ICES WGEIM Report 2006

ICES Mariculture Committee
ICES CM 2006/MCC:03
Ref. ACME

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

24–28 April 2006

Narragansett, Rhode Island, USA

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

H.C. Andersens Boulevard 44-46
DK-1553 Copenhagen V
Denmark
Telephone (+45) 33 38 67 00
Telefax (+45) 33 93 42 15
www.ices.dk
info@ices.dk

Recommended format for purposes of citation:

ICES. 2006. Report of the Working Group on Environmental Interactions of Mariculture (WGEIM), 24-28 April 2006, Narragansett, Rhode Island, USA. ICES CM 2006/MCC:03. 195 pp.

Contents

Executive Summary

WGEIM met at the University of Rhode Island from 24–28, April 2006. The meeting was attended by 11 members and one observer from six countries. Six terms of reference were addressed by the group and are summarised below.

The WG continued to develop a series of documents concerning risk analysis of the consequences of genetic interaction between farmed fish and conspecific wild stocks, as part of a joint project with GESAMP WG31. The documents consist of an introductory paper describing the risk analysis approach, and case studies of five non-salmonid aquaculture species (cod, halibut, sea bass, sea bream and turbot). The WG agreed that the papers were nearing the standard and completeness necessary for submission for peer reviewed publication. The WG recommended that the documents be completed intersessionally and submitted to an appropriate journal.

The WG continued to review existing (EU) legislation or proposed legislation and assess the impacts of the legislation on mariculture activities. Under the Water Framework Directive, no obvious mariculture impacts were highlighted based upon the implementation activities within the intersessional period. Proposed legislation is the European Marine Strategy (EMS), the Strategy for Sustainable Development of European Aquaculture and amendments to the EU Data Collection Regulation all of which might have some impacts on aquaculture activites.

The group continued to investigate the applicability of sustainability indicators for aquaculture. SI's are different from "impact" indicators in that they are more comprehensive, including considerations of not only environmental but also social and economic sustainability. Sustainability indices (SIs) are needed by aquaculture resource managers who must sort through large amounts of scientific information and make numerous environmental decisions. SIs offer a means to prioritize those aquaculture systems most in need of immediate management attention and allow scarce management assets to be applied in the most cost-effective manner. SIs are also valuable for owners of seafood businesses who wish to procure "sustainable seafoods" for this rapidly growing consumer movement. The group considered and evaluated the current status and suitability of SIs for mariculture and selected a suite of SIs that are simple, flexible and cost effective. The group used a matrix approach that maps "sustainability trajectories" and was applied to salmon farming in New Brunswick, Canada and Norway. A number of recommendations were forthcoming from this term of reference. WGEIM will continue to refine the sustainability trajectories approach using the traffic light system of the UK and Canada and apply this example to salmon and shellfish aquaculture farming systems.

Integrated Multi-Trophic Aquaculture (IMTA) represents a global aquaculture sector of growing interest and potential development in the world. Although much of this interest has been expressed through ongoing research initiatives, there has been some movement towards commercialization through large-scale testing of these opportunities. The evolution from monoculture aquatic food production systems to integrated, multi-species systems is envisioned as a shift towards a sustainable approach, and one that has seen parallel developments occur in the terrestrial food production industries.

Results of research programs in North America and Europe would suggest that IMTA has high commercial potential, but that there are still some challenges remaining in terms of R&D, regulatory framework development, and product acceptability. Experimental and pilot-scale systems are providing growing evidence that there are low-level risks associated with contaminant transfers among Integrated Aquaculture components, and that these periodic risks are manageable in terms of husbandry practices and with appropriate regional/international regulatory requirements for seafood inspection. The legal frameworks that currently apply to the aquaculture industry, in most jurisdictions, are considered sufficiently flexible as to accommodate the development of Integrated Aquaculture systems and it is clear that the

environmental, economic and social benefits associated with this *sustainable* approach to aquatic food production outweigh the risks.

The potential advantages and disadvantages of integrated mariculture, based on our assessment of the environmental, social and economic considerations for this sector were presented. In addition, number of information gaps were highlighted and research and development initiatives were recommended.

One of the obvious short-comings of research to date is that studies on substitution of fish meal and fish oil have been mutually exclusive. Many of the promising results from substitution of fish oils with plant oils have been due, in part, to inclusion of high levels of fish meal in the same diet. The fish meal provides some of the essential fatty acids that would have normally been provided directly by the fish oil. Although other novel sources of essential fatty acids are available, they must become more economical before they can sustain the needs of the aquaculture industry. Nevertheless, great advances in reducing, if not eliminating, the reliance upon wild fisheries resources for aquaculture feed ingredients are being made.

The primary conclusion is that during the intersession WGEIM will carry out a review and evaluation of recent advances on alternative sources of lipid and protein to fish oil and fish meal in aquafeed. It is proposed that WGEIM review a draft manuscript at the 2007 meeting that is to be submitted for publication in a peer reviewed scientific journal.

To date, the interaction of mariculture with exotic species and more specifically unintentional species introductions has received limited attention. This is despite the fact that exotic species are having significant impacts on the aquaculture industry worldwide and more particularly for the shellfish aquaculture industry.

The importance of bivalve culture in the promotion and transfer of exotic aquatic species as well as the importance of these exotic species to bivalve culture and the environment. Specifically, we focused on exotic species with an emphasis on those that become invasive and nuisance. Management implications and mitigation strategies are also addressed. It should be noted that the majority of the existing literature addresses the issues as they relate to oyster culture, probably because this appears to be the single greatest vector for all types of introductions (planned or otherwise) in bivalve aquaculture. There is little published information about other bivalve species with respect to their role as vectors for exotic species.

1 Welcome and opening of the Meeting

Francis O'Beirn (Chair) opened the 2006 meeting of the Working Group on the Environmental Interactions of Mariculture (WGEIM) at the University of Rhode Island, Naragannset, Rhode Island from 24–28 April. This year's meeting was attended by 12 scientists from six countries (Annex 1). Included, as an observer, was the Chair of the Working Group on Pathology and Disease in Marine Organisms, Dr Sharon McClean.

The group was welcomed to the URI by the Dean of the Graduate School of Oceanography, Dr David M. Farmer who expressed his delight that the group was meeting at the University of Rhode Island (the first time in 19 years that the group has met in the United States of America). Dr Barry Costa-Pierce, as local host, continued the welcome and highlighted the extensive facilities that were available to the group throughout the week, including, inter alia, the library of the National Sea Grant Program and the Pell Marine Science Library. The Chair, on behalf of the group, expressed considerable gratitude to Dr Costa-Pierce and his assistance Heather Rhodes for their preparations and providing facilities for the meeting.

In the intersessional period, it was noted that a paper originally considered as a term of reference and prepared during WGEIM 2005 "Review of recent carrying capacity models for bivalve culture and recommendations for research and management" by Chris McKindsey, Helmut Thetmeyer, Thomas Landry and William Silvert was submitted for publication in, the Journal Aquaculture.

The working arrangements were described, whereby a series of sub-groups were formed each to address a specific term of reference. A sub-group leader was assigned, who would be responsible for compiling the contributions of the others within the group. There were no deviations from the list of members of the subgroups assigned prior to the meeting.

2 Adoption of Agenda

A draft agenda was circulated in advance the meeting and with minor modification was accepted by the group. The adopted agenda is presented in Annex 2.

3 Update on Workshop on Review of the ICES Committee and Expert Group Performance (WKREP)

The Chair took the opportunity to provide a short verbal report on the outcome of the Workshop on Review of the ICES Committee and Expert Group Performance (WKREP), which was held at the EEA in Copenhagen on 15 March 2006. The meeting was attended by 39 people however, many attendees were either from the secretariat or would have been there for ConC or MCAP meetings anyway. In total there were on 12 chairs of WGs or SGs or delegates. In the overview the Chair outlined the twin processes within ICES that are the Advisory and Science Programmes. The workshop was convened to discuss the structure of the Science program within ICES. It was acknowledged that the advisory structures within ICES appeared to be better organised and more streamlined that the science programs. However, given the requirement to provide integrated advice (i.e. the ecosystem approach) the harmonisation of the programmes should form an important component of any re-structuring within ICES.

Probably the most important outcome of the workshop was the conclusion that the expert group process appears to be functioning well, in the sense that attendances at the group meetings are good and the content of the reports is typically of a high standard. However, it was further pointed out that a constraint is the poor communication with the science program. This lack of communication is apparent at all levels, between the working groups, between the working groups and the science committees. In addition, it was highlighted that the role of the Science Committees appeared to ill-defined in that much of the time was focused upon administrative responsibilities and not on strategic issues pertaining to the subject area. One mechanism to improve the relevant communication flow and transfer of important information and recommendations to the committees is the production of Executive summaries from the WG reports. It was further suggested that Expert Group Chairs should have full member status on Science Committees. This was broadly supported by the attendees and may encourage greater participation and commitment at the Committee level. Communication between Expert Groups could also be achieved by promoting joint meetings (e.g. a one day overlap with plenary discussions on broader issues).

The report of the workshop was made available to the group and the Chair agreed to keep a watching brief on developments regarding restructuring proposals within ICES and to keep the Working Group informed.

4 Terms of Reference

The group considered six approved Terms of Reference during the meeting (Annex 3). A working group member was assigned as the lead prior to the meeting and each provided a brief outline of the workplan and identified the goals that would be achieved by the end of the meeting. In addition, one member Helmut Thetmeyer provided an update on the EU funded project on sustainable impact indicators – Ecosystem Approach to Sustainable Aquaculture (ECASA). The presentations of the group were:

- ToR a – Risk Assessment – Edward Black;
- ToR b – EU Legislation – Francis O'Beirn;
- ToR c – Sustainability indices – Barry Costa-Pierce;
- ECASA Overview – Helmut Thetmeyer;
- ToR d – Multitrophic Aquaculture systems – Steve Cross;
- ToR e – Update on Alternative feeds – Kats Haya;
- ToR f – Fouling hazards – Chris McKindsey.

5 Review the outcome of the GESAMP WG 31 on the aquaculture risk analysis methodologies and finalise case studies examining the potential impacts of escaped non-salmonid farmed fish (cod, sea bass, sea bream, halibut, turbot) (ToR a)

The WG continued to develop a series of documents concerning risk analysis of the consequences of genetic interaction between farmed fish and conspecific adjacent wild stocks, as part of a joint project with GESAMP WG31. The documents consist of an introductory paper describing the risk analysis approach, and case studies of five non-salmonid aquaculture species (cod, halibut, sea bass, sea bream and turbot). The introductory paper is dervide primarily from the document generated and presented in the WGEIM 2005 report and is not included here. The five case studies are presented in Annex 5 of this report

The WG agreed that the papers were nearing the standard and completeness necessary for submission for peer reviewed publication. The WG recommended that the documents be completed and submitted to an appropriate journal intersessionally.

6 Provide an update report on developments in implementation of the Water Framework Directive, the European Marine Strategy, the EU Strategy for sustainable aquaculture and assess their implications for Mariculture (ToR b)

This is a recurring term of reference that examines the implementation progress of existing and proposed European Union legislation and their implications for mariculture. To date, three policy initiatives have been described:

- The Water Framework Directive (WFD);
- The European Marine Strategy (EMS); and
- A Strategy for Sustainable Development of European Aquaculture (SSEA).

For each an update has been provided on the state of the implementation (WFD) or development of the regulations (EMS, SSEA). If the knowledge permitted conclusions on the potential impacts on mariculture were forwarded. The development of these policy initiatives has continued throughout 2005, between meetings of the WGEIM. In addition, it was highlighted that the EU Data Collection Regulation (EC/1543/2000) is to be revised with additional requirements to report on aquaculture. This development will be included in this review.

6.1 The Water Framework Directive

The Water Framework Directive (2000/60/EC) is a comprehensive piece of legislation whose primary goal is to ensure all waterbodies in member states achieve good ecological status by 2015 (good chemical status and quantity in the case of ground waters). Surface waters refer specifically to all streams, rivers, lakes, transitional (estuarine) and coastal waters. As previously, highlighted (WGEIM 2005) an additional goal of the directive is to bring about the effective co-ordination of water environment policy and regulation across Europe in order to:

- to protect and enhance the status of aquatic ecosystems (and terrestrial ecosystems and wetlands directly dependent on aquatic ecosystems);
- to promote sustainable water use based on long-term protection of available water resources;
- to provide for sufficient supply of good quality surface water and groundwater as needed for sustainable, balanced and equitable water use;
- to provide for enhanced protection and improvement of the aquatic environment by reducing / phasing out of discharges, emissions and losses of priority substances.

These goals will be achieved through a combination of assessment, monitoring, classification and implementation of measures to ensure appropriate improvement in ecological and chemical quality of water bodies by the year 2015.

While there were no specific deadlines for 2005 highlighted in the legislation, a number of tasks were ongoing in order to meet deadlines for 2006. For Example in Ireland, these 2006 deadlines (as dictated by national legislation transposed from the EU Directive) are given in Table 6.1.

Table 6.1: WFD implementation deadlines for 2006 in Ireland (Source: WFDIreland.ie).

22 June 2006	Develop Classification systems for surface water and groundwater
22 June 2006	Establishing and maintaining appropriate Monitoring Programmes - Such monitoring must cover both surface and groundwater and must be operational by 22nd December 2006.
22 June 2006	Prepare and publish a work Programme and Timetable for the production of River Basin Management Plans (RBMP).

Intercalibration of Classification Tools

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 communities, etc). Countries are presently defining values for their assessment tools that 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. 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 the status class boundaries are consistent with the normative definitions of water quality status given in the Directive, and therefore consistent among countries.

For the purposes of intercalibration exercise, a Europe-wide list of water body types has been generated and comparisons are being made on the basis of specific geographic regions of which there are four:

- Mediterranean Sea (microtidal – euhaline);
- Baltic Sea (microtidal, oligo – polyhaline);
- NE Atlantic complex (NE Atlantic, North Sea, Barents Sea, Norwegian Sea);
- Black Sea (microtidal, oligo – polyhaline).

The progress of the North East Atlantic Geographic Intercalibration Group (NEA GIG) will be used as an example of this implementation task.

The intercalibration process is to be completed by June 2006. Comparisons are being made using data provided by member states for each the ecological quality elements (plants, benthic macro-invertebrates and fishes). The data selected are representative of a number of pressure gradients, e.g. nutrient enrichment, organic enrichment, pollution by hazardous substances, habitat degradation. Consequently, supporting information is expected to accompany the data provided to the process. Table 6.2 provides a list of the specific tools that are being compared among member states in NEA GIG highlighting their status in development and progress towards defining agreed good/moderate boundaries.

Table 6. 2.

ECOLOGICAL QUALITY ELEMENT	TOOLS USED IN INTERCALIBRATION EXERCISE	WHETHER THE QUALITY ELEMENT IS ADDRESSED FULLY?	STATUS OF METHODS IN MEMBER STATES; IN TERMS OF ACCEPTANCE UNDER NATIONAL LEGISLATION?	WHETHER GOOD/MODERATE CLASS BOUNDARIES CAN BE SET FOR THESE METHODS IN INTERCALIBRATION EXERCISE?
Phytoplankton	Chlorophyll-a; Cell counts (Total and indicator species)	Partly	All - Under development Portugal – officially accepted	Y (boundary levels agreed at GIG level)
Macroalgae and angiosperms	Macroalgae depth limits and species richness; seagrass bed extent; saltmarsh extent.	Partly	All - Under development	Y (boundary levels agreed at GIG level)
Benthic invertebrate fauna	Soft sediment habitat; numerous indices compared using quantitative sampling methods. Sensitive taxa metric (hard substrate-Spain only)	Partly	All - Under development	Y (boundary levels have been set separately within member states)
Fish	Numerous tools.	Partly	All - Under development	Y (boundary levels agreed at GIG level)

While the process is being completed and progress is obviously being made, there appears to be some inconsistencies with the process that may make the complete harmonisation difficult to achieve. Of particular concern is the extent of the quality element being addressed. For example, in the UK and Ireland there are nine tools proposed for monitoring coastal and transitional waters in the phytoplankton quality element. However, for the intercalibration process only two specific ones are being compared (hence, the "partly" designation in column 3 of Table 6.2). Consequently, while the intercalibration process may provide clear consistency for the tools selected, it is not clear whether the combination of tools will constitute a harmonisation of methods. Similarly, for the benthic classification tools the metrics being compared have been developed from soft sediment data (habitat specific). However, it is not clear how the metric will behave in coarser (or more exposed) habitats. The groups acknowledge these potential shortfalls. Specific means to address them have not been forwarded; although, a second intercalibration process has been proposed.

Chemicals

One of the primary aims of the WFD is to eliminate discharges of priority hazardous substances. The Directive contains a list of 33 such substances, which inclue various pesticides, brominated flame retardants, chlorinated compounds, PAHs, metals etc. In general, none of these substances are now significant inputs to mariculture, either fish or shellfish cultivation. The list includes tributyltin, but its use in aquaculture was prohibited more than 10 years ago.

In addition to this list, which is common to all Member States, the Directive also includes at Annex VIII a wider and more general list of hazardous substances. Member States are required to manage discharges of these substances, and the quality of the receiving waters, through a system of EQSs. Member States are required to identify from that list particular substances that are of concern in their own waters, and to develop appropriate monitoring and control systems.

In the case of the UK, this list under Annex VIII currently contains 16 substances including copper, zinc and ammonia. At least two of these substances (copper and ammonia) can potentially be released from aquaculture facilities in locally significant quantities in comparison to other inputs; copper primarily from antifouling and ammonia from fish metabolism. The proposed second tranche of seven substances for the UK includes cypermethrin, which is currently approved for use as a sea lice control medicine, although the primary reason for its selection is probably use in sheep dip.

Implications for Mariculture

It is not yet clear whether the inclusion of these substances will result on any additional constraints on the aquaculture industry. Their use and discharge are currently controlled through systems of approvals for use and permits for discharge.

6.2 European Marine Strategy (EMS)

WGEIM 2005 provided an overview of the EMS and highlighted that the publication of a strategy document was imminent. On 24 October 2005 the commission published both:

- The thematic Strategy on the Protection and Conservation of the Marine Environment (i.e. a background document providing context for the EMS); and
- The Proposed Directive establishing a Framework for Community Action in the field of Marine Environmental Policy (i.e. the Marine Strategy Directive).

All the relevant documents can be found on the EU website,

http://europa.eu.int/comm/environment/water/marine.html

In summary, the EMS will be consistent with the goals of the WFD and in effect provide a continuation in terms of environmental stewardship by the member states. Many of the definition and goals are similar between the two pieces of legislation. However, the proposed directive is not as prescriptive as the WFD. EU Member States share responsibility for a number of different marine areas, each of which has its own distinctive environmental characteristics. Three marine regions are proposed:

1) the Baltic Sea;
2) the North-East Atlantic;
3) the Mediterranean Sea.

The Northeast Atlantic and the Mediterranean Sea regions may be further sub-divided in to four sub-regions each, in order to take into account the specificities of the particular sub-region.

To take account of regional differences the Commission proposal sets out common objectives and standardized approaches to assessment – but these are to be implemented at the level of marine regions (or sub-regions). This means that the Member States sharing a marine area will be responsible for working in close cooperation to develop plans designed to ensure good environmental status in their respective marine waters by 2021. These plans are to include a detailed assessment of the state of the environment as well as defining what achieving good environmental status means in the context of each regional sea.

It is proposed that Initial assessments of the marine environment will be completed by 2010. This assessment will be a partial assessment focusing on within regional sea objectives and how the preliminary stage of the implementation of the directive are being addressed. A pan-European assessment is proposed for 2015 allowing for comparison among regional seas. They will also contain clear environmental targets and monitoring programmes. Annex II (Table 1) of the proposed directive provides a list of characteristics that would form the basis of the initial and pan-European assessment. These characteristics will also form the basis of monitoring programs for which good environmental status must be defined. The characteristic broadly cover:

- Physical and Chemical features;
- Habitat features;
- Biological features;
- Other features (including, *inter alia*, state of nutrient and chemical contamination).

No specific management measures will be set down at EU level, but plans must be checked and approved by the Commission. Management measures must be responsive to the pressures imposed upon the region. A list of pressures that might be considered are provided in Annex II Table 2 of the directive.

For EU Member States that share marine areas with non-EU countries and an important part of achieving good environmental status will involve close co-operation with these third countries. In addition, member States will be encouraged to work within the framework of existing regional seas conventions which have extensive expertise in protecting the marine environment (e.g. OSPAR and HELCOM) and to ensure the programs are consistent with the objectives and measures imposed by the WFD.

Each Member State will be required to draw up a programme of cost-effective measures aimed at delivering good environmental status of the marine environment. Impact assessments (regulatory and environmental), including detailed cost-benefit analyses of the measures proposed, will be required prior to the introduction of any new measure.

Implications for Mariculture

Within the list of pressures identified within the directive (Annex II), mariculture is referred to in the context of non-toxic contamination. Specifically, it is considered a potential source of organic enrichment. While this aspect and the consequences have been well described previously, the scale of mariculture operations (existing or proposed) are such that their significance, in terms of impact, at the region or sub-region level proposed in the EMS, will likely be very small.

However, some proposed integrated activities such as mussel culture within the confines of offshore wind parks (Buck *et al.*, 2004) may require some closer attention in terms of management and monitoring under this new directive. Given the large footprints associated with wind parks the direct impacts may prove significant. Cumulative impacts as a consequence of activities carried out jointly in these areas might be elucidated?

Also of potential relevance might be the impact of dredge fisheries that supply seed for inshore culture plots. This activity is carried out, outside the boundaries for the WFD, in the Irish Sea (NE Atlantic – Celtic Seas). Dredging is a potential source of physical pressure identified in the directive which could lead to physical destruction of habitats and increased siltation (redistribution of sediments) with adverse consequence on adjacent habitats. Again assessment of the pressures might include this activity, specific to mariculture operations. The follow-on impacts on the receiving waters (inside the boundaries on the WFD) should also be assessed, particularly into areas where there are no existing mussel beds.

6.3 A Strategy for Sustainable Development of European Aquaculture

As a result of its rapid growth in recent years, the European Aquaculture industry is facing a number of challenges in terms of market and of the environment. Its future will depend on its ability to become economically self-sufficient and its capacity to respond to environmental constraints. In September 2002, the European Commission presented to the Council and to the European Parliament a communication on a strategy for the sustainable development of European aquaculture. The main aim of the strategy is the maintenance of competitiveness, productivity and sustainability of the European aquaculture sector. The strategy aims to create the conditions that will enable the aquaculture producers to offer a healthy product in the quantities required by the market, while being environmentally non-degrading.

The strategy identifies three objectives:

- Creating secure employment;
- Providing safe and good quality fisheries products and promoting animal health and welfare standards;
- Ensuring an environmentally sound industry.

To meet these objectives, the Commission proposes the following three primary measures:

1) Creating secure employment focusing upon a number of different initiatives designed to increased production, tackle competition for space, stimulate the market, consider social considerations and improving governance;

2) Safety of aquaculture products and animal welfare to ensure high standards for public health, animal health and animal welfare;

3) An environmentally sound aquaculture, with specific focus upon reducing the impact of waste, tackling the risk from alien species and genetically modified organisms, pollution prevention and control and environmental impact assessment: and promoting research.

One of the measures, cited above, is designed to ensure greater biodiversity protection and has taken the form of proposals to regulate the introduction of non-native species in aquaculture. While it is acknowledged that introductions of aquaculture species have proved very beneficial economically, there are risks inherent in these practices. The Commission has published in April 2006 the proposal for regulations concerning the use of locally absent species in aquaculture. In the preamble, aquaculture is defined to include aquaculture is taken to include activities such as bottom cultivation of mussels and both stocking and put-and-take fisheries, which use aquaculture techniques as their basis. As justification for introducing this legislation, a loophole was identified in the EU Habitats Directive (92/43/EC). The Habitats Directive refers to deliberate introductions of species not native to the territory. However, both the accidental, non-deliberate introductions and the introductions into non-wild environments are not covered.

The proposed regulations define a procedure for the establishment at national level of a system of permits for all new species introduced for aquaculture. Compliance with the requirements of the regulation will be carried out by a ‘competent authority’ designated by the member state. The competent authority will appoint and national advisory committee. Under the proposed measures, all projects to introduce a non-native species into an area, for culture purposes, would have to be submitted for approval to a national advisory committee. This committee would have to determine whether the proposed introduction was ‘routine’, or not. In the case of non-routine introductions, an environmental risk assessment (ERA) would have to be carried out. Only movements which are assessed as being low risk could then be granted a permit. If the risk was considered to be medium or high, the advisory committee would enter into dialogue with the applicant to see whether adequate mitigation procedures or technologies which could reduce the risk to an adequately low level were available.

In the case of non-routine movements, the proposal provides for quarantine procedures, and in certain cases, the national authorities may also require a pilot release to be implemented prior to full-scale commercial introduction. The proposed regulation also sets out a number of requirements concerning contingency plans, monitoring procedures, and the keeping of national registers.

The scope of the current proposal is limited to movements of fish stocks which fall under the Common Fisheries Policy (aquaculture species of fish and shellfish are covered by the CFP). Ornamental fish are therefore not concerned by these measures. The spreading of parasites and pathogens is already covered by Community legislation on animal health, so this issue is not addressed here either. The Commission is aware of the problems potentially posed by genetically modified organisms, but believes that these are best addressed by the substantial and evolving Community legislation specific to this field. The regulations build on the voluntary codes of practice formulated by the International Council for the Exploration of the Sea (ICES) and the European Inland Fisheries Advisory Commission (EIFAC).

These regulations allow the practice of introducing shellfish and finfish into EU member states to continue, but provides an assessment of the risks inherent in this practice. In summary the regulations outlines the;

- application and evaluation procedures;
- information requirements of an application; as well as
- the procedures for risk evaluation; including both an evaluation of the risk the target species poses as well as the risk of introducing non-target species.

It is appreciated that much of the information sought during the application process will not be available. For instance, information on the impacts of introduction of non-native species into the wild is not readily available and consequently, determination of risk could prove very difficult. In an effort to address this issue, the EU has called for a proposal to carry out a review of the environmental impacts of alien species for aquaculture. The evaluation of this call is currently ongoing (as of April 2006) and the outcome will be reported at WGEIM 2007. It is expected that the project will provide guidelines for environmentally sound practices for introductions and translocations in aquaculture and specific recommendations and guidelines on quarantine procedures.

Implications for Mariculture

The implications for mariculture of these proposed regulations are obvious, as they are specifically developed to address movements of species used in aquaculture.

While the regulations attempt to address perceived loopholes in the habitats directive (non-deliberate introductions) another appears to present itself. There are practices whereby aquaculture product can be moved large distances (across international boundaries) from one growing area to another for relaying (in the case of mussels). These animals are still within the range of the target species, and the target species are established in the receiving environment (Figure 6.1.A – below). However, the movement may cross boundaries that might have previously prevented the expansion of non-target species. Consequently, a non-target species may be introduced in a non-deliberate fashion (Figure 6.1.B). There is no mechanism in the regulations to carry out a risk evaluation of this practice.

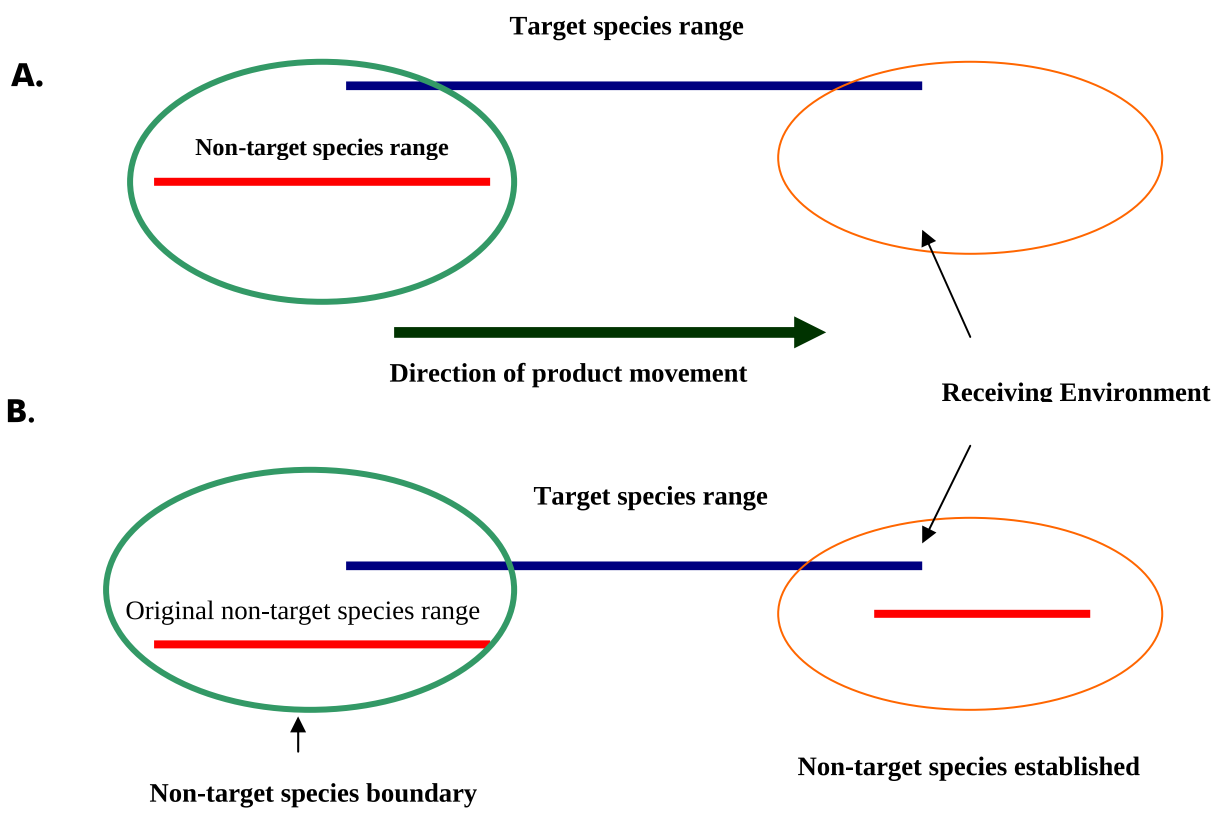


Figure 6.1 – A and B.

The definition that comes closest describing this practice in the proposed regulations is 'Translocation' which is defined as – 'the process by which an aquatic organism is intentionally moved within its natural range for its use in aquaculture to an area where it previously did not exist because of biogeographical reasons'. The term 'previously did not exist' is problematic as it sets the definitional boundaries within which all other conditions are constrained. Two conditions requiring risk evaluation are; translocations between ecoregions defined by the WFD and where scientific advice determines there are potential environmental threats due to translocation. Given that the target species occurs in the receiving environment, these conditions for evaluation are not applicable. However, the risk of introducing a non-target species into a novel area does present itself and the proposed regulations do not appear to provide a mechanism to examine this risk.

A possible solution to this is to exclude the terms 'previously did not exist' in the definition of translocation such that the conditions outlined previously (ecoregions and expert judgement) can cover this situations and potential non-target introductions.

6.4 EU Data Collection Regulation

EU Council Regulation 1543/2000 established a community framework for the collection and management of the data needed to conduct the common fisheries policy. Each member state is obliged to collect data on the biology of the fish stocks, on the fleets and their activities and on economic and social issues. The programmes are evaluated by a panel of experts and by the Scientific Technical and Economic Committee on Fisheries (STECF). The programmes are drawn up for six-year periods, the first covers the years 2002 to 2006. A revised proposal is being considered to cover the period 2007–2013 with the goal of meeting demands generated by fisheries-based management approaches and towards an ecosystem approach. It is expected that the revised framework will cover the collection of information from the sea, the market and the end user.

Implications for Mariculture

Aquaculture is not included in the current data collection framework. It is proposed, in the revised regulations, that information on the performance of this sector will be collected. The exact nature of this information will be determined at a STECF workshop during 2006.

6.5 Reference

Buck, B.H., G. Krause, H. Rosenthal. 2004. Extensive open ocean aquaculture development within wind farms in Germany: the prospect of offshore co-management and legal constraints. Ocean and Coastal Management. 47: 95–122

6.6 Recommendation

The WG recommends to the MCC, that the Term of Reference b (Review of EU legislation) be put in abeyance. It is further recommended that members of the group adopt a watching brief on EU and other legislation until 2008. In the interim, if a piece of legislation is identified that has relevance to Mariculture it can be reported and discussed during subsequent meetings as a Term of Reference or in AOB until it is more fully dealt with in 2008.

7 Evaluate examples of sustainability indices proposed for mariculture operations and provide specific recommendations on utility of proposed (ToR C)

7.1 Abstract

Sustainability indicators are different from “impact” indicators in that they are more comprehensive, including considerations of not only environmental but also social and economic sustainability. Sustainability indices (SIs) are needed by aquaculture resource managers who must sort through large amounts of scientific information and make numerous environmental decisions. SIs offer a means to prioritize those aquaculture systems most in need of immediate management attention and allow scarce management assets to be applied in the most cost-effective manner. SIs are also valuable for owners of seafood businesses who wish to procure “sustainable seafoods” for this rapidly growing consumer movement. We considered and evaluated the current status and suitability of SIs for mariculture and have selected a suite of SIs that are simple, flexible and cost effective. We use a modified traffic light approach that maps “sustainability trajectories” and apply it to salmon farming in New Brunswick, Canada..

7.2 The Many Definitions of Sustainability

There are many definitions of “sustainability”, both general definitions and those that define environmental sustainability (Table 7.1). The most popular definition of sustainable development is to "meet present needs without compromising the ability of future generations to meet their needs" adopted at a UN conference in 1987 (WECD, 1987). Robert Gillman, editor of *In Context* magazine, extends this goal-oriented definition by stating, “sustainability refers to a very old and simple concept – The Golden Rule **–** do onto future generations as you would have them do onto you**.”**

The many definitions of sustainability embody the concepts of "stewardship", "design with nature," the concepts of “polluter-pays”, the “precautionary principle”, and as well as "carrying capacity," the latter a highly developed modeling technique used by scientists and planners. As well, sustainability includes considerations of:

- more comprehensive planning for multiple impacts, with greater involvement of multiple disciplines in decision-making, and considering not only economic impacts but environmental and social as well;
- better planning for long term consequences of present development options; and
- incorporation of externalities in planning for site-specific developments.

Frankic and Hershner (2003) stated that sustainability refers to the ability of a society to continue functioning in the future without being forced into decline through exhaustion or overloading of key resources on which society’s systems rely.

7.2.1 The Evolutionary Transition to Sustainability

“We suspect that living in true harmony with the natural world, in a manner sustainable over the long run, is something no modern human society has yet learned how to do. The survival of the natural world, however, and likely our survival as a species, depends on our learning to do this. It will be a unique experience in human history." (Bock and Bock, 2000)

The sustainability transition started with the publication of “Our Common Future” (WECD, 1987) is one of the great movements of our time. Sustainability is a concept much broader than planning for site-specific impacts; it also accounts for systemic impacts off site. As a result, sustainability indices are much different than environmental “impact” indices. WECD (1987) stated that sustainability is using and not harming renewable resources and unique human-environmental systems of a site – air, water, land, energy, and human ecology – and/or those of other [off-site] sustainable systems. The sustainability transition is well underway in heavy industry (Krajnc and Glavic, 2005).

The world needs the rapid development of aquaculture, but this accelerated development must be done in a sustainable manner. Meeting basic human needs for protein foods in the future will be a difficult challenge. Approximately 1.3 billion people live on less than a dollar a day—the cost of a half a pint of beer—and half of the world's population lives on less than 2 dollars a day (Watson, 1999). Since 1950, there has been a 100% increase in the per capita demand for fish, a 40% increase for grain, and 33% for wood. FAO (2000) predicts world fish consumption to increase from 16 kg (1997) to 19–20 kg by 2030, raising total human use of aquatic foods to 150–160 million tons. Capture fisheries can provide no more than 100 million tons, so bulk of the increase will need to come from aquaculture.

We believe that the accelerated development of aquaculture can occur as an prime example of sustainability, and that in the 21st century, well integrated, ecologically integrated aquaculture systems that have *positive* impacts on both natural and social ecosystems will evolve widely. Costa-Pierce (2002) states that “as an infant enterprise the world over, aquaculture can ill afford to recreate the sorry history of commercial agriculture where huge toxic, nutrient and chemical loads were (and still are) washed down a primitive path of the "solution to pollution is dilution". Aquaculture should be pro-active, promote and develop itself as the world's most ecologically integrated industry, and adopting a new strategy—that of a community-based, sustainable, ecological aquaculture industry that produces ecologically and socially certified produce—adopting input management strategies (Odum, 1989) and “codes of practice”. The Food and Agriculture Organization of the United Nations (FAO) Code of Conduct for Responsible Fisheries (FAO, 1995) contains a key recommendation that: “States should produce and regularly update aquaculture development strategies and plans, as required, to ensure that aquaculture development is ecologically sustainable and to allow the rationale use of resources shared by aquaculture and other activities.” Additional key strategies have been the development of the Holmenkollen Guidelines for Sustainable Aquaculture (Moe and Svennevig, 1998); the Bangkok Declaration and Strategy (NACA/FAO, 2000); and the Global Aquaculture Alliance's Responsible Aquaculture Program that has produced criteria for environmentally certified cultured shrimp. Bertollo (1998) argues that codes of conduct and guidelines for certifying sustainability are too complex. We see the need to develop simple, scientifically credible sustainability indices for mariculture as a cost effective complement to these codes.

Aquatic resource managers are flooded with large amounts of information for about aquaculture and aquatic area management – environmental, economic and social including governance and policy matters. SIs may offer cost-effective methodologies for these managers to help simplify and prioritize their decisions regarding aquaculture development proposals that reach their desks.

In addition, buyers of farmed aquatic foods want unambiguous, unbiased and science-based information on sustainability of aquaculture operations in order to make the best choices for their customers who are increasingly concerned with many issues regarding the overall sustainability of seafood, the ocean environment and the employees of the world’s working waterfronts including issues of worker safety, health, and social justice in addition to environmental issues and farming practices.

Sustainability is an overused word with abundant pedagogy but little practice. There has been much lip service given to the concept but little progressive action has been taken, especially in situations where there are time-worn political "turf wars". Sustainability is hampered when there is a clear need for interdisciplinary actions but there is little scientific knowledge, or where there are clear winners and losers from environmental action.

7.2.2 The Development of Indicators

Well developed and scientifically credible SIs are important tools which can make monitoring, data collection, research enterprises, and communications efforts better organized and targeted, and thereby more cost effective.

Performance indicators must be selected to cover the three components of sustainability. Kjanc and Glavic (2005) argue that indicators should be as quantitative as possible but that qualitative descriptions may be more appropriate foe describing the social aspects of sustainability.

The Integrated Coastal Area Management (ICAM) guidance of UNESCO (2003) defines an indicator as a "parameter or value, which provides succinct information about a phenomenon". The ICAM guidance has three basic categories of indicators:

- Environmental: reflect trends in the state of the environment; are descriptive in nature; and become performance indicators if they compare actual conditions to desired conditions expressed in terms of environmental targets;
- Socioeconomic: represent the demographics of humans in the coastal zone and measure quality of life issues;
- Governance: measure the performance of the state of implementation, measuring the progress and quality of interventions of the governance process in relation to program goals set at the outset.

In relation to environmental policy-making, environmental indicators are used for three major purposes:

- To supply information on environmental problems in order to enable policy-makers to value their seriousness;
- To support policy development and priority setting by identifying key factors that cause pressure on the environment;
- To monitor the effects of policy decisions (Smeets and Weterings, 1999).

Indicators must be measurable objects that can be simplified by aggregation and calculation. Outcomes from theoretical models cannot be considered as indicators. Nevertheless, models may help to indicate the most relevant factors to be monitored. Ideally, indicators must address the following issues:

- Continuity of supply (environmental, economic, and social services),
- Social, economic and environmental costs to provide this continuity of supply,
- Long-term aspects,
- Financial viability,
- Social and ecological impacts,
- Global efficiency.

Communication is the main function of indicators - they should enable or promote information exchange regarding the issue they address. Our body temperature is an example of an indicator we regularly use. It provides critical information on our physical condition. Likewise, environmental indicators provide information about phenomena that are regarded typical for and/or critical to environmental quality (Smeets and Weterings, 1999).

Environmental indicators may be used as a powerful tool to raise public awareness on environmental issues. Providing information on driving forces and impacts and connecting them to policy responses is an important strategy to strengthen public support for environmental policy measures. However, communication demands simplicity, and are important tools to focus attention on certain environmental aspects that are regarded relevant to society, and on which credible data are available. Indicators always simplify complex realities, and their significance goes beyond that obtained directly from the observed properties. Environmental indicators communicate those aspects regarded critical or typical for the complex interrelation between natural species and abiotic components of the environmental system.

There is a great deal of on-going activity on the development of indicators for coastal areas throughout the world (Table 7.2). European Union Member States are currently developing standards and indicators at both the national and regional scales as part of their collective work towards the Water Framework Directive. Ireland has completed a review of the application of marine environmental indicators to that nation's marine ecosystems (Boelens *et al.*, 2004). The Department of Fisheries and Oceans Canada (DFO) is engaged in developing ocean indicators (environmental, socio-economic and governance) that will be used to follow trends and make decisions regarding a number of diverse coastal settings in support of the integrated management of Canadian oceans (DFO, 2004). DFO intends to make a long-term commitment to indicator development as part of its iterative cycles of planning for both reporting and performance evaluation (DFO, 2004).

At the outset it is important to discern the differences between "sustainability" indicators and "impact" indicators. Sustainability indicators should be able to track more than aquaculture's impacts on the environment - deterioration and recovery – and be able to monitor economic, social, and cultural externalities, as well as evaluate governance impacts of policies and regulatory measures on aquaculture. Once accepted, SIs need to be ultimately included in codes of best practices, decision support systems, and should be used in steering of the directions of aquaculture development by the authorities. They may also be used to monitor institutional changes and impacts of the policies (*ex ante* and *ex post*).

7.3 Composite Sustainability Indices

While it is important to assess sustainability with several indicators, it may sometimes be difficult to make decisions and comparisons among sectors, production systems or companies based on a large number of performance measurements. To help decision makers in this respect, it may be useful to use composite sustainable development index, linking many sustainability issues and so reducing the number of decision-making criteria that need to be considered.

SIs must be flexible enough to be adapted to the local environment in which they will be used. There is no chance that a single set of "generic" indicators may be universally applicable and used in all the situations in the aquaculture sector. In addition, SIs may be of use to address the interactions with other users of marine resources, locally or internationally because of the opening of global markets.

In recent years, international research has focused on the development of composite indices mostly for cross-national comparisons of economic, societal, environmental and/or sustainable progress of nations in a quantitative fashion (Krajnc and Glavic, 2005a).

Krajnc and Glavic (2005b) proposed a mathematical model for the determination of a composite sustainability index that will enable comparisons of companies in specific sector regarding sustainability performance and applied it to the oil industry. The proposed model reduces the number of indicators by aggregating them into a composite sustainable development index (ICSD).

The procedure of calculating the ICSD is divided into several hierarchical parts: selecting, grouping, weighting, judging, normalizing indicators, calculating sub-indices and combining them into the ICSD. Weighting the indicators is a sensitive operation. A pair-wise comparison technique was used to calculate the ICSD in order to derive relative weights of each indicator practically (the Analytic Hierarchy Process (AHP), Saaty (1995)). The comparisons were made by posing the question to a panel of experts which of the two indicators was more important with respect to the sustainable development (in a range from 1 to 9). Normalizing the indicators was the main difficulty because it imposed to aggregate indicators which may be expressed in different units into the ICSD. In this study, economic (4 indicators), environmental (6 indicators), and societal (4 indicators) indicators have been used, but their number is not limited. They were aggregated into sustainability sub-indices for the case companies and finally aggregated into the ICSD.

The various ICSD indicators can be used to highlights the progress towards sustainability achieved with time by a sector or a company, or to compare different options. It can be used to inform decision makers and individuals of trends in development. The decision makers can easily interpret the ICSD and its corresponding sub-indices, rather than trying to find a trend in many separate indicators of sustainable development. The possible disadvantage of the model may be the way in which the weights of indicators are determined, which could lead to a high degree of subjectivity. However, it is simple to re-evaluate the weights and to operate a sensitivity analysis to counterbalance this issue. The second possible weakness relies on the a selection of indicators. To avoid this, it requires a transparent process to determine how and who will select the indicators.

7.4 The Development of SIs for Mariculture

Sustainability concepts rely upon considerations of the fundamental components of societies throughout the world – the environment, the economy, and the society (the “3 P concept” of people, profit, planet).

In France, a program on life cycle analysis is being developed based upon models developed originally for intensive farming (Papatryphon *et al.*, 2004). Indicators are based on the analysis of labor and energy required for each component of the production system, including the use of production factors, intermediate products, marketing and supply, as well as long-term investments for infrastructure and decommissioning. These indicators will allow the comparison of various production systems and the consideration of different technical solutions prior to decision-making.

A recent publication by Dewulf and Van Langenhove (2005) theorized the use of this approach using the second principle of thermodynamics. The paper presents a set of five environmental sustainability indicators for the assessment of products and production pathways, integrating industrial ecology principles. The indicators, all scaled between 0 and 1, take into account the renewability of resources, toxicity of emissions, inputs of used materials, recoverability of products at the end of their use, and process efficiency corresponding to the basic processes related to an overall product life. Applied to aquaculture systems this set should be able to assess the environmental (*in sensu lato*) sustainability of technology options in a quantitative way, improving the sustainability debate through quantitative information.

7.4.1 Efforts to Develop SIs for Mariculture

SIs for environmental and aquaculture management are still in the early stages of development. We suggest there is a need to pilot a simple, set of easily understood indicators that policy makers and seafood buyers can understand. We share the concern of Hammond *et al.* (1995) who question whether or not sustainability is a “bounded concept with measurable goals and objectives”. Bertollo (1998) expressed the concern that codes of conduct and guidelines for certifying sustainability in environmental management are much too complex.

Along with concerns about too much complexity, there are concerns about the costs associated with monitoring multiple indicators that could be irrelevant to managers and the public. Useable indicators must be more than just a description of state and should have diagnostic properties that lead to some insights into processes taking place, and towards greater understanding of when and why things “go “wrong”.

Pullin *et al.* (2001) suggested a simple set of easily quantifiable indicators for sustainability in aquaculture:

- Biological: domestication, trophic level, nutrient/energy conversion;
- Ecological: footprint, emissions, escapes;
- Intersectoral: water-sharing, diversity, cycling, stability, and capacity.

Caffey *et al.* (2001) used a Delphi survey technique to develop sustainability indicators for aquaculture in the southeastern USA. The Delphi approach was started by the Rand Corporation in 1948 to develop strategy and forecasts during the Cold War (Sackman, 1975; Schmidt, 1997), and has been applied to a range of fields from agriculture (Walter and Reisner, 1994) to fisheries (Zuboy, 1981). The Delphi technique yielded 31 indicators of aquaculture sustainability: 12 environmental, ten economic, and nine social indicators (Table 7.3). Respondents identified two paramount environmental indicators: resource use and pollution. Resource use indicators included: conservation of land, energy, protein, water, and wetlands. Pollution (environmental externality) indicators included: reduction of chemical use, effluent BOD control, controls of ammonia-nitrogen, phosphorus, suspended solids, and use of non-native species in aquaculture. Top economic indicators were profitability, risk, efficiency, and marketing issues. Social indicators of top importance were job availability, compensation rates, benefits, and worker safety.

7.4.2 Existing Efforts to Develop SIs

The WGEIM has considered several examples of existing attempts to evaluate/judge sustainability of mariculture that fall under three general categories:

Decision Support Systems

Traffic light system in Canada (Hargrave, 2002)

MOM-Lenka in Norway (Ervik *et al.*, 1997; Hansen *et al.*, 2001)

MARAQUA in Europe for siting and monitoring mariculture (Fernandes *et al.*, 2000).

Current EU efforts (ECASA) to evaluate environmental indicators

An EU project titled, “Ecosystem Approach for Sustainable Aquaculture” (ECASA) led by Dr Kenneth Black of the Scottish Association for Marine Science is considering the ability of indicators to discriminate between aquaculture and other anthropogenic sources of perturbation in the marine environment. Annual, national meetings with stakeholders are being held to allow two-way interaction ensuring the practical relevance of the work, and also ensure that the “user community” achieves ownership of the project’s outputs. Objectives of the EU project are to:

- Identify quantitative indicators of the effects of aquaculture on ecosystems through a process of expert working groups, workshops, and meetings;
- Identify indicators of the main drivers of ecosystem change affecting aquaculture, including natural and environmental pressures;
- Assess sets of indicators using existing datasets – project partners collectively have extensive data archives – considering each in the context of appropriate selection criteria;
- Develop a range of tools, particularly models, that encapsulate best process understanding at a wide range of scales;

- Test models and indicators in a wide variety of field locations across Europe (~10 locations) that encompass major cultured species and technologies, and covering a wide spectrum of environment types, selected according to criteria developed during the project;
- Use the collected data to test and select the final "tool pack" of models and indicators, including appropriate decision support tools to guide users to effective implementation.

To date, ECASA has evaluated 53 indicators of ecosystem change and will conduct fieldwork to select the best environmental indicators in three categories:

- Benthic fauna
- Sediment
- Water Quality

NGO-based consumer-oriented indices

(Monterey Bay Aquarium, Sustainable Seafood Forum, Seafood Choices Alliance).

There are a number of coastal environmental indicators that are in current use as to their relevance for assessing the environmental sustainability for mariculture, namely:

- the Infaunal Trophic Index (ITI) used by Scottish Environment Protection Agency,
- the AZTI Marine Biotic Index (Borja *et al.*, 2000; Borja *et al.*, 2003),
- the Benthic Quality Index (BQI) for classification of marine benthic quality according to the European Union Water Framework Directive (Rosenberg *et al.*, 2004), and,
- the measurement of free sulfides to predict the biodiversity of benthic macrofauna (Hargrave, in prep.).

Of these, the BQI has the most utility in environmental quality assessment since it allows comprehensive assessment across different benthic ecosystems (Rosenberg *et al.*, 2004).

However, all of these indicators are too complicated and expensive, require considerable taxonomic expertise, are not easily implemented, and are therefore not in current, routine use by managers or aquaculture operators, being limited to use by academics. The free sulfide method of Hargrave (Figure 7.1) is a significant step forward in this regard. However, further statistical sampling research is required for assessments since there is a high spatial and temporal variability of redox measurements (K. Haya, pers. communication).

7.5 Sustainability Trajectories for Aquaculture: A Matrix Approach

Sustainability cannot be defined in a stark "black/white" manner – labeling an aquaculture operation as "sustainable or not sustainable" adds little to the overall goal to make aquaculture compatible with the modern world. Sustainability is an iterative process of improvement of management practices and procedures. We argue that there are "sustainability trajectories" for a small set of important indicators for which scaled comparisons can be made between operating procedures that are "best" practices, and that these can be distinguished from "average" and "poor" practices. We believe that these "trajectories" using a matrix approach are a more effective way of plotting evolutionary changes in practices over time. Indicators translate sustainability issues into quantifiable measures (Azapagic, 2004). Sustainability matrix approaches allow the flexibility needed to assess multiple factors more simultaneously and comprehensively so that rapid determinations and comparisons can be made.

SIs must be able to detect the linkages between the 3P's – people, profit, planet. We have adapted a process similar to the Canadian work of (Hargrave, 2002) and the UK Department of the Environment and Rural Affairs (Anonymous, 2005) that incorporate a “traffic light system” to assess progress towards sustainability for the seas around the UK to measure progress over time (Table 7.4). An example using this approach is accomplished for the Brunswick salmon farming industry in Table 7.5.

7.6 Recommendations

SIs are more comprehensive than environmental impacts assessments. They incorporate and try to integrate the “triple bottom line” concept (social, economic, and environmental) assessments. We recognize hat the strength of ICES is in the environmental and ecological fields, but to develop SIs further, ICES needs to broaden the scope of investigations to incorporate social and economic assessments. The WGEIM is only fully qualified to comment on the ecological/environmental aspects, as such we recognize that, for maximum utility, any environmental indices that ICES recommends should be placed in the context of the three factors/measures/dimensions. We recommend the efforts of Krajnc and Glavic (2005) to develop composite sustainability indices as a potential method to combine environmental, economic and social sub-indices (Figure 7.2).

The ECASA program is developing environmental SIs of the highest scientific credibility that will be peer reviewed for analyses of precision, accuracy, reliability, and consistency. We recommend that ICES evaluate their findings and that once there is agreement that these environmental SIs be featured widely in monitoring and management protocols.

Continue to refine the sustainability trajectories approach using the traffic light system of the UK and Canada., and apply this to example aquaculture farming systems (Table 7.5). We suggest that complete sustainability assessments are best when done in a collaborative, multidisciplinary manner.

We would like to emphasize that SIs must be sustainable themselves. The production of information must be practicable at a low cost for the government and industry, and understandable to the public. Data from SIs must provide credible, meaningful long-term data series. These time series will need to be housed in data management frameworks at the institutional level, but be universally accessible.

7.7 References

Anonymous. 2005. *Charting Progress: An Integrated Assessment of the State of UK Seas*. UK Department of the Environment and Rural Affairs, London, UK.

Azapagic, A. 2003. Systems approach to corporate sustainability – a general management framework. Trans. IChemE 81, Part B, 303–316.

Bertollo, P. 1998. Assessing ecosystem health in governed landscapes: a framework for developing core indicators. Ecosystem Health, 4: 32–51.

Bock, C., and Bock, J. 2000. The View From Bald Hill: Thirty Years in an Arizona Grassland. University of California Press, Berkeley, California.

Boelens, R., Gray, J., and Parsons, A. 2004. Review and evaluation of marine environmental indicators in Ireland. NDP Marine RTDI Desk Study Series DK/01/006. Galway, Ireland.

Borja *et al.*, 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Marine Poll. Bull., 40: 1100–1114.

Borja *et al.*, 2003. The application of a biotic index to different impact sources affecting soft-bottom benthic communities along European coasts. Mar. Poll. Bull., 46: 835–845.

Caffey, R.H., Kazmierczak, R.F., and Avault, J.W., Jr. 2001. Developing consensus indicators for sustainability for southeastern United States aquaculture. LSU Ag Center Bulletin Number 879, Baton Rouge, LA, USA.

Costa-Pierce, B.A. 2002. Ecological Aquaculture. Blackwell Science, Oxford, UK.

Dewulf, J., Van Langenhove, H., 2005. Integrating industrial ecology principles into a set of environmental sustainability indicators for technology assessment. Resources, Conservation and Recycling, 43: 419–432.

DFO. 2004. Federal – provincial – territorial coastal and ocean indicators in support of the integrated management of oceans. Fisheries and Oceans Canada, Ottawa.

Ervik, A., Hansen, P.K., Stigebrandt, A.J., Johannessen, P., and Jahnsen, T. 1997. Regulating the local environmental impact of intensive, marine fish farming. I. The concept of the MOM system (Monitoring-Ongoing fish farms-Monitoring). Aquaculture, 158: 85–94.

Fernandes, T.F., Miller, K.L., and Read, P.A. 2000. Monitoring and regulation of marine aquaculture in Europe. Journal of Applied Ichthyology, 16: 138–143.

Frankic A. and Hershner, C. 2003. Sustainable aquaculture: developing the promise of aquaculture. Aquaculture International, 11: 517–530.

Goodland, R., and Daly, H. 1996. Environmental sustainability: universal and non-negotiable. Ecological Applications, 6: 1002–1017.

Hansen, P.K., Ervik, A., Schaanning, M., Johannessen, P., Aure, J., Jahnsen, T., and Stigebrandt, A. 2001. Regulating the local environmental impact of intensive, marine fish farming. I. The monitoring programme of the MOM system (Monitoring-Ongoing fish farms-Monitoring). Aquaculture, 194: 75–92.

Hammond, A., Adriaanse, A., Rodenburg, E., Bryant, D., and R. Woodward. 1995. Environmental indicators: A systematic approach to measuring and reporting on environmental policy performance in the context of sustainable development. World Resources Institute, Washington, DC, USA.

Hargrave, B.T. 2002. A traffic light decision system for marine finfish aquaculture siting. Ocean & Coastal Management, 45: 215–235.

Hargrave, B.T. *in prep*. Effects of free sulfides in marine sediments on benthic macrofauna biodiversity. Department of Fisheries and Oceans Canada, Bedford Institution of Ocanography, Dartmouth, N.S., Canada.

Krajnc, D., and Glavic, P. 2005a. A model for integrated assessment of sustainable development. Resour. Conserv. Recycl., 43: 189– 208.

Krajnc, D., and Glavic, P. 2005b. How to compare companies on relevant dimensions of sustainability. Ecological Economics, 55: 551–563.

Papatryphon E., Petit, J., Kaushik, S.J., and Van der Werf H.M.G., 2004. Environmental impact assessment of salmonid feed using Life Cycle Assessment. Ambio, 33: 316–323.

Pullin, R., Froese, R., and Pauly, D. 2001. Indicators for the sustainability of aquaculture. *In* Ecological and Genetic Implications of Aquaculture Activities. Ed by T. Bert. Kluwer Press.

Rosenberg, R., Blomqvist, M., Nilsson, H.C., Cederwall, H., and Dimming, A. 2004. Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. Marine Pollution Bulletin, 49: 728–739.

Saaty, T.L. 1995. Decision making for leaders: the analytic hierarchy process in a complex world. RWS Publications, Pittsburgh.

Sackman, H. 1975. Delphi critique: expert opinion, forecasting, and group process. DC Heath and Company, Lexington, Mass., USA.

Schmidt, R.C. 1997. Managing Delphi surveys using nonparametric statistical techniques. Decision Sciences, 28.

Smeets, E. and Weterings, R. 1999. Environmental indicators: Typology and overview. Technical Report No 25. European Environmental Agency, Copenhagen, Denmark.

UNESCO. 2003. A reference guide on the use of indicators for integrated coastal management. ICAM Dossier 1. IOC Manuals and Guides Number 45. UNESCO, Paris, France.

Walter, G. and Reisner, A. 1994. Midwestern land grant university scientists definition of sustainable agriculture: A Delphi study. American Journal of Alternative Agriculture, 9.

Zuboy, J.R. 1981. A new tool for fisheries managers: The Delphi technique. North American Journal of Fisheries Management, 1: 55–59.

Table 7.1: Definitions of sustainability on the Web.

