true even if escapes involved relatively large numbers of fish. On the other hand, a significant impact could occur if the farmed stock were a variety of non-local origin with a narrow genetic base (i.e., a high degree of inbreeding), and it escaped and mixed with a highly structured stock with a restricted local population.

This risk of release of genetic material from farms (either as gametes or as escaped fish) could be minimized by harvesting fish before they reach maturity; using sterile fish; or using pump-ashore sites where the effluent water can be filtered or sterilized. Use of sterile fish on cod farms would eliminate any possibility of genetic interaction with wild stocks. The use of triploid fish has been investigated in salmon culture; however the cost and low efficiency could be problems. More research into other methods of producing sterile fish is required. Studies at the University of St Andrews, Memorial University of Newfoundland, The Institute of Aquaculture at Stirling University, and the Institute of Marine Research in Sweden are investigating photoperiod control of maturation in cod. The British Marine Finfish Association website reports that “Recent research has shown that continuous light can delay sexual maturation and improve growth, making the utilization of photoperiod manipulation a viable option”. This suggests that husbandry practices could significantly reduce the risk of release of viable gametes.

Recently, there has been considerable interest and action concerning the possibility of recapture of escaped salmon, since escaped salmon tend to remain in the area of the cages for some time after escapement. This is thought to be due to their tendency for schooling behavior, and imprinting on artificial “prey” (i.e., feed pellets). The potential to recapture escaped cod has not been analyzed; but is an important area for research. It is also important to discuss this with the public early in a development program, and to derive the risk management triggers and contingency plans in an open and transparent manner, for each area where a wild cod sub-population can be identified.

4 Risk Mitigation

Genetic studies of cod ranching experiments should be identified and used to formulate new studies into behavior and potential interactions of wild and farmed cod. Modeling the risk of genetic mixing will help the decision makers and managers. Lacroix *et al.* (1998) show modelling approaches to estimate genetic introgression into the genome of wild stocks for salmonids and such approaches should be considered to be employed in studies of non-salmonids as well. Genetic information will be required to identify and characterize wild cod stocks at the subpopulation levels, in order to assess potentials for long-term effects on interbreeding with cultured stocks. More information on genetic variability of wild stocks, including information identifying the genetic integrity of stocks in spawning and nursery areas. Information on the mating success of wild and cultured cod is required. Size-related male-male aggression has been shown to be a factor in mating success. Studies addressing what levels of escapes will cause problems for local populations; impacts of escapees on different life-cycle stages. Information on the location and importance of nursery areas for juvenile cod is required.

The behavior of cultured cod has not been intensively investigated. There is a need to better understand not only the behavior of wild individuals, but also the behavior of released culture fish when returning to the wild. This would give an assessment of the degree of risk. The maintenance of sufficient wild populations (managing fisheries pressure, enhancement, etc.) is an efficient tool to mitigate the effects of increasing quantities of released aquaculture individuals. Many fish populations have suffered a substantial reduction in number during the last century, due to anthropogenic disturbance such as habitat degradation and overexploitation.

Investigations on the use of reproductively sterile fish are required. High priority should be given to devising alternative means of sterilization that could be achieved at low cost. Additional investigations on the findings that some sterile fish have been reported to mimic the reproductive aspects of wild fish, impairing the natural reproduction of their conspecifics, are required.

Use of local, unselected cod stocks will decrease the risk of potential genetic interaction. Programs using local cod populations as broodstocks for founder populations for selective breeding programs which maintain a large genetic diversity in hatcheries are important.

The improvement of aquaculture engineering for the containment of cod fry, growout stocks, and breeders is required. This applies to both floating cages (mooring, net quality, resistance of the raft to waves, avoidance of predators' effects on the nets, choice of locations), and to land-based facilities (screening and treatment of effluents). Development of closed systems, on land or floating, should be encouraged.

5 Risk Communication

The potential to recapture escaped cod has not been analyzed; but is an important area for research. It is also important to discuss this with the public early in a development program, and to derive the risk management triggers and contingency plans in an open and transparent manner, for each area where a wild cod subpopulation can be identified.

The positions of farms and cod stocks held in each country should be recorded by the regulating body in each Member State. Monitoring for sexual maturity and spawning activities should be carried out on farms that rear cod beyond the normal age of sexual maturity (two years). Monitoring would address the question of whether photoperiod manipulation is effective in delaying sexual maturation and identify where the potential risk of egg releases could occur.

A precautionary approach seems warranted that would have as one part a legislative framework for monitoring fish farm escapes of non-salmonid species, at least until the potential for problems is better understood. It has been reported that a high percentage of the escapees are due to releases of cultured fish during usual husbandry operations (sorting, local cage transfer, bathing for vaccination or chemical treatments). Through the expanded use of Code of Conducts and Code of Practice, the husbandry standards have to be improved in order to decrease the risk.

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Atlantic Halibut (*Hippoglossus hippoglossus* L)

1. Hazard Identification

1.1 Life History Description of Wild Populations

1.1.1 Distribution

The Atlantic halibut is a boreal species with a wide north-south distribution in the NW Atlantic. Although a rare species, it is more commons along the northern and western coasts of Norway, the Barents Sea, Iceland, Greenland, and Canada. Immature and mature halibut reside in different habitats, with immature fish occupying coastal areas at depths of 20–60 m, then migrating to waters as deep as 1000 m as adults.

1.1.2 Movements

Several tagging experiments have shown that Atlantic halibut have widespread movements throughout the NW Atlantic, moving hundreds of km and undertaking both short and long distance spawning and feeding migrations. A fish tagged at Spitzbergen was caught 8 months later off Western Norway 1000 km to the south. Migration patterns have a distinct seasonality (Haug, 1990). Mark-recapture studies show that adults may return annually to the same spawning grounds, forming breeding populations. Adults appear to return to the same site to spawn every autumn, but this seasonal regularity of movement depends on local oceanographic conditions. When water temperatures in surface waters are too low during winter (halibut seem to avoid water temperatures below 3^{o}C), halibut migrate to deeper waters, returning to the coastal areas in the warmer summer months.

1.1.3 Growth

Halibut are the largest of all the flatfishes. Maximum sizes are more than 3.5 m in length and weights exceeding 300 kg. Halibut exhibit strong sexual dimorphism, with females larger and longer-lived than males. Maximum reported sizes of male fish are 170 cm and 70 kg for a 27 year old fish, whereas females can exceed 3.5 m in length and 300 kg in weight. The maximum age for a female has been reported at ~50 years old.

Halibut are eurythermal, showing good growth over a wide range of temperatures (7–15^{o}C). Females grow much more rapidly and to a larger maximum size than males (Figure A6.5).

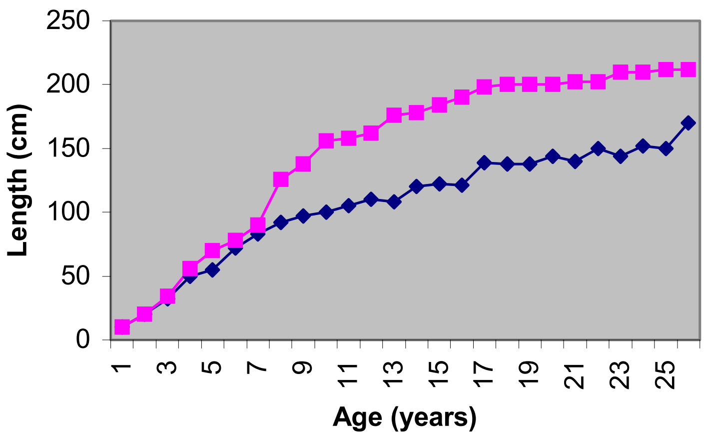

Figure A6.5. Observed growth of male (blue) and female (pink) halibut captured from a spawning area SW of the Faroe Bank in 1983–86. (Jakupsstouv and Haug, 1988).

There are reported differences in growth capacity between populations at different latitudes, with fish from high latitudes having a higher growth capacity than fish from lower latitudes (Jonassen *et al*., 2000). Northern populations also have a lower optimal temperature for growth when compared to southern populations. Studies of several fish species showed that the optimal temperature for growth is positively correlated with long or increasing photoperiod. In halibut, growth was correlated positively with day length at 11°C (Jonassen *et al*., 2000).

Growth rates of halibut decrease with increasing size, as shown for many fish species (size dependent growth). Juvenile growth rates vary throughout the year with most rapid rates in summer and autumn. In a three year study of juveniles in Faxa Bay, Iceland, year one fish grew from 12–15 cm in May–June to ~26 cm in December, remaining at ~26 cm sizes all the next winter (January–May), but reaching 35–39 cm by the end of the second year. Stationary growth continued throughout the winter of year 3. Juveniles reached 50–56 cm by December of year 3 (Sigurdsson, 1956). Growth rates of juvenile halibut vary widely across the North Atlantic, and even within different fjords (Sigurdsson 1956). Optimal temperatures for growth decrease with increasing fish sizes. Bjornsson and Tryggvadottir (1996) showed a 4°C decrease in optimal temperature as halibut grew from 10 g to 5 kg. Jonassen *et al*., (2000) showed that growth rates for juvenile halibut are influenced significantly by temperatures and fish sizes (Table A6.3).

Table A6.3. Optimal temperatures for growth of different class sizes of juvenile halibut (Jonassen *et al*. 2000).

Fish Sizes (g)	Optimal Growth Temps. (°C)
5–10	14.9
20–25	13.9
40–50	13.0
60–70	12.7

1.1.4 Diets

Halibut change their feeding preferences as they age. Juveniles less than 30 cm have a diet comprised almost exclusively of crustaceans (mysids, hermit crabs, prawns, and other small crabs) (McIntyre 1953). As they grow to a size of 30 to 60 cm, they become more piscivorous, and juvenile stomachs contain a mixture of fish and crustaceans. Small gadiods, young cod, and sand eels become more prevalent in the diet. This switch in dietary composition has been found in studies of young halibut from throughout the north Atlantic (Haug 199).

Adult halibut are ambush predators; however, they are not restricted to the seabed, hunting also in the pelagic, and preying heavily on fish. Adults have a remarkably narrow prey spectrum, with a special affinity for *Sebastes* (Haug, 1990). *Sebastes marinus* occupied 65–81% of stomach contents in a study of seasonal food contents of adult halibut in Icelandic waters (McIntyre, 1953). In the winter when it occupies deeper waters its diet will contain more shrimp and other benthic crustaceans.

1.1.5 Reproduction and spawning

Halibut spawn over deepwater soft clay or mud bottoms off the Norwegian coast (300–700m depths). Halibut appear to have a remarkable homing ability that allow adult fish to return to the same spawning sites each year where they form spawning aggregations. Spawning aggregations have been observed in Norway and a restricted area along the southwestern slope of the Faroe Bank at 700–1000 m. It is likely they also spawn in deepwater slope areas along the continental shelf in other parts of the North Atlantic. Stobo *et al.* (1988) has suggested similar homing to specific spawning areas where spawning aggregations form also occurs in Canadian waters.

Male halibut reach sexual maturity at a younger age and smaller size than females (Haug, 1990). Average ages (50% levels), lengths and total weights at which males matured were 4.5 years, 55 cm and 1.7 kg; and in females, 7 years, 110–115 cm and ~18 kg, but there is much variability (Table A6.4).

Table A6.4. Variability in Sexual Maturity in Halibut (taken from Haug 1990).

Location	Sex	Age (y)	Length (cm)	Weight (kg)
Faroe Islands	Males	4.5	55	1.7
NE Atlantic	Males	4–6		
Norway	Males	12 (range 7–17)		
Faroe Islands	Females	7.0	110–115	~18
NE Atlantic	Females	4–6		
Norway	Females	13 (range 8–18)		

There was a large reduction in age at first maturity reported from northern Norwegian halibut populations from the years 1936–1960 to 1981–1985. In the 1936–1960 data sets, average ages were 12 years for males and 13 years for females, which declined to 7 years for males and 8 years for females by 1981–1985. It was suggested that fishing pressure decreased halibut population densities, causing an increased growth rates. If so, this would imply that age at sexual maturity is more a function of growth rate and size than of age, which is a common feature for fish that mature at old ages (Roff, 1982).

Halibut spawn at 300–700 m at 4.5–7.0 °C and salinities of 33.8–35.0 ppt. Along the Norwegian coast, a spawning migration takes place at Christmas time from shallower coastal areas to deeper waters at the ends of fjords, where spawning takes place from December to May. Apart from these deep holes along the Norwegian coast, the most important spawning grounds are at the western side of the ridge from Scotland to the Faroe Islands and to Greenland.

Halibut are proportional spawners, spawning in intervals with ~70 hours between each spawning. Halibut have an enormous egg production, with a single mature female able to produce millions of eggs; the total number of eggs in one season may reach 2–3.5 million. Halibut eggs are exceptionally large for a marine teleost. In Norway, egg diameters vary from 3.06 to 3.49 mm (Haug 1990). There is evidence that egg diameters decrease during the spawning season.

Spawning takes place on the seabed. Eggs have positive buoyancy, ascending to reach neutral buoyancy in the bathypelagic, then hatching at ~100–200 m within 12–18 days at ~5°C. The halibut yolk sac is not absorbed until 1.5–2.0 months after hatching. Larvae are ~6–7 mm at hatching, but are poorly developed. Over the next 40 days, the internal organs, functional mouth and gut parts develop. During this period, larvae rise into the upper part of the water column. The extended period of larval development in the pelagic insures a long distance distribution of fish from its spawning areas.

