EUROPEAN INLAND FISHERIES AND AQUACULTURE ADVISORY COMMISSION

ICES ADVISORY COMMITTEE ON FISHERIES MANAGEMENT

ISSN 2221-6650

ICES CM 2011/ACOM:18

EIFAAC OCCASIONAL PAPER NO. 48

REF. WGRECORDS, SSGEF, EIFAAC AND ACOM

Report of the 2011 Session of the Joint EIFAAC/ICES Working Group on Eels

Lisbon, Portugal, 5–9 September 2011

Report of the 2011 Session of the Joint EIFAAC/ICES Working Group on Eels

Lisbon, Portugal, 5–9 September 2011

European Inland Fisheries and Aquaculture Advisory Commission
Food and Agriculture Organization of the United Nations
Rome

International Council for the Exploration of the Sea
Copenhagen

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS

Rome, 2011
https://doi.org/10.17895/ices.pub.19280822

INTERNATIONAL COUNCIL FOR THE EXPLORATION OF THE SEA

COPENHAGEN, 2011

E-ISBN 978-92-5

Preparation of this document

This publication is the Report of the 2011 session of the Joint European Inland Fisheries and Aquaculture Advisory Commission (EIFAAC) and International Council for the Exploration of the Sea (ICES) Working Group on Eels, which was held in Lisbon, Portugal, from 5 to 9 September 2011.

The Working Group would like to acknowledge ICES for undertaking the editing and formatting of this publication.

Contact addresses:

European Inland Fisheries and Aquaculture Advisory Commission
Food and Agriculture Organization of the United Nations
Viale delle Terme di Caracalla
00153 Rome, Italy
Telephone (+39) 06 5705 4376
Telefax (+39) 06 5705 3360
www.fao.org publications-sales@fao.org

International Council for the Exploration of the Sea
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

Abstract

Indications are that the eel stock remains at an historical minimum, continues to decline and is outside safe biological limits. Recruitment of both glass eel and young yellow eel continues to decline and shows no sign of recovery. Current levels of anthropogenic mortality, thought to be high on juvenile (glass eel) and older eel (yellow and silver eel), are not sustainable and there is an urgent need to reduce these until there is clear evidence that the stock is increasing.

The effect of implemented management actions under the European Union (EU) Regulation initiated in 2009 have not yet led to any discernible changes in recruitment. It is likely that such changes will not be statistically detectable for some years, or even up to a decade or more. The loss of recruitment-series will weaken the power to detect any changes in the overall recruitment pattern or trend.

Fisheries on all life stages are found throughout the distribution area. Impacts vary from almost nil to heavy overexploitation. With the implementation of the management plans and the decline in the stock, a progressive restriction or collapse of local small-scale fisheries is foreseen. This change will come to the detriment of culture and heritage (e.g. fishing techniques, skills, gastronomy). There is also an increased risk of illegal fishing. Landings data continues to be unreliable and reporting under the Regulation and Data Collection Framework (DCF) is incomplete. Reported landings data in the Country Reports to WGEEL showed a great heterogeneity. Because landings data were incomplete, with some years missing for some of the countries, missing values were estimated and this shows that landings continue to decline.

Scientific reference points have not been previously set for eel. The EU Regulation sets a long-term escapement objective for the biomass of silver eel escaping from each management area at 40% of the pristine biomass (B_0) or B_{lim}. However, no explicit limit on anthropogenic impacts A_{lim} was specified, even though current biomass is (far) below B_0 and B_{lim}. The biomass reference point of B_{lim} = 40% of B_0 corresponds to a lifetime mortality limit of ΣA_{lim} = 0.92, unless strong density-dependence applies. As an initial option, it is recommended to set $B_{MSY\text{-}trigger}$ (value which should trigger a mortality reduction) at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ correspondingly. Allowing for natural variation in B_0 and for uncertainty in the estimates of status indicators and reference points, the resulting reference points (B_{lim}, $B_{MSY\text{-}trigger}$ and A_{lim}) should be considered as somewhat optimistic or unsafe. Noting the relationship between biomass stock reference points $B_{current}$, $B_{MSY\text{-}trigger}$ and mortality reference point ΣA_{lim}, the actual value for ΣA_{lim} below $B_{MSY\text{-}trigger}$ must be determined on a country (or Eel Management Unit) basis.

A framework is presented in the report for calculating lifetime anthropogenic mortality (ΣA) at the catchment level. As there are several different types of anthropogenic mortality (e.g. fishing, turbines and pumps, pollution, barriers and obstacles), this total mortality is expressed as the sum of those for all types, ΣA. Where direct estimates are not currently available, mortality information is collated in this report which may be used as alternatives in the interim.

With the need for statistical and scientific assessments of the reported management actions in 2012 and subsequent years and post-evaluation of the eel stock at the international level, WGEEL recommends that a (series of) planning workshop(s) be held to provide support and coordination for data collection, analysis and reporting. This should include updating the data reporting requirements for eel in the DCF, to include improved fisheries dependent sampling and fisheries independent surveys.

The European Eel Quality Database (EEQD) integrates data on contaminants, diseases and parasites, and fat content and this was updated with 2011 records. There is a need for data standardization across Europe and for the development of sensitivity thresholds in order to assess the impact of different contaminants/infections on the ability of the eel to migrate and breed successfully. The levels of some hazardous substances are so high is some cases that an effect on reproduction is likely to occur, but scientific evidence (dose/response studies) is still not available. Some direct fish kills due to pollution have been observed but the effect on the overall stock is not known. *Anguillicoloides crassus* continues to spread (e.g. in Scotland and Ireland) and is now quasi-omnipresent over Europe. Fisheries for eel (and other species) have been closed in the south of Belgium in 2006 and in many important rivers of France and the Netherlands in 2010 and 2011, because pollution levels are so high as to be a risk to the health of consumers. The eels protected by these measures are in general of lower quality and hence their contribution to the restoration of the stock might be limited.

Declared glass eel total catch from fisheries in 2011 was approximately equal to the current requirements for stocking listed in eel management plans submitted under the EU Regulation. The best estimate is that glass eel fisheries in 2011 distributed 12% of their total catch to restocking, 30% to Aquaculture and the fate of the remainder is unknown. There are insufficient traceability systems in place to improve this analysis making it difficult to provide accurate advice on the fate of glass eel.

Giving priority to the recovery of the European stock, the objective of any stocking exercise should be to maximize net benefit to the stock as a whole until clear signs of recovery. However, stocking with an element of fishery support, combined with maintaining some spawner escapement, is not excluded in the EU Regulation. Given the current assessment of the overall stock, stocking, where it occurs, should be in conjunction with reductions in fisheries (yellow and silver) mortality and other direct mortalities (e.g. turbine, pumping stations) affecting the stocked eels. Stocking should not be seen as a substitute for reducing mortality, but as an additional measure.

There is an obligation that up to 60% of the catch of eel less than 12 cm is used for stocking. WGEEL makes the management recommendation that this 60% should be stocked in areas where anthropogenic mortality is minimal and environmental quality is high. Those wishing to stock to support fisheries, or to mitigate against other anthropogenic mortalities, should draw on stock from the remaining 40% allocated by management to other uses. The burden of proof that stocking will generate net benefit in terms of spawner escapement rests with those taking the stocking action. Prior to, or for continuing existing, stocking, a risk assessment and net benefit analysis should be conducted.

The MSFD requires the development of a marine strategy, specific to its own waters, that reflects the overall perspective of the marine region or subregion concerned. The definition of criteria and the selection of indicators to assess GES is an important step in this directive. As the eel is a diadromous species with a wide distribution and unusual life cycle some caution has to be taken when selecting the life stage (glass eel, yellow or silver eel) to assess the environmental status of marine regions. The descriptors selected relevant to eel by the WG included: **D1**-*Biological Diversity is maintained*; **D3**-*Populations of all commercially exploited fish and shellfish are within safe biological limits*; **D9**-*Contaminants in fish and other seafood for human consumption do not exceed levels established*; and **D11**- *Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*.

FAO European Inland Fisheries and Aquaculture Advisory Commission; International Council for the Exploration of the Sea.

Report of the 2011 session of the Joint EIFAAC/ICES Working Group on Eels. Lisbon, Portugal, from 5 to 9 September 2011. EIFAAC Occasional Paper. No. 48. ICES CM 2011/ACOM:18. Rome, FAO/Copenhagen, ICES. 2011. 246p.

Executive summary

This report summarizes the presentations, discussions and recommendations of the 2011 session of the Joint EIFAAC/ICES Working Group on Eel which took place in Lisbon, Portugal, hosted by the University of Lisbon (Faculdade de Ciências, Universidade de Lisboa), from 5 to 9 September 2011.

In this section, the main outcomes from the report are summarized, a forward focus is proposed in the light of observed declines in many Anguillid stocks and the implementation of the EU Regulation for the Recovery of the Eel Stock and the main recommendations are presented by WGEEL.

The Working Group in 2011, along with Study Group on International Post-Evaluation on Eels (SGIPEE), has focused on six main themes, updating the recruitment and stocking time-series, including a power analysis on the ability to detect a change in recruitment or silver eel output, a discussion on biological reference points and setting reference biomass and mortality limits, providing support for local population assessments especially in describing anthropogenic mortalities, updating the EEQD and making preliminary assessments of spawner quality and making some observation on eel and the Marine Strategy Framework Directive.

The objective of eel stock assessment is to quantify the biomass of silver eel escaping in order to assess compliance with the EU target of 40% of pristine biomass without anthropogenic mortality. Given that it will be impractical to directly assess silver eel biomass and mortality in many rivers, yellow eel stock assessment will also be required. In conjunction with SGIPEE and pilot projects to estimate potential and actual escapement of silver eel (POSE), the Working Group has identified a number of areas where gaps in knowledge existed (i.e. silver eel assessment, yellow to silver transformation, quantification of habitat) and focused on these in order to provide support for local stock assessments. To fill the gap between implemented reductions in mortality and the subsequent changes in silver eel biomass, it is also recommended that a measure of lifetime anthropogenic mortality be determined.

Summary of this report

From the information available to the Working Group in 2011, indications are that the eel stock remains at an historical minimum, continues to decline and is outside safe biological limits. Recruitment continues to decline and shows no sign of recovery. Current levels of anthropogenic mortality, thought to be high on juvenile (glass eel) and older eel (yellow and silver eel), are not sustainable and there is an urgent need to reduce these until there is clear evidence that the stock is increasing. Recruitment in the last five years has been particularly low with an index average of less than 1% in the continental North Sea and less than 5% elsewhere in Europe compared to the mean for 1960–1979 levels. Young yellow eel recruitment series also remain low at around 10% of their mean for the 1960–1979 levels.

The effect of implemented management actions under the EU Regulation initiated in 2009 have not yet led to any discernible changes in recruitment. Power analyses have shown that changes in recruitment (WGEEL) or silver eel escapement (SGIPEE) will not be statistically detectable for some years, or even up to a decade or more. This emphasizes the need for information on anthropogenic mortality rates in the short term in support of focusing on the status of the stock. The loss of recruitment series will weaken the power to detect any changes in the overall recruitment pattern or trend.

Landings data continues to be unreliable and reporting under the Regulation and DCF is incomplete. Reported landings data in the Country Reports to WGEEL showed a great heterogeneity. Some countries operate an official system; some report total landings while others report landings by Management Unit or Region, and some countries don't have a centralized system. Furthermore, some countries have revised their dataseries, with extrapolations to the whole time-series, for the necessities of the Eel Management Plan compilation. Because landings data were incomplete, with some years missing for some of the countries, an estimate of the missing values was provided by glm extrapolation, with year and countries as the explanatory factors. Landings continue to decline.

Fisheries on all life stages are found throughout the distribution area. Impacts vary from almost nil to heavy overexploitation. The EU Regulation delegates the processes of assessing and managing the fisheries to the Member States, and quantification of fishing mortalities is foreseen by 2012. With the implementation of the management plans and the decline in the stock, a progressive restriction or collapse of local small-scale fisheries is foreseen. This change will come to the detriment of culture and heritage (e.g. fishing techniques, skills, gastronomy). In some places, there is also an increased risk of illegal fishing.

Eel was first included in the DCR in 2004 and the EC held a workshop in Sweden in 2005 with the objective of specifying minimum requirements on sampling levels for fishery-dependent and fishery-independent data, for the three exploited life stages (glass eel, yellow eel and silver eel), in both inland and coastal waters. At the time of the 2005 workshop, the mid- and long-term targets of the EC action plan on eel were not yet defined. In the absence of clear targets, it was not possible to implement any monitoring programmes in the DCR. The EC acknowledged that this would modification in line with the explicit management goals when defined. Given the reporting requirements for Member States in 2012 and subsequent years and the need for statistical and scientific assessments of the reported management actions and post-evaluation of the eel stock at the international level, the WGEEL recommends that a

(series of) data workshop(s) be held to provide support and coordination for data collection, analysis and reporting. This would include updating the data reporting requirements for eel in the DCF, to include improved fisheries dependent sampling and, among others, fisheries-independent surveys.

No scientific reference points have been previously set for eel. The EU Regulation sets a long-term escapement objective for the biomass of silver eel escaping from each management area at 40% of the pristine biomass (B_0) or B_{lim}. However, no explicit limit on anthropogenic impacts A_{lim} is specified, even though current biomass is (far) below B_0 and B_{lim}. For long-lived species (such as the eel) with a low fecundity (unlike the eel), biological reference points are often formulated in terms of numbers, rather than biomass. Though numbers-based and biomass-based reference points will differ slightly, a mortality-based reference point that results in 40% of the pristine stock *numbers* is proposed. The biomass reference point of B_{lim} = 40% of B_0 corresponds to a lifetime mortality limit of $\Sigma A_{lim} = 0.92$, unless strong density-dependence applies. In the latter case, a more complex assessment will be required, and a limit of $\%SPR_{lim}$ = 40% can be applied. As an initial option, it is recommended to set $B_{MSY\text{-}trigger}$ (value which should trigger a mortality reduction) at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ correspondingly. Allowing for natural variation in B_0 and for uncertainty in the estimates of status indicators and reference points, the resulting reference points (B_{lim}, $B_{MSY\text{-}trigger}$ and A_{lim}) should be considered as somewhat optimistic or unsafe. Noting the relationship between biomass stock reference points $B_{current}$, $B_{MSY\text{-}trigger}$ and mortality reference point ΣA_{lim}, the actual value for ΣA_{lim} below $B_{MSY\text{-}trigger}$ must be determined on a country (or Eel Management Unit) basis.

A framework is presented in the report for calculating lifetime anthropogenic mortality (ΣA) at the catchment level. As there are several different types of potential anthropogenic mortality (e.g. fishing, turbines and pumps, pollution, barriers and obstacles), this total mortality is expressed as the sum of those for all types, ΣA. Where direct estimates are not currently available, mortality information is collated in this report for each of these impacts which may be used as alternatives in the interim.

The European Eel Quality Database (EEQD) integrates data on contaminants, diseases and parasites, and fat content. New data were incorporated in 2011 for 471 records of contaminants, diseases or parasites, but the data do not yet support a comprehensive overview on the quality of eel throughout its distribution. There is a need for data standardization across Europe and for the development of sensitivity thresholds in order to assess the impact of different contaminants/infections on the ability of the eel to migrate and breed successfully. The levels of particular hazardous substances are so high is some cases that an effect on reproduction is likely to occur, but scientific evidence (dose/response studies) is still not available. Some direct fish kills due to pollution have been observed but the effect on the overall stock is not known. *Anguillicoloides crassus* continues to spread (e.g. in Scotland and Ireland) and is now quasi-omnipresent over Europe.

Fisheries for eel (and other species) have been closed in the south of Belgium in 2006 and in many important rivers of France and the Netherlands in 2010 and 2011, because pollution levels are so high as to be a risk to the health of consumers. The eels protected by these measures are in general of lower quality, and hence, their contribution to the restoration of the stock might be limited.

Stocking with glass eel has decreased strongly since the early 1990s and is now to be at a relatively low level and still decreasing. However, this has partly been compensated for by an increasing number of young yellow eels stocked since the late 1980s, the amount of which has varied widely in recent years.

Declared glass eel total catch from fisheries in 2011 was approximately equal to the current requirements for stocking listed in eel management plans submitted under the EU regulation. The best estimate of WGEEL is that glass eel fisheries in 2011 distributed 12% of their total catch to restocking, 30% to Aquaculture and the fate of the remainder is unknown. There are insufficient traceability systems in place to improve upon this analysis. This poor data reporting makes it difficult to provide accurate advice on the fate of glass eel and the proportions and mortalities of glass eel set aside and used for stocking.

Giving priority to the recovery of the European stock, the objective of any stocking exercise should be to maximize net benefit to the stock as a whole until clear signs of recovery. However, stocking with an element of fishery support, combined with maintaining some spawner escapement, is not excluded in the EU Regulation. Given the current assessment of the overall stock, stocking, where it occurs, should be in conjunction with reductions in fisheries (yellow and silver) mortality and other direct mortalities (e.g. turbine, pumping stations) affecting the stocked eels. Stocking should not be seen as a substitute for reducing mortality, but as an additional measure.

The Regulation contains an obligation that up to 60% of the catch of eel less than 12 cm is used for stocking. WGEEL 2011 makes the management recommendation that this 60% should be stocked in areas where anthropogenic mortality is minimal and environmental quality is high. Those wishing to stock to support fisheries or to mitigate against other anthropogenic mortalities should draw on stock from the remaining 40% allocated by management to other uses.

The burden of proof that stocking will generate net benefit in terms of spawner escapement rests with those taking the stocking action. Prior to, or for continuing existing, stocking, a risk assessment should be conducted, taking into account fishing, holding, transport and post-stocking mortalities and other factors such as disease and parasite transfers. WGEEL 2011 offer the TRANSLOCEEL model as a framework for assisting with such risk assessments. The best available parameters or estimates of mortality in source and supplied areas should be used as inputs to the model.

Preliminary results of the latest tracking studies show, that as far as we can track any eel, coastal and oceanic migration routes and behaviour patterns of eel of stocked origin are indistinguishable from those derived from naturally immigrated recruits.

The MSFD requires that each Member State develops a marine strategy, specific to its own waters that reflects the overall perspective of the marine region or subregion concerned. The definition of criteria and the selection of indicators to assess GES is an important step in this directive. The eel is classified as threatened by OSPAR and HELCOM conventions. As the eel is a diadromous species with a wide distribution and unusual life cycle some caution has to be taken when selecting the life stage (glass eel, yellow or silver eel) to assess the environmental status of marine regions. The descriptors selected relevant to eel by the WG included: **D1**-*Biological Diversity is maintained*; **D3**-*Populations of all commercially exploited fish and shellfish are within safe biological limits*; **D9**-*Contaminants in fish and other seafood for human consumption do not exceed levels established*; and **D11**-*Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*. However, when trying to set indicators for each descriptor selected it was decided to exclude D11 due to difficulties associated with assessing the impact of renewable energy on the eel. Despite the existence of scientific evidence showing that eels can use electromagnetism for orientation when migrating to reproduce it is currently not possible to quantify their impact on marine species.

Forward focus

This report is a further step in an ongoing process of documenting stock and fisheries of the eel (*Anguilla anguilla*) and developing methodology for giving scientific advice on management to effect a recovery in the European eel stock. In 2007, a European plan for recovery of the stock was adopted by the EU Council of Ministers (Council Regulation No. 110/2007). In accordance with this plan, Member States developed Eel Management Plans (2008) for the stock on their territory, aiming at a silver eel (spawner) escapement of 40% in biomass terms, relative to the pristine state (no anthropogenic mortality, historical high recruitment). By July 2012, Member States will be required to report on the actions taken, the reduction in anthropogenic mortalities achieved, and the state of their stock relative to its targets. By the end of 2013, the EU Commission will present a report to the European Parliament and the Council with a statistical and scientific evaluation of the outcome of the implementation of the Eel Management Plans.

In recent years, ICES has advised on the (alarming) state of the stock; has provided technical consultations of the national management plans; and has developed a methodology for international post-evaluation based on national status reports. The WGEEL considers that the scientific evaluation foreseen by 2013 will require scientific advice on the reduction in anthropogenic mortalities achieved throughout Europe and on the state of the whole European stock relative to its overall targets. In the light of the cascaded approach of the Eel Regulation, it is noted that stock-wide evaluation critically depends on the exchange of data collected at the national level (within the framework of the Data Collection Framework, the Water Framework Directive and others), for which no coordinated structure exists yet for eel. Additionally, routine assessments for eel stocks have rarely been made, and standard ICES methodology has proven to be less than suitable in many cases. Consequently, there is an urgent need for planning (data exchange and methodologies), and for tuning expectations and opportunities. This international planning process can commence independently of the national post-evaluation and reporting by July 2012.

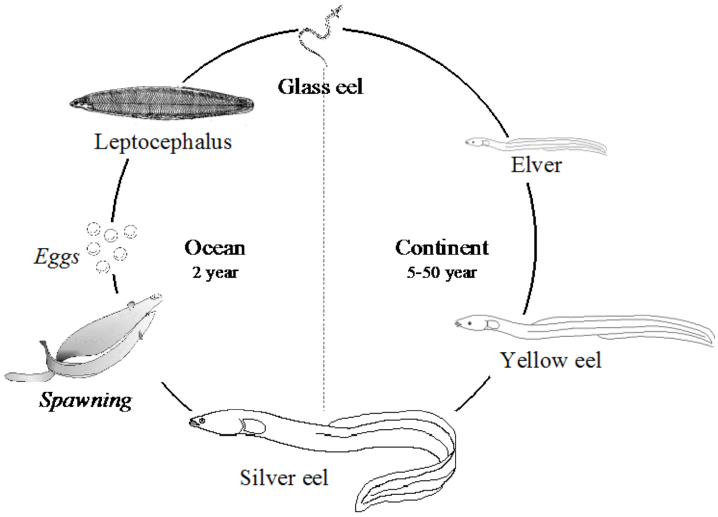
Scientific advice in 2012 and 2013 will critically depend on this planning process. The WGEEL expresses its concern that, without an improved database and timely consideration of assessment procedures, the scientific advice will be restricted to recruitment trend analysis. In that case, the effect of major changes in anthropogenic impact will only be significantly demonstrable at a time-scale of decades.

Main recommendations

- Because overall recruitment remains at an all time low since records began, the stock continues to decline and stock recovery will be a long-term process for biological reasons, all negative anthropogenic factors impacting on the stock and affecting the production/escapement of silver eels should be reduced until there is clear evidence the stock is increasing;
- Given the current low abundance, glass eel landings are unlikely to support the total commitments for stocking given in the Eel Management Plans. This situation is foreseen to continue as a consequence of the continued decline of recruitment, the implementation of conservation measures reducing the glass eel catch and competing demands from aquaculture. The use of stocking within the management actions of the EU Regulation 1100/2007 should not be used as a reason for not taking actions to reduce anthropogenic mortality;
- The urgent need to plan and coordinate the data collection and tool development for the 2012 post-evaluation is re-iterated;
- With the need for statistical and scientific assessments of the reported management actions in 2012 and subsequent years and post-evaluation of the eel stock at the international level, WGEEL recommends that a (series of) planning workshop(s) be held to provide support and coordination for data collection, analysis and reporting. This should include updating the data reporting requirements for eel in the DCF, to include improved fisheries dependent sampling and fisheries independent surveys.

Glossary

Eels are quite unlike other fish. Consequently, eel fisheries and eel biology come with a specialized jargon. This section provides a quick introduction for outside readers. It is by no means intended to be exhaustive.



The life cycle of the European eel. The names of the major life stages are indicated. Spawning and eggs have never been observed in the wild.

Glossary of terms

Glass eel	Young, unpigmented eel, recruiting from the sea into continental waters
Elver	Young eel, in its first year following recruitment from the ocean. The elver stage is sometimes considered to exclude the glass eel stage, but not by everyone. Thus, it is a confusing term.
Bootlace, fingerling	Intermediate sized eels, approx. 10–25 cm in length. These terms are most often used in relation to stocking. The exact size of the eels may vary considerably. Thus, it is a confusing term.
Yellow eel (Brown eel)	Life stage resident in continental waters. Often defined as a sedentary phase, but migration within and between rivers, and to and from coastal waters occurs. This phase encompasses the elver and bootlace stages.
Silver eel	Migratory phase following the yellow eel phase. Eel characterized by darkened back, silvery belly with a clearly contrasting black lateral line, enlarged eyes. Downstream migration towards the sea, and subsequently westwards. This phase mainly occurs in the second half of calendar years, though some are observed throughout winter and following spring.
Eel River Basin or Eel Management Unit	"Member States shall identify and define the individual river basins lying within their national territory that constitute natural habitats for the European eel (eel river basins) which may include maritime waters. If appropriate justification is provided, a Member State may designate the whole of its national territory or an existing regional administrative unit as one eel river basin. In defining eel river basins, Member States shall have the maximum possible regard for the administrative arrangements referred to in Article 3 of Directive 2000/60/EC [i.e. River Basin Districts of the Water Framework Directive]." EC No. 1100/2007

River Basin District	The area of land and sea, made up of one or more neighbouring river basins together with their associated surface and groundwaters, transitional and coastal waters, which is identified under Article 3(1) of the Water Framework Directive as the main unit for management of river basins. Term used in relation to the EU Water Framework Directive.
Stocking	Stocking is the practice of adding fish [eels] to a waterbody from another source, to supplement existing populations or to create a population where none exists.
Trap & transport	Traditionally, the term trap and transport referred to trapping recruits at impassible obstacles and transporting them upstream and releasing them. Under the EMPs, trap and transport (or catch and carry) now also refers to fishing for downstream migrating silver eel for transportation around hydropower turbines.

Eel reference points/population dynamic	
Anthropogenic mortality after management (A_{post})	Estimate of anthropogenic mortality after management actions are implemented
Anthropogenic mortality before management (A_{pre})	Estimate of anthropogenic mortality before management actions are implemented
Best achievable biomass (B_{best})	Spawning biomass corresponding to recent natural recruitment that would have survived if there was only natural mortality and no restocking; that is
Interim Target for biomass ($B_{interim}$)	Pragmatic intermediate goals for spawner escapement biomass; set by managers.
Interim Target for mortality ($A_{interim}$)	Pragmatic intermediate anthropogenic mortality goal; set by managers.
Limit anthropogenic mortality (A_{lim})	Anthropogenic mortality, above which the capacity of self-renewal of the stock is considered to be endangered and conservation measures are requested. (Cadima, 2003)
Limit spawner escapement biomass (B_{lim})	Spawner escapement biomass, below which the capacity of self-renewal of the stock is considered to be endangered and conservation measures are requested. (Cadima, 2003)
Precautionary anthropogenic mortality (A_{pa})	Anthropogenic mortality, above which the capacity of self-renewal of the stock is considered to be endangered, taking into consideration the uncertainty in the estimate of the current stock status.
Precautionary spawner escapement biomass (B_{pa})	The spawner escapement biomass, below which the capacity of self-renewal of the stock is considered to be endangered, taking into consideration the uncertainty in the estimate of the current stock status.
$B_{msy\text{-}trigger}$	Value of spawning–stock biomass (SSB) which triggers a specific management action, in particular: triggering a lower limit for mortality to achieve recovery of the stock.
Pristine biomass (B_0)	Spawner escapement biomass in absence of any anthropogenic impacts..
Spawner escapement biomass after management (B_{post})	Estimate of spawner escapement biomass after management actions are implemented
Spawner escapement biomass before management (B_{pre})	Estimate of spawner escapement biomass before management actions are implemented

Spawner per recruitment (SPR)	Estimate of spawner production per recruiting individual.
%SPR	Ratio of SPR as currently observed to SPR of the pristine stock, expressed in percentage. %SPR is also known as Spawner Potential Ratio.
Anthropogenic mortality after management after (A_{post})	Estimate of anthropogenic mortality after management actions are implemented
Anthropogenic mortality before management (A_{pre})	Estimate of anthropogenic mortality before management actions are implemented
Best achievable biomass (B_{best})	Spawning biomass corresponding to recent natural recruitment that would have survived if there was only natural mortality and no restocking;
Interim Target for biomass ($B_{interim}$)	Pragmatic intermediate goals for spawner escapement biomass; set by managers.
Interim Target for mortality ($A_{interim}$)	Pragmatic intermediate anthropogenic mortality goal; set by managers.

Contents

1 Introduction

1.1 The 2010 WGEEL

At the 98th Statutory Meeting of ICES (2010) and the 26th meeting of EIFAC (2010) it was decided that:

2010/2/ACOM18 The **Joint EIFAAC/ICES Working Group on Eels** (WGEEL), chaired by Russell Poole*, Ireland and Cedric Briand, France, will meet in Lisbon, Portugal, 5–9 September 2011, to:

a) Assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans; examine criteria for defining a recovery;

b) Develop and test methods to post-evaluate effects of management actions at the stock-wide level (in conjunction with SGIPEE), including quality assurance checking of Eel Management Unit biomass estimates;

c) Develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures; test data scenarios at the local level;

d) Provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods;

e) Review and develop approaches to quantifying the effects of eel quality on stock dynamics and integrating these into stock assessments; develop references points for evaluating impacts on eel;

f) Respond to specific requests in support of the eel stock recovery Regulation, as necessary; and

g) Report on improvements to the scientific basis for advice on the management of European and American eel;

h) Identify elements of the EGs work that may help determine status for the eleven Descriptors set out in the Commission Decision (available at http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2010:232:0014:0024:EN:PDF;

i) Provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status;

j) Take note of and comment on the Report of the Workshop on the Science for area-based management: Coastal and Marine Spatial Planning in Practice (WKCMSP) http://www.ices.dk/reports/SSGHIE/2011/WKCMSP11.pdf;

k) Provide information that could be used in setting pressure indicators that would complement biodiversity indicators currently being developed by the Strategic Initiative on Biodiversity Advice and Science (SIBAS). Particular consideration should be given to assessing the impacts of very large renewable energy plans with a view to identifying/predicting potentially catastrophic outcomes;

l) Identify spatially resolved data, for e.g. spawning grounds, fishery activity, habitats, etc.

Additions to the ToR:

1) Review the glass eel catches for the past two years,

assess quantities caught in the commercial fishery

exported to Asia

used in stocking

used in aquaculture for consumption

consumed direct

mortalities

compare with the EMP commitments to stocking

2) Review the latest information on stocking, including previous reviews by WGEEL, new scientific reports/publications on the uses for stocking, the success of stocking as a method of fisheries support and/or as a means of increasing silver eel escapement and spawner biomass.

WGEEL will report by 21 September 2011 for the attention of WGRECORDS, SGEF and ACOM and EIFAAC.

45 people attended the meeting, from seventeen countries (see Annex 1).

The current Terms of Reference and Report constitute a further step in an ongoing process of documenting the status of the European eel stock and fisheries and compiling management advice. As such, the current Report does not present a comprehensive overview, but should be read in conjunction with previous WGEEL reports (ICES, 2000; 2002; 2003; 2004, 2005a, 2006, 2007, 2008, 2009 and 2010a) and with the SGIPEE reports (ICES 2010b and 2011).