General Definitions	Web References
Meeting the needs of the present without compromising the ability of future generations to meet their own needs	www.afsc.org/trade-matters/learn-about/glossary.htm
A state or process that can be maintained indefinitely. The principles of sustainability integrate three closely interlined elements of the environment, the economy, and the social systems into a system that can be maintained in a healthy state indefinitely.	www.edo.or.blm.gov/infms/HTML/GLOSSARY/S.HTM
The ability to provide for the needs of the world's current population without damaging the ability of future generations to provide for themselves. When a process is sustainable, it can be carried out over and over without negative environmental effects or impossibly high costs to anyone involved.	www.sustainabletable.org/intro/dictionary/
A concept and strategy by which communities seek economic development approaches that benefit the local environment and quality of life. Sustainable development provides a framework under which communities can use resources efficiently, create efficient infrastructures, protect and enhance the quality of life, and create new businesses to strengthen their economies. A sustainable community is achieved by a long-term and integrated approach to developing and achieving a healthy community by addressing economic, environmental, and social issues. Fostering a strong sense of community and building partnerships and consensus among key stakeholders are also important elements.	www.ci.austin.tx.us/zoning/glossary.htm
The ability of a community or society to develop a strategy of economic growth and development that continues to function indefinitely within the limits set by ecology and is beneficial to all stakeholders and the environment.	www.thecorporatelibrary.com/Help/glossary/glossary.asp
The term originally applied to natural resource situations, where the long term was the focus. Today, it applies to many disciplines, including economic development, environment, food production, energy, and lifestyle. Basically, sustainability refers to doing something with the long term in mind, (several hundred years is sufficient). Today's decisions are made with a consideration of sustaining our activities into the long term future	ag.arizona.edu/futures/home/glossary.html
Sustainable development is the process of conducting business and commerce in a resource conservative and resource efficient manner such that operations do not compromise the ability of future generations to meet their own needs. The essential elements of this trend are the promotion and maintenance of business and community development strategies that lead to a better business environment in the future; one sustained by stable, healthful communities within a clean, safe environment. The operative concept underlying this growing trend is an emphasis on fostering community and business activity that is driven by long range goals, often met through pollution prevention.	www.mass.gov/epp/info/define.htm
The long-term health and vitality — cultural, economic, environmental, and social — of a community. Sustainable thinking considers the connections between various elements of a healthy society, and implies a longer time span (i.e, in decades, instead of years)	mapp.naccho.org/mapp_glossary.asp

GENERAL DEFINITIONS	WEB REFERENCES
Indicates that a plan, initiative or physical development project can be implemented and supported over time without depleting or adversely affecting the resources and management capabilities available to it.	www.uvm.edu/~plan/masterplan/glossary.html
Sustainability is an economic, social, and ecological concept. It is intended to be a means of configuring civilization and human activity so that society and its members are able to meet their needs and express their greatest potential in the present, while preserving biodiversity and natural ecosystems, and planning and acting for the ability to maintain these ideals indefinitely. Sustainability affects every level of organization, from the local neighborhood to the entire globe.	en.wikipedia.org/wiki/Sustainability
Economic development that takes full account of the environmental consequences of economic activity and is based on the use of resources that can be replaced or renewed and therefore are not depleted.	biology.usgs.gov/s+t/SNT/noframe/zy198.htm
The measure by which a human activity can be continued without relying upon limited resources, such as fossil fuels, or by leaving waste behind, and also giving nature the chance to replenish itself	www.ecohealth101.org/glossary.html
Definitions of Environmental Sustainability	Web References
The ability of an ecosystem to maintain ecological processes and functions, biological diversity, and productivity over time.	www.umpqua-watersheds.org/glossary/gloss_s.html
The use of ecosystems and their resources in a manner that satisfies current needs while allowing them to persist in the long term.	research.amnh.org/biodiversity/symposia/archives/seascapes/glossary.html
Meeting the resource needs of the present population without damaging the functionality of the area's ecosystem or its ability to meet the resource needs of future populations.	www.fairus.org/Research/ResearchList.cfm
Use of resources in a manner that allows the resources to be replenished by natural systems, as well avoidance of pollution that damages biological systems. Use of resources in such a manner that they will never be exhausted.	web-savvy.com/river/Schuylkill/glossary.html

Table 7.2: International/National Efforts Developing Coastal Environmental Indicators

STATE/ORGANIZATION (DATE)	INDICATOR REPORTS
Australia (2001)	State of the Environment report includes a chapter on coasts and oceans
Canada (2003a,b; 2004)	Environment and Sustainable Development Indicators of the National Roundtable on Environment and Economy (2003a); National Environment Indicator Series (2003b); Federal-Provincial-Territorial Coastal and Ocean Indicators in Support of the Integrated Management of Oceans
USA (2001, 2002)	State of the Nation's Ecosystems; US National Coastal Condition
Global Programme of Action	On going
Organization for Economic Cooperation and Development	On-going in its PSR Framework
UN Global Environmental Outlook	On-going
Millennium Ecosystem Assessment	On-going
ICAM/UNESCO	ICAM Guidance on Use of Coastal Indicators Worldwide (UNESCO, 2003)
EU	Water Framework Directive

Table 7.3: Indicators Identified for Aquaculture Sustainability Using a Delphi Technique (Caffey *et al.*, 2001).

ENVIRONMENTAL	ECONOMIC	SOCIAL
Quantity of Land Used	Gross revenue	Local consumption of product
Quantity of Energy Used	Total variable production cost	Use of local inputs
Animal Fraction of Supplemental Protein Used	Fixed costs of production	Value of job benefits
Quantity of Chemicals Used	Overall profit	Worker safety
Quantity of Water Discharged	Return on investment	Local ownership
BOD of Effluents	Variability in annual profits	Wage levels
Supplemental Feed Protein Used	FCRs	Jobs/employment
Total ammonia nitrogen in effluents	Cost of regulatory compliance	Competition with local industries
Culture of non-native species	Per capita consumption	Perception of local aquaculture industry
Total phosphorus in effluents	Market outlets	
Production of natural wetlands		
Suspended solids in effluents		

Table 7.4: The traffic light system for tracking progress towards sustainability for the seas around the UK (Anonymous, 2005).

	Key factors and pressures	What the evidence shows	Trend	Status (now)	Confidence in Assessment *	Reason for overall status
Water Quality	Riverine inputs and direct discharges of specified metals, lindane and PAH from point and diffuse sources	Reduction in inputs of metals and other contaminants since 1990 moving towards the OSPAR 2020 cessation target for OSPAR priority substances.	✓		III	On the basis of monitored substances water quality status is improving due to inputs falling. The open seas are generally not affected by pollution. The main contamination problems which are identified are in part due to the legacy of the past and are generally observed at higher levels in industrialised estuaries or areas local to the activity. However, some persistent chemicals are not routinely monitored and mixtures of chemical substances and diffuse inputs may pose a problem.
	Radionuclides	New anthropogenic emissions to marine environment highly controlled and meet internationally accepted exposure levels.	✓		III	
	Inputs from point and diffuse sources	Some persistent chemicals are not routinely monitored and mixtures of chemical substances and diffuse inputs may pose a problem.	?		0	
	Oil from accidental spills	No major spills in recent times.	✓		II	
	Oil from refineries and offshore oil and gas	Controls on deliberate inputs show that oil pollution only affects localised areas.	↔		II	
	Sewage discharges and microbiological	Improvements in sewage treatment infrastructure have given greater compliance with EU standards for bathing waters and shellfish waters, but some shellfish quality still fail the standards due to diffuse pollution.	✓		III	
	Discharges and emissions of nutrients from human activities	Direct inputs of nutrients from point sources discharging directly to the sea and atmospheric emissions of nitrogen have reduced by 35% since 1990. (NB direct inputs only account for roughly 25% of all nutrients inputs). Overall inputs of diffuse sources to the sea are unquantified.	✓		II	
Coastal habitats	Coastal development, erosion, sea level rise and climate change	A number of areas around our coast are vulnerable to erosion. This may be increased by rising sea levels and development on the coast. A number of key coastal habitats are under threat.	✗		III	Increasing development and sea level rise around our coastline leads to a narrowing of the coastal zone where natural processes may occur.
	Beach litter and human debris	Litter on beaches is totally preventable and yet quantities of debris are not falling.	↔		II	
Benthic communi-ties and associated sea floor habitat	Human activities causing physical disturbance	Benthic communities are adversely affected by human activities which have a physical impact on the sea floor such as fishing and dredging. Bottom trawling activity is the greatest impact since it results in direct mortality, can be over large areas of the sea bed and repeated frequently.	?		I	We have a very diverse range of benthic habitats and species but there are many threats which cause localised damage.
	Chemical contamination	Overall there is no evidence of broad scale impacts of nutrients or hazardous substances on benthic communities. However, some species do show signs of contamination in local areas, often close to the source of the pollution. Endocrine disruption (hormone change) has been detected in dogwelks.	?		II	
Fish	Commercial fishing	Many species of commercial fish adversely affected by exploitation with many stocks outside safe biological limits in particular regions.	↔		III	Our seas are some of the most productive in the world but many fish stocks are threatened by over exploitation.
	Industrial activities and contamination	Although the levels of disease in fish are higher than naturally expected in some UK waters it is unclear if human activities such as pollution are causing this.	↔		II	

* The confidence is in the quality and amount of data used to underpin the statements made.

Table 7.5: Example of the traffic light sustainability matrix for salmon farming in New Brunswick.

I. Environmental Indicators	Key factors and pressures	What the evidence shows	Trends over the last 5 years	Status	Confidence assessment	Reasons
Resource Use						
Energy	Use of electricity, petroleum and petroleum-based products	Primary use is in fish and feed transportation	?	NP	I	No specific studies for New Brunswick salmon farming; total energy use in salmon farming is high
Protein	Use of fish meals and oils in production	Decreased FCRs has increased production efficiency	+	GREEN	III	Improved research on feed formulations; improved production efficiency; improved feeding methods; substitution of fish meals/oils by industry
Habitats	Removal or degradation of benthic habitats	Improved video and benthic monitoring programs by both government and industry	+	GREEN	III	No/limitied areas of Beggiatoa mats; no evidence of irreversible harm; wide indication of improved oxygenation in sediments
Pollution						
Chemicals	Use of medicines and therapeutants	Strict monitoring and reporting requirements by government and industry	+	GREEN	II	Vaccination of all fish; stock year class separation has decreased needs
Water Quality	Inputs and discharges of nutrients and metals for point and non-point sources	No monitoring or reporting requirements	+	GREEN	II	No evidence of eutrophication or HABs from salmon farming in New Brunswick
Alien Species	Use and discharge of alien species	Strict control by government on the use and movement of alien species	+	GREEN	III	Industry uses native species
Diseases	Presence and spread of salmon diseases such as ISA	Large amount of monitoring and research by government and industry	+	YELLOW	II	Bay management measures instituted, but ISA still present. Concerns present re: continued movement of ISA between Maine and New Brunswick
Interactions with Marine Mammals	Presence and impacts on mammals	Unknown	?	NP	0	Unknown
Other Interactions with Marine Biota and Ecosystems	Presence, spread and vector of sea lice	Large amount of monitoring and research by government	-	YELLOW	III	Although well controlled and monitored, the use of slice is major issue in sustainability

I. ENVIRONMENTAL INDICATORS	KEY FACTORS AND PRESSURES	WHAT THE EVIDENCE SHOWS	TRENDS OVER THE LAST 5 YEARS	STATUS	CONFIDENCE ASSESSMENT	REASONS
Economic Indicators						
Profitability	Economic viability in regional, national and international contexts	Economists engaged with planning processes at industry and government levels	X	YELLOW	III	Higher prices in recent years have offset severe competition from Chile
Risk	Risk to investors and the public	Little known	X	YELLOW	III	No studies
Value Added	Planning for the value chain	Little known	?	NP	0	Unknown
Social Indicators						
Local Jobs	Planning for local employment	Little known	?	NP	0	Stewart (2001) report shows favorable impact on rural communities in province
Compensation	Comparable or better compensation with other primary production employment in the region	Little known	?	NP	0	Stewart (2001) report shows favorable impact on rural communities in province
Benefits	Social safety net/health care comparable or better to regional primary production employment	Little known	?	NP	0	Unknown
Safety	Adherence/concern to worker safety rules/international standards	Little known	?	NP	0	Unknown
Community Engagement	Regular engagement with community leaders	Little known	?	NP	0	Unknown
User Conflicts	Interactions with fishing, recreational and other industries	Little known	?	NP	0	Good fishery data on the herring and lobster fisheries. Studies on-going on integrated management using GIS layering
Laws and Regulations	Good governance and regulations insure positive benefits for ecosystems and society	Many federal, provincial and self-regulatory mechanisms in place	+	GREEN	III	Fisheries Act Oceans Act CSSP CFIA Regulation PMRA Regulations Health Canada CEA

Assessment Criteria are RED = unacceptable; YELLOW = room for improvement; GREEN = acceptable; NP = not possible to assess at this time. Confidence assessment are III = high; II =satisfactory; I = low; 0 = not possible. Trends are + = shows improvement; X = deterioration; - = no change; ? = no trend.

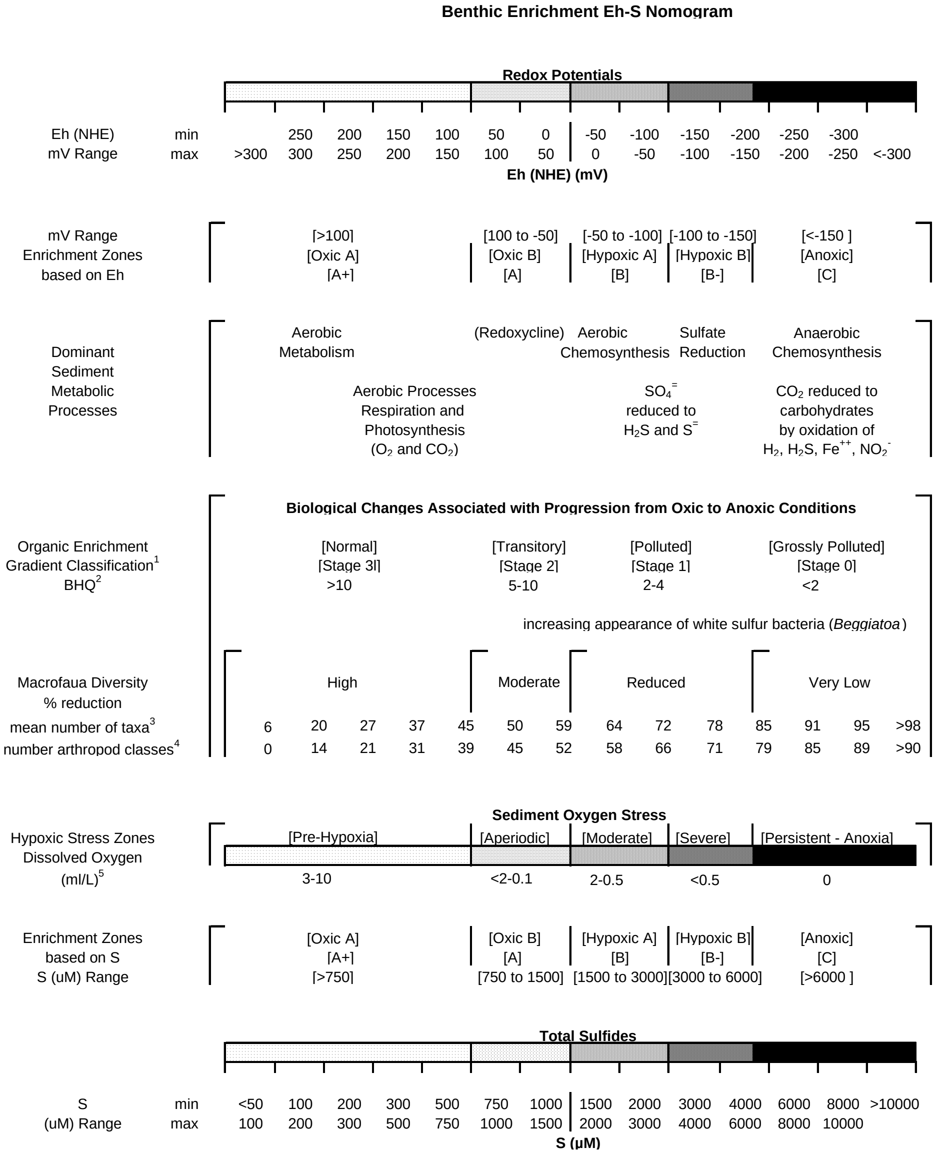


Figure 7.1: The Hargrave method for using free sulfide concentrations to predict marine benthic biodiversity. Modest losses (< 30%) in biodiversity occur between 100–500 µM S. Severe losses of biodiversity (70 to >90%) occur at free sulfide concentrations of 1500 to 3000 µM S.

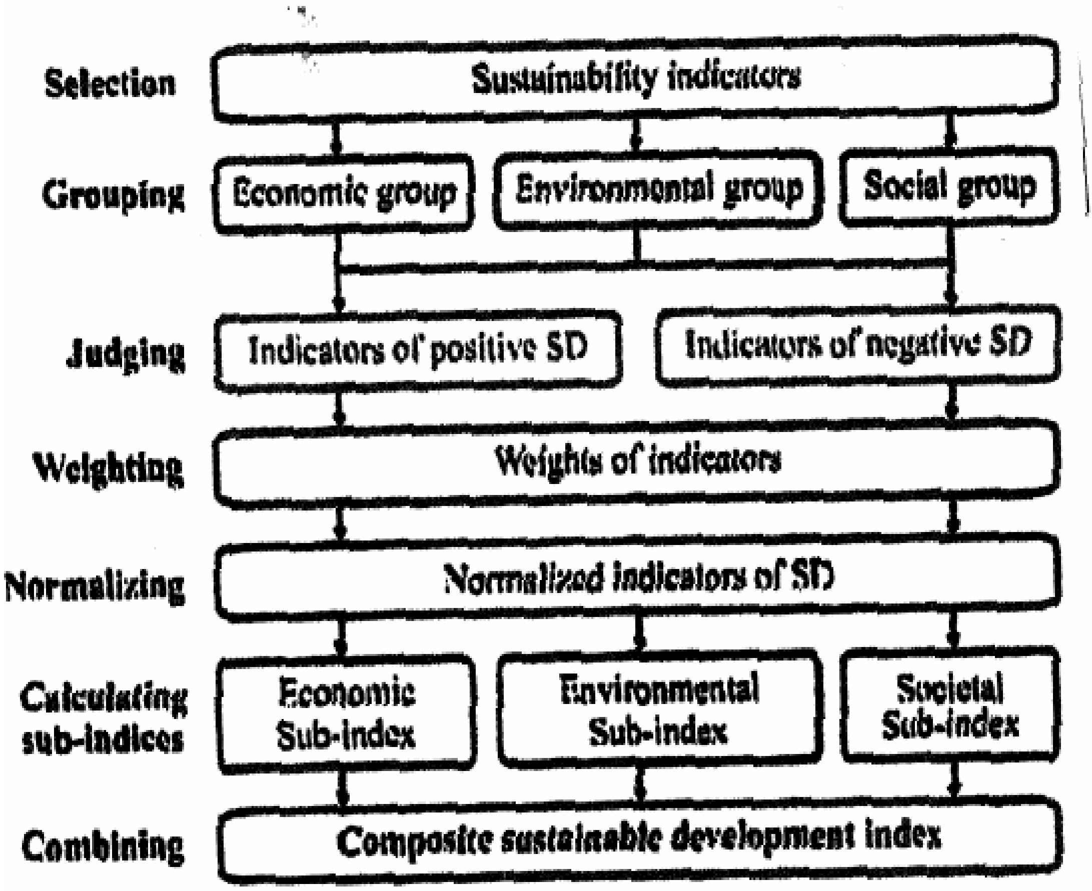


Figure 7.2: Procedure used for the development of a composite sustainable development index for industry (Kranjc and Glavic, 2005).

8 Evaluate the environmental impacts of integrated (multi-trophic) culture systems and provide recommendations on changes to EU regulatory frameworks that are required to accommodate this form of aquaculture operation (ToR d)

8.1 Abstract

Integrated Aquaculture, or Multi-Trophic Aquaculture, represents a global aquaculture sector of growing interest and potential development. Although much of this interest has been expressed through ongoing research initiatives, there has been some movement towards commercialization through large-scale testing of these opportunities.

Results of research programs in North America and Europe would suggest that there are low-level risks associated with contaminant transfers among Integrated Aquaculture components, but that these periodic risks are manageable in terms of husbandry practices and regional/international regulatory requirements for seafood inspection. The legal frameworks that currently apply to the aquaculture industry, in most jurisdictions, are considered sufficiently flexible as to accommodate the development of Integrated Aquaculture systems and it is clear that the environmental, economic and social benefits associated with this *sustainable* approach to aquatic food production outweigh the risks.

The advantages and disadvantages of integrated mariculture, based on our assessment of the environmental, social and economic considerations for this sector includes:

Advantages

- Reduction in net effluent discharges – environmental improvement;
- Shared operational resources is more cost-effective thus increasing profitability;
- Production intensification without environmental degradation;
- Diversification of production - market potential;
- “Sustainable” approach to aquaculture - public acceptance benefits;
- Development of aquaculture in remote coastal communities (economies of scale);
- Improvement to overall water quality (biofiltration effects) may reduce likelihood of disease outbreaks and Harmful Algal Bloom impacts.

Disadvantages

- Technically more complex - higher capital costs;
- Greater scope of technical expertise required to operate system;
- Approach and efficiency could vary significantly among sites;
- Handling (e.g., grading, harvesting) of individual species components may be more difficult;
- Monitoring & control of disease organisms may be more difficult;
- Public perception of growing conditions may require consumer education;
- Achieving optimal environmental conditions (e.g., temperature, salinity, water exchange, habitat) for each species within the system;
- Potential water quality impacts/transfer among integrated components (e.g., antibiotic use) will require management and the appropriate modification to jurisdictional regulatory procedures in consideration of seafood safety/inspection.

8.2 Introduction

The global demand for seafood continues to rise, and it is predicted that by 2030 nearly 50% of the world's seafood will come from aquaculture sources (Neori *et al.*, 2004). Systems ecology approaches are used to develop aquaculture production for the target species in a highly diversified, segmented manner, with numerous interconnections supplying inputs and outputs using local resources and recycled wastes and materials, planning for maximal job creation, and closing leaky loops of energy and materials that can potentially degrade natural ecosystems. The principles of ecological aquaculture are that it treats and recycles its own wastes rather than relying on natural environmental process and management process to mitigate cumulative environmental effects. It integrates people with technologies in new synergies to create new employment and biotechnological advances with a global view, integrating ecological principles with technological innovation in the global marketplace.

Two major categories of animals are cultured intensively around the world: suspension feeders such as bivalves, which feed directly on natural phytoplankton populations, bacteria and detritus; and species such as marine finfish and shrimp, which require an exogenous input of food for growth. Modern aquaculture systems are typified by intensive culture of a single species in open-sea net pens in coastal areas and in land based systems (ponds, tanks). There have been concerns about intensive aquaculture operations being feedlots (Wohlfarth and Schroeder, 1979), that they are energy intensive (Weatherly and Cogger, 1977), produce nutrient pollution loads comparable to human sewage (Bergheim and Sivertsen, 1981; Bergheim *et al.*, 1982), and that they lead to accelerated eutrophication, harmful algal blooms, and unacceptable modification of benthic ecosystems (Beveridge *et al.*, 1991; Pullin *et al.*, 1993; Folke *et al.*, 1994; Costa-Pierce, 1997). An approach to mitigate these environmental impacts, while optimizing production, is by integrating fed aquaculture (finfish, shrimp) with inorganic and organic extractive aquaculture (seaweed, shellfish, detritivores) whereby the wastes from one resource user becomes a resource (fertilizer, food) for the other.

At the farm level, the term integration can be understood under two main concepts: (i) rearing various species in the same production unit; or (ii) rearing a single, or multiple species downstream from one another. While using a combination of these two approaches is also considered integration, the rearing of different species in parallel in different rearing units is not. At a greater scale (e.g. an embayment), integration may address the optimization of shared resources among various aquaculture users (e.g. shellfish or seaweed farms around fish farms), but assumes that the integrated components (species) are situated within the influence of the system component upon which it directly depends for waste transfer and utilization.

In exploring the scientific literature regarding the co-culture of marine species, a range of terminology has been developed and used interchangeably to refer to this form of mariculture. These include:

- Polyculture;
- Integrated Aquaculture;
- Multi-Trophic Aquaculture;
- Ecological Aquaculture;
- Sustainable Aquaculture;
- Sustainable Ecological Aquaculture (SEA) Systems.

In this document, these terms may be used interchangeably to reflect the position and preferences of the international researchers, but should be regarded as comparable in our overview of the concepts, types and effectiveness/benefits of these integrated multi-species/multi-trophic systems.

8.3 Benefits of *Integrated-MTA* Systems:

The development of *integrated*-MTA systems, in favour of present monculture approaches to aquatic food production, offers a variety of environmental, social and economical benefits, the combination of which will help define the sustainability of the proposed system.

8.3.1 Environmental Benefits

Intensive fed aquaculture (finfish and shrimp) throughout the world has raised concerns about the environmental impacts of such mono-specific practices, especially where activities are highly geographically concentrated or located in suboptimal sites whose assimilative capacities are poor and consequently prone to being exceeded. Traditional methods of treating aquaculture effluent have been built on technological solutions such as mechanical separation of solid particles using screens, sedimentation, filtration (Cripps, 1994) and biological filtration of dissolved nutrients. Integrated farming methods are built on ecological engineering practices, where “extractive” (i.e. bivalves, macro algae, polycaetes, sea cucumbers) and “fed” (salmon, sea bream) species are grown simultaneously, and have been proposed as a means for recycling the nutrients and particulate wastes from fish cage farming, shellfish facilities, and land bases farms.

All of the compounds in fish food as well as the by-products of metabolism are potential waste products, and are lost via two pathways. Organic carbon and nitrogen compounds can be lost directly (due to uneaten food pellets), and indirectly (due to faeces), (Gowen and Bradbury, 1987), while other nitrogenous wastes (ammonia and urea) as well as phosphate, are dissolved into the water column surrounding the farms. It is the potential ability of these compounds to cause hypernutrification and eutrophication (Folke and Kautsky, 1992), particularly in the case of poorly sited and managed marine aquaculture, in both the benthic and pelagic realms that is of concern. Although some studies of aquaculture have indicated that ecological impacts may be localized and reversible by fallowing (Stewart, 1998; Newell, 2004), the management and treatment of aquaculture effluents remains an important issue.

Poorly sited and managed marine aquaculture operations have caused environmental impacts, but assessments of impacts have too often been based upon out-dated literature, scientific misinterpretation, and advocacy. Review of studies of benthic impacts of marine aquaculture has shown ecological impacts to be localized and reversible by fallowing (Stewart, 1998, Hargrave, 2003; Newell, 2004). After extensive studies, the Net Pen Advisory Work Group of the Washington Department of Ecology (WDOE) found that benthic impacts of salmon farming in Puget Sound was limited to within 30 m of the net pen perimeter, and that impacts were reversible by fallowing. Based upon their data the WDOE decided to mange salmon pens by allowing a sediment impact zone within a 30 m of the edges of the cages. Outside of this perimeter, water quality and benthic "performance standards" would have to be met (Rensel, 2001). However, additional research needs to be conducted since very few data exist to date on the long-term assimilative capacity of benthic communities in different climatic regions. For example, Angel *et al.* (1992) found that organic matter decomposition in sediments under fish cages in the Gulf of Aqaba may be 3–4 times greater than in temperate waters. In the case of shellfish farming, worst case scenarios have shown that the sedimentation rate under a suspended longline mussel (*Mytilus edulis*) culture site in a Swedish fjord was almost four times that of control areas and that sediment deposition could be up to 1000 g organic carbon $m^{-2} \cdot y^{-1}$ (Dahlback and Gunnarsson,1981). This is resulting from changes in the hydrodynamics and the production of faeces and pseudofaeces.

In concept, the design of an integrated aquaculture system will provide a balance of biological components (co-cultured species) such that the production of wastes from one component is used (extracted, ingested) in a manner that optimizes the growth of a second, the second providing inputs to a third, etc. In a well-balanced system this relationship provides the environmental benefits associated with polyculture, and is the basis of definitions such as Sustainable, or Ecological Aquaculture.

Initiatives on the east coast of Canada (New Brunswick) have recently evaluated the performance of mussels (*Mytilus trossolus*) and large macrophytes (*Laminaria*) cultured within the infrastructure of an open net-cage salmon (*Salmo salar*) aquaculture facility. Chopin *et al.* (1999, 2001) and Neori *et al.* (2002) demonstrated that these integrated species perform significantly better within the influence of the salmon net-cage systems as compared with a monoculture arrangement removed from the apparent effluent effects of the finfish system. In contrast, a study in Tasmania, Australia (Cheshuk *et al.*, 2003) indicated that mussels (*Mytilus planulatus*) grown within 70 meters of a salmon (*Salmo salar*) farm revealed only very minor improvements in growth (shell height) and condition over the 14-month grow-out period. Stirling and Okumus (1995) also showed slight increases in mussel culture performance, grown at two salmon farm sites in Scotland, and suggested that enrichment of the seston field by organic material from the salmon farm was likely contributing to this observed elevation in growth. Cross (unpublished) observed that oysters (*Crassostrea gigas*) and scallops (*Patinopectin yessoensis*) showed neither a positive nor negative growth change as a result of co-culture with Pacific (*Oncoryhnchus tshawytscha)* or Atlantic *(Salmo salar*) salmon.

Although the combination of species proposed for a Multi-Trophic Aquaculture (MTA) system will determine to what degree the transfer or organic waste materials are effectively used among the co-cultured species, the literature has suggested that these direct environmental benefits are highly variable among systems and may in fact represent a smaller benefit in open marine systems than anticipated. However, the environmental benefits of MTA are not constrained solely to the direct assimilation of waste constituents among the co-cultured species, but will also be achieved indirectly through the physical design/configuration and orientation of such a system with respect to adjacent, and potentially sensitive marine habitats. There are no known field studies investigating the possibility of applying MTA from the basis of a shellfish farm. This approach, however, has been investigated though modeling evaluations focusing mainly on the addition of macroalgae culture to shellfish operation (Duarte *et al.* 2003, Nunes *et al.* 2003). Although this would not be an optimal approach for MTA, it could provide similar benefits for area where finfish farming is not feasible or permitted, such as shallow coastal areas. However, the main benefit would be through the addition of detritivores for minimizing the impact of biodeposits.

8.3.2 Social Benefits

The social benefits associated with the development of marine integrated aquaculture include: (i) optimizing potential culture opportunities in jurisdictions that are constrained by available space (e.g., New Brunswick, Canada; small EU countries); (ii) provision of development opportunities in remote coastal regions that are otherwise constrained by operational logistics (e.g., north coast of western Canada, southern coast of Chile, north coast of Norway); (iii) provision of product diversification at the coastal community level that could stimulate the development of a larger and more diverse secondary industry support system; and/or (iv) improving public approval of aquaculture or aquaculture environmental accountability.

In southern Europe, where coastal zones have been already heavily impacted, the restoration of abandoned wetlands and the optimal use of existing ponds is a coastal zone management issue. Maintaining these sites is costly and cannot be handled by public funds only. The Common Agriculture Policy and Common Fishery Policy requires primary users of the natural resources (e.g. agriculturists, fishermen, aquaculturists) to implement an ecosystem approach in the management and conservation of the environment and landscape. It considers polyculture (MTA) as a utilization approach for these areas that could provide restoration at a lower cost for society.

8.3.3 Economic Benefits

While most of the scientific community assessing the potential for Multi-Trophic Aquaculture agree that this approach has considerable merit in terms of environmental benefits (at least in theory), the question remains: "*why has this not yet been widely accepted and developed at the commercial level*?"

It is clear that while there is *potential* for MTA, commercialization is based on an evaluation of not only opportunity, but of economic risk in terms of associated capital/operational costs, performance certainty, impact and integration of multi-products to existing markets and sales pathways, personnel requirements, differential/fluctuating component species pricing, and profitability. The widespread commercial development of MTA by industry has not yet occurred most likely due to one or a number of these business uncertainties. The remaining challenges facing future research and development of these systems include initiatives that will address the practical aspects of commercial-scale MTA facilities, and to offer results that can be assessed by the investment and corporate community that will allow these development risks to be properly considered.

The economic benefits offered by the commercial development of integrated MTA systems will be evaluated using factors that contribute directly to the cost-effectiveness of these aquaculture systems over current monoculture approaches, and will have important implications to MTA system design and engineering. Profitability, versus system function (potential interferences among components), operational logistics, capital expense, and training requirements (complexity of employee knowledge-base), will jointly determine the level and acceptability of investment and commercial development risk.

In remote coastal areas operational efficiencies become critical in determining the economic viability of a proposed shellfish aquaculture facility, and are often cited as the economic constraints to such development (despite optimal growing conditions). The development of an MTA system provides the opportunity to capitalize on the infrastructure and operational activities/schedules available through the other culture components (e.g., finfish). In particular, transportation costs (e.g., for crew, feed delivery, supplies, seed, harvests) represents a significant, and usually limiting factor for developing these types of monoculture operations in remote coastal regions. With the development of an integrated finfish-shellfish aquaculture system based on a modified (stretched) 12-cage steel net-cage facility, Cross (2004) estimated that the capital and operational costs realized by the shellfish component of the system to be between 66 and 79% of that of an independent shellfish operation of similar size. Furthermore, his projections suggested that profitability of the shellfish aquaculture component ranged from 0.8–20% (net profits), compared to that of an independently operated shellfish operation of similar size that would otherwise realize a net loss – these margins would vary depending upon distance from an operational base (port).

8.4 MTA and Product Quality Issues

The introduction of Integrated Aquaculture systems to the aquatic food production industry has a number of potential issues associated with product quality and safety. The issues related to contaminant transfer among system components, the organoleptic characteristics of co-cultured seafood, and/or the market (public) perception of seafood grown within a cascading waste stream, will determine whether such systems can successfully be developed.

8.4.1 Contaminant Transfer & Seafood Safety

Cross (2004) completed a three-year research program that assessed the interactions between salmon (*Oncoryhnchus tshawytscha, Salmo salar*) and shellfish (*Patinopectin yessoensis, Crassostrea gigas*), with a focus on the possible water quality and hence seafood safety issues associated with co-culture of these species. Continual release of micronutrients (trace metals:

Cu, Zn, Se, etc.) associated with feed inputs over an entire production cycle revealed no significant tissue accumulations in shellfish grown adjacent to and downstream of the salmon organic waste discharge. However, the periodic entire-farm treatment with antibiotics (oxytetracycline, in-feed) did result in a downstream affect on shellfish tissues, with elevations of OTC residues detected as far as 150 metres from the salmon farm. The tissue levels of OTC were all below USFDA safe limits, and the clearance of these residues from all of the shellfish occurred within 14–30 days depending upon seasonal influences (water temperature, shellfish feeding rates, etc.).

The EC *GENESIS* project (European Community Contract n° IPS-2000-00102) is currently addressing the development of a generic approach to sustainable integrated marine aquaculture for European environments and markets, and has included a project component to evaluate an assurance of product quality and safety for fish-microalgae-bivalve integrated systems (cascading pond system) in accordance with EU regulations. Results from these studies have shown no biomagnification effects from trace metals (Cd, Cu, Zn, Pb) available to the shellfish component (clams, oysters) via the fish component, although medicated feed treatments with oxolinic acid resulted in elevated levels in both of the shellfish species. Decrease of OA concentration was very quick after the end of treatment (62 $ng.g^{-1}$ after 2 days), and less than the Minimum Residue Level. The OA concentrations observed in bivalves (0.04 $\mu g.g^{-1}$ in C. gigas and 0.033 $\mu g.g^{-1}$ in clams) were closely related to OA seawater concentrations.

In terms of potential pathogen transfers among MTA system components, the EU GENESIS project also documented *E. coli, Salmonella*, and *Vibrio spp*. levels in seawater, in oyster, and in the fish feed used in the integrated system. No *Salmonella* or *E. coli* contamination was observed in water or in shellfish flesh. Low concentrations of *E. coli* contamination in clams was observed, but these were well below the critical level. Observed *Vibrio* spp. in the inflow was attributed to *Vibrio alginolyticus*, *which is v*ery frequent in brackish and marine water systems and is not considered pathogenic.

Initial results of research programs in North America and Europe would suggest that there are low-level risks associated with contaminant transfers among Integrated Aquaculture components, but that these risks are manageable in terms of husbandry practices and regional/international regulatory requirements for seafood inspection.

8.4.2 Organoleptic Properties

The growth of the non-fed components of an MTA system (e.g., shellfish) will be determined, in part, by the nutritional qualities of the organic waste stream supplementing their diet. Hence, the question of palatability of MTA products has been raised as an issue potentially affecting marketability and economic viability of these integrated aquaculture approaches.

Cross (2004) conducted organoleptic tests (66 participants) to determine if taste, odour or texture differences could be detected between samples of scallops and oysters grown adjacent to a salmon aquaculture facility from those cultured at monoculture shellfish sites (no species interaction effects). Results of this study revealed no significant difference in the taste evaluations, suggesting that a consumer would not be able to detect a difference in products grown within an MTA system.

In eastern Canada, Robinson (unpubl.) conducted a similar organoleptic study, but in this case used the blue mussel (*Mytilus edulis*) – also grown adjacent to a salmon aquaculture facility. Results were comparable, and indicated that no significant difference could be detected among shellfish samples grown in monoculture or polyculture systems.

8.4.3 Public Perception of *Integrated*-MTA

Despite scientific evidence that supports the *safe* development of Integrated Aquaculture systems, given an appropriate operational and regulatory framework, the success of this approach to aquatic food production will inevitably rely upon market (public) response and acceptance.

An eastern Canada AquaNet project (Barrington *et al.*, 2005) compiled responses to a survey/questionnaire that queried respondents as to their opinions regarding the benefits of MTA. Once educated as to what and how MTA works, the study respondents were generally supportive of the idea and of the inherent environmental and socio-economic benefits of MTA. All of the study participants (100%) showed a willingness to eat seafood products grown in proximity to salmon, yet most felt that appropriate testing be conducted on the harvested products (particularly trace metals, antibiotic residues, and potential pathogens).

In the EU GENESIS project, a study was conducted to estimate the awareness of consumers to eat shellfish and fish coming from integrated aquaculture systems. It was based on the "focus group" methodology. While the French and UK consumer did not indicate any difficulties to consume fish, French consumers showed some reluctance to consume shellfish grown downstream of a fish farm when compared to shellfish cultivated in the usual culture environments.

8.5 Current Examples of *Integrated-MTA* Systems:

Table 8.1 provides a summary of the current research and pilot-scale integrated aquaculture facilities. Many of these are based on small experimental systems, within the R&D process, and may be of questionable sustainability and economic viability.

Four examples of commercial-scale and/or projects in development at pilot-scale are described below.

The following are examples of pilot or commercial scale MTA projects/developments. Selected results specific to each of the projects are presents. It is assumed that the general environmental and socio-economic benefits described above are applicable to each.

8.5.1 Mediterranean

Integrated aquaculture is practiced in the Mediterranean with, to our knowledge, no pilot or commercial-scale systems currently occurring in Europe north of the English Channel. This could be explained by the fact that integration is easier to develop from extensive systems where they do exist than from intensive ones. These systems represent a range of intensification from the extensive to the intensive forms (Table 8.2).

Innovation EC project "GENESIS" concentrates on prototypes of integrated systems including finfish (sea bass or sea bream) and bivalves (oysters and Manila clam), where bivalves ponds were downstream of fish ponds using a flow-through system, in France and Israel. The systems included fish ponds, settling tanks for organic matters, phytoplancton production units, and oyster and clam ponds. Both biological (biotoxins and bacteria) and chemical (heavy metals, veterinary drug residues, and antibiotic resistances) hazards have been investigated, revealing no biomagnification issues associated with the trace metal constituents but some periodic influences of bacteria and antibiotic residues (although at low levels of accumulation). Recommendations from this project involved aspects of MTA farm operation, including: (i) minimizing antibiotic or chemotherapeutic treatments; (ii) synchronization of the emptying of the ponds at the end of treatments to facilitate bivalve depuration; (iii) preference for pond design that avoided ones with bottom sediment, thereby reducing risk of contaminant/bacterial persistence within the system; and (iv) consideration of procedures for removal of the solid matter from the ponds post-treatment.

Extensive production of fish in Lake Quarun in Egypt (El Gayar, 2003) - The technique is based on restocking a salt lake with various juvenile species which are mainly caught from the wild. Production in the lake is estimated at 23 000 tons, and is extensive (yield of 150 kg/ha per year). The main species produced are mullets (all species), seabream, seabass and shrimp. Adjacent earthen ponds are used for rearing mullet juveniles, using fertilisers to enhance their productivity. The major threat to sustainability is the capture of wild fry, particularly mullet, whose reproduction cycle is not yet closed. Mullet production in Egypt is around 160 000 tons per year.

Extensive production of fish in Valliculture in Italy - Around 43 000 ha of earthern ponds in brackish waters are cultivated, mainly in the Po River delta. These areas produce 3000 tons of mullets, 1000 t of seabream, 1000 t of seabass and 200 tons of eels per year. Seabass and seabream fry come from hatchery production, while mullet and eel are from the wild. The main income in these areas is more from tourism and hunting rather than from aquaculture. Nevertheless, aquaculture makes the enterprise profitable. The valli are traditionally extensive systems (only 11 has also semi-intensive and 3 intensive structures), with a productivity ranging between 30 kg/ha/year (Valli di Comacchio) and 150 kg/ha/year (Valli Venete). Productivity of valliculture is today significantly reduced, mainly because of the strong impact of ichthyophagous birds, especially on juvenile stages - a single individual can eat 100 to 400g of fish per day. Production costs are higher compared to intensive marine systems and economic sustainability of valliculture is strongly depending on the improvement of production cycles and on product differentiation.

Semi-intensive production of fish in Spain (Andalucia) and Portugal - The production system is a combination of various systems based on levels of intensification. Extensive, semi intensive and intensive technologies are mixed to produce seabream, seabass, mullets, eel, sole and shrimps: 60% of the 6700 tons annual production is from the semi-intensive units, the bulk of it being represented by sea bream from land based hatcheries. The remaining is intensive production (34%) and extensive (300 tons, 6%). Sole, mullets, shrimp and eels are produced in the extensive system. One advantage of this system is the water reuse from the more intensive part to the extensive one, thus reducing the need for water. It has been observed that the nutrient and organic matter contents in the effluent from the intensive part sustain the production of worms (for soles) and other preys in the extensive ponds. Attempts are undergoing to cultivated clams (*Tapes decussatus*) in the same ponds. Some of these farms charge tourists for admission (aquatourism).

Semi intensive production of shrimps and oyster in Southern France - The level of production is very low (60 tons of *Paeneus japonicus* per year), but is sustainable, having been in operation for 20 years. Five years ago oysters (*Crassostrea gigas*) were incorporated within the same ponds. Oysters are able to utilise the phytobenthos that is resuspended by shrimp foraging activity. By increasing the income through aquaculture the combined system has proved to increase the economic sustainability of the farm.

8.5.2 Israel

Description of a land based pilot scale facility aiming at integrating fish-shellfish-algae - The most promising developments in this respect have been realized in Israel using seabream as the fish species and *Ulva* for the seaweed, which ultimately feeds abalone (*Haliotis discus*) or urchin (Shpigel, 1996). Abalone fed with *Ulva* are reared on inflowing water, then seabream are reared in intensive circular tanks, and ultimately *Ulva* (fed to abalone) are cultivated in raceways using effluent water from seabream, which are downstream from the sedimentation tank. Water from *Ulva* can be re-used for seabream rearing. The main features of the system are summarized (Table 8.3).

Contrary to rearing various species in one polyculture system, the integration of various monoculture through water transfer alleviates one of the deficiencies of the former: a smaller yield of each organism. This is made possible because the fish and the algae have opposite effects on the water quality i.e. CO2, O2, dissolved nitrogen and phosphorus, dissolved heavy metals, pH.

The nitrogen budget is very promising since the uptake efficiency by Ulva was about 90% of the dissolved nitrogen on an annual average, at a ratio of 3–5 g of nitrogen per square meter per day. Instead of disposing 175g of nitrogen per month (for 10 tons of fish production at an average ammonia concentration of 12 $mg.l^{-1}$) in the environment, the algal treatment allows only 25g (2 $mg.l^{-1}$ of ammonia) to enter the water body (to be compared to the 5g from the inflowing water). The calculation made by Neori *et al.* (2000) indicates that to depurate nitrogen, a farm producing 1000 ton of seabream per year would need around 15 ha of Ulva biofilter and 7 ha of tanks supporting the production of 660 tons of abalone which is, however, more efficient than sea urchin in the same context.

This system is well adapted to conditions in Israel (particularly temperature and light) but, these results have to be taken cautiously for any further extension. Comparable pilot scale experiments in southern France (Deviller *et al.*, 2004), in less favorable conditions, the seaweed growth is susceptible to seasonal variations. In more northern regions, the most probable developments would occur by using phytoplankton for bioreactors and filtering bivalves for secondary production (Hussenot, 2004).

Even if the investment breakdown is equally shared between the three stages, it appears that the main revenue comes from the abalone production. Based on the figures of Table 8.3, the income from the farm would be 1.05 million Euros from the sea bream and 6.5 million Euros from the abalone. The farm would be barely profitable without producing abalone, the addition of this unit raised the expected profit from nearly zero to 2.5 million Euros. At any case, labor costs are critical to this system, mainly for the abalone unit: the expected needs in work force are 10–12 permanent employees. In addition the capital to be invested is very high. Again based on the figures from Table 8.3, the initial investment would be 1.3 million euros.

These figures have to be compared to the global revenues from a similar cage farm. According to Neori *et al.* (2004), the production cost are comparable if the cost incurred for water treatment would be added in the form of taxes (according to the polluter-pays principle). From these results, a commercial farm has been put in operation in 2004.

Clearly it is a matter of governance which will decide whether these types of systems will be developed. The potential negative effects of releasing nutrients in the environment has a cost and a risk analysis will be required.

8.5.3 Eastern Canada

An experimental commercial scale project in the Fundy Isles Region, New Brunswick, Canada on the feasibility of the integrated multispecies aquaculture by combining inorganic extractive aquaculture of the kelp, *Laminaria saccharina*, and organic extractive aquaculture of the blue mussel, *Mytilus edulis*, with the fed aquaculture of salmon, *Salmo salar* is in progress. The project in co-operation with the Atlantic salmon Aquaculture industry and is investigating the incorporation of mussel and kelp culture facilities on existing commercial salmon culture stities. Food safety and physical/chemical modelling (especially of the oxygen budget) and the socio-economic studies are included (Chopin *et al.*, 2002; 2003; 2004).

Results from these studies suggest that the mussels and kelp are utilizing the wastes from the salmon culture to their benefit as well as the environment. Kelps grown in the vicinity of salmon farms increased their growth rates by 46% in comparison to kelps grown at reference sites. Blue mussel, *Mytilus edulis*, was developed to show that mussels are not only capable of capturing excess food particles from the fish farm but also increase their feeding rates in response to the presence of these particles. Seston levels at salmon farms are elevated by a factor of 2 to 4 over ambient levels and are of very high quality (up to 90% organic). Enhanced growth rates at farm sites (50% more than that of mussels at reference sites) and accelerated production times to commercial size (approximately 18 months from socking) reflect this increase in food energy, as mussels ingest fish food particles with approximately the same efficiency as phytoplankton species.

In 2006 the Eastern Canada research program will expand to evaluate the efficacy of IMTA at two commercial-scale farm sites. The five-year program will continue to examine the combination of salmon-mussel-kelp, but will assess the processes affecting system performance at commercial levels of production. The proposed research will document environmental influences, economics, health issues, and system component performance.

8.5.4 Western Canada

The Pacific SEA-Lab is currently being developed as a commercial-scale R&D site for evaluating the efficacy of Sustainable Ecological Aquaculture (SEA) systems. This multi-trophic aquaculture facility will be based on current salmon aquaculture infrastructure (steel netcage system) and will comprise a fed component of sablefish (*Anoplopoma fimbria*), an inorganic extractive component of kelp (*Laminaria saccharina*), with a 2-layer assemblage of scallops (*Patinopectin yessoensis*) and mussels (*Mytilus edulis*) to extract the fine particulate organic fraction. To accommodate the loss of settleable organic solids from the system, the potential use of sea cucumbers (*Parastichopus californicus*) and nereid polychaetes will be evaluated.

The Pacific SEA-System research program (2006–2010) will focus on commercial MTA development issues including: (i) balance of system components; (ii) system performance – environmental and socio-economic; (iii) disease persistence/transfer; (iv) water quality effects; (v) system optimization – engineering, operational; and (vi) product quality. The western Canada initiative is being conducted in parallel with the eastern Canada program, with a joint national body (Canada SEA-Lab) using these regional R&D programs to facilitate regulatory reform for MTA development in Canada.

8.6 Policy & Regulatory Constraints

The legal instruments (policies and regulations) that currently apply to the aquaculture industry, in most jurisdictions, are considered sufficiently flexible as to accommodate the development of Integrated Aquaculture systems. In a comparative legal analysis, White and Glenn (2005) conclude that the legal frameworks that govern aquaculture across Europe can, in most cases, apply to the installation and subsequent management of biofiltration components (e.g., shellfish, macrophytes, sea cucumbers) with little or no significant modification. The current European legal frameworks will allow the introduction of biofilters in order to facilitate environmental impact mitigation (waste reduction), but are also viewed as being able to consider the regulatory issues associated with harvest of these biofilters (e.g., shellfish) as a secondary (or tertiary) level of production within an integrated aquaculture system.

The dilemma faced in regulating the introduction of biofilters to a finfish monoculture system relates to governance procedures that: (i) define waste discharge limits/standards through a permitting process; and (ii) establish specific levels of farm site production. When introducing biofilters, this culture component presumably reduces (changes) organic loading (waste impact mitigation) but at the same time increases site production, albeit across more than a single species. How to accommodate this apparent contradiction has thus become the focal point of regulatory reform discussions in countries considering commercialization of Integrated Aquaculture, yet currently operate using these regulatory procedures.

In North America, regulation of the environmental effects of aquaculture has moved towards a performance-based approach, with operational limitations focussed primarily on achieving environmental (benthic, water quality) standards. The inclusion of Integrated Systems within a performance-based paradigm should therefore be less problematic in terms of licensing/operation, assuming that improved environmental performance resulting from the introduction of biofiltration components would continue to satisfy the established standards or performance thresholds (despite increased overall site production).

The need to satisfy jurisdictional and international agreements/regulations regarding seafood safety (regarding bacterial, antibiotic, chemical contaminant loading in MTA products) will require procedural modifications to reflect the co-culture of species. However, given that the use of such treatments in the fish component typically has sufficient procedural safe-guards (e.g., prescribed treatment dosage/applications, required tissue clearance periods, acceptable product tissue levels), it is assumed that these protocols should be adaptable to incorporate the other species of an Integrated Aquaculture system that may be exposed to the residues released during and immediately following treatment.

White and Glenn (2005) suggest that while additional administrative protocols or procedures will necessarily evolve in response to the development of MTA within individual jurisdictions, that this added bureaucracy should not be prohibitive. In fact, these *adjustments* will most likely be determined by regional politics and by the options available given scientific support of their effectiveness, as well as by the economic and financial considerations.

While the policy and regulatory constraints to incorporating Integrated Aquaculture into existing legal frameworks does not appear prohibitive, the premise that this approach to aquaculture attempts to move towards system *sustainability* should be viewed by society as positive, and an approach that should be encouraged. Robinson (2004) suggests that the role of government, in reforming aquaculture policy to incorporate MTA, be one of encouragement for industry sectors that follow these tenets. He further recommends that incentives or penalties, similar to those that have been applied to environmental or health behaviour of people in land-based systems (e.g., fuel or cigarette taxes, higher insurance premiums for high-risk activities, pollution tax, etc.) be considered for Integrated Aquaculture systems.

8.7 Recommendations:

The introduction of Integrated Aquaculture systems to the aquatic food production industry is viewed as a positive development given the inherent environmental, social and economic benefits associated with the design of such systems. However, there are many technical details of integrated mariculture that need to be addressed through further research. In terms of the management of pond-based integrated aquaculture systems these efforts include, for example, the development of:

- Algal control strategies;
- Nutritional strategies, including fertilization and supplemental feeds (micoralgae, zooplankton, artemia, polychaetes);
- Methods for mass production of juveniles for system stocking; and
- Optimal fish stocking and fertilization (through modelling).

With respect to open, coastal integrated aquaculture (intensive or extensive), similar such research and development initiatives need to be completed. These comprise, for example:

- Evaluating the efficacy of these systems in terms of environmental impact mitigation;
- Assessing the commercial viability/profitability of these systems
- Determining an appropriate number and species composition of trophic components (balancing organic materials transfers and minimizing the net system loss of wastes);
- Identifying and developing management approaches for potential water quality interaction effects (e.g., antibiotic residues); and
- Providing science-based recommendations for changes to regulatory procedures to accommodate Integrated Aquaculture development.

However, to successfully transfer the concept of integrated aquaculture to industry the technical challenges associated with the practical aspects of commercial-scale facilities must be addressed, and these results presented in a context that could be assessed by the investment community that would consider these development opportunities. Ongong research should consider multi-disciplinary, commercial-scale testing of integrated aquaculture systems to permit all of the environmental, social and economical issues to be addressed accordingly.

8.8 References

Ahlgren, M. 1998. Consumption and assimilation of salmon net pen fouling debris by the red sea cucumber *Parastichopus californicus* : implications for aquaculture. J. World Aquac. Soc. 29: 133–139.

Ahn, O., Petrell, R.J. and Harrisin, P.J. 1998. Ammonium and nitrate uptake by Laminaria saccharina and Neocystis luetkeana originating from a salmon sea cage farm. J. Appl. Phycol. 10: 333–340.

Angel, D., Krost, R., Zuber, D., Mozes, N. and Neori, A. 1992. The turnover of organic matter in hypertrophic sediments below a floating fish farm in the oligotrophic Gulf of Eliat. Bamidgeh 44: 143–144.

Barrington, K., Ridler, N., Chopin, T., Robinson, S., Page, F., MacDonald, B., and Haya, K. 2005. Development of integrated aquaculture (finifsh/shellfish/seaweed) for environmentally balanced diversifiction and social acceptability: scoia perceptions of integrated MTA. AquaNet Project EI-17 Final Report. 80 pp.

Benson, J. A., Neushul., M., Harger, B. W. W., Braden, E., and MacDonald, T. 1986. Polyculture of abalone and algae in the sea. NSF/ISI-86094, 23pp.

Bergheim, A. and Sivertsen, A. 1981. Oxygen consuming properties of effluents from fish farms. Aquaculture 22: 185–187.

Bergheim, A., Sivertsen, A., and Selmer-Olsen, A. 1982. Estimated pollution loadings from Norwegian fish farms. I. Investigations 1978–1979. Aquaculture 28: 347–361.

Beveridge, M., Phillips, M., and Clarke, R. 1991. A quantitative assessment of wastes from aquatic animal production. *In*: Aquaculture and Water Quality. (Ed by D. Brune and J. Tomasso), pp. 506–533. Advances in World Aquaculture, Volume 3. World Aquaculture Society, Baton Rouge, LA

Buschmann, A., Troell, M., and Kautsky, N. 2001. Integrated algal farming: a review. Cah. Biol. Mar. 42: 83– 90.

Buschmann, A.H., Mora, O.A., Gomez, P., Bottger, M., Buitano, S., Retamales, C., Vergara, P.A. and Guiterrez, A. 1994. *Gracilaria chilensis* outdoor tank cultivation in Chile: use of land-based salmon culttre effluents. Aquacul. Eng. 13: 283–300.

Buschmann, A.H., Troell, M., Kautsky, N. and Kautsky, L. 1996. Integrated tank cultivation of salmonids and *Gracilaria chilensis* (Gracilariales, Rhodophyta). Hydrobiologia 326/327: 75– 82.

Buschmann, A.H., Westermeier, R., and Retamales, C. A. 1995. Cultivation of *Garcilaria* on sea bottom in southern Chile: A review. J. Appl. Phycol. 7: 291–301.

Cheshuk, B.W., Purser, G.J., and Quintana, R., 2003. Integrated open-water mussel (*Mytilus planulatus*) and Atlantic salmon (*Salmo salar*) culture in Tasmania, Australia. Aquaculture 218: 357–378.

Chopin, T., and Bastarache, S. 2002. Finfish, shellfish and seaweed mariculture in Canada. Bull. Aquat. Assoc. Can. 102: 119–124.

Chopin, T. and Bastarache, S. 2004. Mariculture in Canada: finfish, shellfish and seaweed. World Aquacult. 35 (3): 37–41.

Chopin T., Bastarache, S., Belyea, E., Haya, K., Sephton, D., Martin, J. L., Eddy, S., and Stewart, I. 2003. Development of the cultivation of *Laminaria saccharina* as the extractive inorganic component of an integrated aquaculture system and monitoring of therapeutants and phycotoxins. 2003. J. Phycol. 39 (S1) 10.

Chopin, T., Buschmann, A.H., Halling, C., Troell, M., Kautsky, N., Neori, A., Kraemer, G.P., Zertuche-Gonzales, J.A., Yarish, C., and Neefus, C. 2001. Integrating seaweeds into marine aquaculture systems: a key toward sustainability. J. Phycol. 37, 975–986.

Chopin, T., Yarish, C., Wilkes, R., Belyea, E., Lu, S., and Mathieson, A. 1999. Developing *Porphyra*/salmon integrated aquaculture for bioremediation and diversification of the aquaculture industry. Journal of applied Phycology 11: 463–472.

Chow, F., Macciavello, J., Santa Cruz, S. and Fonck, O. 2001 Utilization of *Gracilaria chilensis* (Rhodophyta: Gracilariaceae) as biofilter in the depuration of effluents from tank cultures of fish, oyster, and sea urchins. J. World Aquac. Soc. 32 : 214– 220.

Chung, I., Kang, Y. H., Yarish, C., Kraemer, G. P. and Lee, J. 2002. Application of seaweed cultivation to the bioremediation of nutrient-rich effluent. Algae 17 : 187–194.

Cohen, I., and Neori, A. 1991. Ulva lactuca biofilters for marine fishpond effluents: I. Ammonia uptake kinetics and nitrogen content. Bot. Mar. 34 : 475– 482.

Cross, S.F. 2004. Finfish-shellfish integrated aquaculture: water quality interactions and the implications for integrated multi-trophic aquaculture policy development. Bull. Aquacul. Assoc. Canada, 104-3: 44–55

Costa-Pierce, B. A. 1997. Environmental impacts of nutrients discharged from aquaculture: towards the evolution of sustainable, ecological aquaculture systems. *In*: Aquaculture and Water Resource Management, D. J. Baird, M. C. M. Beveridge, L. A. Kelly and J. F. Muir (eds), pp. 81–113. Blackwell Science, Oxford, UK.

Cripps, S. J. 1994. Minimizing output: treatment. J. Appl. Ichthyol. 10: 284–294.

Danakusumah, E., Kadowaki, S. and Hirata, H. 1991. Effects of coexisting *Ulva pertusa* on the production of Kruma prawn. Nippon Suisan Gakkaishi, 578, 1597.

Deviller, G., Aliaume, C., Nava, M. A. f., Caseillas, C. and Blancheton, J. P. 2004. High-rate algal pond treatment for water reuse in an integrated marine fish recirculating system: effect on water quality and sea bass growth. Aquaculture 235: 331–344.

Duarte P., Meneses, R., Hawkins, A.J.S., Zhu, M., Fang, J., and Grant, J. 2003. Mathematical modelling to assess the carrying capacity for multi-species culture within coastal waters. Ecological Modeling. 168. 109–143

El-Gayar, O.F. 2003. Aquaculture in Egypt and issues for sustainable development. Aquacult. Econ. Manage. 7: 137–154.

Evans, F., and Langdon, C. J. 2000. Co-culture of dulse, *Palmaria mollis* and red abalone, *Haliotis rufescens* under limited flow conditions. Aquaculture 185: 137–158.

Folke, C., Kautsky, N., and Troell, M. 1994. The costs of eutrophication from salmon farming: implications for policy. J. Environ. Manage. 40: 173–182.

Folke, C., and Kautsky, N. 1992. Aquaculture with its environment: Prospects for sustainability. Ocean and Coastal Management, 17: 5–24.

Gowen, R. J., and Bradbury, N.B. 1987. The ecological impact of salmon farming in costal waters: A review. Oceanogr. Mar. Biol. Annu. Rev., 25: 563–575.

Haglund, K., and Pedersen, M. 1993. Outdoor pond cultivation of the subtropical marine red algae *Gracilaria tenuistipitata* in brackish water in Sweden. Growth, nutrient uptake, co-cultivation with rainbow trout and epiphyte control. J. Appl. Phycol., 5: 271–284.

Hargrave, B.T. 2003. Far-filed environmental effects of marine finfish aquaculture. *In*: Fisheries and Oceans Canada, A scientific review of the potential environmental effects of aquaculture in aquatic ecosystems, Can. Tech. Rept. Fish. Aquat. Sci., 2450: 1–49.

Hussenot, J.M. 2003. Emerging effluent management strategies in marine fish-culture farms located in European coastal wetlands. Aquaculture, 226: 113–128.

Jara-Jara, R., Pazos, A.J., Abad, M., Garcia-Martin, L.O., and Sanchez, J.L. 1997. Growth of clam seed (*Ruditapes decussates*) reared in the waste water effluent from a fish farm in Galicia (N. W. Spain). Aquaculture, 158: 247–262.

Jimenez del Rio, M., Ramazanov, Z. and Garcıa-Reina, G. 1996. *Ulva rigida* (Ulvales, Chlorophyta) tank culture as biofilters for dissolved inorganic nitrogen from fishpond effluents. Hydrobiologia 326/327: 61– 66.

Jones, A.B., Dennison, W.C., Preston, N.P. 2001. Integrated treatment of shrimp effluent by sedimentation, oyster filtration and macroalgal absorption: a laboratory scale study. Aquaculture, 193: 155–178.

Katz, T., Genin, A., Porter, C. B., Krost, P., Gordin, H., and Angel, D.L. 1996. Preliminary assessment of grey mullet (Mugil cephalus) as a forager of organically enriched sediments below marine fish farms. Isr. J. Aquac. – Bamidgeh, 48: 47–55.

Krom, M.D., Ellner, S., van Rijn, J., and Neori, A. 1995. Nitrogen and phosphorus cycling and transformations in a prototype ‘‘non-polluting’’ integrated mariculture system, Eilat, Israel. Mar. Ecol., Prog. Ser., 118: 25– 36.

Lefebvre, S., Barille, L., and Clerc, M. 2000. Pacific oyster (*Crassostrea gigas*) feeding responses to a fish-farm effluent. Aquaculture 187 : 185–198.