There is little known about the movements of juvenile halibut. However, it is presumed that they are carried inshore by currents and occupy well-defined nursery areas; which are shallow coastal areas with sandy bottoms of 20–60 m depth. Nursery areas are known from the Faroe Islands, Faxa Bay on the west coast of Iceland, and Sable Island Gully off Nova Scotia (Trumble *et al*., 1993).

2 Risk Assessment

Status of Atlantic Halibut Fisheries

Having the characteristics of being a large, slow growing and long-lived top predator with a late onset of sexual maturity, halibut are vulnerable to overfishing. Indeed, halibut is now on the IUCN Red List as "endangered" (the listing is "endangered A1d", which is defined as an "observed, estimated, inferred or suspected reduction of at least 50% over the last 10 years or 3 generations, whichever is the longer, based on actual or potential levels of exploitation"). Today, the Atlantic halibut fishery off Canada has been determined as "practically extinct", producing just ~1,000 tons. Based on the ICES STATLAN data from 1991–2000, the total catch in the Northern Atlantic and Southern Arctic oceans fell from 3,988 to 1,847 tons. Based upon a maximum fishing mortality of 0.2 (ICES 2002), the total wild stock of halibut could be estimated at 7,833 tons. Rice and Cooper (2003) have comprehensively reviewed the management of flatfish fisheries and conclude that unsustainability in a "common feature of these fisheries".

2.1 Release assessment

2.1.1 Aquaculture of Halibut

The first aquaculture trials of Atlantic halibut started in the 1980's, pioneered by Norway. Progress has been slow due to the difficulties in high mortalities experienced in the transition from eggs to juveniles, high rates of infections and diseases at fry and juvenile stages, and lack of adequate quality formulated feeds.

Significant constraints to development exist in broodstock maintenance and performance, larval rearing and juvenile survival, and the development of economically viable and high performing feeds for halibut at all rearing sizes. Production of juvenile halibut remains a delicate process more akin to an art rather than a science. The hatchery operator must use live feeds and carefully balance essential fatty acid compositions for diets as fish grow. During the first few weeks of hatchery production, fish survival is highly uncertain (Olsen *et al.,* 1999). However, hatchery production is becoming more predictable and juvenile production is increasing steadily (Berg, 1997) but juvenile production is still too costly; in addition, demands for juveniles by growout operators remain limited.

In 2003, Norway produced ~500,000 fry and ~500 tons from aquaculture. Fry are produced in intensive, closed system production units. In Norway, most growout takes place in net pens having stacks of false net bottoms ("net trampolines") that increase the bottom surfaces on which the demersal fish can rest. Maintenance of halibut broodstock and juvenile production is conducted on land in recirculating systems, but all commercial production is currently conducted in coastal net pens. Depths of net pens range from deep, ~35 m nets in deel fjords in Norway to shallower 6 m deep pens in Scotland. Video cameras are used to monitor the fish. Net pens are located at protected sites with favourable temperature conditions; escapes are thus less likely to occur than in salmon farms located at exposed sites. A limited amount of aquaculture is also being conducted in tanks which are also provided with "shelves".
Reports of malpigmented, "albino" or discontinuous pigmentation patterns observed in some adult halibut from culture have been attributed to an incorrect amino acid balance in enriched <u>Artemia</u> given at first feeding. Normal, continuous pigmentation has been achieved using cultured zooplankton (C. Greathead, pers. comm.).

Projected halibut aquaculture production could exceed 20,000 tons in 10 years (by 2014), with the UK (Scotland), Norway and Iceland as centers of research, development and production (Table A6.5). Nearly all of the future aquaculture production will be conducted in net pens.

Table A6.5. Status and Projections for the Development of Halibut Aquaculture.

Nations	Status	Planned Production
Norway	~700,000 juveniles (2002), ~1,000 tons production (2004)	9,000 tons by 2010
Iceland	178 licenses, 10–15 active farmers, 1 company (Fiske), ~1,000,000 juveniles produced (2004), 100 tons production (2001)	Not available
UK	7 companies, 12 sites, 4 hatcheries, ~300 tons in 2003	10,000 tons by 2012
Canada	Limited production from just 2 farms	Not available
Chile, Ireland, USA	Experimental only	Not available

2.1.2 Genetic Structure of Wild Halibut Populations

Several tagging experiments have revealed that the Atlantic halibut is highly migratory, but mark-recapture studies suggest that adults return annually to the same spawning grounds forming distinct breeding populations; however, small, local breeding stocks also exist.

Some variations in electrophoretic characteristics between halibut from three spawning locations along the Norwegian coast have been reported (Mork and Haug, 1983); however, later analyses gave support to a hypothesis of homogeneity over a larger geographic scale (North Norway to Greenland). Cluster analysis however indicated that a sample from mid- Norway could be different from the others (Haug & Fevolden, 1986). Other studies using allozymes gave some indication of two reproductively isolated groups: northern Norway/Barents Sea and Faroes-Iceland-Greenland (Foss *et al*., 1998). More recent studies of Atlantic halibut along the Norwegian coast, and the first to utilize microsatellite DNAs, to analyze populations in eastern Canadian and Icelandic waters has shown that stocks may be comprised of a single "panmictic" stock and do not indicate any reproductive isolation (Reid *et al*., submitted).

2.3 Exposure Assessment

The downward trend recorded by the ICES fisheries landing data for Atlantic halibut in the North Atlantic between 1991 and 2000 will likely continue which, in turn, means that the wild stock will become even more endangered.

There have been no studies carried out on the interaction between wild and cultured halibut but based on the known reproductive biology and fish ecology, any interactions will unlikely occur until the escaped fish mature. Wild halibut females do not mature until large sizes are reached (Table A6.4); a size which is much larger than current and projected market sizes for cultured fish. As a result, spontaneous spawning from mixed sex populations contained in net pens is very unlikely.

Given the propensity for halibut to travel extensive distances, escapes are likely to disperse widely from their point of escape. It is not known whether the escaped fish will have the sensory clues to allow them to find the spawning areas where it is suggested that halibut congregate.

It has been shown that in some populations of halibut, that the red fish, *Sebastes marinus,* can make up to 80% of the diet of adult halibut. A consequence of significant releases of adult halibut could be a negative impact on red fish populations.

It is not known whether halibut pair during spawning or whether there is a massed spawning event. It is therefore not possible to predict whether escaped males will be less successful in mating with wild females.

2.4 Consequence Assessment

Halibut fisheries are in poor shape. The species has been classified in an "endangered" category by IUCN; as a result, the species is uncommon throughout its natural range. Halibut are prized by consumers and command high prices. It is unlikely that expansion of halibut aquaculture will experience significant price or volume competition from restored wild capture fisheries in the foreseeable future.

Halibut is a highly migratory species that is widely dispersed across the North Atlantic. Some variation in growth capacity of populations within the north Atlantic has been reported, but studies report a very low amount of genetic differentiation within its range. Halibut broodstock and juvenile production is performed in containment on land. While there is no information available on annual numbers of escapees from net pens, its low population density in the wild and the low level of genetic differentiation found throughout its range (Reid *et al*., submitted), suggests a minimal possibility for negative genetic effects on the wild populations.

Since spawning of fish in net pens is very unlikely, there is little need to be concerned about halibut aquaculture operations delivering fertile eggs to the marine environment and potentially impacting wild populations.

In the wild, adult halibut have a mixed diet of fish and crustaceans, with a special affinity for *Sebastes*. Since escapee halibut from expanded net pen aquaculture would be conditioned to eat a pelleted diet, it is unknown how quickly they would return to their wild diet.

2.5 Risk Management

Given the endangered status of halibut populations, the species' widespread movements and abilities for wide dispersion and the reported small amount of genetic differentiation between these widely dispersed populations within its natural range, escapees from expanded halibut aquaculture present little risk to the remnant wild populations of the north Atlantic.

Since halibut are demersal and net pen operations are located in more protected areas than salmon farms, escapement from net pen operations would likely occur only during catastrophic losses of the entire structures, with smaller releases occurring during transfers of juveniles for stocking and harvests of adult fish for market.

In 2003, there was a reported escape of 3000 6 kg size fish (18 tons) from a halibut farm in Scotland due to a seal attack. There are no reports of any negative (or positive) impacts on local halibut stocks due to this event.

It is predicted that, due to the factors reviewed above, annual losses for expanded halibut aquaculture would be much lower than, for example, the 20–25 incidents per year reported from 1998 to 2003 in salmon net pen aquaculture in Scotland (an escapement rate estimated 0.1–1% of smolts stocked; I.M. Davies, pers.comm.). It is debatable if halibut escapement from aquaculture would present any negative impacts on the presently unsustainable halibut fishery in the north Atlantic. Genetic impacts are forecasted to be negligible, since north Atlantic populations are, at the present time, considered "panmictic" (Reid *et al*., submitted).

2.6 Risk Communication

The wild fishery for Atlantic halibut can be regarded in many instances as a "bycatch", and it is predicted that those fisheries that are returning significant tonnages of this species at present will also start to decrease in the future. It is considered that fishing pressure is a far more significant risk to this species than that of any potential impact from aquaculture escapees.

If successful culturing of halibut continues to expand in the North Atlantic, this may in turn reduce the pressure on the wild fish populations. Part of this expansion would be the development of a code of practice for this species and the prioritization of research topics that need to be undertaken. This should be designed to have the least negative impact and the most positive impact on the wild population of halibut.

In the North East Atlantic, the halibut is an open water, oceanic, deeper water species, and recreational aspects to this fishery in this area are therefore low.

The communication of this information must be inclusive and all potential stakeholders must be included in the transfer of information so that a full discussion can be maintained on predicted low potential risks of halibut aquaculture.

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Annex 7 A review and assessment of environmental risk of chemicals used for the treatment of sea lice infestations of cultured salmon

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1 INTRODUCTION

Fin-fish marine aquaculture is a relatively new but important food industry that had a World wide production amounted to 3.79M metric tonnes in 2002 of which 47.5% was salmonid species valued at $4.9B (US) (Food and Agriculture Organization of the United Nations, 2004). Cultured Atlantic salmon comprised 60% of the salmonid species production, and 91% of this was produced in Canada, Chile, Norway and United Kingdom. The demand for cultured fisheries products is increasing with the continued decline in catchable wild fisheries and increase in demand by consumers (Food and Agriculture Organization of the United Nations, 1999). The challenge for the aquaculture industry is to expand and yet remain environmentally sustainable and socially acceptable (Commission of the European Communities, 2002).

Cultured salmon in the crowded and stressful conditions of aquaculture are susceptible to epidemics of infectious bacterial, viral and parasitic diseases. Sea lice are ectoparasites of many species of fish and are a serious problem for salmon aquaculture industries (Roth *et al.,* 1993c; MacKinnon, 1997). The species that infest cultured Atlantic salmon are *Lepeophtheirus salmonis* and *Caligus elongates*. Infestations result in skin erosion and sub-epidermal haemorrhage which, if left untreated would result in significant fish losses, probably as a result of osmotic stress and other secondary infections (Wootten *et al.,* 1982; Pike, 1989). Sea lice are natural parasites of wild Atlantic salmon, and infestations have occurred routinely in European aquaculture since the 1970’s (Roth *et al.,* 1993c). The first severe epidemic in Atlantic Canada occurred in 1994 (Hogans, 1995; O'Halloran and Hogans, 1996). Sea lice reproduce year round and the aim of successful lice control strategy must be to pre-empt an internal infestation cycle becoming established on a farm by exerting a reliable control on juvenile and preadult stages, thus preventing the appearance of gravid females (Treasurer and Grant, 1997). Effective mitigation, management and control of sea lice infestations requires good husbandry, linked to the use of natural predators such as wrasse and effective anti-parasitic chemicals (Rae, 2000; Read *et al.,* 2001; Eithun, 2004).

Aquaculture, like all forms of intensive food production, will potentially generate environmental costs. Chemicals used in the treatment of sea lice infestations are normally subsequently released to the aquatic environment and may have impact on other aquatic organisms and their habitat. The present paper will review the chemical therapeutants available to control sea lice and assesses their risks to the aquatic ecosystem. The review will be limited to those chemicals that are currently authorized for use by the salmon aquaculture industry in Europe and North America.

2 SEA LICE BIOLOGY

The species that infects Atlantic salmon are *Lepeophteirus salmonis* and *Caligus elongatus.* Both species are ectoparasitic on salmonids, the former being the more destructive. A grown female of *L. salmonis* is 8 to 12 mm in length, while the male is about half of this size. The sea lice in fish farms tend to be a bit smaller. Likely sources of new infections of fish farms are planktonic stages, which may originate from sea trout, wild salmon or rainbow trout or salmon that have escaped form captivity, although *L. salmonis* has also been found on other marine species.