In addition to documenting the status of the stock and fisheries and compiling management advice, in previous years the Working Group also provided scientific advice in support of the establishment of a recovery plan for the stock of European Eel by the EU. In 2007, the EU published the Regulation establishing measures for the recovery of the eel stock (EC 1100/2007). This introduced new challenges for the Working Group, requiring development of new methodologies for local and regional stock assessments and evaluation of the status of the stock at the international level. Implementation of the Eel Management Plans will likely introduce discontinuities to data trends and will require a shift from fisheries-based to scientific survey-based assessments. This challenging situation continues in 2011 with the introduction of an trade ban into and out of Europe under CITES and the need to set biological reference points for the eel stock in order to support post-evaluation of eel management actions.

The structure of this report does not strictly follow the order of the Terms of Reference for the meeting. The meeting, and consequently the report, was organized in five subgroups using the Agenda in Annex 2. The subgroups, under the headings of "Data and trends", "International Stock Assessment, reference points and post-evaluation", "Local Assessments and mortality", "Assessment of eel quality”, Glass eel resources and stocking" and "Marine Strategy Framework Directive" and “the eel fishery resource” addressed the Terms of Reference as follows:

Chapter 2 presents trends in recruitment, stock, fisheries and aquaculture (ToR a). Chapter 2 presents a power analysis of recruitment time-series to test the ability to detect a change in recruitment trends over time (ToR a, d). This chapter also provides suggestions for the improvement of the DCF with respect to eel and comment is made on the socio-economics of eel fisheries.

Chapter 3 continues the line of development on the concept of post-evaluation and stock assessment at the international level. Chapter 3 presents a reassessment of the precautionary diagram using standard ICES practices and applies it to the limits set in the EU Regulation.

Chapter 3 also provides guidelines on the data requirements in the reporting of the Eel Management Plans in 2012 (ToR a, b and f).

Chapter 4 provides support for locally based stock assessment and post-evaluation of the impact of local management actions on anthropogenic mortalities. Gaps in methodologies and data/information are addressed. (ToR a, c and d).

Chapter 5 updates the European Eel Quality Database (EEQD) and discusses the importance of the inclusion of spawner quality parameters in stock management advice (ToR d and e).

Chapter 6 examines the trade and movement of glass eel within Europe, provides an update on the most recent scientific knowledge in relation to stocked eel and develops and presents a model on evaluating the net benefit of stocking (ToR a, c, g and additions 1 and 2 to the ToR).

Chapter 7 provides information in support of the Marine Strategy Framework Directive with respect to eel (ToR h, i, j, k, l).

Terms of Reference a. (revision of catch statistics) is the follow-up of the analysis made in the Report of the 2004 meeting of the Working Group (ICES 2005, specifically Annex 2). Following that meeting, a Workshop was held under the umbrella of the European Data Collection Regulation (DCR), in September 2005, in Sånga Säby (Stockholm, Sweden). The Workshop report presented catch statistics in greater detail than had been handled by this Working Group before. It is envisaged that additional and improved data will become available under the Eel Regulation and when the Data Collection Framework is fully implemented across Europe. An initial review of the data available was undertaken in 2009 and recommendations for data reporting to the EU in 2012 are made by ICES (2010 SGIPEE and 2011) and these are further developed in Chapter 2.9 with a proposal for a workshop on improving the inclusion of eel data in the DCF, especially for fisheries independent data.

1.2 Workshop on Age Reading of European and American Eel (WKAREA 2)

The **Workshop on Age Reading of European and American Eel [WKAREA-2]** (Chair: Françoise Daverat, France) exchanged information by correspondence in 2010 and met in Bordeaux, France in March 2011:

a) to exchange samples (>100 per species) of European and American eel otolith pictures, including known age eels, with samples prepared using different protocols and representing a range of eel subpopulations, and environment types encountered in both species range;

b) to apply the age estimation criteria defined during the previous meeting in an inter-calibration process involving the exchanged images and a significant number of readers (>20);

c) to analyse readings and interpret the results of the inter-calibration of European and American eel age reading;

d) to make recommendations and feedback on the age estimation criteria to increase age estimation precision and accuracy and improve the inter reader agreement;

e) to incorporate the findings with the report and manual developed by WKAREA 2009 for formal publication; and

f) to address the generic ToRs adopted for workshops on age calibration (see 'PGCCDBS Guidelines for Workshops on Age Calibration').

The workshop commenced with the analysis of the results of the experienced reader inter calibration exercise that had been carried out several months previous the meeting. This intercalibration exercise was based on image exchange for both species. The readings had been performed on a web platform device allowing the positioning of age checks on the pictures and recording the number of checks identified by each reader. A total of 21 readers participated to the exchange. A collection of 117 European eel pictures and 44 American eel otolith pictures were used for the exchange. The overall agreement rate of the readings with the modal age ranged from 66.2% to 13.2%. The results showed that more agreement would have been obtained if the reading rules had been applied more consistently. Some readers discarded some "difficult" otoliths. The absence of metadata such as the location, date of capture and habitat type of the otolith was also identified as a source of misinterpretation of growth patterns. It was recognized for future readings that metadata should be included and that all otoliths would be read, with the addition of a reading confidence parameter. A reference collection composed of 38 *A. Anguilla* and 19 *A. rostrata* known age otolith pictures was set up, with one blind file and one fully annotated file. The manual was updated with more details included for the different preparation protocols. A protocol for age reading and training age reading and routine age reading was proposed, including the use of the reference collection.

The WKAREA2 recommended:

Recommendation	For follow up by:
1.Set up new validation projects to obtain known age eels from different locations using direct (mark recapture of marked otoliths)	ICES
2. Provide indirect estimation of age with direct estimation of growth rate (mark recapture of fish)	ICES
3. Investigate alternative methods of age estimation such as otolith chemistry (lead radium decay)	ICES
4. Validate new methods of otolith preparation	Other members
5. the validation projects should be funded at international level	ICES
6. DCF should support/manage the reference collection	WGDIM (group on Data and Information management)

The Working Group endorsed the recommendations of WKAREA2 in the wider context of age and growth being one element of the data required for reporting and post-evaluating the EU Regulation. It is important that eel age determination is viewed as one, and not the only aspect, of the DCF for eel.

1.3 Study Group on International Post-Evaluation on Eels (SGIPEE)

2009/2/SSGEF20: The **Study Group on International Post evaluation on Eels** (SGIPEE), chaired by Laurent Beaulaton, France, met in London, UK, 24–27 May 2011 to:

a) Review stock assessment and post-evaluation methods available for species of eels, and those used by ICES Expert Groups on other species, that could be successfully applied to eels at the stock-wide level in 2012;

b) Adapt methods for stock-wide post-evaluation of *Anguilla anguilla* and apply them to data collated by WGEEL at its annual meetings; (this may include aggregation of EMU post-evaluation);

c) Analyse sensitivity of the selected methods to stock improvement or deterioration using simulated data;

d) Submit recommendations to WGEEL on: the best available post-evaluation method for 2012; gaps in data or knowledge that need to be filled before 2012; and methods that should be developed and data that should be collected after 2012 for the next stock-wide evaluation.

This Report summarizes the presentations, discussions and recommendations of the 2011 session of ICES Study Group on International Post evaluation on Eels which took place in London, UK, hosted by the Environment Agency, from 24th to 27th of May 2011. This study group was chaired by Laurent Beaulaton (France) and involved 13 people from eight countries.

SGIPEE was intended to design, test, analyse and report on a method of scientific ex post evaluation at the stock-wide level of applied management measure for eel restoration. After a first meeting in 2010 mainly focused on designing the appropriate framework and the methods for eel ex post evaluation and reviewing available data, the 2011 meeting tested the reliability of this framework.

The scientific basis and the applicability of the modified ICES precautionary diagram have been improved. The possibilities of data deficiencies and inconsistencies have been explored and a first draft of a quality control sheet has been designed. Additionally a power analysis has been conducted to see the ability to detect any change in stock status indicator (recruitment and silver eel biomass). It shows that, given the high natural variability of biological processes, the probability to detect any change, even in case of strong management measures, is very low in 2012 but increase with time. As a consequence, in the short term, the most important parameter to post-evaluate the result of implemented eel management measures is anthropogenic mortality ($\sum$A) because most effects on biomass will only show up after several years.

The Study Group on International Post Evaluation on Eels (24–27 May 2011 in London) recommends that:

1) Because short-term post-evaluation of eel management is primarily focused on (achieved and intended) mortality levels (rather than biomass-levels), SGIPEE recommends that WGEEL considers the relation between

biomass reference point and mortality reference point, taking into account the objective of the EU Eel Regulation and previous ICES advice.

2) Because short-term post-evaluation is primarily focused on mortality levels and long-term post-evaluation on future recruitment trends, SGIPEE recommends that the power-analyses (on simulated silver eel escapements in this report) are extended to cover mortality estimates and recruitment trends.

3) the spatial coverage of the international stock assessment done by the Joint EIFAAC/ICES Working Group on Eels is improved through the participation of countries throughout the distribution area, particularly through integration of ICES, EIFAAC and GFCM eel assessment and advice.

4) assessments of anthropogenic impacts and the dynamics of the stock (current, past and future) are improved.

The following two recommendations are copied from WGEEL 2010 (ICES, 2010b) and endorsed by the study group:

1) The 2001 meeting of WGEEL (ICES 2002b) recommended the formation of an international commission that could act as a clearing house for handling and coordinating data collection and storage, stock assessment, management and research. Noting the urgent need to plan and coordinate the data collection and tool development for the 2012 post-evaluation; this recommendation is reiterated.

2) In particular, it is recommended to organize a (series of) workshops in relation to local eel stock monitoring, with a focus on standardization and coordination, preparing for the 2012 post-evaluation, setting the scene for the 2013 international stock assessment. The study group also underline that wetted area data are of utmost importance and should be collected and made publically available in priority.

NOTE: these recommendations are picked up in Chapters 3 and 4.2 and also in the Forward Focus.

1.4 Workshop-BaltEel

At the 98th Statutory Meeting of ICES (2010) it was decided that:

A **Workshop on BALTic EEL** [WKBALTEEL] (Chair: Willem Dekker, Sweden) met in Stockholm, Sweden 2–4 November 2010 to:

a) assess the status of the eel stock in the Baltic, to identify available data, to summarize the documentation available in national management plans;

b) prepare the work of SGIPEE as regards the Baltic by assessing the status of the eel stock in the Baltic region as a whole, following the assessment framework developed by WGEEL/SGIPEE, and to make the required data available to WGEEL/SGIPEE;

c) assess the anthropogenic impacts on the stock in the Baltic, focusing on international interactions between countries/rivers, and to relate that to the targets/limits of the (national) Eel Management Plans and the (international) EU recovery plan;

d) consider data requirements for the assessment of the international interactions, and to identify data and knowledge gaps.

WKBALTEEL reported for the attention of WGEEL and PGCCDBS and ACOM.

20 people attended the meeting, from nine countries. Unfortunately, Russia was not represented, but otherwise all countries around the Baltic participated in this Workshop. In the preparatory process for this Workshop, contacts were made and information exchanged with the Kaliningrad State Technical University, Russia.

The objective of this Workshop has been to document and present the information on the eel stock in the Baltic currently available, to standardize stock status assessments (cf. SGIPEE), to initiate a common assessment for the whole Baltic stock, to identify and quantify interactions between management measures taken in different countries, and to suggest future improvements by means of further standardization, cooperation and integration of monitoring and assessment efforts; and identify future data requirements and current knowledge gaps.

The impact of coastal fisheries in the countries around the outlet of the Baltic has been quantified using information from tag-recapture studies; these studies have addressed the national fisheries only. Though impacts on the escaping silver eels from other countries have been documented in long-running tagging programmes, these impacts so far have remained unquantified. To quantify the impact of the outlet-countries on the total Baltic stock, international tagging experiments are required, in which eels are tagged on the east-side and recaptured in the west. Such an experiment cannot be organized by individual countries, neither east nor west. A joint initiative for a pan-Baltic tagging programme is required (high information tracking studies; mass-marking methods for quantification).

The scientific documentation of the stock status and advice on potential management actions will benefit from further integration and coordination in monitoring and research. To this end, field programmes can be (further) integrated, expertise be shared, a central database designed (or a standardized data exchange procedure developed), and a joint assessment of stock status developed. Because the interactions between countries in the Baltic are essentially regional in character, a regional monitoring and assessment procedure will relieve the truly international assessment addressed by SGIPEE and WGEEL.

The first post-evaluation of the Eel Regulation is foreseen in 2012. Individual countries will report on the status of their stock and fisheries, other anthropogenic impacts and protective measures. Standardization (of the data and/or the reporting) will greatly facilitate the international post-evaluation process. As a pragmatic interim goal for further integration of eel stock management in the Baltic, a full integration of the data collection and analysis by 2012 is recommended. An integrated assessment will set the scene for joint management advice, as a basis for strengthening cooperation between HELCOM States with regard to protection of eel in the Baltic Sea.

The Workshop on the Baltic Eel recommends:

- To coordinate, standardize, integrate and jointly organize eel stock monitoring in the Baltic;
- To set up data exchange/storage procedures for data on the Baltic eel stock (recent and historical data);
- To initiate (new) field programmes to quantify the interactions between management areas in the Baltic (marking restocked eels, international silver eel tagging experiments);
- To organize a series of practical workshops on eel data collection and working procedures, to support local programmes, to coordinate and standardize, and to explore post-evaluation methods for local eel stocks;

- To evaluate the status of the stock, the anthropogenic impacts and the effect of protective measures by 2012 on a pan-Baltic level;
- To develop pan-Baltic management advice by 2012.

2 Data and trends

Chapter 2 addresses the following Terms of Reference:

a) Assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans; examine criteria for defining a recovery.

and also links to:

d) Provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods.

2.1 Recruitment

2.1.1 Temporal trends in recruitment

Information on recruitment is provided by a number of datasets, relative to various stages, (glass eel and elver, yellow eel), recruiting to continental habitats (Dekker, 2002). The recruitment time-series data in European rivers and a description of the dataseries are presented in Annex 4, Tables 2.1 and 2.2.

The time-series used for recruitment analysis are coming from 48 rivers in eleven countries (Figure 2.1). They were updated to the last season available, which is for most of the cases in 2011. 14 series were available for recruitment analysis in 2011 out of a maximum of 26 in 2006, 7 out of 11 were available for yellow eel (Figure 2.2) and this number was judged adequate, even for the recent years. Some of the series have been discontinued, due to the lack of recruits in the case of fishery based survey (the Ems, Germany) and some others due the lack of financial support (the Tiber, Italy). Series based on fisheries data were discontinued in France as the catch statistics no longer report the precise location of the catch, but only give details by Eel Management Unit.

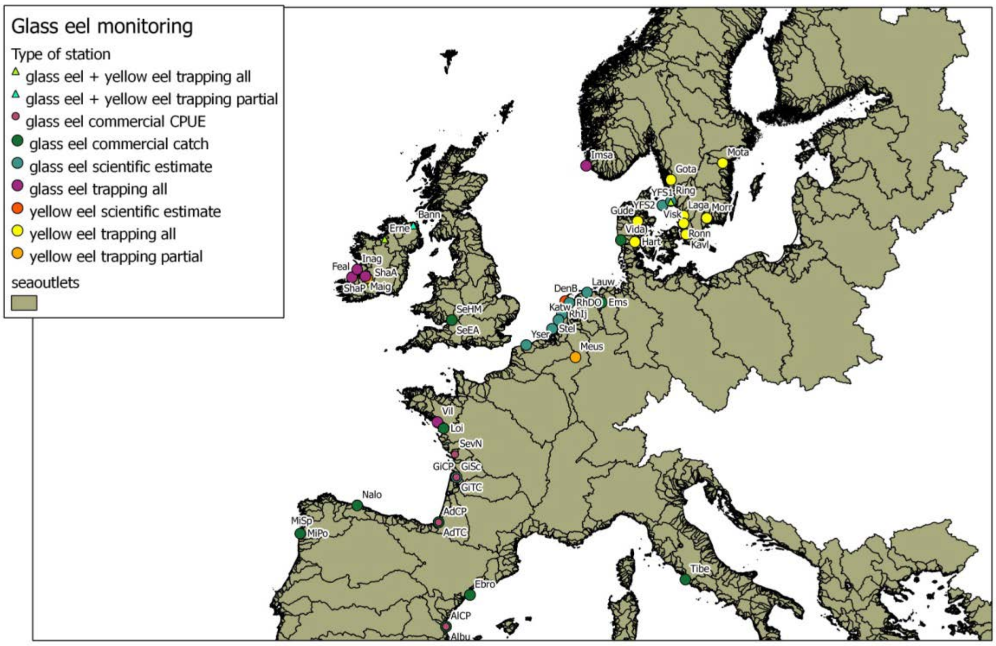


Figure 2-1. Location of the recruitment monitoring sites in Europe. The code of the stations and their short description is given in Annex 4, Table 2.1.

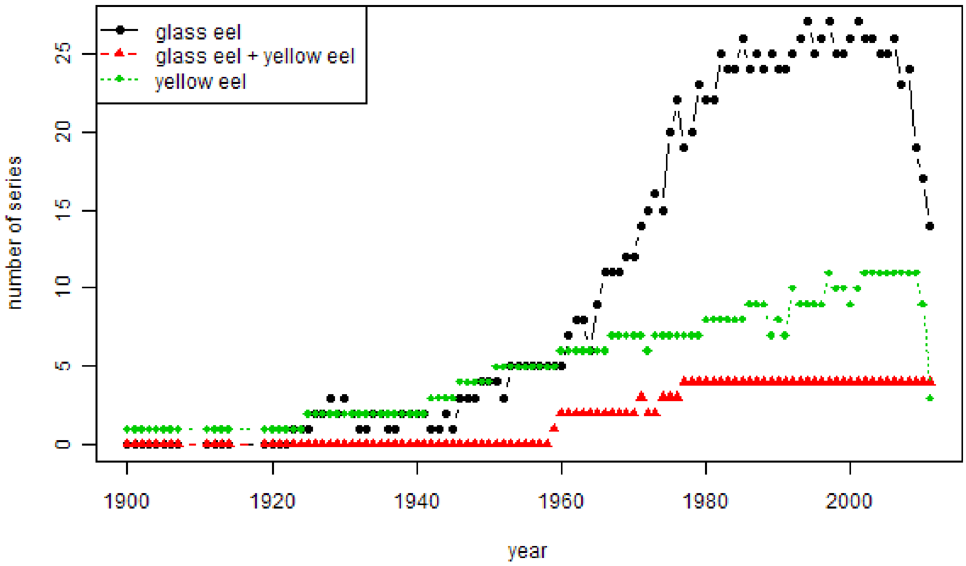


Figure 2-2. Trend in number of available dataseries per life stage.

The recruitment time-series data were derived from fishery-dependent sources (i.e. catch records) and also from fishery-independent surveys across much of the geographic range of European eel. The series cover varying time intervals and only those series covering >35 years where selected for a final analysis of the trend. Some series date back as far as 1920 (glass eel, Loire, France) and even to the beginning of 20th century (yellow eel, Göta Älv, Sweden).

The series have been classified according to the type of data: commercial cpue, commercial total catch, scientific estimate, trapping partial (i.e. only a part of the glass eel or yellow eel are caught) and trapping all (all glass eel and yellow ascending a particular point of the river are caught). The glass eel recruitment-series have also been classified according to area: continental North Sea, elsewhere Europe, with the two different trends identified by spatial analysis (ICES 2010a). The Baltic area does not contain any pure glass eel series. The yellow eel recruitment-series are comprised of either a mixture of glass eel and young yellow eel, or as in the Baltic, are only of young yellow eel.

For graphical presentation, the series are scaled to 1979–1994 as it is not possible to set an appropriate reference earlier than 1980 for most of the series. But the reconstructed values when using the GLM analysis (Generalised Linear Model) are given in reference to the mean reconstructed estimate of the 1960–1979 period. Declining trends are still evident over the last three decades for all time-series. After high levels in the late 1970s, there was a rapid decrease that still continues to the present time (Figures 2.3–2.6; note the logarithmic scale). The glass eel landings data in 2010 and 2011 were higher than in 2009, but remain at a low level.

For the last five years the WGEEL recruitment index average between less than 1% (continental North Sea) and 5% (elsewhere in Europe) of 1960–1979 levels respectively (Annex 4, Table 2.3, Figures 2.5 and 2.6). The revision of the index calculated for the Netherland series had consequences on the scaling of the data on the 1979–1994 average, but not on the overall trend of recruitment.

The series for yellow eel recruitment remain at a low level around 10% of their mean of 1960–1979 levels (Figure 2.6; Annex 4, Table 2.3).

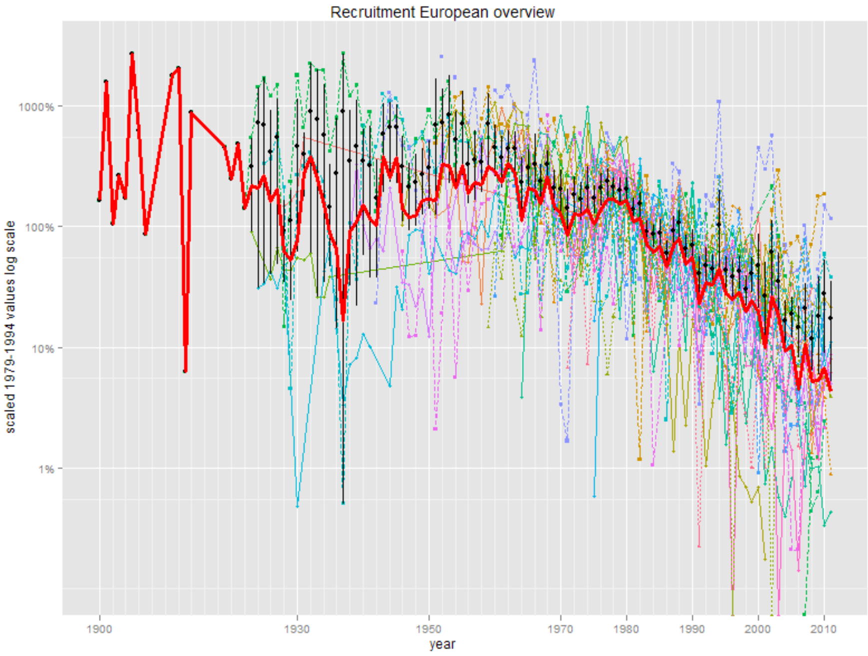


Figure 2-3. Time-series of glass eel and yellow eel recruitment in European rivers with dataseries >35 years (26 rivers). Each series has been scaled to its 1979–1994 average. Note the <u>logarithmic scale</u> on the y-axis. The mean values and their bootstrap confidence interval (95%) are represented as black dots and bars. Note: for practical reasons, not all series are presented in this graph. Geometric means are presented in red.

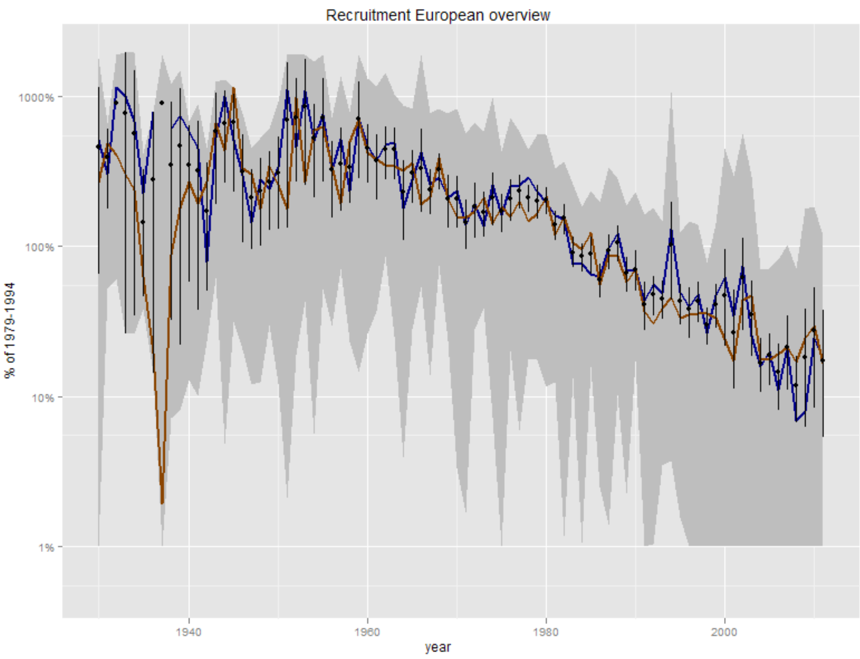


Figure 2-4. Time-series of glass eel and yellow eel recruitment in European rivers with dataseries >35 years (26 rivers). Each series has been scaled to its 1979–1994 average. Note the <u>logarithmic scale</u> on the y-axis. The mean values of combined yellow and glass eel series and their bootstrap confidence interval (95%) are represented as black dots and bars. The brown line represents the mean value for yellow eel, the blue line represents the mean value for glass eel series. The range of the series is indicated by a grey shade. Note that individual series from Figure 2.3 were removed for clarity.

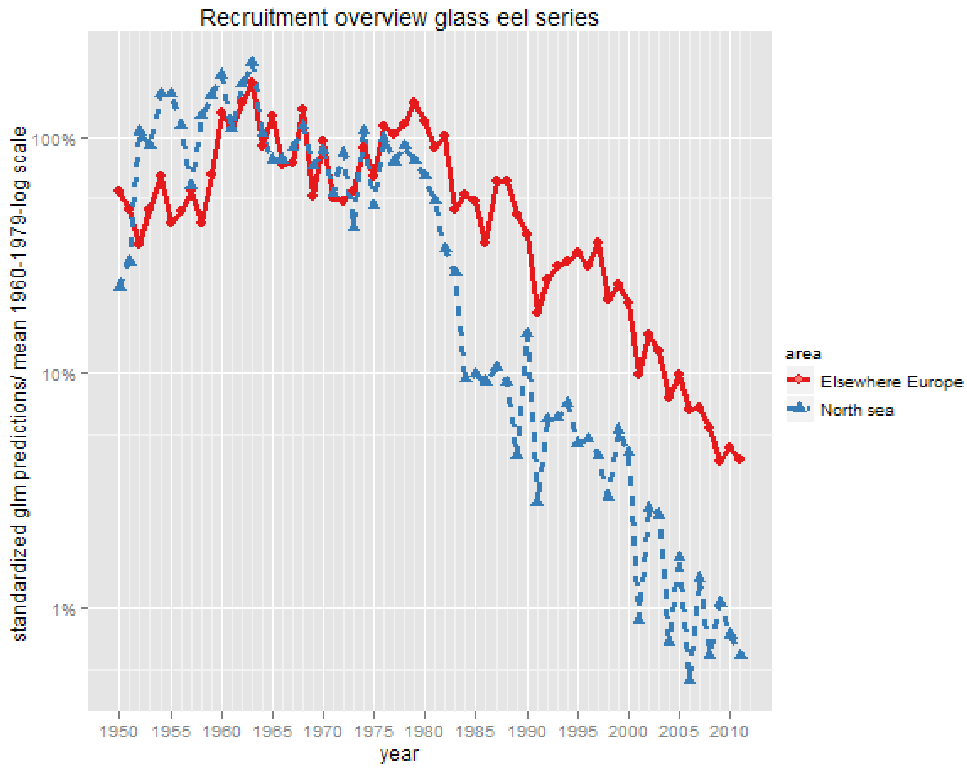


Figure 2-5. WGEEL recruitment index: mean of estimated (GLM) glass eel recruitment for the continental North Sea and elsewhere in Europe. The GLM (recruit=area:year+site) was fitted to all glass eel series available and scaled to the 1960–1979 average. No series for glass eel are available in the Baltic area. Note <u>logarithmic scale on the y-axis</u>.

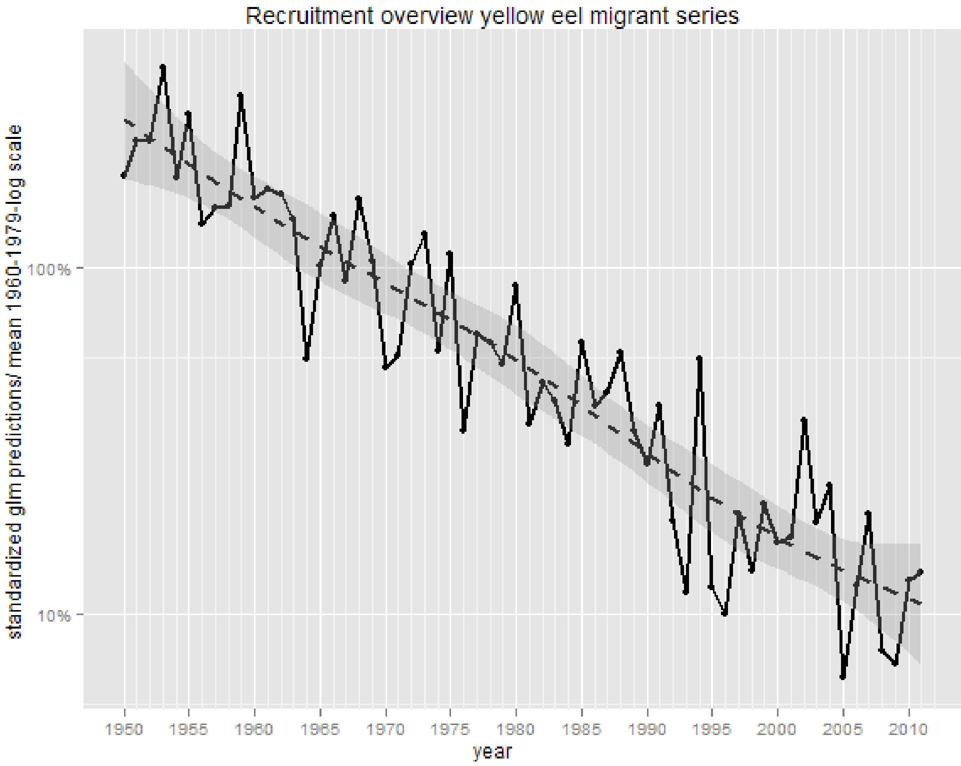


Figure 2-6. Mean of estimated (GLM) yellow eel recruitment and smoothed trends for Europe. The GLM (recruit=area:year) was fitted to all yellow eel series available and scaled to the 1960–1979 average. Note logarithmic scale on the y-axis. Bands show 95% point-wise confidence interval of the smoothed trend.

2.1.2 Exponential trend in recruitment

The calibration of population dynamics models for eels is based on the exponential trend of the WGEEL recruitment indices.

$$R(t) = R(t_0)e^{-D(t-t_0)}$$

The D coefficient was calculated for 1980–2011 as D_{NS}=-0.1285 for the continental North Sea and D_{EE}=-0.0982 elsewhere in Europe.

The effect of management plans initiated in 2009 have not yet led to any changes in recruitment. The test of a change in slope *lm(lEE~year+pmax(year, 2009))* was not significant p=0.47.

2.1.3 Data discontinuities

It was cautioned by the EIFAC/ICES Working Group on Eel (ICES, 2008, 2009, 2010b) that data discontinuities, particularly related to data from commercial fisheries, can be expected following implementation of EMPs (e.g. management measures affecting fishing effort, season quota, size limits), and CITES restrictions. However, from 2008 the loss of four long-term recruitment-series in France is only the consequence of changes brought in the catch report compilation. The only data currently available is at the EMU level and this prevents separating out recruitment data collected in the estuaries.