Lefebvre, S., Probert, I., Lefrancois, C., and Hussenot, J. 2004. Outdoor phytoplankton culture in a marine fish-phytoplankton-bivalve integrated system: combined effects of dilution rate and ambient conditions on growth rate, biomass and nutrient cycling. Aquaculture 240: 211–231.

Martinez, A., and Buschmann, A.H. 1996. Agar yield and quality of *Gracillaria chilensis* (Gigartinales, Rhodophyta) in tank culture using fish effluents. Hydrobiologia 326–327: 341–345.

Miller, I.M. 1989. Commercial culture of the giant read sea urchin, *Strongylocentrotus franciscanus*, in Hawaii. J. Shellfish Res. 8: 414–415.

Nelson, G.S., Glenn, E.P., Conn, J., Moore, D., Walsh, T. and Akutagawa, M. 2001. Cultivation of *Gracilaria parvispora* (Rhodophyta) in shrimp-farm effluent ditches and floating cages in Hawaii: a two-phase polyculture system. Aquaculture 192: 239– 248.

Neori, A., Cohen, I. and Gordin, H. 1991. *Ulva lactuca* biofilters for marine fishpond effluents: II. Growth rate, yield and C:N ratio. Bot. Mar. 34 : 483– 489.

Neori, A., Ellner, S.P., Boyd, C.E. and Krom, M.D. 1993. The integration of seaweed biofilters with intensive fish ponds to improve water quality and recapture nutrients. *In*: Moshiri GA (ed), Constructed wetlands for water quality improvement. Lewis Publishes, Boca Raton, FL, pp 603–607.

Neori, A., Ragg, N.L.C. and Shpigel, M. 1998. The integrated culture of seaweed, abalone, fish and clams in modular intensive land-based systems: II. Performance and nitrogen partitioning within an abalone (*Haliotis tuberculata*) and macroalgae culture system. Aquac. Eng. 17: 215– 239

Neori, A., Chopin, T., Troell, M., Buschmann, A.H., Kraemer, G.P., Halling, C., Shpigel, M. and Yarish, C. 2004. Integrated aquaculture: rationale, evolution and state of the art emphasizing seaweed biofiltration in modern aquaculture. Aquaculture 231: 361–391.

Neori, A., Krom, M., Ellner, S., Boyd, C., Popper, D., Rabinovitch, R., Davison, P., Dvir, O., Zuber, D., Ucko, M., Angel, D. and Gordin, H. 1996. Seaweed biofilters as regulators of water quality in integrated fish-seaweed culture units. Aquaculture 141: 183–199.

Neori, A., Shpigel, M. and Ben-Ezra, D. 2000. A sustainable integrated system for culture of fish, seaweed and abalone. Aquaculture 186: 279–291.

Nunes, J. P., Ferreira, J.G., Gazeau, F., Lencart-Silva, J., Zhang, X. L., Zhu, M.Y., and Fang, J.G. 2003. A model for sustainable management of shellfish polyculture in coastal bays. Aquaculture, 219: 257–277.

Pagand, P., Blancheton, J. P., Lemoalle, J., and Casellas, C. 2000. The use of high rate algal ponds for the treatment of marine effluent from recirculating fish rearing system. Aquac. Res. 31: 729– 736.

Petrell, R.J., and Alie, S.Y. 1996. Integrated cultivation of salmonids and seaweeds in open systems. Hydrobiologia 326/327 : 67– 73.

Phang, S. M., Shaharuddin, S., Noraishah, H., and Sasekumar, A. 1996. Studies on *Gracilaria changii* (Gracilariales, Rhodophyta) from Malaysian mangroves. Hydrobiologia 326/327 :347– 352.

Pullin, R., Rosenthal, H., and Maclean, J. 1993. Environment and Aquaculture in Developing Countries. ICLARM Conference Proceedings 31. ICLARM, Manila.

Rensel, J. 2001. Salmon net pens in Puget Sound. The Advocate, February 2000, 66–69.

Robinson, S.M.C. 2004. Defining the appropriate regulatory and policy framework for the development of integrated Multi-Trophic Aquauclture practices: summary of the workshop and issues for the future. Bull. Aquacul. Assoc. Canada, 104–3: 73–83

Sandifer, P., and Hopkins, S. 1996. Conceptual design of a sustainable pond-based shrimp culture system. Aquacultural Engineering 15: 41–52.

Shpigel, M. and Neori, A. 1996. The integrated culture of seaweed, abalone, fish and clams in modular intensive land-based systems: I. Proportions of size and projected revenues. Aquacultural Engineering 15: 313–326.

Shpigel, M. Neori, A. and Marshal, A. 1996. The suitability of several introduced species of abalone (Gastropda; *Haliotidae*) for land-based culture with pond-grown seaweed in Israel. Isr. J. Aquacult/Bamidgeh 48: 192–200.

Shpigel, M., Neori, A., Popper, D. and Gordin, H. 1993. A proposed model for “environmentally clean” land-based culture of fish, bivalves and seaweeds. Aquaculture 117: 115–128.

Stewart, J. 1998. Sharing the waters: an evaluation of site fallowing, year class separation and distances between sites for fish health purposes on Atlantic salmon farms. Can. Tech. Rept. Fish. Aquat. Sci., 2218.

Stirling, H.P., and Okumus, I. 1995. Growth and production of mussels (*Mytilus edulis* L.) suspended at salmon cages and shellfish farms in two Scottish sea lochs. Aquaculture 134: 193–210.

Subandar, A., Petrell, R. J., and Harrison, P. J. 1993. Laminaria culture for reduction of dissolved inorganic nitrogen in salmon farm effluent. J. Appl. Phycol., 5: 455– 463.

Troell, M., Halling, C., Nilsson, A., Buschmann, A., Kautsky, N., and Kautsky, L. 1997. Integrated marine cultivation of *Gracilaria chilensis* (Gracilariles, Rhodophyta) and salmon cages for reduced environmental impact and increased economic output. Aquaculture, 156: 45–61.

Troell, M., Kautsky, N., and Folke, C. 1999a. Applicability of integrated coastal aquaculture systems. Ocean and Coastal Management, 42: 63–69.

Troell, M., Ronnback, P., Halling, C., and Buschmann, A. 1999b. Ecological engineering in aquaculture: use of seaweeds for removing nutrients from intensive aquaculture. J. Appl. Phycol. 11: 89–97.

Vandermeulen, H., and Gordin, H. 1990. Ammonium uptake using Ulva (Chlorophyta) in intensive fishpond systems: mass culture and treatment of effluent. J. Appl. Phycol. 2: 363–374.

Walker, R.L, Heffernan, P.B, Crenshaw Jr, J.W. and Hoats, J. 1991. The effects of mesh size, stocking density and depth of the growth and survival of pearl net cultured bay scallops, *Argopecten irradians concentricus*, in shrimp ponds in South Carolina, USA., J. Shellfish Res., 10: 465–469

Wang, J.K. 1990. Managing shrimp pond water to reduce discharge problems. Aquacul. Eng. 9: 277–288.

Weatherly, A., and Cogger, B. 1977. Fish culture: problems and prospects. Science, 197: 427–430.

White, H., and Glenn, H. 2005. Environmental impact mitigation and bi-culture: a comparative legal analysis of flexibility within European legal regimes – biofilter deployment. Aquaculture International, 21 p.

Wohlfarth, G., and Schroeder, G. 1979. Use of manure in fish farming – a review. Agricultural Wastes, 1, 279–299.

Table 8.1: Integrated Mariculture Systems – Experimental and Pilot-Scale.

WATER BASED SYSTEMS	REFERENCES
Two Phyla Systems	
Sea cucumber to process fish wastes	Ahlgren (1998)
Abalone-seaweed combination	Benson *et al.* (1986)
Grey mullet in bottom cages underneath commercial sea bream cages	Angel *et al.* (1992), Katz *et al.* (1996)
Cultivation of seaweeds (*Laminaria saccharina*, *Nereocystis luetkeana*, *Gracilaria*, *Porphyra*) with salmon cage aquaculture	Ahn *et al.* (1998); Chopin *et al.* (1999, 2001); Buschmann *et al.* (1994, 1995, 1996, 2001); Chung *et al.* (2002); Troell *et al.* (1997, 1999a, 1999b); Petrell and Alie (1996)
Salmon (*Salmo salar*), scallops (*Patinopectin yessoensis*) and oysters (*Crassostrea gigas*)	Cross (2004)
Three Phyla Systems	
Salmon (Salmo salar), mussels (Mytilus edulis), seaweed (Laminaria saccharina)	Chopin *et al*. (2002, 2003, 2004)
Land Based Systems	
Two Phyla Systems	
Integrated shrimp-oysters	Wang (1990)
Integrated Shrimp-scallops	Walker *et al.*, 1991
Integration of fish culture (sea bream, salmon) with seaweed (*Ulva, Gracilaria, Laminaria)*	Cohen and Neori (1991); Krom *et al.* (1995); Jimenez del Rio *et al.* (1996); Neori *et al.* (1991, 1993, 1996, 2000); Buschmann *et al.* (1994, 1996); Martinez and Buschmann (1996); Haglund and Pedersen (1993); Subandar *et al.* (1993); Pagand *et al.* (2000); Vandermeulen and Gordin (1990)
Integration of abalone and sea urchins	Miller (1989)
Integration of abalone and seaweeds (*Gracilaria, Ulva, Palmaria*)	Neori *et al.* (1998); Evans and Langdon (2000)
Integration of fish (turbot, sea bass, sole) and bivalves (clams, oysters)	Jara-Jara *et al.* (1997); Lefebvre *et al.* (2000)
Integration of shrimp and seaweeds (*Gracilaria, Ulva*)	Danakusumah *et al.* (1991); Nelson *et al.* (2001); Phang *et al.* (1996)
Three Phyla Systems	
Integration of shrimp, oysters and seaweed (*Gracilaria edulis*)	Jones *et al.* (2001)
Integrated shrimp- fish (mullet)-oysters	Sandifer and Hopkins (1996)
Integrated culture of fish (sea bream), bivalves (*Crassostrea gigas, Tapes semidecussatus, Haliotis tuberculata*) and seaweed (*Ulva, Gracilaria)*	Shpigel *et al.* (1993, 1996); ; Neori (1996); Neori *et al.* (2000)
Four Phyla Systems	
Integration of fish-oysters-sea urchins and seaweeds	Chow *et al.* (2001)

Table 8.2: Classification aquaculture by degree of intensity (Hussenot, 2003)

	DESCRIPTION	TYPICAL AREA (PER PRODUCTION UNIT)	PRODUCTIVITY (PER YEAR) STANDING BIOMASS (KG/M3)
Extensive	Traditional extensive culture used for eels (*Anguilla* spp.),grey mullets (*Mugilidae*), sea bass, sea bream	1–100 ha	0.1 t/ha 0.1 – 0.5
Semi intensive	Semi-intensive earthen ponds producing sea bass or sea bream	0.1–1 ha	20–50 t/ha 1.0– 4.0
Intensive	Unit Intensive grow-out of sea bass and sea bream in concrete tanks or ponds covered by greenhouses or inflated structures.	0.01–0.3 ha	200–400 t/ha 5.0–30.0

Table 8.3. Production from a pilot scale integrated multispecies aquaculture site in Israel (Neori *et al.*, 2004).

ORGANISM	POND SIZE RATIO/HA	YIELD (MT Y^{-1})	YIELD (KG M^{-2} Y^{-1})
Seabream	1	265	22
Ulva	3.5	2215	64
Abalone	1.85	185	10
or Sea urchin	2.75	275	10
TOTAL	*6.3*	*450*	*30*

9 Assess and report on the state of knowledge of alternatives to fish for use in formulated feeds for finfish aquaculture (ToR e)

9.1 Introduction

WGEIM 2002 and other ICES groups have previously evaluated the need for alternative sources of fish meal and fish oil for aquaculture feeds and their impact the world supply of reduction fisheries. The sustainability of utilising fish meal and oil based feed products for marine fish farm activities continue to be questioned by environmentalists in the media and clarification is sought continuously. Fish feed manufacturers continue to evaluate alternative sources. The goal of this report is to provide an update on the progress being made in identifying alternatives to fish meal and oil as protein and lipid sources for feed used in finfish aquaculture.

Total finfish and crustacean aquaculture production finfish and crustacean aquaculture production increased from 10.9 to 31.8 million tonnes from 1992 to 2004 (Figure 9.1) and more than double since 1994 (14 100 thousand tonnes; FAO, 2005). Of this farmed salmon and marine-brackishwater reared rainbow trout in 2004 was 1950 thousand tonnes and sea bream and sea bass were 140 thousand tonnes (Figure 9.2; FAO, 2005). Aquaculture production is predicted to increase along with the world population and demand for food. Salmon aquaculture is expected to reach over 2000 thousand tonnes by 2010 (Forster, 2003) and total global aquaculture production is expected to exceed total capture fisheries production by 2015. Growth in salmon and aquaculture production will likely increase the demand and price for fishmeal and fish oil that are ingredients of aquafeed. Also, past and projected growth of the aquaculture industry is putting great pressure on the need for safe food products and sustainable practices.

9.2 Need for Alternative Sources of Fish Meal and Fish Oil

The quantities of landed fish and shellfish from capture fisheries destined for reduction into meals and oils and other non-food purposes has increased over seven-fold from 3000 thousand tonnes in 1950 (representing 16.1% total capture fisheries landings) to 21 370 thousand tonnes in 2003 or 23.4% total capture fisheries landings (FAO, 2005). However, the world supply has been fairly constant since 1976 (Figures 9.3 and 9.4). However, this figure only refers to whole fish destined for reduction, and so excludes other fish scraps and processing wastes. In fact, industry estimates for the total quantity of whole fish and trimmings reduced into meals and oils in 2002 have been given as 33 million tonnes (includes 27 400 thousand tonnes of whole fish caught by dedicated fishing fleets and 5600 thousand tonnes of trimmings and rejects from food fish; FIN, 2004). The supplies of these resources are limited and will not meet the future demands of the industry. It is therefore imperative that more sustainable sources of protein and lipid be found.

Traditional salmonid and marine fish feeds, for instance, are composed mainly of fish oil (lipid) and fish meal (protein) harvested from wild fish stocks. The total estimated amount of fish oil and fish meal used within compound aquafeeds has grown from 234 to 802 thousand tonnes and from 963 to 2936 thousand tonnes from 1994 to 2003, respectively. Carnivorous finfish species consumed 52.8% and 81.9% of the total fishmeal and fish oil used in compound aquafeeds in 2003, with farmed salmon alone consuming 13.9% and 51.0% total fishmeal and fish oil used within aquafeeds, respectively. Clearly, however, if the sector for carnivorous finfish species is to be sustainable in the long-run it must reduce its dependence upon these finite commodities. In the short term this is of most concern for fish oil, and could be partly resolved through the use of plant oils and animal fats as dietary energy sources supplemented with marine fish oils reserved only as dietary providers of essential fatty acids. Apart from the use of fish oils for farmed aquatic animals as a source of dietary energy and essential fatty acids fish oils are also used for human and animal consumption and specific technical applications, such as in the manufacture of non-food products

The apparent higher dependency of marine/brackishwater carnivorous finfish and crustacean species for fishmeal and fish oil is primarily due to their more exacting dietary requirements for high quality animal protein, essential fatty acids and trace minerals (Hardy *et al.* 2001; Pike, 1998). Advances in feed formulation (Larrain *et al.* 2005), feed manufacturing technology (Kearns, 2005), and on-farm feed management (Larrain *et al.* 2005) have all resulted in increased fish growth, reduced fish production costs, and reduced feed conversion ratios (FCRs).

Between 50 and 75% of commercial salmon feeds are currently composed of fishmeal and fish oil and any price increases in these finite commodities will have a significant effect on feed price and farm profitability as salmon feeds and feeding representing between 60 to 70% of total farm production costs. In general, the effect of increasing prices on fishmeal and fish oil use include 1) fishmeal: increased substitution with cheaper dietary protein sources, and increased dietary supplementation within limiting essential nutrients, such as amino acids and trace elements and 2) fish oil: increased substitution with cheaper dietary plant and/or terrestrial animal lipid sources.

Public health concerns have been raised about the potential accumulation of environmental contaminants within farmed salmon from the feeding of aquafeeds containing contaminated fishmeals and fish oils. The mean reported lipid content of farmed Atlantic salmon is currently almost twice that of wild Atlantic salmon (17–19% versus 8–10%. Although the total essential fatty acid (EFA) content of farmed salmon flesh may be higher than that of wild salmon (due to the use of EFA-rich dietary fish oils), by the same token farmed salmon also runs the risk of containing higher levels of environmental contaminants from increased fish oil use. Thus, apart from having almost twice the body burden of contaminants by virtue of their higher carcass lipid content, dietary fish oils may also be contaminated with persistent organic pollutants. Feed manufacturers have used fishmeals and fish oils from other less contaminated regions of the world or purchased more expensive *decontaminated* oils and meals.

Due to the perceived attitudes and opinions of consumers towards food safety and `wholesomeness' or `quality' (including farmed fish), there has been a growing trend in some countries for major salmon producers and/or leading salmon retailers/supermarket chains to set guidelines to feed manufacturers as to what can be used within salmon feeds, including levels of maximum fishmeal and fish oil substitution. For example, the lower levels of fish oil and to a lesser extent fishmeal substitution within salmon feeds in the UK has been in part due to the formulation constraints imposed by leading salmon producers and/or retailers, including what ingredients or levels of substitution are considered acceptable (Huntington, 2004). Concerns raised about the possible transfer of mammalian infectious agents such as Bovine Spongiform Encephalopathy (BSE) and other Transmissible Spongiform Encephalopathies (TSEs) through the use of rendered animal by-product meals within compound animal feeds (including aquafeeds; FAO, 1998, 2001; FIN, 2004; Pearl, 2000; SCAHAW, 2003; SSC, 2003) has led to introduction of stricter feed assurance schemes, including codes of practice concerning fishery products, fishmeal and feed manufacture and the development of improved rendering techniques and safer animal by-product meals (Woodgate, 2004) and that the by-products of farmed finfish should not be fed to farmed finfish. One approach is to reduce fish oil levels (and therefore potential contaminant levels) through dietary substitution with less contaminated vegetable and/or other terrestrial land animal fats and oils. Plant sources of protein and lipid have shown some promise in this regard. However, direct substitution of plant ingredients for fish ingredients is not straight-forward and research to determine suitable levels and combinations of these ingredients to meet the nutrient and energy requirements of the various cultured fish species is required. Furthermore he substitutes must not affect the taste of the product, nutritional value, product safety and fillet processing quality.

Sustainable fisheries concerns have been raised considering the long-term sustainability and ethics of using potentially food-grade fishery resources for animal feeding rather than for direct human consumption (Best, 1996; Goldburg and Naylor, 2005; Tacon, 1997). In particular, in some major fishmeal and salmon producing countries such as Chile there has been small shift toward selling a portion of the fish catch for direct human consumption to African countries rather than for reduction (Wray, 2001; Zaldivar, 2004). In addition to the above, there has been increasing public awareness and concern for the health and management of marine fisheries stocks and ecosystems, and the growing demand for assurance/certification schemes that fishery products are obtained from sustainable sources, including the increasing demand for traceability, labeling and transparency (FIN, 2004; Hole, 2004; Huntington, 2004; Huntington *et al.* 2004; Wessells *et al.*, 2001). The dietary substitution of fishmeal and/or fish oil with less digestible plant and animal protein and lipid sources will result in increased nutrient loading and potential loss in fish growth, survival, palatability and feed efficiency. However, such negative impacts could be greatly reduced by selecting the use of highly digestible ingredient sources, through the use of enzyme treated plant proteins, exogenous dietary feed enzymes, supplemental amino acids, flavouring, and selective breeding of plant strains with desirable qualities.

9.3 Overview of Alternatives to Fish Oil in Aquafeeds

9.3.1 Effects on fatty acid composition and quality of flesh

One of the side-effects of substituting plant oils for fish oils in fish diets is an unintentional alteration in the fatty acid composition of the diet, which in turn affects the fatty acid composition of the fish. Plant oils contain high levels of shorter-chain monounsaturated fatty acids such as 18:1n-9, and polyunsaturated fatty acids (PUFA) such as 18:2n-6 and 18:3n-3, and low levels of saturated fatty acids (SFA) compared to fish oil, and the relative accumulation or depletion of these fatty acids in flesh of fish fed plant oil-based diets has been shown (Glencross *et al.* 2003a; Montero *et al.* 2005). These alterations in nutrition status can have surprising effects on the health of the fish. Jobling and Bendiksen (2003) suggested that Atlantic salmon parr fed plant oils in place of fish oils may be better able to withstand exposure to low temperature as a result of improved membrane fluidity (higher UFA:SFA ratio) while having membrane polar lipids that were less susceptible to oxidative damage (implied from lower unsaturation indices).

Plant oils are devoid of the n-3 highly unsaturated fatty acids (HUFA), 22:6n-3 (docosahexaenoic acid) and 20:5n-3 (eicosapentaenoic acid), and the n-6 HUFA, 20:4n-6 (arachidonic acid), that are present in fish oil and have physiological importance for carnivorous fish. Studies have illustrated depletion of 20:5n-3 and 22:6n-3 in the flesh when plant oils are substituted for dietary fish oil in Atlantic salmon (Bell *et al.*, 2005; Berntssen *et al.*, 2005) and seabass (Montero *et al.*, 2005). However, some species of marine fish such as red seabream (Glencross *et al.*, 2003a) and gilthead seabream (Izquierdo *et al.* 2003) tend to selectively retain 20:5n-3 and 22:6n-3 in their tissues when fed diets deficient in these fatty acids, illustrating that these fatty acids are essential for these species.

In addition to effects on specific fatty acids, some studies have shown that fatty acid deficient diets tend to decrease overall n-3 HUFA levels in species such as red seabream (Glencross *et al.*, 2003a), gilthead seabream (Izquierdo *et al.*, 2003) and seabass (Montero *et al.*, 2005). This may lead to cumulative negative effects on the n-3/n-6 fatty acid ratios, which are suggested to be as important as the absolute amounts of the specific fatty acids (Berntssen *et al.*, 2005; Glencross *et al.* 2003a).

The n-3 HUFA, particularly 20:5n-3 and 22:6n-3, are considered highly beneficial for human health (Bell *et al.*, 2005; Izquierdo *et al.*, 2003). Negative changes in the 20:5n-3, 22:6n-3 and overall n-3 levels of fish fed plant oils may have implications for human nutrition. Furthermore, alterations in the total SFA and PUFA levels can affect the flesh characteristics and consumer acceptance. Both issues may influence the market values of these fish, so strategies to counter these issues need to be considered.

The possibility of using “finishing” diets containing fish oil to reverse the changes in fatty acid composition caused by feeding with plant oils has been examined. Bell *et al.* (2005) successfully used a finishing diet prior to harvest to restore Atlantic salmon flesh 20:5n-3 and 22:6n-3 concentrations to 80% of the levels in salmon fed fish oil diets throughout their growth cycle. Glencross *et al.* (2003b) showed a reversal of the fatty acid composition of fish previously fed plant oil, towards one more consistent with that of a fish fed fish oils. Montero *et al.* (2005) restored 22:6n-3 levels, but not 20:5n-3 levels in sea bass. For fish previously fed plant oil diets, the time taken to achieve fatty acid composition recovery varies not only depending on what the original oil source was, but also for which fatty acid the composition recovery is being evaluated, and depending on fish species and size (Glencross *et al*., 2003b).

In some instances, finishing diets may not be truly necessary. Despite a reduction of n-3 HUFAs, cardiovascular patients consuming Atlantic salmon farmed on plant oil diets still displayed beneficial effects of this consumption with regard to the disease development (Berntssen *et al.* 2005). Izquierdo *et al.* (2003) reported that the HUFA content of gilthead seabream and seabass fed plant oils as partial replacement of anchovy oil were not much lower than the corresponding levels if the fish oil had been a typical Scandinavian fish oil, and that the levels of n-3 HUFA were still high compared to levels found in other animal protein sources.

In terms of fillet quality, fillets of gilthead seabream and seabass fed plant oils as partial replacement of fish oils were well accepted by trained judges when assessed cooked (Izquierdo *et al*., 2003). In another study, sensory assessment, by an Australian taste panel, of red seabream fed the fish oil reference, or the 100% replacement by canola or soybean diets showed a preference in order canola oil > soybean oil > fish oil fed fish (Glencross *et al.*, 2003a). However, both of these studies were conducted with young fish and further flesh quality tests on market-sized fish are needed.

9.3.2 Effects on reproduction, growth and health of fish

Atlantic salmon can be cultured over the whole production cycle using diets in which 100% of the added fish oil has been replaced with a blend of linseed oil and rapeseed oil without detrimental effects to growth performance (Bell *et al.*, 2005). Juvenile red seabream were fed diets in which 100% of the fish oil was replaced with refined canola or soybean oil without significant effect on growth (Glencross *et al.*, 2003a). It is likely that only partial replacement of fish oil in diets for some marine fish will be achieved. Provided there is a minimum content of essential fatty acids in the diet, it is possible to replace up to 60% of the fish oil with a mixture of plant oils (soybean, rapeseed and linseed oil) in diets for gilthead seabream and seabass without compromising growth, feed intake, feed utilization or fish health (Izquierdo *et al.*, 2003; Montero *et al.*, 2003). However, when a single plant oil is used over a long period of feeding, detrimental effects may occur in terms of immunosuppression or stress resistance (Montero *et al.* 2003). Waagbo (1997) reviewed the impact of nutritional factors on the immune system in fish. Poly-unsaturated fatty acids of the ω3 and ω6 series are precursors of eicosanoids which are signal substances. They regulate physiological reaction on both non specific (e.g. phagocyte production) and specific immunity (e.g. antibody production by B and T cells). Kiron *et al.* (1995) demonstrated that the non-specific immunity against *Aeromonas salmonicida* (measured by the number of killed bacteria) was increased in rainbow trout fed with ω3 rich fish oil compared to ω6 rich plant oil. Simultaneously the antibody production in rainbow trout vaccinated against *Aeromonas* was increased in fish fed fish oil rather than plant oil. In the same way, increased resistance to vibriosis was observed in Atlantic salmon (Waagbo, 1997).

Caballero *et al.* (2004) observed that liver morphology of sea bream was altered by diet containing a single vegetable oil, but that the alterations were reversible by re-feeding the fish with a diet containing fish oil. In another study, Welker and Congleton (2003) concluded that diets made with plant oils high in n-6 fatty acids may have beneficial effects on the parr-smolt transformation and marine adaptation of salmonids but they could potentially have negative effects on fish health, especially by enhancing sensitivity to stressful stimuli. McKenzie et al (1999) and Agnisola (1996) demonstrated in Sturgeon a decrease in the cardiac performance of fish with reduced levels of ω3 in the diet and consequently their susceptibility to low oxygen levels. Further research is needed to examine the relationship between dietary fatty acids and stress response, disease pathology and oxidative stress of fish.

Atlantic salmon broodstock were cultured for one year prior to spawning with diets in which 50% of the fish oil was substituted with rapeseed oil (Rennie *et al.* 2005). While the fatty acid profiles of both the eggs and the fry were altered by the change in the broodstock diets, there were no significant differences in egg number, weight or proximate composition or fry weight at first feeding. Although the fatty acid profiles of the eggs and fry were changed by the diets, the changes did not affect the fertilisation, eyeing, hatching or the first feeding survival (Rennie *et al.* 2005). The authors observed that 20:5n-3 and 22:6n-3 (important for egg and fry development) seemed to be selectively incorporated into the eggs in the same ratio as in eggs from broodstock fed a fish oil based diet. Ratios of n-3/n-6 were likewise adjusted by Atlantic salmon. The 22:6n-3/20:5n-3 ratio has been linked to fry susceptibility to external stressors (Rennie *et al.*, 2005), which was not broached in this study. Although the dietary manipulations did not affect egg and fry survival and growth, possible effects on later fry development should be examined.

9.3.3 Effects on concentrations of dioxins and dioxin-like PCBs (DLPCB)

Dioxins and dioxin-like PCBs are highly persistent, fat soluble environmental pollutants that are readily biomagnified in the food chain (Berntssen *et al.* 2005). Concentrations of PCDD/F and DLPCBs in fish oil vary greatly depending on factors such as seasonal variation, fish species, age and geographical origin (Berntssen *et al.* 2005). When fish oil is used as a primary lipid ingredient in aquaculture feeds, the contaminants present in the fish oil are subsequently accumulated in the cultured species destined for human consumption (Bell *et al.* 2005; Berntssen *et al.* 2005). Although it's been suggested that salmon cultured on diets based on fish meal and oil attain flesh dioxin concentrations that are <14% of the European Commission limit of 2.25 ng TEQ (toxic equivalent values)/kg (Bell *et al.*, 2005), consumer concern over the potential adverse health effects on humans has generated interest in developing novel ways to reduce exposure to these contaminants in our food.

The levels of dioxins and dioxin-like PCBs in plant oils are considerably lower than in fish oils (Jacobs *et al.*, 1998; 2002). Therefore, substitution of plant oils in place of fish oils as the primary lipid ingredient in feeds for aquaculture fish has been explored with promising results. Substituting a plant oil blend (55% rapeseed oil, 30% palm oil, and 15% linseed oil) for capelin oil in a diet fed to Atlantic salmon over a period of 22 months reduced whole salmon concentrations of dioxins and DLPCBs by eight and twelve times, respectively (Berntssen *et al.*, 2005). In a second study, herring oil replacement with plant oils (linseed oil: rapeseed oil 1:1) in Atlantic salmon diets from first feeding to harvest (115 weeks) has been shown to decrease the concentrations of dioxins and DLPCBs in the feed and in the flesh of farmed salmon by 64–75% (Bell *et al.*, 2005). The same study proceeded to feed a finishing diet containing high fish oil (35% capelin) for 24 weeks to restore the HUFA levels depleted during feeding with the plant oil diet. After 24 weeks, levels of dioxin and dioxin-like PCBs in fish previously fed plant oil diets significantly increased, but the concentrations were still significantly lower (47–60% lower) than in fish fed fish oil diets throughout (Bell *et al.* 2005).

In diets where the main source of contaminants is substituted, additional sources might become more important (Berntssen *et al.*, 2005). Fish meals, soybean meals, and mineral premixes, for instance, may all be contaminated with dioxins. Therefore, other means to reduce transfer of contaminants from the fish feed to the cultured fish should be implemented. Feed conversion ratio is positively correlated with dioxin and DLPCB levels in fish. Growth rate is negatively correlated with PCDD/F and DLPCB levels in fish. Thus maintaining an efficient feed conversion ratio and a high growth rate will keep the level of persistent organic pollutants as low as possible in farmed fish (Berntssen *et al.*, 2005).

One of the obvious short-comings of research to date is that studies on substitution of fish meal and fish oil have been mutually exclusive. Many of the promising results from substitution of fish oils with plant oils have been due, in part, to inclusion of high levels of fish meal in the same diet. The fish meal provides some of the essential fatty acids that would have normally been provided directly by the fish oil. Although other novel sources of essential fatty acids are available, they must become more economical before they can sustain the needs of the aquaculture industry. Nevertheless, great advances in reducing, if not eliminating, the reliance upon wild fisheries resources for aquaculture feed ingredients are being made.

9.4 Overview of Alternate Protein Sources for Marine Fish

Fish meals are commonly utilized as protein ingredients in prepared feeds for many species of fish and livestock as a result of its protein density, unique balance of amino acids, high digestibility and effect on palatability of the complete feed. However due to increased demand as a result of global increases in the production volume of cultured finfish (FAO 2004) and major fluctuations in the catches of species used for the production of fish meals, the price of fish meals has been unstable and generally on the rise. From 1990–2000, average annual fishmeal prices were reported to range between $325 to 650 USD (Hardy 2000). Recent reports from September 2005 to the end of March 2006 saw an increase in the average cost of Peruvian FAQ 65 from $625 to $810 USD, (Bacon, 2005; 2006).

To address supply limitations and the unstable cost of fish meal, many studies have been conducted with alternative protein sources to replace fish meals in diets for most commercially grown species of fish (Hardy *et al.*, 2001; Hardy, 2004). Performance of fishes when fed these alternative protein feeds is dependent upon the natural diet of the species fed, composition and nutrient availability of the test ingredient, possible presence and level of anti-nutritional factors, and effect on the palatability and pellet stability of the prepared feed. Suitable alternative protein ingredients must not compromise fish health, growth or feed efficiency and must be economically competitive and available in large quantities (Hardy, 2004).

The majority of research that has been conducted with alternative protein sources for marine fish has focused on the use of proteins from plant origin (Table 9.1; Carter and Hauler, 2000; Hardy, 2004; Krishnankutty, 2005). However these sources are often deficient in one or more essential amino acids and may contain anti-nutritional factors which can reduce palatability, protein utilization and growth (Francis *et al.*, 2001).

In response, these protein sources frequently require further processing in order to concentrate the protein, remove anti-nutritional factors (Oliva-Teles *et al.*, 1994) or diets using these ingredients may be supplemented with crystalline amino acids (Goff and Gatlin III, 2004; Choi *et al.*, 2004), or with enzymes that increase nutrient availability (Yoo *et al.*, 2003). In addition, diets that contain these high protein sources are also often supplemented with feeds such as krill or shrimp to enhance the palatability of the prepared feed (Lee and Meyers, 1997). Although these practices may increase nutrient availability and acceptance of the feed, they may also increase the cost of using the chosen alternative protein feedstuff. Evaluation of alternative protein feeds from animal origin has also been conducted with many species of fish, but to a lesser degree then with alternative plant proteins (Tibbetts *et al.*, 2004; Riis, 2004). Animal meals are typically higher in protein and lower in carbohydrate concentration compared to those from plant origin (NRC 1993). However, the future use and availability of ingredients from bovine and avian origin may decrease in response to current global concerns over bovine spongiform encephalitis (BSE), foot and mouth disease, and the recent avian flu outbreak in south Asia.

9.5 Recommendations

WGEIM have identified a need to conduct further studies on:

1) Consumer acceptance of fish fed alternative oils and proteins (taste panels);

2) Effects of alternative feed on disease resistance (general fish welfare issues) under culture conditions;

3) Heritability and selective breeding potential for ability to use all feeds;

4) Development of feed additives to enhance nutrition of products (e.g. omega 3);

5) Use of fish offal and by products in aquafeeds;

6) Accumulation of contaminants and therapeutants from feed under culture conditions.

The main recommendation from WGEIM is that during the intersession, WGEIM lead a review and evaluation of recent advances on alternative sources of lipid and protein to fish oil and fish meal in aquafeed. It is proposed that a WGEIM review a draft manuscript at the 2007 meeting that is to be submitted for publication in a peer reviewed scientific journal.

9.6 References

Agnisola, C., McKenzie, D. J., Taylor, E. W., Bolis C. L., and Tota, B. 1996. Cardiac performance in relation to oxygen supply varies with dietary lipid composition in sturgeon. Am. J. Physiol. 40, 417-425.

Bacon, W. 2005. Hammersmith marketing Limited. Weekly Report. http://www.hammersmith.blogspot.com

Bacon, W. 2006. Hammersmith marketing Limited. Weekly Report. http://www.hammersmith.blogspot.com

Bell, J.G., McGhee, F., Dick J.R., and Tocher, D.R. 2005. Dioxin and dioxin-like polychlorinated biphenyls (PCBs) in scottish farmed salmon (salmo salar): Effects of replacement of dietary marine fish oil with vegetable oils. Aquaculture 243(1–4): 305–314.

Berntssen, M. H. G., Lundebye, A., and Torstensen, B.E. 2005. Reducing the levels of dioxins and dioxin-like PCBs in farmed atlantic salmon by substitution of fish oil with vegetable oil in the feed. Aquaculture Nutrition 11(3): 219–231.

Best, P. 1996. Should fish meal be cowed by the ethics of environmentalism? Feed International, 17(8): 4.

Caballero, M.J, Izquierdo, M.S., Kjorsvik, E., Fernandez, A.J., and Rosenlund, G. 2004. Histological alterations in the liver of sea bream, sparus aurata L., caused by short- or long-term feeding with vegetable oils. recovery of normal morphology after feeding fish oil as the sole lipid source. Journal of Fish Diseases 27(9): 531–541.

Carter, C.G., and Hauler, R.C. 2000. Fish meal replacement by plant meals in extruded feeds for Atlantic salmon, *Salmo salar* L. Aquaculture, 185: 299–311.

Catacutan, M.R., and Pagador, G.E. 2004. Partial replacement of fish meal by defatted soyabean meal in formulated diets for the mangrove red snapper, *Lutjanus argentimaculatus* (Forsskal 1775). Aquacult. Res., 35: 299–306.

Choi, S., Wang, X., Park, G., Lim, S., Kim, K., Bai, S.C. and Shin, I. 2004. Dietary dehulled soybean meal as a replacement for fishmeal in fingerling and growing olive flounder *Paralichthys olivaceus* (Temminck et Schlegel). Aquaculture Research, 35: 410–418.

Fishmeal Information Network. 2004. Fishmeal facts and figures – November 2004 update (http://www.gafta.com/fin/finfacts.html)

Food and Agriculture Organization of the United Nations (FAO). 1998. Animal feeding and food safety. Food and Nutrition paper 69. Rome, FAO. 48p. (http://www.fao.org/WAICENT/FAOINFO/ECONOMIC/ESN/ animal/ animapdf/ contents.htm).

Food and Agriculture Organization of the United Nations (FAO). 2001. Good aquaculture feed manufacturing practice. FAO Technical Guidelines for Responsible Fisheries. No.5.1. Rome, FAO. 2001. 50p.

Food and Agriculture Organization of the United Nations (FAO). 2004. The state of world fisheries and aquaculture. http://www.fao.org

Food and Agriculture Organisation of the Untited Nations (FAO). 2005. FAO Fisheries Department, Fishery Information, Data and Statistics Unit. Fishstat Plus: Universal software for fishery statistical time series.

Forster, J. 2003. Salmon farming: aquaculture for the mass market. The Global Aquaculture Advocate, 6(1): 52–54.

Fournier, V., Huelvan, C. and Desbruyeres, E. 2004. Incorporation of a mixture of plant feedstuffs for fish meal in diets of juvenile turbot. Aquaculture, 236: 451–465.

Francis F., Makkar H.P.S., and Becker, K. 2001. Antinutritional factors present in plant derived alternative fish feed ingredients and their effects in fish. Aquaculture, 199: 197–227.

Glencross, B., Hawkins, W., and Curnow, J. 2003a. Evaluation of canola oils as alternative lipid resources in diets for juvenile red seabream, pagrus auratus. Aquaculture Nutrition 9(5): 305–315.

Glencross, B., Hawkins, W. and Curnow, J. 2003b. Restoration of the fatty acid composition of red seabream (pagrus auratus) using a fish oil finishing diet after grow-out on plant oil based diets. Aquaculture Nutrition, 9(6): 409–418.

Goldburg, R., and Naylor, R. 2005. Future escapes, fishing, and fish farming. Front. Ecol. Environm., 2005; 3(1): 21–28.

Goff, J. B. and Gatlin, III, D.M. 2004. Evaluation of different sulfur amino acid compounds in the diet of red drum, *Sciaenops ocellatus*, and sparing value of Cystine for Methionine. Aquaculture, 241: 465–477.

Hardy, R.W. 2000. New developments in aquatic feed ingredients, and potential of enzyme supplements. Ed. by L.E. Cruz-Suárez, D. Recque-Marie, M. Tapia-Salazar, M.A. Olvera-Novoa, and R. Civera-Cerecedo. *In* Avances en Nutrición Acuicola V. Memorias del V. Simposium Internacional de Nutrición Acuicola. 19–22 Noviembre, 2000. Merida, Yucatán, Mexico. pp. 216–226.

Hardy, R.W. 2004. New Developments in Alternative Proteins. Aquaculture Magazine, 30(3): 56–59.

Hardy, R.W., Higgs, D.A., Lall, S.P. and Tacon, A.G.J. 2001. Alternative dietary protein and lipid sources for sustainable production of salmonids. Fisken og Havet, No.8, 53pp. (31 May 2001: Institute of Marine Research, Bergen, Norway; http://www.imr.no)

Hole, R. 2004. Food safety: a question of assessment and communication. Aquafeed International, 7(1): 40–43.

Huntington, T.C. 2004. Feeding the fish: sustainable fish feed and Scottish aquaculture. Report to the Joint Marine Programme (Scottish Wildlife Trust and WWF Scotland) and RSPB Scotland. Poseidon Aquatic Resource Management Ltd, Lymington, Hampshire, UK. August 2004.(http://www.wwf.org.uk/ filelibrary/ pdf/feedingthefish.pdf)

Huntington, T., Frid, C., Banks, R., Scott, C., and Paramor, O. 2004. Assessment of the sustainability of industrial fisheries producing fish meal and fish oil. Report to the Royal Society for the Protection of Birds (RSPB). Poseidon Aquatic Resource Management Ltd, Lymington, Hampshire, UK. June 2004. (http://www.rspb. org. uk/Images/fishmeal_tcm5-58613.pdf)

Izquierdo, M., Obach, A., Arantzamendi, L., Montero, D., Robaina, L. and Rosenlund, G. 2003. Dietary lipid sources for seabream and seabass: Growth performance, tissue composition and flesh quality. Aquaculture Nutrition, 9(6): 397–407.

Jacobs, M.N., Santillo, D., Johnston, P.A., Wyatt, C.L. and French, M.C. 1998. Organochlorine residues in fish oil dietary suppliements: Comparison with industrial grade oils. Chemosphere 37: 1709–1721.

Jacobs, M.N., Ferrario, J. and Byrne C. 2002. Investigation of polychlorinated dibenzo-*p*-dioxins, dibenzo-*p*-furans and selected coplanar biphenyls in Scottish farmed Atlantic salmon (*Salmo salar*). Chemosphere 47: 183–191.

Jobling, M. and Bendiksen, E.A. 2003. Dietary lipids and temperature interact to influence tissue fatty acid compositions of atlantic salmon, salmo salar L., parr. Aquaculture Research 34(15): 1423–1441.

Kearns, J.P. 2005. Current production methods for the production of salmon feeds. International Aquafeed, 8(1): 28–33.

Kiron, V., Fukuda, H., Takeuchi, T., and Watanabe, T. 1995. Essential fatty acid nutrition and defence mechanisms in rainbow trout *Oncorhynchus mykiss*. Comp. Biochem. Physiol. 111A, 361–367.

Krishnankutty, N. 2005. Plant proteins in fish feed. Current Science, 88: 865–867.

Lee, P.G. and Meyers, S.P. 1997. Chemoattraction and feeding stimulation. Ed. by L.R D'Abramo, D.E. Conklin and D.M. Akiyama. *In* Crustacean Nutrition, Advances in World Aquaculture, World Aquaculture Society, 6: 292–352.

Larraín, C., Leyton, P., and Almendras, F. 2005. Aquafeed country profile – Chile and salmon farming. International Aquafeed, 8: 22–27.

McKenzie, D.J., Piraccini, G., Agnisola, C., Steffensen, J. F., Bronzi, P., Bolis, C.L., Tota, B. and Taylor, E.W. 1999. The influence of dietary fatty acid composition on the respiratory and cardiovascular physiology of Adriatic sturgeon (*Acipenser naccarii*) : a review. J. Applied Ichthyology 15: 265–269.

Montero, D., Kalinowski, T., Obach, A., Robaina, L., Tort, L., Caballero, M. J. and Izquierdo, M. S. 2003. Vegetable lipid sources for gilthead seabream (sparus aurata): Effects on fish health. Aquaculture 225(1–4): 353–370.

Montero, D., Robaina, L., Caballero, M. J., Gines, R., and Izquierdo M. S. 2005. Growth, feed utilization and flesh quality of european sea bass (dicentrarchus labrax) fed diets containing vegetable oils: A time-course study on the effect of a re-feeding period with a 100% fish oil diet. Aquaculture 248(1–4): 121–134.

National Research Council. 1993. Nutrient Requirements of Fish. National Academy Press, Washington, D.C. pp. 114.

Pearl, G. 2000. Rendering role in biosecurity and emerging diseases. Render, 29(2): 46–54.

Pike, I.H. 1998. Future supplies of fish meal and fish oil: quality requirements for aquaculture. International Aquafeed Directory and Buyers Guide 1998. pp. 39–49.

Rennie, S., Huntingford, F.A., Loeland, A.L. and Rimbach, M. 2005. Long term partial replacement of dietary fish oil with rapeseed oil; effects on egg quality of Atlantic salmon salmo salar. Aquaculture, 248: 135–146.

Riis, T. 2004. Spray-dried porcine blood meal – a valuble protein source for fish feeds. Fish Farmer, 27: 28–29.

Scientific Committee on Animal Health and Animal Welfare (SCAHAW). 2003. The use of fish by-products in aquaculture. Report of the Scientific Committee on Animal Health and Animal Welfare, Adopted 26 February 2003. European Commission, (http://europa.eu.int/comm/food/fs/sc/scah/out87_en.pdf)

Scientific Steering Committee (SSC). 2003. Opinion on the feeding of wild fishmeal to farmed fish and recycling of fish with regard to the risk of TSE. Report of the 71 Scientific Steering Committee, Adopted 6–7 March 2003. European Commission (http://europa.eu.int/comm/food/fs/sc/ssc/out320_en.pdf)

Sitjà-Bobadilla, A., Pena-Llopis, S., Gómez-Requeni, Medale, F., Kaushik, S. and Pérez-Sánchez, J. 2005. Effect of fish meal replacement by plant protein sources on non-specific defence mechanisms and oxidative stress in gilthead seabream (*Sparus aurata*). Aquaculture, 249: 387–400.

Tacon, A.G.J. 1997. Feeding tomorrow's fish: keys for sustainability. In: Feeding Tomorrow's Fish (A.G.J. Tacon and B. Basurco, editors). Cahiers Options Mediterraneennes, Vol. 22. Zaragoza, Spain. pp. 11–33.

Tibbetts, S. M., Lall, S. P. and Milley, J. E. 2004. Apparent digestibility of common feed ingredients by juvenile haddock, *Melanogrammus aeglefinus* L. Aquaculture Research, 35: 643–651.

Waagbo, R. 1997. The impact of nutritional factors on the immune system in Atlantic salmon, *Salmo salar* L.: A review. Aquac. Fish Manag. 25, 175–197.

Welker, T.L and Congleton, J.L. 2003. Relationship between dietary lipid source, oxidative stress, and the physiological response to stress in sub-yearling chinook salmon (oncorhynchus tshawytscha). Fish Physiology and Biochemistry, 29(3): 225–235.

Wessells, C.R., Cochrane, K., Deere, C., Wallis, P. and Willmann, R. 2001. Product certification and ecolabelling for fisheries sustainability. FAO Fisheries Technical Paper No. 422. Rome, FAO. 2001. 83 p.

Woodgate, S. L. 2004. Terrestrial animal by-products: the new European Union regulation on animal by-products: an update. International Aquafeed, 7(1): 29–32.

Wray, T. 2001. Making the most of jack mackerel. Seafood International, July 2001, pp. 39–43.

Yoo, G.Y., Wang, X, Choi, S., Han, K., Kang, J-C., and Bai, S. C. 2003. Dietary microbial phytase increased the phosphorus digestibility in juvenile Korean rockfish *Sebastes schlegeli* fed diets containing soybean meal. Aquaculture, 243: 315–322.

Zaldivar, M.J. 2004. Review of the commercial situation of fishmeal and fish oil and the new requirements imposed on them. Paper presented at Tecnica Aqua Sur 2004 (Chile), March 2004 (http://www.iffo.org.uk/tech/TecnAqua.htm)

Table 9.1: Published usage of alternative protein sources in diets of various species.

Species	Alternative Protein Source	Results	Reference
Salmon (*Salmo salar*)	Soybean meal (20 or 27% replacement) and DL-methionine (0.3 or 0.5%)	No difference in weight gain or feed efficiency compared to control diet	Carter and Hauler, 2000
Gilthead seabream (*Sparus aurata*)	Mixture of corn gluten meal, wheat gluten, extruded wheat and indispensable amino acids (up to 100% replacement)	Up to 75%: No adverse effects on growth; each increase in dietary plant protein caused decreased dry matter intake and increased feed efficiency; 100%: feed efficiency and growth were significantly reduced; feed intake declined	Sitjà-Bobadilla *et al.*, 2005
Olive flounder (*Paralichthys olivaceus*)	Dehulled soybean meal (DSBM); DSBM + lysine + methionine (with and without attractant)	DSBM: up to 20% replacement DSBM + lys + meth: up to 30% replacement	Choi *et al.*, 2004
Red drum (*Sciaenops ocellatus*)	Sulphur amino acid compounds and cysteine (in place of methionine)	Sulphur amino acid compounds were effectively used; cysteine could replace ~50% of dietary methionine	Goff and Gatlin III, 2004
Turbot (*Psetta maxima*)	Mixture of lupin, corn gluten meal, wheat gluten meal and amino acids	Decreased feed intake, growth, nitrogen retention and hepato-somatic index	Fournier *et al.*, 2004
Mangrove red snapper (*Lutjanus argentimaculatus*)	Defatted soybean meal (50% replacement)	No difference in growth or feed efficiency; protein efficiency was slightly lower than control	Catacutan and Pagador, 2004
Haddock (*Melanogrammus aeglefinus*)	Herring meal; Soybean meal; Corn gluten meal; Canola meal; Crab meal; Shrimp meal	Digestible protein coefficients: Herring meal, 95.0%; Soybean meal, 92.3%; Corn gluten meal, 92.3%; Canola meal, 83.0%; Crab meal, 82.0%; Shrimp meal, 73.5%	Tibbetts *et al.*, 2004

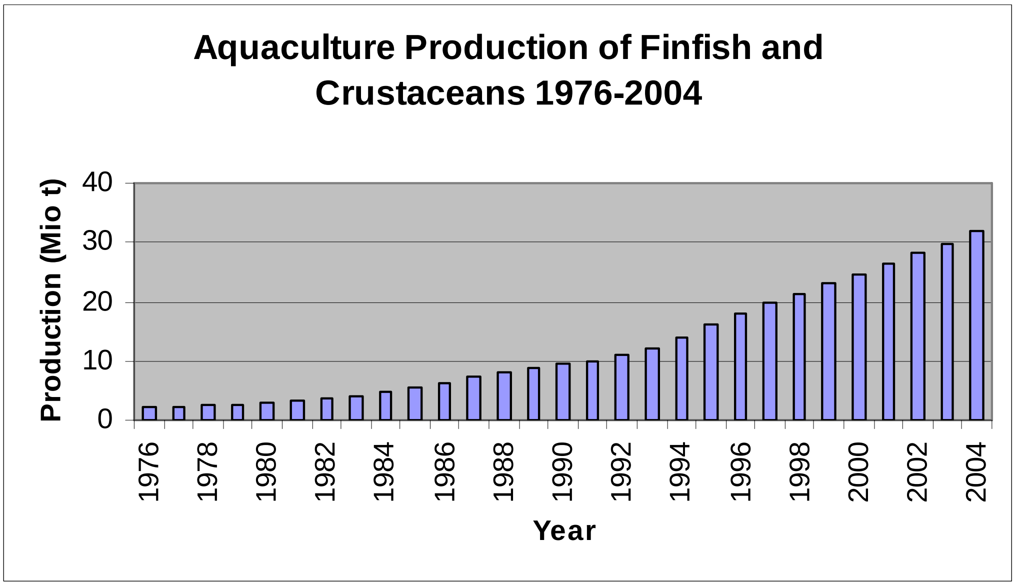


Figure 9.1.

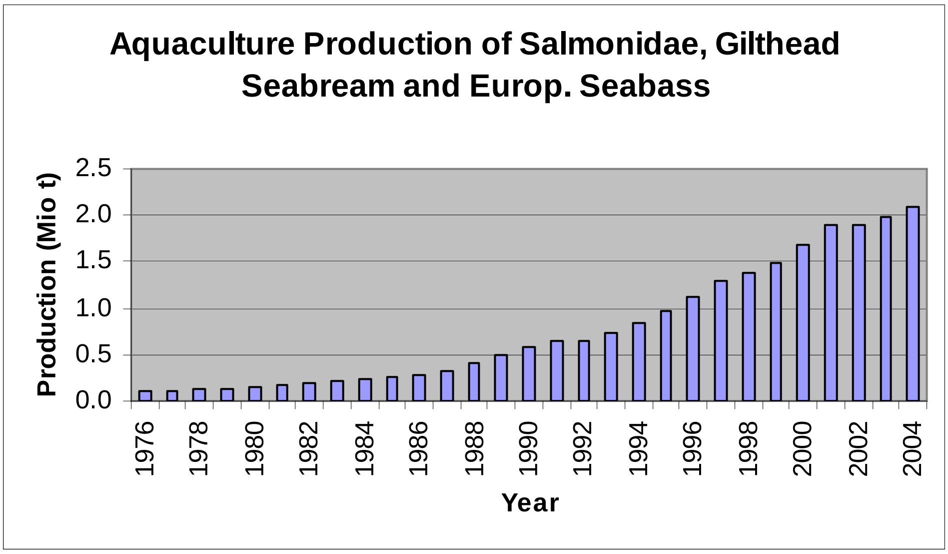


Figure 9.2.

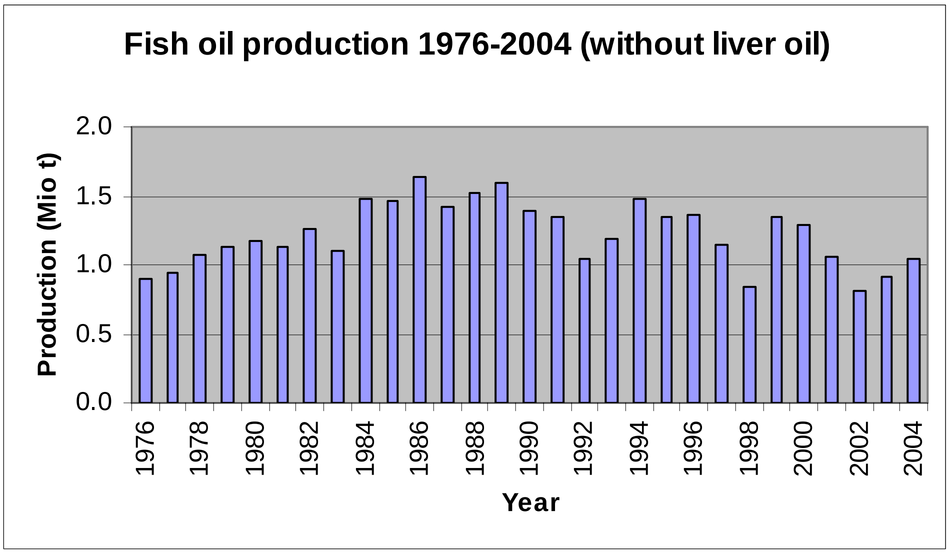


Figure 9.3.

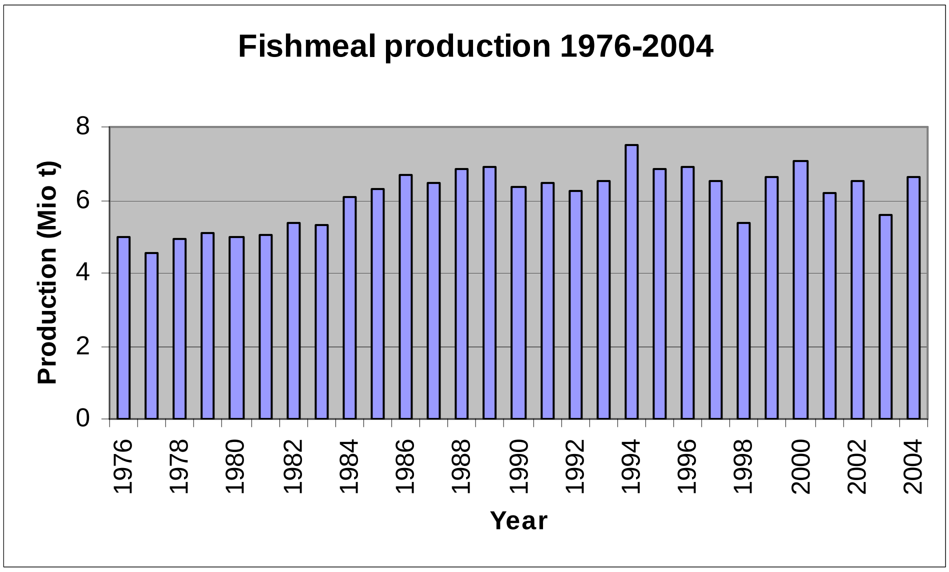


Figure 9.4.

10 Investigate fouling hazards associated with the physical structures used in Mariculture and assess their potential for the introduction of invasive/nuisance species into the local environment (ToR f)

10.1 Introduction

The topic of aquatic exotic species is regularly discussed by the ICES Working Group on Introduction and Transfer of Marine Organisms (WGITMO) and the Study Group on Ballast and Other Ship Vectors (SGBOSV). The WGITMO deals mainly with intentional introductions for e.g. aquaculture purposes, and works to reduce unintentional introductions of exotic and deleterious species through a risk assessment process and quarantine recommendations. Meanwhile, the SGBOSV focuses mostly on unintentional species introductions with e.g. ballast water and hull fouling. To date, the interaction of mariculture with exotic species and more specifically unintentional species introductions has received limited attention. This is despite the fact that exotic species are having significant impacts on the aquaculture industry worldwide and more particularly for the shellfish aquaculture industry. For clarity, the terminology used in this text will be based on that proposed by Binggeli (1994) as follows:

- Native (indigenous): species naturally occurring in an area since prehistorical time;
- Exotic (alien, introduced): deliberate or accidental release of a species into an area in which it has not occurred in historical times;
- Invasive (naturalised, neophyte, adventive): the establishment of self-regenerating, usually expanding, populations of an introduced species in a free-living state in the wild;
- Nuisance (pest): any species, either native or introduced, that interferes with the objectives or requirements of people.

The aim of the section is to examine the potential importance of bivalve culture in the promotion and transfer of exotic aquatic species as well as the importance of these exotic species to bivalve culture and the environment. Specifically, we focus on exotic species with an emphasis on those that become invasive and nuisance. Management implications and mitigation strategies are also addressed. It should be noted that the majority of the existing literature addresses the issues as they relate to oyster culture, probably because this appears to be the single greatest vector for all types of introductions (planned or otherwise) in bivalve aquaculture (Carlton, 1992b). There is little published information about other bivalve species with respect to their role as vectors for exotic species. The following discussion is thus largely based on oyster-oriented literature but has been expanded where possible to include other taxa.

10.2 Background

There has been much discussion and debate about the importance of aquaculture as a vector for the introduction and spread of exotic species (Carlton 1992a, 1992b; Naylor *et al.* 2001; Streftaris *et al.* 2005). There are two broad classes of introductions that may result from bivalve aquaculture. First, there is the establishment and spread of non-endemic species that have been intentionally introduced into an area for aquaculture purposes, the “target” species. Classic examples of this include the establishment of the Pacific oyster (*Crassostrea gigas*) on the Pacific coast of North America (Ruesink *et al.* 2005) and in various countries throughout Europe (Grizel and Heral, 1991; Reise, 1998; Drinkwaard, 1999) and of the Mediterranean mussel (*Mytilus galloprovincialis*) in South Africa (Branch and Steffani, 2004). We do not consider this aspect further but a recent review may be found in Landry *et al.* (in press). Second, there is the establishment and spread of species that are associated with the introduced bivalves (Carlton, 1989; 1999). These species may include both "hitchhiking" species - animals and plants that grow associated with the bivalves, and diseases and parasites that may cause outbreaks in the same or other species (Barber, 1996). This acts at two spatial scales: at an inter-regional or international scale with respect to the initial introduction of hitchhiking species and also at a regional scale, where the transfer of stock among sites may be very important to the spread of established exotic species locally (Bourque *et al.*, 2003a). The provision of novel habitat by the species being cultured may also allow for the establishment or amplification of exotic species that may be introduced through other vectors or of native species that thrive in the novel habitat (Carver *et al.* 2003; Rodriguez, 2005; Locke *et al.*, submitted).