The eggs of the sea lice hatch directly into the water from egg strings fastened to the genital segment of the female lice (Figure A7.1). The larvae are free-swimming naupli through one moult and then become infective copepodids. These are about 0.7 mm long and 0.3 mm wide, and it is this stage that can recognize and become attached to a host fish. It is however observed that grown sea lice can transfer from fish to fish. The dispersion of the nauplii is dominantly passive as the larvae drift in the water, but the vertical movements of the larva (copepodids are positively phototoxic) will also influence there position in a water body. In total, the seal lice pass through 10 stages, with one moult between each stage (Rae, 1979)

The development of the sea lice is dependent on the sea temperature. It takes a male 42 days and a female 50 days to develop from egg to adult at 10^{o}C, a temperature that is normal in salmon farming areas. The sea slice can, however, tolerate relatively large range of temperatures and can hatch and develop at as low as 2^{o}C (Boxaspen and Naess, 2000).

The period during which a copopodid can infect a fish is called the infective window and is crucial in the combat of sea lice. It has been shown that the larvae can infect fish from day one after moulting, but they appear to be more infective after a few days. After this period the copopodid exhausts its energy reserves, and becomes less successful in infecting susceptible host fish. Calculations on empirical data indicate that the last day a larvae can infect a fish will be 32.5 days after hatching at 6^{o}C and 17 days at 12^{o}C. Such long infective pelagic stages show that *Lepeophtheirus* *salmonis* has a great potential for dispersion by tidal ans other currents, and that it can infect fish over a wide area away from the source. This explains the massive infection problems that have been encountered by the salmon farming industry and emphasizes the need for efficient husbandry strategies and chemical agents to contol infections on fish farms and reduce the potential for transfer of lice between farms.

3 THERAPEUTANTS IN USE

Chemicals currently authorized for the treatment of sea lice infestation may be classified into two groups, based on their route of administration (Table A7.1). Organophosphates (azamethiphos), pyrethroids (cypermethrin and deltamethrin) and hydrogen peroxide are administered by bath techniques, while the avermectins (emamectin benzoate) and chitin synthesis inhibitors (telflubenzuron and diflubenzuron) are administered as additives in medicated feed. The number of chemicals authorized for use is limited because of the high cost of development and licensing for a small market relative to other markets for pesticides and medicinals.

Bath treatments are conducted by reducing the depth of the net in the salmon cage, thus reducing the volume of water. The net-pen (and enclosed salmon) is surrounded by an impervious tarpaulin and the chemical is added to the recommended treatment concentration. The salmon are maintained in the bath for a period of time (usually 30–60 minutes) and aeration/oxygenation may be provided. After treatment, the tarpaulin is removed and the treatment chemical is allowed to disperse into the surrounding water. Bath treatments are considered a topical application as the therapeutant is absorbed by the sea lice from the water.

Medicated feed is prepared by adding concentrated pre-mix containing the active ingredient to feed during the milling and pelletisation processes. The chemical is administered by calculating the dosage based on the feed consumption rate of the salmon. Generally, the medicated feed is given on the first feeding of the day as this can counteract any reduction in appetite of infected fish, or any tendancy of the fish to discriminate against medicated feed. The therapeutant is absorbed though the gut into the blood stream of the salmon and is then transferred to the sea lice as they feed on the skin of the salmon. The advantages of in-feed preparations compared to bath treatments are that releases to environment are much slower and less direct. Treatment is less stressful to the fish, the dosage can be more accurately controlled, the oral preparations are not toxic to farmers, and it requires less labour. One disadvantage is that stressed or diseased fish often feed less than healthy fish and therefore may not receive a fully effective dose.

4 BATH TREATMENTS

4.1 Organophosphates

4.1.1 Efficacy and Mechanism of Action of Organophosphates

Four organophosphate compounds have been used in the treatment of infestations of sea lice: malathion, trichlorfon, dichlorvos (DDVP) and azamethiphos (Roth *et al.,* 1993c). Malathion was tested experimentally but the concentration required to effectively remove lice was so high that treated fish became lethargic. Trichlorfon also had a narrow margin of safety for salmon (Horsberg *et al.,* 1989). Trichlorfon degrades into the more toxic and effective DDVP, but the rate of transformation is dependent on water temperature and pH (Roth *et al.,* 1993c). The incocnsistency of this transformation, the acute toxic risk to salmon and the increase in use of DDVP resulted in the gradual cessation of use of trichorfon. For a number of years, DDVP was the treatment of choice against infestations of sea lice. However, frequent use led to the resistance to DDVP in sea lice in some areas (Tully and Mcfadden, 2000). This, coupled with narrow therapeutic margin (about 4 times the recommended treatment concentration of 1.0 mg/l), resulted in the product being phased out as an anti-louse therapeutant. Herring larvae were reported to tolerate azamethiphos better than DDVP (Roth *et al.,* 1993c).

Azamethiphos is an organophosphate insecticide and the active ingredient in the formulation Salmosan®. It is used as a bath treatment at 0.1 mg/l for up to 1 hour and has a fairly small therapeutic index (dose toxic to salmon/dose used to treat sea lice). Azamethiphos is registered for use in Chile, Ireland, Norway and Scotland. Recently Novartis, the producer of azamethiphos applied to discontinue the use of their product in Canada from April 1, 2002. Aquauculturists may continue to use azamethiphos until April 1, 2005 (Cathy Morris, Health Canada, personal communication). Similar initiatives have occurred in other countries.

Atlantic salmon can tolerate one hour exposures to 0.5 mg/l of azamethiphos, and three one hour doses repeatedly weekly to 0.3 mg/l of azamethiphos. Deaths were observed after 1 hour exposure to 1.0 mg/l (Roth *et al.,* 1993c). Azamethiphos has neuro-toxic action, acting as an acetylcholinesterase (AChE) inhibitor. In Atlantic salmon that died during the treatment with azamethiphos, the AChE levels in the brain were reduced by 74% but the depression of AChE by azamethiphos is not cumulative in fish after repeated exposures. Exposure at 1 mg/ml for 24 hours resulted in 15% mortality of Atlantic salmon after 24 hours (Sievers *et al.,* 1995). Azamethiphos has been shown to be mutagenic in several *in vitro* tests (Committee for Veterinary Medicinal Products 1999). The high alkylating potency of azamethiphos explains this mutagenicity and it was recommended that biological effects studies on non-target biota should include tests for delayed effects (Zitko, 2001).

The sensitivity of lice to azamethiphos is variable, and some populations of lice are more sensitive to this compound than others (Roth *et al.,* 1996). Development of resistance to organophosphates is common and has been shown to include azamethiphos (Levot and Hughes, 1989). In sensitive populations of lice, azamethiphos is effective in removing >85 % of adult and pre-adult lice but is not effective against the earlier life stages of the parasite (Roth *et al.,* 1996).

4.1.2 Distribution and Fate of Organophosphates

Azamethiphos is soluble in water (1.1 g/l) and has a low octanol-water partition coefficient (log K_{ow} = 1.05)(Scottish Environmental Protection Agency 1997). Consequently, azamethiphos is likely to remain in the aqueous phase on entering the environment. It is unlikely to accumulate in tissue or in sediment. Azamethiphos decomposes by hydrolysis in natural water with a half-life of 8.9 days. Dispersion studies indicated that after release of an experimental treatment (200 µg/l), the concentration of azamethiphos was below detection (0.1 µg/l) in a short period of time. It was not detected below 10 m depth and it was suggested that it is unlikely that azamethiphos would accumulate in sediment.

The bioaccumulation of azamethiphos by salmon is low and depletion of total azamethiphos in salmon is rapid, withdrawal time is 24 hours (Committee for Veterinary Medicinal Products 1999).

4.1.3 Biological Effects of Organophosphates

4.1.3.1 Laboratory Studies with Organophosphates

Lobster and shrimp were the most sensitive species to azamethiphos in laboratory acute toxicity tests, while bivalves such as scallops and clams were unaffected (Burridge and Haya, 1998). Adult lobsters held within the tarpaulin during an operational treatment did not survive. The 48-h LC50 has been estimated for the first four larval stages of the American lobster and adults of the same species (Burridge *et al.,* 1999). The values are as follows: Stage I 3.57 µg/l, Stage II 1.03 µg/l, Stage III 2.29 µg/l, Stage IV 2.12 µg/l, and Adults 1.39 µg/l. There is no statistically significant difference between these values. There is a seasonal aspect to sensitivity of lobsters to azamethiphos. Females lobsters are significantly more sensitive to azamethiphos in the summer than at any other time of year (Burridge *et al.,* forthcoming). Adult and Stage IV lobsters were exposed repeatedly (up to nine times) for varying lengths of time to four concentrations of azamethiphos(Burridge *et al.,* 2000c). The No Observed Effect Level (NOEL) was nine exposures of 120-min each over three days to 1 µg l^{-1} of azamethiphos. In addition to observed lethality, several surviving lobsters showed significant behavioral responses to repeated exposure to concentrations greater than 10 µg/l. Research commissioned by Ciba Geigy shows that azamethiphos is only lethal to several groups of invertebrates (molluscs (bivalves and gastropods), amphipods, and echinoderms) at concentrations greater than the prescribed treatment concentration of 100 µg/l (Scottish Environmental Protection Agency, 1997). Exceptions to this include copepod, decapod and mysid crustaceans. The 24 h LC_{50} of azamethiphos to the copepod, *Temora longicornis*, is reported to be > 10 µg/l. The 96 h LC_{50} for lobster larvae (*Homarus gammarus*) is 0.5 µg/l and is in general agreement with the 48 h LC_{50} for the American lobster (Burridge *et al.,* 1999). Finally, the 96 h LC_{50} for the mysid shrimp is reported as 0.52 µg/l (Scottish Environmental Protection Agency, 1997).

In laboratory studies, lobsters exposed to azamethiphos (5.0–10.0 µg/l) became quite agitated, often 'flopping' erratically around the exposure tank (Burridge *et al.,* 2000b). They were also aggressive to other lobsters and reacted very quickly to any movement. They seemed to lose control of their claws and eventually flipped onto their backs and died within hours. Some affected lobsters remained moribund for periods of time ranging from hours to days. The consequences of behavioral responses such as these on organisms and populations in the natural environment are unknown.

Laboratory studies were conducted to investigate possible sublethal effects of azamethiphos exposure on the American lobster. Preovigerous females were exposed for 1 h biweekly to 10 µg/l azamethiphos and monitored for spawning success and survival (Burridge *et al.,* 2000b; Waddy *et al.,* 2002b). Surprisingly, even with such infrequent exposures, up to 100% of the animals exposed to this concentration died during the experiment: some expired after only three treatments. A significant number of the surviving lobsters failed to spawn. A laboratory study indicated that shelter use behavior could be affected by azamethiphos (Abgrall *et al.,* 2000). However, exposure to concentrations of azamethiphos in water greater than five times the recommended treatment concentration for periods of several hours was necessary.

The response of mussels to stimuli was unaffected by exposures to 10.0 µg/l for up to 24 h (Scottish Environmental Protection Agency, 1997). The inhibition of AChE by azamethiphos is not cumulative in fish (Roth *et al.,* 1993c), however cumulative inhibition of AChE may have occurred in lobster in the studies above (Burridge *et al.,* 2000b). Mussel closure rate was affected at concentrations above 100 µg/l and exposure to 46.0 µg/l resulted in 50% inhibition of AChE activity. AChE activity in herring yolk sac larvae and post-yolk sac larvae was inhibited by 96 h exposure to azamethiphos at 33.4 and 26.6 µg/l, respectively.

4.1.3.2 Field Studies with Organophosphates

During 1995, a study was conducted to determine the effects of single operational azamethiphos treatments on juvenile and adult lobsters, shrimp, clams and scallops suspended at two depths and varying distances from the treated cage.

During two of the treatments, lobsters held within the treatment tarpaulin died. No other treatment-related mortalities were observed (Chang and McClelland, 1996). In addition, lobsters were suspended at three depths at 20 sites surrounding a salmon cage site that was conducting operational treatments with azamethiphos. No treatment-related mortalities were observed. Mussels deployed during field trials in Scotland were unaffected. Mortality among lobster larvae was 27% but was not correlated to distance form the treatment cage (Scottish Environmental Protection Agency, 1997).

Finally, survival of lobsters suspended at mid-depth and near bottom at four sites in the salmon farming area of Lime Kiln Bay, New Brunswick, Canada, plus a control site, was monitored for nine weeks during August–October 1996. There were no apparent differences in lobster survival between the experimental and control sites (Chang and McClelland, 1997). No residues of azamethiphos were detected in water samples collected weekly from the five sites (Detection Limit = 50 pg/l). Diving surveys at a lobster nursery area located near a salmon farm in early August, September and late October of 1996 found no apparent changes in lobster populations over time, and the area was found to have a considerable population of juvenile lobsters.

Measurements of primary productivity and dissolved oxygen were made before, during and after chemical treatments at salmon farms in southwest New Brunswick in August–September 1996. There were no evident effects on dissolved oxygen and chlorophyll *a* levels, indicating no impact on primary production (D. Wildish, St. Andrews Biological Station, St. Andrews, NB, unpublished data).

4.2 Pyrethroids and Pyrethrins

4.2.1 Efficacy and Mechanism of Action of Pyrethroids and Pyrethrins

Pyrethrins are the active constituents of an extract from flower heads of *Chrysanthemum cinerariaefolium*. This mixture of chemically related compounds has been used for their insecticidal activity since the late 19th century (Davis 1985). The pyrethrins decompose readily as they are susceptible to catabolic enzymes and sunlight. In the early 1960s synthetic analogues that were more persistent than the natural pyrethrins were developed and referred to as pyrethroids were developed (Barthel, 1961). It was their high degradability, low toxicity to mammals and high toxicity to crustaceans that led to the initial interest in pyrethins as treatments for sea lice infestations.