The paucity of recruitment data for the Mediterranean is again highlighted by the working group, as there are only three series remaining, each from commercial fisheries which may change in future.

The yellow eel time-series remain largely unaffected by any changes due to the implementation of management measures: none have closed and only two appear vulnerable.

The expected changes to the recruitment time-series due to the implementation of management measures, particularly the glass eel time-series, would reduce the data available for analysis by almost half. This means the provision of scientific advice on changes to the stock based on recruitment-series will in the next years become vulnerable and it is unlikely that statistical modelling will be able to correct for this.

2.2 Power to detect a change in the trend

2.2.1 Introduction

A power analysis was carried out on silver eel escapement trends (ICES 2011). It was concluded that no change in silver eel escapement could be shown for at least a decade, even if huge changes in management occurred. However, as the collection and analysis of silver series is still preliminary (see Section 2.3), the most appropriate way for the WGEEL to follow the status of stock has so far been the level of recruitment. The objective of this analysis was to pursue a power analyses on recruitment-series using simple assumptions on the projected forward trend.

The main idea can be brought forward in the following example. Assuming that at a given year (let's say 2009, the official entry into force of eel management plans) the recruitment stops declining at the same rate or even starts to increase. However the natural variability (the decline is not simply a line), the spatial variation (more or less in a given site this year for some local reasons) or the measurement error, will impair the detection of immediate change. On the other hand, if the recruitment continues to decline at the same rate, it might even generate false detection of change. In statistical terms, the latter is called the type I error, noted α (and usually fixed at 5%) and the former is the type II error, noted β. The ideal recruitment survey should minimize type I error: i.e. avoiding thus to produce false hope and type II error: i.e. ensuring thus the detection of change with a high probability, this probability being the power of the test (1- β).

In this Section, the test of change in the slope of the recruitment trend is analysed. The power of this test is assessed for ways of possible optimization. For that, the characteristics of recruitment-series are analysed. They are then used to simulate various futures (declining, stabilizing or increasing recruitment). We finally analyse those simulations to infer the power of our recruitment analysis.

2.2.2 Estimating the characteristics of recruitment data

The assumption made for the remainder of the analysis is that the recruitment index of the WGEEL is a reflection of the true trend in recruitment, which is just an exponential negative. For simplicity, the analysis was restricted to the "Elsewhere Europe" recruitment-series, thus excluding the recruitment-series from the continental North Sea. The recruitment index provides an exponential trend in the decline ('exponential rate' D_{NS}=-0.0982) (Section 2.1.2) and a year-to-year variation around this trend ('overall variance' V_o = 0.27). At a given site, local conditions and sampling errors cause the recruitment-series to deviate from the overall recruitment and can be characterized by its variance around the WGEEL recruitment index ('site-specific variance' V_s and SD_s for the resulting standard deviation). All data were log-transformed and all values provided in the next paragraph correspond to log transformed values.

The site-specific variances have been estimated as the variance of the residual of site recruitment-series compared to WGEEL recruitment index (Table 2.4). The highest standard deviation (the Ebro) is fourfold higher than the lowest standard deviation (the Vilaine). WGEEL 2010 computed Mandel's coefficient which provided a computation of the trend in the series relative to the overall trend (was the series decreasing faster or more slowly than the mean trend?). Compared to last year's analysis (ICES 2010a, Ch. 2.1.3.1.4), series with the lowest Mandel's coefficient (thus closest to the general trend) corresponded to those having low SD_s (e.g. Vilaine, Loire, Gironde) but also to some having large SD_s (e.g. Bann and Ebro). In all cases, a high Mandel's coefficient (thus series deviating from the general trend) leads to high SD_s (e.g. Erne or Nalon).

Table 2-4. Site-specific standard deviations.

short name	SDs
Vil	0.23
Loi	0.32
GiCP	0.36
GiTC	0.40
MiSp	0.45
AdCP	0.53
MiPo	0.55
Nalo	0.61
Albu	0.65
Erne	0.71
SeEA	0.72
Bann	0.83
Ebro	0.87

2.2.3 Simulating possible futures

Five possible futures have been envisioned:

- the declining rate accelerates (not tested here);
- the recruitment continues to decline at the same rate (called here 'BAU' – Business As Usual – scenario);
- the recruitment continues to decline but at a lower rate (not tested here);
- the recruitment remains at a constant rate (called here 'stable' scenario);
- the recruitment increases (called here 'increase' scenario).

For the 'increase' scenario, we have arbitrarily used an exponential increase rate of 0.05. The simulations cover the 1980–2030 period with a trend at the 'exponential rate' from 1980 to 2009 and one of the above options from 2010 onward. For each scenario, 1000 simulations have been made according to the following two steps:

- an overall recruitment-series is generated from the trend and the overall variance;

- 'Elsewhere' recruitment dataseries are generated from this overall recruitment-series and by adding the site-specific variance.

From each simulation (see Figure 2.7 for an example) we have computed from the WGEEL recruitment index from all series, the four 'best' (lowest SD_s: Vil, Loi, GiCP, GiTC) series or the four 'worst' (highest SD_s: Ebro, Bann, SeEA, Erne) series following the usual procedure (Ch. 2.2.1). We have finally computed from the WGEEL recruitment index a 2-slopes regression[1], with a year of change fixed at 2009. The coefficient and the p-value of slope after 2009 have been stored.

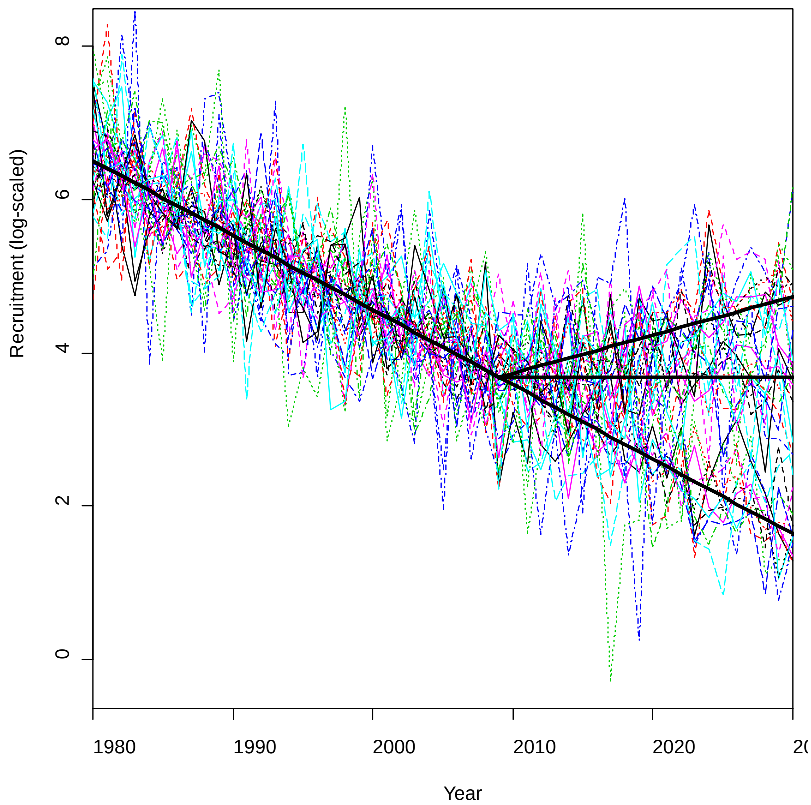


Figure 2-7. A simulation (randomly chosen) for the three scenarios (black lines). Each curve is a site-specific recruitment-series.

2.2.4 Analysis of the simulation

For each year between 2011 and 2030, the percentage of simulations for each scenario that shows a significant slope was calculated (taking α=0.05) (Figure 2.8).

According the time, the number of significant detections are increasing for 'stable' and 'increase' scenarios. For the 'BAU' scenario, this number is stable at around 5% corresponding to our chosen α. The trend computed from series with the lowest SD_s ('best') and the trend derived from 'all' series are those providing the earlier detection (about 97% in 2014 for the increase scenario and 91% in 2015 for the stable scenario), while it takes much more time with the lowest SD_s recruitment-series (worst = 96% in 2018 for the increase scenario and 92% in 2021 for the stable scenario).

[1] as explained in SGIPEE 2011 (Ch. 4.3) and tested in Ch. 2.2.2.

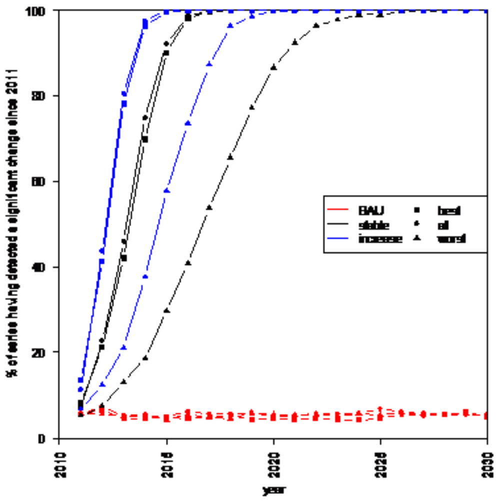


Figure 2-8. Percentage of significant post-2009 change in slope (α=0.05) for three scenarios: business as usual (BAU), stable trend after 2009 (decrease) and increase after 2009 ((rate = 0.05) and three batches of recruitment-series: all series (all), the four lowest SD_s (best) and the four highest SD_s (worst).

Another use of the results described above is to compute the power of our recruitment analysis. The principle is to compare the distribution of slope coefficient with the null hypothesis (BAU scenario) to the distribution with the alternative hypothesis (stable or increase scenario). More precisely, we count the percentage of slope coefficient for the alternative scenarios that are below the 95-percentile (α=0.05) of slope coefficient for the null scenario (1-tail test). This percentage is the power of the test for a variation in the recruitment trend slope (Figure 2.9).

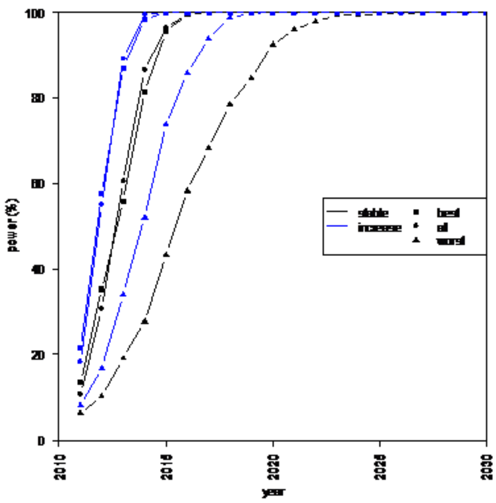


Figure 2-9. Power test to detect a variation in the slope of the recruitment trend (α=0.05) for the stable and increase scenarios and taking all series (all), the series with the four lowest variance (best) and the series with the four highest variance (worst).

As above, the best and all series performed better than the worst series: for example with the stable scenario, the usual 80% level is reached as soon as 2013 for best and all series, but in 2016 only for series with highest SD's.

A way to shorten the period before the detection of a variation in trend is to relax the type I error (increase the α level). For instance, if α is taken as 0.20, the change is detected in 2012 for all series and stable scenario. However, this comes at an increased risk of false detection; see also Section 2.2.5.

2.2.5 Implications for management

In the best situation, the detection of a change in the recruitment trend will take four years. Moreover, management action taken from 2009, even if they concern the silver eel phase, will bring about a change in recruitment only two to three years later, given the delay between silver eel migration and the return of glass eel offspring to our coasts. Moreover many countries have taken progressive management measures leading to (if any) a slow recovery process and certainly to a slow change (if any) in the recruitment trend while we have simulated a clear-cut point in recruitment data from 2009. Thus if any change (hopefully an increase) in recruitment would occur, it is highly likely that it would not be detected for many years (if not a decade or more). This confirms SGIPEE's results from the power analysis on silver eel escapement (ICES, 2011, Ch. 4) and emphasizes the need to work on anthropogenic mortality in the short term instead of focusing only on the status of the stock (either recruitment or silver eel).

Finally, the risk of losing recruitment-series mainly in the central part of the distribution due to change in fisheries monitoring and management was highlighted in Section 2.2.3. Those series are among the closest to the general trend and the lowest SD's make them of a high value for the recruitment analysis. This loss (if it occurs) will weaken the Working Groups ability to detect any change in the overall recruitment pattern or trend. This highlights the importance of a structured recruitment monitoring programme.

2.3 Time-series of yellow and silver eel

Eleven scientific time-series of silver eel escapement and three time-series of yellow eel have been included or updated in the WGEEL database. A detailed analysis of the trawl surveys from the Baltic and the North Sea has been provided to the Working Group (Annex 5). Further data collection and analysis is required before the time-series can be used to assess the status of the stock in different geographical areas.

2.4 Data on landings

In WGEEL 2010, data on total eel landings obtained from country reports were presented, without data on official eel landings from FAO sources which completely differs from country report data.

Implementation of the EU Eel Regulation requires Member States to implement a full catch registration system, along with the Data Collection Framework. This would lead to considerable improvement of the coverage of the fishery, i.e. underreporting will probably reduce markedly. Not all countries developed that system under the DCF (Table 2.13).

However, at the present 2011 status, dataseries from the Country Reports still continue to be unreliable. A review of the catches and landing reports in the CR showed a great heterogeneity in reporting landings data, with countries making reference to an official system, some of which report total landings, others report landings by Management Unit or Region, and some countries haven't any centralized system. Furthermore, some countries have revised their dataseries, with extrapolations to the whole time-series, for the necessities of the Eel Management Plan compilation (Poland, Portugal).

Because landings data were incomplete, with some years missing for some of the countries, an estimate of the missing values is provided by simple glm extrapolation (after Dekker, 2003), with year and countries as the explanatory factors (Figure 2.10).

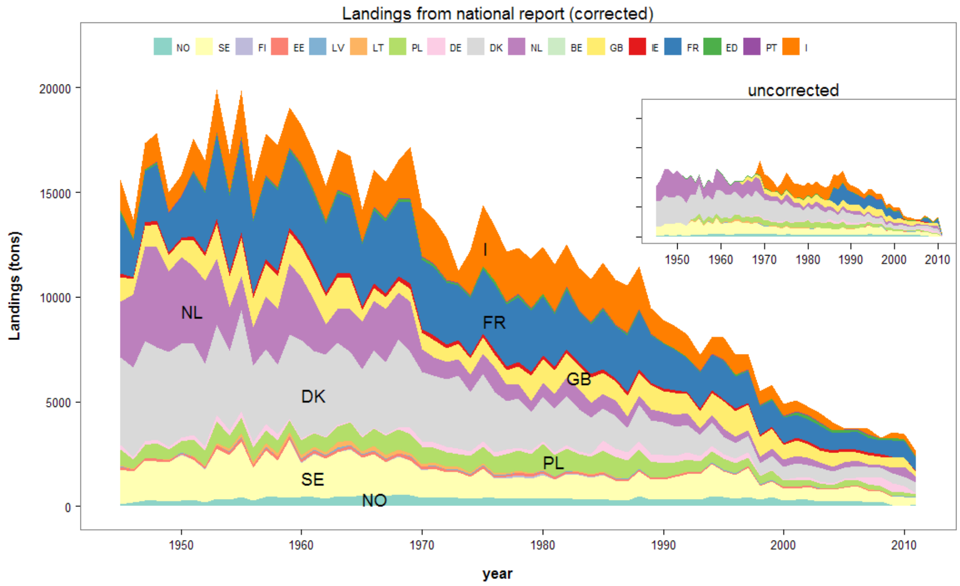


Figure 2-10. Total landings (all life stages) from 2011 Country Reports (not all countries reported); the corrected trend has missing data filled by GLM.

2.4.1 Collection of landings statistics by country (from CRs)

Landings data are presented in Annex 4, Table 2.5.

Norway provided official landing statistics (Fisheries Directorate) calculated according to the number of licences.

Sweden: Data on eel landings in coastal areas are based on sales notes sent to the appropriate agency, while landings data from freshwaters come from a logbook system. Fishing for eels in private waters was not reported before 2005. Data from logbooks and journals are stored at the Swedish Agency for Marine and Water Management**.**

Finland: Eel catch data available for period from 1975–2010. Data from professional fishermen collected by logbooks, recreational by questionnaire.

Lithuania: no data available.

Estonia: The catch statistics are based on logbooks from inland and coastal fisheries.

Latvia: Eel landing statistics were collected on coastal fishery by voluntary reporters in period from 1924–1938, by fishing enterprises (state and cooperative) official reports from 1946–1992, by monthly logbooks (daily records of catch) from legal and private persons using professional fishing gear until now. Eel landing statistics in inland waters were collected from state fishing companies from 1946–1992, by monthly logbooks (records by fishing day catch) from legal persons using professional fishing gear until now. Formats of logbooks are formalized and defined by Cabinet regulation. Coastal eel fishery data are stored in ICIS database administered by Department of Fisheries Ministry of Agriculture.

Poland: The data on inland catches were obtained by surveying selected fisheries facilities, and then extrapolating the results for the entire river basin. These data are thus approximate. The data from the lagoons and coastal waters were drawn from official catch statistics (logbooks).

Germany: Eel landings statistics form coastal fishery is based on logbooks. The obligation to deliver the inland catch statistics separate for both stages has only recently been established in most of the States. Fishermen have to deliver the information at least on a monthly basis to the authorities.

Denmark: The yellow and silver eel catches are reported by commercial fishermen.

Netherlands: For Lake IJsselmeer, statistics from the auctions around Lake IJsselmeer are now kept by the Fish Board. For the inland areas outside Lake IJsselmeer, no detailed records of catches and landings were available until 2010. In January 2010 the Ministry of Economic Affairs, Agriculture and Innovation introduced an obligatory catch recording system for inland eel fishers.

Belgium: There are no commercial eel fishing in Belgium.

UK: Environment Agency and it is a legal requirement that all eel fishermen submit a catch return. Licensees are required to give details of the number of days fished, the location and type of water fished, and the total weight of eel caught and retained, or a statement that no eel have been caught. Annual eel and elver net licence sales and catches are summarized by gear type and Agency region (soon to be RBDs) and reported in their "Salmonid and Freshwater Fisheries Statistics for England and Wales" series (www.environment-agency.gov.uk/research/library/publications/33945.aspx).

Ireland: Until 2008 eel landing statistics in Ireland collected from voluntary declarations, from 2009 commercial and recreational fishing of eel are closed.

France: Level of glass eel landings were processed from the database from OFIMER for marine fishermen and ONEMA for fluvial fishermen. Three sources of data can be used: landings, trader statistics (unofficial) and EU trade statistics. Landings data are not available for 2009. Data for 2010 are official. Yellow and silver eel catch statistics are estimated due to the paucity of official statistics.

Spain: Data on eel landings in country mostly are collected from fishermen's guild reports and fish markets (auctions). The precision of the information of the catches and landings differs greatly among Autonomies.

Portugal: Fisheries managed by DGPA have obligatory landing reports, contrary to catches from inland waters, which are not reported.

Italy: Detailed data on catches and landings (by life stage, by type of fishing gear, by EMU, commercial and recreational, etc.) are available only from 2009, when the DCF has been definitively put in place.

Morocco: No data available.

2.5 Recreational and non-commercial fisheries

Data for recreational catch, (via angling methods), and non-commercial landings for 2010 are not presented by each country/ region. As a result, updates are still not available for the recreational and non-commercial data presented in the WGEEL 2008 report (Tables 2.1 and 2.3). Therefore, recreational and non-commercial components are presented as a status in terms of each life stage (Table 2.6). However some estimations of catch volume are available for Norway, Poland Denmark, Netherland and Spain.

The legal framework for collection of recreational fisheries data by EU Member States is given by the EU Data Collection Framework (Council Regulation (EC) No 199/2008 and Council Decision 2008/949/EC). The species for which recreational fishery data are to be collected in each area are:

- Baltic (ICES Subdivisions 22–32): Salmon, cod and eel;
- North Sea (ICES Division IV and VIId) and Eastern Arctic (ICES Division. I and II): cod and eel;
- North Atlantic (ICES Division V–XIV): Salmon, sea bass and eel;
- Mediterranean and Black Sea: bluefin tuna and eel.

Estimates of recreational fisheries catches will be used in the near future in ICES stock assessment for the above mentioned species where relevant. The main aim of the new Working Group on Recreational Fisheries Surveys is to develop guidelines for survey design and analysis methods for the estimation of recreational fishing effort and catch totals.

Table 2-6. Status and catch volume (if available; in t) of recreational and non-commercial eel fishing in 2010 'Prohibited' (by law), 'Active' (permitted under regional angling licence), 'n/a' (not applicable due to non-occurrence in the region).

Country	Glass eel	Yellow eel	Silver eel
Norway	Prohibited	Prohibited	Prohibited
Sweden	Prohibited	Prohibited	Prohibited
Finland	n/a	no catches	no catches
Latvia	n/a	Active	Active
Poland	n/a	Active/70	Active
Germany	n/a	Active	Active
Denmark	n/a	Active/100	Active
Netherlands	Prohibited	Prohibited	Prohibited
Belgium	Prohibited	Active/30	Active
UK	Prohibited	Active**	Active**
Ireland	Prohibited	Active	Active
France	Prohibited	Active	Prohibited
Spain	active*/0.7	Active	Active
Portugal	Prohibited	Prohibited	Prohibited
Italy	Prohibited	Active	Active
Estonia	n/a	Active	Active

*** Estimates for Basque inner basins RBD and Cantabria.**

**** Recreational angling for eel in UK (England and Wales) is only under "Catch and Release" so all eels must be returned to the water alive.**

2.6 Trends in stocking

Data on stocking were obtained from a number of countries, separated for glass eels and for young yellow eels.

An overview of data available up to 2010 (partly 2011) is compiled in Annex 4, Tables 2.7 and 2.8. Note that various countries use different size and weight classes of young yellow eels for restocking purposes.

2.6.1 Stocking review notes

Sweden: since 2006 only imported and quarantined glass eels are eligible for stocking supported with public money. From 2009 all glass eels are marked with strontium chloride ($SrCl_2$) in their otoliths. Since 2010 glass eels are imported exclusively from the Bay of Biscay (Charente-Maritime in France)

Finland: In 2011, only 200 000 individual eels were stocked (37% of the amount in EMP).

Lithuania: No data available.

Estonia: Historical database available on restocking of glass eel/young yellow eel in Estonia since 1950. During the period 2011–2014 the stocking of eel into L. Peipsi basin will supported by EFF up to 255 000 EUR (co-financing up to ⅓ of total annual financing). In 2011, 680 000 glass eels were stocked (UK glass eels).

Latvia: Historical database of restocking collected from different sources. Data from 1945–1992 obtained from archives of USSR institution Balribvod responsible on fish restocking and fisheries control in former USSR. Since 1992 every restocking of fish in natural waterbodies in Latvia must be reported to BIOR by special papers.

In 2011, Latvia started restocking in accordance with Latvia's EMP, 100 kg glass eels were restocked (in late May) in the river Daugava basin lakes connected with Gulf of Riga (ICES Subdivision 28). Glass eel were imported from Glass Eel UK by a supplier from Czech Republic.

Poland: In 2011 Poland started restocking within EMP framework. Because of ice coverage in glass eel catch season, about 6 tons of fingerlings (average of 5 grammes) were restocked in August in various waterbodies. Data on private stakeholders restocking comes from eel importers.

Germany: There is no central database on restocking, but some data are available. The quantity of young yellow eels stocked to the waterbodies is significant.

Denmark: Glass eels are imported mostly from France and are grown to a weight of 2–5 gramme in heated culture before they are stocked. Restocking is done as a management measure.

Netherlands*:* Glass eel and young yellow eel are used for re-stocking inland waters since time immemorial, mostly by local action of stakeholders.

Belgium: Glass eel restocking is proposed as a management measure in the EMP for Flanders.

UK (England & Wales), (Scotland): About 37 kg of UK origin glass eel were stocked in rivers of England & Wales in 2010. No eel stocking takes place in Scotland.

UK, N. Ireland: In 2010 the 996 kg of glass eel purchased from Glass Eel UK originated from fisheries in San Sebastian, Spain and the west coast of France.

Ireland: No stocking of imported eel takes place in Ireland.

France: The first restocking nationally organized action started in 2010 in the Loire River. Glass eel comes from a CITES seizure. It further extends in 2011.

Spain: no stocking on a national level. Each autonomy has their own rules and experience concerning restocking.

Portugal: no stocking on a national level.

Italy: No data available.

Morocco: No data available.

Stocking with glass eel has decreased strongly since the early 1990s and appears now to be at a very low level (Figure 2.11). However, this has partly been compensated for by an increasing number of young yellow eels stocked since the late 1980s. During the 1990s stocking of young eel showed an increase but dropped again in the late 1990s (Figure 2.12). During recent years, another increase in stocking young yellow eels was observed. In 2010 stocking of glass eels were the highest since 1999. In case yellow eel highest ever, despite incomplete data from some countries.

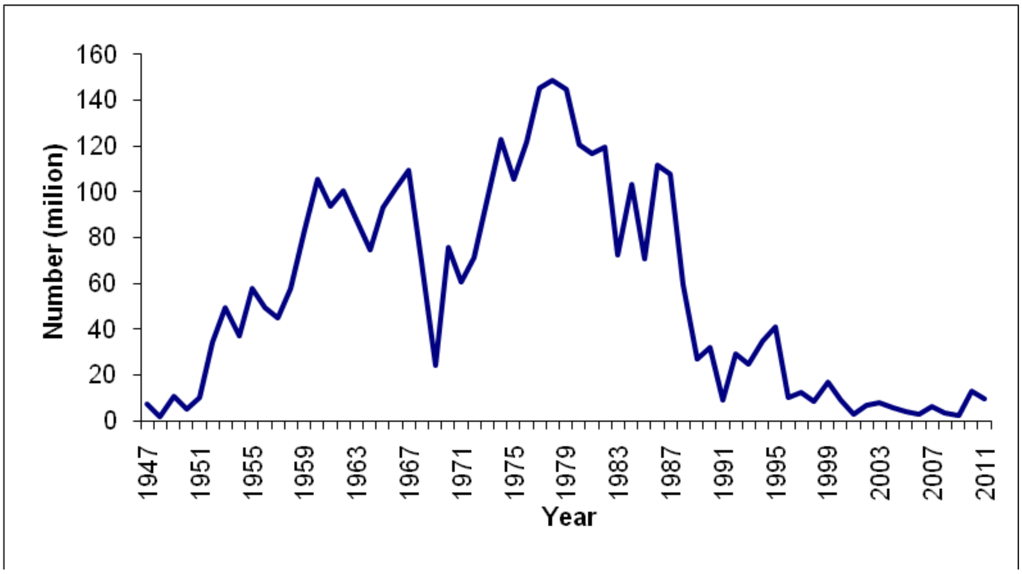


Figure 2-11. Stocking of glass eel in Europe (Sweden, Finland, Estonia, Latvia, Lithuania, Poland, Germany, the Netherlands, Belgium, Northern Ireland, France and Spain) in millions stocked.

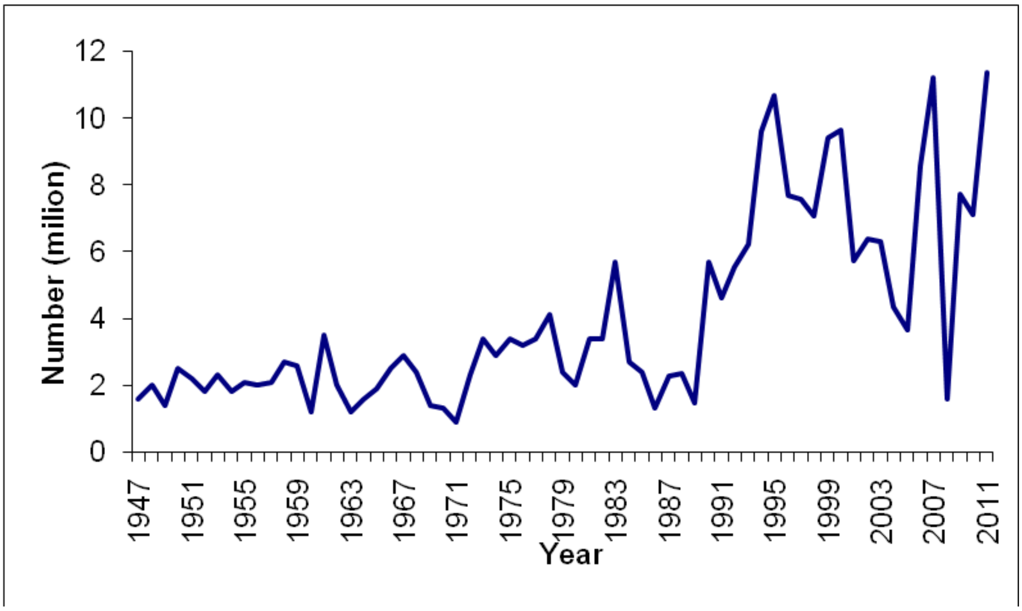


Figure 2-12. Stocking of young yellow eel in Europe (Sweden, Finland, Estonia, Latvia, Lithuania, Poland, Germany, Denmark the Netherlands, Belgium, and Spain), in millions stocked.

2.7 Aquaculture production

Aquaculture production data for European eel limited to European countries from 2003 to 2010 are compiled by integrating different sources, Country Reports to WGEEL 2011 (Table 2.9), FAO (Table 2.10) and FEAP (Table 2.11). Some discrepancies still exist between databases and the national reports annexed to this report, but overall the trend in aquaculture production is decreasing (Figure 2.13). Some of the discrepancies between FAO and the country reports data result from eel used for re-stocking not being reported to the FAO.

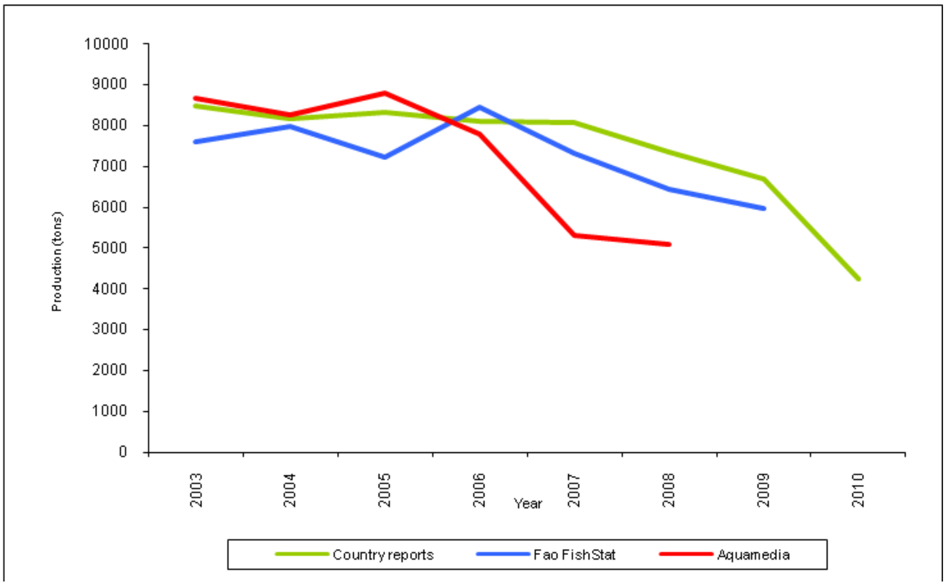


Figure 2-13. Different sources of data for aquaculture production of European eel in Europe from 2003 to 2010, in tonnes.