Bivalves have been grown and introduced for culture throughout the world for hundreds of years (Mann, 1983; Chew, 1990). The first records of bivalve transfers date back to 1714 in Europe for the European flat oysters (*Ostrea edulis*) (Wolff and Reise, 2002). Thereafter, transfers of this same species became routine, and attempts at introducing other species were also initiated: American oysters (*Crassostrea virginica*) (1870s), Portuguese oysters (*Crassostrea angulata* – actually a strain of *C. gigas* likely introduced with ship fouling) (imported from Portugal to France in latter half of 19^{th} century), and Pacific oysters (*Crassostrea gigas*) (1903) (Wolff and Reise, 2002). Although there have been sporadic efforts to introduce non-indigenous bivalves for aquaculture purposes to the northwest coast of North America, especially *O. edulis*, from 1949-1961 in the eastern United States and 1957-1959 in eastern Canada (Chew 1990; Carlton 1992a; Shatkin *et al.* 1997; Vercaemer *et al.* 2003; Ruesink *et al.* 2005), the bivalve aquaculture industry there is largely based on endemic species. Consequently, introductions associated with bivalve aquaculture are relatively scarce in the general area (Carlton 1999).

Introductions of the *C. gigas*, and to a lesser extent *C. virginica* and other oyster species, outside of their native range for aquaculture have been suggested to be one of the greatest single modes of introduction of exotic species world-wide (Wasson *et al.*, 2001; Ruesink *et al.* 2005). For example, transfer of organisms with bivalves has been suggested to be the most important source of exotic species in northern Europe (Minchin 1996; Streftaris *et al.* 2005) and among the most important vectors elsewhere in that continent (Ribera Siguan 2003; Streftaris *et al.* 2005; Gollasch 2006). In the northeast Pacific, some authors suggest that oyster introductions have even been the major source of introduction of exotic molluscs (Carlton 1992a) and invertebrates in general (Wonham and Carlton, 2005), historically contributing at least as many of the exotic species in that area as has international shipping.

10.3 Aquaculture structures and bivalves as habitat

All types of bivalve culture have the potential to increase the 3D structure of the physical environment. This is due to the physical structure of the equipment used (buoys, lines, trays, bags, rafts, netting, etc.) as well as the cultured bivalves themselves. All bivalves that are currently cultured to any major extent may be considered as engineering (Jones *et al.* 1994 – species that modulate the availability of resources to other species by causing physical state changes in biotic or abiotic materials and thus modify, maintain or create habitats) or "foundation" (Dayton 1972 – inferring that the species is relatively large, dominant in terms of biomass or abundance, and has a positive effect on community inhabitants via its presence and not its actions) species.

Bruno and Bertness (2001) suggest that bivalves, as foundation species, have a number of ways in which they facilitate or otherwise influence benthic communities, including general habitat creation, providing refuge from predation, reducing physical and physiological stress, enhancing settlement and recruitment, and increasing food supply. These are described in detail in McKindsey *et al.* (in press) and will not be discussed further here. The physical structure associated with bivalve aquaculture affords both foraging and refuge opportunities for different species, either directly or else indirectly by colonizing species (Bartol and Mann, 1997; O'Beirn *et al.* 2000; Shumway *et al.* 2003). In addition, many studies have noted great abundances and biomass of organisms living associated with bivalves in suspension and in bottom and off-bottom culture (Tenore and González, 1976; Castel *et al.*, 1989; Khalaman, 2001; Luckenbach, 2001; LeBlanc *et al.*, 2002; Dealteris *et al.*, 2004; O'Beirn *et al.*, 2004). Indeed, bivalve farmers are constantly searching for ways to reduce the abundance of fouling organisms on their bivalves and equipment in order to increase their growth, to facilitate field maintenance and processing (see reviews in LeBlanc *et al.*, 2003; Ross *et al.*, 2004) and increase marketability. For example, Tenore and González (1976) found over 100 species on mature mussel socks in Spain and up to 430 g dw of fouling organisms m^{-1} mussel sock. Guenther *et al.* (in press) reported over 30 species recruiting to pearl oyster valves over a period of 16 weeks. O'Beirn *et al.* (2004) studied the assemblages associated with floating bag culture for *C. virginica* in Virginia and found a total of 45 (mean = 29) taxa on 50 oysters in each bag with annelids and molluscs being the most abundant groups. Dealteris *et al.* (2004) also studied *C. virginica* in floating bag culture but in Rhode Island and found the average number and abundance of associated taxa > 5 mm in size to vary between about 15 and 23 and about 1000 and 2500 individuals $cage^{-1}$, respectively.

The organisms growing on bivalves in suspension may in turn attract other organisms, such as fish and more mobile macroinvertebrates. For example, Carbines (1993) found a positive correlation between algal cover and the number of young spotties (*Notolabrus celidotus*) on mussel lines and noted that the fish also associated with encrusting invertebrates and algae on mooring ropes and particularly mooring blocks in New Zealand mussel farms. Similarly, Lawrence *et al.* (2000) reported the presence of a dinoflagellate growing associated with an algae that grew on mussels in culture in Nova Scotia, eastern Canada.

The effect of adding suprabenthic bivalves to benthic soft-bottom communities that lack such engineering species are pretty much as would be expected from the ecological literature. The presence of bivalves on the seabed increases complexity and can provide refuge from predatory organisms (Thiel and Dernedde, 1994) as well as substratum for epifaunal settlement (Albrecht and Reise, 1994; Kröncke, 1996). Consequently, the addition of exotic oysters to soft-sediment areas may lead to an increase in the abundance of most groups of organisms. We know of no similar studies that have been done on sea-ranched scallops or other suprabenthic bivalve species.

In summary, with respect to the infaunal and epifaunal organisms associated with bivalve culture, the installation and associated species function more or less like a normal benthic hard-bottom community, what McKindsey *et al.* (in press) refer to as a "pelagic hard-bottom benthic community". In soft-bottom bivalve aquaculture, the addition of epibenthic species will typically increase the abundance of most groups of organisms.

10.4 Exotic hitchhikers

The majority of aquatic marine exotic species are benthic and, more specifically, hard-bottom associated species (Gollasch, 2006). Further, the majority of aquatic exotic species are also associated with coastal areas, particularly estuaries and lagoons (Reise *et al.* 2006), and exotics in general are commonly exploit novel and/or disturbed habitats (Ruiz *et al.* 2000). Thus, as bivalve culture sites are commonly sited in areas that have this suite of conditions and/or help create these conditions, bivalve culture may serve to focus exotic species. What's more, because of the great diversity of associated species in bivalve culture, relaying or stock transfers among regions may be an important vector for the introduction and/or spread of exotic species.

There are a number of ways in which exotic species may be introduced into a new environment when bivalves are transferred for aquaculture. Exotics may be present within the bivalves, on the bivalves, in water or on equipment (such as ropes, socking material, cages) transferred with the bivalves, within sediment transferred, within empty shells of dead individuals, or associated with other hitchhiking species. The importance of the different modes of transfer varies with culture type and stage of both the bivalves and the exotic species (Buhle *et al.*, 2005).

There are three major classes of exotic hitchhikers that are of concern with respect to bivalve aquaculture and introductions and transfers:

1) exotic macrospecies including algae and animals;
2) exotic phytoplankton (toxic and otherwise); and
3) exotic disease species.

Each of these may influence the bivalve species being cultured or the surrounding ecosystem. At the time of writing this, there are a number of worrisome invasive species associated with bivalve culture in Canada. These include the suite of invasive ascidians that is plaguing the mussel industry in Prince Edward Island (PEI) (the solitary tunicates *Styela clava* and *Ciona intestinalis* and the colonial species *Botrylloides violaceus* and *Botryllus schlosseri*, known commonly as the clubbed, vase, violet, and golden star tunicates, respectively) and another species that is fouling bivalve culture sites in British Columbia, *Didemnum* sp., which has also been reported off the coast of Nova Scotia as well as the northeast coast of the United States (Kott, 2002; 2004). Other macroscopic species of concern include the green crab (*Carcinus maenas*) the skeleton shrimp (*Caprella mutica*) and the green alga (*Codium fragile* ssp. *tomentosoides*). There are also a number of worrisome disease organisms, mostly associated with oyster culture, including MSX disease (*Haplosporidium nelsoni*) (Burreson and Ford, 2004) and *Bonamia ostreae* in European flat oysters (Bower and McGladdery, 2003). Although these exotic species may not necessarily have been introduced initially with bivalve aquaculture, they are primarily associated with this activity (given the commercial implications) and thus bivalve culture may play an important role in their secondary spread.

10.4.1 Exotic macrospecies

Exotic macrospecies of invertebrates and algae may impact the culture bivalves with which they are associated and the environment in general once introduced. Once again, it must be pointed out that the literature dealing with this subject is quite limited and much of the available information is only available in the “grey literature”, including reports and conference proceedings.

The most obvious and immediate impact of exotic macrospecies on aquaculture is the fouling of the bivalves that are being cultured and the equipment (lines, cages, buoys, *etc*.) used to do this. The tunicates and others listed above are classic examples from Canada (and elsewhere for some species). At least some of these tunicates have been suggested to have been introduced and/or spread through bivalve aquaculture (Lambert and Lambert, 1998). Fouling organisms such as tunicates likely compete directly with bivalves in culture for food and space, potentially reducing growth rates and increasing stress and mortality (Lesser *et al.*, 1992; Bourque *et al.*, 2003a; Carver *et al.*, 2003). That being said, different filter feeders, both bivalves and the fouling tunicates, feed on different types of food such that competition between mussels and tunicates is species-specific. For example, *S. clava* and *M. edulis* feed on similar sized food (Bourque *et al.*, 2003b) whereas *C. intestinalis and M. edulis* feed on different sizes of food (Lesser *et al.*, 1992). Under the latter scenario, Lesser *et al.* (1992) suggest that the mussel and fouling species are not likely strong competitors for food and that the latter should not influence mussel yield unless food is a limiting factor. However, this does not take into account the simple physical barrier that the tunicates create that may reduce the availability of food to the mussels underneath. The presence of such large filter-feeders may also filter out large quantities of food and potentially change the local carrying capacity of a given area for bivalve culture. The presence of such abundant and large macrospecies in association with or adjacent to bivalve culture operations also greatly impacts general operations within the culture sites and for processing as all the lines, *etc*. used are all much heavier and the tunicates can impede the efficiency of the processing equipment.

The green crab *Carcinus maenas* is also of concern for bivalve aquaculture in northwestern North America. It is a voracious predator and has a preference for bivalves (Behrens Yamada, 2001). On the Atlantic coast of North America, it has been blamed, in part, for the decline of the soft-shell clam population (Glude, 1955). The soft-shell clam aquaculture industry in the Gulf of St. Lawrence is thus under threat from this species. Floyd and Williams (2004) suggest that farmers will have to protect the young clams until they reach a size at which they are no longer vulnerable to the crab. Although green crabs are not known for their climbing ability, it is also common on mussel lines and scallop cages in areas where it is widespread (McKindsey, personal observations) so it may also have an impact on these as well. The invasive skeleton shrimp (*C. mutica*) seems to be widespread along the northeastern Atlantic coast and in the southern Gulf of St. Lawrence and is thought by some farmers to be responsible for a decline in mussel spat-fall in Canada and elsewhere (Cook *et al.*, 2004). Once again, little or no research has addressed these points. A number of exotic species introduced with bivalve culture are also having significant effects of that same industry in Europe. For example, the eastern North American oyster drill *Urosalpinx cinerea* was first introduced to Europe with shipments of *C. virginica* and is now a common nuisance species in bivalve culture there (Pratt, 1974). The slipper shell *Crepidula fornicate* was also introduced to Europe with *C. virginica* and is now also considered a pest on commercial oyster beds in the United Kingdom and elsewhere (Barton and Heard, 2005).

The transfer of bivalves is also a well-known and important vector for macroalgae introductions (Critchley and Dijkema, 1984; Rueness, 1989; Neushul *et al.*, 1992; Wallentinus, 2002; Ribera Siguan, 2003; Mineur *et al.* 2004). In eastern Canada, the green algae *Codium fragile* ssp. *tomentosoides* (hereafter, *Codium*) is one such species. *Codium* is thought to have originally been transferred to north-eastern North America via oyster culture (Malinowski and Ramus, 1973) and to Atlantic Canada with shellfish from the United States (Campbell, 1997). Currently, in Canada it is found in Nova Scotia, New Brunswick and PEI (Hubbard and Garbary, 2002) and in the Magdalen Islands, Quebec (Simard *et al.* in press) and throughout the northeast United States and often in association with bivalve aquaculture. Bivalves on which *Codium* grows are often dislodged because of the increased drag they impart unto the animals (Trowbridge, 1998). The alga has also been shown to smother blue mussels and bay oysters in eastern North America by attaching to the valves of the animals and keeping them shut (Fralick (1970), cited in Trowbridge 1998) and may also render afflicted bivalves more susceptible to predation (Ramus, 1971). Afflicted bivalves may also have lower meat yields (Galtsoff, 1964, cited in Trowbridge 1998) and presumably growth rates. Thus, it is a concern for bivalve culture operations.

Hanisak (1979) suggests that *Codium* may be nitrogen-limited for a good part of the growing season. Bivalves increase the concentration of nitrogen-based compounds in the water directly through excretion and indirectly through mineralization of settled pseudofaeces and faeces in the surrounding sediments (e.g., Prins *et al.*, 1998), this being particularly true in aquaculture situation (Dame, 1993). Thus, it is reasonable to predict that association with bivalves in culture may increase the growth and productivity of macroalgae in some sort of cascading effect. This has also been suggested for the endemic brown algae *Pilayella littoralis* that was growing on mussels and equipment in Nova Scotia, eastern Canada. It was shown to grow quicker on mussel lines than on control mussel lines with dead mussels (Lawrence *et al.*, 2000).

In Europe, exotic macroalgae are also commonly associated with bivalve culture sites. In fact, Wallentinus (2002) suggests that bivalve stock transfer is the single greatest vector for exotic macroalgae in Europe. For example, Verlaque (2001) has reported 45 species of exotic macroalge from the Thau lagoon in southern France, many of which were suggested to have been introduced with or are associated with the intensive bivalve (mostly oyster, some mussel) culture there. Of these, all but two have a likely Pacific origin and Verlaque (2001) suggests that most of these likely arrived with imported *C. gigas*. Similar claims were also made by Maggs and Stegenga (1999), who suggested that most species of exotic red algae in the North Sea were also introduced via oyster culture. Critchley and Dijkema (1984) also suggest that one of the most invasive species of algae in Europe at this time, *Sargassum muticum*, is believed to have been introduced with *C. gigas* and although believed to be spread secondarily by floating thalli it has also been observed growing on *O. edulis*. A similar case has been made for the introduced algae *Undaria pinnatifida* in the Mediterranean with both shipping and subsequent oyster cultivation believed to have aided in spreading the alga from the Thau Lagoon where it was originally introduced to Europe with *C. gigas* spat (see Curiel *et al.*, 2001).

All the species associated with bivalve culture discussed above may also have an influence on the surrounding ecosystem. However, the importance of the different exotics on the surrounding ecosystem is not well studied. Further, when they have been studied, they have usually been studied as a part of the surrounding ecosystem, not as an influence on it (for tunicates, see Osman and Whitlatch 1995b, 1995a, 1995c; Stachowicz *et al.* 1999; Stachowicz *et al.* 2002; Osman and Whitlatch 2004), although there have been some exceptions (Whitlatch *et al.*, 1995; Bullard *et al.*, 2005; Getchis, 2005). The influence of these species on the ecosystems in may vary among locations. Observations in numerous embayments in PEI have shown *B. violaceus* overgrowing eelgrass (*Zostera marina*) and various algae in the areas where it is prevalent in mussel farms and it has also been observed (A. Locke, personal communications) on a large proportion of the rock crabs (*Cancer irroratus*) in one bay that was examined in 2005. The influence of hitchhikers on the functioning of the ecosystem may be considerable. For example, Cloern (1982) suggests that, together, three exotic bivalve species (*Tapes philippinarum*, *Gemma gemma*, and *Musculista senhousia*) that arrived with oyster introductions (Carlton 1992a) may filter the entire volume of water of South San Francisco Bay within one day. Similarily, the slipper shell (*Crepidula fornicata*), originally introduced into England with *C. virginica*, has had great impacts on some benthic communities in Europe, particularly in France (see review by Goulletquer *et al.* 2002), where it has displaced important commercial bivalves, such as the great scallop (*Pecten maximus*) in some areas (Chauvaud *et al.* 2003), but has had little effect in others (De Montaudouin *et al.* 2001).

10.4.2 Exotic toxic and nuisance phytoplankton

Although historically thought to be largely associated with introduction from ballast water (Simard and Hardy, 2004), the importance of shellfish introductions to the introduction and spread of phytoplankton that cause harmful algal blooms and other detrimental ecosystem effects is now being recognized (O'Mahony, 1993; Kaiser and Beadman, 2002). There have been a number of experimental studies that have shown that phytoplankton may be transported via the transfer and introduction of bivalves for aquaculture. Although any stage may be transferred, the concern may be greatest for the resting stages (spores and cysts) as these are the most robust.

Toxic and other nuisance phytoplankton may obviously be transferred with water or as cysts or other resting stages in sediments in bivalve transfers but they may also be transferred on the external surfaces of bivalves (Minchin, 1996). In one study, O'Mahony (1993) identified 67 species of phytoplankton associated with oysters transferred from France to Ireland. As was suggested above for macroalgae, there may be some feedback whereby excretion from mussels in culture stimulates the growth of associated phytoplankton. Following a DSP outbreak, Levasseur *et al.* (2003) studied the abundance of the dinoflagellate *Prorocentrum lima*, the presumptive causative species for the observed DSP associated with mussel socks in the Magdalen Islands, Quebec, eastern Canada. They found this species and a further previously unobserved congener, *P. mexicanum*, associated with the epibionts growing on the socks and in the guts of the mussels. Both these studies show that *P. littoralis* may indeed live in association with mussels in culture and thus may be transferred along with mussels during stock transfers. For example, Lawrence *et al.* (2000) studied the relationship between macroalgae and mussel farming and found the toxic (responsible for diarrhetic shellfish poisoning - DSP) dinoflagellate *Prorocentrum lima* growing associated with the *P. littoralis* growing on mussels and equipment in Nova Scotia.

A number of studies have shown that phytoplankton may also be transferred within bivalves with stock transfers (Bricelj and Shumway 1998). Laing and Gollasch (2002) discuss how the nuisance diatom *Coscinodiscus wailesii* may have arrived in Europe with bivalve importations, possibly having been transported within the gut or pseudofaeces of oysters in the form of resting cells. This has also been suggested as a possible vector for the exotic toxic dinflagellate *Alexandrium catenella*, which is now found in the Thau Lagoon, France (Lilly *et al.*, 2002; Penna *et al.*, 2005). Penna *et al.* (2005) further suggest that even if the transfer of bivalves in aquaculture is not the initial vector for toxic phytoplankton, it may be for the secondary spread of a species. Tsujino (2002) found abundant viable cysts of the toxic dinoflagellate *Alexandrium* spp. in bivalves faecal pellets in Japan, suggesting that this genus may also be transferred with bivalves for aquaculture purposes. This was further supported by work by Bricelj *et al.* (1993) who showed that the faeces of *M. edulis* can contain viable *Alexandrium fundyense* cells and Hallegraeff (1993) has reported resistant resting stages from the digestive tracts of bivalves. Upon dissection of numerous mussels following an outbreak of paralytic shellfish poisoning (PSP), the potentially toxin-producing dinoflagellates *Gonyaulax excavata* (=*Alexandrium tamarense*) and *Prorocentrum minimum* were found on the gills and in the digestive tract of mussels from areas where the suspect mussels originated (Langeland *et al.* 1984). Scarratt *et al.* (1993) did an experiment to determine the potential of *A. tamarense* being transferred with scallop (*P. magellanicus*) and mussel (*M. edulis*) spat. They showed that live cells were released from the bivalves after spending six hours under simulated transfer conditions. Subsequent work has shown how these and other species of phytoplankton may all pass through a variety of bivalve species and remain viable (Laabir and Gentien, 1999; Bauder and Cembella, 2000; Harper *et al.*, 2002; Springer *et al.*, 2002; Hégaret *et al.*, 2006), highlighting the possibility of introducing toxic or otherwise harmful phytoplankton with bivalve transfers.

10.4.3 Exotic parasites and diseases

Diseases in many species of bivalves in culture and in fisheries are well known throughout the world (Harvell *et al.*, 1999, see also the special issue on bivalve diseases in Aquatic Living Resources 17(4) 2004). In fact, as pointed out by Figueras (2004: 395), “bivalve diseases are one of the critical bottle necks causing important and recurrent losses in bivalve culture.” Thus, with respect to diseases of oysters, both Farley (1992) and Ruesink *et al.* (2005) suggest that most mass mortalities have resulted from the transfer of infectious stock. Indeed, it has been suggested that one of the more infamous bivalve diseases in Canadian history, the outbreak of Malpeque Bay disease in oysters in PEI in 1915, resulted from a transfer of *C. virginica* stock from New England (Barber, 1996). Similarly, MSX disease (*Haplosporidium nelsoni*) is thought to have been introduced to the east coast of North America and elsewhere via transfer of infected *C. gigas* stock (Burreson *et al.*, 2000). In Europe, stocks of *C. angulata* declined after the introduction of an iridovirus with contaminated *C. gigas* stock from Japan and British Columbia (see Sindermann, 1984). Elston *et al.* (1986) and Grizel *et al.* (1988) report that *O. edulis* stocks in Europe have similarly been impacted by the inadvertent introduction of the protistan parasite *Bonamia ostreae* with contaminated *O. edulis* juveniles from California. However, many diseases have only recently been described, are cryptic and may not become expressed once an introduction has taken place (Minchin, 1996). In general, species of concern fall into one of 4 main taxa: viruses, bacteria, protozoans, and higher invertebrates. Reviews of the major species may be found in Bower *et al.* (1994) and Bower and McGladdery (2003). Specific reviews for pathogens of oysters and their effects may be found in Shatkin *et al.* (1997), Anonymous (2004), and Ruesink *et al.* (2005).

10.5 Management issues

From the above, it is clear that the introduction and transfer of bivalves for aquaculture purposes is a major source of introduction of exotic species. Although not extensively studied, it is also clear that such introductions may have profound effects on bivalve culture itself and on the receiving ecosystems. It is also clear that once established, exotic species are rarely eliminated from their new habitat (Mack *et al.*, 2000). Thus, exotic species must be checked before they arrive in a new area. Appropriate governance must be established to ensure that risks of introductions are minimized. This is being addressed by the European Union with the publication, in April 2006, of a proposal for council regulation concerning the use of alien and locally absent species in aquaculture. The goal of this regulation is to ensure a full risk assessment is carried out for proposed introduction on aquaculture species, incorporating an assessment on the introduction of non-target species (See ToR B above).

10.5.1 Risk assessment

The first line of defence lays in completing effective risk assessment for any proposed stock transfers (Rosenfield, 1992; Minchin, 1996; Minchin and Rosenthal, 2002; Wolff and Reise, 2002; Anonymous, 2004; Forrest *et al.*, 2004; Ruesink *et al.*, 2005). At this time, most approaches internationally are, for the most part, voluntary and based loosely on the *ICES Code of Practice on the Introductions and Transfers of Marine Organisms* (ICES 1995). In short, the *Code* gives a flowchart to follow to ensure that the chances of introductions are minimized. The following chart is summarized from Ruesink *et al.* (2005) and ICES (2005) and emphasizes the need for five main steps:

1) An extensive understanding of the functioning of the receiving ecosystem (predator-prey interactions, competition, diseases, environmental responses, *etc*) and of the basic requirements of the exotic species that may potentially be introduced with stock transfers. Use this information for Steps 2–3, below.

2) Determine the probability of (i) colonization and establishment of any potential exotic species in the target area and (ii) the potential for them to spread.

3) Estimate the impact of the introduction of any potential exotic species on the receiving ecosystem, including trophic interactions, habitat transformations, and interactions with native species of concern (threatened or declining).

4) Establish quarantine and disinfection protocols to help prevent the introduction of undesirable hitchhikers, possibly with the release of only proven uncontaminated progeny into the environment, and the development of a contingency plan to withdraw the species should this become necessary.

It should be highlighted that the complete information required in Step 1 is rarely available. That being said, the identification of crucial knowledge gaps in this step is important in guiding future research effort in this area. It seems that the utilisation of such assessments can help curb the influx of exotic species in a given area. Prior to 1960, the ecological implications of large-scale introductions of exotic bivalves were largely ignored; transfers, *etc*. occurred without much foresight (Wolff and Reise, 2002). Since then, many codes of practice have been implemented with respect to shipping, bivalve transfers, *etc*., and there has been a concomitant decrease in the rate of exotic species introductions, at least in Europe (Streftaris *et al.*, 2005). Transfers are thus less important to the introduction of novel species today but are still important on a regional scale both within Europe (Wolff and Reise, 2002) and eastern Canada (Bourque *et al.*, 2003a). In contrast, when such logic is not followed unwanted introductions may occur. A good example of this concerns *Mytilicola orientalis*, a parasitic copepod from Japan that occurs in the lower intestine of oysters and mussels. Britain and

Ireland were initially free of the parasite because of historic quarantines for *C. gigas*. However, the block on introduction of half grown oysters was deemed a block to an EU free trade directive (91/67/EC). Consequently, half-grown oysters were transferred from France to Ireland which led to the introduction of this parasite and others and a variety of other exotic species (Minchin *et al.*, 1993; O'Mahony, 1993; Minchin, 1996; see also Minchin and Rosenthal, 2002, for other impacts of the EU directive).

10.5.2 Caveats with respect to spread and predictions of ecological effects

Predictions of the risk of spread of exotic species with bivalve aquaculture and of ecological effects are only as good as the information that is available to predict them. The requirements of most associated species are not well known and thus their role in any new environment will be hard to predict. Further, novel interactions within a new environment may also limit the accuracy of predictions based solely on information from elsewhere.

To use a recurring example, the suite of tunicates creating problems in PEI seems to have become a fairly unassuming part of the ecosystem in the central part of the New England states according to the ecological literature (see above). However, it appears that their influence in bivalve culture sites in the northeastern United States (Bullard *et al.*, 2005; Getchis 2005) and PEI is not so benign. This may be because many northeastern United States (Altieri and Witman, 2006) and PEI (Meeuwig *et al.*, 1998) embayments are hyper-eutrophic because of catchment basin land-use patterns. There has also been the suggestion that another invasive species, the green crab, may be facilitating their time there (Locke *et al.*, 2005; Locke *et al.*, submitted). A number of authors have suggested a link between disturbance in the form of eutrophication and the susceptibility of a system to invasion by exotic species (Ruiz *et al.*, 1999; Ruiz *et al.*, 2000) Indeed, invasive species have been found to out-compete native species or fill vacant niches (see Herbold and Moyle, 1986) under such conditions. Examples may be found for macrophytes (Bertness *et al.*, 2002), algae (Wikström and Kautsky, 2004), phytoplankton (Smayda and Reynolds, 2001) and invertebrates (Currie *et al.*, 2000). These factors interact in a myriad of ways to modify ecosystem processes and communities. However, most of these interactions are only theoretical and are little studied (Cloern, 2001). Whatever the cause, the fact that exotic tunicate species become nuisance species in some bivalve culture areas but not in other parts of the ecosystem underlies the point that prediction of impacts may not be made simply by comparing similar situations.

10.5.3 Quarantine, disinfection and other protocols to limit risk

One of the first lines of defence to limit the introduction of exotic species with aquaculture practices should be to establish quarantine and/or disinfection protocols. The first choice for introductions should be to use hatchery-raised and tested stock grown in “clean” areas (Minchin and Rosenthal, 2002). However, this is not always feasible in day to day operations of bivalve culture sites as stock is often relayed among sites at a regional scale. Thus, treatments must be done to limit the risk of transferring hitchhikers along with the stock and/or limit their spread in the environment. As pointed out by Buhle *et al.* (2005), very different methods may be appropriate for different life stages and cost-effectiveness studies may minimize the cost of an overall management strategy.

Using once again the suite of tunicates described above for PEI as an example, a number of treatments have been evaluated around the world, including dipping the mussel lines and equipment used in the culture operations in or spraying on acetic and other acids, brine or lime solutions or fresh water or else using high-pressure sprays, drying, heat, *etc*. (Boothroyd *et al.*, 2002; Anonymous, 2003; Bourque *et al.*, 2003a; Carver *et al.*, 2003; Forrest *et al.*, 2004; Mineur *et al.*, 2004; Thompson and MacNair, 2004; MacDonald *et al.*, 2005; Swan *et al.*, 2005). To date, different producers have employed different management strategies with lesser or greater success. For example, Mineur (2004) examined the efficacy of using pressure washing to clean oysters in an experiment that simulated “normal” operational culture conditions. After washing, the oysters were then incubated for 40 days under laboratory conditions with a clean water source, following which time about 20 species of algae were observed to be growing on the oyster shells, including a few exotic species found only in that culture site so far. Minchin and Rosenthal (2002) discuss how a shipment of *C. gigas* from Japan to France led to the introduction of a number of species into Europe. This was despite the fact that the oysters were subjected to a brine dip upon arriving in France in order to kill the organisms attached to the shell. Minchin and Rosenthal (2002) temper this observation by suggesting that the invaders may have been within the mantle cavities or tissues of live oysters or within the shells of dead ones. However, Shatkin (1997) outlines how similar transfers from Japan and British Columbia to France that were treated with freshwater baths and inspected led to the establishment of a number of species, including barnacles and algae that were stuck to the outside of the oysters. In short, disinfection of bivalves for external hitchhikers is not always effective. Further, the influence of these treatments on the environment is unknown, even though some of the products being used are know to be harmful to a variety of organisms at low concentrations. Although the use of biocontrol measures has been discussed for some time (see Lafferty and Kuris, 1996), to our knowledge, few attempts of this have been tried.

The use of dips, *etc*., does not address the problem of introducing organisms that live within living bivalves or the shells of dead ones and thus most parasites, bacteria, viruses, and protozoan diseases as well as some phytoplankton will not be addressed using these methods (Minchin, 1996). The alternative here is to use depuration so that the target bivalves can clear themselves of the organisms of concern. Although long-used to purge bivalves of toxins associated with, among various factors, toxic phytoplankton and for coliforms and other noxious human-associated microbes (Otwell *et al.*, 1991; Sekiguchi *et al.*, 2001; Blanco *et al.*, 2002; Lee and Younger, 2002), such an approach has also been shown possible for toxic phytoplankton themselves (Scarratt *et al.*, 1993; Dijkema, 1995, cited in Kaiser and Beadman 2002), although efficacy is both bivalve- and phytoplankton species-dependent (Hégaret *et al.*, 2006). Similarly, recent work by Bushek *et al.* (2004) has also shown that depuration or quarantine of shucked oyster shells prior to being used as oyster cultch is important to limit the potential spread of the protozoan parasite *Perkinsus marinus* among regions. Depuration will not however work for organisms that are not released by bivalves over time. This includes many parasites, bacteria, and other bivalve-related pathogens. In these instances, quarantine and growth of F_1 individuals for introduction is prescribed (Minchin and Rosenthal, 2002). This approach is also however ineffective for vertically transmitted pathogens. Barber (1996) gives an example of how a protozoan parasite, *Perkinsus karlssoni*, persisted for 10 generations in quarantined *A. irradians* populations. Further, any monitoring to see if stock is “clean” is only as good as the test used for monitoring (Carnegie *et al.*, 2003) and hitherto unknown species that are only expressed once in a new environment cannot be screened for (Minchin, 1996).

The efficacy of the above protocols to limit risk is obviously a function of how well any guidelines are followed. As pointed out by Minchin and Rosenthal (2002), unauthorized transfers and introductions of bivalves is a serious issue that poses a risk to future bivalve production and ecosystem integrity. They (Minchin and Rosenthal, 2002) give an international (United States to Ireland) example but the same issues exist at much more regional scales where bivalves are transported among sites for grow-out or relaying (Wasson *et al*., 2001).

10.6 Knowledge gaps and recommended research

- Preliminary risk analyses, as outlined in Section 5a, should be done to identify knowledge gaps with respect to exotic species in bivalve culture.
- Directed research should be used to address these knowledge gaps prior to the introduction of bivalves into a system for aquaculture.
- Obtain baseline information on the receiving environment (physical and biological) to make predictions with respect to exotics and to evaluate and understand their influence. Concomitantly, the tolerance of culture species and potential hitchhikers to physical stresses might be elucidated.
- Predict the ability of exotics to establish and spread in the receiving environment. More information is needed with respect to the relative importance of natural (currents, dispersion rates, etc.) and anthropogenic (stock transfers, processing, hull fouling, etc.) spread of exotic species.
- Predict the impact of exotic species on receiving ecosystems, including interactions with local species, habitat modifications, energy flow, etc.
- More information is needed on the requirements and influence of hitchhiking species in the environment. This is particularly true for a number of currently problematic species (e.g., tunicates).
- More information is needed with respect to the natural history of most exotic species.
- Remedial measures need be developed to mitigate impacts and minimize spread.
- Research is needed to understand the cumulative impacts of exotic species and other stressors in the environment (e.g., eutrophication, climate change, fishing activities, contamination, etc.).
- Can one (or a group) of exotics be representative of others (i.e. surrogates) in terms of predicted response in an area?

10.7 Recommendations

The WGEIM recommends to the Mariculture Committee, that a meeting is facilitated and organised with the participation of key representatives from ICES groups dealing with AES, (i.e. WGEIM, WGMASC, WGITMO and SGBOSV). This meeting (group) would be tasked to prepare a joint document highlighting, among other things, an update on the extent of introductions related specifically to aquaculture activities, the mechanism and interactions of the exotics with their new environment and; on the basis of these, identify information gaps and recommend specific research goals to fill these gaps.

It is clear that ICES could play a key role in addressing the growing need for information and advice on the management of AES. Presently, AES is mainly being addressed on national basis with significant inconsistencies in data collection, monitoring and management approaches. ICES could address some of these challenges and offers an international forum to provide coherent advice for the North Atlantic Zone.

Further to this, WGEIM recommends to ICES, that a business case is prepared to organize a symposium to initiate the discussion among member countries on working collaboratively to address identified research gaps, collection and sharing of data and provide advice on the mitigation and management of AES and their impacts on aquaculture.

10.8 References

Albrecht, A., and Reise, K. 1994. Effects of *Fucus vesiculosus* covering intertidal mussel beds in the Wadden Sea. Helgol. Meeresunters. 48: 243–256.

Altieri, A.H., and Witman, J.D. 2006. Local extinction of a foundation species in a hypoxic estuary: integrating individuals to ecosystem. Ecology, 87: 717–730.

Anonymous. 2003. Clubbed tunicate use options. Prince Edward Island Agriculture, Fisheries and Aquaculture. Technical Report 232: iv + 30 p.

Anonymous 2004. Nonnative oysters in the Chesapeake Bay. National Academies Press, Washington, 344 p.

Barber, B.J. 1996. Impacts of Shellfish introductions on local communities. *In* Exotic species workshop: issues relating to aquaculture and biodiversity. Edited by J. Pedersen. MIT Sea Grant College Program (MITSG 96-15), Cambridge. pp. 18–21.

Bartol, I., and Mann, R. 1997. Small-scale settlement patterns of the oyster *Crassostrea virginica* on a constructed intertidal reef. Bull. Mar. Sci., 61: 881–897.

Barton, E., and Heard, J. 2005. Alien, non-native and invasive marine species. The Marine Biological Association of the United Kingdom *and* The Marine Life Information Network for Britain and Ireland. http://www.marlin.ac.uk/PDF/MLTN_alien_non_natives.pdfs

Bauder, A.G., and Cembella, A.D. 2000. Viability of the toxic dinoflagellate *Prorocentrum lima* following ingestion and gut passage in the bay scallop *Argopecten irradians*. J. Shellfish Res. 19: 321–324.

Behrens Yamada, S. 2001. Global invader: the European green crab. Oregon Sea Grant, Corvalis, 123 p.

Bertness, M.D., Ewanchuk, P.J., and Silliman, B.R. 2002. Anthropogenic modification of New England salt marsh landscapes. Proc. Natl. Acad. Sci. USA 99: 1395–1398.

Binggeli, P. 1994. Misuse of terminology and anthropomorphic concepts in the description of introduced species. Bull. Brit. Ecol. Soc. 25: 10–13.

Blanco, J., Acosta, C.P., de la Puente, M.B., and Salgado, C. 2002. Depuration and anatomical distribution of the amnesic shellfish poisoning (ASP) toxin domoic acid in the king scallop *Pecten maximus*. Aquat. Toxicol., 60: 111–121.

Boothroyd, F.A., MacNair, N.G., Landry, T., Locke, A., and Davidson, T.J. 2002. Dealing with an aquatic invader: the clubbed tunicate (*Styela clava*) in Prince Edward Island waters. *In* Aquaculture Canada 2002. Charlottetown, PEI, 17–20 September 2002.

Bourque, D., Landry, T., Davidson, J., and McNair, N. 2003a. Impact of an invasive tunicate in Atlantic Canada: Recruitment and competition. J. Shellfish Res. 22: 320.

Bourque, D., LeBlanc, A., and Landry, T. 2003b. Inter-specific competition between an invasive filter-feeding tunicate (*Styela clava*) and cultivated mussels (*Mytilus edulis*) in Prince Edward Island, Canada. *In* Aquaculture Canada 2003. Victoria, BC, 29 October–1 November 2003.

Bower, S.M., and McGladdery, S.E. 2003. Synopsis of infectious diseases and parasites of commercially exploited shellfish http://www-sci.pac.dfo-mpo.gc.ca/shelldis/title_e.htm

Bower, S.M., McGladdery, S.E., and Price, I.M. 1994. Synopsis of infectious diseases and parasites of commercially exploited shellfish. Ann. Rev. Fish Dis., 4: 1–199.

Branch, G.M., and Steffani, C.N. 2004. Can we predict the effects of alien species? A case-history of the invasion of South Africa by *Mytilus galloprovincialis* (Lamarck). J. Exp. Mar. Biol. Ecol., 300: 189–215.

Bricelj, V.M., Greene, M., and Cembella, A.D. 1993. Growth of the blue mussel *Mytilus edulis* on toxic *Alexandrium fundyense* and effects of gut passage on dinoflagellate cells. *In* Toxic Phytoplankton Blooms in the Sea. Edited by T.J. Smayda and Y. Shimizu. Elsevier, New York. pp. 371–376.

Bricelj, V.M., and Shumway, S.E. 1998. Paralytic shellfish toxins in bivalve molluscs: occurrence, transfer kinetics, and biotransformation. Rev. Fish. Sci., 6: 315–383.

Bruno, J.F., and Bertness, M.D. 2001. Habitat modification and facilitation in benthic marine communities. *In* Marine community ecology. Edited by M.D. Bertness, S.D. Gaines and M.E. Hay. Sinauer Associates, Inc., Sunderland. pp. 201–218.

Buhle, E.R., Margolis, M.M., and Ruesink, J.L. 2005. Bang for buck: cost-effective control of invasive species with different life histories. Ecol. Econ., 52: 355–366.

Bullard, S., Whitlatch, R.B., Shumway, S.E., and Osman, R.W. 2005. Scientists crying foul: sea squirts are invading Long Island Sound! Wrack Lines, 5: 2–6.

Burreson, E.M., and Ford, S.E. 2004. A review of recent information on the Haplosporidia, with special reference to *Haplosporidium nelsoni* (MSX disease). Aquat. Living Resour., 17: 499–517.

Burreson, E.M., Stokes, N.A., and Friedman, C.S. 2000. Increased virulence in an introduced pathogen: *Haplosporidium nelsoni* (MSX) in the Eastern oyster *Crassostrea virginica*. J. Aquat. Anim. Health, 12: 1–8.

Bushek, D., Richardson, D., Bobo, M.Y., and Coen, L.D. 2004. Quarantine of oyster shell cultch reduces the abundance of *Perkinsus marinus*. J. Shellfish Res., 23: 369–373.

Campbell, M. 1997. National report for Canada. *In* Report of the working group on introductions and transfers of marine organisms (WGITMO), ICES CM 2000/ACME: 07. Edited by ICES Advisory Committee on the Marine Environment. pp. 24–30.

Carbines, G.D. 1993. The ecology and early life history of *Notolabrus celidotus* (Pisces: Labridae) around mussel farms in the Marlborough Sounds. Thesis (MSc), University of Canterbury, Christchurch, 134 p.

Carlton, J.T. 1989. Man's role in changing the face of the ocean: biological invasions and implications for conservation of near-shore environments. Conserv. Biol., 3: 265–273.

Carlton, J.T. 1992a. Introduced marine and estuarine mollusks of North America: an end-of-the-20th-century perspective. J. Shellfish Res., 11: 489–505.

Carlton, J.T. 1992b. Dispersal of living organisms into aquatic ecosystems as mediated by aquaculture and fisheries activities. *In* Dispersal of living organisms into aquatic ecosystems. Edited by A. Rosenfield and R. Mann. Maryland Sea Grant, College Park. pp. 13–46.

Carlton, J.T. 1999. Molluscan invasions in marine and estuarine communities. Malacologia, 41: 439–454.

Carnegie, R.B., Barber, B.J., and Distel, D.L. 2003. Detection of the oyster parasite *Bonamia ostreae* by fluorescent in situ hybridization. Dis. Aqaut. Organisms, 55: 247–252.

Carver, C.E., Chisholm, A., and Mallet, A. 2003. Strategies to mitigate the impact of *Ciona intestinalis* (L.) biofouling on shellfish production. J. Shellfish Res., 22: 621–631.

Castel, J., Labourg, P.J., Escaravage, V., Auby, I., and Garcia, M.E. 1989. Influence of seagrass beds and oyster parks on the abundance and biomass patterns of meio- and macrobenthos in tidal flats. Estuar, Coast. Shelf Sci., 28: 71–85.

Chauvaud, L., Thouzeau, G., Grall, J., and Paulet, Y.M. 2003. Exploitation et surexploitation des ressources marines vivantes, #17: La crepidule en rade de Brest: un paradoxe pour le devenir de la coquille Saint-Jacques. Rapp. Sci. Technol., Acad. Sci. Paris, 17: 307–318.

Chew, K.K. 1990. Global bivalve shellfish introductions. World Aquacult., 21(3): 9–22.

Cloern, J.E. 1982. Does the benthos control phytoplankton biomass in south San Francisco Bay? Mar. Ecol. Prog. Ser., 9: 191–202.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser., 210: 223–253.

Cook, E., Willis, K., and Ashton, G. 2004. Population dynamics and development of an invasive caprellid amphipod - *Caprella mutica*. *In* 13th International Conference on Aquatic Invasive Species. Ireland, pp. 59.

Critchley, A.T., and Dijkema, R. 1984. On the presence of the introduced brown alga *Sargassum muticum*, attached to commercially imported *Ostrea edulis* in the S. W. Netherlands. Bot. Mar., 27: 211–216.

Curiel, D., Guidetti, P., Bellemo, G., Scattolin, M., Marzocchi, M. 2001. The introduced alga *Undaria pinnatifida* (Laminariales, Alariaceae) in the lagoon of Venice. Hydrobiol., 477: 209–219.

Currie, D.R., McArthur, M.A., and Cohen, B.F. 2000. Reproduction and distribution of the invasive European fanworm *Sabella spallanzanii* (Polychaeta: Sabellidae) in Port Phillip Bay, Victoria, Australia. Mar. Biol., 136: 645–656.

Dame, R.F. 1993. Bivalve filter feeders in estuarine and coastal ecosystem processes. Springer-Verlag, Berlin, 579 p.

Dayton, P.K. 1972. Toward an understanding of community resilience and the potential effects of enrichments to the benthos at McMurdo Sound, Antarctica. *In* Colloquium on Conservation problems in Antarctica. Blacksburg, VA, 10–12 September, 1971. Edited by B.C. Parker. pp. 81–96.

De Montaudouin, X., Labarraque, D., Giraud, K., and Bachelet, G. 2001. Why does the introduced gastropod *Crepidula fornicata* fail to invade Arcachon Bay (France)? J. Mar. Biol. Ass. U. K., 81: 97–104.

Dealteris, J.T., Kilpatrick, B.D., and Rheault, R.B. 2004. A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. J. Shellfish Res., 23: 867–874.

Dijkema, R. 1995. Large-scales recirculation systems for storage of imported bivalves as a means to counteract introduction of cysts of toxic dinoflagellates in the coastal waters of the Netherlands. *In* 2nd Conference Internationale sur la Purification des Coquillages, Rennes (France), 6–8 April 1992. Edited by R. Poggi and J.Y. Le Gall. pp. 355–367.

Drinkwaard, A.C. 1999. Introductions and developments of oysters in the North Sea area: a review. Helgol. Meeresunters., 52: 301–308.

Elston, R.A., Farley, C.A., and Kent, M.L. 1986. Occurrence and significance of bonamiasis in European flat oysters *Ostrea edulis* in North America. Dis. Aqaut. Organisms, 2: 49–54.

Farley, C.A. 1992. Mass mortalities and infectious lethal diseases in bivalve molluscs and associations with geographic transfers of populations. *In* Dispersal of living organisms into aquatic ecosystems. Edited by A. Rosenfield and R. Mann. Maryland Sea Grant, College Park. pp. 139–154.

Figueras, A. 2004. Foreward. Aquat. Living Resour. 17: 395.

Forrest, B., Dodgshun, T., and Blakemore, K. 2004. Vector management tools for invasive marine species: reducing the spread of biofouling pests with aquaculture transfers. *In* 13th International Conference on Aquatic Invasive Species. Ennis, Ireland, 20–24 September 2004. pp. 159.

Fralick, R.A. 1970. An ecological study of *Codium fragile* subsp. *tomentosoides* in New England. Thesis (MSc), University of New Hampshire, Durham.

Galtsoff, P.S. 1964. The American Oyster *Crassostrea virginica* Gmelin. U.S. Fish Wildl. Service, Fish. Bull., 64: 1–480.

Getchis, T.S. 2005. What's putting some aquaculturists in a "foul" mood? Fouling organisms are taking their toll on marine aquaculture. Wrack Lines, 5: 8–10.

Glude, J.B. 1955. The effects of temperature and predators on abundance of the soft-shell clam, *Mya arenaria*, in New England. Trans. Am. Fish. Soc., 84: 13–26.

Gollasch, S. 2006. Overview on introduced aquatic species in European navigational and adjacent waters. Helgol. Mar. Res. 60: 84–89.

Goulletquer, P., Bachelet, G., Sauriau, P.G., and Noel, P. 2002. Open Atlantic coasts of Europe — a century of introduced species into French waters. *In* Invasive aquatic species of Europe: Distribution, impacts and management. Edited by E. Leppäkoski, S. Gollasch and S. Olenin. Kluwer Academic Publishers, Dordrecht. pp. 276–290.

Grizel, H., and Heral, M. 1991. Introductions into France of the Japanese oyster (*Crassostrea gigas*). J. Cons. Int. Explor. Mer 47: 339–403.

Grizel, H., Mialhe, E., Chagot, D., Boulo, V., and Bachère, E. 1988. Bonamiasis: a model study of diseases in marine molluscs. *In* Disease processes in marine molluscs. Edited by W.S. Fisher. American Fisheries Society, Special Publication #18, Bethesda. pp. 1–4.

Guenther, J., Southgate, P.C., and de Nys, R. in press. The effect of age and shell size on accumulation of fouling organisms on the Akoya pearl oyster *Pinctada fucata* (Gould). Aquaculture.

Hallegraeff, G.M. 1993. A review of harmful algal blooms and their apparent global increase. Phycologia, 32: 79–99.

Hanisak, M.D. 1979. Nitrogen limitation of *Codium fragile* ssp. *tomentosoides* as determined by tissue. Mar. Biol., 50: 333–337.

Harper, F.M., Hatfield, E.A., Thompson, R.J. 2002. Recirculation of dinoflagellate cysts by the mussel, *Mytilus edulis* L., at an aquaculture site contaminated by *Alexandrium fundyense* (Lebour) Balech. J. Shellfish Res., 21: 471–477.

Harvell, C.D., Burkholder, J.M., Colwell, R.R., Epstein, P.R., Grimes, D.J., Hofmann, E.E., Lipp, E.K., Osterhaus, A.D.M.E., Overstreet, R.M., Porter, J.W., Smith, G.W., Vasta, G.R. 1999. Emerging marine diseases - climate links and anthropogenic factors. Science, 285: 1505–1510.

Hégaret, H., Shumway, S.E., Wikfors, G.H., Pate, S., Burkholder, J.M. 2006. Relocation of bivalve molluscs can be vectors of transport of harmful algae. *In* 35th Benthic Ecology Meeting. Quebec City, QC, 8–12 March 2006. Edited by L.E. Johnson and M. Cusson. pp. 26–27.

Herbold, B., and Moyle, P.B. 1986. Introduced species and vacant niches. Am. Nat. 128: 751–760.

Hubbard, C.B., and Garbary, D.J. 2002. Morphological variation of *Codium fragile* (Chlorophyta) in eastern Canada. Bot. Mar. 45: 476–485.

ICES. 1995. ICES Code of Practice on the Introductions and Transfers of Marine Organisms 1994. ICES Co-operative Research Report No. 204. 19 p.

ICES. 2005. ICES Code of Practice on the Introductions and Transfers of Marine Organisms 2005. ii + 30 p.

Jones, C.G., Lawton, J.H., Shachak, M. 1994. Organisms as ecosystem engineers. Oikos 69: 373–386.

Kaiser, M.J., and Beadman, H.A. 2002. Scoping study of the carrying capacity for bivalve cultivation in the coastal waters of Great Britain. The Crown Estate. Interim Report 39 p.

Khalaman, V.V. 2001. Succession of fouling communities on an artificial substrate of a mussel culture in the White Sea. Rus. J. Mar. Biol., 27: 345–352.

Kott, P. 2002. A complex didemnid ascidian from Whangamata, New Zealand. J. Mar. Biol. Ass. U. K., 82: 625–628.

Kott, P. 2004. A new species of *Didemnum* (Ascidiacea, Tunicata) from the Atlantic coast of North America. Zootaxa, 732: 1–10.

Kröncke, I. 1996. Impact of biodeposition on macrofaunal communities in intertidal sandflats. P. S. Z. N. I: Marine Ecology, 17: 159–174.

Laabir, M., and Gentien, P. 1999. Survival of toxic dinoflagellates after gut passage in the Pacific oyster *Crassostrea gigas* Thunburg. J. Shellfish Res., 18: 217–222.

Lafferty, K.D., and Kuris, A. 1996. Biological control of marine pests. Ecology 77: 1989-2000.

Laing, I., Gollasch, S. 2002. *Coscinodiscus wailesii* - a nuisance diatom in European waters. *In* Invasive aquatic species of Europe: Distribution, impacts and management. Edited by E. Leppäkoski, S. Gollasch and S. Olenin. Kluwer Academic Publishers, Dordrecht. pp. 53–55.

Lambert, C.C., and Lambert, G. 1998. Non-indigenous ascidians in southern California harbors and marinas. Mar. Biol. 130: 675–688.

Landry, T., Tremblay, R., Sevigny, J.M., Archambault, P., Hartog, F., Courtney, S., McKindsey, C.W., and Skinner, M. in press. State of knowledge of mollusc aquaculture: interactions between wild and cultured species. DFO Can. Sci. Advis. Sec. Res. Doc. 2006/xxx: xxx p.

Langeland, G., Hasselgaard, T., Tangen, K., Skulberg, O.M., and Hjelle, A. 1984. An outbreak of paralytic shellfish poisoning in western Norway. Sarsia, 69: 185–193.

Lawrence, J.E., Grant, J., Quilliam, M.A., Bauder, A.G., and Cembella, A.D. 2000. Colonization and growth of the toxic dinoflagellate *Prorocentrum lima* and associated fouling macroalgae on mussels in suspended culture. Mar. Ecol. Prog. Ser., 201: 147–154.

LeBlanc, A.R., Landry, T., and Miron, G. 2002. Fouling organisms in a mussel cultivation bay: their effect on nutrient uptake and release. Can. Tech. Rep. Fish. Aquat. Sci. 2431: vii + 16 p.

LeBlanc, A.R., Landry, T., and Miron, G. 2003. Identification of fouling organisms covering mussel lines and impact of a common defouling method on the abundance of foulers in Tracadie Bay, Prince Edward Island. Can. Tech. Rep. Fish. Aquat. Sci. 2477: vii + 18 p.

Lee, R.J., and Younger, A.D. 2002. Developing microbiological risk assessment for shellfish purification. Int. Biodeterior. Biodegrad., 50: 177–183.

Lesser, M.P., Shumway, S.E., Cucci, T., and Smith, J. 1992. Impact of fouling organisms on mussel rope culture: interspecific competition for food among suspension-feeding invertebrates. J. Exp. Mar. Biol. Ecol., 165: 91–102.

Levasseur, M., Couture, J.-Y., Weise, A., Michaud, S., Elbrächter, M., Sauvé, G., and Bonneau, E. 2003. Pelagic and epiphytic summer distributions of *Prorocentrum lima* and *P. mexicanum* at two mussel farms in the Gulf of St. Lawrence, Canada. Aquat. Microb. Ecol. 30: 283–293.

Lilly, E.L., Kulis, D.M., Gentien, P., and Anderson, D.M. 2002. Paralytic shellfish poisoning toxins in France linked to a human-introduced strain of *Alexandrium catenella* from the western Pacific: evidence from DNA and toxin analysis. J. Plankton Res., 24: 443–452.

Locke, A., Hanson, J.M., Ellis, K.M., Thompson, J., and Rochette, R. submitted. Invasion of the southern Gulf of St. Lawrence by the clubbed tunicate (*Styela clava* Herdman): potential mechanisms for invasions of Prince Edward Island estuaries. J. Exp. Mar. Biol. Ecol.

Locke, A., Hanson, J.M., Thompson, J., and Ellis, K.M. 2005. Did green crabs facilitate the clubbed tunicate (*Styela clava*) invasion of Prince Edward Island? *In* International invasive sea squirt conference. Woods Hole Oceanographic Institution, 21–22 April 2005. Edited by M. Carman. pp. 16/59.

Luckenbach, M.W. 2001. Ecological interactions associated with cultured and restored bivalve populations in Chesepeake Bay. *In* Puget Sound Research 2001.

MacDonald, J., MacCallum, G., and MacNair, N. 2005. Biofouling control in mussel farming on Prince Edward Island. *In* Aquaculture Canada 2005. St. John's, NF, 3–6 July 2005.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M., and Bazzaz, F.A. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecol. Appl., 10: 689–710.

Maggs, C.A., and Stegenga, H. 1999. Red algal exotics on North Sea coasts. Helgol. Meeresunters., 52: 243–258.

Malinowski, K.C., and Ramus, J. 1973. Growth of the green alga *Codium fragile* in a Connecticut estuary. J. Phycol., 9: 102–110.

Mann, R. 1983. The role of introduced bivalve mollusc species in mariculture. J. World Maricul. Soc., 14: 436–559.

McKindsey, C.W., Barnes, P., Anderson, M.R., Courtenay, S., Landry, T., and Skinner, M. in press. Effects of shellfish aquaculture on fish habitat. DFO Can. Sci. Advis. Sec. Res. Doc. 2006/011: xxx p.

Meeuwig, J.J., Rasmussen, J.B., and Peters, R.H. 1998. Turbid waters and clarifying mussels: Their moderation of empirical chl:nutrient relations in estuaries in Prince Edward Island, Canada. Mar. Ecol. Prog. Ser., 171: 139–150.

Minchin, D. 1996. Management of the introduction and transfer of marine molluscs. Aq. Conserv: Mar. Freshw. Ecosyst., 4: 229–244.

Minchin, D., Duggan, C.B., Holmes, J.M.C., and Neiland, S. 1993. Introduction of exotic species associated with Pacific oyster transfers from France to Ireland. ICES CM 1993/F:27 11 p.

Minchin, D., and Rosenthal, H. 2002. Exotics for stocking and aquaculture, making correct decisions. *In* Invasive aquatic species of Europe: Distribution, impacts and management. Edited by E. Leppäkoski, S. Gollasch and S. Olenin. Kluwer Academic Publishers, Dordrecht. pp. 206–215.

Mineur, F., Maggs, C., Johnson, M.P., Belsher, T., and Verlaque, M. 2004. Vectors for introduction of alien macroalgae in Europe: oyster transfers. *In* 13th International Conference on Aquatic Invasive Species. Ennis, Ireland, pp. 158.

Naylor, R.L., Williams, S.L., Strong, D.R. 2001. Aquaculture - a gateway for exotic species. Science 294: 1655–1656.

Neushul, M., Amsler, C.D., Reed, D.C., Lewis, R.J. 1992. Introduction of marine plants for aquaculture purposes. *In* Dispersal of living organisms into aquatic ecosystems. Edited by A. Rosenfield and R. Mann. Maryland Sea Grant, College Park. pp. 103–135.

O'Beirn, F.X., Luckenbach, M.W., Nestlerode, J.A., Coates, G.M. 2000. Oyster recruitment as a function of substrate type and tidal height. J. Shellfish. Res. 19: 387–395.

O'Beirn, F.X., Ross, P.G., Luckenbach, M.W. 2004. Organisms associated with oysters cultured in floating systems in Virginia, USA. J. Shellfish. Res., 23: 825–829.

O'Mahony, J.H.T. 1993. Phytoplankton species associated with imports of the Pacific oyster *Crassostrea gigas*, from France to Ireland. ICES CM 1993/F:26 8 p.

Osman, R.W., Whitlatch, R.B. 1995a. The influence of resident adults on larval settlement: experiments with four species of ascidians. J. Exp. Mar. Biol. Ecol., 190: 199–220.

Osman, R.W., Whitlatch, R.B. 1995b. The influence of resident adults on recruitment: a comparison to settlement. J. Exp. Mar. Biol. Ecol., 190: 169–198.

Osman, R.W., Whitlatch, R.B. 1995c. Ecological factors controlling the successful invasion of three species of ascidians into marine subtidal habitats of New England. *In* Proceedings of the Northeast Conference on Non-Indigenous Aquatic Nuisance Species. Cromwell, Connecticut, January 25, 1995. Edited by N.C. Balcom. pp. 49-60.

Osman, R.W., and Whitlatch, R.B. 2004. The control of the development of a marine benthic community by predation on recruits. J. Exp. Mar. Biol. Ecol., 311: 117–145.

Otwell, W.S., Rodrick, G.E., and Martin, R.E. 1991. Molluscan shellfish depuration. CRC Press, Boca Raton, 384 p.

Penna, A., Garcés, E., Vila, M., Giacobbe, M.G., Fraga, S., Lugliè, A., Bravo, I., Bertozzini, E., and Vernesi, C. 2005. *Alexandrium catenella* (Dinophyceae), a toxic ribotype expanding in the NW Mediterranean Sea. Mar. Biol., 148: 13–23.

Pratt, D.M. 1974. Attraction to prey and stimulus to attack in the predatory gastropod *Urosalpinix cinerea*. Mar. Biol., 27: 37–45.

Prins, T.C., Smaal, A.C., and Dame, R.F. 1998. A review of the feedbacks between bivalve grazing and ecosystem processes. Aquat. Ecol., 31: 349–359.

Ramus, J. 1971. *Codium*, the invader. Discovery, 6: 59–68.

Reise, K. 1998. Pacific oysters invade mussel beds in the European Wadden Sea. Senckenb. Marit., 28: 167–175.

Reise, K., Olenin, S., and Thieltges, D.W. 2006. Are aliens threatening European aquatic coastal ecosystems? Helgol. Mar. Res., 60: 77–83.

Ribera Siguan, M.A. 2003. Pathways of biological invasions of marine plants. *In* Invasive species: vectors and management strategies. Edited by G.M. Ruiz and J.T. Carlton. Island Press, Washington. pp. 183–226.

Rodriguez, L.F. 2005. Novel habitats created by non-indigenous species: The role of oysters and ascidians as biological substrata for fouling communities. *In* International invasive sea squirt conference. Woods Hole Oceanographic Institution, 21–22 April 2005. Edited by M. Carman. pp. 17/59.

Rosenfield, A. 1992. Risks associated with translocations of biological agents. *In* Dispersal of living organisms into aquatic ecosystems. Edited by A. Rosenfield and R. Mann. Maryland Sea Grant, College Park. pp. 3–12.

Ross, K.A., Thorpe, J.P., Brand, A.R. 2004. Biological control of fouling in suspended scallop cultivation. Aquaculture, 229: 99–116.

Rueness, J. 1989. *Sargassum muticum* and other introduced Japanese macroalgae: Biological pollution of European coasts. Mar. Pollut. Bull., 4: 173–176.

Ruesink, J.L., Lenihan, H.S., Trimble, A.C., Heiman, K.W., Micheli, F., Byers, J.E., Kay, M.C. 2005. Introduction of non-native oysters: ecosystem effects and restoration implications. Ann. Rev. Ecol. Syst., 36: 643–689.

Ruiz, G.M., Fofonoff, P., Hines, A.H., and Grosholz, E.D. 1999. Non-indigenous species as stressors in estuarine and marine communities: Assessing invasion impacts and interactions. Limnol. Oceanogr., 44: 950–972.