The mechanism of action of the pyrethrins involves interference with nerve membrane function, primarily by their interaction with Na channels (Miller and Adams, 1982) which results in depolarization of the nerve ending. This interaction results in repetitive firing of the nerve ending in the case of the pyrethroids, cypermethrin and deltamethrin.

A method used for delousing salmon with pyrethrins is to put an oil based 10.0 mg/l solution of pyrethrins in a 5 meter tube then pass salmon through the solution in the tube. The time for the salmon to pass through the solution is approximately 5–30 sec and over all effectiveness has been reported as 96% (Boxaspen and Holm, 1992). An advantage of this tube method is that the treatment solution can be recovered from the tube and not released to the marine environment.

In the autumn of 1989, a modified version of the bath method for delousing was tested (Boxaspen and Holm, 2001). The technique was based on using pyrethrins mixed in oil, instead of adding a synthetic emulsifier to make a water soluble solution. The oil based solution was allowed to float on top of the water in a cage and the sea lice are exposed as the infested salmon jump out of the water. The water solubility of salmon mucus was expected to protect the fish but the salmon louse with a lipid layer in the cuticle should selectively absorb the pyrethrin mixture. Tests suggest that three jumps would give acceptable delousing (85%). However this method was considered too sensitive to changes in fish behaviour and the amount of decomposition of the pyrethrins with variations in sunlight.

The pyrethrins act only on adult and pre-adult life stages (Roth *et al.,* 1993b), and aquaculturalists have therefore used several pyrethroids in conventional bath treatment techniques. Deltamethrin and cypermethin (Excis®) are approved for use in Norway, and cypermethrin is approved in Ireland and United Kingdom. Cypermethrin had temporary registration in the United States but it has recently been withdrawn. The application for use in Canada for treatment of sea lice was not approved by the Canadian Pest Management Authority.

The recommended treatment of salmon against sea lice is a 1 hour bath with cypermethrin at a concentration of 5.0 µg/l, and for deltamethrin it is 2.0–3.0 µg /l for 40 minutes (Scottish Environmental Protection Agency, 1998) Cypermethrin is effective against all attached stages including adults, and therefore less frequent treatments should be required than with organophosphates, 5–6 week intervals rather then 2–3 week intervals, respectively.

In one of five Norwegian salmon sites that used deltamethrin for the treatment of sea lice there was a significant decrease in effectiveness of the treatment with an increase in the number of treatments(Sevatadal and Horsberg, 2003). Bioassays with preadult stage II sea lice underlaboratory conditions verified that of resistance contributed to treatment failure and that the EC50 was 25 times higher than at an area previously unexposed.

4.2.2 Distribution and Fate of Pyrethroids and Pyrethrins

Synthetic pyrethroids are unlikely to be accumulated to a significant degree in fish and aquatic food chains since they are rapidly metabolized (Kahn, 1983). This author warns, however, that pyrethroids such as cypermethrin can persist in sediments for weeks and may be desorbed and affect benthic invertebrates. While there is a large amount of knowledge regarding the ecotoxicology of cypermethrin in the freshwater environments (Kahn, 1983; Hill, 1985; Haya, 1989), knowledge is more limited for marine species.

The pyrethrins are unstable and a greater than 30% loss of pyrethrins in sea water after 1 hour (Leahey, 1985) and a half life of 5hrs (Burridge and Haya, 1997) have been reported. A 10mg/l solution will lose effectiveness after one hour for the treatment of sea lice after one hour, but will remain effective if an antioxidant (piperonylbutoxide) is added (Clark *et al.,* 1989). The concentration of cypremethrin decreases rapidly on release from a cage site after treatment. Data collected in Loch Eil Scotland showed that the highest concentration found was 187 ng/l 25 minutes after release 25 m from the site in the direction of the current flow(Scottish Environmental Protection Agency, 1998). Cypermethrin remained above 0.031 ng/l up to 50 min after release and above 0.074 ng/l for 30 min (Hunter and Fraser, 1995 through (Pahl and Opitz, 1999)). Mussels exposed inside a treated cage (5.0 ug/l cypermethrin) accumulated 133 ug/g. Mussels 2 m from cages accumulated 9.2 ng/g after 7 treatments and cypermethrin was only occasionally barely detectable 100 m from cage. There were no effects on *Crangon crangon* used as sentinel species near the cage site. Organisms in the vicinity of the cages would be exposed to concentrations which fall to 50 ng/l within one hour of release (Scottish Environmental Protection Agency, 1998).

In aerobic sediments, cypermethrin biodegrades with a half life of 35 and 80 days in high and low organic sediment, respectively. It degrades much more slowly in anaerobic sediments (Scottish Environmental Protection Agency, 1998). The rapid disappearance of deltamethrin from water (60% in 5 min), its high adsorption on sediment and its low bioconcentration capacities (in daphnia, *Chlorella asellus*) indicate that this molecule will not accumulate through food chains. Nevertheless, its high toxicity and rapidity of action may cause significant harm to limnic ecosystems after direct treatment (Thybaud, 1990). The adsorption of pyrethroids onto suspended solids can produce dramatic reductions in the apparent toxicity of the compound. The 96 h LC50 value of rainbow trout is 1.0–0.5 µg /l (Associate Committee on Scientific Criteria for Environmental Quality, 1986). When trout were caged in a pond containing 14–22 mg/l suspended solids, the 96 h LC50 was 2.5 µg /l. In a pond sprayed with deltamethrin containing 11 and 23 mg/l suspended solids, detamethrin partitioned rapidly to suspended solids, plants, sediment and air with a half life if 2–4 h in water (Muir *et al.,* 1985).

Because pyrethroids tend to adsorb onto particulate matter chronic exposures may not occur other than in laboratory studies. Cypermethrin absorbed by sediment was not acutely toxic to grass shrimp until concentrations in sediment were increased to the point where partitioning into the overlying water resulted in acutely lethal concentrations (Clark *et al.,* 1987). For example, the 96 h LC50 for cypermethrin to grass shrimp is 0.016 µg/l, but grass shrimp could tolerate cypermethrin concentrations in sediment of 10.0 µg/kg for 10 day.

4.2.3 Biological Effects of Pyrethroids and Pyrethrins

The lethality (96h LC50) of cypermethrin to lobster (*Homarus americanus*) and shrimp (*Crangon septemspinosa*), was 0.04 µg /l and 0.01 µg /l, respectively (McLeese *et al.,* 1980). The 24 h LC50 was 0.14 µg /l for adult lobster. For other marine invertebrates, 96h LC50 values range from 0.005 µg/l for mysid shrimp (Hill, 1985) to 0.056 µg/L for the same species (Clark *et al.,* 1989). The 96 h LC50 for five other marine crustaceans ranged from 0.016µg/l for grass shrimp to 0.20 µg/l for fiddler crab. Oysters were relatively insensitive, with a 48 h EC 50 of 2.3 mg/l based on larval development. For marine fish, the 96 h LC50 of cypermethrin to Atlantic salmon was 2.0 µg/l (McLeese *et al.,* 1980) and for sheephead minnow was 1.0 µg/l (Hill, 1985). Exposure of Atlantic salmon to a 10 mg/l solution of mixed pyrethrins for 6 min was 100% lethal, and some deaths occurred if the period of exposure was greater than 2 minutes (Clark *et al.,* 1989).

Larvae are often considered the most susceptible life stage to environmental or chemical stress. The 12h LC50 of cypermethrin for stage II lobster larvae at 10 and 12^{o}C was 0.365 and 0.058 µg /l, respectively (Pahl and Opitz, 1999). At sublethal concentrations effects on swimming ability and responsiveness of the lobster larvae were observed. The 48 h LC50 of a cypermethrin to the three larval stages (I, II, and III) of the American lobster (*Homarus americanus*) and to the first post-larval stage (IV) was 0.18, 0.12, 0.06, 0.12 µg /l of respectively (Burridge *et al.,* 2000a). Thus, cypermethrin was lethal to larval lobsters over 48 h at approximately 3 % of the recommended treatment concentration. In a study with larval lobsters and a formulation of pyrethrins and piperonyl butoxide there were significant differences in sensitivity between larval stages (Burridge and Haya, 1997). Stage I larvae were more tolerant of the pyrethrins formulation than Stage II, and both were more tolerant than Stages III and IV (48h LC50 = 4.42, 2.72, 1.39, 1.02 µg /l, respectively). On the other hand soft shell calm larvae, green sea urchin larvae and rotifers were tolerant of cypermethrin and 12 hour LC 50 values were greater than 10 mg/l (Pahl and Opitz, 1999).

The impact of pyrethroids and natural pyrethrins on non-target aquatic animals, especially invetebrates has been reviewed (Mian and Mulla, 1992). In general pyrethroids are more toxic to non-target insects and crustaceans than to other phylogenetically distant invertebrates. Among arthropods, however, crustaceans are phylogenetically closer to insects than molluscs and showed noticeable sensitivity. The isopod, *Asellus aquaticus* and the mysid shrimp, *Mysidophsis bahia* have shown even higher sensitivities than crustaceans to pyrethroids, including cypermethrin and permethrin. Spray operations on ponds have resulted in 95% reduction of arthropod fauna such as crustaceans, insects and arachnids. The residue profile of cypermethrin in water immediately after application, coupled with rapid decay (4–24h), explained the limited effect of pyrethroids on populations of non-target aquatic invertebrates in some case studies. On the other hand, invertebrates in habitats subjected to frequent treatments are likely to be more affected especially those species that show greater sensitivity. However populations of affected organisms generally recovered to pretreatment levels within weeks to months of the exposure.

In freshwater studies cypermethrin had a significant sublethal impact on the pheromone-mediated endocrine system in mature Atlantic salmon parr (Moore and Waring, 2001). It was suggested that cypermethrin acts directly on the Na channels and inhibits nervous transmission within the olfactory system and thus the male salmon is unable to detect and respond to the priming pheromone. In the marine environment it may reduce homing abilities of retuning adult salmon and increase straying rates between river systems.

Shrimp (*Crangon crangon*) were deployed in cages at various distances and depths from the cages during treatment with cypermethrin at two salmon aquaculture sites in Scotland during treatment with cypermethrin. The only mortalities were to shrimp held in treated cages (Scottish Environmental Protection Agency, 1998). Shrimp in drogues released with the treated water were temporarily affected but recovered. In an American field study, cypermethrin was lethal to 90% of the lobsters in the treatment cage but no effect was observed in those located 100–150 m away. There was no effect on mussels placed outside or inside the cages. Similar field studies indicated that cypermethrin was lethal to lobsters and planktonic crustaceans in the treatment tarpaulin but not to mussels, sea urchins or planktonic copepods.

Cypermethrin induced glutathione *S*-transferase (GST) activity in shore crab, *Carcinus maenas*, exposed to a solution of 50 and 500 ng/l of cypermethrin or injected intra-cephalothoracically with 10ng (Gowland *et al.,* 2002). However, activity of the enzyme returned to base levels after 36 h and there was no clear dose response and so GST activity may not be a useful biomarker of exposure to cypermethrin.

4.3 Hydrogen Peroxide

4.3.1 Efficacy and Mechanism of Action of Hydrogen Peroxide

Hydrogen peroxide is a strong oxidizing agent that was first considered for the treatment of ecto-parasites of aquarium fish (Mitchell and Collins, 1997). It is widely used for the treatment of fungal infections of fish and their eggs in hatcheries (Rach *et al.*, 2000). With the development of resistance to dichlorvos by sea lice (Jones *et al.,* 1992) there was move towards the use of hydrogen peroxide to treat infestations of mostly *Lepeophtheirus salmonis* but also *Caligus elogatus*. Hydrogen peroxide was used in salmon farms in Faroe Islands, Norway, Scotland and Canada in the 1990's (Treasurer and Grant, 1997). Hydrogen peroxide (Paramove®, Salartect®) is still authorized for use in all countries but it is not the normal treatment of choice and there is no record of usage in 2003. There may be renewed interest the use of hydrogen peroxide, in conjunction with the use of wrasse, as part of a strategy to allow sites to maintain "organic salmon aquaculture" accreditation status.

The suggested mechanisms of action of hydrogen peroxide are mechanical paralysis, peroxidation by hydroxyl radicals of lipid and cellular organelle membranes, and inactivation of enzymes and DNA replication (Cotran *et al.*, 1989). Most evidence supports the induction of mechanical paralysis when bubbles form in the gut and haemolymph and cause the sea lice to release and float to the surface (Bruno and Raynard, 1994).