Table 2-9. Aquaculture production of European eel in Europe from 2003 to 2010, in tonnes. Source: Country Reports.

	2003	2004	2005	2006	2007	2008	2009	2010
Denmark	2050	1500	1700	1900	1617	1740	1707	1537
Estonia	18	26	19	27	52	45	30	20
Germany	372	328	329	567	740	749	667	681
Netherlands	4200	4500	4500	4200	4000	3700	3200	2000
Portugal	4.7	1.5	1.4	1.1	0.5	0.4	1.1	no data
Sweden	170	158	222	191	175	172	139	91
Poland	1	1	1	1	1	1	1	1
Italy	1325	1220	1131	807	1000	551	587	no data
Spain	339	424	427	403	478	385	370	?
Total	8475	8159	8330	8096	8064	7343	6705	4330

Table 2-10. Aquaculture production of European eel in Europe from 2003 to 2009, in tonnes. Source: FAO FishStat.

	2003	2004	2005	2006	2007	2008	2009
Denmark	2012	1823	1673	1699	1614	895	1659
Estonia	15	7	40	40	45	47	30
Germany	50	322	329	567	440	447	385
Netherlands	4200	4500	4000	5000	4000	3700	2800
Portugal	5	2	1	2	1	0	0
Sweden	170	158	222	191	175	172	0
Poland	0	0	0	0	0	0	0
Italy	200	186	177	151	102	158	167
Spain	399	424	427	403	479	534	488
Greece	544	557	372	385	454	489	428
Lithuania						11	12
Romania				6	7		
Total	7595	7979	7241	8444	7317	6453	5969

Table 2-11. Aquaculture production of European eel in Europe from 2003 to 2008, in tonnes. Source: Aquamedia.

	2003	2004	2005	2006	2007	2008
Denmark	2050	1500	1610	1760	1870	1500
Estonia						
Germany						
Netherlands	4200	4500	4500	4200	3000	3000
Portugal						
Sweden	194	158	170	170	170	170
Poland						
Italy	1400	1200	1200	1200		
Spain	315	390	805	440	280	430
Greece	500	500	500			
Hungary	20	20	20	20		
Total	8679	8268	8805	7790	5320	5100

2.8 Socio-economics of fisheries

An evaluation of what data are available and quality assessment of the country reports were carried out. The Data Collection Framework under the CFP requires the collection of data on eel fisheries since 2004, and specifically for eel in inland waters and recreational fisheries since 2009 (Regulation 199/2008).

For most of country reports, the basic indicators on the status of eel fisheries (e.g. fishing capacity, fishing effort, catch) were missing or incomplete. The inaccuracy and poor representativeness of these indicators leads to wide uncertainties, and prevents any comparison between the situation in 2008 (before implementation of EMP's) and 2010 (reference year of the table) (Table 2.12).

The best estimate is of 7750 professional fishermen and an unknown number of recreational fishermen landing 2280 t and 1650 t respectively, giving a total of around 4000 t of eel in Europe. Despite decreasing abundance of eel and awareness of contamination, **fisheries are still important** and widespread over Europe involving both full-time and part-time professionals, "hobby" professionals, eel specific, multispecies (including eel), life stage specific (glass eel), amateur fishermen as well as, anglers and illegal fishermen (poachers). Poaching was often widespread in some countries and might, in some places, continue despite legal closures of the fishery. As such, there is a risk of an increase in illegal fishing as legal fisheries decline.

An assessment of the present status of eel fisheries at the European level is not possible at the national or local level. The inaccuracy and poor representativeness of these indicators leads to wide **uncertainties**, and prevents any comparison between the situation in 2008 (before implementation of EMP's) and 2010 (reference year of the table).

The accurate description and analysis of the current status of eel fisheries, and the possible consequence of management actions and stock decline on their economic durability, and sustainable use of the resource, necessitates further work. For this reason, a refined outline of a proposal for an independent study group, possibly under the auspices of EIFAAC, is proposed as in annex to this report (Annex 6).

Table 2-12. Quantitative data available from the WG Country Reports (year 2010).

		NO	SE	FI	LV	EE	PL	DE	DK	NL	BE	UK	IE	FR	ES	PL	I	M'CO
Fishing capacity	Total Commercial	55	387	na	115	?	1311	2382	?	na	0	604	0	1700	?	1727	na	346 [2]
	Total Recreational (anglers)	NA	na	na	100 000		na	706 423 [3]	18 768 [4]	na	114 374	na	na	1 128 919	N	na	536 660	
	Total Recreational (amateurs)									1 690 000		na		6339				
Fishing effort	Total Commercial	449 319		na	115	na	1311	2382	401	na	na	604	0	1058	?	486	1152	684
	Total Recreational	NA		na	?	na	na	na	Na	na	na		na	?	N	na	6392	
	Total Recreational (amateurs)													?				
Catch	Total GE commercial	0	0	0	0	0	0	0	0	na	0	3.64	0	31t	1.5	1.5	0	0
	Total Yellow commercial	32	418	2.2	9.2		178	446	108	447	0	361	0	800t	44.8	11	95.26	15,16[5]
	Total Silver Commercial					28.8			314		0	100	0				154.12	
	Total Yellow recreational	NA	na	na	4	0.84	70	97	100	200	30	na	0	1000	N	na	136.72	na
	Total Silver Recreational	NA	na	na								na	na		N	na	12.80	na

[2] boats

[3] recreational plus anglers

[4] partial

[5] cum

2.9 The Data Collection Framework

2.9.1 Eel in DCR

Eel was first included in the DCR in 2004 and the European Commission held a workshop in Sanga Saby, Sweden, in 2005. The objective of the Workshop was to specify minimum requirements on sampling levels for fishery-dependent and fishery-independent data, for the three exploited life stages (glass eel, yellow eel and silver eel), in both inland and coastal waters. The EC required national sampling programmes on European eel to be established by 2006 within the Data Collection Regulation framework, but these were restricted to sampling commercial fisheries and fisheries independent surveys were ignored.

At the time of the 2005 workshop, the mid- and long-term targets of the EC action plan on eel were not yet defined. In the absence of clear targets, it was not possible to implement any monitoring programmes in the DCR. The EC acknowledged that this programme would have to be developed in line with the explicit management goals when defined.

2.9.2 Eel stock regulation

Subsequently, the EU brought in a Regulation for the Recovery of the Eel Stock (EC No. 1100/2007) which required Member States to implement, by 2009, Eel Management Plans (EMP), including eel monitoring, stock surveys and traceability of eel imports and exports. Strong links were suggested with the Water Framework Directive (2000/60/EC).

Under the Eel Regulation, Member States are required to report to the Commission in 2012 on the outcome of the first three years of the EMPs. These "*Reports shall outline monitoring, effectiveness and outcome, and in particular shall provide the best available estimates of: for each Member State, the proportion of the silver eel biomass that escapes to the sea to spawn, or the proportion of the silver eel biomass leaving the territory of that Member State as part of a seaward migration to spawn, relative to the target level of escapement set out in Article 2(4)…*"

Technical evaluation of the reports to the EU in 2012 and a statistical and scientific post-evaluation of the outcome of the report on the implementation of the Regulation are planned under Article 9 of the Regulation and this will require transparent reporting of Eel Management Unit data. These data may be collected within the national eel specific monitoring programme, the DCF and/or the WFD. Harmonization between these programmes is lacking and no individual programme, or combination of programmes, will currently provide sufficient data or detail to facilitate a complete international stock assessment for eel. There is no mandatory reporting mechanism for these data.

2.9.3 Silver eel biomass

SGIPEE (ICES 2010b, ICES 2011) and the WGEEL (ICES 2008, 2009 and 2010a) have been developing a method of international stock assessment for eel to support the EU in the need to evaluate the outcome of the Regulation.

WGEEL (ICES 2010a; Annex 5) therefore recommended that EMP reporting must provide the following biomass and anthropogenic mortality data:

a) $\mathbf{B}_{post}$, the biomass of the escapement in the assessment year;

b) $\mathbf{B}_0$, the biomass of the escapement in the pristine state. Alternatively, one could specify $\mathbf{B}_{lim}$, the 40% limit of $\mathbf{B}_0$, as set in the Eel Regulation;

c) $\mathbf{B}_{best}$, the estimated potential biomass in the assessment year, assuming no anthropogenic impacts (and without stocking) have occurred and from all potentially available habitat;

d) $\sum\mathbf{A}$ the estimation of $\mathbf{B}_{best}$ will require an estimate of $\mathbf{A}$ (anthropogenic mortality (e.g. catch, turbines)) for density-independent cases, and a more complex analysis for density-dependent cases.

2.9.4 Eel and current DCF

The WGEEL have previously addressed the issues of data collection under the various frameworks in its reports of 2008 (Ch. 3.3.5) and 2010 (Ch. 4.3) and ICES Study Group (SGIPEE) also addressed the issue in its 2010 report (Ch. 4).

In summary, the general conclusion has been that the current requirements for eel in the DCF are inadequate and at the wrong spatial scale. To date there are only 14 member states with a DCF programme in place however not all of these data are in a format that can be used in the assessment process (Table 2.13). Five countries are currently using the DCF collected data in their stock assessments (Table 2.14). The DCF is, therefore, not providing the most efficient support for national stock assessments for eel or for EU post evaluation of the implementation of the Regulation. The current inclusion of eel in the DCF is for sampling commercial catches. The cross-compliance link between the Eel Regulation and the DCF process is a useful provision for stock assessment purposes. The current DCF driven data provision for eel is, however, dependent on continuation of commercial and recreational eel fisheries. There is no requirement for any fishery-independent eel sampling in the current DCR, or for any sampling to continue where and when fisheries close. Continuation of commercial eel fishing is far from guaranteed given the continuing downward trends in catches, the possibility of approaching economic extinction, and the probability of widespread cuts in eel fishing activity as a consequence of MS or RBD scale failure to meet the "40%" silver escapement targets required in the eel regulation.

In addition there are several other non-fishery forms of anthropogenic impact on eel production that must be accounted for in stock assessment. Assessment protocols in some member states rely on scientific surveys of stock and not just on fishery data. The present DCF requirement for eel does not take these into account. SGIPEE (ICES 2010b) concluded that many countries are collecting, or intending to collect, fisheries-dependent data. In contrast, few countries are developing fisheries-independent surveys or eel specific monitoring.

For a full international eel stock assessment to be achieved, against which a post-evaluation of the implementation of the EU Regulation can be measured, there is a need for the establishment of a nationally maintained eel stock database of key stock descriptors, including fishing effort, which is made available for international compilation and analysis. These descriptors are; emigrating silver eel numbers, biomass and sex ratio, and recruitment in terms of larvae glass eel or young of the year numbers and biomass. The component and compiled data must be annually updated to enable examination of any stock–recruitment relationship by the assessment working group. Only when such data exist will it be possible to bring eel population and stock–recruitment assessments to the level given to most other major internationally exploited fish species.

Table 2-13. Country participation in the Working Group, Country Reports and DCF and whether DCF data are reported to the WG.

WG: WGEEL				
Country:	Present at WG	Country Report	DCF Implemented	DCF data in Country Rpt to WG
Belgium	Yes	Yes	No	No
Denmark	Yes	Yes	Yes	No
Estonia	Yes	Yes	Yes	Yes
Finland	No	Yes		
France	Yes	Yes	Yes	Yes
Germany	Yes	Yes	Yes	Yes
Greenland	No			
Iceland	No			
Ireland	Yes	Yes	No	No
Latvia	Yes	Yes	Yes	No
Lithuania	No	No	No	No
Netherlands	Yes	Yes	Yes	Yes
Norway	Yes	Yes	Yes	Yes
Poland	Yes	Yes	Yes	Yes
Portugal	Yes	Yes	Yes	Yes
Russia	No			
Spain	Yes	Yes	No	No
Sweden Freshwater	Yes	Yes	Yes	Yes
Sweden Marine	Yes	Yes	Yes	Yes
UK	Yes	Yes	Yes	Yes
UK-North Ireland	Yes	Yes	Yes	Yes
Faroe Islands	No			
Italy	Yes	Yes	Yes	Yes
Morocco	Yes	Yes	No	No

Table 2-14. Summary of the number of Countries applying DCF data to Eel assessments.

	Landings: Age	Landings: Length	Landings: Weight	Landings: Maturity	Landings: Sex Ratio
A. Not relevant	2	2	2	2	2
B. Data relevant but not available to ICES	2	1	1	3	5
C. Data available to ICES but not used in assessment	2	3	3	3	2
C.1. Data available but time-series too short	1	1	1	1	1
C2. Data available but not relevant in the model we use for assessment for the time being	1	1	1	1	
C.3. Data are available from this country but because other countries do not submit it we cannot use it					
C.4. Data are available but they have insufficient representativity or quality for us to use them	4	4	4	4	3
D. Data available and used in assessment *	5	5	5	3	4
Total	17	17	17	17	17
%	29	29	29	18	24

*** D: Used in National Assessments, not in assessments by WGEEL.**

2.9.5 Other information not collected in DCF

The EU Regulation requires other information to be reported but to date this has not been coordinated and the DCF might provide a suitable framework for some or all of these. These are:

Reg. Article 7–5 *"The Commission shall annually report to Council on the evolution of market price of eel <12 cm in length. Member States shall establish an appropriate system to monitor prices and shall report annually to the Commission".*

Reg. Article 10 *"Control and Enforcement in waters other than Community Waters".*

Reg. Article 12 *"Control and enforcement concerning imports and exports of eel."*

Take the measures necessary to identify origins and ensure traceability of all imports and exports and make efforts to ensure these were captured in a manner consistent with Community conservation measures.

2.10 Conclusion on DCF

Eel specific fisheries-independent surveys are lacking in most member states and need international impetus and coordination. Fisheries-dependent surveys, especially recruitment data, are becoming increasingly vulnerable. Analysis of recruitment time-series data has been the main tool in the past for assessing the overall status of the stock. SGIPEE (ICES 2010b) and WGEEL (ICES 2010a) identified that out of 47 time-series available, four have ceased and 14 are vulnerable or have already changed; ten of which are in the Bay of Biscay area where recruitment is concentrated. The loss of these recruitment data jeopardizes the scientific advice on the stock status and the ability to measure a recovery (see Ch. 2.1.3).

It is proposed that a workshop be convened to assess and make recommendations for improved fisheries-dependent sampling and scientific surveys in the DCF.

2.11 Conclusion on data and trends

The WGEEL recruitment index shows a continuing decline and is currently at its lowest historical level, less than 1% for the continental North Sea and 5% elsewhere in the distribution area with respect to 1960–1979. Commercial landings have also declined to a low level to less than 4000 t. Aquaculture production has declined to 4000 t in 2010. As a result of entry into force of EMPs restocking starts to increase with about 10 millions of glass eels and 11 millions of yellow eels restocked in 2011.

The Working Group has begun to collect yellow and silver time-series. This work needs to be extended in order to enable an analysis of those series.

The analysis of recruitment-series shows that, because of all sources of variability, it will take some years (maybe decades) to see a change in recruitment-series (if any) as a result of implemented management actions.

With the implementation of the management plans and the decline in the stock, a progressive restriction or collapse of local small-scale fisheries is foreseen. This change will come to the detriment of culture and heritage (e.g. fishing techniques, skills, gastronomy). In some places, there is also an increased risk of illegal fishing. A study is proposed.

Recommendation	For follow up by:
1. From 2008 four glass eel recruitment-series have been stopped in France. This adds up to four series lost elsewhere in Europe. This is only the consequence of changes brought in the catch report compilation. The only data currently available is the EMU and this prevents from doing any analysis at the estuarine level. The recommandation is for the previous details to be reported again in the data collection procedures.	ICES ? DCF, France.
2. A small number (three) of recruitment-series are available in the mediterranean area. Change this.	GFCM DCF
3. Catch and effort data be collected and made available to the working group	DCF
4.1 The 2001 meeting of WGEEL (ICES 2002) recommended the formation of an international commission that could act as a clearing house for handling and coordinating data collection & storage, stock assessment, management and research. Noting the urgent need to plan and coordinate the data collection and tool development for the 2012 post-evaluation; this recommendation is reiterated. 4.2 In particular, it is recommended to organize a (series of) workshop(s) in relation to local eel stock monitoring, with a focus on standardization and coordination, preparing for the 2012 post-evaluation, setting the scene for the 2013 international stock National surveys of eel stocks should now be included in the DCF under the following headings: Recruitment Surveys (Time-Series), internationaly coordinated Silver Eel Escapement Indices, including biomass estimates Yellow Eel Stock Surveys, including collection of biological and eel quality data	ICES, EU, PGCCDBS, PGMED, DCF
5. National data on trend in silver eel and yellow abundance be made available to the working for an analysis next year	WGEEL 2012
6. It is unlikely that we detect a change brought by management actions to the recruitment level and silver eel escapement in the short term. It is of utmost importance to work on anthropogenic mortalities and interim targets.	ICES, EU
7. Establish a project on the Eel Fisheries Resource - a social and economic perspective	EU, EIFAAC

3 Objectives, targets and reference values

Chapter 3 addresses the ongoing issue of a lack of biological reference points by developing a line of reasoning using stand ICEs practices:

b) Develop and test methods to post-evaluate effects of management actions at the stock-wide level (in conjunction with SGIPEE), including quality assurance checking of Eel Management Unit biomass estimates;

f) Respond to specific requests in support of the eel stock recovery Regulation, as necessary;

and has links to:

c) Develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures; test data scenarios at the local level;

e) Provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods.

3.1 The framework for assessment

The EU Eel Regulation sets a long-term general objective (“the protection and sustainable use of the stock of European eel“), while delegating the local management, the implementation of protective measures, the monitoring, and the local post evaluation to its Member States (EU 2007; Dekker, 2009). An escapement objective is set for the biomass of silver eel escaping from each management area, at 40% of the pristine biomass. Eel management plans (EMPs) have been submitted by Member States in 2008/2009 and a post evaluation of EMPs is required every three years, the first in 2012.

CITES regulates the import/export of eels from EU territory and from individual non-EU countries, by means of a requirement for a so-called Non-Detriment Finding NDF. Minimal conditions for an NDF are that the import/export is “not detrimental to the survival of the species” and “that species is maintained throughout its range at a level consistent with its role in the ecosystems in which it occurs”. The CITES checklist for making NDFs indicates that import/export should “not result in unplanned range reduction, or long-term population decline...”, but no quantitative criteria defining a range reduction or long-term population decline are spelled out. Noting that all exploitation leads to some decline in population abundance, the NDF criterion is here understood to mark a situation of severe declines and/or secondary effects such as reduced reproductive capacity. Interpreted this way, these criteria correspond to those applied by ICES, and those agreed upon in the EU Eel Regulation.

Due to the panmixia of the eel (i.e. local silver eel production contributes an unknown fraction to the entire European eel spawning stock, which in turn generates new glass eel recruitment), the efficacy of local protective actions (single EMPs, national export regulation) cannot be post-evaluated without considering the overall efficacy of all protective measures taken throughout the distribution range. This requires an international post-evaluation, as planned by WGEEL. ICES (2010b) considered two different approaches for this. The first is to conduct a central assessment with data from all areas (spatial lumping of data); the second consists of regional stock assessment and the *post hoc* summing up of indicators. The approach of regional stock assessment and *post hoc* summing up of indicators for total stock assessment appears to be more pragmatic than the "central assessment" and to relate more directly to the approach taken in the EU Eel Regulation and CITES procedures. For quality control, however, even a regionalised assessment will require that data are made available.

ICES (2010a Vincennes, 2011 London) derived a framework for *post hoc* summing up of stock indicators, based on four estimates:

a) B_{post}, the biomass of the escapement in the assessment year;

b) B_0, the biomass of the escapement in the pristine state. Alternatively, one could specify B_{lim}, the 40% limit of B_0, as set in the Eel Regulation;

c) B_{best}, the estimated biomass in the assessment year, based on the recently observed recruitment, but assuming no anthropogenic impacts have occurred (neither positive nor negative impacts);

d) $\sum A$, the lifetime anthropogenic mortality rate, or %SPR, the ratio of actual escapement B_{post} to best achievable spawner escapement B_{best}. ICES (2011 London) indicated that estimates of either $\sum A$ or %SPR usually refer to anthropogenic impacts in the most recent year, not to impacts summed over the life history of any individual or cohort in the current stock.

In compiling Eel Management Plans (2008), Member States have provided estimates of pristine biomass and of current anthropogenic impacts, and thus have set **reference points** for their own EMP(s), to which the state of the local stock and efficacy of their actions can be compared.

In the 2010 Report of ICES Study Group on International Post-Evaluation of Eel (SGIPEE), a pragmatic framework to post-evaluate the status of the eel stock and the effect of management measures has been designed and presented, including an overview of potential post-evaluation tests and an adaptation to the eel case of the classical ICES precautionary diagram. In the Precautionary Diagram, annual fishing mortality (averaged over the dominating age groups) is plotted vs. the spawning–stock biomass. In the modified Precautionary diagram proposed by Dekker (2010), lifetime anthropogenic mortality ΣA (or the spawner potential ratio %SPR on a logarithmic scale) is plotted against silver eel escapement (in percentage of B_0). This modified diagram allows for comparisons between EMUs (%-wise SSB; lifetime summation of anthropogenic mortality) and comparisons of the status to limit/target values, while at the same time allowing for the integration of local stock status estimates (by region, EMU or country) into status indicators for larger geographical areas (ultimately: population wide). However, the Modified Precautionary Diagram shown in ICES (2010a,b *Vincennes, Hamburg*) implicitly quantifies a number of management reference points, for which no value had been agreed. Below, we will analyse the ICES framework for setting reference values, and suggest specific values.

3.2 Quantifying specific values for biological reference points

3.2.1 Biological reference points specified in the Eel Regulation

The Eel Regulation sets a limit for the escapement of (maturing) silver eels, at 40% of the natural pristine escapement B_0 (that is: in the absence of any anthropogenic impacts and at historic recruitment). Because current recruitment is far below pre-1980 levels and is assumed to be so due to anthropogenic impacts, return to this limit level is not expected before decades or centuries even if all anthropogenic impacts are removed (FAO EIFAC and ICES 2006, 2007; Åström and Dekker, 2007).

Regulation Article 2.4 specifies the limit as "The objective of each Eel Management Plan shall be to reduce anthropogenic mortalities so as to permit *with high probability* the escapement to the sea of at least 40% of the silver eel biomass relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock. The Eel Management Plan shall be prepared with the purpose of achieving this objective in the long term". This indicates that the *true* escapement should exceed 40%, while uncertainties in the estimate of pristine or current biomass and natural variation under pristine conditions should be compensated for by raising the target escapement level. Hence, the EU Eel Regulation sets a limit reference point, relating to the desirable state rather than to agreed indicator values.

The EU Regulation sets a clear limit for the spawning–stock biomass B_{lim}, as a percentage of B_0. However, no explicit limit on anthropogenic impacts A_{lim} is specified, even though current biomass is (far) below B_0 and B_{lim}. As noted by Åström and Dekker (2007), even a full stop to all anthropogenic impacts will probably not lead to a recovery of the stock to target level. It will take more than one generation for the stock to recover. This implies that the international management objective B_{lim} is unachievable, at least for the coming generation. In addition to the biomass limit, a mortality reference point will also be required.

As an alternative to mortality-based management regimes, a system based on ***escapement quota*** might be considered, in which escapement limits are set based on comparisons between systems and the relation to covariates (latitude, distance, trophic status, etc). However, the number of cases that allow a direct quantification of escapement is extremely limited; that is: in most cases there would be no indicator variable matching the reference point (and definitely no real-time indicator, cf. salmon spawner escapement limit). Hence, escapement quotas are not a realistic basis for managing eel stocks.

3.2.2 Mortality reference point corresponding to the EU Regulation

The Eel Regulation specifies a limit reference point (40% of pristine biomass B_0) for the biomass of the spawning stock. For long-lived species (such as the eel) with a low fecundity (unlike the eel), biological reference points are often formulated in terms of numbers, rather than biomass. Though numbers-based and biomass-based reference points will differ slightly, a mortality-based reference point will be derived here, that results in 40% of the pristine stock *numbers*.

If no density-dependent processes substantially affect the stock abundance in the continental phase, the number of silver eels escaping to the ocean equals[6]:

---

[6] Notation in these equations:

X* parameter X as applied in the silver eel stage. Hence: A* is the anthropogenic mortality (A) in the silver eel stage.

Esc silver eel escapement. the number of silver eels leaving the area towards the ocean.

t time, in years

a age, in years since recruitment to the continent

$$Esc_t^* = N_t^* \times \exp^{-Z_t^*} = N_t \times \exp^{-Z_t^* - S_t} = R_{t-a} \times \exp^{-S_t - \sum_{i=0}^{a} M_{t-a+i,i}} \times \exp^{-Z_t^* - \sum_{i=0}^{a} A_{t-a+i,i}}$$

Without anthropogenic mortality, the last factor ($\exp^{-Z_t^* - \sum_{i=0}^{a} A_{t-a+i,i}}$) vanishes. Hence, the number of silver eels escaping, as a percentage of the number that would have escaped without anthropogenic impacts is

$$\%SPR_t = \exp^{-Z_t^* - \sum_{i=0}^{a} A_{t-a+i,i}} \quad (\times 100\%)$$

This is independent of the number of recruits and the natural mortality (unless density-dependence is significant). If the limit reference point on the *number* of silver eels escaping is set at 40%, it follows that

$$Z_t^* + \sum_{i=0}^{a} A_{t-a+i,i} = -\ln(\%SPR) \leq -\ln(40\%) = 0.92$$

i.e. the sum of all anthropogenic impacts, summed over the entire continental lifespan, should not exceed a fixed value of 0.92.

In cases where density-dependent processes substantially influence continental stock dynamics, no general mortality reference point can be derived. Here, anthropogenic mortality will be compensated for by reduced density-dependent natural mortality; biomass production and silver eel escapement become stable through compensatory survival. A much higher anthropogenic mortality, however, will eventually reduce the production and escapement of silver eels. When and where this occurs, a more elaborate analysis of the density-dependent dynamics will be required, referring directly to escapement levels and %SPR. As a rule of thumb, this more complex analysis will only be required in areas where stock production and/or silver eel escapement has not declined over time. However, density-dependence may still influence young eel stages in areas where they recruit in relatively large numbers (i.e. some west coast estuaries and associated lower river stretches); even if inland densities are too low to influence subsequent eel production dynamics.

For reference points based on biomass rather than on numbers, the relationship between relative spawner escapement %SPR and mortality ∑A is much more complex, but numerical simulation indicates that the relationship comes close to that specified above.

Mortality based indicators and reference points routinely refer to mortality levels assessed in (the most) recent years. ICES (2011 *Sgipee London*) noted that the actual spawner escapement will lag behind, because cohorts contributing to recent spawner escapement have experienced earlier mortality levels before. As a consequence, stock

---

%SPR ratio of spawner per recruit (SPR), the current SPR as a percentage of SPR in the pristine state.

A anthropogenic mortality (fishing F & other anthropogenic mortality H)

M natural mortality.

N number of eels in the stock; N* is the number of silver eels produced (before mortality)

R recruitment

S instantaneous rate of the silvering process, i.e. the silvering process expressed as a rate indicators based on assessed mortalities do not match with those based on measured spawner escapement. The time-lag applies to mortality based indicators as well as to %SPR-based indicators. It will be in line with the conventional ICES procedures and the standard Precautionary Diagram to focus on immediate effects ($\sum A$), ignoring the inherent time-lag in spawner production. This will show the full effect of management measures taken (on the vertical mortality axis) even though the effect on biomass (horizontal) has not yet fully occurred.

3.2.3 Mortality reference values derived from historical time-trends

Åström and Dekker (2007) developed a first whole life cycle model, and argued that, to bring the decadal decline to a hold, a minimal reduction in anthropogenic mortality is required that compensates for the decline in recruitment actually observed. Quantifying the rate of decline in recruitment, and using the estimate of total anthropogenic mortality of Dekker (2000b), they came to a limit for anthropogenic mortality of $\sum A$ =0.48 over the lifetime of the eel. However, there are inconsistencies between the data used (Dekker, 2000b) and the analyses made by Åström and Dekker (2007). The mortality estimates by Dekker (2000b) are crucially based on the assumption of a steady state, while Åström and Dekker (2007) analysed the trend in recruitment, and derived estimates of the <u>lack</u> of a steady state in the population, using the mortality estimates by Dekker (2000b) without correction. Dekker (2010) indicates that the mortality estimates given will probably underestimate the true mortalities and therefore there is some concern about the validity of this estimate.

The Simple Eel Dynamics Model SED (Lambert, 2008) is an adaptation of Åström and Dekker (2007) model applied to French Atlantic coasts. It assumes an eel population with spawners only produced in the French Atlantic, uses a continental lifespan of nine years (instead of 16 years) and an anthropogenic mortality level ΣA of 1.83 instead of 3.24, resulting in a threshold anthropogenic mortality of $\sum A$ =0.73.

FAO EIFAC and ICES (2007) noted that the outcomes of these models depend crucially on assumed parameter values, finding extremes of $\sum A$ = 0.03 or 2.9 (two extremes for north and south Europe).

3.2.4 Reference points used or implicated in previous ICES advice

Since 1998 (ICES 1999 through to ICES 2010), ICES has given advice[7] that the stock has shown a long-term decline and therefore management is not sustainable; that

---

[7] ICES (1999) advised "The eel stock is outside safe biological limits and the current fishery is not sustainable. (…) Actions that would lead to a recovery of the recruitment are needed. The possible actions are 1) restricting the fishery and/or 2) stocking of glass eel."

ICES (2000) recommended “that a recovery plan should be implemented for the eel stock and that the fishing mortality be reduced to the lowest possible level until such a plan is agreed upon and implemented.”

ICES (2001) recommended “that an international rebuilding plan is developed for the whole stock. Such a rebuilding plan should include measures to reduce exploitation of all life stages and restore habitats. Until such a plan is agreed upon and implemented, ICES recommends that exploitation be reduced to the lowest possible level.”

ICES (2002) recommended “that an international recovery plan be developed for the whole stock on an urgent basis and that exploitation and other anthropogenic mortalities be reduced to as close to zero as possible, until such a plan is agreed upon and implemented.”

fishing and other anthropogenic impacts should be reduced; that a recovery plan should be compiled and implemented; that preliminary reductions in mortality to as close to zero as possible are required until such a plan is implemented, respectively until stock recovery has been achieved.

ICES (2002a) discussed a potential reference value for spawning–stock biomass: "a precautionary reference point for eel must be stricter than universal provisional reference targets. Exploitation, which provides 30% of the virgin (F=0) spawning–stock biomass is generally considered to be such a reasonable provisional reference target. However, for eel a preliminary value could be 50%." That is: ICES advised to set B_{lim} above the universal value of 30%, at a value of 50% of B_0. ICES (2007) added: "an intermediate rebuilding target could be the pre-1970s average SSB level which has generated normal recruitments in the past."