Ruiz, G.M., Fofonoff, P.W., Carlton, J.T., and Hines, A.H. 2000. Invasion of coastal marine communities in North America: apparent patterns, processes, and biases. Ann. Rev. Ecol. Syst., 31: 481–531.

Scarratt, A.M., Scarratt, D.J., and Scarratt, M.G. 1993. Survival of live *Alexandrium tamarense* cells in mussel and scallop spat under simulated transfer conditions. J. Shellfish Res., 12: 383–388.

Sekiguchi, K., Sato, S., Ogata, T., Kaga, S., and Kodama, M. 2001. Accumulation and depuration kinetics of paralytic shellfish toxins in the scallop *Patinopecten yessoensis* fed *Alexandrium tamarense*. Mar. Ecol. Prog. Ser., 220: 213–218.

Shatkin, G., Shumway, S.E., Hawes, R. 1997. Considerations regarding the possible introduction of the pacific oyster (*Crassostrea gigas*) to the Gulf of Maine: a review of global experience. J. Shellfish Res., 16: 463–477.

Shumway, S.E., Davis, C., Downey, R., Karney, R., Kraeuter, J., Parsons, J., Rheault, R., Wikfors, G. 2003. Shellfish aquaculture - in praise of sustainable economies and environments. World Aquacult., 34: 15–18.

Simard, N., and Hardy, M. 2004. The Laurentian Channel as an alternative ballast water exchange zone: risks, analysis and recommendations. DFO Can. Sci. Advis. Sec. Res. Doc. 2004/120: iii + 74 p.

Simard, N., Paille, N., and McKindsey, C.W. in press. *Codium fragile* ssp. *tomentosoides*: revue de littérature et situation aux Îles-de-la-Madeleine. Can. Manuscr. Rep. Fish. Aquat. Sci. 2006/? vii + 39 p.

Sindermann, C.J. 1984. Disease in marine aquaculture. Helgol. Meeresunters., 37: 505–532.

Smayda, T.J., Reynolds, C.S. 2001. Community assembly in marine phytoplankton: application of recent models to harmful dinoflagellate blooms. J. Plankton Res., 23: 447–461.

Springer, J., Shumway, S.E., Burkholder, J.M., Glasgow, H.B. 2002. Interactions between the toxic estuarine dinoflagellate, *Pfiesteria piscicida*, and two species of bivalve molluscs. Mar. Ecol. Prog. Ser., 245: 1–10.

Stachowicz, J.J., Fried, H., Osman, R.W., Whitlach, R.B. 2002. Biodiversity, invasion resistance, and marine ecosystem function: reconciling pattern and process. Ecology 83: 2575–2590.

Stachowicz, J.J., Whitlach, R.B., and Osman, R.W. 1999. Species diversity and invasion resistance in a marine ecosystem. Science, 286: 1577–1579.

Streftaris, N., Zenetos, A., and Papathanassiou, E. 2005. Globalisation in marine ecosystems: The story of non-indigenous marine species across European seas. Oceanogr. Mar. Biol. Ann. Rev., 43: 419–453.

Swan, K., Davidson, J., and Landry, T. 2005. The effect of treatment regimes on the physiological fitness of the Prince Edward Island cultured blue mussel *Mytilus edulis* population. *In* AquaNet V. Victoria, BC, 18–21 October 2005. pp. E41.

Tenore, K.R., and González, N. 1976. Food chain patterns in the Ria de Arosa, Spain: an area of intense mussel aquaculture. *In* Proceedings of the 10th European Symposium on Marine Biology, Ostend, Belgium, 17–23 September 1975. Vol 2: Population dynamics of marine organisms in relation with nutrient cycling in shallow waters. Edited by G. Persoone and E. Jaspers. Universa Press, Wetteren. pp. 601–619.

Thiel, M., and Dernedde, T. 1994. Recruitment of shore crabs *Carcinus maenas*, on tidal flats: mussel clumps as an important refuge for juveniles. Helgol. Meeresunters., 48: 321–332.

Thompson, R., and MacNair, N. 2004. An overview of the clubbed tunicate (*Styela clava*) in Prince Edward Island. Prince Edward Island Agriculture, Fisheries and Aquaculture. Technical Report 234: viii + 29 p.

Trowbridge, C.D. 1998. Ecology of the green macroalga *Codium fragile* (Suringar) Hariot 1889: invasive and non-invasive subspecies. Oceanogr. Mar. Biol. Annu. Rev., 36: 1–64.

Tsujino, M., Kamiyama, T., Uchida, T., Yamaguchi, M., and Itakura, S. 2002. Abundance and germination capability of resting cysts of *Alexandrium* spp. (Dinophyceae) from faecal pellets of macrobenthic organisms. J. Exp. Mar. Biol. Ecol., 271: 1–7.

Vercaemer, B., Spence, K., Kenchington, E., Mallet, A., and Harding, J. 2003. Assessment of genetic diversity of the European oyster (*Ostrea edulis*) in Nova Scotia using microsatellite markers. Can. Tech. Rep. Fish. Aquat. Sci. 2453: v + 30 p.

Verlaque, M. 2001. Checklist of the macroalgae of Thau Lagoon (Herault, France), a hot spot of marine species introduction in Europe. Oceanol. Acta, 24: 29–49.

Wallentinus, I. 2002. Introduced marine algae and vascular plants in European aquatic environments. *In* Invasive aquatic species of Europe: Distribution, impacts and management. Edited by E. Leppäkoski, S. Gollasch and S. Olenin. Kluwer Academic Publishers, Dordrecht. pp. 27–52.

Wasson, K., Zabin, C.J., Bedinger, L., Diaz, M.C., and Pearse, J.S. 2001. Biological invasions of estuaries without international shipping: the importance of intraregional transport. Biol. Conserv., 102: 143–153.

Whitlatch, R.B., Osman, R.W., Frese, A., Malatesta, R., Mitchell, P., and Sedgwick, L. 1995. The ecology of two introduced marine ascidians and their effects on epifaunal organisms in Long Island Sound. *In* Proceedings of the Northeast Conference on Non-Indigenous Aquatic Nuisance Species. Cromwell, Connecticut, 25 January 1995. Edited by N.C. Balcom. pp. 29–48.

Wikström, S.A., and Kautsky, L. 2004. Invasion of habitat-forming seaweed: effects on associated biota. Biol. Invasions, 6: 141–150.

Wolff, W.J., and Reise, K. 2002. Oyster imports as a vector for the introduction of alien species into Northern and Western European coastal waters. *In* Invasive aquatic species of Europe: distribution, impacts and management. Edited by E. Leppäkoski, S. Gollasch and S. Olenin. Kluwer Academic Publishers, Dordrecht. pp. 193–205.

Wonham, M.J., and Carlton, J.T. 2005. Trends in marine biological invasions at local and regional scales: the Northeast Pacific Ocean as a model system. Biol. Invasions, 7: 369–392.

11 Recommendations

RECOMMENDATION	ACTION
The WG recommended that the documents relating to risk analysis of non-salmonid species be completed and submitted to an appropriate journal intersessionally	WGEIM members
WGEIM recommends to the MCC, that the Term of Reference B (Review of EU legislation) be put in abeyance. It is further recommended that members of the group adopt a watching brief on EU and other National legislation until 2008. In the interim, if a piece of legislation is identified that has relevance to Mariculture it can be reported and discussed during subsequent meetings as a Term of Reference or in AOB until it is more fully dealt with in 2008.	MCC
WGEIM recommends to the MCC and ICES that the efforts of Krajnc and Glavic (2005) be adopted to develop composite sustainability indices as a potential method to combine environmental, economic and social sub-indices. Sustainability Indices are more comprehensive than environmental impacts assessments. They incorporate and try to integrate the “triple bottom line” concept (social, economic, and environmental) assessments. We recognize that the strength of ICES is in the environmental and ecological fields, but to develop SIs further, ICES needs to broaden the scope of investigations to incorporate social and economic assessments as highlighted in the Action Plan No. 3.12. This experience should be brought to WGEIM to deal with this ToR specifically.	MCC, ICES
The ECASA program is evaluating environmental Sustainability Indices of the highest scientific credibility that will be peer reviewed for analyses of precision, accuracy, reliability, and consistency. WGEIM recommend that ICES evaluate the findings of this EU funded project that appropriate environmental SIs arising form this program might feature in recommended monitoring and management protocols.	ICES
WGEIM recognise that the technical challenges associated with the practical aspects of commercial-scale multi-trophic aquaculture systems must be addressed and these results presented in a context that could be assessed by the investment community and consider these development opportunities in order to successfully transfer the concept of integrated aquaculture to industry. Ongoing research should consider multi-disciplinary, commercial-scale testing of integrated aquaculture systems to permit all of the environmental, social and economical issues to be addressed accordingly. Considering that forthcoming results from these projects will offer more insight into the benefits and challenges of IMTA, the WGEIM recommends that this topic be revisited as a term of reference in two years (2008).	MCC
WGEIM recommend to the MC that members of the working group will lead a review and evaluation of recent advances on alternative sources of lipid and protein to fish oil and fish meal in aquafeed. It is proposed that a WGEIM review a draft manuscript at the 2007 meeting that is to be submitted for publication in a peer reviewed scientific journal.	MCC
WGEIM recommends to the Mariculture Committee and ICES, that a meeting is facilitated and organised with the participation of key representatives from ICES groups dealing with the Aquacultural Engineering Society (AES), i.e. WGEIM, WGMASC, WGITMO and SGBOSV. This meeting (group) would be tasked to prepare a joint document highlighting, among other things, an update on the extent of introductions related specifically to aquaculture activities, the mechanism and interactions of the exotics with their new environment and; on the basis of these, identify information gaps and recommend specific research goals to fill these gaps.. It is clear that ICES could play a key role in addressing the growing need for information and advice on the management of AES. Presently, AES is mainly being addressed on national basis with significant inconsistencies in data collection, monitoring and management approaches. ICES could address some of these challenges and offer an international forum to provide coherent advice for the North Atlantic Zone. Further to this, WGEIM recommends to ICES, that a business case is prepared to organize a symposium to initiate the discussion among member countries on working collaborately to address identified research gaps, collection and sharing of data and provide advice on the mitigation and management of AES and their impacts on the marine ecosystem.	MCC, ICES

12 WGEIM Draft Resolution 2006

The ICES Working Group on Environmental Interactions of Mariculture [WGEIM] (Chair F. O'Beirn, Ireland) will meet in Kiel, Germany, from 16–20 April 2007 to:

a) discuss the status of the risk assessment papers on non-salmonid mariculture species and the outcome of the GESAMP WG 31 meeting in November 2007.

b) further evaluate the examples of sustainability indices proposed for mariculture activities and critically evaluate those SIs recommended by WGEIM and other forums.

c) further the review on alternative feeds with a view to generating a manuscript to be submitted for publication in a peer reviewed scientific journal.

d) further investigate fouling hazards associated with the physical structures used in Mariculture and assess their potential for the introduction of invasive/nuisance species into the local environment

e) review the role and tasks of WGEIM in relation to ICES Strategic Plan and action plan as well the key tasks of the Mariculture Committee and prepare a draft future work plan.

WGEIM will report by 15 May 2007 for the attention of the Mariculture Committee and ACME.

Supporting Information

Priority	The activities of this group are fundamental to the work of the Mariculture Committee. The work is essential to the development and understanding of the effects of man-induced variability and change in relation to the health of the ecosystem. The work of this ICES WG is deemed high priority.
Scientific justification and relation to action plan	Action Plan references: a) 2.5, 2.6, 2.10, 3.3, 3.11 b) 2.2, 3.2, 3.3, 3.5, 3.12, 4.7, 5.3 c) 3.8, 3.9, 3.10, 3.11, 4.11.3 d) 1.2, 1.10, 2.11, 3.6 e) 7.1, 7.2, 8.1 As identified previously by WGEIM, regulatory actions that limit the transportation and utilization of mariculture species can be viewed as a non-technical barrier to trade under international trade agreements. Risk analysis is one method of identifying environmental risks associated with the utilization of new species in culture and of justifying environmentally based constraints on the transfer and use of the species. GESAMP WG31 is developing methodologies for analyzing environmental risks associated with aquaculture activities. Their application to the environmental risks associated with culturing new mariculture 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. WGEIM have produced 6 papers on this issues: one is an introductory paper that introduces the template for risk analysis followed by five case studies on five different aquaculture species. These papers are close to final condition and will be completed intercessionally and submitted for publication all together. An update will be provided at WGEIM 2007. Lead: Edward Black (Canada) Sustainability indexes have, among other uses, been offered as a methodology to integrate large amounts of scientific information to underpin management and regulatory decisions. Some current research in the EU are evaluating an extensive range of environmental indicators and assessing their utility relating to aquaculture systems. This research will be reviewed and the utility of any indices proposed will be evaluated in light of the criteria for an acceptable sustainability index outlined by WGEIM 2005. Additional indices will be assessed with priority foucusing upon composite indices incorporating economic, environmental ans social aspects. The importance of multidisiciplinary approaches to defining sustainability will be also assessed. Lead: Barry Costa-Pierce, USA WGEIM 2003 and other ICES group have previously reviewed this issue. However, the sustainability of utilising fish based oil in feed products for marine fish farm activities continue to be questioned and justification continues to be sought. Feed producing companies are apparently endeavouring to find alternative sources. The goal of this work package is to provide and update on the progress in identifying alternatives to fish oil for feed in finfish aquaculture. Intercessional communication with industry sources and other working groups WGMAFC will be carried out and reported upon at the meeting. Lead: Kats Haya, Canada

	Structure associated with mariculture activities can provide considerable surface area for colonisation of species not typically found in the culture area. This is presumably due to the increased habitat complexity and appropriate substrate for epifuanal organisms. The question is raised, do these structures have to potential to provide a pathway for the introduction of an exotic nuisance species to a system, which could potentially spread over larger geographical area once established. Existing examples will be examined and mechanisms elucidated more clearly. The management implications and potential mitigation strategies will also be addressed. Lead: To clearly identify the value of the topics covered in the WGEIM and ensure they are relevant to the ICES Strategic pan and action plans. More specifically, there are 10 goals outlined ICES Strategic plan and the relevance of the work of the group will be examined in light of these goals. The relevance of information emerging from WGEIM will be also assessed and it's relevance evaluated in light of the requirements of ICES client organisations or user groups. In addition, the products of WGEIM will be considered in relation to those of other ICES groups beyond the current mariculture groups so as to modify the integrated advice model currently being developed by ICES. Finally, this exercise will provide a fuller understanding of the working arrangements and outputs of the group such that is has clear relevance to marine management issues in each member state.
Resource Requirements	None
Participants	The Group is normally attended by some 12–15 members and guests
Secretariat Facilities	None
Financial	No financial implications
Linkages to Advisory Committees	ACME
Linkages to other committees or groups	WGEIM interacts with WGMASC, WGAGFM, MARC
Linkages to other organisations	The work of this group is undertaken in close collaboration with the DFO Gesamp group, BEQUALM, OIE, EU, EAS
Secretariat marginal Costs	ICES

13 Close of Meeting

The meeting was closed on April 28 at 12:30 pm. The Chair expressed thanks to all the members of the group for their input in to the work during the week. The local host Dr. Barry Costa-Pierce was thanked for hosting the meeting. Special thanks were reserved for Heather Rhodes of the Rhode Island Sea Grant Program for her considerable efforts to facilitate the smooth running of the meeting.

Annex 1: List of participants

NAME	ADDRESS	PHONE/FAX	EMAIL
Francis O'Beirn (Chairman)	Marine Institute Rinville, Oranmore, Galway, Ireland	+353 91 387250/ +353 91 387 201	francis.obeirn@marine.ie
Edward Black	Department of Fisheries and Oceans Aquaculture Science Branch, 200 Kent St. Ottawa, Ontario Canada, K1A 0E6	+1 613 990 0272/ +1 613 993 7665	blacke@dfo-mpo.gc.ca
Barry Costa-Pierce	Rhode Island Sea Grant College Program, Graduate School of Oceanography, University of Rhode Island, USA	+1 401 874 6802/ +1 401 789 8340	bcp@gso.uri.edu
Ian M. Davies	Fisheries Research Services, Marine Laboratory P.O. Box 101 375 Victoria Road Aberdeen AB11 9DB United Kingdom	+44 1 224 876 544/ +44 1 224 295 511	daviesim@marlab.ac.uk
Antoine Dosdat	IFREMER European Affairs 155 Rue Jean Jacques Rousseau 92138 Issy Les Moulineaux France	+33 146 482 256/ +33 146 482 121	adosdat@ifremer.fr
Kats Haya	Dept. of Fisheries & Oceans, Biological Station 531 Brandy Cove Road, St Andrews, N.B. ESB 2L9 Canada	+1 506 529 5916/ +1 506 529 5862	hayak@mar.dfo-mpo.gc.ca
Helmut Thetmeyer	IFM-GEOMAR Duesternbrooke Weg 20, D-24105 Kiel Germany	+49 4316004569/ +49 1212513060164	hthetmeyer@ifm-geomar.de
Stephen Cross	Aquametrix Research Ltd., 2541 Conrad Rd., Courtenay, B.C. Canada V9N9N8	+1 250 338 9255/ +1 250 338 9288	cross@aquametrix-research.com
Chris McKindsey	Fisheries and Oceans Canada, 850 Route de la Mer Mont-Joli, QC G5H 3Z4, Canada	+1 418 775 0667/ -	mckindsey@dfo-mpo.gc.ca
Thomas Landry	DFO/MPO, Gulf Fisheries Centre, 343 Universite Ave. Moncton, N.B. E1A 4Y1, Canada	+1 506 851 6219/ +1 506 851 2079	landry@dfo-mpo.gc.ca
David Bengtson	Dept. Fisheries, Animal and Veternary Science, University of Rhode Island, Kingston RI 02881, USA	+1401 874 2668/	bengtson@uri.edu

NAME	ADDRESS	PHONE/FAX	EMAIL
Sharon McClean Chair WGPDMO (Observer)	Narragansett Laboratory NMFS/NOAA 28 Tarzwell Drive Narragansett, RI 02882-1199 USA	+1 401-782-3258	smaclean@mola.na.nmfs.gov

Annex 2: WGEIM Agenda

Working Group on Environmental Interactions of Mariculture (WGEIM)

Narragansett, Rhode Island, USA, 24–28 April 2006

Monday, 24 April

09:00 House-keeping and support arrangements – Heather, Barry
Welcome from URI - Barry
9:45 Introduction of Participants, Review of Terms of Reference and Designation of Rapporteurs and drafting group members – Chair, All.
10:30 *Comfort Break*
11:00 Brief presentations and discussion on individual ToR from leaders:
ToR A – Risk Assessment – Edward Black
ToR B – EU Legislation – Francis O'Beirn
ToR C – Sustainability indices – Barry Costa-Pierce
- ECASA Overview – Helmut Thetmeyer
ToR D – Multitrophic systems – Steve Cross
ToR E – Update on Alternative feeds – Kats Haya
ToR F – Fouling hazards – Chris McKindsey
12:00 ***LUNCH***
13:00 Break out to drafting groups
15:00 *Comfort Break*
15:15 Return to Drafting Groups
16:45 Plenary – Progress update

Tuesday, 25 April

08:30 Plenary Session – overview of work to be carried out - All
09:15 Drafting groups reconvene
10:00 *Comfort Break*
10:30 Drafting groups reconvene
12:00 *LUNCH*
13:00 Drafting groups reconvene
15:00 *Comfort Break*
15:15 Drafting groups reconvene
16:45 Plenary – Progress update

Wednesday, 26 April

08:30 Plenary – Progress to report? - All
09:15 Drafting groups reconvene
10:00 *Comfort Break*
10:30 Drafting groups reconvene
12:00 *LUNCH*
13:00 **Field Trip to American Mussel Harvesters**

Evening Reception at Almafi Restaurant, sponsored by RI Sea Grant

Thursday, 27 April

08:30 Progress distributed and read and discussed
09:15 Drafting groups reconvene
10:00 *Comfort Break*
10:30 Drafting groups reconvene
12:00 *LUNCH*
13:00 Drafting groups reconvene
15:00 *Comfort Break*
15:15 Days progress distributed and read
15:45 Presentation of Progress and discussion

Friday, 28 April

09:00 Leaders pass executive summary text, draft recommendations and 2005 ToR proposals to the Chair

Drafting of final document - groups reconvene

10:00 *Comfort Break*

10:15 Plenary

- Update on joint Session (with WGMASC) for ASC 2007 on Ecological Carrying capacity.
- Adoption of the scientific text of the report
- Discussion on Recommendations
- Discussion on new Terms of Reference
- Location of next meeting
- Any other business

12:00 End of 2006 meeting

LUNCH

Annex 3: WGEIM Terms of Reference for 2006

2005/2/MCC03 The **Working Group on Environmental Interactions of Mariculture** [WGEIM] (Chair: F. O’Beirn, Ireland) will meet in Narragansett, Rhode Island, USA, from 24–28 April 2006 to:

a) review the outcome of the GESAMP WG 31 on the aquaculture risk analysis methodologies and finalise case studies examining the potential impacts of escaped non-salmonid farmed fish (cod, sea bass, sea bream, halibut, turbot);
b) provide an update report on developments in implementation of the Water Framework Directive, the European Marine Strategy, the EU Strategy for sustainable aquaculture and assess their implications for mariculture;
c) evaluate examples of sustainability indices proposed for mariculture operations and provide specific recommendations on utility of proposed
d) evaluate the environmental impacts of integrated (multi-trophic) culture systems and provide recommendations on changes to EU regulatory frameworks that are required to accommodate this form of aquaculture operation;
e) assess and report on the state of knowledge of alternatives to fish for use in formulated feeds for finfish aquaculture.
f) investigate fouling hazards associated with the physical structures used in Mariculture and assess their potential for the introduction of invasive/nuisance species into the local environment.

WGEIM will report by 20 May 2006 for the attention of the Mariculture Committee and ACME.

Supporting information

Priority:	WGEIM addresses many important issues of the ICES Strategic Plan.
Scientific Justification and relation to Action Plan:	Action Plan references: a) 2.6, 2.7, 3.3 b) 4.6 c) 4.11.3 d) 3.11, 4.7 e) 3.8 f) 2.10 Sustainable development of coastal and marine aquaculture is dependent upon the diversification of production into new species with an effort 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 sea bream), 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 template and will recommended mitigative strategies with some worked examples of the aforementioned finfish species. GESAMP WG31 is developing methodologies for analyzing environmental risks associated with aquaculture activities. Their application to the environmental risks associated with culturing new mariculture 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. The current document will be finalised intersessionally by GEASAMP WG31 in December 2005 and reviewed in WGEIM 2006. (Lead: Ian Davies, Scotland) 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. The EU 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 WG will continue to monitor developments in these areas and assess their implications for mariculture. (Lead: Francis O’Beirn, Ireland) Sustainability indexes have, among other uses, been offered as a methodology to integrate large amounts of scientific information to underpin management decisions.

	Some current research in the EU are evaluating an extensive range of environmental indicators and assessing their utility relating to aquaculture systems. This research will be reviewed and the utility of any indices proposed will be evaluated in light of the criteria for an acceptable sustainability index outlined by WGEIM 2005. (Lead: Barry Costa-Pierce, USA) Integrated aquaculture systems (multi-trophic-species systems) are a way to utilise the waste materials from the primary species being farmed to create additional products of significant commercial value while reducing the overall environmental impacts of the site. Soluble nutrients from fish farms can support macro-algae production while the particulate solid wastes are filtered and used by bivalve molluscs. Some practical developments are starting to occur, and the EU has supported work in this area. However, the benefits do need to be fully elucidated for both open or closed systems. The nature of these culture methods does create some regulatory conflicts that will identified and addressed in the report. (Lead: Stephen Cross, Canada) WGEIM 2003 and other ICES group have previously reviewed this issue. However, the sustainability of utilising fish oil based feed products for marine fish farm activities continue to be questioned by environmentalists in the media and clarification is sought continuously. Fish feed manufacturers continue to evaluate alternative sources. The goal of this report is to provide an update on the progress being made in identifying alternatives to fish oil as a lipid source and fish meal as a protein source for feed used in finfish aquaculture. Intercessional communication with industry sources and other working groups WGMAFC will be carried out and reported upon at the meeting. (Lead: Kats Haya, Canada) The physical structure(s) associated with mariculture activities provides a 3 dimensional surface area for the colonisation of species not typically found in the culture area (e.g. artificial reef). This is due to the increased habitat complexity and the provision of a physical substrate for the growth of epifuanal (fouling) organisms. The question is raised, do these structures have to potential to provide a pathway for the introduction of an exotic nuisance species to a system, which could potentially spread over larger geographical area once established. Existing examples will be examined and mechanisms elucidated more clearly. The management implications and potential mitigation strategies will also be addressed. (Lead: Chris McKindsey, Canada)
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,

Annex 4: A risk analysis approach to assessment of the potential genetic interactions of non-salmonid marine finfish escapes from aquaculture with local native wild stocks

1. Introduction

The development of aquaculture is regularly challenged by the complexity of its environmental interactions. The more common and highly debated issues concern the environmental effects, disease interactions and genetic interactions. The complexities of these issues make them susceptible to inconsistencies of assessment, both regionally and internationally, that can lead to economic and trade consequences.

Regulators and developers need to make decisions on investment and development opportunities and proposals, taking account of the wide range of issues concerned. In many cases, full information on the probability of undesirable environmental interactions may not be available, leading to the need to make clear and transparent decisions in situations of high uncertainty.

United Nations Group of Experts on the Scientific Aspects of Marine Protection (GESAMP) identified this as an area where improved systems and advice were required. A joint project between GESAMP Working Group 31 on Environmental Impacts of Coastal Aquaculture and the International Council for the Exploration of the Sea (ICES) Working Group on Environmental Interactions of Mariculture (WGEIM) was initiated to develop improved risk analysis procedures to assist stakeholders in the coastal zone to come to decisions regarding coastal aquaculture proposals.

An important tool in designing and justifying regulatory actions in the international market place is risk analysis. For example, the Office International des Epizootic (OIE) manual for disease control uses risk analysis as the basis for justifying restrictions on movement of aquatic animals in response to concerns about disease transfer and control. Their intent is to provide guidelines and principles for conducting transparent, objective and defensible risk analyses for international trade. Furthermore, the International Council for the Exploration of the Sea (ICES) has embraced this approach in their 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.”

This document and the attached case studies present an illustrated approach to the use of risk analysis in the evaluation of the environmental effects of some aquaculture activities. The intention in doing so is to create a model of a clear and transparent process that regulators can use to come to justifiable decisions regarding the management of their resource allocation decisions.

2 Risk analysis model

Risk analysis can be broken down into four components:

1) Risk Communication;
2) Hazard Identification;
3) Risk Assessment; and
4) Risk Management.

The process and its components are represented diagrammatically in Figure 1. It provides an objective, repeatable, and documented assessment of risks posed by a particular course of action and answers the following questions:

- What can go wrong? – Hazard Identification,
- How likely is it to go wrong and what would be the consequences of it going wrong? – Risk Assessment, and
- What can be done to reduce the likelihood or consequences of it going wrong, or the level of uncertainty in our prediction of the outcome? - Risk Management.

The Risk Assessment component is further broken down into four subcomponent steps: (i) Release Assessment, (ii) Exposure Assessment, (iii) Consequence Assessment and (iv) Risk Estimation. The following risk analysis process has been largely adapted from the process used by the OIE to analyze risks associated with introduction and transfer of aquatic diseases.

It should be noted that the analysis outlined in this paper does not discuss hazards arising from the culture of new exotic species. Regulators should subject new aquaculture species that are exotic to the 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. In the European Union, statutory regulations built on the ICES code are under discussion. The risk analysis outlined below also does not address potential disease interactions, other than to encourage regulators to apply the aforementioned ICES Code and the OIE protocols.

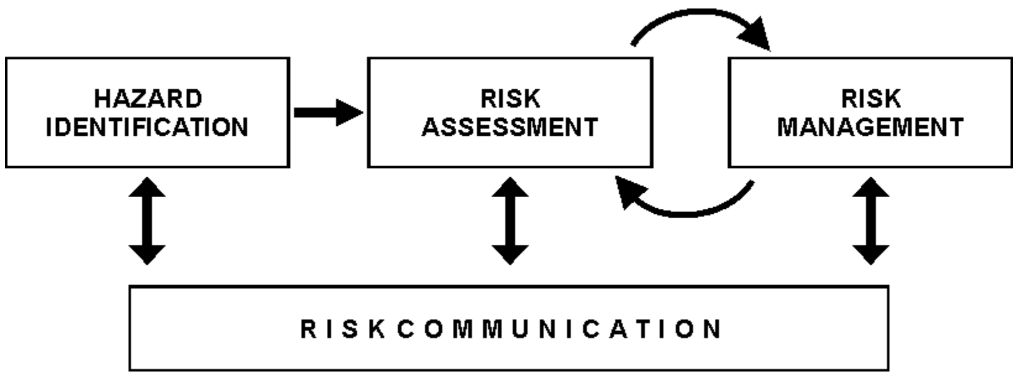


Figure A4.1: The four components of risk analysis (after OIE 2003).

2.1 Risk communication

Risk communication is the process by which information and opinions regarding risks are gathered from potentially affected and other interested parties (stakeholders) 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 stakeholders. It is a multidimensional and iterative process and should begin at the start of the risk analysis process and continue throughout to help ensure involvement in, and acceptance of, the process by all interested parties. The principal participants in risk communication in the current context of aquaculture include regulators, local authorities and other stakeholders such as recreational and commercial fishermen, conservation and wildlife groups, consumer groups, and relevant domestic and foreign industry groups.

It therefore follows that a risk communication strategy should be put in place at the start of each risk analysis. To be successful, the strategy for communication of risk should be an open, interactive, iterative and transparent exchange of information that may continue after a final decision is reached. The communication strategy should ensure that the parties concerned become aware of the assumptions and uncertainties in the model, model inputs and in the overall risk analysis. Peer review of the risk analyses is an essential component of risk communication in that it can provide an independent scientific critique aimed at ensuring that the data, information, methods and assumptions are the best available.

2.2 Hazard identification

Hazard identification involves identification of those aspects of the cultivation process that could potentially produce adverse consequences for the local environment. The risk assessment should be terminated if hazard identification fails to identify an increased risk of an effect associated with culture of the species.

The analysis must include a determination of whether the risk under consideration might also occur as a result of some other activity already present in the local environment. In the case of genetic interactions of wild and farmed fish, deliberate release of individuals raised in captivity with the aim of enhancing local wild populations is an example of a hazard that might carry the same or similar risks as those arising from escapes from commercial aquaculture facilities. Such events are likely to reduce the incremental change in risks arising from aquaculture escapes. At the same time one should recognize that past enhancement activities can also contribute significantly to information on the consequences of releases (deliberate or accidental) of cultured animals in that ecosystem.

2.3 Risk assessment

Preparing a risk assessment requires:

- a clear and useable statement of the question under examination utilizing a quantifiable endpoint parameter
- clear definition of terms for quantification of risk and uncertainty
- a understanding of the structure of the information required for the analysis; and,
- a predefined and clear statement of what constitutes an acceptable levels of protection for the risk in question.

2.3.1 Defining the Endpoint

Prior to initiating a risk analysis, it is very important to clearly identify the end point characteristic(parameter) to be managed for. The endpoint should be a clearly measurable parameter which is readily identifiable by those who wish to be protected by the risk analysis. In most matters of environmental governance, where it is the public is who wish to be protected not the government, those parameters usually involve the abundance, distribution or diversity of a resource. An example of an end point might be “a change in the abundance of wild salmon as a result of the abundance of sea lice on salmon farms”.

Confusion sometimes arise in the difference between predicting a change in the value of a parameter that is part of the sequence of events (abundance of sea lice on salmon farms) and that of estimating the overall probability (together with its associated uncertainty) of the actual environmental risk being expressed (the affect that the numbers of sea lice on the farm have on the abundance of wild salmon). The true end point is the abundance of the wild salmon populations, not the very contentious and often referred to abundance of sea lice that may or may not affect the abundance of the salmon. (This example is illustrated in detail in an article by McVicar (2004).)

2.3.2 Describing Risks

The terminologies associated with description of the severity of environmental changes and the probability of the changes occurring must be explicitly defined. These attributes determine the importance of the risk and the nature of the resultant management decisions and actions that are to be taken.

Terms used in the Australian *Import Risk Analysis on Non-Viable Salmonids and Non-Salmonid Marine Finfish* (AQUIS 1999) are used here to provide a template for these definitions. In that analysis, there are five categories of severity. The definition of each level of severity is determined by, three factors:

1) the degree of change experienced in the affected ecosystem or species,
2) the geographical extent of the change, and
3) the temporal duration of the change (from transient to irreversible).

Table A4.1: An example of definitions used to categorize severity of environmental change. Note: The term ecosystems refers here to water bodies of such size that water quality processes occurring in them largely function independently of the processes in adjoining water bodies. For example, a bay or estuary with relatively short water residence time would not be considered an ecosystem. In contrast, a fjord or an inland sea with a more protracted residence time might be considered an ecosystem for the purposes of these definitions.

Catastrophic:	Irreversible change to ecosystems performance at the faunal province level or the extinction of a species or rare habitat.
High:	High mortality of an affected species or significant changes in the function of an ecosystem. Effects would be expected to occur at the level of a single coastal or oceanic water body. Effects would be felt for a prolonged period after the culture activities stop (e.g. greater than the period during which the new species was cultured or 3 generations of the wild species, whichever is the lesser time period). Changes would not be amenable to control or mitigation.
Moderate:	Changes in ecosystem performance or species performance at a regional or subpopulation level that would not be expected to affect whole ecosystems. Changes associated with these risks would be reversible Change that has a moderately protracted consequence. Changes may be amenable to control or mitigation at a significant cost or their effects may be temporary.
Low:	Changes are expected to affect the environment and species at a local level and would be expected to have a negligible effect at the regional or ecosystem level. Changes that would be amenable to control or mitigation and would be of a temporary nature.
Negligible:	Changes expected to be restricted to the production site and to be of a transitory nature. Changes that are readily amenable to control or mitigation.

Attributes of the anticipated severity of change often are characterized by over more than one severity class; the overall severity is the average of the severity categories. For example, if the predicted effect is high mortality of a subpopulation of a species that would be reversible over a couple of generation, then,

- High mortality of a species is an attribute associated with HIGH severity.
- Effects at subpopulation level only is associated with MEDIUM severity and,
- the anticipated duration of a couple of generations is a MEDIUM severity characteristic.
- The final assessment of severity of the change would therefore be the average of HIGH+MEDIUM+MEDIUM, i.e. MEDIUM As will be discussed in later sections, the response to a particular severity of change is determined by three other factors; the probability of it happening, the uncertainty associated with that prediction, and the desired level of protection.

The assignment of probabilities to particular risks is a critical part of the risk analysis process. In some cases, a fully quantified approach can be taken, but in most cases knowledge of probabilities associated with each of the steps between the initial driver and the final expression of the risk will not be available. Generally, it will be necessary to adopt semi-quantified or qualitative approaches to estimation of the probability. For example, the probability of change due to enrichment of the sea bed below fish culture units in Scotland is high (based on monitoring data), but the same degree change for the same rate of organic carbon release from fish cages level in the oligotrophic areas of the Aegean Sea may be less probable (moderate to low probability).

Previous experience, scientific knowledge, and expert judgment, will be the important factors in assessing the probability of the risk being expressed. However, there will inevitably be a degree of imprecision and uncertainty in the final assigned probability. Expression of the probability of a risk being expressed can be achieved in a number of ways. These may be expressed precisely in numerical form or more qualitatively. As numerical quantification is seldom available, the definitions below (Table A4.2) are of a more qualitative nature.

Table A4.2: Qualitative definitions of levels of probability.

High:	The risk is very likely to occur.
Moderate:	The risk quite likely to be expressed.
Low:	In most cases, the risk will not be expressed
Extremely Low:	The risk is likely to be expressed only rarely
Negligible:	The probability of the risk being expressed is so small that it can be ignored in practical terms.

The number of categories used to describe the severity and probability of a risk may vary. There is nothing dictating that it should be 5; it could be more or less. The greater the number used, the more difficult it will be to clearly attribute any particular risk to a specific category. The fewer the number, the more extreme the evaluation is likely to be.

2.3.3 The structure of the information required for the analysis

For each of the environmental effects or risks, prior to undertaking an analysis, we outline what is known of the process that leads to the expression of the effect. Often we lack a complete understanding of that process. However, there is usually an understanding of many of the factors involved. To the degree possible, explicitly identify sources that act to seed the phenomenon (one of the sources will be the aquaculture activities under consideration but there may be other sources also), factors that contribute to the change (drivers), its likely geographic extent, the temporal duration of the change once the hazard has been removed, and what factors modify or prevent the change(modifiers). It is very important to identify what other human activities in the area might contribute to the expression of the same risks.

For example the structure of the data required to analyze for the endpoint “A significant decline in fitness (survival) due to genetic changes resulting from interbreeding with cultured organisms.” might look like table III.

TableA4.3: The reduction of survival due to interbreeding of wild fish and fish that have escaped from a cultivation facitity.

SOURCES	OTHER FISHFARMS IN THE AREA
	Strays from other endemic populations
	Genetic effects of Stock improvement Transfers Enhancement Genetic selection associated with fishing activities
Drivers	Proportion of wild population interbreeding with organisms escaping culture
	Relative difference in the genome of the wild and cultured fish
Modifiers	Proportion of genetic vs. environmental contribution to population differences
	the size of the wild population (its ability to avoid effects of drift and inbreeding on effective selection)
	The effects on selection by other human activities such as enhancement activities.
	Metapopulation structure of the wilf fishes
Temporal expression	Where intergradation has an effect on survival it is likely to affect the f1 and to a lesser extent the f2 generation.
	Impact beyond the f2 generation are unclear.
Geographical extent	Dependent on migratory behaviour and breeding distribution but most likely in areas adjacent to escape.
Outcomes	Reduced survival of feral population/s

2.3.4 Risk assessment steps

2.3.41 Release assessment

The release assessment describes the likelihood of the 'release' of the hazard agent under a specified set of conditions and in respect to amounts and timing. Release assessment includes a description of the pathway(s) necessary to 'release' (that is, introduce) a hazard agent (e.g. escaped fish) into a particular environment (e.g. the pathway for the escape of fish from a cage might include the boat backing into the cage, a tear occurring in the net, the fish finding the whole before employees can repair it and fish leaving the cage into an environment where con-specifics dwell.) and estimating the likelihood of that complete process occurring. It should also note how these might change as a result of various actions, events or measures.

The information in this step should be sufficiently detailed to support the consequence assessment (described in a later section).

An example of the kind of information that may be required in the release assessment for escaped fish might include:

a) Biological factors

- Species, strain or genotype, sex and age of animals,

b) Area Specific factors

- Density of culture facilities, numerical abundance in each containment unit
- Siting details such as physical conditions of the site and details of the nature of the installation being considered.

- Evaluation of surveillance and control programs, and zoning systems of local authorities.
- Potential location of releases due to transport, culture and treatment,

c) Species specific factors

- Schooling behavior,
- Exploratory behavior,
- Jumping behavior
- Rubbing or nibbling behavior
- Effect of handling behavior (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 probability of release, the risk analysis should be terminated.

2.3.4.2 Exposure assessment

Exposure assessment estimates the likelihood of exposure(s) occurring, and of the spread or establishment of the hazard agent. It describes the pathway(s) necessary for the hazard agent to be exposed to the resource of concern (e.g. wild fish). For example it might include the navigation of the escaped fish to a breeding ground at the correct time of year and the interaction with individuals of the wild population.

The information in this step should be sufficiently detailed to link to the information in the release assessment and support the consequence assessment (described in a later section).

The likelihood of exposure 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 for evaluation the risk of wild and escaped fish interbreeding are:

a) Biological factors

- Genotype of con-specifics
- Properties of the cultured fish that would affect interbreeding (e.g. mate preference, timing of spawning, survival to spawning.
- 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, predators such as seals, the 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 con-specific strains,
- Waste disposal practices.
- If the exposure assessment demonstrates no significant risk, the risk analysis should be terminated at this point.

2.3.4.3 Consequence assessment

Consequence assessment consists of identifying the potential biological consequences of a release of a hazard agent into the environment. A causal process must exist by which exposures to the agent results in undesirable changes. In the case of the genetic interactions of farmed and wild fish, the links between the various stages or processes leading from the release of the cultured fish to the potential measurable outcome (reduced survival of wild fish) form a risk pathway. This can be expressed as a logic model which lists the stages or processes involved as a series of steps. For example, a logic model for genetic interactions of farmed cod with wild cod stocks could comprise the following:

Process of concern: Changes in fitness of wild populations of cod due to genetic intergradation

End Point of Concern: Significant decline in survival in wild cod populations due to interbreeding with escaped cultured cod.

Logic model steps:

1) Cod farms are established in coastal waters.
2) There are phenotypic differences between the wild and cultured cod populations.
3) These phenotypic differences arise primarily for genetic rather than environmental reasons.
4) The primary route for genetic interaction (interbreeding) between cultured and wild cod is through escapes of cod from cages.
5) Cultured cod escape from cages.
6) Cultured cod interbreed with wild cod.
7) The progeny of this interbreeding (hybrids) show reduced fitness.
8) Sufficient gene flow to affect survival rates of cod in individual fisheries management units, *i.e.* the population structure of wild cod is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.
9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), *i.e.* Genetic interaction between wild and populations of escaped cultured cod causes significant declines in survival in wild cod populations.
10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, *i.e.* Escapes of cultured cod cause significant decreases in wild/feral cod stocks.

For each of these steps, consequence assessment evaluates three attributes; severity/intensity of the step if it occurs, probability of occurrence, and the level of uncertainty in the prediction of its occurrence. Assessment of the severity/intensity incorporates three aspects; the degree of change, the geographical extent of the expression of the risk, the duration of the effect.

2.3.4.4 Risk estimation

Risk estimation consists of integrating the results of the release assessment, exposure assessment, and consequence assessment with those unique attributes of the proposal/development and local environmental management systems that might affect the probability of the expression of the risk 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, of a specific hazard under a specific set of regulatory and management practices, from hazard identified to unwanted outcome.

A common feature of risk analysis is that the analyst finds that all the information desired for the analysis may not exist. Under such situations, the analysis is based on the available information combined with expert opinion and/or experience in other similar situations.

A qualitative risk assessment should always be performed. Quantitative assessments can then be used to add precision to the outcome of the qualitative assessment. An example of an output format is shown in Table A4.4.

Table A4.4: Summary of the output from the Risk Assessment stage of a Risk Analysis of escapes of farmed cod interbreeding with wild cod, leading to significant decline in survival in wild cod populations.

STEPS IN THE LOGIC MODEL	SEVERITY/ INTENSITY (C,H,M, L, OR N) [1]	PROBABILITY (H,M,L,EL, OR N) [2]	UNCERTAINTY (H,M, OR L)
Step 1	H	M	L
Step 2	H	H	M
Step 3	M	H	M
Step 4	H	H	L
Step 5	H	M	L
Step 6	H	M	M
Step 7	L	M	M
Step 8	H	L	H
Step 9	L	L	M
Step 10	L	L	L
Final Rating [4]	L[5]	L[5]	H[6]

Explanatory notes:

1 Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

2 Severity = C – very intense, H – high, M – Moderate, L – Low, N – Negligible There are three components of severity that should be commented on: the duration of the activity, the degree of change, and the geographic extent of the change.

3 Uncertainty = H- Highly certain, M – Moderately certain, L – Low Uncertain

4 The final rating for the Probability is assigned the value of the element with the lowest level of probability.

5 The final rating for the Severity (intensity of interaction) is assigned the value of the step with the lowest risk rating (e.g., Medium and Low estimates for the logic model steps would result in an overall Low rating). The final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

6 The final rating for the Uncertainty is assigned the value of the element with the most uncertainty level (i.e. the least certainty).

Because of its more demanding nature, quantitative analysis is necessarily more focused and may deliver a more precise assessment, but potentially of no greater accuracy than a qualitative assessment. In contrast qualitative analyses tend to be more inclusive of factors that are difficult to quantify and that tend to improve the accuracy of the prediction.

For a quantitative assessment, the final outputs may also include:

- Definitions of the various populations of aquatic animals likely to be affected by intergradation 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.

2.4 Risk management components

Once the Risk Evaluation has been completed, it is possible to undertake a process of risk management. The first stage of this is evaluation of the risk management options available. This process includes identifying, evaluating the efficacy and feasibility of possible measures to reduce the risk or uncertainty associated with cultivation of the species under consideration. The efficacy is the degree to which an option reduces the likelihood and/or magnitude of adverse 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 of feasibility normally focuses on technical, operational and economic factors affecting the implementation of the risk management options.

Option evaluation addresses what might be done to reduce the probability of a risk being expressed or the uncertainty in the prediction of the expression of a risk. This can be addressed through a review the logic model and the identification, for each step, of what could be done to reduce the probability of that step occurring. These actions would directly mitigate possible effects. A further contribution to increasing the effectiveness of the risk analysis would be to reduce the uncertainty associated with predicting that the step will happen. Usually this involves further research or development. Table A4.5 below identifies both mitigative and research or development steps that could be in addressing risks associated with genetic interactions arising from cod culture.

Table A4.5: An example of a table of risk management options that might be derived from the evaluation of the risks of cultured cod escaping confinement and breeding with wild cod.

Step	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Uncertainty	Research/Development
1	Cod farms are established in coastal waters	M	Where feasible move to land- based production	L	Develop economically competitive land-based technologies.
2	There are phenotypic differences between the wild and cultured cod populations.	H	For each generation recruit all grow-out stock from juveniles captured in the wild	M	
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.	H		M	Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival.
4	The primary route for genetic interaction (interbreeding) between cultured and wild cod is through escapes of cod from cages.	H	Recovery plan for escaped fish	L	Identify factors that will limit dispersion of escapees
5	Cultured cod escape from culture	M	Improve containment design and/or build in fail-safe measures	L	

STEP	LOGIC MODEL STEP	PROBABILITY	MITIGATION (REGULATE/DESIGN/ MODIFIED PRACTICES)	UNCERTAINTY	RESEARCH/DEVELOPMENT
6	Cultured cod interbreed with wild cod	M	Use of sterile fish	M	
7	The progeny of this interbreeding (hybrids) show reduced fitness	M	For each generation recruit all grow-out stock from juveniles captured in the wild	M	
8	Sufficient gene flow to affect survival rates of cod in individual fisheries management units, *i.e.* the population structure of wild cod is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.	L	Limit the distribution of cod farming to either proximity to small value stocks or very large stocks.	H	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), *i.e.* Genetic interaction between wild and populations of escaped cultured cod causes significant declines in survival in wild cod populations.	L		M	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , *i.e.* Escapes of cultured cod cause significant decreases in wild/feral cod stocks	L	Limit the distribution of cod farming in relation to the distribution of the species or meta population	L	Identify dynamics of genome at the meta population or species level.

It is important that the implementation of risk management actions, (*i.e.* the process of following through with the risk management decisions and ensuring that the risk management measures are in place) are accompanied by a planned process of monitoring and review. By this process, the risk management measures are continuously audited to ensure that they are achieving the results intended, and to allow establishment of an action cycle in which management actions can be reviewed and amended.

2.5 Levels of Protection and the Precautionary Approach

Prior to initiating a risk analysis, it is important to develop an explicit enunciation of what constitutes an acceptable level of protection for the identified risk. Acceptable level of protection (ALOP) will vary, as jurisdictions differ in the level of risk that they are willing to accept, depending on their social and economic conditions. Within the context of justification for trade restrictions, this is likely to be internationally acceptable as long as the restrictions are equally applied to all traders whether the goods and services in trade are created within the jurisdiction or externally and exported into the jurisdiction. In national or more local regulatory contexts, it implies that regulators can be explicit in the standards that they adopt, and deliver transparent and consistent decisions.

Based on the severity and probability of a risk being expressed, an explicit table for making decisions can be constructed (Table A4.6) that provides working guidelines on the acceptable level of protection against the occurrence of an event at risk. Such a table might be used to assist resource managers to decide if a license should be issued to operate a farm in a certain location (Accept) or not (Reject).

Table A4.6: An example of a table illustrating the relationship of management decisions and the combination of the severity of a risk and the probability of it being expressed.

		Severity				
		C	H	M	L	N
Probability	H	Reject	Reject	Reject	**Accept**	Accept
	M	Reject	Reject	**Accept**	Accept	Accept
	L	Reject	**Accept**	Accept	Accept	Accept
	EL	**Accept**	Accept	Accept	Accept	Accept
	N	Accept	Accept	Accept	Accept	Accept

Severity = C – Catastrophic, H – high, M – Moderate, L – Low, N – Negligible

Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

Reject = e.g. Reject a request for a permit to undertake the proposed activity

Accept = e.g. Accept the risks associated with permitting the culture to be undertaken

In recent years, the precautionary principle has emerged as a popular approach to deal with uncertainty in science-based decision making. Article 15 of the United Nations 1992 RIO Conference on Environment and Development defined the precautionary principle as that a "lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation". While the broad sentiment behind the statement is generally agreed upon, the principle has never been accepted as a general principle of international law. A number of factors appear to contribute to this reticence. The precision of the definition has been problematic. Ronald Doerling, former Vice President of the Canadian Food Inspection Agency, illustrated several of these in an invited plenary speech at Aquaculture 2004 in Montreal, Canada. Among his comments, he pointed out that working interpretations of the principle varied significantly, and that the Swedish philosopher Sandin documented no less than 19 variations in the principle's definition in laws, treaties and academic writings. The versions differed in the interpretation of how scientific uncertainty was evaluated, how severity of consequences is considered, how the costs and risks are to be balanced and, from a legal perspective, how the onus shifts to the proponent to prove (if that is ever possible) that the process or product is safe.

Table A4.6 combines severity and probability to derive consistent and transparent decisions. However, the table does not take account of uncertainty. A probability that has high uncertainty indicates that the true expression of the risk may differ from the assigned assessment. A risk assessed as of low probability of occurrence, with a high degree of uncertainty, may actually have an extremely low or moderate probability of occurring. The precautionary principle indicates that such uncertainty should be taken into account in the assessment and decision-making processes. In terms of Table A4.6, any risk probability with a high degree of uncertainty should, as a consequence, be considered as equivalent to an assessment of a higher probability of occurrence. Consequently, the acceptable levels of risk identified in Table A4.6 can then be modified as shown in Table A4.7 and are shown in bold where a higher degree of uncertainty has changed a decision from one of accept to one of reject.

Table A4.7: Table A4.6 adjusted to allow for high uncertainty in the assessment of the probability of change. Differences from Table A4.6 are shown in bold.

		Severity				
Probability		C	H	M	L	N
	H	Reject	Reject	Reject	**Reject**	Accept
	M	Reject	Reject	**Reject**	Accept	Accept
	L	Reject	**Reject**	Accept	Accept	Accept
	EL	**Reject**	Accept	Accept	Accept	Accept
	N	Accept	Accept	Accept	Accept	Accept

Risk analysis does not overcome all the difficulties in the definition and application of the precautionary principle, but it does make the assumptions and value judgments much clearer and explicit. If, however, definitions and an explicit elucidation of what constitutes an acceptable level of protection are not well made, the uncertainties and misuse that can be associated with the application of the precautionary principle also become a threat to the objectivity attainable through risk analysis.

Before undertaking any risk analysis it is important that the regulators who will apply the results of the analysis define *a priori* and explicitly what is their acceptable level of protection for wild stocks, 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 ultimate value of the analysis.

Annex 5: Case studies examining the potential impacts of escaped non-salmonid farmed fish (cod, sea bass, sea bream, halibut, turbot)

Risk analysis of the potential interbreeding of wild and escaped farmed cod (*Gadus morhua)*

Introduction

The cod (*Gadus morhua)*) is a benthopelagic species found on both the eastern and western sides of the North Atlantic from Greenland and the Barents Sea south to Cape Hatteras and the Bay of Biscay. It feeds on both invertebrates and fish. The maximum reported age is around 25 years, with males reaching 200 cm and 96 kg, although large specimens are now rare. Cod has a long tradition as an important commercial species, enormous stocks having existed in the past in areas such as the Grand Banks. Stocks in the North east Atlantic are also currently at a low level, and measures are being taken to attempt to restore them. The traditional popularity of cod for human consumption, low stock levels, and the fishs' high growth rate made the species an attractive target for aquaculture development.

Risk analysis

Hazard Identification

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 described below, 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 series of 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 (Jørstad, 1994).

During the same series of experiments, 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.

Similarly, 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 behavior of newly released cod is poorer than wild conspecifics, they adopt similar feeding behavior to wild fish within five months after release.

Svåsand *et al.* (2000) reviewed these studies of 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 (*Trisopterus minutus*) 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, and can contribute to assessment of the 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.

Risk Assessment

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 A5.1, intensive fry production is the dominant production method

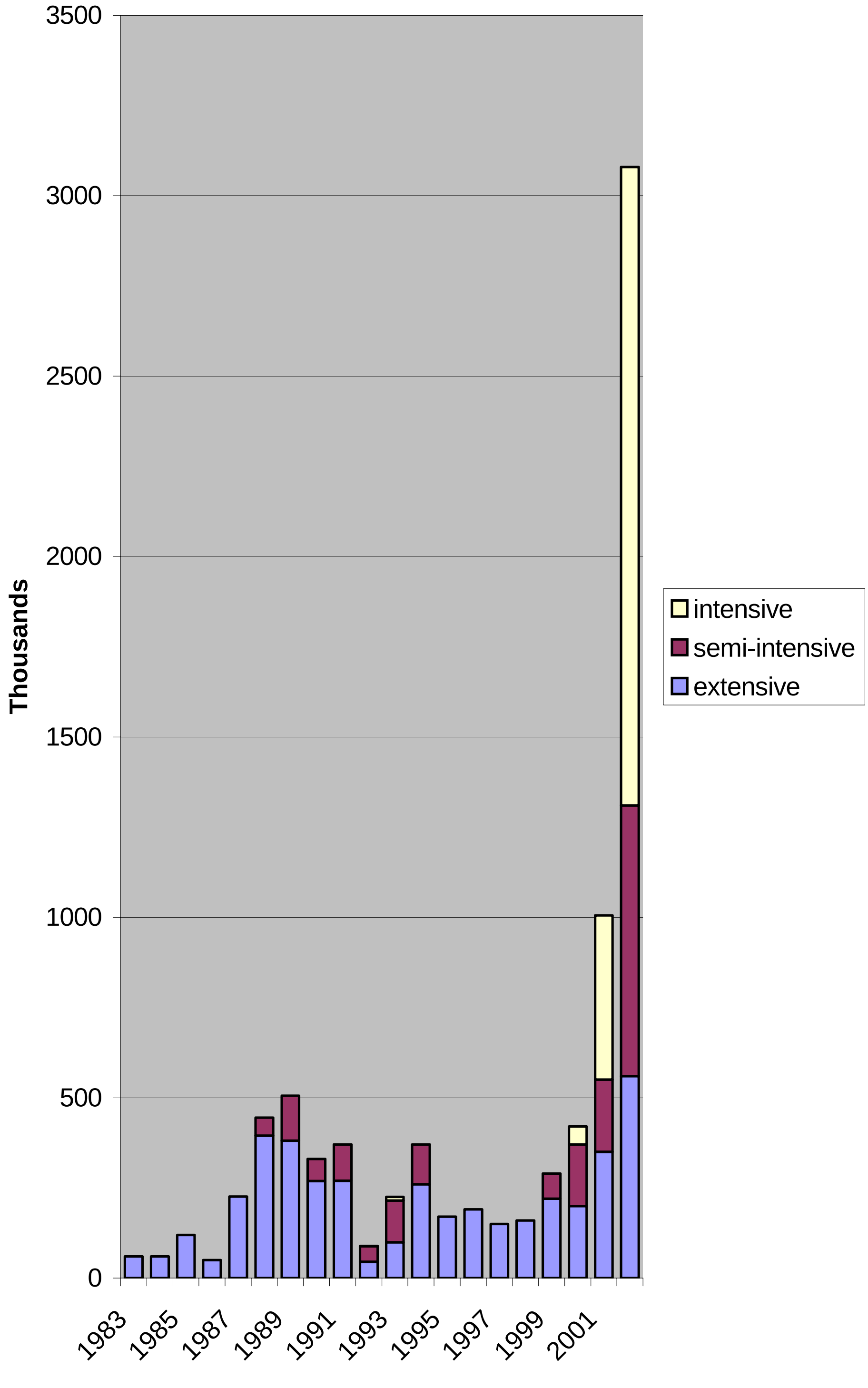


Figure A5.1: Total production of cod fry in Norway 1983 –2002 (Karlsen and Adoff, 2003).

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 350 tons of production is predicted in 2005. 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 as a preliminary step to develop an industry.

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, 20 out of 483 registered farms have multi-species licenses and therefore have the potential to stock and produce cod. No aquaculture licenses for cod have yet been issued in Ireland, although several applications are being evaluated. Cod reared in pump ashore facilities, particularly those employing treatment of discharge water (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 1 m 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 past five years, there have been 20–25 escapes per year from Scottish fish farms, mostly from Atlantic salmon farms. Rates of escapes of salmon from saltwater sites in Scotland have been between 76 000 and 411 000 growing fish (1–4 kg) per year (Table A5.1).

Table A5.1: Numbers of salmon smolts put into salt water on-growing units, and numbers of escapees for 1999–2004. The percentages are calculated from the smolt inputs in single years. As the production cycle is approximately 2 years, the escape rates expressed against the total fish numbers in cultivation will be approximately one half of the percentages in this table.

	Number (millions) of salmon smolts put to sea	Numbers escaped in salt water (thousands)	% Escapees
1999	41.1	257	0.63
2000	45.2	411	0.91
2001	48.6	76	0.16
2002	50.1	376	0.75
2003	43.8	104	0.24
2004	38.1	83	0.22

This table suggests that the rate of escape is around 0.1–0.5% of the total number of individuals in cultivation. If this rate is also applicable to cod, this suggests an escape rate of between 20 000 and 150 000 cod per annum at an annual input to on-growing of around 10 000 000 juveniles per annum.

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), particularly with the use of double nets;
- 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

Exposure Assessment

Life History of Wild Populations

Distribution and movements

Much of the information in this section refers to populations around Scotland or the UK, or the North Sea area. It is not the purpose of this paper to undertake a thorough review of cod throughout its range. It is considered that the principles and patterns established in this limited area are broadly applicable to cod in other areas, for example off the Norwegian or Canadian coasts. Cod in the North Atlantic appear to have a number of subpopulations. Studies suggest that, off the coasts of Canada, Iceland and Norway, cod have differentiated in to a number of subpopulations (Imsland and Jonsdottir 2003). Ruzzante *et al.* 1996 has demonstrated in Canadian waters there is genetic differentiation between onshore and offshore populations of cod off the coast of Newfoundland. Later work by Ruzzant (1998) suggested when inshore and offshore populations were considered as many as 14 subpopulations may exist. In the Eastern North Atlantic Neilsen *et al.* 2001 identified 3 distinct subpopulations (North East Artic Ocean, North Sea and Baltic). In the North Sea recent microsatellite DNA Hutchinson *et al* (2001) suggests that there may be 4 distinct subpopulations. The amount of information supporting 4 rather than the traditional 3 subpopulations is limited and but an FP5 project (METACOD) is investigating this issue.

Clearly the precise number of genetically differentiated populations is an ongoing discussion. Smedbole and Wroblewski (2002) have framed the discussion of cod population differentiation in terms of metapopulations. A metapopulation is composed of set of local subpopulations. The degree of genetic differentiation among subpopulations may range from slight to almost complete isolation. The spatial patterning of subpopulations within a metapopulation is temporally dynamic; subpopulations may undergo extinction and recolonization, and new subpopulations may develop. Extinction, recolonization and differentiation of subpopulations will be affected by abundance in the metapopulation and recent studies (Beamish, In Press, Transactions American Fisheries Society) suggests that oceanic regime shifts may have as large an affect on population abundance of marine fishes as fishing pressure.

Given the above complexity some assumptions must be made about the structure of cod populations. Currently, the main areas where the aquaculture industry is actively engaged in seeking to develop cod farming are Canada, Scotland, Norway and Ireland. Imsland and Jonsdottir (2003) identify groupings of spawning areas of the east coast of Alantic Canada, Scotland and Ireland as well as of the North coast of Scotland. Gene flow between populations is generally expected to be highest between populations whose spawning areas are closer together. These spawning area aggregations may thus form the basis for metapopulations with subpopulations derived and maintained by individual spawning areas within an aggregation of spawning areas. On this basis for the duration of this analysis population structure will be assumed to be composed of separate populations in the North West Atlantic, Ireland, Scotland and Norway (North of Stavanger).