The recommended dosage for bath treatments is 0.5 g/l for 20 min but the effectiveness is temperature dependent and the coompound is not effective below 10^{o}C. Treatments are rarely fully effective but 85–100% of mobile stages may be removed (Treasurer *et al.,* 2000). The first farm treatments in Scotland in October 1992 removed 83% of the mobile stages of sea lice. The recommended course is to repeat the tretment at 3–4 week intervals. This usually results in low numbers of sea lice for 8 weeks following the third treatment (Treasurer and Grant, 1997). Hydrogen peroxide has little efficacy against larval sea lice and its effectiveness against preadult and adult stages has been inconsistent (Mitchell and Collins, 1997). Effectiveness can be difficult to determine on farms as the treatment concentration varies due to highly variable volumes of water enclosed in the tarpaulin. Temperature and duration also influence the efficacy. Ovigerous females are less sensitive that other mobile stages (Treasurer *et al.,* 2000). It is possible that a proportion of the eggs on gravid female lice may not be viable after exposure to hydrogen peroxide (Johnson *et al.,* 1993). Hydrogen peroxide was less efficacious when treating sea lice infestation on salmon in a cage that had been treated regularly for 6 years than in cages where the sea lice were treated for the first time. This suggested that *L. salmonis* had developed some resistance to hydrogen peroxide (Treasurer *et al.*, 2000).

In a laboratory experiment, all adult and pre-adult sea lice exposed to 2.0 g/l hydrogen peroxide for 20 min became immobilized, but half had recovered two hours post-treatment (Bruno and Raynard, 1994). The recovered sea lice swam normally and may have been able to reattach to the host salmon (Hodneland *et al.,* 1993). Therefore it was the recommended that floating lice should be removed. However, re-infection has not been noticed in practice (Treasurer *et al.,* 2000) as the removed sea lice generally show little swimming activity. Re-infection in the field is less likely because the free sea lice will be washed away with the tidal flow or eaten by predators. After treatment of a cage with approximately 1.5 g/l hydrogen peroxide at 6.5 ^{o}C, all the sea lice that were collected from surface water of treated cages were inactive but recovery commenced within 30 minutes and 90–97% of the sea lice were active 12 hours post-treatment (Treasurer and Grant, 1997). In this study, a higher proportion of pre-adult sea lice was removed than of adult sea lice.

4.3.2 Distribution and Fate of Hydrogen Peroxide

Hydrogen peroxide is generally considered environmentally compatible because it decomposes into oxygen and water and is totally miscible with water. At 4 °C and 15 °C, 21% and 54% respectively of the hydrogen peroxide has decomposed after 7 days in sea water. If the sea water is aerated the amount decomposed after 7 days is 45% and 67%, respectively (Bruno and Raynard, 1994). Field observations suggest that decomposition in the field is more rapid, possibly due to reaction with organic matter in the water column, or decomposition catalyzed by other substances in the water, such as metals. In most countries, hydrogen peroxide is considered a low environmental risk and therefore of low regulatory priority.

4.3.3 Biological Effects of Hydrogen Peroxide

There is little information of the toxicity of hydrogen peroxide to marine organisms. Most toxicity data are related to the potential effects on salmonids during treatment of sea lice infestations. Experimental exposure of Atlantic salmon to hydrogen peroxide at varying temperatures shows that there is a very narrow margin between treatment concentration (0.5 g/l) and that which causes gill damage and mortality (2.38 g/l) (Kiemer and Black, 1997)

Toxicity to fish varies with temperature; for example, the one hour LC50 to rainbow trout at 7°C was 2.38 g/l, at 22°C was 0.218 g/l (Mitchell and Collins, 1997) and for Atlantic salmon increased five fold when the temperature was raised from 6°C to 14°C (Roth *et al.,* 1993c). There was 35% mortality in Atlantic salmon exposed to hydrogen peroxide at 13.5°C for 20 min. There was a rapid increase in respiration and loss of balance, but if the exposure was at 10°C there was no effect (Bruno and Raynard, 1994). Hydrogen peroxide is not recommended as a treatment for sea lice infestations at water temperatures above 14°C. Whole bay treatments in the winter should reduce the need for treatments in the summer (Rach *et al.,* 1997).

The method of application of hydrogen peroxide is not standardized but is a balance between achieving consistently effective treatments and toxicity to fish. For example, high concentrations were used (2.5 g/l for 23 minutes) to treat a farm for 6 years, which achieved 63% removal of sea lice. Exposure periods longer than this were the used in an attempt to increase removal, but caused 9% mortality in the salmon (Treasurer *et al.,* 2000). There is evidence that the concentrations of hydrogen peroxide used in sea lice treatments can cause gill damage and reduced growth rates for two weeks post treatment (Carvajal *et al.,* 2000).

5. IN-FEED TREATMENTS

5.1 Avermectins

5.1.1 Efficacy and Mechanism of Action of Avermectins

Two avermectin compounds have been used to treat sea lice infestations. Ivermectin and emamectin benzoate (SLICE®) are semi-synthetic derivatives of a chemical produced by the bacterium, *Streptomyces avermitilis*. Ivermectin is manufactured by Merck, Sharp and Dohme (MSD) and the company has made it clear they do not wish to have the product licensed for use as an anti-louse treatment (Davies and Rodger, 2000). In Canada, ivermectin has been used under veterinarian prescription to treat sea lice as an 'off-label' drug treatment under veterinary prescription (Burridge, 2003). This means the drug (and product) has regulatory status from Health Canada but is not registered for the specific treatment. In the UK and Europe a similar regulation exists (the Cascade Principle) by which veterinarians can prescribe ivermectin if no other effective licensed product is available. The subsequent availability of emamectin benzoate as a treatment against sea lice infestations should eliminate the need for the use of such 'off-label' prescriptions.

Emamectin benzoate, Slice® has been available in Canada as an Emergency Drug Release (EDR) from Health Canada since 1999 and is used to treat salmon against sea lice in eastern Canada. (DI Alexander, Health Canada. Veterinary Drugs Directorate, personal communication). SLICE is registered for use in the UK, Norway, Ireland, Iceland, Chile and the Faroes**.**

The avermectins are effective in the control of internal and external parasites in a wide range of host species, particularly mammals (Campbell, 1989). The avermectins generally open glutamate-gated chloride channels at invertebrate inhibitory synapses. The result is an increase in chloride concentrations, hyperpolarization of muscle and nerve tissue, and inhibition of neural transmission (Roy *et al.,* 2000; Grant, 2002)). Avermectins can also increase the release of the inhibitory neurotransmitter γ-amino-butyric acid (GABA) in mammals (Davies and Rodger, 2000).

Ivermectin is effective against chalimus as well as adult stages of the parasite giving it a wider efficacy than the organophosphates and hydrogen peroxide (Johnson and Margolis, 1993) (Davies and Rodger, 2000). The 'standard treatment' is 0.1 mg/kg divided into two treatments of 0.05 mg/kg separated by 3 or 4 days (Palmer *et al.,* 1987). This treatment regimen reduced the numbers of sea lice by up to 93% (Smith *et al.,* 1993). When fish were treated weekly at a dose of 0.02 mg/kg for 3 months, ivermectin was shown to be effective in preventing re-infection for about 4 weeks after the termination of treatment (Johnson and Margolis, 1993).

The optimum therapeutic dose for emamectin benzoate is .05 mg/kg fish/day for seven consecutive days (Stone *et al.,* 1999). This dose has been shown to reduce the number of motile and chalimus stages of *L. salmonis* by 94–95% after a 21 day study period (Stone *et al.,* 1999; Ramstad *et al.,* 2002). Four cage sites with a total of 1.2 million first year class fish were treated. Although there was a slight depression of appetite at two of the four sites, appetite was normal when top-up rations were supplied. *Caligus elongatus* were present in low numbers and results suggested that they were also affected by the treatment. The number of motile lice was reduced by as much as 80% at the end of the 7–day treatment period. In a field trial emamectin benzoate reduced sea lice counts on treated fish by 68–98% and lice numbers remained low compared to control fish for at least 55 days (Stone *et al.,* 2000a; Stone *et al.,* 2000b).

5.1.2 Distribution and Fate of Avermectins

Ivermectin reaches the marine environment in one of two ways: on uneaten feed pellets or as waste products from the fish (faeces and biliary excretion). Ivermectin is depurated from fish in two phases: an initial biliary excretion of unchanged ivermectin followed by a slower excretion after enterohepatic cycling (Davies and Rodger, 2000). A relatively high proportion of ivermectin passes through the gut unabsorbed (Hoy *et al.,* 1992). Ivermectin has a low solubility in water and a strong affinity to lipid, soil, and organic matter (Davies and Rodger, 2000). It is readily photo-degraded, but the half life for hydrolysis in the dark is quite long (Grant and Briggs, 1998). Within the marine environment, ivermectin is expected to be associated with sediments and particles and to show low mobility. The half life of ivermectin in sediment is at least 3 months (Davies *et al.,* 1998). The octanol-water partition coefficient for ivermectin is 1651 (Halley *et al.,* 1989) and the calculated bioconcentration factor of ivermectin is 74 for fish and 750 for mussels (Davies and Rodger, 2000). A “withdrawal period” of 1000 degree days for the elimination of ivermectin from edible tissue of was suggested prior to harvesting the Atlantic salmon (Roth *et al.,* 1993a).

Emamectin benzoate also has low water solubility and relatively high octanol-water partition coefficient, indicating that it has the potential to be absorbed to particulate material and surfaces and that it will be tightly bound to marine sediments with little or no mobility (Scottish Environmental Protection Agency, 1999b). The half life of emamectin benzoate is 193.4 days in aerobic soil and 427 days in anaerobic soil. In field trials, emamectin benzoate was not detected in water samples and only 4 of 59 sediment samples collected near a treated cage had detectable levels of emamectin benzoate. The emamectin benzoate persisted in the sediment; the highest concentration was measured at 10 m from the cage 4 months post-treatment. In Canada, emamectin benzoate was not detected in sediment samples collected near an aquaculture site for 10 weeks after treatment with SLICE® (W.R Parker, Environment Canada, personal communication) Mussels were deployed and traps were set out to capture invertebrates near aquaculture sites undergoing treatment with emamectin benzoate. While detectable levels of emamectin benzoate and metabolites were measured in mussels (9 of 18 sites) one week after treatment, no positive results were observed after 4 months (Scottish Environmental Protection Agency, 1999b). Emamectin benzoate was found in crustaceans during and immediately after treatment. Species showing detectable levels of emamectin benzoate for several months after treatment are scavengers which are likely to consume faecal material and waste food. The withdrawal period prior to slaughter of salmon in Canada is 25 days (DI Alexander, Health Canada. Veterinary Drugs Directorate, personal communication).

Biological Effects of Avermectins

A body of literature exists for LC50s and LD50s for ivermectin to fish and marine invertebrates (Davies and Rodger, 2000). Unfortunately, very few of these studies involve exposure of test organisms to ivermectin either in feed or in the sediments. Most researchers have exposed experimental animals through immersion in spiked water.

Over a 27 day period, there was a cumulative mortality of 10% and 80% of the Atlantic salmon (wt = 800g) exposed to 0.05 and 0.2 mg/kg ivermectin in food, respectively (Johnson *et al.*, 1993). Atlantic salmon was the most sensitive of several salmonid species tested and behavioral changes, such as cessation of feeding and lethargy, were observed in fish exposed to lower concentrations. The 96 h LD50 was 0.5 mg/kg for Atlantic salmon administered ivermectin by intubation and the 96 h LC50 was 17 µg/l when the salmon were immersed in a sea water solution of ivermectin (Kilmartin *et al*., 1997).

Sand shrimp (*Crangon septemspinosa*) were exposed to fish feed treated with various concentrations of ivermectin for 96 h in running seawater (Burridge and Haya, 1993). When the food was accessible to the shrimp, mortality occurred. When the feed was present in the water but not accessible by the shrimp, no mortality occurred, suggesting that the feed must be ingested by the shrimp before lethality occurs. The nominal 96-h LC50 was 8.5 mg/kg food and the No Observed Effect Concentration (NOEC) was 2.6 mg/kg food.

The 10-day LC50 for ivermectin in sediment to the marine amphipod, *Corophium valuator* was estimated to be 180 µg/kg dry weight (Davies *et al.,* 1998). The NOEC was 50 µg/kg. The 10-day LC50s to *Arenicola marina* and *Asterias rubens* were 23 and 23 600 µg/kg dry weight, respectively (Thain *et al.,* 1997).

Toxicological studies have shown that emamectin benzoate is less toxic than ivermectin in all taxa tested (Scottish Environmental Protection Agency, 1999b). The treatment concentrations of emamectin benzoate on salmon feed range from 1 to 25 µg kg^{-1} (Roy *et al.,* 2000). Feeding emamectin benzoate to Atlantic salmon and rainbow trout at up to ten times the recommended treatment dose resulted in no mortality. However, signs of toxicity, lethargy, dark coloration and lack of appetite were observed at the highest treatment concentration.

The lethality of emamectin benzoate treated fish feed to adult and juvenile American lobsters is estimated as 644 and >589 µg/kg of feed, respectively (Burridge *et al.,* 2004). The lethality of emamectin benzoate to other aquatic invertebrates (for example, *Nephrops norvegicus* and *Crangon crangon*) was >68 mg/kg (Scottish Environmental Protection Agency 1999b). In laboratory studies, prawns and crabs were offered feed medicated with emamectin benzoate at concentrations up to 500 mg/kg (Linssen *et al.,* 2002). There was no acute mortality. However, the crabs appeared to avoid medicated feed pellets. Ingestion of emamectin benzoate induced premature molting of lobsters

(Waddy *et al.,* 2002b). This molting response of lobsters to emamectin benzoate may involve an inter-relationship of a number of environmental (water temperature), physiological (molt and reproductive status) and chemical (concentration/dose) factors (Waddy *et al.,* 2002a). In a 7 day subletal test, there was significant reduction of egg production in the adult marine copepod, *Acartia clauii* (Willis and Ling, 2003) The concentrations necessary to elicit these responses were above the Predicted Environmental Concentration (PEC) (Willis and Ling, 2003).