The Eel Regulation (Council Regulation 1100/2007) sets a limit for the escapement of (maturing) silver eels, at 40% of the natural escapement (that is: in the absence of any anthropogenic impacts and at historic recruitment). That is: EU decided to set B_{lim} at 40% of B_0, in-between the universal level and the level advised. ICES (2008) noted that its 2002 advice was "higher than the escapement level of at least 40% set by the EU regulation."

ICES has not advised on specific values for mortality-based reference points, but the wordings "the lowest possible level" and "as close to zero as possible" imply that F_{lim} and therefore A_{lim} should be set close to zero. Over the years, the implied time frame for this advice has changed from "until a plan is agreed upon and implemented", to "until stock recovery is achieved" and "until there is clear evidence that the stock is increasing". The first and third phrases are more interim precautionary mortality advice than clear reference point related to any biomass.

3.3 ICES approach for fisheries advice

ICES (2009, 2010) provides advice on fish stock management; in the introduction, the general approach is explained. This section and the next one copy that framework and consider how to apply it to the eel case.

> ICES. 2009. Report of ICES Advisory Committee, 2009. ICES Advice, 2009. Books 1–11. 1,420 pp.

> ICES. 2010. Report of ICES Advisory Committee, 2010. ICES Advice, 2010. Books 1–11. 1928 pp.

ICES (2006) advice read: "An important element of such a recovery plan should be a ban on all exploitation (including eel harvesting for aquaculture) until clear signs of recovery can be established. Other anthropogenic impacts should be reduced to a level as close to zero as possible."

ICES (2008a) concluded "There is no change in the perception of the status of the stock. The advice remains that urgent actions are needed to avoid further depletion of the eel stock and to bring about a recovery."

ICES (2009) reiterated its previous advice that "all anthropogenic impacts on production and escapement of eels should be reduced to as close to zero as possible until stock recovery is achieved".

ICES (2010c) reiterated its previous advice that "all anthropogenic mortality (e.g. recreational and commercial fishing, barriers to passage, habitat alteration, pollution, etc.) affecting production and escapement of eels should be reduced to as close to zero as possible until there is clear evidence that the stock is increasing."

ICES provides fisheries advice that is consistent with the broad international policy norms of the Maximum Sustainable Yield approach, the precautionary approach, and an ecosystem approach while at the same time responding to the specific needs of the management bodies requesting advice.

When information for determining reference points is poor or absent, ICES (2009) advises that provisional reference points are set.

For long-lived stocks with population size estimates, ICES bases its advice on attaining an anthropogenic mortality rate at or below the mortality that corresponds to long-term biomass targets. However, $B_{MSY\text{-}trigger}$ is a biomass level triggering a more cautious response. Below $B_{MSY\text{-}trigger}$, the anthropogenic mortality advised is reduced, to reinforce the tendency for stocks to rebuild. Below $B_{MSY\text{-}trigger}$, ICES applies a proportional reduction in mortality reference values (i.e. a linear relation between the mortality rate advised and biomass). The determination of an appropriate value for $B_{MSY\text{-}trigger}$ requires contemporary data in the normal range of fluctuations around the long-term biomass target. As an initial option, ICES sets $B_{MSY\text{-}trigger}$ at B_{pa} (unless there is a sound basis for using a different value). For the eel, a decadal decline has been observed. Contemporary estimates of escapement biomass have only recently become available, but this relates only to the stock in its current, depleted state. Consequently, there is no basis to advice on an appropriate margin between B_{lim} and B_{pa}.

3.4 Precautionary advice, uncertainty in estimates

The EU Eel Regulation sets a lower limit to the escapement of silver eel, at 40% of the pristine escapement, and leaves it up to its Member States to derive actual quantitative estimates. As indicated above, the generic 40% value is a limit reference point, to which a margin should be added accounting for natural fluctuations in the pristine level, and statistical uncertainties in the estimates of pristine and current escapement levels. Because of the structure of the EU Regulation (delegating quantification to the Member States) there are no generic precautionary reference points B_{pa} and A_{pa}. The level of uncertainty in the estimates may vary from Member State to Member State. This leaves ICES with two options: either to abstain from providing advice; or to provide advice based on B_{lim} and A_{lim} rather than B_{pa} and A_{pa}, pointing at the required uncertainty margins throughout the advice. The second option seems to be the more productive alternative, and this option will be followed here.

ICES (2002a) discussed a potential reference value for spawning–stock biomass: "a precautionary reference point for eel must be stricter than universal provisional reference targets. Exploitation, which provides 30% of the virgin (F=0) spawning–stock biomass is generally considered to be such a reasonable provisional reference target. However, for eel a preliminary value could be 50%".

The Eel Regulation (Council Regulation 1100/2007) set a limit for the escapement of (maturing) silver eels (B_{lim}), at 40% of the natural escapement (that is: in the absence of any anthropogenic impacts and at historic recruitment), in-between the universal level and the 50% level advised by ICES (2008).

If the evaluation of a management plan indicates that a stock has a low probability (e.g. less than 5%) of being below B_{lim} in the medium term, ICES considers the plan in accordance with the precautionary approach even when the stock is below the precautionary biomass level B_{pa} or above the precautionary mortality level A_{pa}. Noting the current state of the eel stock, full recovery is not expected within a few generations, even if all anthropogenic mortality would be reduced to zero (Åström and Dekker, 2007). Hence, reductions in anthropogenic mortality below the ultimately sustainable level of A_{lim} and therefore A_{pa} are required to reinforce the tendency for the stock to rebuild.

3.5 Unquantified effects

In the Modified Precautionary Diagram, only quantitative effects are represented. Pollution, for instance, is only included if it has a quantified effect on survival during the continental stage or on growth rates (but little is known of either impact; see Ch. 5). In turn this means that only management measures which act on such quantitative parameters can be evaluated. Oceanic factors are also not directly included (only via potential effects on recruitment, which is not explicitly shown in the diagram). Therefore, the diagram only shows quantified effects of management measures during the continental phase. This selective presentation, however, matches with the selective obligations in the Eel Regulation, mentioning but not enforcing currently unquantifiable management actions.

3.6 Recommended reference values

Summarizing the above:

ICES (2002a) considered a precautionary reference limit, corresponding to B_{lim} is 30% of B_0. Because of the many uncertainties in eel biology and estimation, a more cautious level of 50% was advised.

The EU Eel Regulation set a limit, corresponding to B_{lim} = 40% of B_0.

According to the EU Eel Regulation, quantification and implementation of these reference points is up to EU Member States. Because uncertainties may vary from Member State to Member State, no universal values for the precautionary reference points can be provided. Hence, it is recommended that ICES abstains from advising precautious reference points, cautioning for the required extra margin on all reference points instead.

The biomass reference point of B_{lim} = 40% of B_0 corresponds to a lifetime mortality limit of $\Sigma A_{lim} = 0.92$, unless strong density-dependence applies. In the latter case, a more complex assessment will be required, and a limit of $\%SPR_{lim} = 40\%$ can be applied.

As an initial option, it is recommended to set $B_{MSY\text{-}trigger}$ at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ correspondingly. Allowing for natural variation in B_0 and for uncertainty in the estimates of status indicators and reference points, the resulting reference points (B_{lim}, $B_{MSY\text{-}trigger}$ and A_{lim}) should be considered as somewhat optimistic, incautious.

In accordance with the structure of the EU Eel Regulation, the reduction on A_{lim} below $B_{MSY\text{-}trigger}$ may be applied on an area by area basis.

These reference values are summarized in the Modified Precautionary Diagram (see ICES 2010b) in Figures 3.1 and 3.2, the available biomass data with respect to 2008 EMPs are presented.

Note that the ICES approach of reducing A_{lim} in proportion to biomass below $B_{MSY\text{-}trigger}$ does not take into account the logarithmic nature of mortality rates. Hence, the proportional mortality reduction below $B_{MSY\text{-}trigger}$ shows up as a curved relation in this diagram. Working out four examples:

- At $B_{current} \geq 40\%$ of B_0, the limit mortality $A_{lim} = 0.92$ applies, corresponding to an escapement of $\%SPR = exp^{-0.92} = 40\%$, i.e. a minimal escapement of 40% of the currently best achievable escapement B_{best} is taken as a limit on mortality.
- At $B_{current} = 20\%$ of B_0, the limit mortality is halved, $A_{lim} = 0.92 \times \frac{20\%}{40\%} = 0.46$, corresponding to an escapement of $\%SPR = exp^{-0.46} = 63\%$, i.e. a minimal escapement of 63% of the currently best achievable escapement B_{best} is taken as a limit on mortality.
- At $B_{current} = 10\%$ of B_0, the limit mortality becomes $A_{lim} = 0.92 \times \frac{10\%}{40\%} = 0.23$, corresponding to an escapement of $\%SPR = exp^{-0.23} = 79\%$, i.e. a minimal escapement of 79% of the currently best achievable escapement B_{best} is taken as a limit on mortality.
- At $B_{current} = 1\%$ of B_0, the limit mortality becomes $A_{lim} = 0.92 \times \frac{1\%}{40\%} = 0.023$, corresponding to an escapement of $\%SPR = exp^{-0.023} = 98\%$, i.e. a minimal escapement of 98% of the currently best achievable escapement B_{best} is taken as a limit on mortality.

This diagram (Figure 3.2) presents the status of the stock (horizontal, low vs. high spawning–stock biomass determining whether the stock has achieved full reproductive potential) and the impact of anthropogenic mortality (vertical, low vs. high anthropogenic mortality determining whether the mortality, including fisheries exploitation, is sustainable or not).

Recommendation	For follow up by:
The EU Eel Regulation set a limit, corresponding to $B_{lim} = 40\%$ of B_0. According to the EU Eel Regulation, quantification and implementation of these reference points is up to EU Member States. Because uncertainties may vary from Member State to Member State, no universal values for the precautionary reference points can be provided. Hence, it is recommended that ICES abstains from advising precautious reference points, cautioning for the required extra margin on all reference points instead.	ACOM
As an initial option, it is recommended to set $B_{MSY\text{-}trigger}$ at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ correspondingly.	ACOM
The biomass reference point of $B_{lim} = 40\%$ of B_0 corresponds to a lifetime mortality limit of $\Sigma A_{lim} = 0.92$, unless strong density-dependence applies. In the latter case, a more complex assessment will be required, and a limit of $\%SPR_{lim} = 40\%$ can be applied.	ACOM

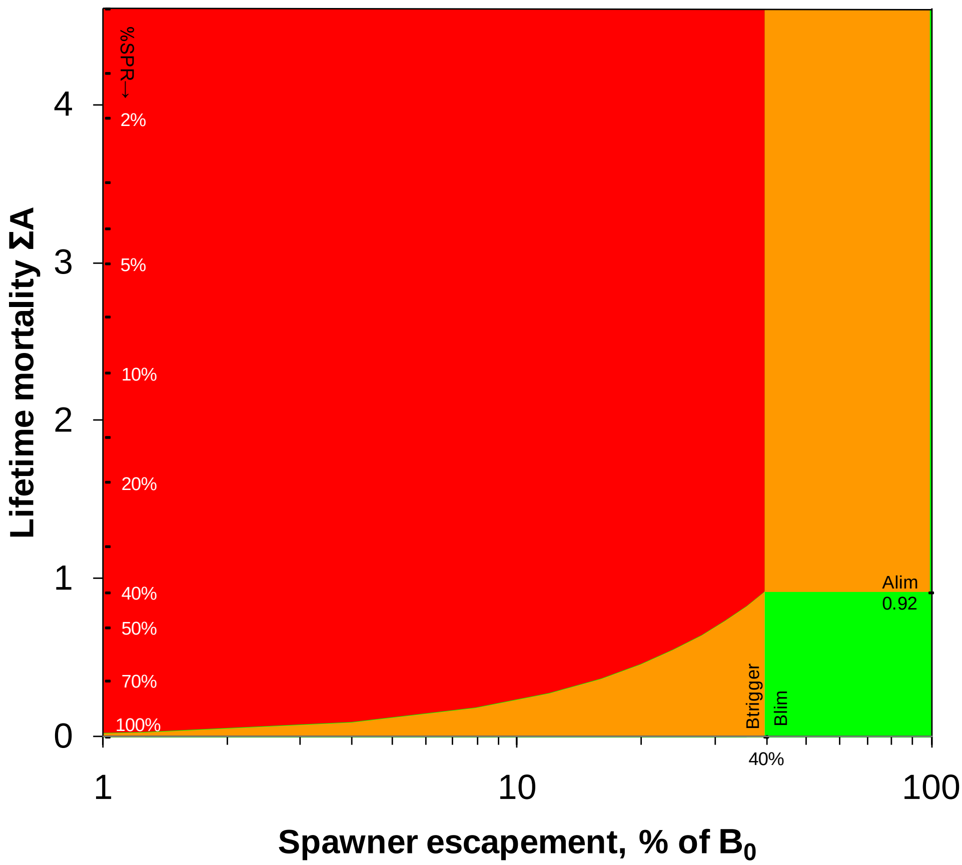


Figure 3-1. Modified Precautionary Diagram, summarizing the suggested reference points. Note that statistical uncertainty and natural variation could not be taken into account, and hence the suggested reference points are conservative: in practice, a higher biomass and a lower mortality will be required, creating a safety margin for uncertainty and natural variation. %SPR = spawner potential ratio, a measure for the survival to silver eel relative to pristine conditions.

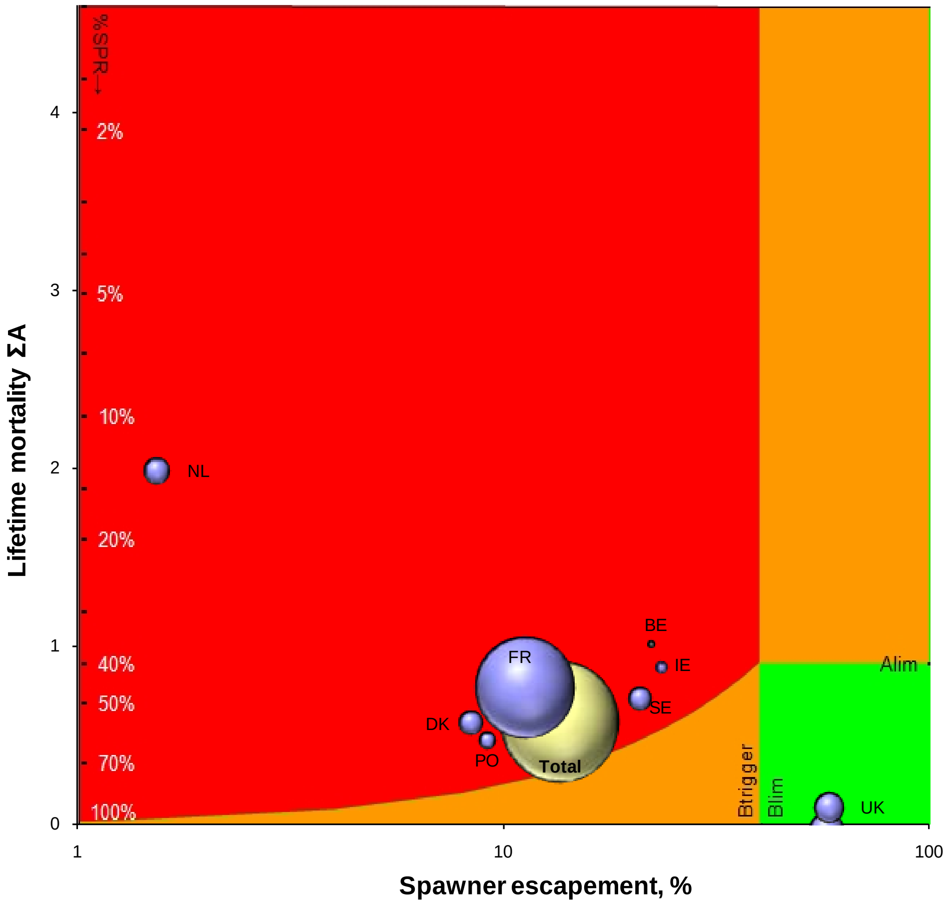


Figure 3-2. Modified Precautionary Diagram, presenting the status of the stock and the anthropogenic impacts, per country as presented in the Eel Management Plans in 2008. For each, the size of the bubble is proportional to B_{best}, the best achievable spawner escapement given the recent recruitment, while the center of the bubble gives the stock status relative to the targets/limits. The horizontal axis represents the status of the stock in relation to pristine conditions, while the vertical axis represents the impact made by anthropogenic mortality. Data from national Eel Management Plans, supplemented by Country Reports; not all countries supplied estimates. Reference points as derived in the text.

4 Quantitative assessment of the status of local eel populations

Chapter 4 addresses the following Terms of Reference:

a) assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans;

c) develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures (in conjunction with SGAESAW 2);

d) provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and the review and development of recommendations on inclusion of data quality issues, including the impact of the implementation of the eel recovery plan on time-series data, on stock assessment methods.

and has links to:

b) develop methods to post-evaluate effects of management actions at the stock-wide level (in conjunction with SGIPEE);

f) respond to specific requests in support of the eel stock recovery Regulation, as necessary.

4.1 Introduction

In order to assess the status of (local) eel stocks, good quantitative information and an understanding of all anthropogenic mortalities occurring over an eel's life time is necessary. This chapter will start with describing the process for estimating lifetime anthropogenic mortality rates and will provide an overview of quantified non-fishing mortalities such as hydropower and pumping stations. In addition to anthropogenic mortality and following on from the information on size and age of silver eel discussed in 2010 (ICES 2010a), a start is made in this chapter to understand what determines sex-ratios in eel.

4.2 Estimating lifetime anthropogenic mortality rates

4.2.1 Introduction

In addition to the eel biomass metrics (the three B's), international stock assessment requires an estimate of the total anthropogenic mortality. As there are several different types of potential anthropogenic mortality (e.g. fishing, turbines and pumps, pollution, barriers and obstacles, etc.), this total mortality is expressed as the sum of those for all types, ΣA.

There are several issues that must be considered in developing an approach to express the total losses due to a variety of anthropogenic impacts distributed across a river basin, and further, to express those for a complete River Basin District or Eel Management Unit. These issues are summarized as follows:

- That losses occur across the lifetime of the eel;
- That losses in each life stage have an effect on subsequent life stages;
- That losses at one location within a basin affect eel in other locations because of the migratory nature of the eel;

- That restocking confounds mortality assessments because its relative contribution to production depends on the local circumstances;
- Summing the losses across river basins.

The following text expands on some of these issues, to support scientists and managers in assessing anthropogenic mortalities and deriving an assessment indicator for this metric, but the issues are complex and this is an area that requires careful consideration.

For assessment purposes, this mortality rate sums all mortality between the point of recruitment and the point at which silver eels leave the assessment area, so includes the anthropogenic mortalities affecting the eel from glass eel recruitment to silvering and emigration (cf the percentage spawner per recruit, %SPR). This is commonly referred to as the cumulative or 'lifetime' anthropogenic mortality rate for eel, though this can be misleading as it does not include mortalities (assumed natural) between egg and recruitment, or between silver eel leaving continental waters (management boundary) and spawning. Conceptually, the estimate of lifetime anthropogenic mortality for assessment purposes ought to include these prerecruit and post-escapement mortalities, but these are two parts of the life cycle for which our knowledge is especially limited.

There are two general approaches to deriving this 'lifetime' rate:

1) The longitudinal approach: an estimate for a specific cohort of eels, such as a year class, combining the mortalities experienced by this cohort at different compartments of its life from recruits to emigrating silver eels. These compartments could be fixed periods of time, e.g. years, or life stages, e.g. elvers, undifferentiated yellow eels, differentiated yellow eels, silver eels.

2) The cross sectional approach: an estimate at a fixed point in time, such as in year 2011, derived from combining the mortalities affecting each class of eel present in the population at that time; the class can be age cohorts or life stages (as 1. above).

Approach 1 is the stronger approach because it allows for variations in the population state over time affecting that specific cohort, for example because of variations in recruitment affecting local densities and hence density-dependent factors, or changes in mortality impacts over time such changes in management. But this approach requires a long-term investment in research and data collection because it requires knowledge of the past mortality rates affecting the cohort over a time period equivalent to an eel generation (or at least the continental growth period of the life cycle). Conversely, the rate is specific to a particular cohort so studies must follow the life of several cohorts to track temporal trends in anthropogenic mortalities.

Approach 2 is 'quicker' to calculate because the rates for each age or stage can be estimated in the same year. But, in combining these rates, it is assumed that they do not change over time, i.e. for a continental 'life' generation period of ten years, the mortality rate for age 1 eels is the same throughout that ten year period. Thus, the main drawback of this approach is that it reflects past management regimes and does not provide feedback on measures taken recently. This assumption of the 'steady state' is particularly weak (unsafe) during periods when recruitment is changing significantly from year to year and therefore affecting local densities, and/or when changing management actions, such as fishery closures or turbine screens, cause significant changes in anthropogenic rates.

In practice, studies are likely to adopt both approaches, using approach 2 in the first instance but repeating the study every year so as to construct a longitudinal study of several cohorts.

Where these mortalities derived in 1 and 2 were standardized and independent of each other, it would be a relatively simple step to sum them all to derive the total anthropogenic mortality. However, there a number of reasons why this is not immediately possible for a complex management unit such as a river basin. Figure 4.1 presents a virtual river network with a complex distribution of anthropogenic impacts (yellow and silver eel fisheries, turbines) which we will use to illustrating these reasons.

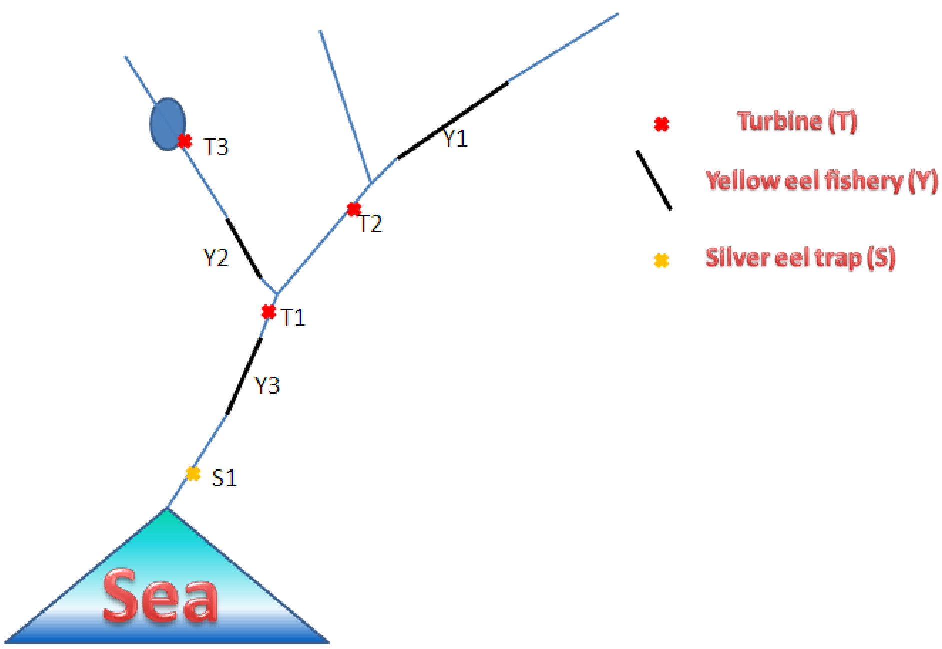


Figure 4-1. Schematic of river network showing locations of example anthropogenic mortalities: turbines, and yellow and silver eel fisheries. The turbines and silver eel trap are discrete locations, and impact eels moving downstream past these locations: the "migrants". The yellow eel fisheries exploit a local area, represented by the black sections of river, but have little impact on eels in the other areas: the "residents".

First, there is a spatial element to consider because anthropogenic mortalities are usually associated with particular locations or areas, and parts of the eel population move into and out of such areas during their continental life and therefore become more or less susceptible to the mortality. The spatial nature of the anthropogenic impacts can be categorized into 2 groups (though there are subtleties and similarities between some examples of the 2 groups):

1) Impacts on resident eels in the surrounding area, where the area may extend both upstream and downstream. A yellow eel fishery is an example of an impact on residents.
2) Impacts on migrating (*) eels are location specific, typically those passing in one or other direction, but also possibly in both directions. A turbine is an example of an impact on movers.

(*) – some dispersals are not strictly migrations but we use the term here to simplify the classification.

The differences between these two mortality groups determine the manner in which the mortality from multiple sites around a river network can be combined to estimate a total mortality for a particular type.

Those mortalities affecting resident eels generally have little or no effect on eel outside the area and therefore they can be treated as independent values and summed accordingly.

In contrast, the mortalities affecting migrants may have a sequential effect, are therefore not independent, and the (log) rates must be combined in an additive manner.

For example, 'T1' the turbine most downstream in Figure 4.1 is identical in design to the other two turbines further upstream, so has the same rate of impact on the silver eels migrating past this location, but it is only those eels that survive passage at T2 and T3 plus the silver eels produced in the river area between the three turbines that are impacted by T1.

Second, with the exception of some pollution events, the mortality rarely affects all the eels in the same way; there is an element of 'selectivity' to the mortality. In fisheries, this selection is typically related to the design of fishing gears and acts on fish length such that capture efficiency (cf. mortality) declines with length below a threshold (occasionally efficiency declines with increasng length also). This selectivity may be truncated by a minimum or maximum landing size. Turbines (power generating) and pumps (water transfer) are also selective on fish lengths, though these can be more difficult to quantify compared to fisheries because some eels may survive passage through the turbine or pump but die later because of injuries caused during the passage.

Third, stock structure generally varies across the river network, typically with densities declining in the upstream direction while boundaries of length distribution and the average length increase upstream (Ibbotson *et al*., 2002). This variation in stock structure across the network interacts with the selective nature of the impact to the result that a mortality event has a different consequence on groups of eel at different locations throughout the network. For example, while the average mortality associated with passage through a turbine is 28% (Table 4.2)), it is incorrect to infer the same loss in number of eels passing two turbines when one is 5 km from the sea and the other is 50 km from the sea.

Finally, we must account for the fact that although we wish to express the total anthropogenic mortality at the time when silver eels emigrate from the management unit (or ideally at spawning), most of the mortalities impact the earlier stages (glass, elver, yellow)/younger eels and therefore are not equivalent to losses of silver eels. For example, a catch of 100 yellow eels to a fishery does not equal a loss of 100 silver eels, because some of these yellow eels would have died for natural reasons. The model of natural mortality presented by Bevacqua *et al*. (2011) provides a tool to derive silver eel equivalents when combined with knowledge of the typical growth rates (Daverat *et al*., in press) and length of silver eels in the management unit. However, the approach is complex because we must take account of the effects of all the mortality events that may occur between the event in question and the point at which each eel would become a silver eel!

To bring all these points together in terms of mortality at each location (exploitation and selectivity) requires a very complex series of equations and we do not attempt to develop these here.

4.2.2 Summing mortalities using eel quantity instead of rate

The stock assessment metrics are biomass (EC 1100/2007), and the impact of fishing (the catch) is often expressed as a weight. However, analysis of anthropogenic mortality in units of biomass obscures the variations in stock structure around the management unit, and the selectivity of mortality types. Therefore, it is more practical to work with eel numbers and length distributions. The approach relies on the ability to estimate the standing stock of eel in terms of their number (density) and lengths distributed around the management unit in relation to the distribution of impacts (see Figure 4.2).

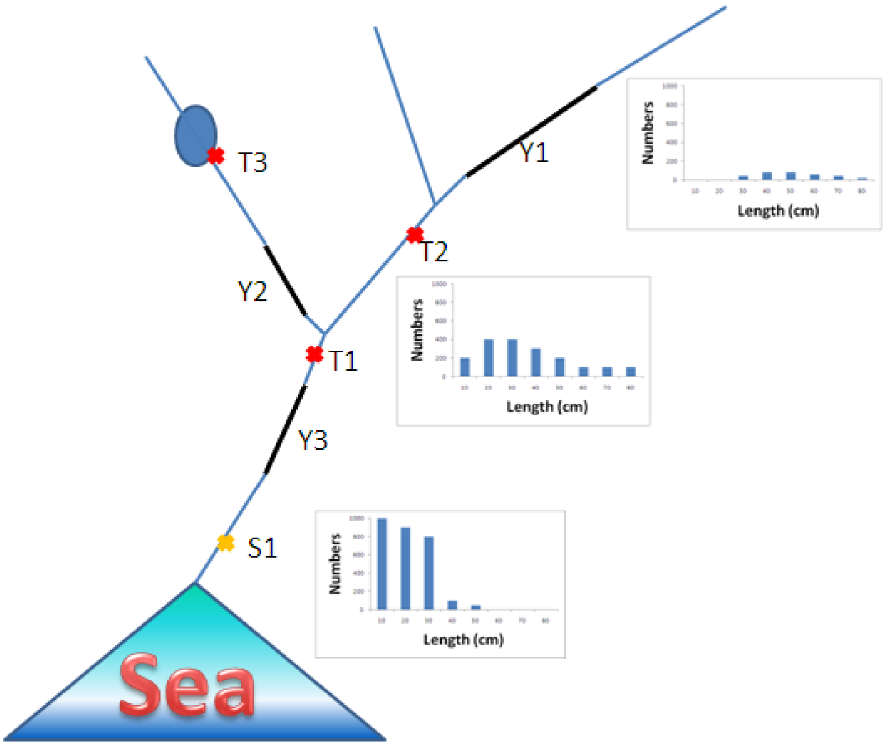


Figure 4-2. Schematic of river network showing locations of example anthropogenic mortalities (turbines, and yellow and silver eel fisheries; see Figure 4.1 for key) and a stereotypical distribution of numbers and length frequencies of yellow eels throughout the network.

Where the impacts of the anthropogenic mortality 'events' are known or estimated in terms of the numbers of eel and their length distribution, these numbers-at-length can be subtracted from local standing stock estimates for those mortality events that impact on residents, in order to derive the mortality 'rate'. The problem remains, however, to extrapolate from resident yellow eels, to silver eels that are impacted when encountering anthropogenic mortality events during their emigration (e.g. turbines, silver eel fisheries, acute pollution events).

Describing the standing stock of eels across the management unit is typically based on extrapolating from survey data collected from a few, relatively small sites and assuming these are representative of much larger areas and in some cases of very different environments (cf shallow rivers vs. lakes). At least two models have been developed taking this approach, the Environment Agency (England & Wales) Fisheries Classification Scheme II (FCS II) and the Eel Density Analysis (EDA: Beaulaton *et al*.). Both models develop relationships between eel densities from surveys and habitat descriptors, and then apply these relationships to derive eel standing stock throughout the remainder of the study area. However, neither approach addresses length distributions of eel, nor eel in still waters and they also rely on a generic silver rate to convert from yellow eel densities to silver eel numbers.

Second, it may be difficult to measure the mortality due to some impacts in terms of the numbers and length distribution of eels. Practically, this is possible for fisheries and pollution events by counting and measuring the catch or kill, although these may be very labour intensive. It is far less practical to do this for the impacts of turbines and pumps, where the killed eels are not routinely sampled.

In conclusion, however, a combined approach expressing losses due to anthropogenic mortality factors as rates for some mortality causes, and in numbers of eel for other causes is almost always going to be required. For example, consider an eel fishery located upstream from a turbine. The loss to the fishery can be expressed as numbers of eel, whereas the loss to the turbine will more likely be expressed as a %. The closure of the fishery will reduce the mortality due to fishing, but increase the numbers of eels impacted by the turbine. Thus, although the number of eel dying in the turbines has increased, the mortality rate is still the same when expressed as a percentage.