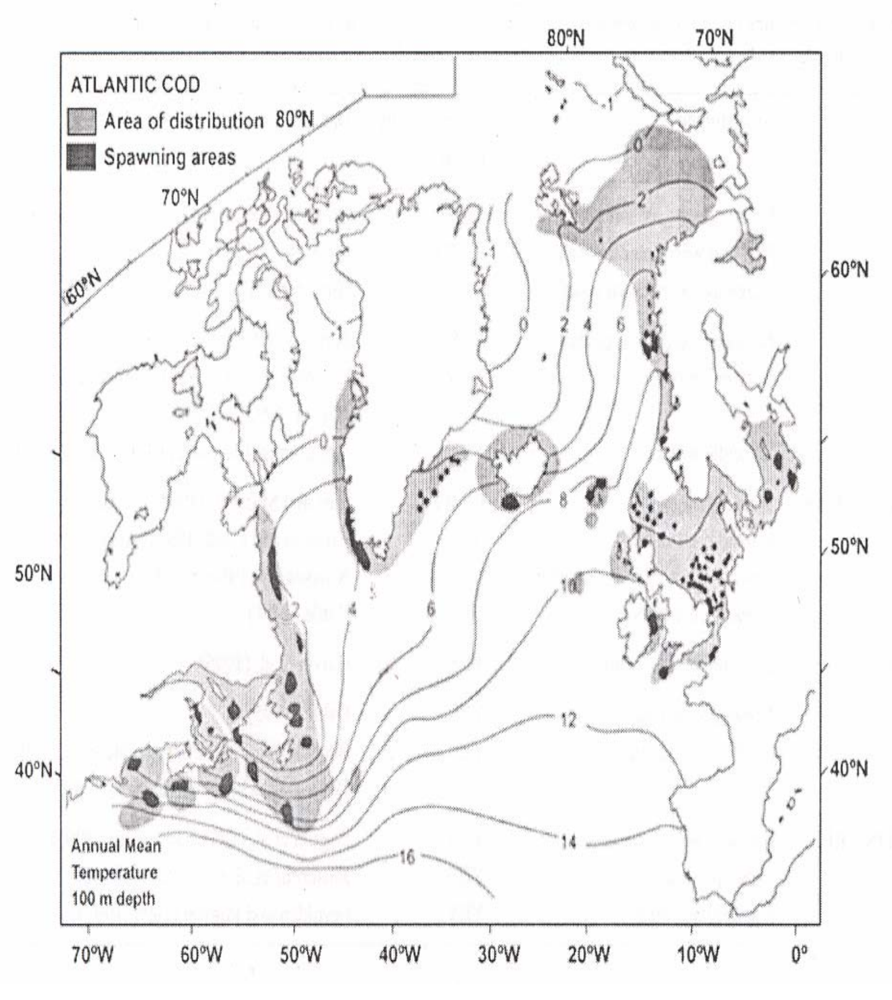


Figure A5.2: Cod distribution and spawning areas (After Imsland and Jonsdottir 2003).

Off Scotland in the 1970s during spawning season (January–April) cod eggs and larvae were found through out the west and north coasts. By the 1990s, this area had diminished to areas off the west coast of the Western Isles and the northern North Sea (Heath *et al.* 1994). In this area juvenile cod during their first year are close inshore or around the mouths of sea lochs and fjords. 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.

East of the UK after hatching at a length of about 0.4 cm length 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 have identified regional populations of cod Moray Firth, at Flamborough Head, in the German Bight, in the Southern Bight and in the English Channel that separate during the spawning season and, in some cases, inter-mix during the feeding season. The Clyde has also been identified as a preferred area for juvenile cod. From the evidence of NW Atlantic stocks we might expect that the different reproductive units might intermix to some extent during the summer.

There is some understanding of the movements of cod to the west of Scotland. As elsewhere, 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. 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 in the NW Atlantic cod migrate along clines of preferred ambient temperatures (Rose 1993). Some coastal aggregations of cod appear to have very limited migration and these are most likely to be the most sensitive to interactions with farmed stocks. Cod reach maturity at 2 years and on the west coast can also spawn at this age. Although maturity at age varies by region, all cod are spawning by 6 years of age. 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. Tagging studies carried out over several decades have also shown that the maximum distance traveled 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.

Tagging data from Scotland 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).

The above discussion has mainly concentrated on cod stocks round the United Kingdom. The available information indicates a degree of uncertainty in understanding of migration and other behaviour, and the existence of structure within the overall population to the east of the UK. Although published information is lacking, it is likely that cod populations in other areas relevant to aquaculture, such as the west of Norway and Scotland, and east of Canada will show a similar degree of differentiation. They will probably show similar sensitivity to integradation with farmed stocks.

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:

Table A5.2.

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. Since 1980, the fishing mortality on North Sea stocks has been around 1.0, although it has varied rather more since 2000 (0.5–1.2) at a time when stocks have been reduced to such a level that productivity is impaired, and a formal stock recovery plan has been introduced at EU level (ICES, 2005).

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.

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).

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 54 400 tons, with 168 000 000 recruits at age 1. The most recent complete data on numbers of individuals present in the North Sea/Skaggerak/E Channel stock assessment area are for January 1 2003 (Table A5.3).

Table A5.3: ICES estimates of numbers of cod at age in the North Sea, Skaggerak and E Channel combined, 1 January 2003.

AGE	NUMBER OF INDIVIDUALS
1	50 037 000
2	63 059 000
3	14 034 000
4	13 234 000
5	1 542 000
6	260 000
7	122 000

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). Around Iceland, there has been low spawning stock biomass and weak recruitment since the mid-1980s.

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. Egg, 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, 2002a,b).

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).

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 (2002a,b) 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.

Synthesis

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.

Consequence Assessment

Accidental release of fish from culture sites may engender change in local populations by affecting wild cod at all life-cycle stages including exposing wild fish to feed competition and behavioral stresses. Behavioral stress will be particularly intense when territorial competition is a key component controlling population density in a given habitat. If the decline in abundance of cod during the 1900s is primarily a result of fisheries, it would seem likely that food and habitat resources are unlikely to limit survival of cod. This conclusion has been reiterated by Blaxter in a recent (2000) 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”.

The genetic effect of escapes will likely be minimal if farming involves the rearing of wild-caught juveniles from a local stock which was widespread and abundant, and which has a regional rather than local population structure. On the other hand, a significant change may occur if farmed fish are of non-local origin with low genetic diversity (e.g., from small populations with a high degree of inbreeding), and mix with a population that is not differentiated into a series of distinct local populations. 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 into 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 these differences will be, and to what degree will they impact wild populations when cultured and wild populations interact. Experience with cod culture (as an enhancement activity) dates back to the middle of 1800s however, actual investigations into the differences between wild and cultured cod are primarily from more studies in the 1980s–1990s. Our knowledge of the differences is further 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 hybridization involving escaped farmed cod is not thought to be a problem (FRS, unpub.). An extensive e-journals literature search found no reference to any literature on cod hybridization. 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) differential 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 is likely to create 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 that these escaped fish will contribute to the recovery of wild cod stocks will need to be investigated more seriously. 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 behavior, 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 outcompeting 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. A similar problem of dispersion of a time-limited viable agent is dealt with in management of diseases on fish farms in Scotland. There the criterion used is the predicted dispersion of an agent over a tidal cycle (12 hrs). That has been translated to a “rule of thumb” of a 5 km separation distance between groups of farms in a disease management unit. A similar approach might be useful to separate farms from cod breeding areas.

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.

Logic model

The series of steps and processes leading from the establishment of cod farms in coastal waters to significant decreases in wild cod stocks as a result of genetic interactions between the two groups of cod can be summarized in a logic model as below:

Process of concern:	Changes in fitness of wild populations of cod due to genetic intergradation
End Point of Concern:	Significant decline in survival in wild cod populations due to interbreeding with escaped cultured cod.

Logic model steps:

1) Cod farms are established in coastal waters.

2) There are phenotypic differences between the wild and cultured cod populations.

3) These phenotypic differences arise primarily for genetic rather than environmental causes.

4) The primary route for genetic interaction (interbreeding) between cultured and wild cod is through escapes of cod from cages.

5) Cultured cod escape from cages.

6) Cultured cod interbreed with wild cod.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

8) Sufficient gene flow to affect survival rates of cod in individual fisheries management units, i.e. the population structure of wild cod is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured cod causes significant declines in survival in wild cod populations.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e escapes of cultured cod cause significant decreases in wild/feral cod stocks.

The information presented in the preceding sections of this risk analysis allows annotation of each step in the logic model to indicate the likelyhood that each step has been, or will be, completed.

1) Cod farms are established in coastal waters.

Highly probable. - Cod farms are already established in Norway and Scotland. Considerable growth in production is planned for the coming years. Where active cod farms will tend to aggregate, as salmon farming has done, it is likely that the density of the farms will, over time, move from low to high but occupy only part of the wild cod's coastal habitat. Once in place, the farms tend to become a long term feature of the coastal environment, however they can be moved or removed. For this evaluation, the intensity in this step is considered to be moderate. The uncertainty associated with this prediction is low, as development has already been initiated.

2) There are phenotypic differences between the wild and cultured cod populations.

Highly probable – (particularly as the industry continues to develope). Present practice throughout the industry is to use mainly wild-caught mature adults as broodstock. This will tend to mitigate against genetic differences between wild and farmed fish. There is evidence from restocking experiments that cultured juveniles can be selected to show phenotypic differences from the wild stocks. It is likely that cod cultivation systems will progressively become independent of input of genetic material from wild populations. This will make it easier to select broodstock for desirable genetically determined phenotypic traits (intentional or otherwise) desireable for cultured fishes. Similarly, start-up hatcheries in the EU tend to obtain fish or eggs from other disease-free hatcheries. In both cases, over time, greater differences are then likely to develop between farmed and wild stocks. To allow for this, the intensity of this step will be high. The uncertainty is considered to be moderate.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

Moderately likely - (Particularly as the industry continues to develope.) As indicated above, the current use of wild-caught broodstock will limit the potential for such differences to develop, but differentiation will be favoured by a move to culturing cod through their entire lifecycle using cultured broodstock. Over time as culture activities rely less on wild broodstock, the differences will, to a greater extent, be genetic as aquaculture will experience greater reliance on genetics to differentiate populations in a increasingly homogeneous environment supplied by the farmer. The intensity of this differentiation will likely be high and this is likely to be an ongoing feature of cod husbandry throughout the industry. There is moderate uncertainty in this evaluation.

4) The primary route for genetic interaction (interbreeding) between cultured and wild cod is through escapes of cod from cages. .

Highly probable. - It is possible for farmed cod to release eggs or milt inside cages, and for these products to interact with eggs or milt from wild fish outside the cages. Current commercial preferences are selecting against animals that would mature while in the grow out cages. At present the probability of interaction by escaped fish compared to that occuring by dispersion of gametes from a cage is high. It is therefore likely that escapes will be a more significant route for interaction. This is type of interaction is likely to occur over a restricted portion of the wild populations range but will probably continue for the forseeable future. The likely intensity of this step is high. There is low uncertainty in this evaluation.

5) Cultured cod escape from cages.

Highly probable. - It is highly likely that some cod will escape from cages. Experience with other species indicates that accidents happen. Data for the period 1999 – 2004 suggest an escape rate of 0.1 – 0.5% of salmon from cages in Scotland. Loss of fish from the cage will be strongly avoided so escape rates are likely to remain at least as low as that of salmon farming (0.5-1.0%) and while the number in culture will grow for the immediate future (5 years) the likely number of escapes will be small relative to the abundance of wild fishes. Escapes will be a general feature of the industry but will be highly selected against. Intensity of this is considered moderate. There is low uncertainty in this assessment.

6) Cultured cod interbreed with wild cod.

Highly probable. - Studies of released juvenile cod in fjord environments in Norway have detected no differences in behaviour (migration patterns) between released and wild cod. As the cultured fish genetically differentiate from wild stock, this may result in reduced competence to follow the wild fish to the breeding ground or to maintain the necessary spawning behavior. This is likely to be a general feature across the industry. Intensity of the interaction is considered moderate. Uncertainty for this prediction is moderate.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

Low probability. - There is no evidence to support this contention for cod. Reduced fitness of the progeny from interbreeding has been shown in salmon thus it is anticipated that with increased independence from wild stock this may happen in cod. With unlikely differences between wild and cultured fish, the immediate severity of reduced hybrid survivorship is low. As with the probability, this may increase over time. The intensity for this step is moderate. Our uncertainty in this prediction is moderate.

8) Sufficient gene flow to affect survival rates of cod in individual fisheries management units, (i.e. the population structure of wild cod is such that the rate of interbreeding is sufficient to affect population fitness, at managed stock level.)

Highly likely for very localized small stocks. - Knowledge of the detailed population structure of cod in the Atlantic is incomplete. Currently, stock management is based on large geographical areas (eg North Sea, Skaggerak and Channel combined). However, there are suggestions that inshore populations in some fjords or sealochs may be to some degree distinct from the more open sea populations. If this is true, the small size of individual inshore populations may mean that sufficient escapees may be available to significantly change the feral population genome. At present, the concentration of farms is such the gene flow into these populations is likely to be intermittent, quantitatively small and the genetic differences small so the interaction will be of low intensity (though with an increased number of farms in an area this would increase). This has been shown to occur for salmon (McGinnity, 2004) and brown trout (Skara, ref.) This prediction is highly uncertian because of lack of ability to predict the number of farms in an area.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), (i.e. Genetic interaction between wild and populations of escaped cultured cod causes significant declines in survival in wild cod populations.)

For the same reasons as in step 8, the probability is low. The severity is likely to remain low as the effect is spread over a greater number of fish and the uncertainty of this prediction decreases.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e Escapes of cultured cod cause significant decreases in wild/feral cod stocks.

For the same reasons as in step 8, the probability is low. The severity is likely to be low and, because the effect is spread over an even greater number of fish, the uncertainty is decreased.

Table A5.4.

Steps in the logic model	Intensity (C,H,M,L, or N) [1]	Probability (H,M,L,EL, or N) [2]	Uncertainty (H,M, or L)
Step 1	M	H	L
Step 2	H	H	M
Step 3	H	M	M
Step 4	H	H	L
Step 5	M	H	L
Step 6	M	H	M
Step 7	M	L	M
Step 8	L	H	H
Step 9	L	L	M
Step 10	L	L	L
Final Rating [4]	L	L	H

Explanatory notes:

Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

Severity = C – very intense, H – high, M – Moderate, L – Low, N – Negligible There are three components of severity that should be commented on: the duration of the activity, the degree of change, and the geographic extent of the change.

Uncertainty = H- Highly certain, M – Moderately certain, L – Low Uncertainty

The final rating for the Probability is assigned the value of the element with the lowest level of probability.

The final rating for the Severity (intensity of interaction) is assigned the value of the step with the lowest risk rating (e.g., Medium and Low estimates for the logic model steps would result in an overall Low rating). The final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

The final rating for the Uncertainty is assigned the value of the element with the highest uncertainty level (i.e. the least certainty).

Risk Evaluation

Without regulations or farm management practices specific to cod farms there is unlikely to be any difference between the outcome of the Consequence Assessment and that of the Risk Evaluation. Risk management may be able to alter the values in Table A5.4.

Risk Management

Option evaluation in risk management addresses what might be done to reduce the probability of a risk being expresses, or to reduce the uncertainty in the prediction of the expression of a risk. This can be addresses through consideration of the series of steps in the logic model discussed above. The process identifies, for each step, what could be done to reduce the probability of it occurring. These actions would directly mitigate possible effects. A further contribution to increasing the effectiveness of the risk analysis would be to reduce the uncertainty associated with predicting that the step will happen. Usually this involves further research or development. The table below identifies both mitigation and research or development steps that could be in addressing risks associated with genetic interactions arising from cod culture.

Table A5.5.

	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Uncertainty	Research/Development
1	Cod farms are established in coastal waters	H	Where feasible move to land-based production	L	Develop economically competitive land-based technologies.
2	There are phenotypic differences between the wild and cultured cod populations.	H	For each generation recruit all grow-out stock from juveniles captured in the wild, preferably from local and abundant stocks	M	Determine local and regional stock structure.
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.	M		M	Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival.
4	The primary route for genetic interaction (interbreeding) between cultured and wild cod is through escapes of cod from cages.	H		L	Identify factors that will limit dispersion of escapees
5	Cultured cod escape from culture	H	Improve containment design and/or build in fail-safe measures Recovery plan for escaped fish	L	Improve contingency plans for recapture, possibly including prior imprinting, e.g. of prey (pellets)
6	Cultured cod interbreed with wild cod	H	Use of sterile fish Harvest fish before maturity	M	Improve methods of producing sterile fish
7	The progeny of this interbreeding (hybrids) show reduced fitness	L	For each generation recruit all grow-out stock from juveniles captured in the wild	M	Develop models of the impact of interbreeding on fitness.
8	Sufficient gene flow to affect survival rates of cod in individual fisheries management units, i.e. the population structure of wild cod is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.	H	Limit the distribution of cod farming to either proximity to small value stocks or very large stocks.	H	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured cod causes significant declines in survival in wild cod populations.	L		M	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e. Escapes of cultured cod cause significant decreases in wild/feral cod stocks	L	Limit the distribution of cod farming in relation to the distribution of the species or meta population	L	Identify dynamics of genome at the meta population or species level.

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 local 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 low efficiency and market acceptability 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 shooling 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.

Risk Mitigation

As indicated in the risk management table above two broad approaches can be taken to manage risk. The first is direct mitigation which generally reduces the likelihood of a step in the logic model being fully realized. These mitigation measures usually take the form of regulatory strictures such as moving culture to land based facilities or codes of practice utilized by industry. As can be seen under the column labeled mitigation most of these options can be put in place using regulatory or code of practice mechanisms. Some, such as the requirement for geographic limits to the culture of cod (mitigation for logic model step 8) may necessitate a wide planning process.

Where a regulatory approach is taken, care must be given to ensure only those regulatory measures necessary to reduce the level of to an acceptable level of protection are taken. Regulation for an extreme level of protection, where not required, is contrary to the concept of sustainable development. Suggestions such as moving marine culture to land based facilities (mitigation for logic model step 1) should be considered carefully in this context.

The other approach is to managing risk is to reduce sources of high or moderate uncertainty. In this context one of the advantages of risk analysis is that it can assist in identifying priorities for research and development work. For example step 8 in the logic model is associated with a high degree of uncertainty. That uncertainty in the decision making process could be reduced by research that defines gene flow between wild populations which could do much to clarify where specific populations may be at risk due to a low rate of gene flow with other components of the meta population.

Testable models can be useful in the development of knowledge as well as being of immediate assistance to decision makers faced with uncertainty. For example in step 7 and 8 there is a moderate or high level of uncertainty about the effect of hybrids on wild stocks. Lacroix *et al.* (1998) show modeling 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.

It is important to be inclusive as possible in considering how to control risk. Some control of risk may not directly involve the hazard under consideration (e.g. cod farming). For example step 8 talks of sufficient gene flow. That will in part be determined by the relative size of the populations of wild and cultured fishes.

The maintenance of sufficient wild populations (managing fisheries pressure, enhancement, etc.) may be an efficient tool to mitigate the effects of increasing quantities of released aquaculture individuals. That is no mean feat as generations of fisheries managers will tell anyone who will listen. Even so, there is evidence (Rice and Cooper 2004) that strict adherence to advice from fisheries scientists increases the likleyhood of success in this objective. So health wild stocks in and of themselves help limit the effect of interbreeding.

Combinations of regulatory and developmental research can be very powerful approach to mitigation. The critical event of cod escaping containment (Step 5) is very responsive to such an approach. 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. And when economically feasible their use can be encouraged by codes of practice or regulatory tools.

Risk Communication

Risk communication can do much to enhance this and any other risk analysis process. It is first and foremost a dialogue between resource mangers, the public and industry.

The geographic location of cod farms (proposed and existing) and cod breeding stocks should be a matter of public record made available by the regulating bodies in each Member State. This openness helps allay fears of secret collusion between industry and government.

Results from 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) and be available in aggregate for the industry. Such monitoring might assist in addressing the question of whether photoperiod manipulation is effective in delaying sexual maturation and identify where the potential risk of egg releases could occur.

The potential to recapture escaped cod has not been analyzed; but is an important area for research and discussion with the public and farmers. It is important to discuss this with the public early in a development program, and to publicly 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.

Literature Cited

Casey J., Dann J. and Harding D. 1986. Stomach contents of cod and whiting caught during the English groundfish survey of the North sea in 1982 and 1984. ICES Council meting 1986 (Collected Papers), ICES, Copenhagen (Denmark) 1986.

Begg G.A. and Marteinsdottir G. 2000. Spawning Origins of Juvenile Cod. Marine Ecological Progress Series 202: 193–217.

Begg G.A. and Marteindottir G. 2002a. Environmental and stock effects on spatial distribution and abundance of mature cod *Gadus morhua*. Mar. Ecol. Prog. Ser. 229:233–244.

Begg, G. A., and Marteinsdottir G. 2002b. Environmental and stock effects on spawning origins and recruitment of cod *Gadus morhua*. Mar. Ecol. Prog. Ser. 229:245–262.

Blaxter, J.H.S. 2000. The enhancement of Marine Fish stocks. Advances in Marine Biology, 38: 1–54.

Fleming, I. A., Hindar, K., Mjolnerod, I.B., Jonsson, B., Balstad, T., Lamberg, A. 2000. Lifetime success and interactions of farm salmon invading a native population. Proc. R. Soc. Lond. B 267: 15 17–1524.

Fossâ J.H. Nordeide J.T. Salvanes A.G.V. Smestad O. Giske J. 1994. Rearing of Cod in Mafjorden 1982–1992. Can. Transl. Fish. Aqat. Sci.5624.

Huessy, K and Boettcher U. 1997. Food resource utilisation by juvenile Baltic Cod *Gadus morhua*: A mechanism potentially influencing recruitment success at the demersal juvenile stage? Marine Ecological Progress Series. 155: 199–208.

Hutchinson W.F., Carvallo G.R. and Rogers S.I. 2001. Marked genetic structuring in localised populations of cod *Gadus morhua* in the North Sea and adjoining waters. As revealed by micro-satellites. Marine ecology and Progress Series 223: 251–260.

ICES, 2005. ICES advice, October 2005. International Council for the Exploration of the Sea, Advisory Committee on Fishery Management (ACFM), www.ices.dk

Imsland A.K. and Jónsdóttir o.D.B. 2003. Linking Population Genetics and Growth Properties of Atlantic Cod. Fish Biology and Fisheries 13: 1–26.

Jorstad, K.E. 1994. Cod stock enhancement studies in Norway: genetic aspects and the use of genetic tagging. World Aquaculture '94, Book of Abstracts. New Orleans, p. 127.

Jorstad, K.,E., Naevdal, G. 1992. Genetic studies on released and recaptured cod in a fjord system. ICES, C.M. *1992/G:65*, 1–6. (Demersal Fish Cttee, Ref. F).

Jorstad, K.E., Paulsen, 0.l., Naevdal, G., Thorkildsen, S. 1994. Release and recapture of genetically tagged cod fry in a Norwegian fjord system. In: Beaumont, A.R. (ed.). Genetics and evolution of aquatic organisms. p. *5* 19–528. Chapman and Hall, London, England.

Jorstad, K.E., Paulsen, 0.l., N~cvdal, G., Thorkildsen, S. 1994. Genetic studies of cod, *Gadus morhua* L., in Masijorden, western Norway: comparisons between the local stock and released, artificially reared cod. Aquacult. & Fish. Manage. 25 (Suppl. 1): 77–91.

Kitada, S., Taga, Y., Kishino, H. 1992. Effectiveness of a stock enhancement program evaluated by a two-stage sampling survey of commercial landings. Can. J. Fish. Aquat. Sci. *49:* 1573–1582.

Lacroix, G.L., Koman, J., Heath, D.D. 1998. Genetic introgression of the domestic Atlantic salmon genome into wild populations: a simulation of requirements for conservation. Department of Fisheries and Oceans, Can. Stock Assess. Secretariate, Res. Doc. 98/158: 6pp.

SERAD, 2003. Locational Guidelines for the Authorisation of Marine Fish Farms in Scottish Waters. Scottish Executive Rural Affairs Department.

Lund, R.A., Hansen, L.P., Järvi, T. 1989. Identifisering av oppdrettslaks og vill-laks ved ytre morfologi finnestorrelse og skjellkaraktered. (Identification of reared and wild salmon by external morphology, size of fins and scale characteristics.). Norwegian Institute for Nature Research, (NINA) Trondheim, Res. Rep. 1: *1–54*. [In Norwegian].

Lura, H., Økland, F. 1994. Content of synthetic astaxanthin in escaped farmed Atlantic salmon, *Salmo salar* L., ascending Norwegian rivers. Fish. Manage. Ecol. 1: 205–216

Lura, H., Sagrov, H. 1991. A method of separating offspring from farmed and wild Atlantic salmon (*Salmo salar*) based on different ratios of optical isomers of astaxanthin. Can. J. Fish. Aquat. Sci. 48: 429–433.

Maroni K. 2000. Monitoring and regulation of Marine aquaculture in Norway. Journal of Applied Ichthyology. 16:192–195.

Munro, J.L., Bell, J.D. 1997. Enhancement of marine fish resources. Reviews in fisheries Science, 5(2): 185–222.

Nordeide, J.T., Salvanes A.G.V. 1991. Observations on reared newly released and wild cod *(Gadus morhua* L.) and their potential predators. ICES mar. Sci. Syrup. 192: 139–146.

Gubbins M.J., Gillibrand P.A., Sammes P.J., Miller B.S., and Davies I.M. 2003. OSPAR Eutrophication Assessment of Aquaculture Hotspots in Scottish Coastal Waters.

FRS Marine Laboratory Aberdeen Collaborative Report Number 07/03.

Otterlind, G. 1985. Cod migration and transplantation experiments in the Baltic. J. Applied Ichthyology 1(1): 3–16.

Richards, W.J. and Edwards, RE. 1986. Stocking to restore or enhance marine fisheries. pp. 75–80. In: Stroud, H. (Ed.). Fish culture in fisheries management. Am. Fish. Soc., Bethesda, Maryland, USA. (ISBN 0-913235-39-3).

Report of the ICES Working Group on the Environmental Interactions of Mariculture (WGEIM) 2003. International Council for the Exploration of the Sea, Copenhagen. www.ices.dk

Skaala, O., Dahle, G., Jorstad, K.E., Nævdal, G. 1990. Interactions between wild and farmed populations: information from genetic markers. J. Fish Biol. 36. 449–460.

Smedbol R.K. and Wroblewski J.S. 2002. Metaapopulation Theory and Northern Cod Population Stock Structure: Interdependancy of Subpopulations in Recovery of a Groundfish Population. Fisheries Research 55: 161–174.

Steingrund P. and Fernoe A. 1997. Feeding behaviour of reared and wild cod and the effect of learning: Two strategies of feeding on the two-spotted goby. Journal of Fish Biology. 51 No.2:334–348.

Svasand, T. 1993. Cod enhancement experiments in Norway - status and perspectives. Publ.European Aquacult. Soc. 19: 1557. (From Discovery to Commercialization. Abstracts of contributions presented at the Intern. Conference “World Aquaculture ‘93”).

Svåsand T., Kristiansen T.S, Petersen T., Salvanes A.G.V., Engelsen R., Naevdal G. and Nodtveedt M. 2000. The Enhancement of Cod Stocks. Fish and Fisheries, 1: 173–205.

Youngson, A.F., Webb, J.H., Thompson, C.E., and Knox, D. 1993. Spawning of escaped farmed Atlantic salmon (Salmo salar): hybridization of females with brown trout *(Salmo trutta*). Can. J. Fish. Aquat. Sci., 50*:* 1986–1990.

Wright P.J., Gibb F.M., Gibb I.M. Heath M.R., and McLay H.A. 2003. North Sea Spawning Grounds. Fisheries Research Services Marine Lab. Aberdeeen.

Annex 6: Risk analysis of the potential interbreeding of wild and escaped farmed sea bass *(Dicentrachus labrax)*

Introduction

The sea bass (*Dicentrachus labrax)* is primarily a benthic species found in the eastern North Atlantic from Norway to Morocco, including the Mediterranean Sea. It feeds mainly on invertebrates (shrimps and mollusks), often close to shore, but also takes other fish. The maximum reported age is 15 years, and reaches 103 cm length and 12 kg weight. It lives in waters of a wide range of salinities and at depths of 10–100m. Sea bass supports commercial fisheries in some areas, and is well established as an aquaculture species, particularly in the Mediterranean Sea.

Hazard Identification

Known effects of cultured populations

Cultivation of sea bass began in France, Spain and Italy, using local western Mediterranean broodstock. Subsequent developments further east generally initially utilized eggs/juveniles from the established industries, ie western stock. As discussed n more detail below, there is evidence that there are three endemic populations of sea bass covering the Atlantic and the Sea of Alboran, the western Mediterranean, and the eastern Mediterranean Sea (Patarnello *et al.*, 1993; Cesaroni *et al.*, 1997; Naciri *et al.*, 1999).

The only data available on escapes indicates that when sea bass cultured from western Mediterranean populations escaped in the eastern Mediterranean, they established and maintained distinct populations of the western Mediterranean phenotype without intergradation with the local population (Bahri-Sfar *et al.*, 2004). No stock enhancement or voluntary release operations have been reported.

While there is limited experience of 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 (Allegrucci *et al.*, 1997; Lemaire *et al.*, 2000) 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.

Closing the breeder stocks for selective breeding purposes began in the late 1990s in some of the larger aquaculture companies. Sea bass have been escaping from captivity in the Mediterranean for the last 2 decades. The lack of data on sea bass survival covering the period before and after the advent of sea bass culture, and on the degree of genetic differentiation between wild and cultured sea bass (which appears low 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 Channel/Biscay) population could be indicative of nutritional resource limitation, or simply a response to local water temperatures (Pickett and Pawson, 1994).. During the growth period (April to October), surveys of condition index, feeding status and fat content of year 0 and year 1 classes did not show any shortfall in feeding resources, even in 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 fish farming began also precludes comment on whether ecological shifts might result from sea bass culture activities, as currently practiced.

Risk Assessment

Release Assessment

The majority of the sea bass reared in Europe are maintained in floating sea cages, in near shore locations. There is one large production unit (1500 tonnes a year) on land in Northern France (Channel coast) that uses heated effluent from a power plant. Some farmers use salt marshes to produce fish in ponds, for example in the Bay of Biscay, in Spain and in Italy. The bulk of the production is based in Greece, using sea cage technology, but all the Mediterranean countries are now producing sea bass. Farms using the recirculating technology are expending slowly. Usually, 10 to 20 g fish from hatcheries are taken into these farms, where they are reared at a maximum density of 20 kg/m^3. Sorting operations are frequent throughout the production cycle, because sea bass are cannibalistic and show high heterogeneity in growth rates. Sorting requires the fish to be anaesthetised. The typical production cycle duration is 2 years to produce 250-400 g fish. It requires one more year to produce 600-800 g fish for specific markets. The market for sea bass is changing and is asking for larger fish (i.e. three years of age) than the traditional pan size, and at this age sea bass males are mature but females are immature. Consequently, there is an increasing incidence of mature fish in the sea cages.

Recently, a movement of sea cage locations into less protected areas further from the coast occurred in Spain and Greece, using basically the same technology. This was due to environmental pressure on near shore sites. The main factors limiting this move to offshore locations are economic, and then technical. Production is likely to expand in the future, probably by moving increasing numbers of fish cages towards these new offshore locations.

The total aquaculture production of sea bass was about 62000 metric tonnes in 2002, essentially from the Mediterranean Sea, representing a standing stock in cultivations of about 100 000 tonnes. The average weight can be estimated at 200 g, which indicates some 500.10^6 individuals being under cultivation, i.e. approximately twice the estimated number of wild individuals.

No information is available on the actual number of fish that escape from sea bass farms. It is not compulsory for fish farmers to report this information. It is clear that sea cage technology is more likely to give rise to escapes than land based farms, particularly those utilising recirculating systems. It may be assumed that the 0.1 to 3% rate of escape of reared stock from the cages in the salmonid industry is broadly applicable to the sea bass industry. The primary cause for escapes is containment failures, e.g. cages broken by bad weather conditions or accidents, or nets opened by external predators (including humans), or insufficient maintenance. Sorting operations, since they are done on the cages, and transport, may also induce un-intentional releases.

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 possible 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. Conditions at farming sites in the Mediterranean appear more favourable to this last process than sites in the Atlantic, where farms are generally far from sea bass spawning areas and weather conditions are generally less favourable.

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

Exposure assessment

The global number of cultured fish is in the proportion of 2.5 to 1 with wild populations. But it is probably greater than that in areas where fish farms are located, even when considering a potential spread of escapees of up to 50 km away from the fish farm. A rough calculation suggests, on average, around 50,000 wild fish in such areas, where an average fish farm rears 1.0 million individuals. The number of escapees might be 1000–30 000 in these circumstances, i.e. could attain a similar order of magnitude as the wild population. This could have the potential to lead to displacement of wild population in some specific sites.

Sea bass are cannibalistic, so aggregation of small fish with large ones is unlikely. Escapees will not be subjected to direct predation by wild congeners any differently from the wild bass. Being generally larger 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 communities 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 prey for them. Only in a case where escapes had occurred some months previously did Dempster *et al.* (2002) find a significant number of sea bass. This suggests that there is a low probability of escapees mating with wild sea bass around the cages.

Young sea bass are attracted by the high food availability and low salinities near the coast, so that they have the potential to be close to the farms. There is a high probability of escapees of the same age class mixing with the wild individuals, but the degree of mixing of subpopulations is not known. If the persistence of the two populations reported by Lemaire *et al* (2000) is confirmed, genetic mixing could be low when these fish mature. 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 as salt marshes. A significant issue would be whether the available food in these areas could support a sea bass population increased by escapees. Both groups will probably migrate to offshore zones in winter.

The hunting behaviour of reared fish released to the wild is probably not degraded at present, as the majority of these fish are of the first or second generation away from wild breeders. The response to predators 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 the fitness of farmed fish. Counterbalancing this is the observation that sea bass move in large shoals in sea cages. This behaviour acts in favour of better survival in the wild, provided that the food supply would not be limiting. The fact that reared fish can support overcrowding more easily, with lower stress, acts in their favour when confronted by wild predators.

Body weight appears to be a highly heritable trait in sea bass (Saillant *et al.,* in press), which could lead to a high introgression of this character in the wild population, which is nevertheless dependant on a high genetic and environment interaction.

Consequence Assessment

Life history – Description of wild population

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 sometimes be found in rivers. Its salinity preference is 15 (Saillant *et al.*, 2003) and it is not sensitive to flood conditions. Traditionally, it was found in coastal lagoons as well at river mouths. At 12 mm length, larvae actively swim to nursery habitats. Sea bass are pelagic up to the size of 20 mm, and then become demersal. Sea bass hunt for prey 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 from spring to autumn in near-shore coastal areas, in lagoons and/or estuaries. 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 the open sea and into estuaries. When adults leave these areas in autumn, they move to depths less than 50 m. 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 Lyon.

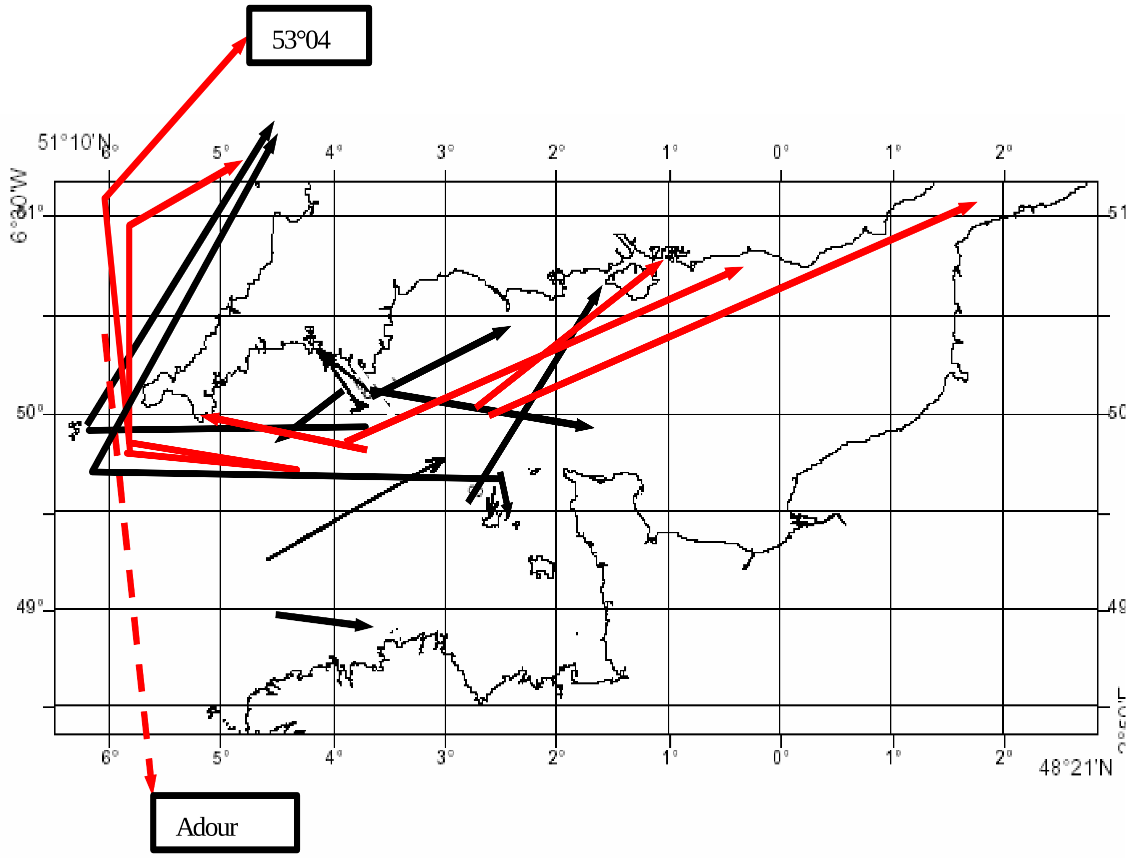


Figure A5.1: Adults displacements from releases in 2002 (red) and 2003 (black). From Morizur (2004).

In the Atlantic, juveniles behave as they do in the Mediterranean, and they also colonize tidal flats and salt marshes. In contrast to the Mediterranean area, adult sea bass 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.

Very few sea bass appear to move between Channel areas and the Bay 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 larger 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 to remain near the farms in significant numbers for some months, demonstrating a relatively high degree of site fidelity.

The stock is said to be “not under threat” in the Atlantic ICES sub-areas. The status in the Mediterranean is not known.

Growth and Survival

Females are 20% larger than males of the same age (Saillant *et al.,* 2002). Mediterranean sea bass are generally 40% bigger than Atlantic sea bass of the same age. Growth almost ceases in colder (below 10°C) winter waters. The table below summarizes the growth (length in cm/weight in grams) of wild females in these two areas (Barnabé, 1989), and for cultured females (Dosdat, personal communication.).

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				

Diet

In both the Atlantic and Mediterranean, the diet of young (first year) sea bass 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 length. At all ages, sea bass are cannibalistic (Pickett and Pawson, 1994). Sea bass can eat other sea bass up to half their weight. In cultivation, it is reported to be cannibalistic even during the very young stages (under 6 months). This behaviour may affect the ability of escapees to survive in the wild. Different year classes of wild sea bass generally do not mix.

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 assumptions of an average weight of 1kg for the landings, and a mortality rate of 0.9 (Pickett and Pawson, 1994), the number of wild fish in the Mediterranean has been estimated at $210*10^6$ individuals. In the Atlantic, sea bass abundance is very variable and appears to be respond to climatic conditions rather than to feed availability. The present level of exploitation in the Atlantic is considered sustainable by ICES (2002). This species is not subjected to Total Allowable Catch or fishing quota controls.

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 one year later in the Atlantic. First spawning occurs even later, particularly in the northern parts of the sea bass range (e.g. 6 years in the Channel).

Spawning areas for the Atlantic stock have been identified in the Channel, in the Celtic Sea, west of Brittany and in the Bay of Biscay. Spawning has been observed (Barnabé, 1976) inshore on rocky bottom in the Mediterranean (at 5-6m depth). Spawning occurs in December to February in the Mediterranean, and in March to June in the Atlantic. Adults are known to concentrate for a period over 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 continue 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 of the eggs has to occur within seconds), but it is not known if the fish form 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 as conspecific wild populations in the area of the cages. Maturation and reproduction are under the control of temperature and photoperiod.

Genetic structure of the populations

There is evidence that there are three endemic populations of sea bass: one covering the Atlantic and the Sea of Alboran, one in the western, and one in the eastern, Mediterranean (Patarnello *et al.*, 1993; Cesaroni *et al.*, 1997; Naciri *et al.*, 1999). This differentiation was described using the allele frequency of six microsatellite loci. Microsatellite analysis is very sensitive in detecting genetic variability, but the relative proportions of intra- or inter-population genetic variability are not always clear. It is suspected that the passive retention of larvae east and west of the Straits of Gibraltar is not a sufficient explanation for the persistence of the pattern that has been detected between Atlantic and Mediterranean stocks. Castilho and McAndrew (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 populations (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 responded 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 populations 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 from 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)* that lives 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.

From the early 1970s to the mid 1980s, broodstock fish for cultivated sea bass were obtained from the wild, primarily from the western Mediterranean stock. Later, cultured broodstock began to be used, with a increasing risk of in-breeding. From the 1990s, genetic improvement through selective breeding occurred in France, Greece and Italy, increasing the genetic distance between wild and farmed population. In fish farms, males generally outnumber females. The exact nature of the mechanism controlling this has not yet been demonstrated, but environmental influences on sex determination have been demonstrated (Saillant *et al.*, 2002).

From various surveys, it appears that the majority of the breeders used in cultivation originated from the Western stock. Transport of non-local stocks has therefore 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 where local stocks are utilized, genetic drift may occur.

Summary and related factors

The high proportion of males in the cages acts to decrease the risk of fish in the cage fertilising the other sex in the wild population because sperm viability in sea water is only some 20 seconds, while it is some 2-3 minutes for eggs. However, to produce bigger fish on the farms, only females that grow faster are selected. So, in the same location, fish cages supporting maturing females would be separated from the rest of the males and non-maturing females. A solution to ensure that any eggs released are not fertilised by captive males would be to separate the relevant cages by a sufficient distance.

Increased sea bass fishing has been reported in Sicily in the regions where sea cages were in use, but there was no increase in other areas (Cannizzaro *et al.*, 1999). Even if the origin of the fish (wild or cultured) was not investigated, this observation indicated that cages could increase the productivity of these oligotrophic areas and/or act as artificial reefs. This does not contradict the observations by Dempster *et al.* (2002). It means that only fish with specific feeding regimes adapted to the new food resources became concentrated around the cages, and therefore become possible prey for (presumably wild) sea bass.

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

There appears only a small risk of successful reproduction between wild and cultured populations in the areas of cage culture, either because only small numbers of wild individuals come to the sea cages 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 routes and patterns. In the absence of long term surveys, the stability of the equilibrium cannot demonstrated.

Bahri-Sfar *et al.* (2004) reported that Greek fish farmers in the island of Leros experienced very great difficulty catching individuals from the eastern population in the environment, even far from fish farms, where 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 to have 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 maintainance of the western genotype. It could also signify the displacement of local populations to the benefit of imported ones.

Logic model

The series of steps and processes leading from the establishment of sea bass farms in coastal waters to significant decreases in wild sea bass stocks through interbreeding can be summarised in a logic model as below:

Process of concern:	**Changes in fitness of wild populations of sea bass due to genetic intergradation**
End Point of Concern:	**Significant decline in survival in wild sea bass populations due to interbreeding with escaped cultured sea bass.**

Logic model steps:

11) Sea bass farms are established in coastal waters.

12) There are phenotypic differences between the wild and cultured sea bass populations.

13) These phenotypic differences arise primarily for genetic rather than environmental reasons.

14) The primary route for genetic interaction (interbreeding) between cultured and wild sea bass is through escapes of sea bass from cages.

15) Cultured sea bass escape from cages.

16) Cultured sea bass interbreed with wild sea bass.

17) The progeny of this interbreeding (hybrids) show reduced fitness.

18) Sufficient gene flow to affect survival rates of sea bass in individual fisheries management units, i.e. the population structure of wild sea bass is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

19) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured sea bass causes significant declines in survival in wild sea bass populations.

20) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e Escapes of cultured sea bass cause significant decreases in wild/feral sea bass stocks.

The information presented in the preceding sections of this risk analysis allows annotation of each step in the logic model to indicate the probability that each step has been, or will be, completed.

1) Sea bass farms are established in coastal waters.

Highly probable with low uncertainty. Sea bass farms are already established throughout the Mediterranean. Considerable growth in production has occurred in recent years, and development into more offshore areas results in further opportunities for increased production. Where the ranges of wild sea bass and cultivation overlap, the intensity is assessed as high.

2) There are phenotypic differences between the wild and cultured sea bass populations.

Highly probable with low uncertainty. Initially, farms used wild caught broodstock from the western Mediterranean stock, resulting in immediate differences between wild and farmed stocks in the eastern Mediterranean. Current movements in the industry towards selective breeding will increase divergence, and therefore the intensity is assessed as high.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

Highly probable with low uncertainty. As indicated above, the use of wild-caught western broodstock immediate resulted in genetic differences between wild and farmed stocks. The differentiation in the wild stock in the eastern Mediterranean increases the potential significance of the differences.

4) The primary route for genetic interaction (interbreeding) between cultured and wild sea bass is through escapes of sea bass from cages. .

Highly probable with low uncertainty. It is possible that larger farmed sea bass may release eggs or milt inside cages, but the opportunities for these products to interact with eggs or milt from wild fish outside the cages is very limited. Similarly, interbreeding in farmed stock can be easily preventing by separation of cages. Therefore, it is likely that escapes will be a more significant route for interaction.

5) Cultured sea bass escape from cages.

Highly probable with low uncertainty. It is highly likely that some sea bass will escape from cages. Experience with other species indicates that accidents happen, and the requirement for frequent handling/grading of sea bass will increase the probability of losses.

6) Cultured sea bass interbreed with wild sea bass.

Moderate probability with moderate certainty. Escapes of western population sea bass have occurred in the eastern Mediterranean for many years. Studies of wild sea bass populations in the eastern area indicate that little intergradation has occurred, i.e. western-type sea bass populations are found. The intensity (severity) of interbreeding is probably low.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

Moderate probability with high uncertainty. There is no evidence to support this contention for sea bass, at least partly because of the difficulty of detecting hybrids in the wild. Reduced fitness of the progeny from interbreeding has been shown in some other species and therefore should interbreeding occur there may be some probability that the progeny will show reduced fitness. The differentiated nature of the eastern stock suggests vulnerability of local stocks to interbreeding, but evidence suggests that escaped western stock fish remain as distinct populations, i.e. intensity of on local sticks is low.

8) Sufficient gene flow to affect survival rates of sea bass in individual fisheries management units, i.e. the population structure of wild sea bass is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

Low probability with moderate uncertainty. The numbers of fish in cultivation probably greatly outnumber the wild populations in production areas, and in some areas wild stocks are highly differentiated. Escapes therefore may numerically have the potential to interact with the wild populations. However, the apparent low rate of intergradation makes the outcome on population fitness unlikely, and of low severity.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured sea bass causes significant declines in survival in wild sea bass populations

Low probability with low uncertainty. There is no evidence to support this contention for sea bass, indeed, such evidence exists that rates of interbreeding are low. The distribution of any effect over a larger number of fish in the wild reduces the uncertainty in the assessment.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e escapes of cultured sea bass cause significant decreases in wild/feral sea bass stocks.

Low probability with low uncertainty. There is no evidence to support this contention for sea bass and the effect would need to be spread over even larger number of wild fish than in 9 above.

STEPS IN THE LOGIC MODEL	INTENSITY (C,H,M,L, OR N) [1]	PROBABILITY (H,M,L,EL, OR N)[2]	UNCERTAINTY (H,M, OR L)
Step 1	H	H	L
Step 2	H	H	L
Step 3	H	H	L
Step 4	H	H	L
Step 5	H	H	L
Step 6	L	M	M
Step 7	L	L	H
Step 8	L	L	M
Step 9	L	L	L
Step 10	L	L	L
Final Rating [4]	L	L	H

Explanatory notes:

Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

Severity = C – very intense, H – high, M – Moderate, L – Low, N – Negligible

Uncertainty = H- Highly uncertain, M – Moderately uncertain, L – Low uncertainty

The final rating for the Probability is assigned the value of the element with the lowest level of probability.

The final rating for the Severity (intensity of interaction) is assigned the value of the step with the lowest risk rating (e.g., Medium and Low estimates for the logic model steps would result in an overall Low rating). Note that the calculation of the final rating follows the multiplication rule of probabilities (i.e., the severity that a given event will occur corresponds to the product of the individual severity). Thus the final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

The final rating for the Uncertainty is assigned the value of the element with the highest uncertainty level (i.e. the least certainty).

Risk evaluation

Without regulations or farm management practices specific to sea bass farms there is unlikely to be any difference between the outcome of the Consequence Assessment and that of the Risk Evaluation. Risk management may be able to alter the values in the above table.

	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Un-certainty	Research/Development
1	Sea bass farms are established in coastal waters	H	Where feasible move to land- based production	L	Develop economically competitive land-based technologies.
2	There are phenotypic differences between the wild and cultured sea bass populations.	H	For each generation recruit all grow-out stock from juveniles captured in the wild.	L	
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.	H		L	Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival.
4	The primary route for genetic interaction (interbreeding) between cultured and wild sea bass is through escapes of sea bass from cages.	H	Improve containment design and/or build in fail-safe measures	L	Identify factors that will limit dispersion of escapees
5	Cultured sea bass escape from culture	H	Improve containment design and/or build in fail-safe measures Recovery plan for escaped fish	L	
6	Cultured sea bass interbreed with wild sea bass	L	Use of sterile fish	M	Development of tools to distinguish wild fish from escapees. Studies to determine the survival of escapees, their migration pattern and behaviour in relation to their location (e.g. inshore or offshore) and the season they are released
7	The progeny of this interbreeding (hybrids) show reduced fitness	L	For each generation recruit all grow-out stock from juveniles captured in the wild	H	Development of tools to distinguish wild fish from escapees and hybrids Studies to investigate the fitness and survival of hybrids
8	Sufficient gene flow to affect survival rates of sea bass in individual fisheries management units, i.e. the population structure of wild sea bass is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.	L	Limit the distribution of sea bass farming to either proximity to small value stocks or very large stocks.	M	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured sea bass causes significant declines in survival in wild sea bass populations.	L		L	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e. Escapes of cultured sea bass cause significant decreases in wild/feral sea bass stocks	L	Limit the distribution of sea bass farming in relation to the distribution of the species or meta population	L	Identify dynamics of genome at the meta population or species level.

Risk Management

Option Evaluation

Option evaluation addresses what might be done to reduce the probability of a risk being expresses or reduce the uncertainty in the prediction of the probability of expression of a risk. A useful way to do this is to look at the logic model and for each step identify what can be done to reduce the probability of it occurring. These are steps to mitigate possible effects. The other contribution would be to reduce the uncertainty associated with predicting that the step will happen. Usually that involves further research or development. The table above identifies both mitigation actions and research or development steps that could be used in addressing risks associated with sea bass culture.

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 can dramatically reduce the probability of escapes occurring.

The present approach involving the selective breeding of fish for cultivation increases the risk of reduced genetic variability in farm stocks unless a sufficient level of heterozygosity is maintained in the cultured broodstocks used for breeding. It can be noted that the use of sea bass broodstock that have been selected for culture is progressively expanding throughout Europe.

The 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 effects on the ability of a local sea bass culture industry to compete internationally. The EU funded project "Heritabolum" is evaluating the relative performances of these various populations under various production constraints. This could lead to recommendations for the use of particular populations with regards to the consequences of environmental interactions.

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

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

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

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

The use of sterile fish to limit both gamete emission and possible interbreeding with wild populations is frequently recommended and would significantly reduce the risk of interactions with wild stocks. Sea bass triploids have been produced on a pilot scale in France, targeting the increased productivity of sterile fish. The triploids showed a decreased occurrence of breeding, but it was not totally prevented (98% success). Experience showed that sea bass triploids were less robust in cultivation than diploids. This technique for supplying triploids was not economically viable, so the industry has not adopted them in their production processes (in contrast to trout farming where production of large triploid trout is less costly than producing diploids). Hybridisation with the related species (*D. punctatus*) is another area for investigation. Hybrids have been reported to have a high survival rate, and, in some cases, to produce spontaneous triploids. However, the sterility of diploid hybrids has not yet been demonstrated.

At present, it appears that governmental 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 sea bass farms to more offshore, or open coastal, locations, where weather conditions could lead to more frequent containment failure. Effective development requires improved designs for mooring systems and containments.

Code of practice - Certification

In all cases, the training of the fish farm operatives should be an essential preoccupation of the fish farmer. The maintenance, replacement and monitoring of nets is of paramount importance to limit accidental escapes. Periodic inspection of containment equipment should be compulsory, and particular attention should be devoted to net replacement either for cleaning or to increase the mesh size. In particularly, producers should give greater attention to improving sorting and treatment operations.

Other aspects of sea bass farming that might be enhanced by a code of practice for a local industry include:

- Advice on best practice for sorting and bath treatments.
- Advice on best practice for transport, applying to both supply of juveniles to the farm as well as to harvested adults at commercial age
- Improved methodologies for net replacement
- Advice on best practice for mooring and anchoring, particularly in more exposed areas
- In the particular case of 3-year old fish, the females 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 personnel

Research

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

- Studies to determine the survival of escapees, their migration patterns 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.
- Monitoring the behaviour of adults and juveniles released in offshore locations. It would be especially helpful 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.

- The efficacy of photoperiod control on maturation.
- Another possible approach, not specific to sea bass, could be to produce fish that genetically do not synthesize some essential dietary component which they can only find in artificial feed. This would make the fish unable to survive in the wild. However this solution is highly hypothetical and needs substantial theoretical development (animal welfare, technical feasibility) but could also be applied to GMO fish if they are adopted by the industry.
- One way to decrease the impact of releases in a given environment would be to maximise the wild stocks in areas where farming activities are based, particularly where wild stocks are scarce. This would require tools to be available to evaluate these wild stocks.
- Tools to enable the recognition of wild fish from escapees are not readily available, and new developments are necessary to implement their monitoring.
- Contingency Planning

Recovering escaped sea bass around the cages within some days/weeks of escape seems possible and efficient, particularly for adults. Increasing the fishing activity (e.g. by commercial fishermen) in the vicinity of the farm after a major release has proved efficient. Fishing techniques could also be adapted (e.g. to include the use of specific devices, perhaps that would otherwise be illegal, under specific authorisation for this purpose). This raises 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 some clarification of the law could ease the process.

The degree to which a fish farm should be monitored must be a function of the degree of risk arising from the farming system. In this respect, the intensity of monitoring should decrease from sea ranching to offshore sea cages, inshore sea cages, flow through land based systems, closed systems and integrated systems.

Literature Cited

Allegrucci, G., Fortunato, C., Sbordoni, V., 1997. Genetic structure and allozyme variation of sea bass (Dicentrarchus labrax and D. punctatus) in the Mediterranean sea. Marine Biol., 128: 347–358.

Bahri-Sfar, L., Lemaire, C., Ben Hassine, O., Bonhomme, F., 2000. Fragmentation of sea bass populations in the western and eastern Mediterranean as revealed by microsatellite polymorphism. Proc. Royal. Soc. Lond., B 269: 929–935.

Bahri-Sfar, L., Lemaire, C., Chatain, B., Divanach, P., Ben Hassine, O., Bonhomme, F., 2004. Impact de l’élevage sur la structure génétique des populations Méditerranéennes de Dicentrarchus labrax. Aquatic Living Ressources (submitted).

Barnabé, G., 1976. Contribution à la connaissance de la biologie du loup Dicentrarchus labrax (L.) de la région de sète. PhD thesis, University of Languedoc, 424 pp.

Barnabé, G., 1989. Aquaculture. Technique et Documentation, Lavoisier, Paris, France, Second edition; Vol. 2, 1308 p.

Beyst, B., Hostens, K., Mees, J., 2001. Factors influencing fish and macrocrustacean communities in the surf zone of sandy beaches in Belgium: temporal variation. Journal of Sea Research , 46 : 281–294.

Bianchini, ML., Casadei, S., Ingle, E., Lombardi, F., 1994. Competition effect on the survival of sea bass fry without functional swim bladder. J. Ecobiol., 6: 1–4.

Bonhomme, F., Naciri, M., Bahri-Sfar, L., Lemaire, C. 2002. Comparative analysis of genetic structure of two closely related sympatric marine fish species Dicentrarchus labrax and Dicentrarchus punctatus. C. R. Biologies, 325 : 213–220.

Cannizzaro, L., Garofalo, G., Bono, G., Scalisi, M., 1999. Dicentrarchus labrax (Linnaeus, 1758) nella fascia costiera da Punta Raisi a Pozzallo. Congress of the SIBM, Ustica, Palermo (Italy). Biologica Marina Mediterranea, 6: 537–539.

Castilho, R., Mac Andrew B., 1998. Population structure of sea bass in Portugal : evidence from allozyme. J. Fish Biol., 53: 1038–1049.

Cesaroni, D., Venanzetti, F., Allegrucci, G, Sbodorni, V., 1997. Mitochondrial DNA length variation and heteroplasmy in natural populations of the European sea bass, Dicentrarchus labrax. Mol. Biol. Evol., 14: 560–568.

Dosdat, A., Chatain, B., La Pomélie, C., 2001. Development and achievements of marine juvenile production. NATO Science Series, Serie A: Life Science, 314: 70–82.

Felip, A., Zanuy, S., Carillo, M., Piferrer F., 1999. growth and gonadal development in triploids sea bass (Dicentrarchus labrax L.) during the first to years of age. Aquaculture, 173 : 387–397.

Fahy, E., Forrest, N., Shaw, U., Green, P., 2000. Observation on the status of sea bass Dicentrarchus labrax stocks in Ireland in the late 1990s. Irish Fish. Investig., 5, 27 pp.

ICES, 2002. Answer to the request from the EC concerning the status of the sea bass in European waters. *ICES Cooperative Research Report*, 255: 309–312.

Koumoudouros, G., Maingot, E., Divanach, P., Kentouri, M., 2002. Kyphosis in reared sea bass (*Dicentrarchus labrax* L.) : ontogeny and effect on mortality. Aquaculture, 209: 49–58.

Lafaille, P., Lefeuvre, J.C., Schricke, M.T., and Feuteun, E., 2001. Feeding ecology of 0-group sea basss, Dicentrarchus labrax, in salt marches of Mont saint Michel Bay (France). Estuaries, 24: 116–125.

Lemaire, C., Allegricci, G., Naciri, N., Bahri-Sfar, L., Kara, H., and Bonhomme, F., 2000. Do discrepencies between microsatellite and allozyme variation reveal differential selection between sea and lagoon in the sea bass (Dicentrarchus labrax) ? Mol. Ecol., 9: 457–467.

Morizur, Y., 2004. Les opérations de marquage du bar. Internal Ifremer report, 4 pp.

Naciri, Y., Lemaire, C., Borsa, P., Bonhomme, F., 1999. Genetic study of the Atlantic/Mediterranean transition in seabass (*Dicentrarchus labrax*). Journal of Heredity, 90, 591–596.

Olla, B.L., Davis, M.W., and Ryer, C.H. 1998. Understanding how the hatchery environment represses or promote the development of behavioural survival skills. Bulletin of Marine Science, 62: 531–550.

Patarnello, T., Bargelloni, L., Caldera, F., and Colombo, L., 1993. Mitochondrial DNA sequence variation in the European sea bass, Dicentrarchus labrax L. (Serranidae): evidence of a differential haplotype distribution in natural and farmed populations. Mol. Mar. Biol. Biotechnol., 2: 333–337.

Pawson, M.G. 1992. Climatic influence on the spawning success, growth and recruitment on bas (Dicentrarchus labrax L.) in British waters. ICES Mar. Sci. Symp., 195: 388–392.

Peruzzi, S., Chatain, B., 2000. Pressure and cold shock induction of meiotic gynogenesis and triplïody in the European sea bass, Dicentrarchus labrax, L.: relative efficiency of methods and parental variability. Aquaculture, 189: 23–27.

Pickett, G.D, Pawson M.G., 1994. sea bass : biology, exploitation and conservation. Chapman and Hall Eds., London ; 337 pp.

Saillant, E., Fostier, A., Menu, B., Haffray, P.Y., and Chatain, B., 2002. Sexual growth dimorphism in sea bass, Dicentrarchus labrax. Aquaculture, 202 : 371–387.