5.2 Chitin Synthesis Inhibitors

5.2.1 Efficacy and Mechanism of Action Chitin Synthesis Inhibitors

Chitin synthesis inhibitors belong to a class of insecticides collectively referred to as insect growth regulators and have been used in terrestrial spray programmes for nuisance insects since the late 1970s. Two of these, diflubenzuron (Lepsidon®) and teflubenzuron (Calicide®, UK; Ektobann®, Norway) were approved as additives in feed to treat sea lice infestations of cultured salmon in Norway (1997) and Scotland (1999). Teflubenzuron use is approved in Ireland and Canada but there has been no recorded use in 2002–2003.

Chitin is the predominant component of the exoskeleton of insects and crustaceans, and the biochemical mechanism by which these insecticides inhibit the synthesis of chitin is unclear (Savitz *et al.,* 1994). The molting stage is the sensitive stage of the life cycle and inhibition of chitin synthesis interferes with the formation of new exoskeleton in a post-molt animal, for example, post-molt blue crabs (Walker and Horst, 1992; Horst and Walker, 1995). Thus the chitin synthesis inhibitors are effective against the larval and pre-adult life stages of sea lice.

Teflubenzuron is effective against *L. salmonis* at a dose to salmon of 10 mg/kg body weight per day for 7 consecutive days at 11–15^{o}C (Branson *et al.,* 2000). Teflubenzuron at this dosage was used to treat commercial salmon farms in Scotland and Norway, and the efficacy was 83.4 and 86.3 % respectively, measured at 7 days post treatment. There were no toxic effects on treated fish or effects on appetite of the fish. In a Norwegian field trial of salmon in a polar circle with 100,000 kg of salmon, the efficacy for a dosage of 8.1 mg/kg body wt/day for 7 days was 77.5% at 5.4^{o}C (Ritchie *et al.,* 2002). The greatest reductions were in chalimus and pre-adult lice and the efficacy was 88% if the calculation was based only on the susceptible life stages of *L. salmonis*. The effects were observed up to 26 days after start of the treatment. A few Norwegian sites successfully used teflubenzuron (Calicide®) in 1997 to remove all developing stages and the sea lice did not return during the further year's growth cycle (Scottish Environmental Protection Agency, 1999a).

Since chitin synthesis inhibitors are effective against the developing copepodids, larval (chalimus) and pre-adult stages of sea lice and less effective against adult lice, treatments are most effective before adult lice appear, or at least are present in only low numbers. In some cases, a prior bath treatment with organophosphates may be useful to remove adult lice or to control recruitment. When used correctly, chitin synthesis inhibitors provide a treatment option that breaks the life cycle of the sea lice and, as a result, the duration between treatments may be several months.

Distribution and Fate of Chitin Synthesis Inhibitors

Teflubenzuron and diflubenzuron have moderate octanol-water partition coefficients and relatively low water solubility, which means that they tend to remain bound to sediment and organic materials in the environment. They are not persistent in fresh water (Fischer and Hall, 1992; Scottish Environmental Protection Agency, 1999a) and a few marine studies suggest that sediment is a significant sink for these compounds in the marine environment.

In a field study, a total of 19.6 kg of teflubenzuron was applied over a 7 day period to treat a salmon cage with a biomass of 294.6 tonnes (Scottish Environmental Protection Agency, 1999a). Teflubenzuron was not detected in the water after treatment and highest concentrations in the sediments were found under the cages and decreased with distance from the cage in the direction of the current flow. It persisted in sediments for at least six months and the half-life was estimated at 115 days. Measurable levels were noted for a distance of 1000m in line with the current flow, but 98% of the total load had degraded or dispersed by 645 days after treatment. There was some indication of re-suspension and redistribution of sediment after several weeks based on concentrations of teflubenzuron found in mussel tissues. Evidence suggested that that there was some risk to indigenous sediment dwelling crustaceans, such as edible crab or Norway lobster, that may accumulate teflubenzuron from the sediment. However, the mussels eliminated teflubenzuron readily.

Diflubenzuron was found to be stable and persistent in anoxic marine sediments under laboratory conditions. There was no significant decrease in concentration (38 and 50 µg/g) after 204 days for diflubenzuron in sediments held in the dark at 4 and 14^{o}C or in sediments in tanks that were flushed with sea water (Selvik *et al.,* 2002). In a field study, salmon were fed medicated feed for 14 days. The concentrations of diflubenzuron found in the sediment did not reflect the high concentrations found in the sediment traps and accounted for only 15% of the total input. It was suggested that the feed and faeces at the sediment surface may have been re-suspended and transported to farther than anticipated or that faulty sampling of the sediment by the grab had lead to under estimation of the amount of diflubenzuron in the sediment near the farm(Selvik *et al.,* 2002). Evidence from soil studies suggests that diflubenzuron may be metabolized by bacteria in sediment (Finkelstein *et al.,* 2001).

In an estuarine microcosm system, an initial concentration of diflubenzuron of 140 µg/l decreased slowly over 3 weeks with a half life of >17 days in one study but only 4 days in another (Fischer and Hall, 1992). Laboratory and field studies demonstrated the importance of substrate in the fate and persistence of diflubenzuron in estuarine systems. Crab larvae exposed in a salt water system to a single treatment of technical grade diflubenzuron resulted in total mortality to solutions that were 1–42 days old and decreased from 86% to 5% for solutions that were 50–59 days old. In a salt marsh treated three times with 45g/ha of diflubenzuron at 2-week intervals, the concentrations of diflubenzuron that remained were >0.4 µg/l in the salt water one week post treatment while concentrations in sediment remained above 100 µg/l. The half-life in the microcosm system containing sediment was 5.3 days due to absorption of diflubenzuron by highly organic matter and 17.8 days in the absence of sediment.

The absorption of teflubenzuron from the gastrointestinal tract of salmon has been found to be poor, with only around 10% of the administered dose being retained by salmon and 90% being released by the fish via feces as well as the uneaten portion of the feed (Scottish Environmental Protection Agency, 1999a). In general, fish accumulate diflubenzuron rapidly during acute exposures but will eliminate the compund within 7days (Fischer and Hall, 1992). For example, the freshwater fish, white crappies, accumulated diflubenzuron from water to 264 ng/g wt weight but after 24 hours in clean water the concentration had fallen to 8 ng/g wet weight (Shaefer *et al.,* 1979). The deposition of diflubenzuron and teflubenzuron, in the vicinity of the treated cage is primarily from waste feed, with a more widespread distribution arising from the dispersion of fecal matter that may extend to 100 m from cages in the direction of the current flow (Scottish Environmental Protection Agency, 1999a).

5.3 Biological Effects Chitin Synthesis Inhibitors

Although teflubenzuron is relatively non-toxic to most marine species (birds, mammals and fish) due to its mode of action, it is potentially highly toxic to any species which undergo molting within their life cycle (Eisler, 1992; Scottish Environmental Protection Agency, 1999a). This includes some commercially important marine animals such as lobster, crab, shrimp and some zooplankton species.

Aquatic toxicity data for diflubenzuron has been compiled for 15 estuarine and marine species, mostly invertebrates (Fischer and Hall, 1992). The premolt stage of grass shrimp was the most acutely sensitive to diflubenzuron (96 h LC50 = 1.1 µg/l) and the mummichog was the most resistant species (96 h LC50 = 33 mg/l). Exposure of a marine harpacticoid copepod indicated that concentrations of diflubenzuron as low as 1.0 µg/l cause adult mortality and inhibited reproduction. The viability of *Acartia tonsa* nauplii to hatch was reduced to <50% during a 12 hour exposure to 1µg/l of diflubenzuron. When brine shrimp were exposed to 2 or more µg/l of diflubenzuron the reproductive life span and numbers of broods produced were significantly less than in controls. The 96 h LC 50 to various life stages of grass shrimp are: larvae, 1.44µg/l; post-larvae, 1.62µg/l and adult, >200µg/l. There was 60% mortality of the resident grass shrimp in a tidal pool treated with 45g/ha diflubenzuron. The borrowing behavior of fiddler crab was significantly reduced by exposure for more than one week to >5.0µg/l of diflubenzuron. However, there was 100% mortality of stone crab larvae exposed to 5.0µg/l; 95% mortality of the blue crab exposed to >3.0 µg/l; 46% mortality of juvenile blue crab after treatment of the tidal pool to 3.6µg/l at one hour after treatment. The lowest reported chronic effect concentration for a salt water organism exposed to diflubenzuron was 0.075µg/l, which significantly reduced reproduction in mysid shrimp.

A secondary effect of diflubenzuron on fish populations has been shown in a littoral enclosure, but not in the open marine environment. Exposure to diflubenzuron (2.5 µg/l) in littoral enclosures adversely affected reproductive success by reducing growth of bluegill larvae by 56 and 86% (Tanner and Moffett, 1995). This reduction in growth was an indirect effect by eliminating or reducing preferred bluegill larvae food (cladocerans and copepods). Decreases in growth of the food larvae may lead to greater starvation, increased predation and lower over-winter survival, which may result in poor recruitment. It was suggested that the early–stage bluegill larvae are more sensitive because this is when their growth rate is most rapid and at this first feeding stage are more selective because of their small mouth and poor swimming activity. The calanoid copepod, *Eurytemora affinis*, is widely distributed in North America and Europe (Savitz *et al.,* 1994), for example in Chesapeake Bay it makes up 20% of the total annual zooplankton crop and is major prey item for white perch and stripped bass larvae. The 48 hour LC50 of diflubenzuron (Dimilin®) to *E. affinis* is 2.2 µg/l. Other studies indicate that diflubenzuron concentrations less than 1.0 µg/l have substantial effects on survival, growth and production of nauplii (Wright *et al.,* 1996).

In a field study, no adverse effects were detectable in the benthic macrofaunal community or indigenous crustaceans and it was concluded that residual teflubenzuron in sediment was not bioavailable (Scottish Environmental Protection Agency, 1999a). There was some evidence of effects on the benthic fauna within 50 m of the treated cages, but no adverse impacts on community structure and diversity including important key sediment re-worker species and crustacean populations. A study at three locations in Scotland included a novel biomonitoring technique whereby juvenile lobster larvae were deployed on platforms at locations around cages. The juvenile lobster mortality was attributed to exposure to the medicated feed at 25m from the cage, but this effect did not occur 100 m from the cage, and it was confirmed that a molt occurred during the study. It was concluded that the "predicted no effect concentration" would not be exceeded 15m from cages. Since crustaceans are largely absent within 15 m of cages, and evidence suggests that teflubenzuron is relatively non-toxic to sediment re-worker organisms such as polychaete worms, the environmental risks in the use of teflubenzuron in the treatment of sea lice infestations were considered to be low and acceptable.

6. RISK ASSESSMENT

RISK ASSESSMENT OF SEA LICE THERAPEUTANTS

All the countries involved in the marine cultivation of salmonid fish operate systems for the regulation of therapeutic chemicals that are needed as components of strategies to control disease. In most cases, an authorization or licence issued by the relevant authority is required before a chemical can become available for use on fish farms. The ecological risk posed by the proposed use of a chemical is normally assessed during the process leading to decisions on the granting of the appropriate licence. Therefore, licencing procedures are an important element of the risk management process relating to the use of sea lice treatment chemicals

For members of the European Union an essential pre-requisite to the use of a medicine on food fish is the granting of a Maximum Residue Limit (MRL) by the European Medicines Evaluation Agency (EMEA), by being annexed to EC Council Regulation 2377/90. The MRL protects the consumer from any possible adverse effects of residues of medicines that might be present in fish presented for consumption. The EMEA seeks to further harmonize medicine authorization and assessment procedures within Europe and beyond. However, applications for Marketing Authorizations are still most commonly made to national authorities rather than directly to the EMEA. Exceptionally, the EMEA Committee for Veterinary Medicinal Products (CVMP) can assess applications for MAs. CVMP has issued European Guidelines for environmental risk assessment of veterinary medicinal products. The EU is party to an initiative (International Co-operation on Harmonisation of Technical Requirements for Registration of Veterinary Medicinal Products, VICH) to harmonise technical requirements between the EU, USA, and Japan.

In Canada, chemicals that are applied as feed additives or by injection are classified as drugs and are approved by the Bureau of Veterinary Drugs (BVD) of Health Canada under the Food and Drugs Act. The BVD is required to ensure that drugs offered for use on animals are safe and effective and do not leave residues in the products that pose a health risk to the consumer. Chemicals applied topically or as bath treatments are classified as pesticides, and are the responsibility of the Pest Management Regulatory Agency (PMRA) of Health Canada and are registered under the Pest Control Products Act. In all countries authorization to apply approved therapeutants for sea lice infestations ultimately requires a veterinary prescription.