4.2.3 Spatial hierarchy to pooling mortalities across a river basin and between basins

There are two approaches to building the hierarchy of anthropogenic mortalities across a river basin.

One option is to estimate losses for each mortality type across the whole river basin and then combine these for the different types. This approach facilitates comparisons between the relative impact (importance) of different mortality types on silver eel escapement, and in prioritizing management actions; management actions are probably similar across a mortality type, but may be different between types.

The other option is to combine the losses from all anthropogenic mortality types within a zone (for example a tributary), and then combine these zone estimates to produce the total for the river basin. This is particularly useful in considering the sequential effects of different mortality types within the zone, and evaluating management actions. Taking the example of the fishery upstream of the turbine (above), the relative mortality of each must be compared directly in considering the most appropriate management action, and the fact that the benefit of closing the fishery is to some extent lost in the turbine must also be taken into consideration. This zonal approach may also be useful where the river basin extends across management boundaries.

The total loss across a River Basin District must be estimated in terms of eel quantity because rates for different river basins may not be equivalent.

4.2.4 Restocking

Stocking can be considered throughout the above texts, if treated as a positive change to the stock as compared to a loss due to a mortality. However, it potentially confounds the derivation of assessment indicators when treated as a positive 'rate' because the relative (positive) impact depends on local circumstances. Therefore, stocking should be considered in terms of quantity of eel and not percentages.

4.3 Sex ratios

Eel sex is not genetically determined (Tesch, 2003) and sex ratio in adult eel can vary to a great extent in eel stocks, in time as well as in space (Parsons *et al.*, 1977; Rosell *et al.*, 2005). There is some evidence that the proportion of females in migrating silver eel may be increasing during the last decades (e.g. Poole *et al.*, 1990, pers. comm.) and that male percentage is higher in the lowest part of catchments (i.e. estuaries and lagoons) (Ibbotson *et al.*, 2002).

To understand the mechanisms behind this variability of sex ratio is crucial when developing models to predict consequences on spawning–stock biomass and abundance of management actions.

Some authors suggest that lower densities would lead to higher fractions of females, which attain bigger size than males (Parsons *et al.*, 1977; Poole *et al.*, 1990; Rosell *et al.*, 2005; Svärdson, 1976). Density-dependent sex ratio of the eel has been suggested to be an adaptive strategy to achieve maximum fitness. Males which exhibit a time-minimizing growth strategy by maturing as soon as possible would predominate at high densities; while at low density levels females, which postpone maturation with a size-maximizing growth strategy to attain higher fecundity, would be favoured (Helfman *et al.*, 1987; Larsson *et al.*, 1990; Vøllestad, 1992). High competition for food might make it difficult for a female to both produce a sufficient number of eggs and to store enough energy to successfully migrate back to the Sargasso Sea.

4.3.1 Assessing sex ratio

4.3.1.1 Main issues in assessing sex ratio

Due to the different age at silvering (e.g. 10 and 20 years old respectively for males and females), males and females experience different cumulative mortality rates during the different lifespan in continental waters. Particularly, female eels are likely to accumulate greater mortality as they spend more time in the feeding/growing yellow phase and their bigger size make them more easily catchable through size selectivity fishing gear. Hence, sex ratio at differentiation of a cohort *SRd* can significantly differ from sex ratio observed in migrating silver eel stock *SRs*.

Moreover, observed *SRs* in year *t* are also influenced by trends in local recruitment. In fact, the abundance of silver eel females at time *t* relies on recruitment at year *t*-20 while abundance of males relies on recruitment at year *t*-10.

To analyse possible dependence of sex ratio upon density, one should understand which density is at play. *SRd* of a cohort might indeed be influenced by:

- cohort abundance/biomass at recruitment;
- cohort abundance/biomass at time of differentiation (*i.e.* around 25–35 cm);
- eel stock abundance/biomass at recruitment;
- eel stock abundance/biomass at time of differentiation (*i.e.* around 25–35 cm);
- fish community abundance/biomass at recruitment;

- fish community abundance/biomass at time of differentiation (*i.e.* around 25–35 cm).

Furthermore, as density is likely to affect food availability, its consequences should depend also on habitat productivity (e.g. a density of 5 kg/ha can be a very low density in high productive Mediterranean lagoons but a high density in oligotrophic Scandinavian streams).

For all these reasons, standardized methods are essential to compare sex ratios between stocks and obtain reliable data to understand mechanisms responsible for observed patterns.

4.3.1.2 Required data

In order to study if density plays a decisive role in determining *SRd* and its variability of time, it would be necessary to know, for different stocks and years:

- sex of a sample (~100) of young yellow eel (25–35 cm);
- an estimate the age, and therefore growth rates, of the eels in the sample;
- an estimate of eel density at the site where the sample is collected;
- estimates of density at other locations downstream of the sample site, as a proxy for densities experienced by the sampled eels prior to them becoming male or female.

These data should be collected in the lower reaches of a catchment. They would permit an assessment of the actual fraction of individuals differentiating as males/females in a given cohort and then look for possible relationships with density levels experienced by individuals during the early phases of their life cycle. The above data could be collected as part of a WFD or other eel sampling programme at one or several fixed stations.

4.3.1.3 Available data

In absence of dedicated sampling schemes, analyses on sex ratio can be carried out on data regarding time variability of silver eel production explicitly considering males and females. For instance, data from the Burrishoole catchment (western Ireland), show a decline in silver eel abundance starting from 1995 (Figure 4.3a), possibly resulting from the recruitment drop of the 1980s, and a contemporary increase in the fraction of female (Figure 4.3b). Although such data might suggest an effect of density over sex ratio of migrating eels (Figure 4.3c), no conclusion can be drawn about the effect of density on determining sex ratio because i) we are observing sex ratio in eels belonging to different cohorts and ii) this value of SRs is upward skewed in case of declining recruitment because silver females would belong to older and more abundant cohort than silver males.

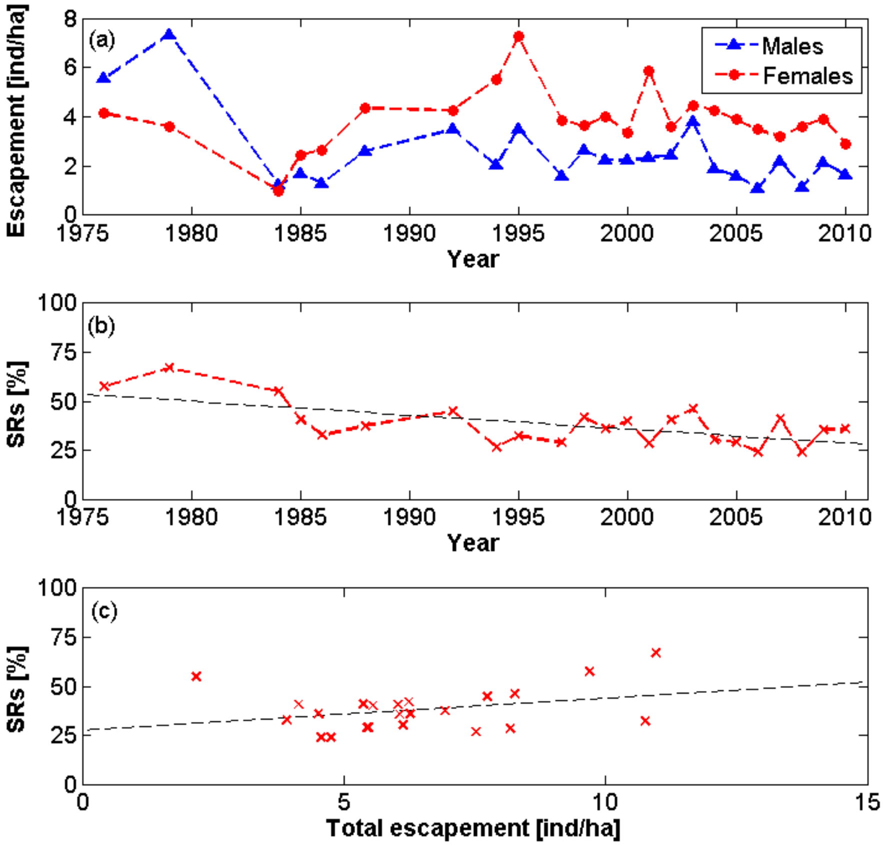


Figure 4-3. Burrishoole catchment: a) Density of silver males and females; b) declining trend (--) of the fraction of males in silver eels and data (red crosses); c) relationship (--) between fraction of males (red crosses) in silver eels and overall abundance of silver eels.

4.4 Non-fishery mortality factors

4.4.1 Hydropower

Mortality rates of downstream migrating eel when passing a hydropower station depend on 1) the proportion of eel moving into the power station intake, 2) the mortality rate of those moving into the power station (turbine mortality, impingement on the trash rack, etc.), and 3) the mortality rate of those using alternative routes (bypass channels, old river bed, etc.). Mortality estimates of downstream migrating eels from hydropower are given in Table 4.1 and summarized in Table 4.2. The table summarizes field studies from several eel species (*A. anguilla, A. rostrata, A. dieffenbachii* and *A. australis*).

Three types of estimations are given: 1) the proportion of eels that went through the turbine vs. the ones that crossed the power station via alternative routes; 2) the mortality of eels passing through the turbines; 3) the total mortality of eels when passing a power station (considering proportions passing through turbines, the mortality of these and the mortality of those using alternative passages).

The most comprehensive estimation comes from a study (Gomes and Larinier, 2008) that developed mortality predictive equations based on body length of eels, turbine diameter, nominal discharge and blade velocity for Kaplan turbines. According to this model based on 71 field studies, damage rate increases with fish length and is generally higher on small turbines with high rotation speeds than on slow, large diameter turbines. Damage is also lower at full opening compared to reduced opening (Gomes and Larinier, 2008).

Mortality is lowered whenever there is a bypass system although the design, location and current speed will determine the efficiency of the bypass. Migrating silver eels are able to swim upstream to look for an alternative passage, if the current in front of the power station intake is not too fast.

Eel mortality due to hydropower stations is variable (Figure 4.4) and highly site-specific. Accurate estimates require on-site studies. If there are no on-site mortality estimates, it is important to select mortality numbers according to the power station characteristics.

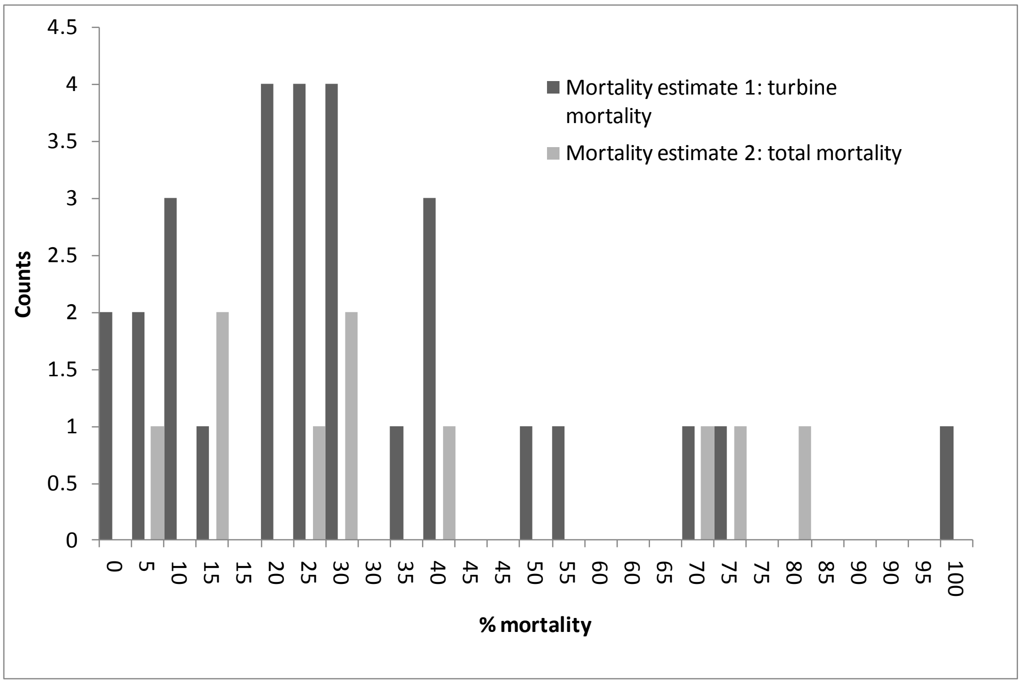


Figure 4-4. Distribution of mortality estimates based on Table 4.1.

Table 4-1. Mortality estimates due to hydropower for several eel species. Blanks in columns 3–6 mean that the type of information "unknown". The reader is referred to the actual publications to obtain more detailed information.

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
1			76	100	76	Kaplan	Canada		>75 cm	*A. rostrata*		Carr and Whoriskey (2008)
2				24		propeller	Canada	Beauharnois Generating station (St Lawrence River)	88–89	*A. rostrata*		Desrochers, 1995
3				16		Francis	Canada	Beauharnois Generating station (St Lawrence River)	88–89	*A. rostrata*		Desrochers, 1995
4					40	Fixed blade propeller and francis	Canada			*A. rostrata*	total estimate for passage of two power stations	Verreault and Dumont, 2002
5	Old, big (6.6 m), and slow rotating turbine	NA	100% (eels put directly into the turbine)	0			Estonia-Russia		Range= 73–97	*A. anguilla*		Järvalt *et al.*, 2010

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
6	Microplant	y	49	0–2.8	14	Francis	France		Mean; Range= 57–93	*A. anguilla*	Results of a 3 year study	Durif *et al*. (2003); Goss *et al*. (2005)
7	Miniplant/ run-of-river; head= 9.6 m		41			Kaplan	France		Range= 50–1100	*A. anguilla*	3 year study	Travade *et al*. (2010)
8				27		Kaplan	Germany			*A. anguilla*		Holzner, 1999, reference in Adam *et al*., 2005
9				50		Kaplan	Germany		Majority : 50–75 cm.	*A. anguilla*		Berg (1986)
10				20		Kaplan.	Germany		Mean = 55; Range= 40–70	*A. anguilla*		von Raben (1964)
11				38			Germany	Neackarzimmern		*A. anguilla*		referred to in Bruijs and Durif, 2009
12				24			Germany	Obernau		*A. anguilla*		referred to in Bruijs and Durif, 2009

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
13				23			Germany	Dettelback??		*A. anguilla*		Oberwahrenbrock *et al*. (1999), refered to in Behrmann-Godel and Eckmann (2003)
14				15–25			Belgium	river Meuse		*A. anguilla*		referred to in Bruijs and Durif, 2009
15				15–25			Belgium	River Moselle		*A. anguilla*		referred to in Bruijs and Durif, 2009
16				24		Kaplan	Netherlands		Mean= 47	*A. anguilla*		Hadderingh and Bakker (1998)
17				6–23		Kaplan	Netherlands		Mean= 49–60	*A. anguilla*		Hadderingh and Bakker (1998)
18				30		Kaplan	Netherlands	Line		*A. anguilla*		Hadderingh and Bruijs (2002)
19					16–26	Kaplan	Netherlands		Range = 64–93	*A. anguilla*	Combined mortality from 2 successive power plants	Winter *et al*. (2006)

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
20					25–34	Kaplan	Netherlands		Range = 64–93	*A. anguilla*	Combined mortality from 2 successive power plants	Winter *et al*. (2007)
21	Yes	y	80	7	5	Kaplan	Netherlands		Mean = 76	*A. anguilla*		Jansen *et al*., 2007*
22	Yes	y	-	-	12	Kaplan	Netherlands			*A. anguilla*		Jansen *et al*., 2007* studies at different power stations
23		y	71	1	71	Francis	New Zealand		87–124	*Anguilla dieffenbachii and Anguilla australis,*		Watene *et al*. (2003)
24	head =178 m/ bypass/netting and transfer of silver eels	y		35		Francis	New Zealand	Manapouri Power Station		*A. dieffenbachii, A. reinardtii, and A. australis*		Boubée *et al*., 2008
25				9–100 *		Francis	Sweden		Range=50–52	*A. anguilla*		Montén (1985)
26				74	70	Francis	Sweden		mean= 74	*A. anguilla*		Calles *et al*. (2010)
27				40		Francis	Sweden		Mean= 74	*A. anguilla*		Montén (1985)

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
28					26	Kaplan	Sweden		Mean =74	*A. anguilla*		Calles and Bergdahl (2009)
29				40–100		Kaplan	Sweden		Mean = 57; Mean= 74	*A. anguilla*	Numbers from eight different powerplants	Montén (1985)
30			77			?	USA		Range= 71–91	*A. rostrata*		Haro *et al*. (2000)
31				27		propeller	USA	St Lawrence	90	*A. rostrata*		NYPA, 1998
32				27		Fixed blade propeller and francis	USA			*A. rostrata*		Normandeau Associates INC. and Skalski, 1998, ref. in Verreault and Dumont, 2002
33				37		vertical propeller and Francis	USA	Raymondville, Raquette River, NY	50–75	*A. rostrata*		NIMO, 1996
34				6		vertical francis	USA	Minetto, Oswego River, NY	50–70	*A. rostrata*		NIMO, 1995
35				9		vertical francis	USA	Luray Project, Shenandoah River, VA	56–112	*A. rostrata*	but 26% of injured (non lethal after 48h) eels	RMC, 1995

Nb	Type of plant	bypass or mitigation system	Proportion that went through the turbine intakes (%)	Turbine mortality %	Total mortality %	Type of turbine	Location	Name of power station/river	Body length (cm)	Species	Comments	Reference
36		y	100	0		Archimedes screw	UK		Range= 43–73	*A. anguilla*		
37				52%; 44% with full blade opening		Kaplan				*A. anguilla*	Review/modelling based on 71 field studies from several countries	Gomes and Larinier (2008)

Table 4-2. Mortality estimates according to type of turbine and presence of a mitigation system (bypass, fish-friendly turbine). The number of studies used to calculate the average mortality rates is given between brackets.

	Turbine mortality %	Total mortality %
Average (all turbines)	28 (29)	36 (10)
Average francis	32 (7)	52 (3)
Average kaplan	38 (9)	28 (6)
Average other turbines (mix, propeller, unknown)	21 (11)	40 (1)
Average no bypass or unknown	32 (24)	44 (6)
Average with bypass	9 (5)	26 (4)

4.4.2 Pumping stations

Pumping stations can negatively influence fish and fish migration as illustrated in Figure 4.5. In the first place pumping stations can cause damage and direct or delayed mortality in fish when passing through a pump. Secondly a pumping station functions as a barrier for migration diadromous fish like eel, both during upstream and downstream migration. Thirdly, pumping stations will increase the predation risk of fish. Damaged and confused fish will be easier to prey on by piscivorous fish or birds. But also the risk of being captured by commercial or recreational fishermen is higher around pumping stations when migratory fish aggregate while searching for an opportunity to pass. In this chapter, however, we will only focus on the impact of pumping stations on the survival of migrating eel when they actually pass through a pumping station.

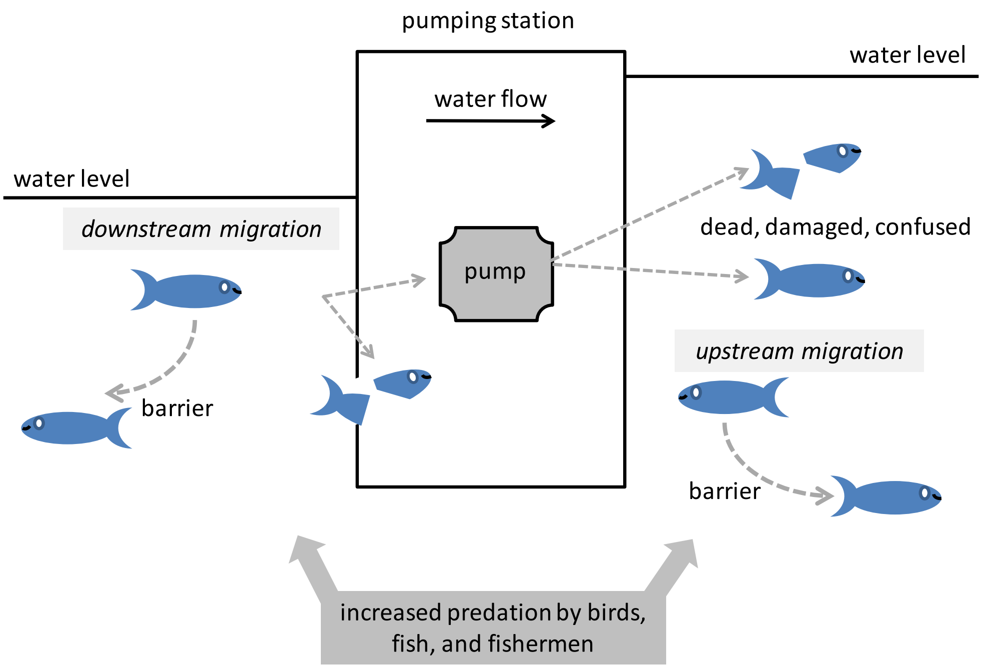


Figure 4-5. Impacts of pumping stations on fish and fish migration (redrawn from STOWA 2010).

Pumping stations can roughly be divided in three groups:

1) water wheels;

2) Archimedes screws;

3) pumps.

Pumps can be subdivided again based on the way the water flows through the pump in following three types:

1) centrifugal pumps (radial water flow);

2) propeller-centrifugal pumps (radial/axial water flow);

3) propeller pumps (axial water flow).

For any of the above mentioned pumps there are of course countless varieties in use based on, for example, capacity, blade velocity, head, blade diameter, etc. Figure 4.6 provides an overview of the distribution of different types of pumping station in the Netherlands (based on a sample of 2813 pumping stations).

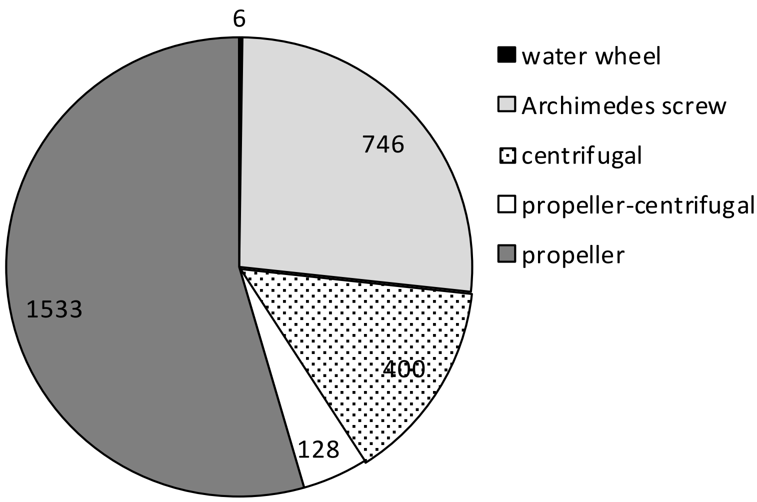


Figure 4-6. Distribution of different types of pumping station in the Netherlands (redrawn from Kunst *et al.*, 2008).

Table 4.3 provides an overview of studies conducted mainly in the Netherlands and Belgium on the impact of different types of pumping stations on the survival of eel. These studies clearly demonstrated that in general propeller pumps with axial or axial/radial water flow caused the highest mortality rates when eel passes through these types of pumps. Unfortunately, at least in the Netherlands, this type of propeller pumps are the most common type used to regulate water levels. On a "fish friendliness" scale propeller pumps are in general regarded as "unfriendly" while water wheels and Archimedes screws are relatively "friendly". Although Archimedes screws are less harmful than propeller pumps, contrary to popular belief, they are not by definition harmless and can still cause significant mortality. Not only the type of pumping station is important when considering eel mortality but also the characteristics of a pump play an important role as summarized in Figure 4.7. For example, Witteveen and Bos (2010b) demonstrated that a propeller pump with a low blade velocity and high capacity caused low (5%) mortality.

No doubt more information on eel mortality when passing pumping stations will become available in the near future. The current studies do, however, already provide reasonable estimates for pump station mortality to be used in models and especially where there is information on the characteristics of their pumping stations (type, head, blade velocity, etc.); it should be possible to make acceptable extrapolations in the models. What remains to be solved is to quantify the effect of a pumping station as a barrier for eel migration. A tagging study with this specific objective is currently being conducted in Friesland (the Netherlands) and the results will hopefully be available during WGEEL 2012.

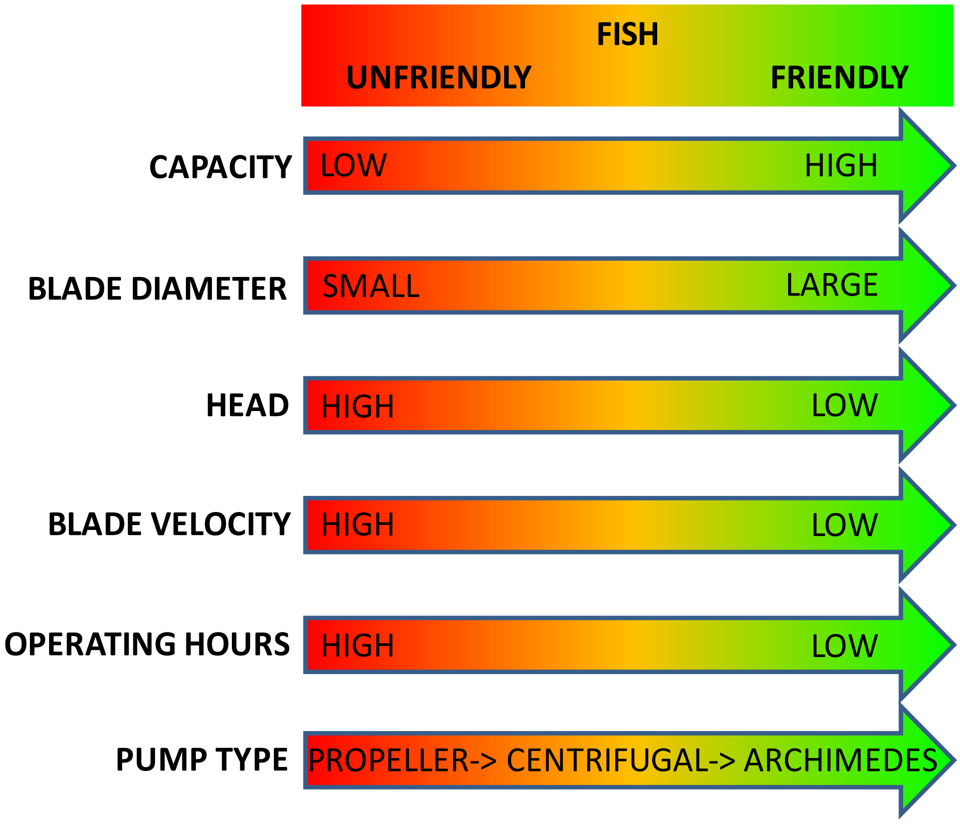


Figure 4-7. Factors influencing the "fish friendliness" of pumping stations (redrawn from Stowa, 2010).

Table 4-3. Overview of eel damage and mortality by different types of pumping station. *underestimation as physically undamaged eels did reveal internal damage after dissection which will result in delayed mortality. Only studies with at least 20 eels that passed through the pumping station during the course of a study are mentioned in the Table with the exception of the water wheel at Spaarndam.

Country	Pump type	Location	Capacity (m^3/min)	rpm	Head (m)	# eel	% damaged	% dead (direct)	Delayed mortality studied	Reference
NL	water wheel	Spaarndam	1920	6	0.3	5	0	0	Yes	Kruitwagen and Klinge, 2008
BE	Archimedes	Sint-Karelsmolen	30	39	2.9	?	10	4	?	Denayer and Belpaire, 1992
BE	Archimedes	De Seine	35	37	3.6	?	37	0	?	Germonpré *et al.*, 1994
BE	Archimedes	Isabella	100	25	?	48		8–19	?	INBO
BE	Archimedes	Isabella	200	21	?	131		13–16	?	INBO
NL	Archimedes	Overwaard	500	17	2.2	43		2	?	Vriese *et al.*, 2010
NL	“de Wit” Archimedes	Halfweg	660	22	0.3	29	0	0	?	Kruitwagen and Klinge, 2008a
UK	Archimedes		?	23–31	?	160	0.6	0	?	Kibel, 2008
GE	Turbine-Archimedes	Bielefeld	?	?	?	22	0	0	?	Spah, 2001
BE	centrifugal	Elektriek-Zuid	60	49	5	287	1.4	1.4	?	Germonpré *et al.*, 1994
NL	centrifugal	Schoute	505	143	-4.8	36	0	0	Yes	Kruitwagen and Klinge, 2008c
NL	centrifugal	Katwijk	1080	59	-4.3	56	0	0	Yes	Kruitwagen and Klinge, 2007
NL	centrifugal	Boreel	400	204	0.9	49		48	?	Vriese *et al.*, 2010
CAN	Hidrostalpump	?	?	890–1200	10	2300	<3	0	NO	Patrick and McKinley, 1987

Country	Pump type	Location	Capacity (m^3/min)	rpm	Head (m)	# eel	% damaged	% dead (direct)	Delayed mortality studied	Reference
NL	propeller-centrifugal	Tonnekreek	170	?	1.52	34		0	?	Vriese *et al.*, 2010
NL	propeller-centrifugal	Schilthuis	350	115	2.8	27		22	?	Vriese *et al.*, 2010
BE	propeller	Woumen	60	500	2.7	?	100	100	NA	Germonpré *et al.*, 1994
BE	propeller	Avrijevaart/Burggraven stroom	100	480	?	39	98%	98	?	INBO
NL	propeller	IJmuiden	15 600	64	0.1–2.3	35	71	52	Yes	Kruitwagen and Klinge, 2008b
NL	propeller	IJmuiden	3000	Variable	0.1–2.3	114	36	36*	Yes	Witteveen and Bos, 2010a
NL	propeller	IJmuiden	3000	Variable	0.1–2.3	251	40.6	41*	Yes	Witteveen and Bos, 2010a
NL	propeller	Hoogland	1500	50	?	77	5	5	Yes	Witteveen and Bos, 2010b
NL	propeller	Lijnden	255	360	5.4	?		100	?	Kruitwagen *et al.*, 2006
NL	propeller	Stenensluisvaart	60	500	2.7	?	100	100	NA	Germonpré *et al.*, 1994
NL	propeller	Den Deel	200	165	0.6	?	30	8	?	Riemersma and Wintermans, 2005
NL	propeller (closed)	Kortenhoeve	60	355	0.8	118		32	?	Vriese *et al.*, 2010
NL	propeller (closed)	Thabor	24	?	1	21		38	?	Vriese *et al.*, 2010

4.4.3 Cooling water intakes

Intakes used for water supply represent another anthropogenic threat to aquatic ecosystems and fish stocks. When water is abstracted from surface waterbodies, there is a risk that fish and other organisms will be drawn in. This may prevent fish from migrating naturally, transfer them to harmful environments and cause death or injury to fish at screens, turbines and pump mechanisms (Environment Agency UK, 2011).