Saillant, E., Fostier, A., Haffray, P.Y., Menu, B., Thimonier, J., Chatain, B., 2002. Temperature effects and genotype-temperature interactions on sex determination in the European sea bass (Dicentrarchus labrax, L.). J. Exp. Zool, 292: 494–505.

Saillant, E., Fostier, A., Haffray, P.Y., Menu, B., Chatain, B., 2003. Saline preferendum for the European sea bass, Dicentrarchus labrax, larvae and juveniles : effect of salinity on early development and sex determination. J. Exp. Marine Biol. Ecol., 287: 103–117.

Saillant, E., Dupont-Nivet, M., Haffray, P., Chatain, B., 2006. Estimates of heritability and genotype–environment interactions for body weight in sea bass (Dicentrarchus labrax L.) raised under communal rearing conditions. Aquaculture (in press).

Annex 7: Risk analysis of the potential interbreeding of wild and escaped farmed sea bream *(Sparus auratus)*

Introduction

The sea bream (*Sparus auratus)* inhabits seagrass beds and sandy bottoms normally down to about 30m depth. It is found in the Mediterranean Sea and also in the East Atlantic from the United Kingdom to Cape Verde. It feeds mainly on shellfish, including bivalves, and can tolerate brackish water in lagoons and estuaries. The maximum reported age is 11 years, reaching 70 cm and 17 kg weight. Sea bream supports commercial fisheries in some areas, and is well established as an aquaculture species, particularly in the Mediterranean Sea.

Hazard Identification

Known effect of culture populations

Only one case of intentional release of cultured sea bream has been reported. It occurred in the southern Atlantic coast of Spain, and in the Bay of Cadiz (Sanchez-Lamadrid, 2002; 2004). These studies suggest that released fishes do not move far from the release point, provided that feeding resources in the form of molluscs are available and that the salinity remains constant. Survival was better for 100g fish than for smaller (15 g) ones, possibly reflecting a higher capacity to adapt. Good growth rates and condition indexes suggested that the behaviour of released fish was adapted to life in the wild. Fish were recaptured only at sites of good water quality (high oxygen, high salinity, low organic load) and at sites where feed was very abundant. 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 Cadiz, one year old released fish moved less than 12 km in few months. The scale of dispersal is probably dependant on the area of release. In these studies, predation by birds was reported to be intensive 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 distant. Released fish were mature and were caught in shoals where they were found mixed with wild conspecifics. They showed the same spawning behaviour as wild specimens after one year in the natural environment, strongly suggesting that gene flows would occur between escaped cultured fish and wild populations.

Risk Assessment

Release assessment

Sea bream are usually reared in the same structures and the same farms as sea bass. The majority of the sea bream reared in Europe are maintained in floating sea cages, into near shore locations. Some farms in Spain and Italy are using salt marshes to produce fish in ponds. The bulk of the production is based in Greece, using the sea cage technology, but all the Mediterranean countries are producing sea bream. They are also grown in Israel in the Gulf of Aqaba. Farms using the recirculating technology are expending slowly. Usually, 10-20 g fish from hatcheries are transferred to these farms, where they are reared at a maximum density of 20 kg/m^3. Sea bream are not cannibalistic, and have a lower dispersion of individual weights within a batch than sea bass, and so sorting operation are less frequent than for sea bass. The typical production cycle is 2 years to produce 250-400 gram fish. It requires one more year to produce 600–800 gram fish for specific markets. The market for sea bream is changing and is requiring bigger fish (three years old) than the traditional pan size. At this age, some of the fish may be mature males, but it is unlikely that there will be any mature females. Consequently, there is an increasing incidence of mature fish in the sea cages.

Recently, pressure on sites for sea cages in near shore locations has led to development of sea bream cages in less protected areas further from the coast in Spain and Greece, using basically the same technology. The total production is likely to expand in the near future, probably by increasing the production at new off-shore locations.

81,000 tons of sea bream were produced in 2002, almost exclusively in the Mediterranean countries. To produce this, the standing stock in cultivation is about 100,000 tons, which is equivalent to around 450.10^6 individuals in cultivation.

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, were apparent in some aquaculture stocks. Genetic drift, probably caused by propagation practices, is most likely to be responsible for the decrease in genetic variation (Palma *et al.*, 2001). A risk of inbreeding has been reported because of the use of a small effective population (Brown *et al.*, 2005) or unique populations (De Innocentis *et al.*, 2005). Mass selection programmes have been developed in France and Israel which could increase the tendency (Gorshkov *et al.*, 2002).

In the case of sea bream, the risk of inbreeding and loss of genetic variability in cultured stocks could be enhanced by the hermaphroditic status of the species. Inter-generic breeding has been used to attempt to produce sterile hybrids of *Pagrus* and *Sparus* (Paspatis *et al.*, 1999). Triploid sea bream have been successfully produced and males proved to be unable to produce spermatozoa. Survival rates are poor and the technique has not been adopted by the industry.

No information is available on the number of sea bream escaping from cultivation, and the evaluation made for sea bass is applicable in the same way. Sorting operations are less frequent for sea bream than for sea bass, and so the risk fof escapes may also be less. On the contrary, sea bream feeding behaviour involves crunching its feed using its powerful jaws. Sea bream thus commonly nibble the nets to feed on the epifauna, and leading to a higher occurrence of damage and holes in the cages than for sea bass, and a greater level of maintenance.

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

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 were reported to be near sea cages in which both sea bream and bass were being reared, in contrast to other sparid species (i.e. *Boops* sp., *Oblada* sp.). This will not encourage competition between 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 conspecifics. The experiments by Sanchez-Lamadrid (2001, 2004) in the Atlantic demonstrate that this can occur. The risk of displacement of wild populations by escapees, and of competition for feed should be the consequence. Aggressive behaviour has been reported in reared sea bream, in the absence of feed limitation. This could be also the case in the wild, and may therefore suggest aggressiveness linked to territoriality.

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.

Consequence assessment

Life history – Description of wild population

Distribution

Sea bream is found from the Mediterranean Sea to the south of England. It is a marine species than can tolerate both reduced salinity, but not as much as the sea bass, and also hypersalination. Its preferred thermal regime is higher than that for sea bass by two degrees C. It is normally found in coastal lagoons during summer (from 1 to 3 years age) and moves to the open sea when the temperature decreases in autumn. They usually stay in near-shore areas. Up to the size of 20 mm, they are pelagic, and then become demersal. Aggregative distributions of 0+ fish near estuaries have been reported in the Mediterranean (Chauvet *et al.*, 1992; Sanchez-Lamadrid, 2002), when their behaviour is gregarious. However, fresh water flooding appears to disrupt this pattern. Sea bream are very sensitive to oxygen depletion and more generally to poor water quality.

In the Mediterranean, age 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 km within 130 days. These movements of 1+ year class fish take place in very near shore waters and juveniles stay in shallow areas of the open sea. Bauchot *et al.* (1986) stated that sea bream had never been found at depths of more than 30 m. The migrations were not erratic, but were generally oriented northwards. In those movements, all the different populations mix.

Bigger fish appear to overwinter in rocky areas inshore, but their locations have not been well described. The proportion returning to lagoons decreases with age (Lasserre, 1976). Fish of more than 4 years are almost absent from 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 all attempts to investigate their behaviour have failed. Some individuals have been caught in water depths as great as 150 m.

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 fish stay in water depths of 5-15 m. Larger individuals are concentrated in deeper areas during winter, and may be caught at water depths up to 100 m.

Overfishing of this species has been reported in southern Spain (Sanchez-Lamadrid, 2002). In other locations, there is no evidence in the last ten years to indicate that the stock is under threat.

Growth

In the Mediterranean, growth is 20 to 30% higher than in sea bass during the first 2 years. The preferred temperature range for growth is 22-24°C, and growth ceases below 13°C. The Bay of Cadiz and coastal lagoons in the Gulf of Lion have been reported to be nursery areas.

The following table (from Lasserre, 1976) summarizes the growth (length in cm/weight in g) for females in these two areas, together with typical performance in cultivation.

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		

Diet

In contrast to sea bass, sea bream is omnivorous, preferring small arthropods and polychaetes in the young life stages, and 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 show 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.

Abundance

The abundance of this species is not well known, particularly because many landing statistics combine all sparid species into a single 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 has not been demonstrated, the standing stocks may be around 25 000 tonnes in this area. This species is caught at any size, from 100 g to 5 kg, and the average weight of the landings is not known.

Reproduction and spawning

Sea bream is a multi-spawning species, which releases eggs over a period of 2 to 3 months. The 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 part of its distribution. Sea bream is a protandric hermaphrodite species, that means that during the first 3 or 4 years of life the individuals are males, and then become females. The first maturation occurs at the age of 2. In the Mediterranean Sea, spawning areas are close to shore, in a maximum of 50 m depth. In the Gulf of Lion, sea bream all migrate during the autumn for hundreds of kilometres to the Rhone delta to reproduce in zones of 5–25 m depth, at temperature under 19°C (Lasserre, 1976). In the southern part of its Atlantic distribution (Spain), breeding areas appeared to be in 50–100m depth. Eggs are pelagic, but are confined under the halocline in winter (Divanach, 1985) where they are protected from the UV radiation.

Genetic structure of the population

Allozyme and microsatellite variations, and variation in mitochondrial DNA, have been studied in sea bream. Fish from six different locations from Portugal to Greece were analysed. Sea bream presented a high degree of genetic variability among wild populations (Alarcon *et al.,* 2004). The reason why such a high variability (2 to 10 times higher than in other sparid species) can be maintained remains unknown. A combination of molecular, demographic and evolutionary factors has been 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 patterns are probably not associated with geographic and/or oceanic factors (Palma *et al.,* 2001; Alarcon *et al.,* 2004). This appears also to be the case in other sparid species. However, the sampling protocols used may not have been sufficiently comprehensive, and additional studies are required to confirm this genetic status. More recently, De Innocentis (2004) suggested that three populations could be found in the Western Mediterranean (Sardinian Sea, Sardinian Channel and Central Tyrrhenian Sea), one in the Adriatic and one in the Atlantic, with a low level of differentiation.

Summary and related factors

All life stages may be impacted by accidental releases of cultured sea bream, either through competition for feed or through 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 a migratory species, and a natural genetic partitioning has not been demonstrated. A single case of a strain showing high growth rate, possibly due to a genetic and environment interaction, has been reported but has not been confirmed. 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 degree to which escapees could displace natural populations or reduce the feed supply is not known and cannot be derived from existing studies. It only can be said that pressure on trophic resources is more likely to occur during the period of high feeding intensity i.e. spring and summer. As released sea bream show the same feeding behaviour as wild fish, it is quite clear that the degree of competition will essentially depend upon the relative numerical abundance of wild and released fish in a given area. The degree to which the fitness of cultured fish may be compromised in the wild is not clear. However, there is some evidence to suggest that it could be less than wild conspecifics due to some physiological deficiencies (dorsal deformations, olfactory abnormalities, etc…) that have been reported (Mana and Kawamura, 2002).

Logic model

The series of steps and processes leading from the establishment of sea bream farms in coastal waters to significant decreases in wild sea bream stocks can be summarised in a logic model as below:

Process of concern:	**Changes in fitness of wild populations of sea bream due to genetic intergradation**
End Point of Concern –	**Significant decline in survival in wild sea bream populations due to interbreeding with escaped cultured sea bream.**

Logic model steps:

1) Sea bream farms are established in coastal waters.

2) There are phenotypic differences between the wild and cultured sea bream populations.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

4) The primary route for genetic interaction (interbreeding) between cultured and wild sea bream is through escapes of sea bream from cages.

5) Cultured sea bream escape from cages.

6) Cultured sea bream interbreed with wild sea bream.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

8) Sufficient gene flow to affect survival rates of sea bream in individual fisheries management units, i.e. the population structure of wild sea bream is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured sea bream causes significant declines in survival in wild sea bream populations.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e Escapes of cultured sea bream cause significant decreases in wild/feral sea bream stocks.

The information presented in the preceding sections of this risk analysis allows annotation of each step in the logic model to indicate the probability that each step has been, or will be, completed.

1) Sea bream farms are established in coastal waters.

Highly probable with low uncertainty. Sea bream farms are already widely established, particularly in the Mediterranean area. Production is expected to increase in the coming years, and may be carried out in more exposed locations on open coasts. Where active sea bream farms tend to aggregate the intensity of the farms will, over time, probably become higher. The intensity is therefore assessed as high.

2) There are phenotypic differences between the wild and cultured sea bream populations.

Moderately probable as the industry develops, with low uncertainty. Some loss of genetic diversity, in comparison with wild stocks, had been reported in farmed stock, probably arising from genetic drift. Present practice is to use wild-caught mature adults as broodstock, and there have been few attempts at genetic improvement. This will tend to mitigate against genetic differences between wild and farmed fish. The severity of the differences is therefore is currently low.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

Moderately probable as the industry develops, with low uncertainty. As indicated above, the current use of wild-caught broodstock will limit the potential for such differences to develop. Current indications therefore are that the differences have only low severity.

4) The primary route for genetic interaction (interbreeding) between cultured and wild sea bream is through escapes of sea bream from cages.

Highly probable with low uncertainty. Sea bream are protandric hermaphrodite, and only males are found in commercial cultivation. This will reduce the probability of genetic products released in cages to interact with wild populations. Deliberately released mature sea bream have been observed to interbreed with wild populations, and so escapes are the most likely mechanism for genetic interaction. As only males will be available as escapes, the severity is assessed as moderate.

5) Cultured sea bream escape from cages.

Highly probable with low uncertainty. It is highly likely that some sea bream escape from cages. The nibbling behaviour shown by sea bream towards fouling communities on the cages will also tend to cause damage to nets and increase the potential for escapes. Wild stocks may amount to around 25,000 tonnes, compared to farmed stock of around 100,000 tonnes (450.106 individuals). A small percentage of escapes may therefore amount to a few percent of the abundance of the wild stocks (moderate intensity).

6) Cultured sea bream interbreed with wild sea bream.

Highly probable with low uncertainty. Breeding of intentionally released mature fish with wild populations one year after release has been reported. They showed the same spawning behaviour as wild specimens, strongly suggesting that gene flows would occur between escaped cultured fish and wild populations. Field observation of interbreeding suggests high intensity of this step.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

Low probability with moderate uncertainty. There is no evidence to support this contention for sea bream, even though interbreeding has been observed. The genetic differences between wild and farmed stock are likely to remain low, and the stock is not highly differentiated. The intensity/severity of this step is also assessed as low.

8) Sufficient gene flow to affect survival rates of sea bream in individual fisheries management units, i.e. the population structure of wild sea bream is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

Low probability with low uncertainty. Knowledge of the population structure of sea bream suggests that there is little differentiation, and therefore the interaction will be of low intensity. If current aquacultural practices are continue, an increased number of farms in an area will not significantly alter this assessment.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured sea bream causes significant declines in survival in wild sea bream populations

Low probability with low uncertainty, and low intensity, as the effect of interactions is being considered over a greater number of fish.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e Escapes of cultured sea bream cause significant decreases in wild/feral sea bream stocks.

Low probability with low uncertainty, and low intensity, as the effect of interactions is being considered over a greater number of fish than in step 9 above.

STEPS IN THE LOGIC MODEL	INTENSITY (C,H,M,L, OR N) [1]	PROBABILITY (H,M,L,EL, OR N) [2]	UNCERTAINTY (H,M, OR L)
Step 1	H	H	L
Step 2	L	M	L
Step 3	L	M	L
Step 4	M	H	L
Step 5	M	H	L
Step 6	H	H	L
Step 7	L	L	M
Step 8	M	L	L
Step 9	L	L	L
Step 10	L	L	L
Final Rating [4]	L	L	M

Explanatory notes:

1 Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

2 Severity = C – very intense, H – high, M – Moderate, L – Low, N – Negligible

3 Uncertainty = H- Highly uncertain, M – Moderately uncertain, L – Low uncertainty

4 The final rating for the Probability is assigned the value of the element with the lowest level of probability.

5 The final rating for the Severity (intensity of interaction) is assigned the value of the step with the lowest risk rating (e.g., Medium and Low estimates for the logic model steps would result in an overall Low rating). Note that the calculation of the final rating follows the multiplication rule of probabilities (i.e., the severity that a given event will occur corresponds to the product of the individual severity). Thus the final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

6 The final rating for the Uncertainty is assigned the value of the element with the highest uncertainty level (i.e. the least certainty).

Risk Management

Option Evaluation

Option evaluation addresses what might be done to reduce the probability of a risk being expresses or our uncertainty in the prediction of the prediction of expression of a risk. A useful way to do this is to look at the logic model and for each step identify what can be done to reduce the probability of it occurring. These are steps to mitigate possible effects. The other contribution would be to reduce the uncertainty associated with predicting that the step will happen. Usually that involves further research or development. The table below identifies both mitigative actions and research or development steps that could be in addressing risks associated with sea bream culture.

	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Uncertainty	Research/Development
1	Sea bream farms are established in coastal waters	H	Where feasible move to land-based production	L	Develop economically competitive land-based technologies.
2	There are phenotypic differences between the wild and cultured sea bream populations.	M	For each generation recruit all grow-out stock from juveniles captured in the wild.	L	
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.	M		L	Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival.
4	The primary route for genetic interaction (interbreeding) between cultured and wild sea bream is through escapes of sea bream from cages.	H	Improve containment design and/or build in fail-safe measures	L	Identify factors that will limit dispersion of escapees
5	Cultured sea bream escape from culture	H	Improve containment design and/or build in fail-safe measures Recovery plan for escaped fish	L	
6	Cultured sea bream interbreed with wild sea bream	H	Use of sterile fish	L	
7	The progeny of this interbreeding (hybrids) show reduced fitness	L	For each generation recruit all grow-out stock from juveniles captured in the wild	L	Develop tools to identify wild/ farmed/ hybrid fish
8	Sufficient gene flow to affect survival rates of sea bream in individual fisheries management units, i.e. the population structure of wild sea bream is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.	L	Limit the distribution of sea bream farming to either proximity to small value stocks or very large stocks.	L	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between	L		L	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations

	wild and populations of escaped cultured sea bream causes significant declines in survival in wild sea bream populations.				
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e. Escapes of cultured sea bream cause significant decreases in wild/feral sea bream stocks	L	Limit the distribution of sea bream farming in relation to the distribution of the species or meta population	L	Identify dynamics of genome at the meta population or species level.

Regulation

Because the wild sea bream stock is not differentiated into localized populations, the use of the local strain for culture purposes need not be 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 the use of sea bream broodstock that have been selected for culture is progressively expanding throughout Europe. Care should be taken to maintain a high level of genetic diversity in farmed stocks, ie to avoid cultivated strains being genetically very different from their parents.

An effective long-term solution to prevent escapes would involve the development of more robust containment technologies. To encouraging implementation, authorities might consider economic incentives such as reduced site licensing fees. Ultimately, taxes on escapees could be also be used as an incentive to make farmers make additional investments in security.

At present, the reporting of sea bream escapees is not compulsory in any country. To reduce uncertainty, the need for regulatory enforcement, improved mandatory reporting should be introduced. Since there is no additional cost involved, it would be of benefit 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 should be encouraged.

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

The use of sterile fish to limit both gamete emission and possible interbreeding with the wild populations has frequently been recommended as a measure that would significantly reduce the risk of interbreeding with wild stocks. Sea bream triploids have been produced by some fish farms, with the aim of securing the genetic progress gained through selective breeding (Dosdat, pers.comm.). The results have not been published. Hybridisation with related species or genera (*Pagrus pagrus, Pagrus major, Dentex dentex*) may also be worthy of investigation. Hybrids have been reported to have lower survival rates, but the sterility of diploid hybrids has not yet been demonstrated. At present, it appears that government-based financial incentives might be necessary to introduce sterile technology. However, even that approach will fail if the product is not acceptable to the consumer.

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

Code of practice - Certification

In all cases, the training of the operators should be an essential preoccupation of the fish farmer. Being omnivorous, sea bream clean their nets by eating the bio-fouling community. The maintenance, replacement and monitoring of nets is of paramount importance to limit accidental escapes, particularly in the case of a nibbling species such as sea bream. Periodic inspection is essential, and particular attention should be given to net replacement, either for cleaning or to increase the mesh size. Being not cannibalistic, the sorting frequency for sea bream is not high. This species is not very susceptible to diseases, so the risks associated with fish manipulation are lower than in the case of sea bass.

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

- Advice on best practice for sorting and bath treatments.
- Advice on best practice for transport: apply to juvenile supply to the farm as well as to extracting adults at harvest
- Improved methods for net replacement
- Advice on best practice for mooring and anchoring, particularly in more exposed areas
- Training of personnel

Research

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

- Studies 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 bream 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 the behaviour of adults and juveniles released at off shore locations. It would be particularly useful to invest in this type of research now, before the cultured stock used by industry has had time to become further genetically differentiated from local wild stocks.
- The effect of photoperiod control in delaying maturation
- Another possible hypothesis, not specific to sea bream, could be to produce fish that genetically do not synthesized an essential dietary component that they can only find in artificial feed. This would make cultured fish unable to survive in the wild. However this solution is highly hypothetical and needs substantial theoretical developments (animal welfare, technical feasibility, etc), but it could also be applied to GMO fish if they ever come into regular use.
- Another way to decrease the impact of releases on wild stocks in a given environment is to maximise the wild stocks in the areas where farming activities take place, particularly if wild stocks are naturally scarce. This requires tools and activities to evaluate the abundance of wild stocks in culture areas.

- Tools to enable recognition of wild fish from escapees are not readily available, and new developments are necessary if they are to be used widely in monitoring programmes.

Contingency Planning

Recovering escaped sea bream around the cages within some days/weeks of escape seems difficult in this species, particularly if adults have been released. Fishing techniques could be developed (possibly including the use of specific or otherwise illegal devices under specific authorisation for this purpose).

The degree to which farms should be monitored must be a function of the degree of risk arising from the farming system. In this respect, the need generally decreases from sea ranching to offshore sea cages, inshore sea cages, flow through land based systems, closed systems and integrated systems.

Literature Cited

Alarcon, J.A., Magoulas, A., Georgakopoulos, T., Zouros, E., Alvarez, M.C., 2004. Genetic composition of wild and cultivated European populations of the gilthead sea bream (*Sparus aurata*). Aquaculture, 230: 65–80.

Arias, A., *1980.* Crecimento, regimen alimentario y reproduction de la dorada *(Sparus aurata* L.) y de1a robalo *(Dicentrarchus labrax* L.) en los esteros de Cadiz. Investigacion Pesquera, 44 : 59–83.

Bauchot, M.L., Hureau, J.C., 1986. Sparidae. Ed. by P.J. Withehead, M.L. Bauchot, J.C. Hureau, J. Nielsen, and E. Tortonese. *In* Fishes of the North-eastern Atlantic and Mediterranean, Vol. 2, Unesco, UK, pp 887.

Brown, R.C., Woollians, J.A., McAndrew, B.J., 2005. Factors influencing effective population size in commercial populations of Gilthead seabream (Sparus aurata). Aquaculture 247, 219–225.

Chauvet, C., Lasserre, G., Bach, P., Besseau, L., 1992. Résultats d'une experience de marquage de trios espèces de poisons côtiers du Golfe du Lion : *Dicentrarchus labrax*, *Sparus aurata* and *Liza aurata*. Cybium, 16, 3–11.

De Innocentiis S., Lesti A., Livi S., Rossi A.R., Crosetti D., and Sola L. 2004. Microsatellite markers reveal population structure in gilthead sea bream Sparus auratus from the Atlantic Ocean and Mediterranean Sea. Fisheries Science 70, 852–859.

De Innocentiis, S., Miggiano, E., Ungaro, A., Livi, S., Sola, L., and Crosetti, D. 2005. Geographical origin of individual breeders from gilthead sea bream (*Sparus auratus*) hatchery broodstocks inferred by microsatellite profiles. Aquaculture 247, 227–232

Gorshkov, S., Gordin, H., Gorshkova, G., and Knibb, W., 2002. Genetic improvement of sea bass and sea bream : practical applications. Isr. J. Aquac., 54 : 53.

Lasserre. 1976. Dynamique des populations ichthyologiques lagunaires : application à Sparus aurata L. PhD Thesis, University of Languedoc, 306 pp.

Mana, R..R., and Kawamura, G., 2002. A comparative study on morphological differences in the olfactory system of red sea bream (*Pagrus major*) and black sea bream (*Acanthopagrus schlegeli*) from wild and cultured stocks. Aquaculture, 209 : 285–306.

Palma, J., Alarcon, J.A., Alvarez, C., Zouros, E., Magoulas, A., and Andrade, J.P. 2001. Developmental stability and genetic heterozygosity in wild and cultured stocks of gilthead seagream. J. Mar. Biol. Assoc. U.K., 81: 283–288.

Paspatis, M., Markakis, G., Koumoudouros, G., and Kentouri, M., 1999. Preliminary results on rearing of *Sparus aurata* x *Pagrus pagrus* hybrids. Performance comparison with the parental species. Aquaculture International, 7: 295–306.

Sanchez-Lamadrid, A., 2002. Stock enhancement of gilthead sea bream (*Sparus aurata*, L.): assessment of season, fish size and pace of release in SW Spanish coast. Aquaculture, 210: 187–202.

Sanchez-Lamadrid, A., 2004. Effectiveness of releasing gilthead sea bream (*Sparus aurata*, L.) for stock enhancement in the bay of Cadiz. Aquaculture, 231: 135–148.

Suau, P., and Lopez, J., 1976. Contribution to the knowledge of the biology of the gilt-head sea bream *Sparus aurata*. Invest. Pesq., 40: 169–199.

Yamada, T., Yamaoka, K., and Taniguchi, N, 1992. Behaviour, distribution and population size change of artificially-bred-group red sea bream released in a small fishing port. Bull. Jap. Soc Sci. Fish., 58, 611–617.

Zouros, E., Paternello, T., Alvarez Herrero, MC., Andrade, JP., and Kentouri, M., 1998. A comprehensive genetic study of cultured and wild stocks of gilthead sea-bream, *Sparus aurata* and genetic assessment of several related species as candidates for aquaculture. Third European Marine Science and Technology conference (MAST conference), Lisbon, 23–27 May 1998: Project synopses Vol. 5: Fisheries and Aquaculture (AIR: 1990–94), p 114–115.

Annex 8: Risk analysis of the potential interbreeding of wild and escaped farmed Atlantic Halibut (*Hippoglossus hippoglossus* L)

Introduction

The Atlantic halibut (*Hippoglossus hippoglossus* L) is primarily a benthic species found on both the eastern and western sides of the North Atlantic. It feeds mainly on other fish, but also takes cephalopods, large crustaceans and other bottom-living animals. The maximum reported age is in excess of 50 years, with males reaching 240 cm and females 300 cm. However, its slow growth rate and late onset of sexual maturity give halibut populations low resilience. Commercial catches are now very low and the species is currently on the IUCN Endangered Species List. The high value and sedentary life style of halibut have made the species an attractive target for aquaculture development.

Risk analysis

Hazard Identification

Known Effects of Cultured Populations

Atlantic Halibut (*Hippoglossus hippoglossus* L) is a relatively new species for cultivation. The main development areas are Norway, Canada and Scotland. There have been no reported observations of interactions between escaped farmed Atlantic halibut and wild stocks.

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”.

Release assessment

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. Cultured halibut are generally marked at sizes of approximately 2 to 3 kg. 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 favorable 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 Artemia 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 3). Nearly all of the future aquaculture production will be conducted in net pens.

TableA8.1: Status and Projections for the Development of Halibut Aquaculture.

NATIONS	STATUS	PLANNED PRODUCTION
Norway	~700,000 juveniles (2002), ~1,000 tons production (2004)	9000 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

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 and 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.*, in press).

Exposure Assessment

Life History Description of Wild Populations

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 common 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.

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.

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 rapily and to a larger maximum size than males (Figure A8.1).

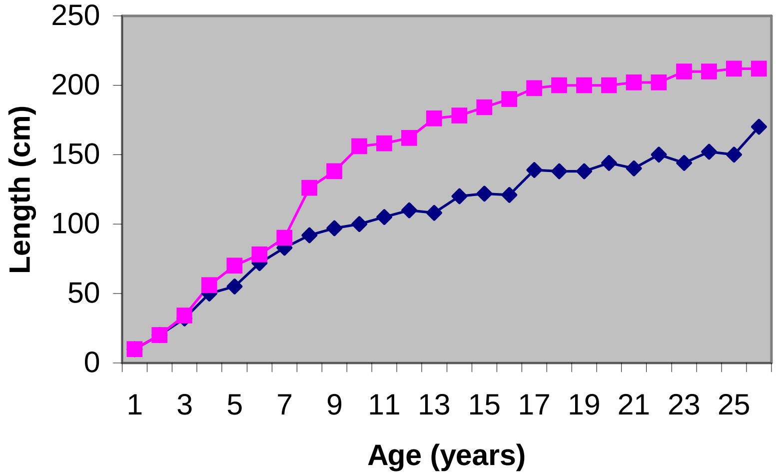


Figure A8.1: Observed growth of male (blue) and female (pink) halibut captured from a spawning area SW of the Faroe Bank in 1983-1986. (Jakupsstouv and Haug 1988)

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 A8.2).

Table A8.2: Optimal temperatures for growth of different class sizes of juvenile halibut (Jonassen *et al.*, 2000).

FISH SIZES (G)	OPTIMAL GROWTH TEMP. (°C)
5–10	14.9
20–25	13.9
40–50	13.0
60–70	12.7

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 1990).

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.

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 by which adult fish 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 2).

Table A8.3: Variability in Sexual Maturity in Halibut (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 between the years 1936–1960 and 1981–1985. In the 1936–1960 data set, 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 ^{o}C and salinities of 33.8–35.0. 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^{o}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).

Summary, and related factors

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 A8.3); size that are 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.

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. There is no information available on annual numbers of escapees from net pens. The wild fishes appear to have a low population density and a low level of genetic differentiation found throughout its range (Reid *et al.*, submitted). Most cultured halibut broodstock is only recently derived from wild stocks, and as such it has had little time to genetically diverge from the allelic patterns of the wild population.

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.

Logic model

The series of steps and processes leading from the establishment of Atlantic halibut farms in coastal waters to significant decreases in wild halibut stocks can be summarised in a logic model as below:

Process of concern:	Changes in fitness of wild populations of Atlantic halibut due to genetic intergradation
End Point of Concern:	Significant decline in survival in wild Atlantic halibut populations due to interbreeding with escaped cultured Atlantic halibut.

Logic model steps:

1) Halibut farms are established in coastal waters.

2) There are phenotypic differences between the wild and cultured halibut populations.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

4) The primary route for genetic interaction (interbreeding) between cultured and wild halibut is through escapes of halibut from cages.

5) Cultured halibut escape from cages.

6) Cultured halibut interbreed with wild halibut.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

8) Sufficient gene flow to affect survival rates of halibut in individual fisheries management units, i.e. the population structure of wild halibut is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured halibut causes significant declines in survival in wild halibut populations.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e Escapes of cultured halibut cause significant decreases in wild/feral halibut stocks.

The information presented in the preceding sections of this risk analysis allows annotation of each step in the logic model to indicate the probability that each step has been, or will be, completed.

1) Halibut farms are established in coastal waters.

Highly likely, with moderate uncertainty, as halibut farms are already established in Norway, Scotland and Canada, although the eastern Canadian industry is in land-based tanks. Considerable growth in production is planned for the coming years, but may be constrained by uncertainties in the production of juveniles. So, for the forseeable future, the number of active halibut farms will probably remain low, although once instituted they are likely to remain in place for an extended period. The geographical distribution will remain limited until production of juveniles is less constrained and more reliable. The intensity is considered to be low..

2) There are phenotypic differences between the wild and cultured halibut populations.

Moderately likely, with low uncertainty. There is little documentation of phenotypic differences between wild and cultured halibut. Individuals with partial albino pigmentation however, are not uncommon in the cultured stock. As the cause is known and is likely to affect the marketing of the product, it is unlikely to continue to be a constant feature of farmed halibut. Other stable pheonotypic differences will be selected for as the industry moves towards maintaining halibut in culture through its entire life-cycle. Thus the intensity of occurrence of this is seen to be moderate.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

Moderately likely, with low uncertainty. Present practice is mainly to use wild-caught mature adults as broodstock. This will tend to mitigate against genetic differences developing between wild and farmed fish. In the near future, it is likely that halibut cultivation systems will progressively move to full life cycle culture and enable selection for desirable traits (intentional or otherwise). Greater differences are then likely to develop between farmed and wild stocks. Given the limited differentiation in the original wild stocks, it seems likely that differentiation may occur slowly so the intensity of occurrence is rated as low. There is strong evidence that the albino pigmentation noted above is principally caused by a combination of dietary and environmental factors.

4) The primary route for genetic interaction (interbreeding) between cultured and wild halibut is through escapes of halibut from cages. .

Highly likely. Halibut spawn in deep water offshore, and so any possible (but unlikely) release of eggs or milt inside cages are very unlikely to interact with mature wild fish. Current commercial practice involves manual stripping of mature fish. Males are likely to mature in cultivation for the table, but females are not. Escapes will probably be the most plausible route for interaction. As the location of the farms and spawning patterns are not likely to change in the immediate future, the intensity will likely remain low. Uncertainty is considered to be low.

5) Cultured halibut escape from cages.

Highly probable. It is highly likely that some halibut will escape from cages through handling errors, storm damage, collision with vessels etc. The likely escape rate is difficult to estimate, and because of the differences in fish behaviour and cage design may be different from that experienced with salmon. Halibut are generally more sedentary in the cage than salmon. Thus they may be likely to less frequently explore the cage perimeter and find opportunities to escape. Domestication may further heighten this effect. Escapes are likely to be intermittent and much emphasis will be put on preventing them because of the economic loss associated with escaped fish. The number of farms is not likely to increase radically until the techology is better.

So the intensity of this factor is low. There is a moderate degree of uncertainty in this evaluation.

6) Cultured halibut interbreed with wild halibut.

Low probability. The separation of cultivation in coastal waters from spawning in deep offshore waters reduces the likelyhood of spawning interactions. The more domesticated the cultured stock become, the more it is likely that behavioural changes in cultured fish will further reduce the probable reproductive success of escaped individuals. At present however, if fish can locate the spawning grounds they would likely be able to succesfully interbreed. Escapes are likely to be intermittent across the limited number of farms. The intensity is considered low. There is a moderate level of uncertainty in this prediction.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

Highly probable, although there is no field evidence to support this contention for halibut. The only model of wild fish interbreeding with cultured individuals is the salmon experience and that would suggets that hybrid progeny would suffer an increased rate of mortality. Those hybrids could reasonably be expected to disappear over a number of years and the extent of mating yielding hybrid offspring would be expected to be low. Intensity is thought to be low. The lack of appropriate data inculcates a high degree of uncertainty.

8) Sufficient gene flow to affect survival rates of halibut in individual fisheries management units, i.e. the population structure of wild halibut is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

Moderately likely. Halibut is currently considered to be an endangered species and the total wild stock could be around 7800 tonnes (2002). There is limited evidence for structure within the N. Atlantic halibut population. The small numbers of wild fish suggest that a small number of interbreeding escapes could have the potential to affect the wild genome. However, frequency and distribution of opportunities for escapes suggest that the intensity will be low. Uncertainty is moderate. The lack of differentiation of subpopulations suggests this evaluation will be the same for the next two steps.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured halibut causes significant declines in survival in wild halibut populations

See step 8 above.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e., escapes of cultured halibut cause significant decreases in wild/feral halibut stocks.

See step 8 above.

SSTEPS IN THE LOGIC MODEL	INTENSITY (C,H,M,L, OR N) [1]	PROBABILITY (H,M,L,EL, OR N)[2]	UNCERTAINTY (H,M, OR L)
Step 1	**L**	**H**	**M**
Step 2	**M**	**M**	**L**
Step 3	**L**	**M**	**L**
Step 4	**L**	**H**	**L**
Step 5	**L**	**H**	**M**
Step 6	**L**	**L**	**M**
Step 7	**L**	**H**	**H**
Step 8	**L**	**M**	**M**
Step 9	**L**	**M**	**M**
Step 10	**L**	**M**	**M**
Final Rating [4]	**L**	**L**	**H**

Explanatory notes:

1 Probability = H – High, M – moderate, L – Low, EL – Extremely Low, N – Negligible

2 Intensity = C – very intense, H – high, M – Moderate, L – Low, N – Negligible There are three components of severity that should be commented on: the duration of the activity, the degree of change, and the geographic extent of the change.

3 Uncertainty = H- Highly uncertain, M – Moderately uncertain, L – Low uncertainty

4 The final rating for the Probability is assigned the value of the element with the lowest level of probability.

5 The final rating for the Severity (intensity of interaction) is assigned the value of the step with the lowest risk rating (e.g., Medium and Low estimates for the logic model steps would result in an overall Low rating). The final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

6 The final rating for the Uncertainty is assigned the value of the element with the most uncertainty level (i.e. the least certainty).

Risk Management

Option evaluation in risk management addresses what might be done to reduce the probability of a risk being expresses, or to reduce the uncertainty in the prediction of the expression of a risk. This can be addresses through consideration of the series of steps in the logic model discussed above. The process identifies, for each step, what could be done to reduce the probability of it occurring. These actions would directly mitigate possible effects. A further contribution to increasing the effectiveness of the risk analysis would be to reduce the uncertainty associated with predicting that the step will happen. Usually this involves further research or development. The table below identifies both mitigative and research or development steps that could be in addressing risks associated with genetic interactions arising from halibut culture.

	Logic Model Step	Mitigate (regulate/design/ modified practices)	Research/Development
1	Halibut farms are established in coastal waters	Where feasible move to land-based production	Develop economically competitive land-based technologies.
2	There are phenotypic differences between the wild and cultured halibut populations.	For each generation recruit all grow-out stock from juveniles captured in the wild	
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.		Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival.
4	The primary route for genetic interaction (interbreeding) between cultured and wild halibut is through escapes of halibut from cages.		
5	Cultured halibut escape from culture	Recovery plan for escaped fish Improve containment design and/or build in fail-safe measures	Identify factors that will limit dispersion of escapees Improve containment design and/or build in fail-safe measures
6	Cultured halibut interbreed with wild halibut	Use of sterile fish	
7	The progeny of this interbreeding (hybrids) show reduced fitness	For each generation recruit all grow-out stock from juveniles captured in the wild	
8	Sufficient gene flow to affect survival rates of halibut in individual fisheries management units, i.e. the population structure of wild halibut is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.		Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured halibut causes significant declines in survival in wild halibut populations.		Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e. Escapes of cultured halibut cause significant decreases in wild/feral halibut stocks		Identify dynamics of genome at the meta population or species level.

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 currently 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 from an expanded halibut aquaculture industry 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).

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 the risk to this species arising from fishing pressure is far more significant than the risk from any potential interbreeding with 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. The code should be designed to lead to the least negative impact and the most positive impact on the wild population of halibut.

In the North East Atlantic, 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.

Literature cited

Berg, L. 1997. Commercial feasibility of semi-intensive larviculture of Atlantic halibut (*Hippoglossus hippoglossus* (L.)). *Aquaculture*, 155: 333–340.

Bjornsson, B., and Tryggvadottir, S.V. 1996. Effect of size on optimal temperature for growth and growth efficiency of immature Atlantic halibut *(Hippoglossus hippoglossus* (L.)). Aquaculture, 142: 33–42.

Foss, A., Imsland, A.K., and Nævdal, G. 1998. Population genetic studies of the Atlantic halibut in the North Atlantic Ocean. Journal of Fish Biology, 53: 901–905.

Haug, T. 1990. Biology of the Atlantic halibut (*Hippoglossus hippoglossus* (L)). *Advances in Marine Biology*, 26: 1–70.

Haug, T., and Fevolden, S.E. 1986. Morphology and biochemical genetics of Atlantic halibut, *Hippoglossus hippoglossus* (L.) from various spawning grounds. Journal of Fish Biology, 28: 367–378.

Jakupsstova, S., and Haug, T. 1988. Growth, maturation and spawning season of Atlantic halibut, *Hippoglossus hippogossus* in Faroese waters. Fisheries Research, 6: 201–215.

Jonassen, T.M., Imsland, A.K., Fitzgerald, R., Bonga, S.W., Ham, E.V., Naevdal, G., Stefansson, M.O., and Stefansson, S.O., 2000. Geographic variation in growth and food conversion efficiency of juvenile Atlantic *halibut* related to latitude. Journal of Fish Biology, 56: 279–294.

McIntyre, A.D. 1953. The food of halibut from North Atlantic fishing grounds. Marine Research, Scotland, 1952(3): 1–20.

Mork, J., and Haug, T. 1983. Genetic variation in halibut, *Hippoglossus hippoglossus* (L.), from Norwegian waters. Hereditas, 98: 167–174.

Olsen, Y., Evjemo, J.O. and Olsen, A. 1999. Status of cultivation technology for production of Atlantic halibut (*Hippoglossus hippoglossus*) juveniles in Norway/Europe. Aquaculture, 176: 3–13.

Reid, D., Pongsomboon, S., Jackson, T., McGowan, C., Murphy, C., Martin-Robichaud, D. and Reith, M . Submitted. Microsatellite analysis indicates an absence of population structure among Atlantic halibut (*Hippoglossus hippoglossus*) in the northwest Atlantic. NRC Institute for Marine Biosciences, Halifax, N.S., Canada.

Rice, J. and Cooper, A. 2003. Management of flatfish fisheries-what factors matter? Journal of Sea Research, 50: 227-243.

Roff, D.E. 1982. Reproductive studies in flatfish: a first synthesis. Canadian Journal of Fisheries and Aquatic Sciences, 39: 1686–1698.

Sigurdsson, A. 1956. Contributions to the life history of the halibut at the west of Iceland in recent years (1936–1950). Meddelelser fra Danmarks Fiskeri-og Havundersogelser, Ny Serie 1(16): 1–24.

Stobo, W.T., Neilson, J.D. and Simpson, P.G. 1988. Movements of Atlantic halibut (*Hippoglossus hippoglossus*) in the Canadian North Atlantic. Canadian Journal of Fisheries and Aquatic Sciences, 45: 484–491.

Trumble, R.J., Neilson, J.D., Bowering, W.R., and McCaughran, D.A. 1993. Atlantic halibut and Pacific halibut and their North American fisheries. Canadian Bulletin of Fisheries and Aquatic Sciences 227, 84 pps. National Research Council of Canada.

Annex 9: Risk analysis of the potential interbreeding of wild and escaped farmed turbot *(Psetta maxima)*

Introduction

The turbot (*Psetta maxima*) is primarily a benthic species found throughout the Mediterranean and along the European coasts to Arctic Circle, including the Baltic Sea. It feeds mainly small benthic fish and to a lesser degree on invertebrates (bivalves and crustacean). The maximum reported age is 25 years, reaching 100 cm and 25 kg weight. It lives in waters of a wide range of salinities and at depths of 10–100m. It is of high value as a commercial fish, and is established in aquaculture in Europe, China and Chile.

Hazard identification

1.1. Genetic variability in broodstock

Only limited evidence exists for reduced genetic variability in farmed strains of turbot, as has been described for other cultivated fish species (Cross and King, 1983; Verspoor, 1988). Bouza *et al.* (1997) observed a reduction in heterozygosity in farmed strains of turbot in comparison with wild populations taken off the Norwegian coast and the Celtic Sea. This was also noticed by Coughlan *et al.* (1998) for farmed turbot from Norway and Ireland. Bouza *et al.* (2002) observed lower allozyme heterozygosity and loss of genetic variation in comparison with samples from the wild. The decrease in differentiation and divergence found in the farmed strains was believed to be caused by genetic drift during culture, due to the use of a limited number of broodstock animals. These results, however, can not be generalized, since broodstocks from other turbot farms in Galicia (Bouza, unpublished data) and France (Estoupe *et al.*, 1998) show much higher genetic diversity values, which were not different from the wild stocks.

Imsland and Jonassen (2001) observed that turbot was sensitive to the length of the light period, with longer light periods showing enhanced growth. But authors also revealed that growth in some cases was enhanced at lower temperatures and longer day lengths. Usually, warmer temperatures enhance growth. They concluded that a strong genotype by environmental interaction must be present.

1.2. Behaviour in the wild of released turbot

In the past, introductions of turbot have been carried out in the former USSR (1930) (FAO, 1997), in Iran (period 1930 – 1931) (Coad, 1995), and in Chili for aquaculture purposes (FAO, 1997, Pérez *et al.*, 2003), but with no successful recapture or establishment of breeding populations. Turbot, however, was successfully introduced (self reproducing) into waters around New Zealand (Muus and Nielsen, 1999). Experimental releases of cultured fry for stock enhancement purposes have been performed in Spain (Iglesias and Rodriguez-Ojea, 1994), Denmark (Nicolajsen, 1993; Støttrup and Paulsen, 1998), and Norway (Bergstad and Folkvord, 1998).

The Sea Fisheries Department in Belgium has started to investigate the possibilities of restocking commercial important North Sea flatfish species, e.g. turbot and sole (*Solea solea*). Turbot was chosen as the first candidate, as reproductive biology and rearing techniques for all life stages are fully understood and under control. Delbare and De Clerck (2000) obtained 3000 juveniles from a commercial fish farm: France Turbot – Adrien Group (Noirmoutier, France) and reared for another 6 months in the pilot nursery system of the Department Sea Fisheries – CLO (Ostend, Belgium). Before release, the juveniles were conditioned for two months to natural live prey organisms, e.g. brown shrimp (*Crangon crangon* L.) and sand gobies (*Pomatoschistus* sp.). Next to that, all juveniles were tagged with a Petersen disk

(Petersen, 1893). The tagged turbot were released in a for fisheries closed area (release position: 51°12’000 N and 02°45’600 E). Approximately 16% of the released turbot was reported back after a period of 1.5 years. At the end of 2004 more than 30% of the released turbot was reported. The migration pattern of the released turbot juveniles is presented in Figure 3. During the first two months after release, the juveniles remained in Belgian coastal waters following the main current towards the Dutch coast. The direction in the two following months (August-September’98) was clearly north – north-east, with the centre of capture on the Flemish sand banks. The same situation was found in October-November’98, although a portion of the animals was migrating into deeper water, i.e. the central part of the Southern North Sea. In the months December’98-January’99 some of the turbot were captured in the proximity of the “Thornton Bank”, while most migrated into deeper waters. Such an off-shore migration pattern, from shallow water during late spring and summer into deeper water during autumn and winter was also observed by Bagge (1987) for turbot in the Kattegat. In February-March’99, the major part of tagged turbot was still captured in deeper waters, with some found in more coastal waters (the Netherlands and the United Kingdom), but also into the Dover Straits, in the proximity of Bologne sûr Mer (France) and Port Rey (United Kingdom). This situation continued in the periods April-May’99, June-July’99 and August-September ’99. However, in the latter period, a concentration of turbot was seen again in the area around the “Thornton Bank”. Further captures (more than 30% of the released juveniles) were found scattered throughout the southern en central North Sea and the English Channel.

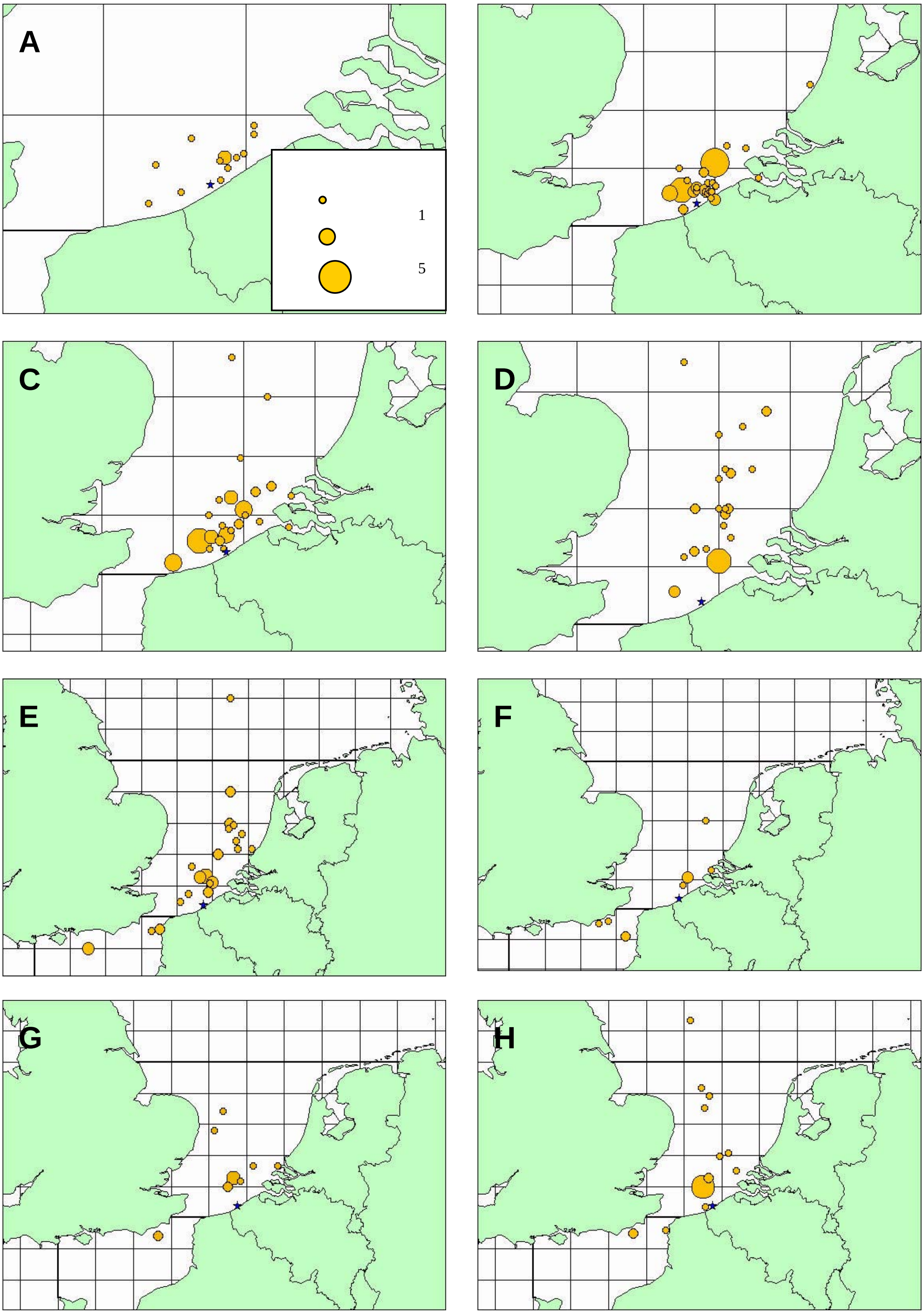


Figure A9.1. Distribution of the released turbot in time : A. June-July'98 ; B. August-September '98 ; C. October-November ; D. December '98-January '99; E. February-March '99 ; F. April-May '99 ; G. June-July '99 ; and H. August-September '99.

The general migration pattern of the released juvenile turbot followed a north – north-west direction into deeper waters of the North Sea, but with a migration to more coastal waters in late-spring and summer. Only a small portion migrated in south - south-western direction into the English Channel. Migration in northern direction started from October 1998 onwards, while tagged turbot in the English Channel were reported from February 1999 onwards. In tagging experiments with other flatfish species (plaice, dab and sole), it was also observed that a small portion migrated from the North Sea into the English Channel (De Clerck and Cloet, 1975; De Clerck, 1984). Growth rate was similar in comparison with the turbot in the wild, although these animals were initially bigger due to the high culture temperatures and *ad libitum* feeding. Other studies on released turbot revealed no differences in growth rate with their wild counterparts (Støttrup and Paulsen, 1998; Støttrup *et al.*, 1998a,b). The stomach analyses showed that the released turbot were able to adapt to the natural food sources. Turbot of the length class 21–23.9 cm fed exclusively on gobies (*Pomatoschistus* sp.). With increasing length, there is a change in prey spectrum, in which other bottom dwelling fish (e.g. lesser weever, *Trachinus vipera* and dragonet, *Callionymus* sp.) and brown shrimp (*Crangon crangon*) were eaten. From 30 cm onwards a significant change in feeding habit occurs, raging from consumption of benthic organisms to hunting for pelagic fish, e.g. bib, *Trisopterus luscus.* The monthly variation in condition factor showed that the animals well adapted to the natural conditions, with a condition factor between 1.8 and 2.2, which was comparable with the range in wild turbot populations (Ongenae and De Clerck, 1998). Furthermore, no major differences in condition factor was noticed between released and wild turbot in the research period.

Several restocking experiments with turbot showed that survival rate of reared turbot in the wild was very high. Survival can, however, be further enhanced by conditioning the reared juveniles to natural conditions. Reared turbot were found to exhibit lower cryptic behavior compared to their wild counterparts. After conditioning the reared animals to a sand bottom, the juveniles exhibited an improved cryptic behaviour and a more efficient burying technique (Støttrup and Nielsen, 1998). Stomach analyses on newly released turbot showed within two months after release lower stomach weights than wild fish of the same size. However, conditioning reared turbot to natural food increased the feeding success after release in the wild (Støttrup and Paulsen, 1998). Studies undertaken to estimate the carrying capacity of habitats along the European coastline revealed that the carrying capacity is rarely reached (van der Veer *et al.*, 1990; van der Veer *et al.*, 1991; Rijnsdorp *et al.*, 1992; Henderson and Seaby, 1994) and could therefore sustain small quantities of released or escaped fish.

1.3. Effect of interbreeding between wild and escaped/released fish

At present no studies have been carried out on the interactions between wild and reared turbot. But for other species extensive data on interbreeding between escaped and wild individuals are available.

Among the main concerns is the loss of genetic variability within and among populations, with a reduction in flexibility to respond to environmental changes. This becomes a serious problem when the genetic variation within a hatchery population is reduced due to inbreeding, selective breeding, or domestication. Even one generation of artificial spawning and hatchery rearing can cause shifts in the genetic make-up (genetic variability and composition), with often detrimental effects to fitness (Allendorf and Ryman, 1987; Cross, 1999).

Interbreeding between wild and escaped domesticated salmon has been observed by Crozier (1993), Webb *et al.* (1993) and Clifford *et al.* (1998). Carr *et al.* (1997) and Saegrov *et al.* (1997) even noticed that in some cases the majority of the fry production in a population was produced by escaped cultured females. Other studies show that for salmon in certain Scottish rivers at least 7% of the spawnings are attributed to farmed female salmon (OSPAR QSR, 2000). Studies with Atlantic salmon demonstrated a significant superior survival of wild strains compared to farmed and hybrid strains under the same natural stream conditions, which means that there is a reduced fitness of the progeny from interbreeding. Fleming and Einum (1997) reported that farming of Atlantic salmon generated rapid genetic change that altered important fitness-related traits relating to behaviour and growth. Skaala *et al.* (1996) reported that survival of young juveniles was nearly three times higher in wild brown trout than in hybrids of wild and introduced (and genetically distinct) trout. Reisenbichler and Rubin (1999) reviewed a number of studies on Pacific salmon and concluded that they provide strong evidence that fitness for natural spawning and rearing can be rapidly and substantially reduced by interbreeding between wild salmon and those produced by artificial propagation.

A difficulty with demonstrating outbreeding depression is that the severity of the action becomes evident in the second and subsequent generation hybrids. Only few studies have continued to monitor the interactions over longer time periods, e.g. Jorstad *et al.* (1994) with cod *Gadus morhua*, and McGinnity *et al.* (1997) with Atlantic salmon *Salmo salar*. Perez-Enriquez *et al.* (2001) studied the genetic diversity of red sea bream (*Pagrus major*) in western Japan, in order to investigate the effects of stock enhancement programs around Shikoku Island on the genetic differentiation among wild stocks. They found significant departures from Hardy-Weinberg equilibrium and significant pairwise F_{st} among locations, indicating genetic instability within this region. It was suggested that stock enhancement caused this genetic instability. For Pacific salmon, Reisenbichler and Rubin (1999) also observed genetic changes from stock enhancement, which affected the productivity and viability in wild stocks. The effect of interbreeding between wild and cultured could cause catastrophic results to wild population in the long run. High numbers of escapees that interbreed with small populations, like in salmonids, can cause genetic incompatibilities between parents, that does only occurs in the second generation, when recombination of the parental genes takes place (Smoker *et al.*, 2004). This, however, provides the possibility of increased hybrid formation until the second generation.

2 Risk Assessment

2.1 Release Assessment

2.1.2. Turbot in aquaculture

Turbot culture has developed rapidly in the last two decades, growing from 4 mt in 1984 to 6748 mt in 2003 in Europe. In China production is estimated at 3000 mt (approximately 33% of total turbot production) and 350 mt in Chile (approximately 4% of total turbot production). The majority of production systems for turbot are land-based recirculation systems for juveniles and ongrowing (Figure 4). Tank volumes can differ according to the farm and depends on the holding system in use. For example, small water volumes are used in “shallow raceway” systems or very high volumes of 3600 m^3 in Puraq`s Sunfish aquaculture (Cambados, Galicia). Maximum stocking densities are presented in Table 5.

Current production is approximately 4000 t per annum in Spain, and 1500 t in France. At a mean weight of 1 kg, these amount to 4.10^6 and $1.5\ 10^6$ individuals respectively.

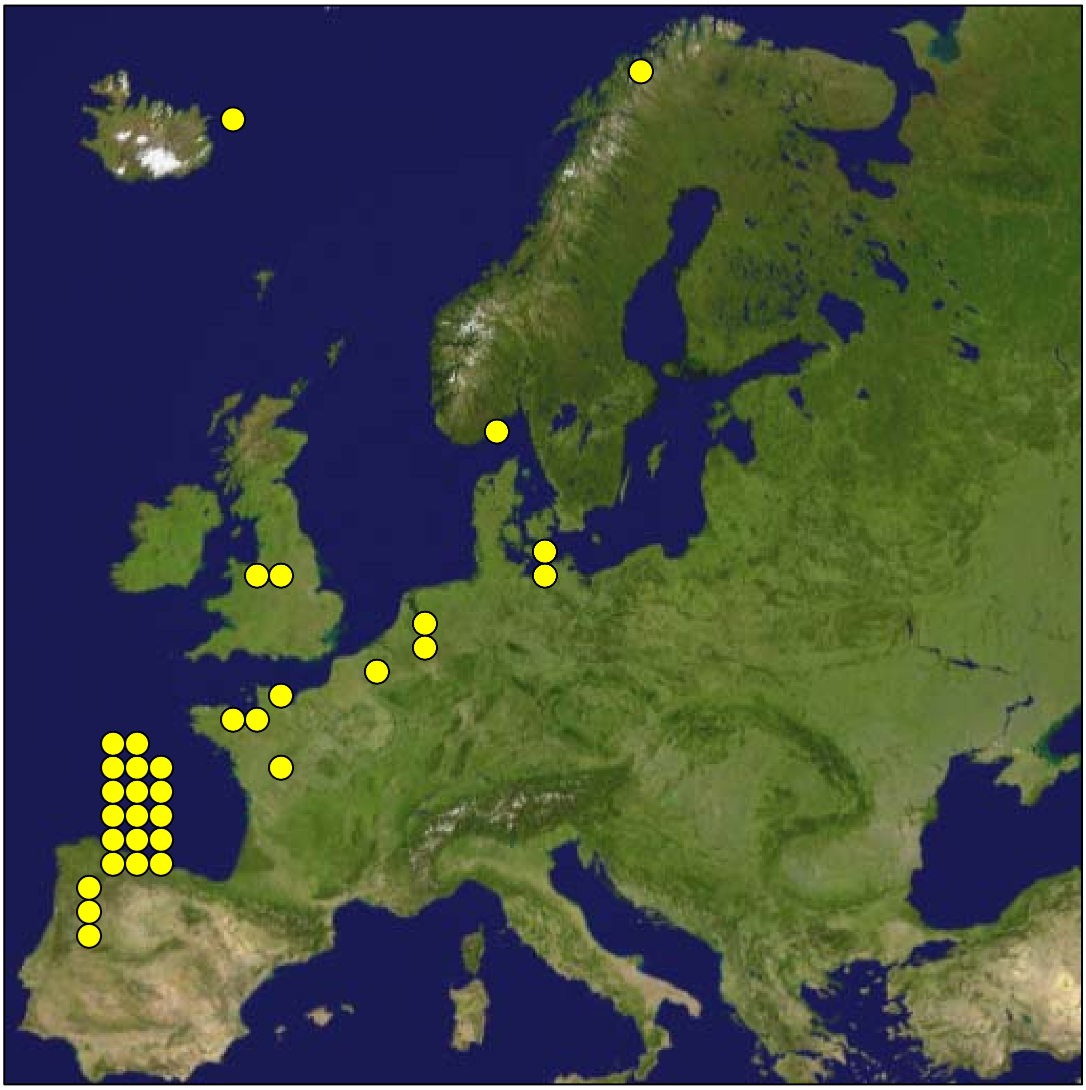


Table A9.1. Maximum stocking densities (kg.m^{-2}) for turbot (Cachelou, 1992; Kamstra, 1992).