Ecological risk assessment is a process for objectively defining the probability of an adverse effect to an organism or collection of organisms when challenged with an environmental modification such as climate change, xenobiotic exposure , infection with a disease organism or some other potential stressor (Roberts Jr *et al.,* 2001). Sea lice therapeutants have the potential to negatively impact the environment through effects on sensitive non-target organisms. There may be a significant body of information relevant to efficacy and safety that is known only to the regulatory authorities and the specific manufactures (Alderman *et al.,* 2004). The absence of these data from the public domain has the unfortunate consequence that neither its quality nor its nature can be debated by those scientists and non-scientists with interests in these areas. Critical evidence quantifying the extent of such impacts when the agents are employed under the conditions of a commercial fish farm is limited. Anti-lice treatments have the potential to significantly alter the population structures of the fauna in the immediate environments.

The central problem presented by anti-lice treatments is their lack of specificity. The properties of lice that present specific site for action of any anti-lice therapeutants are not unique to these lice (Alderman *et al.,* 2004). In particular other crustaceans such as lobster crab, and shrimp may be affected. For example, in the cold waters of the Bay of Fundy hatching of lobsters occurs in July to September (Campbell, 1986) and larval production has been observed as late as September. The larval stages (stage I, II, III) of the lobster are pelagic. The first post larval stage (stage IV) spends at least some of its time in the water column prior to settling to the bottom (Charmantier *et al.,* 1991). It is possible that treatment of lice infested fish and release of pesticide formulations could coincide with the presence of lobster larvae in the water (Burridge *et al.,* 2000a).

The details of the scope of the environmental information required by regulatory authorities varies from country to country. For clarity, one country will be taken as a detailed example. Under UK legislation, any compound applied to an animal for the purpose of disease control, is classified as a medicine, and is licenced under the relevant medicines legislation. A pharmaceutical company seeking approval for a new medicine must show that the medicine is effective (for the purpose for which it is being proposed), of good quality and safe. In the context of this paper. "Safety" includes safety for the animal being treated, for the user and, in the case of food animals, for the consumer (subject to appropriate withdrawal periods). Environmental safety is particularly important for medicines to be used in aquaculture.

The UK operates a tiered approach to the assessment of the environmental safety (Table A7.2), (UK Veterinary Medicines Directorate, 2004)). Applicants are required to present a dossier on the potential risks for the environment resulting from the use of the product. This must include ecotoxicological information, supported by an expert report, to assess the potential harmful effects which the use of the product may cause to the environment and to identify any precautionary measures which may be necessary to reduce such risks. For medicines used in fish farming which will or are likely to enter surface waters, the submission of an ecotoxicity dossier comprising the Phase I and Phase II assessment will always be applicable. An ecotoxicity dossier will also be necessary when applying for an animal test certificate (ATC) to conduct trials using fish medicines.

A progressive, stepwise approach to testing is described, with the data required at one tier being dependent on the results of testing at the previous tier. Where sufficient data are available at any one tier of testing for the environmental risk to be adequately assessed, then there will be no need to conduct further tiers of testing. A general risk assessment strategy is to estimate the predicted environmental concentrations (PEC) that will result from the use of the agent. The estimated PEC is compared to the predicted no effect concentrations (PNEC) derived from toxicity studies with relevant species. Dispersion /advection models for dissolved substances may be used to estimate PEC (Burridge *et al.,* 2000a) (Henderson and Davies, 2001).

The route of administration of the sea lice therapeutants is an important in determining the factors to consider in the risk assessment process of these chemicals. Bath treatments results in the direct release of a solution of the therapeutant and thus the dilution rate (dispersion), spread and direction of flow of the plume (advection) and the life history of sensitive species in the water column are important factors. With medicated feed the critical factors are sedimentation rate of excess food pellets and feces as well as the bioavailabilty of the therapeutant from these particles.

Bath treatments:

The fate and dispersion of cypermethrin, azamethiphos and a dye, rhodamine were determined after simulated bath treatments from a salmon aquaculture site under various tidal conditions in the Bay of Fundy, Canada (Ernst *et al.,* 2001). Dye concentrations were detectable for periods after release which varies from 2–5.5 hours and distances ranged from 900 to 3000 meters depending on the location and tidal flow at the time of release. Concentrations of cypermethrin in the plume reached 1–3 orders of magnitude below the treatment concentration 3–5 hours post release and indicated that the plume retained its toxicity for substantial period of time after release. Water samples collected from the plume were toxic in a 48 hour lethality test to *E. astuarius* for cypermethrin up to 5 hours after release. When azamethiphos was released, none of the water samples from the plume were toxic after 20 minutes. There have been a number of studies where lobsters and shrimps were held in cages near fish pens during treatment for sea lice. It is not known if the caged animals were exposed to the plume from the released bath treatment however the experiments do provide some circumstantial evidence.

Bath treatments require considerable human effort and usually there is only enough staff to treat one cage at a time and up to three cages per day. Thus it is possible that indigenous species could be exposed periodically for several hours to plumes of the released bath treatment from the same aquaculture site or possibly from several sites in the same area.

Organophosphates

Azamethiphos is water soluble and remains in the aqueous phase on discharge to the receiving waters. Azamethiphos decomposes by hydrolysis with a half-life in nature of 8.9 days. Dispersion studies indicated that, after release of the treatment solution the concentration of azamethiphos falls to below detection (0.1 μg/l) in a short period of time. The compound was not detected below 10 m depth. Thus, azamethiphos is unlikely to accumulate in indigenous species or in sediment.

Only lobster larva and shrimp have shown sensitivity to concentrations below treatment, but lethality from a single exposure has been over 48 to 96 hours. Several field studies found no acute effects on caged lobsters held near treated sites or in lobster populations in areas that have been treated for some time.

Laboratory studies have demonstrated sublethal effects on lobster reproduction and mobility from repeated short term exposure to concentrations of azamethiphos below the recommended treatment concentration (10 μg/l for 1 hr). Repeated exposures to higher concentrations could result in significant mortalities and drastic changes in activity level and ability to function normally. The consequences of lethargy or of becoming moribund in the wild are probably severe to individuals, but the number of individuals within the zone of impact during sea lice treatments is likely to be few. Therefore, the risk of ecological effects of azamethiphos being manifested during operational application of the pesticide appears to be quite small. Individual (pelagic) organisms caught in the effluent plume from a bath treatment are likely to be affected but it unlikely that large scale or population effects will occur. Azamethiphos is a moderate risk to individuals of sensitive species but a low risk to populations. However, azamethiphos is not considered the treatment of choice because of the development of resistance to organohosphates by sea lice.

Pyrethroids

The pyrethroids, cypermethrin and deltamethrin are not persistant in marine waters. Both have relatively short half-lives in water and concentrations in the water decreased rapidly (<4h) in some field trials due to decomposition and partitioning to particulate matter. In sediments, the compounds are more persistent with half-lives up to 80 d, and cypermethrin was detected in sediment surveys in near salmon aquaculture sites in Scotland (Scottish Environmental Protection Agency, 2004). However, bioavailability of pyrethroids from sediment is minimal.

Cypermethrin has the potential to release lethal plumes from a single cage treatment. The plume can cover up to a square Km and lethality to sensitive speciecies can last as long as 5 hours. Since treatment of multiple cages is the operational norm, area wide effects of cypermethrin on sensitive species cannot be discounted. Sensitive species include crustaceans such as lobster larvae, shrimp and crabs and the 96 h LC50 for some can be a magnitude less than the treatment concentration. No lethality was observed in shrimp and lobsters deployed in cages during sea lice treatments with cypermethrin.

As with the organophosphates, the development of resistance to pyrethroids by sea lice has been demonstrated. A region in Norway with resistant sea lice had an EC50 which was 25 times higher then that for an area that had not been treated previously with deltamethrin.

Evidence suggests that there is considerable risk to individuals of sensitive species but there is insufficient knowledge to extrapolate to populations. There is sufficient evidence on the development of resistance to advice against routine use of pyrethroids as only means of control. Pyrethroids are not authorized for use in North America for treatment of sea lice infestations. Part of the rationale may be the availability of an unrestricted agricultural product containing cypermethrin that could result in indiscriminate use in marine waters.

Hydrogen Peroxide

Hydrogen peroxide is non-persistent, environmentally friendly and readily dispersed in marine waters. It has negligible risk to marine organisms in the concentrations used for the treatment of sea lice infestations.

In-feed Treatments

Avermectins

The avermectins have low water solubility and are absorbed and tightly bound to particulate matter. They are persistent in sediments, for example the half life of emamectin benzoate is 194 days in aerobic sediment. Ivermectin and emamectin benzoate have been found in some of the sediments sampled near salmon aquaculture sites. Thus, there is a potential for accumulation in sediment and they may pose a risk to sensitive benthic organisms. However, the avermectins may only be absorbed by the benthic organisms if they consume the medicated feed. Emamectin benzoate has been found in crustaceans immediately following treatment and in scavengers several months after sea lice treatments. However, evidence suggests that the amount of medicated feed consumed after use in sea lice control is insufficient to cause mortality. Salmon feed is not a preferred food of crabs and lobsters. There were no effects on polychaete populations near salmon aquaculture sites after sea lice treatments with ivermectin (Black *et al.,* 1997) (Costelloe *et al.,* 1998).

Sublethal effects of emamectin benzoate have been observed in laboratory studies with American lobster and a marine copepod, *Acartia clauii*, however the concentrations required to elicit these responses were above the PEC. The consequence of these sublethal effects on wild populations is unknown. Emamectin benzoate is the preferred to ivermectin because it has a much shorter withdrawal time for salmon. The use of emamectin benzoate is permitted in Canada, Chile and several European countries, and in many cases is the treatment of choice. The use of emamectin benzoate in the treatment of sea lice infestations is also considered to have relatively low risk to the marine ecosystem.

Chitin synthesis inhibitors

The main environmental risk of the chitin synthesis inhibitors to marine environment is likely to arise from the deposition of fish feces and waste feed on the sediments below and around the cages(Scottish Environmental Protection Agency, 1999a). For example 90% of the teflubenzuron administered is not absorbed by salmon and is excreted as parent compound in feces. Chitin synthesis inhibitors are bound by sediment and organic material. They are persistent in sediment with half life estimates to 115 days and there is a moderate risk of build-up in sediment through repeated applications. Field studies suggest that excess feed accumulates near the treatment cages but there is widespread dispersion of the fecal matter that may extend greater than 100 meters from the cages, depending on water depth, current velocities, etc. Proper treatment strategies can reduce this risk by limiting the number of applications required.

Although chitin synthesis inhibitors are specific and of low toxicity to most non-target organisms, there are identified risks to any species that molts, for example crustaceans, that are located near to cage sites during treatment. A case can be made for possible environmental effects of the most sensitive species (pre-molt grass shrimp 96 h LC 50 1.1 μg/l) exposed to the highest reported environmental concentration (1.5 μg/l in water) (Fischer and Hall, 1992). However, based on the short half-life in water, it appears that the concentration decreases rapidly. In addition, there is negligible risk to organisms in the water column due to the tendency of chitin synthesis inhibitors to bind strongly to sediment and organic matter and they are not bioavailable from these bound forms unless ingested. Field studies with teflubenzuron did not detect any adverse effects in benthic biology or crustaceans near treatment sites. The limited data base suggests that the risk adverse environmental effects is minimal.

RISK MANAGEMENT

Registration of sea lice therapeutants after they have been assessed as having acceptable risk makes them available for use. However, farms are located in waters with different capacities to absorb wastes, including medicinal chemicals, without causing unacceptable environmental impacts. Risks therefore have site-specific component, and management of these risks may therefore require site-specific assessments of the quantities of chemicals that can safely be used at each site. The UK environmental authorities (primarily the Scottish Environment Protection Agency, SEPA) operate this further level of control on the use of medicines at fish farms, and provide an example of a risk management plan that should be adopted in all area that use sea lice therapeutants. A medicine or chemical agent cannot be discharged from a fish farm installation unless formal consent under the Control of Pollution Act has been granted to the farm concerned by (in Scotland) SEPA. The main components of the risk management process are:

- Environmental quality standards (Table A7.3) designed to protect non-target organisms in the receiving waters from adverse impacts of the chemical concerned. These are derived from analysis of available toxicological information, and appropriate safety factors.
- Site-specific information on the nature of the environment around the farm, particularly records of water currents over periods normally of at least 15 days.
- Mathematical modelling of the dilution and dispersion of soluble chemicals released at the farm, and of the distribution of particle-bound chemicals on the sea bed.
- The definition of an allowable zone of effect (AZE) around a farm within which the EQS may be breached. The AZE is normally equivalent to approximately 25m distance around the farm.

The output from the above assessment and modelling process is an expression of the maximum amount of a particular chemical that can be safely discharged from a specific farm over a defined period of time (Table A7.3). This is then included as a condition within the overall Discharge Consent for the farm. Exceedence of these quantities, or discharge of a chemical for which Consent has not been granted, would be a breach of the Consent and could lead to prosecution of the farm concerned. Full details of this process are available on the SEPA website (www.sepa.org.uk) in their Fish Farming Manual and in associated policy documents for individual sea lice treatments.

Conclusion

The nature and severity of the environmental risks presented by the use of rthe various chemicals available to control sea lice in farmed salmon vary considerably between treatment compounds. Current regulatory practices, particularly those leading to approvals/authorisations for the use of products for sea lice control, include elements of assessment of the risk to the environment. This is the primary process byt which the environmental risk is managed, ie through the decision where or not grant approval/authorisation, and under what conditions.