Potentially, eels can get caught up in intake flows and screens at any stage of their life. However, they are most at risk during their upstream and downstream migrations within freshwaters (Environment Agency UK, 2011). How they behave in near-shore marine, transitional and freshwaters will determine how vulnerable they are to entrainment during this period.

Intakes–adult silver eels are particularly vulnerable when they actively follow currents downstream ('positive rheotaxis'). Glass eel and elvers are also at risk when they have to pass areas with intakes, which sometimes have enormous capacities for water intake.

Outfalls–juveniles (glass eels, elvers or smaller yellow eels) are more at risk during active migration upstream ('negative rheotaxis').

A risk of entrapment for eel (and fish in general) exists at different places/installations. In Table 4.4, some potential entrapment hazards are listed (taken from: Environment Agency UK, 2011).

Table 4-4. Summary of potential sources of entrapment (Environment Agency, 2011).

Power stations' cooling water systems
Hydroelectric power installations
Pumping stations
Desalination plant
Drinking water abstractions
Water transfer schemes
Industrial abstractions
Industrial discharges
Sewage treatment works
Agricultural abstractions
Flood alleviation schemes
Water level management
Fishfarms
Temporary abstractions

The abstraction of cooling water by power plants causes a wide range of ecological impacts on aquatic communities. Effects are situated both within the cooling-water circuit and in the cooling water receiving waterbody. Thermal loading related to cooling water discharges directly interferes with physiological processes of the biota, such as enzyme activity, feeding, reproduction, respiration, growth and photosynthesis (Hadderingh *et al.*, 1983). Behavioural changes (attraction or avoidance) are commonly observed in organisms subjected to thermal discharges as well (Kennish, 1992), e.g. as advanced spawning time and high fish concentration in the outlet area.

The effects of thermal loading clearly differ in each single case and depend on the actual conditions in the relevant waterbody and on the special characteristics of the respective power plant. A comprehensive literature study on this issue has been conducted by Krieg *et al*. (2010) for the tidal zone of the River Elbe. It is important to note that such effects can be of importance, but they cannot be generalized here.

Of greater potential impact on the aquatic communities than waste heat discharges, however, are probably the losses of various life-history stages of invertebrates and fish due to impingement on intake screens or entrainment through cooling systems (Hadderingh, 1979). The damages and mortalities are mainly caused by mechanical contacts (screen, pump, condenser passages) and pressure fluctuations (pump, condenser).

Whereas these mechanisms and influences are known theoretically, relatively little knowledge exists about quantitative aspects. There are several case studies, in which damage and mortalities were documented for single locations over certain time periods. The results are partly published in "grey reports", which are often not available for the whole scientific community. In addition, they are often written in the language of the respective country, which reduces the international use. Sometimes such reports form the basis for negotiations e.g. about payment for compensation for fishermen. Therefore, the studies often only measured the absolute number of damaged and killed fish. Maes *et al*. (1998) estimated the loss of young-of-the-year fish at the Doel nuclear power plant at the Scheldt estuary to about 50 million fish per year.

Some studies estimate the percentage of fish killed during the entrainment in relation to all fish entrained. Roqueplo *et al*. (2000) state that 9% of the fish entrained at a French nuclear power plant at the Garonne estuary (Blayais) were killed immediately and that mortality increased to 15% after one week. Previously (in the fishing season 1994/1995), the amount of glass eels entrained had been calculated as 4.2 tons. For the Swedish nuclear power plant Ringhals, mortality of eels was estimated at 10% of total entrainment (Jan Andersson, Swedish Board of Fisheries Institute of Coastal Research, personal communication). From these Swedish investigations, some data are available for entrainment and damage of eels at Swedish nuclear power plants. It becomes obvious that in particular glass eels have been entrained and killed in considerable numbers in the Ringhals nuclear power plant. Yellow and silver eels are usually less strong influenced, but the numbers of entrained, damaged or killed fish may still be substantial (Tables 4.5–4.7). However, the available studies revealed a considerable variability of the results and a strong effect on eels has not been shown in all studies.

For example, in a study at the power plant Moorburg at the River Elbe (Germany) in 1994/1995, where mean cooling water intake was 15 m^3/s, in total 67 291 fish of 15 species were found in the collection containers (24 times 1-day-sampling) but eel only represented 0.2% by number (137 individuals) and 0.6% by weight (Rathke, 1996).

Monein-Langle (1985) studied the effects of a water intake at a coal-fired power plant in the Loire and found no evidence of direct mortality or negative effects of the temperature increase on glass eels.

A monitoring of impingement rates of fish at Moor Monkton water abstraction works in the River Ouse was carried out between 1990 and 1991 by Frear and Axford (1991). A total of 16 022 lampreys and 4793 other fish were trapped on the water intake screens during the study period. A significant number of fish died after impingement due to asphyxia caused by low water volumes in both the backwash channel and collection baskets. However, during the study only 48 eels were found impinged (1% of non-lamprey species). Similarly, in a recent study on the River Dee, eel were not impacted to any noteworthy extent (APEM, 2007).

In a German study at the intake of a waste treating facility 602 eels were found at twelve samplings in 1999. Eel was the second most important species by weight and on a third rank by number (Kloppmann, 1999). By far the strongest damaged species in this study was smelt (*Osmerus eperlanus*). Based on the results, the total amount of killed fish during the whole study period was estimated to 5105 fish per day, among them 39 eels with a total weight of 622 g (per day).

In an older study at the nuclear power plant Brunsbüttel (river Elbe, Germany; Rauck, 1980), between 0–82 kg eel were killed per day. Cooling water intake at that time was ca. 30 m^3/s. When the actual amount of cooling water during the study period was considered, an annual loss of eels of more than 6 tons was calculated. The loss of biomass of eels of marketable size was slightly higher than that of stocking size eel. The total annual loss of all species was between 55 and 58 tons. The study also documented the damages and injuries of eels at the fine screen. 31–44% were found undamaged, 48–55% were damaged and of 8–15% only pieces were found. A later study at the same nuclear power plant found slightly lower biomasses of killed eels (Möller *et al.*, 1991), but it should be considered that the biomass of eels in the river Elbe may already have been reduced at that time. This points towards the following problem: the number of damaged and killed fish usually can be measured with a relatively low effort. However, if the results should be related to the total fish population in the respective waterbodies, what would be desirable to assess the real impact of the intake on a population, the effort would increase enormously, in particular in estuaries or downstream regions of large rivers. This is probably the reason why, only few studies relate total losses at an intake to the total stock of the species (or age group) in the respective waterbody. One study, which aimed to establish this relationship, was conducted by Turnpenny *et al.* (2008), who studied the effects of nine raw water intakes at the river Thames. Beside the studies directly at the intakes, the fish stock in the Thames was also considered. The authors show a considerable impact on larvae and smallest juvenile stages of fish (mainly coarse fish). By calculating the *Equivalent Adult Value* it was shown that over the last five years, potentially, the number of fry entrained per year at all intake sites could amount to 31% of the total adult stock. If all intakes abstracted at their maximum rate this could amount to 61% of the total adult stock. However, damages or mortalities of eel were not observed in this study.

In a recent study, Beaulaton and Briand (2007) included the effects of cooling water intake of a French nuclear power plant on glass eels in their modelling of several management scenarios for the eel stock in the Garonne estuary.

The available results show a considerable variability of the effects of water intakes on eel. The biggest effects can probably be expected for glass eels and elvers, but impingement on screens may also affect bigger eels. The degree of damage largely depends on the actual conditions at each location (type of power plant or technical facility in general, capacity of water intake, configuration and design of mitigation measures including screens and behavioural deterrent systems, biological characteristics of the potentially impacted species, etc.) and generalizations are difficult. For a better assessment of the impacts of water intakes on eel stocks an inventory of all intakes for each RBD would be helpful. Furthermore, existing knowledge of damage and mortality rates should be stronger related to the total fish (eel) stock in the respective RBD's. If it has not been done so far, these effects should be included in the modelling of eel stocks in the frame of the eel management plans according to the Regulation (EC) 1100/2007.

Recently, the effort to reduce the damage of fish at technical installations as hydropower turbines, pumping stations or water intakes has increased. To reduce the impact of cooling water abstraction on fish populations in the surrounding waters different methods and devices have been introduced with variable success. The major deflection methods at power stations include visual stimuli (e.g. air-bubble screens, lights or strobe lights), water velocity and pressure changes, electrical shocks and sounds. Mechanical exclusion devices use fine-screens surrounding an intake from which cooling water is drawn (Maes *et al.*, 2004). There exists probably a great amount of approaches and case studies, which cannot be reviewed here. However, a few reviews and best practice approaches have been published (e.g. Ontario Waterpower Association 2010, Environment Agency, 2011).

Table 4.5. Estimated eel mortalities at the Swedish nuclear power plant Oskarshamn (Jan Andersson, pers. comm.).

Estimated mortality in numbers per year (April–September)		
Oskarshamn Reactor 1		
	Eel <40 cm	Eel >40 cm
2003	1264	159
2004	1003	85
2005	93	296
2006	225	0
2007	457	241
2008	335	146
2009	423	821
2010	98	302
Total	3898	2050
Oskarshamn Reactor 2		
	Eel <40 cm	Eel >40 cm
2003	215	80
2004	1161	198
2005	1158	313
2006	792	328
2007	343	344
2008	283	35
2009	367	33
2010	259	360
Total	4578	1691

Table 4.6. Estimated total entrainment of glass eels at the Swedish nuclear power plant Ringhals (four reactors) during February and April. Mortality is estimated as 10% of total entrainment (Jan Andersson, pers. comm.).

Week nr	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
5	4159	3961	5050	8400	1485	1188	0	0	297	0	20720	560	7560	1680	0	5494	0	2800	0	0	0	0
6	7427	3714	15745	12460	1981	2971	0	297	326	0	39620	0	24080	1400	280	3381	560	11620	2240	280	0	0
7	6090	2575	23767	92540	2080	11487	0	6239	2525	2228	74760	840	42980	420	560	17271	700	980	7420	0	0	280
8	15894	743	12180	15960	2228	13517	3119	891	14111	3450	32200	1400	91420	1400	0	6087	560	3360	4760	1120	0	0
9	51692	743	44711	15260	891	48243	0	19013	35056	17231	96320	1260	32620	1400	140	4142	1680	2940	2800	700	0	140
10	42037	4308	131905	33040	2080	62684	1188	56000	28074	33867	105420	700	56000	2800	700	2817	560	7980	8680	560	560	840
11	42037	24658	81252	36400	170674	93284	3714	55554	2228	20202	149240	6160	102480	12180	700	10951	280	22680	31780	840	1120	980
12	40700	11883	131310	149800	112000	159434	7130	16785	49464	44117	59780	6580	148400	14700	4900	45298	3640	106960	10640	4243	2240	9380
13	70408	30897	157454	138600	400318	92789	16785	11883	61645	594	134120	4480	16520	51800	9800	42814	4760	52080	8260	10002	980	10360
14	98334	38621	42186	112700	345952	175130	9358	21687	125369	87788	263620	6160	51660	53760	18060	45292	15400	28140	12040	10305	9380	19600
15	79618	115862	83480	151060	320649	25370	35947	56149	66249	105464	43120	12460	51520	42140	15400	56530	27020	53480	7140	6971	6720	50120
16	33570	71003	39809	147560	173347	17825	20499	13666	26886	22133	83720	6860	14840	20580	25060	80068	36960	5600	3500	6365	25340	15960
17	10398	54069	64764	157920	77836	22430	15745	12180	56594	39512	71820	35700	2100	44100	8820	18504	17220	4900	560	3031	6440	20440
18	0	34759	15448	0	0	0	0	64383	8764	0	0	0	2520	12880	2240	2818	9940	1960	0	0	7700	0
Total	502 366	397 793	849 061	1 071 700	1 611 521	726 352	113 485	334 728	477 589	376 585	1 174 460	83 160	644 700	261 240	86 660	341 468	119 280	305 480	99 820	44 416	60 480	128 100

Table 4.7. Estimated total entrainment of eel at the Swedish nuclear power plant Forsmark, southern Bothnian Sea. Mortality is 100% of entrained eels. Considerable stocking was performed in the 1980s and possibly 1990s (Jan Andersson, Swedish Board of Fisheries Institute of Coastal Research, personal communication).

Total estimated mortality in an eight week period in spring																										
		1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Eel (y+s)	n		388	396	815			283	143	127	93	337	159	342	237	105	576	476	683	726	648	797	643	801	749	836
mean weight	g		42	42	11		0	60	71	116	21	105	105	231	420	11	289	189	147	383	179	305	368	252	189	294
Total estimated mortality in a twelve week period in autumn																										
		1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Eel (y+s)	n	170	34	14	31		30	230	126	420	305	803	399	473	551	735	336	625	1166	672	1008	861	1250	1292	63	641
mean weight	g	26	336	448	74		59	46	111	120	388	121	142	158	285	621	458	774	763	758	835	887	948	911	910	789

4.4.4 Barriers to migration

One of the many modifications that took place in our rivers was the introduction of barriers and the resulting habitat loss has impacted on the structure of our eel stocks. These barriers include culverts, weirs, bridge aprons, dams, hydropower stations, pumping stations, tidal flaps, sluices and surprisingly salmonids passes or fish counter installations (such as crump weirs for resistivity counters). For example, in the St Lawrence River in Quebec, there are 5600 dams impeding upstream eel migration.

Under the Water Framework Directive there is an obligation on Member States to ensure fish passage at all artificial structures. Kemp *et al*. (2008) define barrier porosity as the proportion of fish that encounter an impediment and then successfully pass it (during either an upstream or downstream migration) without undue delay (i.e. the probability of reaching the final destination, e.g. spawning or feeding grounds, is not compromised due to increased energetic expense or predation risk).

Barrier porosity can depend on a number of parameters that can vary throughout the year, such as life stage of species encountering barrier and flow conditions (high or low flow conditions). The impact of temporal barriers and the eel life stage they impact needs to be taken into account when estimating eel production losses.

The presence of a network of barriers along a system will delay the migration of eels further upstream. The energy costs and mortality associated with passing a structure also needs to be taken into account. A delay in migrations will result in increased predation by piscivorous birds, fish and mammals, including recreational and commercial fishermen. Has this factor been taken into account when calculating mortality values within a system?

Ryan (2010) reported that the success of a fish passing an obstruction is based on the physical parameters of the barrier, hydraulic conditions and the swimming abilities of the target fish species during their season of migration. The Environment Agency in the UK and Department of Fisheries and Oceans in Canada (Tremblay *et al.*, 2011) have recently published reports detailing methods for identifying barriers.

These parameters include:

- height of structure;
- width of structure;
- slope of structure;
- material used (masonry, timber, etc.);
- distance from sea;
- edge effects;
- absence/presence of alternative route (fish ladder, bypass channel).

All of these variables are given a ranking value to determine whether a barrier is an impassable structure or not,

- a high risk;
- medium risk,/nearly impassable;
- low risk or/passable at times or with some difficulty;
- not a barrier/passable without apparent difficulty.

Using the barriers database of characterization developed by Tremblay *et al*. (2011), a model approach was proposed to prioritize mitigation schemes in the Rimouski River (Quebec). The Rimouski is 1376 km long, mid-size watershed on which 23 barriers higher than 1 m occur. A diffusion rate of 20 km^2 $year^{-1}$ over a 20 year period was applied in the model. An annual mortality rate of 10% per dam was applied to the estimates. The impact of each barrier was simulated by reducing movements upstream at the barrier, survival among eels blocked downstream, and survival of eels at the barrier during the downstream migration. Model outputs have emphasized the prime importance of a single dam located near the estuary in determining the colonization process whereas mitigating other dams in the watershed did not significantly improve spawning escapement (Lambert *et al*., 2011).

The importance of knowing where a barrier is within a system will allow the accurate assessment of productivity within a system. The presence of a barrier low down in a catchment will result in a lower eel productivity due to the removal of the wetted area above the barrier than if the whole wetted area is evaluated. Restricted upstream migration by juvenile eel limits the available habitat and will increase density-dependent mortality. Churchward (1996) reported that elvers were rarely found to move more than 15–25 km upstream in their 0^+ group year in the Severn. Mediating the effects of barriers within a catchment should start with the first barrier encountered by the elvers.

It is important that countries identify obstacles and evaluate such structures impact on eels. It is recommended that a barrier assessment is carried out to cover requirements under the Eel Directive, Water Framework Directive and Habitats Directive. An opportunity exists for research groups to pool resources and carry out a multi species assessment to benefit a large number of research groups (WFD, Habitats (lamprey), Eels). A national barrier assessment will prioritize barrier mitigation work increasing eel abundance and escapement as directed under the EU Eel regulation.

4.4.5 Predation (natural mortality by cormorants)

Eel populations represent a significant component of the aquatic ecosystems, including their considerable contribution to the diet of other fish (e.g. *Silurus glanis*) and semi-aquatic predators such as, cormorants (*Phalacrocorax carbo*). All the life stages of the eel, in coastal and inland aquatic habitats, can be an important part of the diet for these predators and can therefore add to the natural mortality of the eel. Piscivorous birds may also have a secondary indirect effects on the eel population such as death of fish caused by shock due to birds' incursions and consequent injuries incurred by fish that could cause the death. Moreover fish eating birds such as cormorants may also have a role in extending the distribution of *A. crassus* through fish regurgitation (Wlasow *et al*., 1998). Such secondary effects should therefore also be considered when the impact of different species on the eel stock is assessed.

Phalacrocorax carbo (L.), known as the Great Black Cormorant, lives on western European coasts and south to North Africa, the Faroe Islands, Iceland and Greenland; and on the eastern seaboard of North America. There are two species *P. c. sinensis* than in *P.c. carbo.* In general the cormorant can be considered as an opportunistic predator. However, it has often been suggested to consume large amounts of eel (Carss and Ekins, 2002; ICES, 2007; Zydelis and Kontautas, 2008). Consequently, among potential natural predators, birds are most efficient and are thought to be a key controlling factor in a number of *A. anguilla* populations (ICES, 2007). Moreover this situation generates conflicts, mainly between recreational and commercial fishermen (Jepsen *et al*., 2010). On the other hand, cormorants are currently protected under European legislation (Birds Directive 79/409/EEC).

In particular, the dramatic increase in cormorant (*Phalacrocorax carbo*) numbers that has occurred in Europe since the 1960s, which has resulted in conflicts with commercial fisheries and recreational angling, is well documented but the reasons for the population trend and range expansion are still not fully understood. Possible causal factors are, according to the REDCAFE project (2002), a "non-limiting food supply" and protective legislation such as the EEC Directive 79/409 on Conservation of Wild Birds. The population increases are much greater in the subspecies *P. c. sinensis* than in *P.c. carbo*, which is found mostly in Norway, Britain, Ireland and northwestern France (Kohl, 2010). However, a reliable estimation of the total population size in Europe is missing. Further, significant seasonal changes in local cormorant populations result from movements of adult birds to and from breeding colonies and from longer distance dispersal by immature individuals. The complex population dynamics of these opportunistic predators makes analysis of their impacts on eel populations difficult. Likewise, control measures are still a significant factor affecting their habitat utilization patterns and local movements in different parts of the species range. For example, the REDCAFE project noted that up to 10 000 birds were being shot as "game" in Norway in 1995 and illegal shooting of cormorants by anglers is known to occur in several EU countries.

It is generally accepted that the average food intake of an adult cormorant (*P. c. sinensis*) is about 400–500 g of fish per day (Consolo *et al*., 2009). This value could vary depending on the habitat, the season, the bird's sex and on the energetic value of the prey. However the significance of cormorant predation on fish in restricted areas, such as poundnets, close to the lagoons' barriers or in the hydropower tailraces, where it's easier to prey eels, can readily be established (Dieperink, 1995), whereas the effect on natural fish populations in open areas is more difficult to estimate and there is a lack of scientific documentation of the effect of the birds' predation on fish stocks.

In the current WGEEL Country Reports 2010/2011 not all countries provide estimation of the eel predation by cormorants or refer to local studies survey the proportion of eel in the cormorant diet. Across Europe several studies have been undertaken to quantify the proportion of different prey items in the cormorants' diets. In Table 4.8 a summary is given of selected studies conducted in different countries during the time period of 1974 to the present.

As no standard protocol for the investigation of the cormorants' diet is available the listed studies differ regarding their sample size, sample time and method of diet analysis. In general, the percentage of eel in the diet varied between the different studies (range 0%–46.6% regarding the biomass). These differences can be caused by the habitat characteristics (coastal inland waters, stocking, etc.) or temporal scale (seasons, see Simon (2011)). Furthermore, the used method (regurgitation or fresh stomach content) could cause biased results. Additionally, in light of the eel stock decline it is questionable whether diet estimates prior 2005 should be used to estimate the current impact of the cormorants. Also the lack of time-series data on changing diet composition of cormorants and their impact on the eel stock cannot be provided.

The currently available information on cormorant diet composition, and on local variation in relative abundance of eels at sites where dietary studies have been undertaken in the past, do not allow for a detailed analysis of the kind needed for overall European eel management purposes. A better integration of studies on cormorant population biology is required, with parallel studies on the two subspecies (*P. carbo sinensis* and *P.carbo carbo*), and the eel monitoring being undertaken with respect to both the Water Framework Directive and the EU Regulation for Restoration of European Eel. The INTERCAFE project, currently in progress, should provide a good opportunity for the relevant data on cormorant population biology (abundance, seasonal population dynamics, temporal and geographic variation in clutch size and results of studies on diet) to be compiled. Information on cormorant management in different parts of Europe should also be clarified during the INTERCAFE project and the significance of the high level of cormorant utilization as a game resource in Norway may also be better understood in future. Cormorant management by use of bird-scare technologies, in which concentrations of piscivorous birds are locally reduced by diverting them to waters not regarded as important by anglers or fish-farm owners, could possibly result in changes in the relative importance of eels in their diet. The opportunistic feeding behaviour of cormorants illustrated by previous studies in Ireland and elsewhere, is well documented. The need for caution in interpreting limited data on cormorant diets, or in equating biased field observations of anglers (in which easily identifiable eel are frequently over represented) with results from more systematic studies, needs to be recognized. Likewise, the undue attention being given to this particular avian eel predator, as opposed to herons, bitterns etc., is of concern and it may be better to review the potential impact of cormorants in a wider ecological context in which all eel predators are assessed.

Table 4.8. Selected studies on the proportion of eel in the cormorants' diet.

Study	Years of observation	Country	Inland or Coastal waters	Method of diet analysis	% of eel in the diet (* of biomass or # number of prey items)	Comments
Andersen *et al*. (2007)	1997	Denmark	Coastal	Regurgitation	0.4%#	
Hald (2007)	2005	Denmark	Coastal	Regurgitation	0.3%#	
Sonnesen (2007)	2005	Denmark	Coastal	Regurgitation	0.12%#	
Santoul *et al*. (2004)	2001–2004	France	Inland	Regurgitates	0%#	
Carpentier (2009)	1999–2007	France	Inland	Regurgitation	4.45 %#,5.84 %*	
Ubl (2004)	2002–2003	Germany	Coastal	Fresh Content	3.1%*,0.2%#	
Keller (1995)	1990–1991	Germany	Inland	Regurgitation	6.1%*	
Knösche (2005)	1996–2004	Germany	Inland	Regurgitation	8.35%*	Summary of different studies
Knösche (2005)	1996–2004	Germany	Inland	Fresh content	19.55%*	Summary of different studies
Brämick (2007)		Germany	Inland	Regurgitation	13% *	
Simon (2011)	2006–2009	Germany	Inland	Regurgitation	7.8% *	Differences between seasons
West, Cabot and Greer-Walker (1974)		Ireland	Coastal	Regurgitation	20%#	
Warke and Day (1995)	1992	Ireland	Coastal/Inland	Regurgitation	41%*	
Warke and Day (1995)	1992	Ireland	Inland	Fresh content	3%#	
Warke and Day (1995)	1992	Ireland	Coastal	Regurgitation	0%*	
Doherty and McCarthy (1997)	1995	Ireland	Inland	Observation,Regurgitation	41%#,9%#	
Žydelis and Kontautas (2008)	2001–2002	Lithuania	Coastal	Regurgitates	0%#	
Birzaks (2011)	2009	Latvia	Inland	Not mentioned	0.6%#, 2.6%*	
Dirksen *et al*. (1995)	1989–1992	Netherland	Inland	Regurgitation	0.2%*	
Veldkamp (1995)	1991	Netherlands	Inland	Regurgitation	1.6%*	

Study	Years of observation	Country	Inland or Coastal waters	Method of diet analysis	% of eel in the diet (* of biomass or # number of prey items)	Comments
van Rijn (2001)	1996–2000	Netherlands	Inland	Regurgitation	<1%*	
van Rijn (2005)	2005	Netherlands	Inland	Regurgitation	0%*	
Wright (1986)		N. Ireland	Coastal	Regurgitation	43%*	
Warke and Day (1995)	1995	N. Ireland	Coastal	Regurgitation	44%*	
Lorentsen *et al*. (2004)	2001–2003	Norway	Coastal	Mixture of methods	0.1%*	
Dias (2007)	2005/2006	Portugal	Coastal	Regurgitation	7%#	
Engström (2001)	1998	Sweden	Inland	Regurgitates	0%#	
Wickström *et al*. (2011)	2009–2010	Sweden	Inland, west coast	Fresh content	1–3%*	Differences in area and seasons
Wickström *et al*. (2011)	2009–2010	Sweden	Inland, east coast	Fresh content	<2%*	Differences in area and seasons
Suter (1997)	1985–1992	Switzerland	Inland	Regurgitation	0.3%*	
Schafer (1982)		UK	Coastal	Observation	16%#	
Bearhop *et al*. (1999)	1994–1996	UK	Inland	Fresh content	0%#	
Carss and Ekins (2002)	1992, 1993, 1995	UK	Inland	Regurgitation	24.7%#, 46.6%*	

4.5 Conclusions to Chapter 4

Some information is available to quantify eel mortality induced by non-fishery anthropogenic activities such as eel passage through pumping stations or hydropower plants and predation by cormorants. In the ICES precautionary diagram for eel, only these effects that can be quantified are presented. In turn, this means that only management measures can be evaluated which act on such quantitative parameters. It is important to understand that estimates of life time anthropogenic mortality presented by most countries may have significantly underestimated the true anthropogenic mortality if the impact of (unquantified) anthropogenic activities like e.g. pollution, parasites, viruses, IUU fishing, cooling water intake, and barriers have not been accounted for.

Recommendation	For follow up by:
Life time anthropogenic mortality	
Express anthropogenic mortality events in terms of numbers or % eels, and size-based selectivities	Member states
Collect data on your eel numbers, densities and length distributions making use of WFD	Member states
Analyse the fisheries data collected for the Data Collection Framework (DCF) to estimate fishing-based mortality	Member states
Develop the requirements for eel in the DCF and WFD to reflect these data requirements to support the estimation of anthropogenic mortalities and their summation	Member states
Sex Ratios	
collect sex-ratio data of young yellow (<35cm TL) eel at a fixed location(s) in the lower reaches of a river; provide an estimate of age and density	Member states
Hydropower, Pumping stations, Water intake and other barriers	
conduct an inventory of hydropower, pumping station, water inlet location and characteristics	Member States
undertake studies to quantify the effect of pumping stations as migration barriers for (silver) eel migration undertake; studies to quantify the impact of water intake on eel	Member States
conduct inventory of temporary, permanent, natural and artificial obstacles for eel migration along with estimation of habitat loss for eel above these barriers	Member states
Predation	
Estimation of eel predation by cormorants and put in wider ecological context	EIFAAC WG on cormorants

5 Assessment of the quality of eel stocks

Chapter 5 updates the European Eel Quality Database (EEQD) and discusses the importance of the inclusion of spawner quality parameters in stock management advice. Chapter 5 addresses the following Terms of reference:

d) Provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods;

e) Review and develop approaches to quantifying the effects of eel quality on stock dynamics and integrating these into stock assessments; develop references points for evaluating impacts on eel.

5.1 Introduction

In recent years WGEEL has discussed the risks of reduced biological quality of (silver) eels. The reduction of the fitness of potential spawners, as a consequence of (specific) contaminants and diseases, and the potential mobilization of high loads of reprotoxic chemicals during migration, might be key factors that decrease the probability of successful migration and normal reproduction. An increasing amount of evidence has been presented indicating that eel quality might be an important issue in understanding the reasons for the decline of the species. Previous WG reports have presented an overview and summaries of a variety of reports and data on eel quality. Hence, this chapter should be read in conjunction with the 'eel quality' chapters in WGEEL 2006, 2007, 2008, 2009 and 2010.

During the meeting WGEEL 2011 further updated the European Eel Quality Database (EEQD) in order to analyse some trends. We summarized scientific advancements regarding the better understanding of the status and effects of contamination and diseases in the European eel, in order to facilitate future local assessments of the stock (yellow eels, silver eel and SSB). During this session we further updated the list of areas where fisheries restrictions were issued because contaminant levels in eel were above human consumption safety limits. We made progress in developing a framework for integrating quality of eel factors in local stock assessments.

5.2 Information of eel quality provided by countries and update of database on eel quality related data: the European Eel Quality Database (EEQD)

5.2.1 Introduction

The European Eel Quality Database (EEQD) was created by INBO (Belgium) in 2007 and has been fully described in Belpaire *et al*. (2011a). The database integrates data of contaminants (polychlorine biphenyls, pesticides, heavy metals, brominated flame retardants, dioxins, PFOS), diseases and parasites (such as *Anguillicoloides crassus*, bacteria, and viruses such as EVEX and other lesions), and fitness (fat content).

The database now represents the first comprehensive pan-European compilation of eel health data, including data from over 10 000 eels from approximately 1200 sites over 14 countries. Preliminary work has indicated a number of shortcomings and future developments will be needed. Guaranteeing further development of the database, harmonization of methods, quality assurance, and setting up harmonized eel monitoring strategies over Europe will be a great challenge and will need pan-European cooperative work. Belpaire *et al*. (2011b) included some overview tables and figures about eel quality monitoring over Europe. Specifically, there is a table with an overview of information and eel health descriptors included in the European Eel Quality Database, and a table with the number of records of eel quality data over quality elements reported by European countries and compiled by WGEel (2007–2010) in the European Eel Quality Database. A figure with the densities of records of PCBs and the swimbladder nematode *Anguillicoloides crassus* in eel in European countries is presented. Another figure represents levels of PCBs and prevalences of *A. crassus* in eel from several European countries.

Before and during the 2011 WG meeting the EEQD was updated with new data. These data were retrieved from recently published reports or scientific papers, and from the Country Reports. Table 5.1 summarizes the amount of new data added to EEQD during the WGEel 2011 session.

The following sections give an overview of new information on contaminants or diseases that has become available to WGEEL since the 2010 report. Although new information has been provided on eel quality in several countries, a comprehensive overview on the eel quality over its distribution area is still far from complete.

Table 5-1. Amount of new data included during WGEEL 2011 session.

Contaminant group or pathogen	Number of new records
polychlorinated biphenyls (PCBs)	50
Dioxins	114
Pesticides	43
heavy metals	48
Anguillicoloides	149
viruses, bacteria and other diseases	2
lipid content	65

5.2.1.1 Contaminants

The International Commission for the Protection of the Rhine (ICPR) started collecting and analysing fish contamination (with among others, eel) data from the states along the river Rhine in order to describe spatial distribution and trends over the last ten years (to be published within a year). It will contain relevant data from the Netherlands, Germany, Luxemburg, France, and Switzerland and will be mainly focused on dioxines, furanes, dioxinlike and indicator PCBs. Norms and guidelines used within the report will be adapted from food laws.