Start weight of the fish (g)	End weight of the fish (g)										
	1	5	10	40	75	125	300	600	1000	2000	5000
1		5	5	5	5	5	5	5	5	5	5
5			5	5	5	5	5	5	5	5	5
10				10	10	10	10	10	10	10	10
40					20	20	20	20	20	20	20
75						20	20	20	20	20	20
125							30	30	30	30	30
300								40	40	40	40
600									50	50	50
1000										60	60
2000											60

2.2 Exposure Assessment

Turbot is a widespread species (from Morocco to Norway and into the Mediterranean Sea), but only in low abundances. Total annual turbot production equals the total landing (approximately 7000 mt) of this species, but is concentrated in only a few areas. This means that an accidental release could mean a very sharp increase in turbot numbers in one area.

From a study carried out on turbot by Boon *et al.* (2000) the turbot population size in the North Sea for the period 1981-1989 was estimated at approximately 11000000 individuals. In 1990, however, there was a strong recruitment estimated at 60000000 one year old turbot with a total stock number of 68000000 individuals. The mean CPUE for the North Sea increased after 1990 (Ongenae and De Clerck, 1998).

Taken into account that almost 75% from European turbot landings originates from the North Sea and the CPUE data from the Beam Trawl Survey showing for the English Channel, the Celtic Sea and the Irish Sea rarely 5 ind. per hour fishing in certain ICES rectangles, total stock numbers must be much lower in these areas than in the North Sea and are somewhere in the range of:

- 1 000 000 individuals for the eastern English Channel (0.7 kg per hour fishing);
- 660 000 individuals for the Celtic Sea (1.0 kg per hour fishing);
- 275 000 individuals for the Irish Sea (0.6 kg per hour fishing);
- 770 000 individuals for the Bay of Biscay;
- No estimation available for stock numbers for the Atlantic coast of the Iberian peninsula or the Mediterranean Sea.

It should be noted that these numbers are very crude approximations with considerable uncertainty.

No studies have been carried out on the interactions between wild and reared turbot, but interbreeding is most likely when the escaped/released turbot have matured, although it is not certain if these turbot have the sensory clues to migrate to spawning areas.

Turbot is a predator high on the trophic pyramid and release experiments have shown that reared turbot juveniles are very successful in adapting to conditions in the wild and have no problem in finding prey items (Støttrup and Paulsen, 1998, Støttrup *et al.*, 1998a; Delbare and De Clerck, 2000). The natural predator avoidance strategy in flatfishes is to flee to the bottom, bury into the sediment and remain motionless. It is expected that such cryptic behaviour is not as effective in reared fish as in their wild counterparts, since turbot is cultured in bare bottom tanks. In some cases even lengthy off-bottom behaviour is displayed by cultured Japanese flounder (Tsukamoto *et al.*, 1997). Reared turbot were found to exhibit lower cryptic behavior compared to their wild counterparts. Avoidance of predators through burying in the sand is lower in reared turbot than for their wild counterparts. After conditioning to sandy bottoms, cryptic behaviour can be improved significantly (Støttrup and Nielsen, 1998). Conditioning is only carried out prior to controlled release in the wild. According to Iglesias and Rodriguez-Ojea (1994), however, cultured turbot buried immediately in the sand upon release in their stock enhancement experiments.

Studies on released turbot in the North Sea showed that juveniles dispersed through the North Sea and for a lesser portion moving into the English Channel. Off-shore migration was seen during autumn and winter, while near-shore migration took place in spring and summer. Similar migration patterns of turbot were observed by Bagge (1987) for wild turbot in the Kattegat.

The main areas of overlap of farmed and wild stocks are the Bay of Biscay and the Channel. In these areas, the estimated number of fish produced in cultivation (5.5 10^6 individuals) exceeds the estimated numbers in the wild stock by a factor of approximately 3. No information is available on the actual number of escaped fish from the turbot farms. However, a small percentage of escapes would therefore amount to a larger proportion of the wild stock.

2.3 Consequence Assessment

2.3.1 Distribution

Turbot is distributed throughout the Northeast Atlantic Ocean along the European coastline and is rarer around the Faroe Islands, Iceland and on Rockall Bank. Turbot is also found in the Skagerrak, the Kattegat, the Belt Sea and in the Baltic Sea, but is very scarce in the Gulf of Bothnia, north of the Aaland archipelago, where salinity levels are below 5 psu. The distribution area also extends into the Mediterranean and Adriatic Sea. It is typically found at a depth range of 10 to 70m. Turbot lives on sandy, rocky or mixed bottoms and is one of the few marine fish species that inhabits brackish waters.

2.3.2 Growth and Survival

Turbot is one of the fastest growing flatfish. Only halibut grows faster. During the juvenile phase, growth rates are high, through which the turbot can reach 30 cm in three years. Females grow faster than males. During the first years of life, females grow from 8 to 10 cm a year. Females older than 10 years still grow 1 or 2 cm a year. In male turbot, the growth is already reduced to 2 cm a year at the age of 6 years. Males older than 10 years grow less than 1 cm a year. The difference in length between the sexes increases from 3 cm in 3-year-old turbot to 9 cm in 10-year-old turbot.

The maximum growth rates are obtained in 3, 4 and 5-year-old turbot during the summer (May till October). In these months growth can reach between 2 and 2.6 cm per month. This high rate is comparable with the growth in artificial circumstances. In nature, the ultimate growth rate (on year basis) is lower due to the slowing-down of metabolism during winter.

Ongenae and De Clerck (1998) concluded that in general no major differences in growth could be found among the areas under study. Males and females have a similar growth rate up to age 3. Hereafter the growth rate slows down in the males while the females continue their growth at a higher rate. Asymptotic lengths (L_∞) varied between 47.4 cm (North Sea) and 51.5 cm (Celtic Sea) for male turbot. For females the L_∞ ranged between 68.0 cm (eastern English Channel) and 74.4 cm (Celtic Sea). The asymptotic length thus attained highest values for both sexes in the Celtic Sea. The highest initial growth rate (characterised by the K-value) for both sexes was in the Bay of Biscay region. (Table A9.2).

Tabel A9.2: The von Bertalanffy growth parameters for turbot - $L(t) = L_\infty \{1-\exp[-K(t-t_o)]\}$.

Location	Sex	L_∞ (cm)	K ($year^{-1}$)	t_0 (year)	Reference
North Sea	Male	55.50	0.23	-0.20	Mengi (1963)
	Female	64.10	0.23	-0.16	
	Male	49.20	0.37	-0.51	Jones (1974)
	Female	64.80	0.26	-0.05	
	Male	50.92	0.33	-1.13	Weber (1979)
	Female	68.65	0.23	-0.67	
	Male	47.4	0.44	-0.20	Ongenae and De Clerck, 1998
	Female	74.2	0.19	-0.85	
Eastern English Channel	Male	49.7	0.47	-0.04	Ongenae and De Clerck, 1998
	Female	68.0	0.26	-0.27	
Bay of Douarnenez	Male	65.20	0.32	0.09	Deniel (1990)
	Female	73.60	0.28	0.08	
Celtic Sea	Male	51.5	0.41	-0.08	Ongenae and De Clerck, 1998
	Female	74.4	0.21	-0.44	
Irish Sea	Male	49.1	0.46	-0.14	Ongenae and De Clerck, 1998
	Female	71.5	0.22	-.054	
Bay of Biscay	Male	48.5	0.56	-0.01	Ongenae and De Clerck, 1998
	Female	71.5	0.27	-0.26	
Gulf of Lion (Med)	Male	54.3	0.24	-0.22	Robert and Vianet (1988)
	Female	55.6	0.31	-0.12	
Adriatic Sea	Male	67.7	0.27	-0.86	Arneri *et al.*, 1993
	Female	81.4	0.21	-0.99	
	Male	66.2	0.31	-0.14	Arneri *et al.*, 2001
	Female	81.5	0.21	-0.48	

The growth in weight indicated differences between some areas, but they appeared to be sex-dependent. When comparing the males, it became clear that North Sea turbot had the slowest growth. Bay of Biscay and eastern English Channel male turbot indicated higher initial growth rates while Celtic Sea and eastern English Channel male turbot reached the highest weights (2400 g). For the females, the highest final weights were recorded in turbot from the Celtic and North Sea (8000 g). The asymptotic weight was least for females from the eastern English Channel stock (6300 g) (Table A9.3).

Table A9.3: The von Bertalanffy growth parameters for turbot - $W = W_\infty \{1-\exp[-K(t-to)]\}^b$.

Location	Sex	W_∞ (kg)	K ($year^{-1}$)	t_0 (year)	b	Reference
North Sea	Male	1.91	0.44	-0.20	2.85	Ongenae and De Clerck, 1998
	Female	8.53	0.19	-0.85	3.11	
Eastern English Channel	Male	2.43	0.47	-0.04	3.04	Ongenae and De Clerck, 1998
	Female	6.33	0.26	-0.27	3.04	
Celtic Sea	Male	2.43	0.41	-0.08	3.10	Ongenae and De Clerck, 1998
	Female	8.04	0.21	-0.44	3.18	
Irish Sea	Male	2.14	0.46	-0.14	2.87	Ongenae and De Clerck, 1998
	Female	7.29	0.22	-0.54	3.10	
Bay of Biscay	Male	2.15	0.56	-0.01	3.22	Ongenae and De Clerck, 1998
	Female	6.93	0.27	-0.26	3.15	

Overviewing the parameters of the length/weight relationships for turbot from different regions, it became apparent that the females show a higher allometric coefficient than the males, and this phenomenon occurs in almost every region. Male turbot from the English Channel and Celtic Sea has somewhat higher b-values, which means a slightly higher body weight for the same length compared to other regions. Male turbot from the Bay of Biscay have the highest allometric coefficient and thus the highest weight/length ratio (Table A9.4).

Table A9.4: Weight-length relationships (W = a . Lb) for different areas and for each of the sexes (Ongenae and De Clerck, 1998).

Location	Sex	a	b	R^2
North Sea	Male	0.0325	2.8525	0.84
	Female	0.0133	3.1136	0.97
Eastern English Channel	Male	0.0173	3.0403	0.93
	Female	0.0168	3.0366	0.94
Celtic Sea	Male	0.0121	3.1016	0.97
	Female	0.0089	3.1845	0.98
Irish Sea	Male	0.0302	2.8714	0.96
	Female	0.0131	3.0998	0.98
Bay of Biscay	Male	0.0082	3.2182	0.94
	Female	0.0104	3.1538	0.96

2.3.3 Diet

Turbot is a typical visual feeder and feeds mainly on other bottom-living fishes (common gadoids, sand-eels, gobies, soles, dabs, dragonets, sea breams and boarfish), small pelagic fish (sprats, pilchards) and also, to a lesser extent, on larger crustaceans and bivalves. Large turbot (40 to 70 cm) feed from March till May excessive on herring and sprat (Rae and Devlin, 1972; Wetsteijn, 1981), to build up enough reserve for the subsequent spawning season. During the other nine months 50 to 70 % of the animals were found to have empty stomachs. This percentage was much higher than for most flatfish species. For example, a complete time of fasting, which is characteristic in the life cycle of lemon sole, *Microstomus kitt* is not observed in turbot (Rae and Devlin, 1972). The diet of the juveniles has been shown to consist of copepods, shrimps, barnacle larvae and gastropod mollusc larvae (Jones, 1973).

2.3.4 Genetic structure of the populations

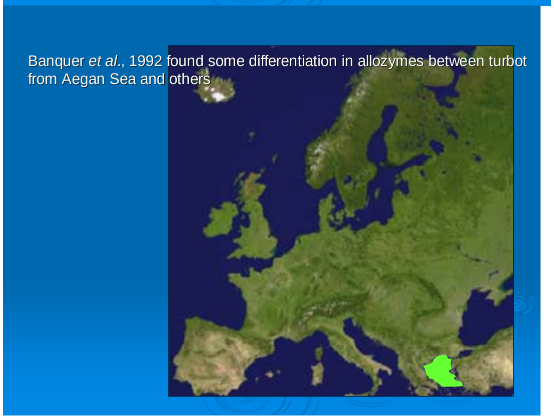
Only limited research on genetic stock analysis on turbot has been performed. In 1986, Renaud *et al.* (1986) showed in a study on allozymes of the cestode parasite, *Bothriocephalus gregarious*, a significant differentiation between the parasites from Atlantic and Mediterranean host turbot. The separation between these two forms was located in southern Portugal, between Lisbon and Faro. Allozyme analysis on 17 loci revealed almost no genetic differences within the complete distribution area turbot, only samples from the Aegean Sea were different from the others (Mediterranean to Kattegat), but with a negligible genetic distance as a result (Blanquer *et al.*, 1992). Also Bouza *et al.*, 1997 found, by the use of 14 allozyme markers a low genetic variability ($P = 0.012$) in both natural and hatchery populations. Imsland *et al.* (1994) did research on blood samples from turbot caught along the Norwegian coast, in the Kattegat, and from the Southwest coast of Iceland. They found some genetic differentiation ($P<0.01$ for Hb-1) based on haemoglobin polymorphisms between Norwegian/Icelandic turbot and turbot from the Kattegat. Studies done with three microsatellite loci on wild and farmed turbot originating from two different locations (Norway and Ireland – Celtic Sea and the Western Approaches) also revealed a lack of significant differentiation between the two wild populations (Coughlan *et al.*, 1998), which is consistent with the low level of genetic differentiation found in the allozyme studies (Blanquer *et al.*, 1992; Bouza *et al.*, 1997). However, Coughlan *et al.* (1998) stressed the importance of further genetic analysis with more microsatellite loci to screen wild turbot across its distribution area. Bouza *et al.*, (2002) found, employing 12 microsatellite and 28 allozyme loci, no differentiation between turbot from the Atlantic Ocean (Burela – 43°40’N, 7°22’W) and the Cantabric Sea area (Vilagarcia – 42°36’N, 8°45’W), areas which are separated by a major oceanographic discontinuity (Harden Jones, 1968). Recent studies carried out by Nielsen *et al.* (2004) on turbot from the Northeast Atlantic and the Baltic Sea (from the Bay of Biscay to the Aaland archipelago) suggests that the presence of multiple hybrid zones in the transition zone (Skagerrak, Kattegat and Belt Sea) between the high saline North Sea and the low saline Baltic Sea. The differentiation between turbot from the North Sea and the Baltic Sea was also observed by Karås and Klingsheim (1997) based on the effects of temperature and salinity on embryonic development of turbot from the two areas. Further research on population structure in the distribution area of turbot was undertaken by Boon *et al.* (2000). The preliminary study, using four microsatellites, showed that turbot from the English Channel appears genetically indistinguishable from the Bay of Biscay. Also turbot from the North Sea was not indistinguishable from the Celtic Sea, while turbot from the Irish Sea appeared to be genetically different from turbot from all other areas under research (Table 4).

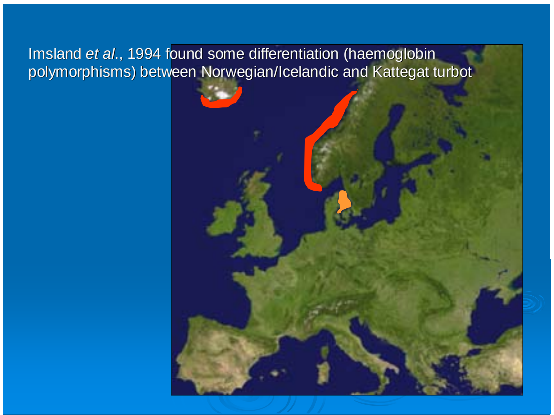
Table A9.5: Matrix of genetic distance (DA) estimates above the diagonal and P-values below the diagonal, between turbot from different fishing grounds (Boon *et al.*, 2000).

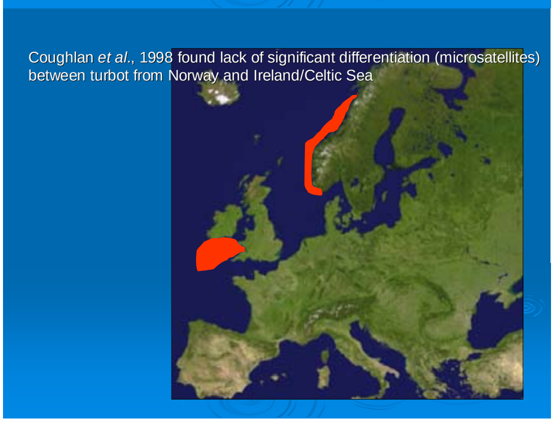
	North Sea	English Channel	Celtic Sea	Irish Sea	Bay of Biscay
North Sea	-	0.169	0.151	0.220	0.171
English Channel	0.052	-	0.196	0.220	0.120
Celtic Sea	0.111	0.005	-	0.235	0.208
Irish Sea	0.002	0.000	0.000	-	0.195
Bay of Biscay	0.019	0.367	0.002	0.001	-

Although samples sizes were small (20 samples per area) and these estimates must be considered as very preliminary, it appears likely, according from the results of the statistical analysis, that there exists a turbot population in the Irish Sea which is genetically different from fish from the other areas studied. This was also noticed by Ongenae and De Clerk (1998) analysing the fishing and landing parameters. There was also a difference found (although not so significant) between turbot from the Celtic Sea and the North Sea, and turbot from the English Channel and the Bay of Biscay. The low genetic differentiation between the North Sea/Celtic Sea and English Channel/Bay of Biscay is caused by the low genetic differentiation between the samples from the English Channel and those from the North Sea. This could mean that the English Channel acts as a transition zone between the Bay of Biscay and the North Sea. Tagging experiments on several flatfish species (turbot, plaice, dab and sole) indicated migrations of small portions from the North Sea into the English Channel (De Clerck and Cloet, 1975; De Clerck, 1984; Delbare and De Clerck, 2000). A similar transition or hybrid zone was found between the North Sea and the Baltic (Nielsen *et al.*, 2004) (Figure A9.2 below).


Renaud *et al.*, 1986 found significant differentiation (allozymes) in *Bothriocephalus gregarius* from Atlantic and Mediterranean host turbot, (between Lisbon and Faro).


Banquer *et al.*, 1992 found some differentiation in allozymes between turbot from Aegan Sea and others


Imsland *et al.*, 1994 found some differentiation (haemoglobin polymorphisms) between Norwegian/Icelandic and Kattegat turbot


Coughlan *et al.*, 1998 found lack of significant differentiation (microsatellites) between turbot from Norway and Ireland/Celtic Sea

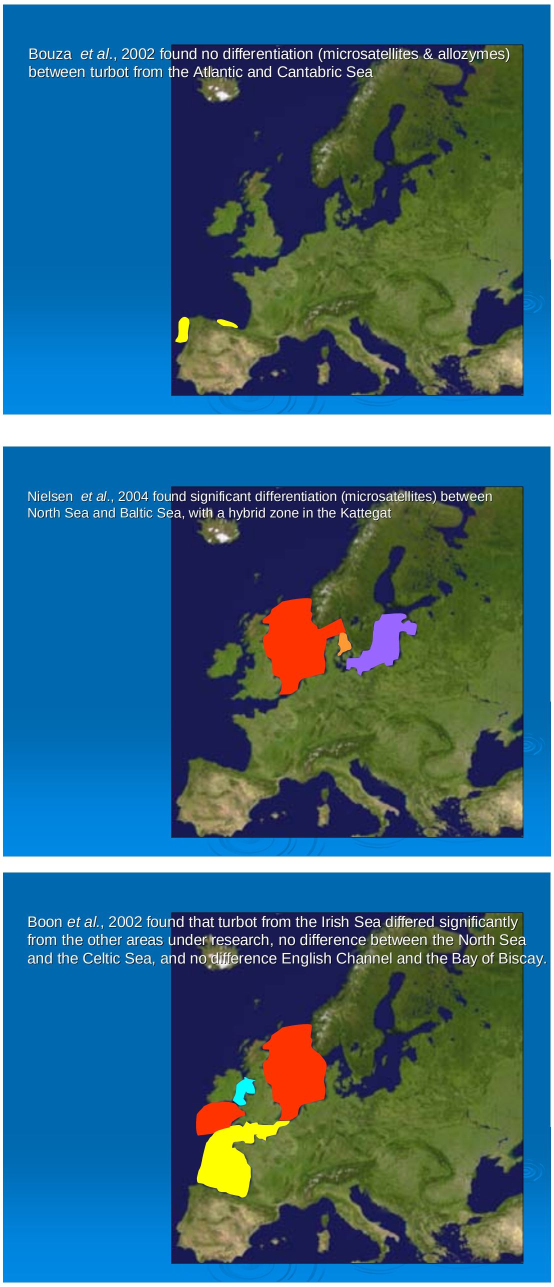


Figure A9.2: Areas which were studied and showed genetic differention.

Compiling all data from different studies, it becomes clear that there are distinct turbot populations in the Baltic Sea and in the Irish Sea. Furthermore, there are indications that turbot from the North Sea, the southern coast of Iceland, the western coast of Scotland and Ireland, and the Celtic Sea (including the Western Approaches - 51°N, 10°W) forms another stock, the northern Atlantic stock. This is different from the stock originating from the Bay of Biscay and the Atlantic site of southern Europe, the southern stock. Transition zones between the northern stock and the southern stock are found in the English Channel and between the northern stock and the Baltic Sea in the Kattegat and the Belt Sea. The situation of turbot stocks in the Mediterranean is still unclear, although there are indications that samples from the Aegean Sea are genetic different from those originating from other areas (Figure 2).

2.3.5 Abundance

Ongenae and De Clerck (1998) observed, from the annual catches per unit effort, that the CPUEs for the North Sea and Celtic Sea with 1.0–1.2 kg/hour fishing were higher than for the

English Channel, Bay of Biscay and the Irish Sea, with 0.5–0.8 kg/hour fishing. Data from the annual Beam Trawl Surveys indicated a high abundance of turbot along the continental coast from Belgium to Denmark, with strong concentrations at the Dogger Bank and near the Wadden Sea and in the German Bight, and to a lesser extent the Scheldt estuary. In the English Channel, Celtic and Irish Sea, the overall abundance of turbot appears to be lower than in the North Sea. Other flatfish, such as sole mostly appear very abundant in the Thames estuary on the UK coast, but this was not the case for the turbot. It could be noted that turbot mainly occurred along the continental coasts of the North Sea. In the central and western part of the North Sea, turbot was much less abundant or even absent. Usually, no turbot were caught in the central part of ICES-region IVb. Catches in the International Bottom Trawl Surveys showed pronounced occurrence of turbot in the central parts of the North Sea and a lower abundance in the German Bight. Another remarkable difference between both survey types lies in the number of turbot caught per rectangle. These were substantially lower for the bottom trawl surveys. For these surveys, the occurrence of turbot along the east coast of the UK was observed in the years 1991-1995. This was not the case for the beam trawl surveys. Year to year comparisons for both surveys pointed out that overall abundance has decreased significantly over the years.

2.3.6 Migration

In general, turbot is rather a sedentary species, but there are some indications of migratory patterns. For example in the North Sea, migrations from the nursery grounds in the south-eastern part to the more northern areas have been recorded, since adult turbot is more tolerant of the colder conditions in northern areas where temperatures are too low for juveniles to survive. A study in the northern Baltic (Aneer and Weston 1990) also indicated that adult turbot might be considered to be very stationary. In this project, a large number of turbot were tagged and released. After recapture, the average distance between first capture and recapture appeared to be very short; only 6 km. Furthermore, more than 90% of the recaptured turbot were caught less than 20 km away from the point of first capture.

It is not clear if juveniles released at some distance from their point of origin would return to their original home grounds.

2.3.7 Reproduction and spawning

Turbot exhibit no sexual dimorphism. The cyclical pattern of reproduction is characterised by massive gonad development and morphological changes (volume and colour), particularly of the ovaries, immediately before the emission of the gametes. In late spring to early summer, males and females gather on spawning beds, which are generally situated above gravel bottoms on the continental shelf. Fish with ripe gonads have been taken in trawls on the North Sea during the months April to July; ripe eggs have been found in the plankton from April to August (Malm, 1877; Möbius and Heincke, 1883; Brook, 1886; Ewart and Fulton, 1889; Fulton, 1892; Holt, 1892). Jones (1974) reported the occurrence of ripe gonads between May and August. In the English Channel, the spawning season is rather long, viz. from May to September (Lahaye, 1972; Deniel, 1990). The eggs are released during the night in one batch and fertilisation is external and at random. It is unclear if there are distinct spawning grounds in the areas identified above as holding distinguishable stocks, or if repeat spawners, or offspring from a spawning event, show fidelity to a spawning area.

The fertilised eggs are buoyant and their diameter varies between 0.9 and 1.2 mm. These eggs are extremely numerous: depending on the size of the female, their number ranges from 5 million up to 10 million per individual. The size-specific fecundity is rather constant. After spawning and feeding season, the turbot moves again to deeper waters.

First maturity for turbot in the North Sea is between ages 4 and 5 for females and age 3 for the males. This conclusion is drawn from a range of studies. Kyle (1926) determined maturity at age 6 or 7 for males as well as females. This (false) result went of course hand in hand with an incorrect age-length key. Ehrenbaum (1936) estimated first maturity at age 5 for both sexes. Length at maturity was determined by this author at 28 cm for the males and 35 cm for the females. Mengi (1963) estimated maturity at age 3, which corresponds to a length of 29-31 cm for males and 35-38 cm for females. In Rae's study (1972), maturity of the females was attained between 31 and 45 cm between the age of 4 or 5. Age of maturity for the males was set between ages 3 and 4. Jones (1974) determined length, weight and age at which 50% of the females reached maturity as follows: 46.01 cm; 2001g and 4.46 years. For males, a length at maturity of 30 cm was recorded. Deniel (1990) determined age and length at first maturation for the females in the English Channel at age 4 and 49 cm.

2.3.8 Further development

The fertilised eggs are carried to the shores by the currents. After aapproximately 7 days, the eggs hatch. At hatching, the larvae are 2.1–2.8 mm (Barnabé); 2.7-3.0 mm (Jones, 1972), 2.14–2.80 mm (Russell, 1976); 2.3–2.8 mm (Al-Maghazachi and Gibson, 1984) in length. Newly hatched turbot larvae possess a large yolk sac containing a single oil globule. This results in the larvae floating upside-down near the water surface during their first 6–12 h of life. At this time the larvae are largely inactive but may occasionally perform energetic wriggling movements. Larval growth and yolk utilisation are affected by temperature. The pelagic phase lasts around 60 days at 16°C (early summer). At the end of the larval phase, the fish undergo metamorphosis, develop asymmetry, and descend to the bottom. Metamorphosis takes place at a length between 13–25 mm (NN); 23 mm (Jones, 1972); 27-39 mm (Jones *et al*, 1974); 38–45 mm (Al-Maghazachi and Gibson, 1984); 19.8 mm (Fukuhara *et al*, 1990). The rates at which morphological changes occur during larval development are partly under genetic control and partly reflect the influence of environmental factors such as temperature, diet and water quality.

Five major developmental stages can be recognised and are characterised as follows:

Stage 1: Larvae symmetrical, yolk sac present

Stage 2: Larvae symmetrical, development of spines and air bladder

Stage 3: Appearance of fin rays, notochord straight

Stage 4: Asymmetry and eye migration, notochord posteriorly slanted dorsally

Stage 5: Completion of eye migration, spines and swim bladder resorbed.

There is no sharp distinction between the successive stages; in general at least half of the features characteristic of a particular stage must be developed before the onset of the next stage. For example, the right eye does not commence its migration until most of the fin rays have formed and the notochord within the caudal fin is inclined dorsally by 45° or more (Al-Maghazachi and Gibson, 1984).

The young fish, carried by the currents towards the shore, start a benthic existence. The juvenile turbot gather together on intertidal nursery grounds, where they remain throughout the summer months. In autumn, they migrate from the coastal areas to deeper waters in the more Northern regions. The juvenile phase is characterised by a high growth rate.

The authors are not aware of any information on the relationship between the total population numbers and genetically effective population size of turbot.

2.3.9 Logic model

The series of steps and processes leading from the establishment of turbot farms in coastal waters to significant decreases in wild turbot stocks can be summarised in a logic model as below:

Process of concern:	**Changes in fitness of wild populations of turbot due to genetic intergradation**
End Point of Concern:	**Significant decline in survival in wild turbot populations due to interbreeding with escaped cultured turbot.**

Logic model steps:

1) Turbot farms are established in coastal waters.

2) There are phenotypic differences between the wild and cultured turbot populations.

3) These phenotypic differences arise primarily for genetic rather than environmental reasons.

4) The primary route for genetic interaction (interbreeding) between cultured and wild turbot is through escapes of turbot from cages.

5) Cultured turbot escape from cages.

6) Cultured turbot interbreed with wild turbot.

7) The progeny of this interbreeding (hybrids) show reduced fitness.

8) Sufficient gene flow to affect survival rates of turbot in individual fisheries management units, i.e. the population structure of wild turbot is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.

9) Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured turbot causes significant declines in survival in wild turbot populations.

10) Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e., escapes of cultured turbot cause significant decreases in wild/feral turbot stocks.

The information presented in the preceding sections of this risk analysis allows annotation of each step in the logic model to indicate the probability that each step has been, or will be, completed.

1. Turbot farms are established in coastal waters.

Highly probable with low uncertainty. Turbot farming is a well established mariculture activity in Europe and some growth in production is foreseen in the near future. Where active turbot farms tend to aggregate (e.g. currently in NW Spain), it is likely that the density of the farms (and production) will increase. However, they will occupy only part of the wild turbot's coastal habitat, and a proportion (perhaps large) will continue to be shore-based tank systems. Once in place, the farms tend to become a long term feature of the coastal environment, but can be moved or removed. For this evaluation, the intensity in this step is considered to be low.

2. There are phenotypic differences between the wild and cultured turbot populations.

Highly probable as the industry develops, with moderate uncertainty. In many turbot farms it is the practice to use wild-caught mature adults as broodstock (Bouza, unpublished data;

Estoupe *et al.*, 1998). There is however evidence that turbot farmers are selecting juveniles with high growth rates and less malpigmentation, in order to increase production outputs. Furthermore, several turbot farmers are obtaining fish from a few selected hatcheries. For turbot, some evidence of lower allozyme heterozygosity and loss of genetic variability exists in farmed strains of turbot (Bouza *et al.*, 1997; Coughlan *et al.*, 1998; Bouza *et al.*, 2002) in some cultivation areas, but not in others. The intensity of this is currently assessed as moderate.

3. These phenotypic differences arise primarily for genetic rather than environmental reasons.

Moderately likely as the industry develops, with moderate uncertainty. As indicated above, the use of wild-caught broodstock will limit the potential for such differences to develop, but differentiation is being favoured by a move to culturing turbot through their entire lifecycle and applying selection for desired traits. The differences will to a greater extent be genetic as farmers fish place greater reliance on genetics to differentiate populations in the farm environment. This is likely to be an ongoing feature of turbot husbandry throughout the industry, i.e. a high severity of change.

4. The primary route for genetic interaction (interbreeding) between cultured and wild turbot is through escapes of turbot from cages. .

Highly probable with low uncertainty and low intensity. Although turbot is mainly cultured in land based systems on recirculation, escapes to the sea are possible through outlets in flow-through systems (when used) or by accident during sorting and handling of turbot. Further interaction with wild stocks could occur through accidental release of fertilized eggs to the environment, since most incubation tanks are run in an open flow-through system. The risk of escape will increase if culture systems are changed from on-land based systems to sea cage culture. This is likely to occur over a restricted portion of the wild populations range but will probably continue for the forseeable future. The intensity of this step is moderate.

5. Cultured turbot escape from cages.

Highly probable with low uncertainty and low intensity. It is very likely that some turbot will escape from land based farms but in very small numbers, especially in land based flow through systems, and also during sorting and handling. The risk becomes much higher for net cage cultured turbot, but it is probable that the rate of escape will still be much lower than experienced with salmon.

6. Cultured turbot interbreed with wild turbot.

Moderately probable with high uncertainty. Although there have been no studies carried out on the interactions between wild and reared turbot, interbreeding is most likely when escaped turbot have matured. Juvenile turbot released into the wild have been shown to exhibit only small differences in feeding and cryptic behaviour (for a short period after release). Released turbot showed a similar off-shore migration during autumn and winter, and a near-shore migration in spring and summer, as seen in wild turbot (Bagge, 1987). However, it is not certain that escaped turbot have the necessary sensory clues to migrate to spawning areas. Direct evidence of interbreeding in turbot is not available, but other observations suggest that it may occur. The intensity is therefore assessed as high.

7. The progeny of this interbreeding (hybrids) show reduced fitness.

Low probability with moderate uncertainty. There is no evidence to support this contention for turbot. With the development of differences between wild and cultured fish, the immediate severity of reduced hybrid survivorship is low. At present, the concentration of farms is such the gene flow into these populations is likely to be intermittent, quantitatively small and the genetic differences are currently small so the interaction will be of low intensity (though with an increased number of farms in an area this could increase). As with the probability, this may increase over time. The intensity for this step is considered moderate.

8. Sufficient gene flow to affect survival rates of turbot in individual fisheries management units, i.e. the population structure of wild turbot is such that the rate of interbreeding is sufficient to affect population fitness, at managed stock level.

Low Probability with moderate uncertainty. There is no evidence to support this contention for turbot. The preponderance of land-based cultivation systems greatly reduces the probability of escapes, and also the likely intensity of gene flow. The severity is therefore assessed as low.

9. Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured turbot causes significant declines in survival in wild turbot populations

For the same reasons as in step 8, the probability is low with moderate uncertainty. There is no evidence to support this contention for turbot. The severity is likely to remain low as the effect is spread over a greater number of fish and the uncertainty of this prediction remains moderate.

10. Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population, i.e Escapes of cultured turbot cause significant decreases in wild/feral turbot stocks.

For the same reasons as in step 8, the probability is low, but with low uncertainty because the effect is spread over an even greater number of fish. There is no evidence to support this contention for turbot. The severity is likely to be low and, because the effect is spread over an even greater number of fish, the uncertainty is decreased.

sSteps in the logic model	Intensity (C,H,M,L, or N) [1]	Probability (H,M,L,EL, or N) [2]	Uncertainty (H,M, or L)
Step 1	L	H	L
Step 2	M	H	M
Step 3	H	M	M
Step 4	M	H	L
Step 5	H	H	L
Step 6	H	M	H
Step 7	M	L	M
Step 8	L	L	M
Step 9	L	L	M
Step 10	L	L	L
Final Rating [4]	L	L	H

Explanatory notes:

1. **Probability = H** – High, **M** – moderate, **L** – Low, **EL –** Extremely Low, **N –** Negligible
2. **Severity = C** – very intense, **H** – high, **M** – Moderate, **L –** Low, **N** – Negligible
   There are three components of severity that should be commented on: the duration of the activity, the degree of change, and the geographic extent of the change.
3. **Uncertainty = H-** Highly uncertain, **M** – Moderately uncertain, **L** – Low Uncertainty
4. The final rating for the **Probability** is assigned the value of the element with the **lowest** level of probability.
5. The final rating for the **Severity** (intensity of interaction) is assigned the value of the step with the **lowest** risk rating (e.g., **Medium** and **Low** estimates for the logic

model steps would result in an overall **Low** rating). The final value for severity for each specific risk is assigned the value of the lowest individual logic model estimate.

6. The final rating for the **Uncertainty** is assigned the value of the element with the **highest uncertainty** level (i.e. the least certainty).

3 Risk evaluation

Without regulations or farm management practices specific to turbot farms there is unlikely to be any difference between the outcome of the Consequence Assessment and that of the Risk Evaluation. Risk management may be able to alter the values in the above table.

4 Risk management

4.1 Option evaluation

Option evaluation addresses what might be done to reduce the probability of a risk being expresses or to reduce the uncertainty in the prediction of the prediction of expression of a risk. A useful way to do this is to look at the logic model and for each step identify what can be done to reduce the probability it occurring. These are steps to mitigate possible effects. The other contribution would be to reduce the uncertainty associated with predicting the probability that the step will happen. Usually that involves further research or development. The table below identifies both mitigation and research or development steps that could be in addressing risks associated with genetic interactions arising from turbot culture.

	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Uncertainty	Research/ Development
1	Turbot farms are established in coastal waters	H	Where feasible maintain the current emphasis on land- based production	L	Ensure that land-based technologies remain economically competitive.
2	There are phenotypic differences between the wild and cultured turbot populations.	H	For each generation recruit all grow-out stock from juveniles captured locally in the wild	M	Development of tools to distinguish escapes and hybrid fish from wild population
3	These phenotypic differences arise primarily for genetic rather than environmental reasons.	M		M	Determine if differences are primarily genetic rather than environmental in origin. Determine if differences are associated with differential survival. Development of tools to distinguish escapes and hybrid fish from wild population
4	The primary route for genetic interaction (interbreeding) between cultured and wild turbot is through escapes of turbot from	H	Improve security of access points for the fish to water supply systems and improve handling practices to minimise losses	L	Identify factors that will limit dispersion of escapees

	Logic Model Step	Probability	Mitigation (regulate/design/ modified practices)	Uncertainty	Research/ Development
	tanks/ponds/cages.				
5	Cultured turbot escape from culture	H	Improve containment design and/or build in fail-safe measures Recovery plan for escaped fish	L	
6	Cultured turbot interbreed with wild turbot	M	Use of sterile fish	H	Studies to determine the number and survival of escapees Determine the frequency and success of interbreeding
7	The progeny of this interbreeding (hybrids) show reduced fitness	L	For each generation recruit all grow-out stock from juveniles captured in the wild	M	Studies of the performance of hybrids in the field
8	Sufficient gene flow to affect survival rates of turbot in individual fisheries management units, i.e. the population structure of wild turbot is such that the rate of interbreeding is sufficient to affect population fitness, at the population or meta-population levels.	L	Limit the distribution of turbot farming to either proximity to small value stocks or very large stocks.	M	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between stocks
9	Genetic interaction caused declines in endemic, evolutionarily significant units (populations), i.e. Genetic interaction between wild and populations of escaped cultured turbot causes significant declines in survival in wild turbot populations.	L		M	Identify those population units that have significant potential to respond to selection. Define rate of gene flow between populations
10	Gene flow is pervasive and persistent enough to affect fitness at the level of species or meta-population , i.e. Escapes of cultured turbot cause significant decreases in wild/feral turbot stocks	L	Limit the distribution of turbot farming in relation to the distribution of the species or meta population	L	Identify dynamics of genome at the meta population or species level.

4.2 Regulation

- There is some indication that there exists several (sub)populations, which probably have there own optimal growth temperature and salinity range, especially for the Baltic and the Irish Sea. However, it is advisable to use as broodstock animals from those stocks that are best fitted for that specific culture location. Decrease in genetic differentiation and divergence was found in the farmed strains, therefore special breeding programs must be set up to guarantee a high level in heterzygosity in farmed strains. Furthermore, it is important to use broodstock animals for restocking that are related to the local turbot (sub)population, in order to minimize adverse genetic interactions with the wild stock..

- The use of triploid turbot would reduce the risk of interactions with the wild stock. Experiments with hybrids between turbot and brill have been carried out to produce only female offspring (Purdom and Thacker, 1980).

- To limit escapees, physical barriers must be installed in all outlets of open flow through systems. Double mesh screens must be installed in the outlet of broodstocks at all times, to prevent loss of fertilized egg. Closed recirculation techniques can further reduce the risks of escapes.

- Particular attention should be paid to robust containment technologies for sea cages, when cage culture of turbot would become feasible.

4.3 Code of practice – certification

- In all cases, the training of operators should be an essential preoccupation of the fish farmer. The maintenance and cleaning of tanks and, in case of cage culture, the replacement and monitoring of nets, is of the outmost importance to limit accidental escapes. Periodic inspection of tanks (outlets and physical barriers) and nets should be compulsory. Special attention should be paid to the procedures of sorting and treatment operations.

- At present, declaration of turbot 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, it would be profitable to both the industry and the environment. In Ireland, salmonid farmers are obliged, immediately following any escape of reared salmonids from a freshwater or marine installation, to fill out a *Reared Fish Escapees — Incident Report Form* and contact the Department of Communications, Marine and Natural Resources (DCMNR), Marine Institute and relevant Regional Fisheries Board(s). The operator is required to report the number of escapees and cause of the escape, if known. The DCMNR collates this information with a view to making recommendations to try and prevent other incidences from happening. Nevertheless, there are no accurate data available for the number of escapes in Ireland. Voluntary Codes concerning escapes: aquaculture industry self-regulation and environmental safeguards through voluntary Codes are effectively worthless forms of governance in the absence of binding legal obligations to enforce rules (See Regulation of Marine Aquaculture). Concerning stock health management, it is a recommended action under the Code to implement the Irish Salmon Growers' Association (ISGA) Code of Practice for the prevention of stock escapes of Irish farmed salmonids (reproduced in Annex III of ECOPACT). The ECOPACT document also annexes the Federation of European Aquaculture Producers (FEAP) Code of Conduct for European Aquaculture.

- EU policy on escapes: in its Communication "A strategy for the sustainable development of European aquaculture" (COM(2002) 511 final, 19/9/02) the European Commission states that 'escaped fish inter-breeding with native populations may induce long-term damage by the loss of genetic diversity'. The Commission proposes developing instruments to tackle the impact of escapees as part of the EU Aquaculture Strategy and states that it 'has financed research on the threats to the diversity of wild Atlantic salmon caused by farm escapees, but further studies are needed. The process started in February 2000 by NASCO and the North Atlantic salmon farming industry to develop guidelines to minimise salmon escapees is particularly worthy of support. The Commission will examine whether such guidelines should be implemented by way of compulsory rules and may extend them to other fish species and strains.'

4.4 Direct Risk Mitigation

Some measures can be identified to reduce the risk of interbreeding of wild and cultivated stocks, e.g.

1) keep all culture on land

Some experiments have been carried out in Scotland for cage culture turbot, but with limited success. Recirculation techniques are improved to be able to culture turbot on full recirculation in order to enhance the control and reduce the dependence on natural water resources and heating costs. Nowadays, most of the turbot is farmed at 75-80% recirculation. It is expected that in the next five years all turbot will be farmed at 100% recirculation. The number of escapees and probability of escapes would be reduced, as would any subsequent the interbreeding between wild and farmed turbot. With land based culture of turbot on full recirculation, the uncertainty of escapes would be negligible.

2) Use sterile fish

Manipulation of sex and ploidy are spreading in fish farming. All-female production is economically advantageous, because in many species female growth rate is higher than that of the males, and first maturation takes place at older age. The market value declines drastically with maturity. For example, all female rainbow trout *Oncorhynchus mykiss* are cultured in Europe (Ingram, 1988) in order to reach greater market size before maturing. Tests have been carried out using hybrids of turbot with brill, *Scophthalmus rhombus*. These hybrids showed a higher survival rate during larval development and metamorphosis. Hybrids between these two species can also been found in nature. Holt (1892) noted three hybrids caught in the North Sea differed significantly from turbot or brill in body form, color, scale and number of fin rays. Hybrids formed between a female turbot and a male brill were all females and could reach a weight of 382 g at natural temperatures. Hybrids formed between a male turbot and a female brill resulted in a all male stock and grew to 289 g in 20 months at the same temperatures (Purdom and Thacker, 1980). Successful experiments to obtain an all-female stock were carried out in the UK and are interesting for the farmer since they exhibit a 12% higher growth rate and only rudimentary development of the ovaries. In turbot females, ovaries can take up 15% of the total body weight (Bye, 1981) and gonad development can divert much energy from somatic growth. Induced triploidy is also used to produce sterile fish, which continue to grow somatically (Ingram, 1988). The severity of the interaction between wild and all-female turbot would remain the same, but the probability would be reduced to extremely low or negligible. The uncertainty would be extremely low to negligible. In Europe, experiments with trout and turbot (Vázquez *et al.* 1996; Cunado *et al.*, 2002; Terrones *et al.*, 2004.) have been carried out, although for turbot this technique is not used on commercial farms.

3) Create dependence on specific food supplements that are not readily available in the wild

Another possible technique is to produce genetically modified turbot that are incapable to synthesize certain nutritional components which are not available in nature. Reared turbot would therefore be totally reliant on the artificial diets given in captivity to provide these essential nutrients. Once a fish has escaped from the rearing system, it could not survive in the wild due to this planned dietary deficiency. This technique would reduce the severity of escapes. The probability that the escapee would find food containing the essential nutrients depends on the chosen component, but could be very low. The uncertainty could be low to moderate, taking into account the good knowledge of natural prey items. This feeding technique is still highly hypothetical and needs substantial theoretical development (animal welfare, technical feasibility, GMO regulations, human welfare, etc.).

5 Contingency Planning

The degree of monitoring all activities on the farm must be a function of the degree of risk in relation to the farming system. In this respect there is a decreasing need for intensive control from sea ranching to inshore sea cages, flow through land based systems to closed recirculation culture systems. Recovery of large number of escapees in a certain area could be carried out by using gill nets in the area shortly after the accident has occurred.

5.1 Research

- Better information on total numbers, spawning areas and the genetic structure of wild stocks is needed to evaluate the severity, probability and uncertainty of the interaction of domesticated escapees on wild populations.

- Studies are needed to determine the number and survival of escapees in the natural environment (as functions of season and location; the impact of escaped turbot in summer may be different from that in winter, due to the presence of natural predators, e.g. cod migrating to the north in winter). This is needed to evaluate the severity, probability and uncertainty of the interaction of domesticated escapees on wild populations.

- Development of tools to distinguish wild fish from wild population. For turbot, the morphology of reared turbot is slightly different to that of their wild counterparts. In turbot, as in other flatfish species, morphological differences are primarily seen as malpigmentation on the blind side to patches with lack of pigment and white pigment on the eyed side. Such malpigmented turbot are found in the wild but at much lower frequencies than in cultured turbot, as it is determined by the larval diet and possibly by the rearing conditions. Further morphological characteristics, like the general form of the turbot, is highly influenced by the stocking densities in the culture tanks. Tagging could be used, but due to high stocking densities in the tanks, these tags could harm other turbots. The most common chemical compounds used to mark otoliths are alizarin compounds (Beckman and Schultz, 1996; Tsukamoto, 1985), calcein (Brooks *et al.* 1994; Wilson *et al.* 1987) and oxytetracycline (Dabrowski and Tsukamoto, 1986; Nagiec *et al.* 1988; Schmitt, 1984). Studies that address what levels of escapees will cause problems for local populations and their impact on the different life-cycle stages of their wild counterparts are required. Discrimination between reared and wild turbot is required to assess impacts after an accidental release of farmed turbot in the wild, in order to reduce the severity.

5.6 Literature Cited

Al-Maghazachi, S.J., and Gibson, R. 1984. The developmental stages of larval turbot, *Scophthalmus maximus* (L.). J. Exp. Mar. Biol. Ecol. 82: 35–51.Aneer, G. and Weston, L. 1990. Migration of turbot (*Psetta maxima* L.) in the northern Baltic proper. Fisheries Research 9: 307–315.

Arneri, E., Colella, S., and Giannetti, G. 2001. Age determination and growth of turbot and brill in the Adriatic Sea: reversal of the seasonal pattern of otolith zone formation. Journal of Applied Ichthyology 17(6): 256.

Bagge, O. 1987. Tagging of turbot and brill in the Kattegat 1965–1970. ICES C.M. 1987/G:10, 27 pp.

Beckman, D.W., and Schulz, R.G. 1996. A simple method for marking fish otoliths with alizarin compounds. Transactions of the American Fisheries Society, 125: 146–149.

Bergstad, O.A., and Folkvord, A. 1998. Dispersal of tagged juvenile turbot *Scophthalmus maximus* on the Norwegian Skagerrak coast. Fisheries Research.

Boon, A., Delbare, D., Ongenae, E., Heesen, H., De Clerck, R., and Rijnsdorp, A. 2000. By-catch species in the North Sea flatfish fishery (data on turbot and brill). Preliminary assessment. Final Report DG XIV 97/078.

Bouza, C., Presa, P., Castro, J., Sanchez, L., and Martinez, P. 2002. Allozyme and microsatellite diversity in natural and domestic populations of turbot (*Scophthalmus maximus*) in comparison with other Pleuronectiformes. Canadian Journal of Fisheries and Aquatic Sciences 59: 1460–1473.

Bouza, C., Sanchez L., and Martinez, P. 1997. Gene diversity analysis in natural populations and cultured stocks of turbot (*Scophthalmus maximus* L.). Animal Genetics, 28: 28–36.

Brook, G., 1886. The spawning period of British food-fishes. Fourth annual report of the fishery board of Scotland for the year, 1885: 242–254.

Brooks, R.C., Heidinger, R.C., and Kohler, C.C. 1994. Mass-marking otoliths of larval and juvenile walleyes by immersion in oxytetracycline, calcein, or calcein blue. North American Journal of Fisheries Management, 14: 143–150.

Bye, V., 1981. Sex control, a method for improving productivity in turbot farming. Fish Farmer International 8(3): 31–32.

Carr, J.W., Anderson, J.M., Whoriskey, F.G., and Dilworth, T. 1997. The occurrence and spawning of cultured Atlantic salmon (*Salmo salar*) in a Canadian river. ICES J. Mar. Sci., 54: 1064–1073.

Clifford, S.L., McGinnity, P., and Ferguson, A. 1998. Genetic changes in Atlantic salmon (*Salmo salar*) populations of Northwest Irish rivers resulting from escapes of adult farm salmon. Can. J. Fish. Aquat. Sci., 55: 358–363.

Coad, B.W. 1995. Freshwater fishes of Iran. Acta Sci. Nat. Acad. Sci. Brno., 29(1): 1–64.

Coughlan, J.P., Imsland A.K., Galvin, P.T., Fitsgerald, R.D., Naevdal, G., and Cross, T.F. 1998. Microsatellite DNA variation in wild populations and farmed strains of turbot from Ireland and Norway: a preliminary study. Journal of Fish Biology, 52: 916–922.

Cross T.F. and King, J. 1983. Genetics effects of hatchery rearing in Atlantic salmon. Aquaculture 33: 33–40.

Crozier, W.W. 1993. Evidence of genetic interaction between escaped farmed salmon and wild Atlantic salmon (*Salmo salar* L.) in a Northern Irish river. Aquaculture 113: 19–29.Cunado, N., Terrones, J., Sanchez, L., Martinez, P. and Santos, J.L. 2002. Sex-dependent synaptic behaviour in triploid turbot, *Scophthalmus maximus* (Pisces, Scophthalmidae). Heredity 89(6):460–464.

De Clerck, R. 1984. Tagging results of mature dab in the Southern Bight. ICES CM. 1984/G:11.

De Clerck, R., and Cloet, M. 1975. Tagging experiments on juvenile plaice off the Belgian coast. ICES CM. 1975/F:25.

Delbare, D., and De Clerck, R. 2000. Release of reared turbot in Belgian coastal waters as a tool for stock enhancement. Presentation of the paper ICES CM 2000/O:02 at the 2000 ICES Annual Science Conference, 28 september 2000.

Deniel, C. 1990. Comparative study on growth of flatfishes on the west coast of Brittany (France). J. Fish Biol. 37 (1): 149–166.

Ehrenbaum, E. 1936. Naturgeschichte und wirtschaftliche Bedeutung der Seefische Nordeuropas, Bd. II. E. Schweitzerbart''sche Verlagsbuchhandlung. 879 pp.

Ewart, J.C., and Fulton, T.W. 1889. Report on the spawning of the British marine food-fishes. Seventh annual report of the fishery board for Scotland for the year 1888, 3 : 186–221.

Estoup, A., Gharbi, K., San Cristobal., M., Chevalet, C., Haffray, P., and Guyomard, R. 1998. Parentage assignment using microsatellites in turbot (*Scophthalmus maximus*) and rainbow trout (*Oncorhynchus mykiss*) hatchery populations. Can. J. Fish. Aquat. Sci. 55: 715–725.

Ewart, J.C., and Fulton, T.W. 1889. Report on the spawning of the British marine food-fishes. Seventh annual report of the fishery board for Scotland for the year 1888, 3 : 186–221.

FAO, 1997. FAO Database on Introduced Aquatic Species. FAO Database on Introduced Aquatic Species, FAO, Rome.

Fleming, I.A., and Einum, S. 1997. Experimental tests of genetic divergence of farmed from wild Atlantic salmon due to domestication. ICES J. Mar. Sci. 54: 1051–1063.

Fukuhara, O., Rosenthal, H., Witt, U., and Quantz, G. 1990. Development and behaviour of larvae and juveniles of turbot, Scophthalmus maximus L., in hatchery tanks. ICES CM/1990/F:18. 21 pp.

Fulton, T.W. 1892. Observations on the reproduction, maturity, and sexual relations of food fishes. Tenth annual report of the Fishery Board for Scotland for the year 1891. 3 : 232–242.

Harden Jones, F.R. 1968. Fish migration. Edward Arnold (Publ.) Ltd., London.

Holt, E.W.L. 1892. Survey of fishing grounds, west coast of Ireland, 1890–1891. Reports on the scientific evidence bearing on the economic aspects of the fishes collected during the survey. Scientific proceedings of the Royal Dublin Society. 7 (4) : 388–477.

Iglesias, J. and Rodriguez-Ojea, G. 1994. Fitness of hatchery-reared turbot, *Scophthalmus maximus* L., for survival in the sea: first year results on feeding, growth and distribution. Aquaculture and Fisheries Management 25: 179–188.

Imsland, A. and Jonassen, T.M. 2001. Regulation of growth in turbot (*Scophthalmus maximus* Rafinesque) and Atlantic halibut (*Hippoglossus hippoglossus* L.): aspects of environment x genotype interactions. Reviews in Fish Biology and Fisheries 11: 71–90.

Imsland, A.K., Scanu, G. and Naevdal, G. 1996. New variants of the hemoglobins of turbot (*Scophthalmus maximus*): possible use in population genetic studies and aquaculture. ICES paper C.M. 1996/F:14 Ref. G. 9 pp.

Jones, A. 1973. Ecology of young turbot, *Scophthalmus maximus* L., at Borth, Cardiganshire, Wales. J. Fish Biol. 5: 367–383.

Jones, A. 1974. Sexual maturity, fecundity and growth of the turbot *Scophthalmus maximus* L.. J. Mar. Biol. Ass. U.K. 54: 109–125.

Karås, P. and Klingsheim, V. 1997. Effects of temperature and salinity on embryonic development of turbot (*Scophthalmus maximus* L.) from the North Sea, and comparisons with Baltic populations. Helgoländer Meeresuntersuchungen 51: 241–247.

Kyle, H.M. 1926. Beobachtungen über das Wachstum des Steinbutts (*Rhombus maximus*). Der Fisherbote. 8. Jg.

Lahaye, J. 1972. Cycles sexuels de quelques poissons plats des côtes bretonnes. Révision des Traveau de l'Institut des Pêches maritimes 36: 191–207.

Malm, A.W. 1877. Göteborgs och Bohunstläns fauna. Ryggradadjuren. Göteborg. 674 pp.

Mengi, T. 1963. Uber das Wachtum des Steinbutts in der Nordsee. Ber. Dt; Wiss. Komm. Meeresforsch. 17 (2): 119–132.

Möbius, K. and Heincke, F. 1883. Die Fische der Ostsee. Jahresbericht der Commission zur Wissenschaftlichen Untersuchung der Deutschen Meere, in Kiel, 1877–1881. Jahrg. 7–11, 197–296.

Muus, B.J. and J.G. Nielsen, 1999. Sea fish. Scandinavian Fishing Year Book, Hedehusene, Denmark. 340 p.

Nicolajsen, H. 1993. Udsaetninger af pighvarrer I Limfjorden, ved Langeland og ved Nordsjaelland 1989–1992. DFH-rapport, Denmark, Nr. 453a, 1993: 1–78 (in Danish).

Ongenae, E. and De Clerck, R. 1998. Collection and modeling of biological data for turbot assessment in support of the common fisheries policy. Final Report DG XIV 95/039.

Purdom, C.E. en Thacker, G., 1980. Hybrid fish could have farm potential. Fish Farmer 3(5) : 35.

Pérez, J.E., Alfonsi, C., Nirchio, M., Muñoz, C. and Gómez, J.A. 2003. The introduction of exotic species in aquaculture: a solution or part of the problem? Intersciencia. 28(4):234–238.

Perez-Enriquez, R. Takemura, M., Tabata, K. and Taniguchi, N., 2001. Genetic diversity of red sea bream *Pagrus major* in western Japan in relation to stock enhancement. Fisheries Science 67: 71–78.

Rae, B.B. and Devlin, S.D.E. 1972. The turbot, its fishery and biology in the Scottish area. Mar. Res. 1: 1–27.

Ramster, J.W., Medler, K.J. and Jones, S.R. 1976. Residual drift regimes in the Southern Bight of the North Sea during the Joint North Sea Data Acquisition Project of 1973. ICES CM 1976/C:5.

Reisenbichler, R.R. and Rubin, S.P. 1999. Genetic changes from artificial propagation of Pacific salmon affect the productivity and viability of supplemented populations. ICES Journal of Marine Science 56: 459–466.

Renaud, F.G. and Pasteur, C.N. 1986. Geographical divergence in *Bothriocephalus* (Cestoda) of fishes demonstrated by enzyme electrophoresis. International Journal for Parasitology 16: 553–558.

Robert, F. and Vianet, R. 1988. Age and growth of *Psetta maxima* and *Scophthalmus rhombus* in the Gulf of Lion. J. Appl. Icht; 4: 111–120.

Saegrov, H., Hindar, K., Kalas, S., and Lura, H. 1997. Escaped farmed Atlantic salmon replace the original salmon stock in the River Vosso, western Norway. ICES J. Mar. Sci. 54: 1166–1172.

Skaala, O., Jorstad, K.E., and Borgstrom, R. 1996. Genetic impact on two brown trout (*Salmo trutta*) populations after release of non-indigenous hatchery spawners. Can. J. Fish. Aquat. Sci. 54: 2027–2035.

Smoker, W. W., Wang, I. A., Gharrett, A. J. and Hard, J. J. 2004. Embryo survival and smolt to adult survival in second generation outbred coho salmon. Journal of Fish Biology 65(1): 254.

Støttrup, J.G., Lehman, K., Nielsen, R., Pedersen, C., Iglesias, J, Rodriguez, G., Howell, B., Baynes, S., Beard, T., Ellis, T., Gibson, R., Batty, R.S., Burrows, M.T., Langsdale, J., Krog, C., Modin, J. and Huges, R. 1998a. Evaluation of Stock Enhancement of marine flatfish. Final report AIR2-CT94-1732.

Støttrup, J.G. and Nielsen, B. 1998. How does cryptic behaviour of reared turbot *Scophthalmus maximus* compare with that of their wild counterparts? Subm. Aquaculture Europe, Bordeaux, France: 257–258.

Støttrup, J.G. and Paulsen, H. 1998b. Stocking of marine fish – a growing market for aquaculture. Farming Marine Fish beyond the Year 2000: Technological Solutions for Biological Challenges. ICES CM 1998/L:3.

Terrones, J. Vidal, S., Cal, R., Matinez, P. and Piferrer, F. 2004. Apoptosis during gametogenesis in diploid and triploid turbot (Scophthalmus maximus). Fish Physiology and Biochemistry 28: 369–370.

Tsukamoto, K., Masuda, R., Kuwada, H. and Uchida, K. 1997. Quality of fish for release: behavioural approach. Bulletin of the National Research Institute of Aquaculture Supplement 3: 93–99.

Vázquez, E., Fernandez, C., Martinez, I., Blanco, G., and Sànchez, J.A. 1996. Rapid flow cytometry method for triploidy determination in turbot (*Scopthalmus maximus* L.). ICES CM 1996/F:8.

Verspoor, E. 1988. Reduced genetic variability in first generation hatchery populations of Atlantic salmon (*Salmon salar*). Canadian Journal of Fisheries and Aquatic Sciences 45: 1686–1690.

Webb, J.H., Youngson, A.F., Thompson, C.E., Hay, D.W., Donaghy, M.J., and McLaren, I.S. 1993. Spawning of escaped farmed Atlantic salmon, *Salmo salar* L., in western and northern Scottish rivers: egg deposition by females. Aquacult. Fish. Manage. 24: 663–670.

Weber, W. 1979. On the turbot stock in the North Sea. ICES CM/1979/G:12. 21 pp.

Wetsteijn, B., 1976. Feeding of North Sea turbot and brill. ICES paper C.M. 1981/G:74.