However, there are considerable differences between the environmental characteristics of fish farm sites and their ability to accept discharges of sea lice treatments without giving rise to unacceptable environmental impacts. For example, differences in tidal currents and other hydrographic factors to dilute and/or disperse chemicals. Such site-specific risks can be managed through the application of appropriate Environmental Quality Standards for the chemicals concerned, and site-specific assessment of the maximum acceptable rate of use of the treatments.

Table A7.1. Chemical therapeutants currently authorized in Europe and North America for the treatment of sea lice infestations in salmon mariculture.

Therapeutant	Quantity Used			
	Canada 2002	Norway 2003	Ireland 2003	Scotland
Bath				
Azamethiphos	15.0 kg			
Cypermethrin		59 kg	107.6 l	
Deltamethrin		16 kg		
Hydrogen Peroxide				
Medicated Feed				
Emamectin Benzoate	25.0 kg	23 kg	4.97 kg	
Teflubenzuron				
Diflubenzuron				

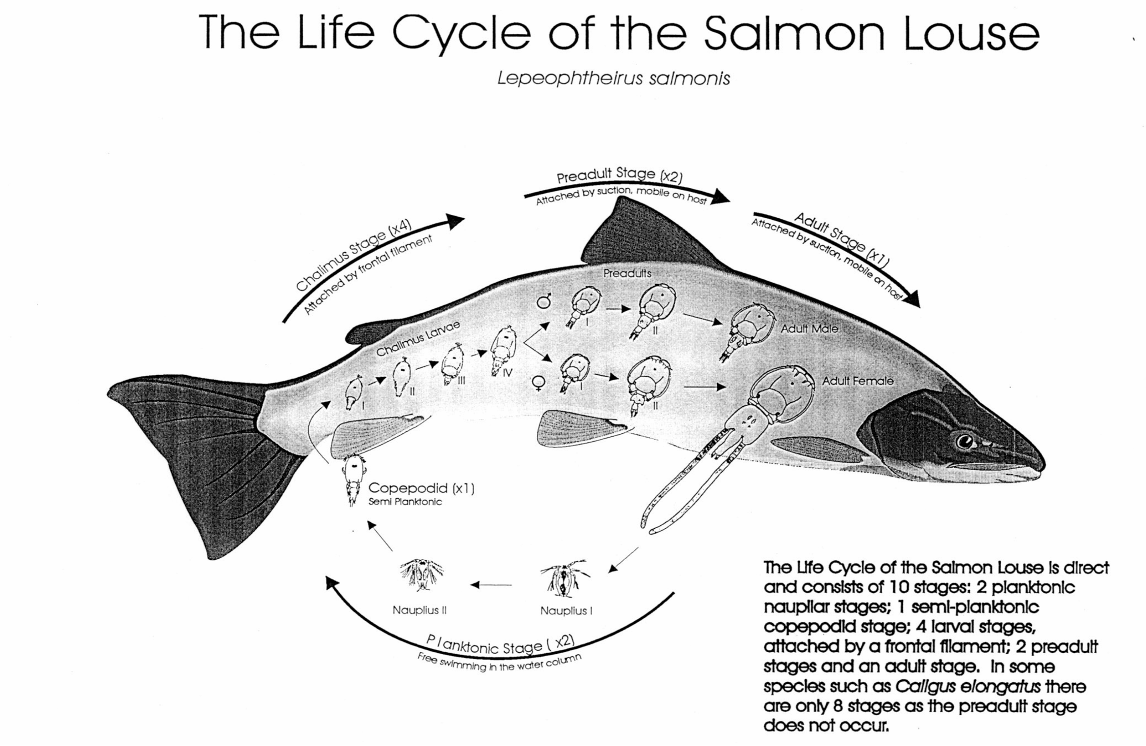
Canada: DI Alexander, Health Canada. Veterinary Drugs Directorate, personal communication
Norway: Norwegian Medical Depot
Ireland: Marine Institute, Galway
Scotland: Scottish Environmental Protection Agency, Dingwall, Scotland.

Table A7.2. Tiered approach to ecotoxicological testing of fish farm medicines in the UK

	Tier 1	**Tier 2**	**Tier 3**
PHYSICO-CHEMICAL PROPERTIES	- Molecular weight - UV/visible absorption spectrum - Melting point - Boiling point - Vapour pressure - Water solubility - Water dissociation constant - Octanol/water partition coefficient (Kow)	Sediment/water adsorption coefficient (if Kow is high)	No further requirements
FATE	- Hydrolysis half-life at pH5, 7 and 9 - Photolysis half-life	- Biodegradation mechanism and halflife in natural sediment-water test systems (if hydrolysis and photolysis slow) - Bioconcentration tests (if Kow is high and exposure likely to be long)	- Dispersion data - Outputs from computer models - Fate in sediments based on microcosms or mesocosms
BIOLOGICAL EFFECTS	- Acute toxicity to one species of juvenile or larval fish - Acute toxicity to one appropriate species and stage of larval crustacean - Toxicity to one species of micro alga	- Chronic fish and crustacean reproduction, earlylife-stage or growth tests (if prolonged exposure likely) - Acute toxicity to a macrophyte (if toxic to algae) - Acute toxicity to juvenile or larval molluscs (if of economic importance in area of use) - Acute and/or chronic toxicity to obligate sediment feeders (crustacea, molluscs or annelids)	- Mesocosm studies of effects on benthic fauna - Bioassays using sensitive taxa - Field investigations - Effects on microbial communities

TableA7.3. Environmental Quality Standards for fish farm medicines applied by the Scottish Environmental Protection Agency.

Active ingredient	Mode of application	Environmental Quality Standards
Azamethiphos	Bath	• Maximum allowable concentration (MAC), 3 hours after release, of 250 ng/l. • 24h MAC of 150 ng/l. • 72h MAC of 40 ng/l
Cypermethrin	Bath	• Short term (3 hour) EQS of 16 ng/l • Maximum allowable concentration (MAC) of 0.5ng/l, applied 24 hours after release • Annual average EQS is 0.05ng/l
Hydrogen peroxide	Bath	None – not considered to be a significant environmental risk
Teflubenzuron	In-feed	• 6.0 ng/l as an annual average in sea water and 30 ng/l as a MAC • 2.0 ug/kg dry wt/5 cm core depth as a general sediment quality standard to be applied as a MAC to surface sediment (cores 5cm depth) at more than 100m from the cages • 10.0 mg/kg dry wt/5cm core depth as a standard applied as an average value within the immediate under cage impact zone defined as surface area under and around cages to a distance of 25m from cage edges.
Emamectin benzoate	In-feed	• Concentrations in sediment should not exceed 0.763 ug kg^{-1} outside the AZE. • Concentrations in sediment should not exceed 7.63 ug kg^{-1} inside the AZE. • Concentrations in sea water should not exceed 2.2 x 10^{-4} ug l^{-1}. • Maximum number of treatments: • three treatments in any 12 calendar months, and • five treatments in any two year growth cycle.



The Life Cycle of the Salmon Louse
Lepeophtheirus salmonis
Preadult Stage (x2)
Attached by suction, mobile on host
Chalimus Stage (x4)
Attached by frontal filament
Adult Stage (x1)
Attached by suction, mobile on host
Preadults
Chalimus Larvae
Adult Male
Adult Female
Copepodid (x1)
Semi Planktonic
Nauplius II
Nauplius I
Planktonic Stage (x2)
Free swimming in the water column
The Life Cycle of the Salmon Louse is direct and consists of 10 stages: 2 planktonic nauplilar stages; 1 semi-planktonic copepodid stage; 4 larval stages, attached by a frontal filament; 2 preadult stages and an adult stage. In some species such as Caligus elongatus there are only 8 stages as the preadult stage does not occur.

Figure A7.1.

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Annex 8 Recommendations

The **Working Group on Environmental Interactions of Mariculture** [WGEIM] will meet for 5 days from 11–15 April 2005 under the Chairship of Dr Francis O'Beirn* (Ireland) in Ottawa, Canada to address the following terms of reference:

a) Continue to prepare a publication on the "state of knowledge" of the potential impacts of escaped aquaculture marine (non-salmonid) finfish species on local native wild stocks (e.g., sea bass, sea bream, cod, turbot, halibut) by;
   i) completing outstanding aspects of risk analyses of escapes of non-salmonid farmed fish (cod, sea bass, sea bream, halibut, turbot).
   ii) preparing risk analyses of releases/escapes of selected strains of farmed shellfish (e.g. Crassostrea gigas, Tapes philippinarum, Pecten maximus, Ostrea edulis, Mytilus edulis, Crassostrea virginica, Crassostrea ariakensis, Mytilus galloprovincialis, Mytilus trosilus)
   iii) preparing plans for publication of risk analyses of escapes/releases of fish and shellfish (items 1 and 2 above).

b) To continue development of risk analyses for new mariculture species in communication with and in relation to the GESAMP WG 31 development of aquaculture risk analysis methodologies;

c) To update report on developments in implementation of WFD and EU Strategy for sustainable aquaculture.

d) To consider recent developments in carrying capacity models for shellfish with a view to proposing a symposium in this area.

e) To receive a presentation and discuss the possibility for developing a "sustainability index" concerning environmental interactions of mariculture.

f) To receive a presentation on the current state of development of integrated culture systems (e.g., fish -invertebrate - seaweed co-culture) with a view to assessing the potential of polyculture to mitigate the environmental effects of mariculture.

Supporting information:

Priority:	WGEIM is of fundamental importance to ICES.
Scientific Justification and relation to Action Plan:	Action Plan Nos.: 2.4, 2.5, 2.6, 2.7, 2.10, 3.3, 3.7, 3.11, 3.14, 4.6, 4.7, 4.14, 5.16 a) In order to foster a sustainable development of coastal and marine aquaculture, there is a need to diversify production and to cultivate new species. A pro-active approach is required to avoid mistakes made previously when salmonid farming was developing. Mitigation strategies based on sound scientific criteria in relation to the species under consideration need to be prepared at an early stage of development. Studies would have to consider the status of the natural stocks in the area, the potential genetic, trophic and behavioural interactions, and, foremost and specifically, the development of methods for recovery of escaped fish in the event of large-scale escapements. This subject seems to be of particular importance for non-migratory fish stocks with small, localised populations (e.g., sea bass and seabream), or migratory species with different migratory patterns than salmonids (e.g., cod, halibut, turbot, and wolfish, and other species). The report will include an overall risk assessment and recommended mitigative strategie b) WGEIM would greatly benefit from inputs of the GESAMP WG31. Regulatory actions that limit the transportation and utilization of mariculture species can be viewed ans a non-technical barrier to trade under international trade agreements. Validation of that view permits punitive tarrifs and trade restrictions. Risk analysis is one method of identifying environmental risks associated with the utilization of new speioces in culture and of justifying environmentally based constriants on the transfere and use of the species. GESAM WG 31 is developing methodologies for analyzing environmental risks associated with aquaculture activities. Their application to the environmental risks associated with culturing new matriculture species will enable better science-based management of existing resources and allow integration of aquaculture into the existing mix of coastal resource users for member states. c) The EC policy on Sustainable Aquaculture sets a new context for the aquaculture industry in the EU. It holds out the possibility, among other things, that Integrated Coastal Zone Management will become the normal approach to the management of the

	aquaculture development, and that new tools and processes will arise from the new policy. The Water Framework Directive will determine the direction of water quality regulation and improvement in the EU over the next 10–20 years. The coincidence of major new policy initiatives in both industrial development strategy and environmental quality presents European aquaculture with a unique set of opportunities and risks. d) Interest in shellfish cultivation is expanding rapidly in several ICES countries, including UK, Canada and Norway. In areas which might have reached capacity for fish farming, perceived lower-impact shellfish farming can offer alternative development potential. As proposed developments increase in size from single-operator part-time ventures to larger, mechanised businesses, questions of carrying capacity arise. The last significant international symposium on shellfish carrying capacity was held around 6–8 years ago. The purpose of this agenda item is to review subsequent developments and to assess the opportunity for a further symposium. e) Acquiring and integrating large amounts of scientific information for environmental resource management creates challenging workloads for resource managers. Methods to triage or reduce the work necessary would improve the effective application of management resources. Sustainability indexes may offer a methodology for monitoring or prioritizing those systems most in need of immediate management attention. This would then allow scarce management assets to be applied in the most cost effective manner. f) Integrated aquaculture systems (encompassing a wide variety of types of multi-species systems) have been proposed as a direct way to utilise the wastes to create additional products of significant commercial/environmental value. Nutrients from fish farms could support algal production; solid wastes from fish farms support bivalve production, etc. Some practical developments are starting to occur, and the EU has supported work in this area. There is very considerable diversity of opinion on the actual and theoretical environmental.
Resource Requirements:	None required, other than those provided by the host institute.
Participants:	Representatives of all Member Countries with expertise relevant to the effects of the environment on aquaculture and aquaculture on the environment.
Secretariat Facilities:	None required
Financial:	None required
Linkages to Advisory Committees:	ACME
Linkages to other Committees or Groups:	MARC, DFC,
Linkages to other Organisations:	BEQUALM, OIE, EU, EAS
Secretariat Cost share	ICES:100 %