Along the lower reaches of many European rivers i.e. Elbe, Rhine, Weser, Ems, and Maas the WHO-TEQ for Dioxines, Furanes, and dl-PCBs in eels are exceeding the European consumption level (EC, 2006; Stachel *et al*., 2006; Geeraerts *et al*., 2011; ML, 2011).

Sweden

SLV (the National Food Administration) has analysed two pooled samples of eel from 2010 in Lake Vänern. They were analysed regarding dioxins, furans and PCB. All values were below allowed limits as well as action levels. However, there were significant differences between eels from the two sites in the lake (Gitte Eskhult, SLV, pers. comm.).

Sweden provided these data for inclusion in the EEQD (see also Figure 5.10 for dioxin levels).

Germany

Nagel *et al*. (2011) examined metabolites of polycyclic aromatic hydrocarbons (PAH's) in the bile of eels from twelve German rivers and discussed their use as biomarkers. In total, 170 yellow eels were analysed. Significant differences occurred in concentrations of PAH metabolites between the rivers and there were differences in the ratio of different PAH metabolites. For all rivers, the dominant PAH metabolite was 1-OH-hydroxypyrene. The individual results for this metabolite ranged from <22.5 ng/ml in the river Uecker to 3724.5 ng/ml in the river Trave.

Two German fish monitoring reports from Baden-Wuerttemberg and Lower-Saxony refer to contaminants in eel (River Rhine and Lake Constance; Rivers Elbe, Ems, and Weser). 14 of 15 eel samples taken from the Rhine in 2008 showed values in the range of or above the maximum level for the sum of PCDD/Fs and dioxin-like PCBs. Five further eel samples from lake Constance showed values below the level (CVUA 2010). See Figures 5.9 and 5.11 for the levels of dioxins in German eels.

Nine eel samples from Elbe, Ems and Weser tributaries also show values above the maximum level for the sum of PCDD/Fs and dioxin-like PCBs, one sample (Ems) remains below (ML, 2011).

The Netherlands

New data issued from the eel contaminant monitoring during 2010 is available and The Netherlands has provided data on contaminants levels in eel for inclusion in the EEQD. The 2010 dioxin levels are summarized in Figure 5.10.

Belgium

Roosens *et al*. (2010) assessed the degree of pollution with the brominated flame retardants PBDEs and HBCDs in pooled eel samples from 50 locations in Flemish waters collected in the period 2000–2006. Concentrations of ∑PBDE ranged between 10 and 5811 ng/g lipid weight (lw). ∑HBCDs ranged between 16 and 4397 ng/g lw, with a median value of 73 ng/g lw. Comparison with previous studies shows that PBDE and HBCD levels in Flemish eels have decreased rapidly between 2000 and 2008 at some particular sites, but also that alarming concentrations can still be found at industrialized hot spots.

Belpaire *et al*. (2011) analysed 30 polychlorinated biphenyl (PCB) congeners in pooled muscle tissue samples of eel collected from 48 sites in Flanders between 2000 and 2007. There was a large variation between individual sites (range 11–7752 ng/g wet weight (ww) for the sum of the ICES 7 PCBs), eels from the River Meuse basin (mean 1545 ng/g ww) being considerably more polluted than those from the River Scheldt (615) and IJzer (61) basins. Overall, PCB 153, PCB 138 and PCB 180 were the most prominent congeners; however PCB patterns varied between the monitored locations.

Analysis of the weight percentage of congeners demonstrates obvious differences in PCB composition between sites, indicating differential sources of pollution. It was shown that atmospheric fallout does not seem to be the main source of the PCB spread, but instead both local and upstream sources linked to industrial activities seem to be the main cause for PCB presence in Flanders. These results emphasize the potential significance of PCBs in the decline of the eel and support (inter)national eel management (e.g. by taking PCB levels into account when designing glass eel restocking programmes).

In order to gain insight in the current status of pollution by dioxins and related compounds, in Flanders, a baseline spatial analysis was conducted in (yellow) eel from 38 locations (Geeraerts *et al*., 2011). Spatial variation in the level of dioxin pollution might indicate areas of concern for these substances. Dioxin concentrations in eel varied considerably between sampling sites. Measured levels of dioxin-like PCBs (DL-PCBs) are much higher than those of the dioxins (PCDDs) and furans (PCDFs). The majority of Flemish eel from this study had levels of dioxins and DL-PCBs considered to be detrimental for their reproduction and therefore a possible contributing causal factor in the decline of the European eel. In almost half of the sampling sites show especially DL-PCB levels exceeding the European consumption level (with a factor 3 on average; Figure 5.9).

The European maximum limit for the sum of dioxins and dioxin-like PCBs (ΣWHO-PCDD/F+DL-PCBs TEQ) in muscle meat of eel and products thereof is expressed in toxicity equivalents. It is set on 12 pg TEQ g^{-1} fresh weight. The levels of this sum varied between 1.14 and 142 pg TEQ g^{-1}. In 42% of the sampling sites the limit was exceeded. The highest human exposure risk is through the consumption of fish, containing more contaminants than most other food products (Leonards *et al*., 2005). Health effects are expected through the long-term exposure of the most exposed part of the human population, i.e. recreational fishermen consuming self caught eel from contaminated locations. So, the Total Daily Intake standard (4 pg WHO TEQ per kg body weight per day (WHO, 2000) aims at lowering the intake of dioxins and related compounds in order to prevent tissue levels from reaching critical concentrations (Hoogenboom *et al*., 2001). Thus, in such cases, an advice to limit consumption of fish from such areas may be the most appropriate risk management option to decrease the intake of dioxins and related compounds (Geeraerts *et al*., 2011).

Morocco

- Heavy metals assessment

This work involves an assessment of the degree of heavy metal contamination (Pb, Cd and Cr) in liver, gills and muscle of eel (*Anguilla anguilla*) inhabiting two ecosystems along the Moroccan Atlantic coast: the Sebou and Loukkos estuaries (Figure 5.1). In these areas *A. anguilla* is widespread and a common predator at the top of the food chain. In this study, heavy metals were determined with flame atomic absorption spectrometry. Metal concentrations reveal high and widespread tissue contamination in eel caught from Sebou estuary and in Loukkos, with preferential accumulation in liver for Cd (chronic accumulation) and in gills for Cr and Pb (recent accumulation).

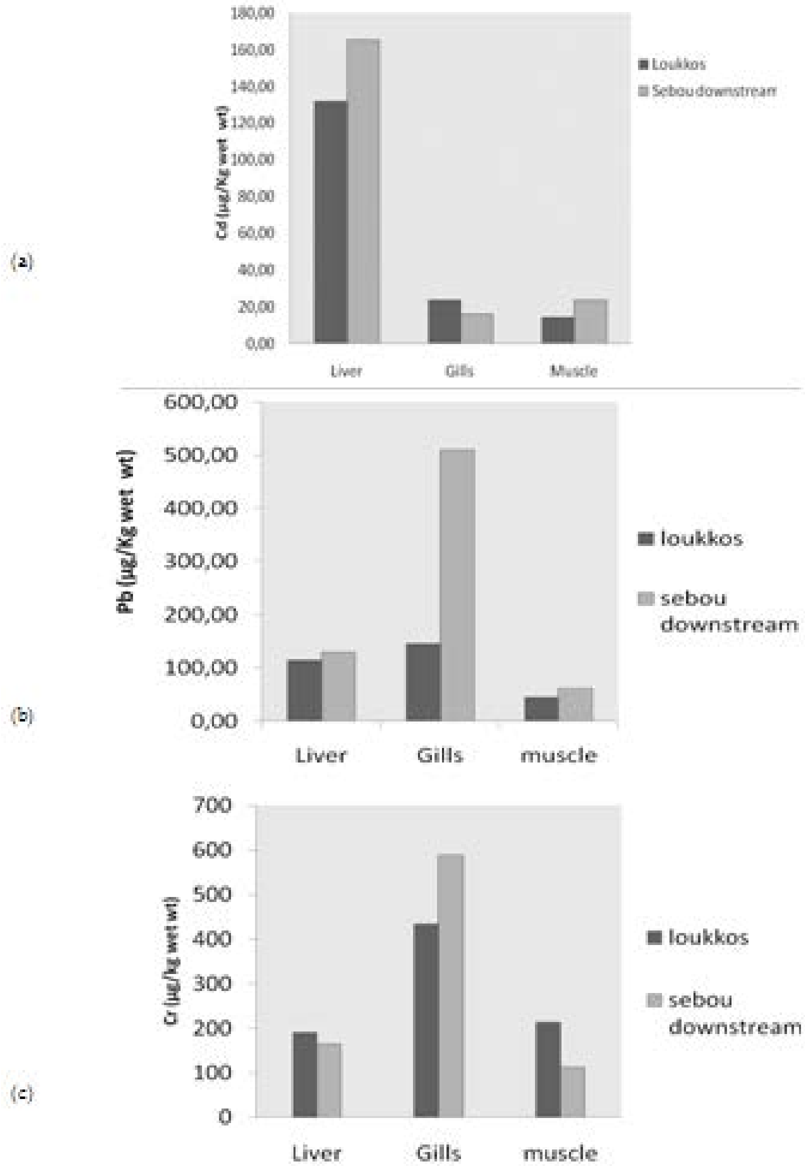


Figure 5-1. The concentrations of lead, cadmium and chromium (µg/g wet wt), in liver, gills and muscle of eels caught from Loukkos and Sebou rivers (Wariaghli *et al.*, 2010; unpublished data).

- PAH metabolites

This study investigated in the usefulness of biliary polycyclic aromatic hydrocarbon (PAHs) metabolites of European eel (*Anguilla anguilla*) as bio-indicator of pollution in Moroccan sites. Eels were collected at two locations (upstream and downstream) in the river Sebou and in the Loukkos estuary in October and November 2009. Biliary 1-Hydroxypyrene, 1-Hydroxyphenantrene and 3-hydroxybenzo[a]pyrene metabolites were measured in eel by HPLC analysis with fluorescence detection. Only 1-OH pyrene and 1-OH phynantrene were detected while 3-OH benzo[a]pyrene was not detected. No statistical differences between the sexes and ages for any of the PAH metabolites or biological parameters could be detected. Data from the three trawls were therefore pooled. These results showed significant differences between sebou upstream and Loukkos sites in mean concentration of 1-OH pyr and 1-OH phen metabolites ($p<0.05$, two sample t-tests), as well as between Sebou downstream and sebou upstream sites ($p<0.05$) which had similar concentrations of PAH metabolites.

Increasing levels of biliary PAH metabolites in eel suggest higher pollution levels downstream in the river Sebou and Loukkos.

Significant relationships were observed between the concentrations of 1-OH pyrene measured and biliverdin concentrations in the bile ($P=0.001$, $p<0.05$).

France

A campaign of PCB analysis in eel (among five other fish) was set up by the French Ministry of Agriculture in order to prioritize sectors of intervention to reduce risk for human food. In general, 290 sites in France were analysed. Results of the set of analyses are published recently (http://www.pollutions.eaufrance.fr/pcb/; http://www.pollutions.eaufrance.fr/pcb/resultats-xls.html http://pollutions.eaufrance.fr/Demo/Resultats_hydro.aspx). PCB concentrations levels are that high that commercial fisheries have been closed in many parts. Many rivers have been closed for the fisheries due to high levels of contamination. A website is giving all available information (including a map): http://www.robindesbois.org/PCB/PCB_peche/restrictions_peche.html).

Heavy metals (Cu, Cd, Zn, Pb, and Ag) were measured in soft tissue of yellow eels in the Adour estuary (Southwest France) and associated wetlands using the European eel (*Anguilla Anguilla*) (Tabouret *et al.*, 2011). Mercury (total Hg and MeHg) and organochlorinated compounds (7 PCBs, 11 OCPs) were analysed in muscle. Concentrations in muscle were in agreement with moderately contaminated environments in Europe and were below the norms fixed for eel consumption for heavy metals and OCPs. Concentrations in liver showed a higher pressure of Ag and Zn in the downstream estuary than in the freshwater sites whereas Cd was lower in the estuary probably because of the salinity influence. According to quality classes 100% of eels from freshwater sites indicated clean or slightly polluted environments. However, total mercury concentrations were close to the thresholds fixed by the European Community in the downstream estuary, whereas the sum of PCBs was found to be greatly above the fixed value. 100% of the individuals from the estuary were classified in quality classes corresponding to polluted or highly polluted sites. These first results highlight the need of further investigations focused on mercury and PCBs in this area taking the seasonal temperature influence into account for a better understanding of the pollution distribution and the possible threat on the eel population from the Adour basin.

Norway

Several reports (in Norwegian) including data of contaminant levels in eel have been made available (Næs *et al.*, 1999; Knutzen *et al.*, 1998, 1999, 2001; Julshamn and Frantzen, 2009).

Sampling is currently being done on eels from coastal Norway (Arne Duinker at NIFES). Results will be available in 2012.

See Figure 5.10 for dioxin levels in eels from a number of Norwegian fjords.

Portugal

Samples of eels caught from five brackish water systems (Aveiro Lagoon, Óbidos Lagoon, Tagus estuary, Santo André Lagoon and Mira estuary), were analysed for some trace metals (Hg, PB, Zn, Cu, Cd) revealing low contamination loads when compared to their European congeners (Passos, 2008; Neto *et al.*, 2011). The most contaminated eels were obtained from the Tagus estuary. However, in this estuary no clear relationships could be established between contaminant concentrations in eel tissues (liver and muscle) and in sediment, probably because of the general heterogeneity in environmental conditions (Neto *et al*., 2011).

A comparative study about the effects of pollution on glass and yellow eels from the estuaries of Minho, Lima and Douro rivers was developed by Gravato *et al*. (2010). Fulton condition index and several biomarkers indicated that eels from polluted estuaries showed a poorer health status than those from a reference estuary, and adverse effects became more pronounced after spending several years in polluted estuaries.

Spain

In 2009 a programme has been developed for toxicological analysis in the Mar Menor lagoon for the first time. In 2010 mercury, lead, cadmium and arsenic levels obtained where below the maximum limit for toxic waste indicated in Regulation 1881/2006. Liver, kidney and muscles of 16 eels were analysed, the total concentration of metals was below the maximum toxic residuals level in all the cases.

Table 5-2. Levels of toxic residuals in mg/kg wet weight found in eel analysed in 2010 in the Mar Menor Lagoon (data: Consejeria de Agricultura y Agua, Murcia).

	Pb	Cd	As	Hg
Liver	3.02	0.17	2.39	0.01
Kidney	8.92	0.69	1.13	0.1
Muscle	Only three individuals with a significant level (average value around 0.2).	Non-significant levels for all the individuals analysed.	2.06	0.01

UK

England & Wales

The Environment Agency provided samples from 35 eels caught in autumn of 2007 in the River Thames between Sunbury and Molesey (upstream of the tidal limit) and in the Thames estuary around Woolwich. These were analysed for 14 organochlorine pesticides and by-products and 41 PCB congeners, including the seven frequently detected congeners commonly used as indicators for PCB contamination (ICES7) (Jurgens, Johnson, Chaemfa, Jones and Hughes, pers. comm. 2009). Most of the investigated chemicals were detectable in every one of the samples although they have all been banned or severely restricted many years ago. However, based on the measured chemicals, all the analysed eels would be considered safe to eat.

Northern Ireland

Data on fat levels and restricted information on contaminants have been provided for inclusion in the EEQD.

Italy

Ferrante *et al*. (2010) measured organochlorine compounds in muscle tissue of European eels from the Garigliano River in Campania (Italy); overall PCBs emerged as the most abundant pollutants, followed by DDTs, Dieldrin and HCB. Target PCBs, IUPAC nos. 118, 138, 153 and 180, were the dominant congeners accounting for 64.2% of total PCBs. Among OCPs, p,p′-DDE was detected in all eels, always with higher concentration levels than other OCPs; p,p0-DDT was frequently detected, about 93.3% of the sample. The high and statistically significant correlations between concentration and length as well as weight of eels suggest that the organochlorine compounds concentrations tend to increase with the size and consequently with the age. Concentrations of DDTs and PCBs detected were similar to those reported in studies in France, UK and Sweden. As regards toxicological risk for human health, in general OCPs residual levels were below the limits established for fish and aquatic products. Conversely, the concentrations of PCBs exceeded the limit set by the EU for terrestrial foods. The results imply that OCPs and PCBs are still important persistent chemical contaminants in Campania freshwaters, although their manufacture and use are banned or highly restricted.

5.2.1.2 Parasites and diseases

Norway

Norway provided information on the prevalences of *A. crassus* in 35 sites along the Baltic states (Figure 5.2). *A. crassus* was present in all case with prevalences between 33 and 93%. In 25 sites prevalence was above 60%. (C. Durif, pers. comm.).

Sweden

The prevalence of the swimbladder parasite *A. crassus* has been monitored in samples taken from commercial catches, in freshwater and coastal areas. The prevalence in yellow eel was generally lower in marine areas along the west coast, going up to 6% in Skagerrak and 13% in the southern Kattegat, while more than 50% of the yellow eels had parasites in both Baltic areas. Silver eel were less infected in general, and differences between sites were smaller. In inland lakes, prevalence was generally much higher (79–94%), although only 27% of the eels in Lake Hjälmaren were infected (Figure 5.3).

Time-trends for the prevalence of *A. crassus* from eels in two lakes are presented below (Figures 5.4 and 5.5)

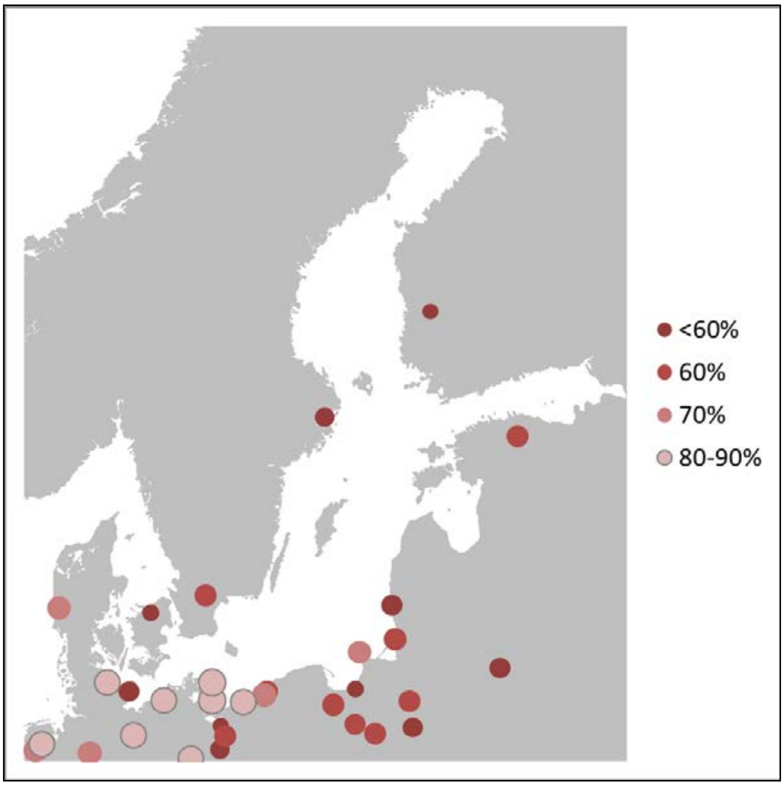


Figure 5-2. Prevalences of *A. crassus* in 35 sites along the Baltic States (Data: C. Durif).

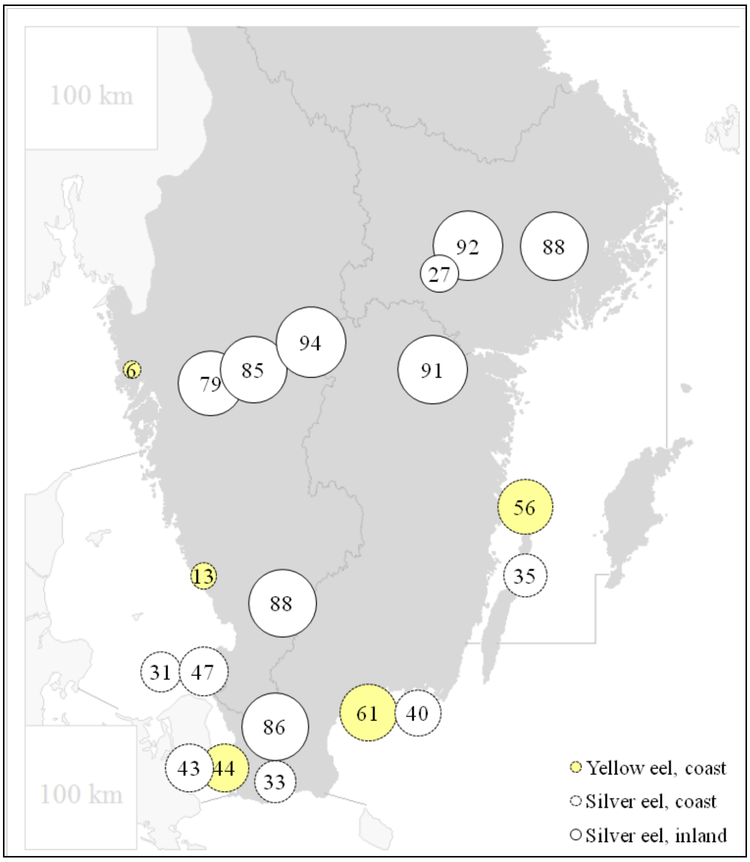


Figure 5-3. Prevalence (%) of the swimbladder parasite *A. crassus* in yellow and silver eel, in the 2000s in Sweden (Swedish CR).

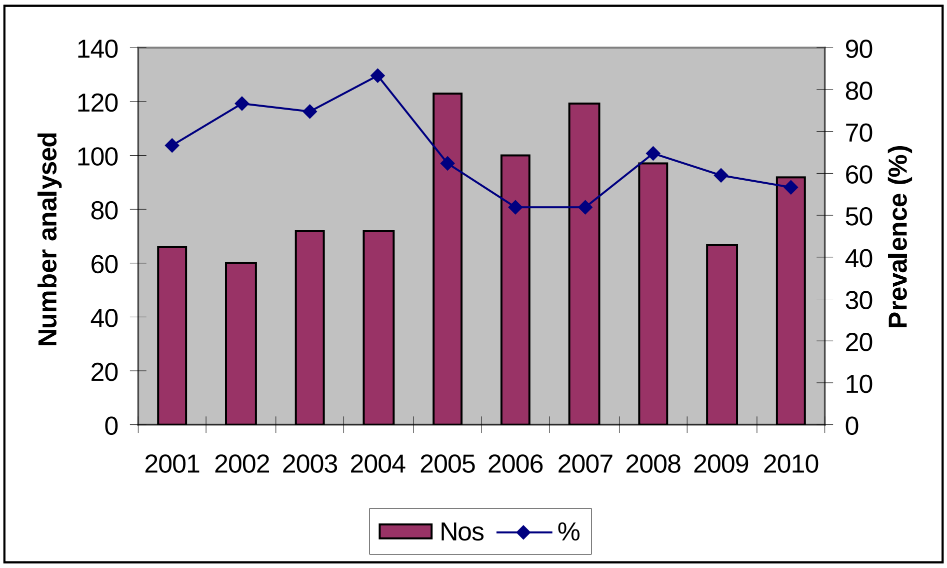


Figure 5-4. ***A. crassus*** **from eels in eels from Lake Mälaren (Swedish CR).**

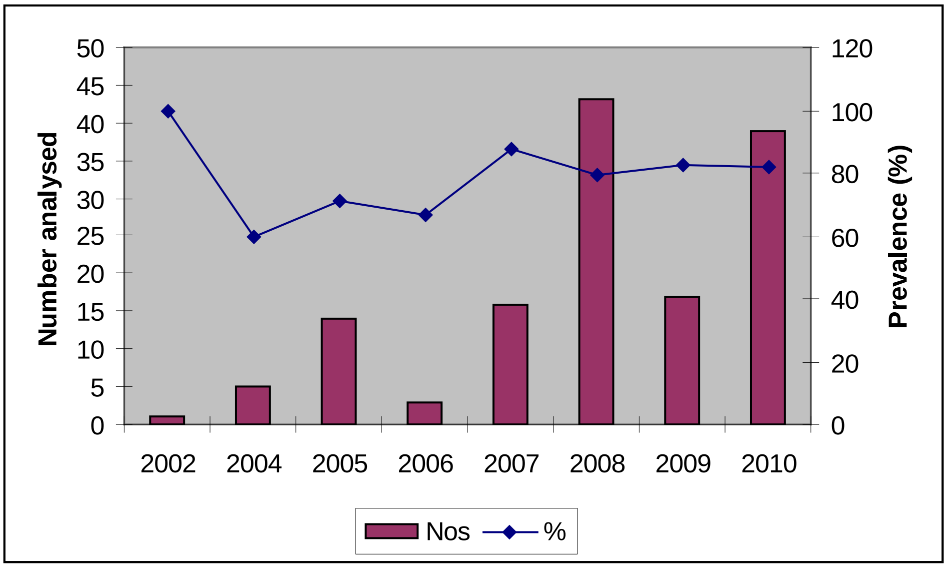


Figure 5-5. ***A. crassus*** **from eels in eels from Lake Ymsen (Swedish CR).**

Latvia

Latvia reports data on the presence *A. crassus* for inclusion in the EEQD.

Netherlands

In 2009, market sampling was initiated for eels captured in fisheries on Lake Ijsselmeer, Friesland and Rivers; no new locations were sampled in 2010. In 2011 the market sampling will be conducted throughout the whole country.

At the Fish and Shellfish Diseases Laboratory of the Central Veterinary Institute of Wageningen UR, in 2010–2011 so far two groups of diseased wild eels (juvenile to adult) were submitted for diagnosis. In August 2010, wild yellow and silver eels from a lake in Friesland (N-Netherlands) showed severe clinical signs: apart from many gill worms *(Pseudodactylogyrus)*, the eels had some cestodes in their gut, and some

A.crassus in their swimbladders. The disease was however caused by two viruses: AngHV-1 (HVA) and EVEX virus, with a bacterial infection by *Edwardsiella tarda*.

In June 2011, wild yellow eels from the Noordzeekanaal had some *Trichodina* as ectoparasite, *Acanthocephalus* in the gut, and *A.crassus* in their swimbladder, not in large numbers, and virus isolation of these eels was negative. (data: Olga Haenen and Marc Engelsma, pers.comm.).

Parasites

A. crassus was introduced in wild stocks of European eels in The Netherlands early 1980s, from SE-Asia. Wild eels showed high prevalence and intensities (no. of parasites per eel), and an acute reaction of the swimbladder by often with severe fibrosis (Banning and Haenen, 1990) raising concerns on the ability of the eels to reach the spawning grounds (Banning and Haenen, 1990; thesis Haenen, 1995). In the 1990s the prevalence decreased as did the severity of pathology, possibly reaching an equilibrium level.

Borgsteede *et al*. (1999) have described the parasitofauna of 361 wild eels of 17–73 grammes in Volkerak, Marker- en IJsselmeer: Various parasites were found, predominantly *Myxidium* sp. (33%), *Pseudodactylogyrus anguillae* (30%), and *Acanthocephalus clavula* (49%). In 2004–2005 Haenen *et al*. (2010) diagnosed 98 wild silver eels from the lower River Rhine, River Merwede, and the IJsselmeer for pathogens and disease: A quarter of the eels had ectoparasites, mostly *Trichodina* species, *Ichthyophthirius multifiliis, Ichthyobodo* species, *Glosattella* species, *Dermocystidium species*(eencelligen), and *Dactylogyrus* species. A quarter also had gut parasites, like cestodes (*Proteocephalus* species) and *Acanthocephalan* sp. Approximately three quarters had *A.crassus* in their swimbladder, with an intensity of five parasites per swimbladder.

Bacteria

In March and April 1997 seven cases and in June 1997 another case of 'red spot disease' were diagnosed in groups of diseased glass eels *Anguilla anguilla*, originating in Southwestern France and Northern Portugal. In all eight cases *Pseudomonas anguilliseptica* Wakabayashi and Egusa (1972), was isolated. The mortalities varied from lower than 5 to 20% in total, within 2–3 weeks. The isolates were sensitive for a list of antibiotics. After the water temperature was raised to 26–27°C, mortalities stopped (Haenen and Davidse, 2001). In wild silver eels, apart from some secondary skin inflammations, some cases of *Aeromonas hydrophila* and *Aer.sobria* were seen (Haenen *et al*., 2010). In hot summers, eels from rivers with a low water level once had a severe *Edwardsiella tarda* infection.

Viruses

Since 1999, both AngHV-1 (HVA, herpesvirus anguillae) and EVEX (Eel Virus European X) virus have been found in wild eels in The Netherlands, but not yet EVE (Eel Virus European, also known as IPNV type Ab or VR299). From silver eels from Lake Grevelingen, EVEX and AngHV-1 were isolated, and AngHV_1 was also found in silver eels of Lauwersmeer (Van Ginneken *et al*., 2005, 2004). In silver eels from the lower River Rhine/Merwede AngHV-1 was detected in 44% of 92 eels, without the eels showing disease (Haenen *et al*., 2010). It is however known, that AngHV-1 may cause disease, when eels are stressed, at ambient water temperatures for the virus. Therefore, it was hypothetized, that AngHV-1 may be a factor in the decline of silver eels, carrying the virus, during their migration to the spawning grounds, when they are stressed and swim at ambient water temperatures for exposition of the viral disease (Haenen *et al*., 2010).

In general, some parasites and the viruses are worrisome in the wild eel. The contact with eel farms should be avoided, as EVE might be introduced into wild eels from positive eel farms in The Netherlands.

France

New data on a study of *A. crassus* will be available in next future.

Ireland

All eels captured in the eel specific fykenet surveys and in the WFD surveys that are sacrificed for age determination will be sexed and examined for parasites.

Parasite data indicated the *A. crassus* was continuing to spread.

Norway

In 2009, silver eel were collected in the River Halselva in Northern Norway (70°N). None of these were infected by *A. crassus* (Davidsen *et al*., 2011). The parasite has previously been recorded in Southern Norway, as far north as River Imsa (58°N).

Estonia

There are no routine programmes monitoring parasites and pathogens of eel in Estonia, except special investigations in the end of 1990s, 2002 and 2008–2009.

Eel fishery in Estonian inland waters depends entirely on the stocking of glass eels or pre-grown (farmed) eels. Via importation of live eels of 20–30 cm length the non-indigenous swimbladder nematode *A. crassus* was probably introduced via Germany into Lake Vortsjarv in 1988, and has since spread to many inland waters of Estonia. In 1992, the parasite was found in eel caught from Lake Vortsjarv. Between 1992 and 2002 and additionally in 2008, in total 870 eels were analysed from Lake Vortsjarv (270 km^2) and in 2008, 63 eels from three small lakes for adult *A. crassus*. The aim of the study was to obtain information on the variation of *A. crassus* infection in eels in Estonian lakes, to determine the temporal dynamics of prevalence and intensity of infection, and to establish a relationship between the length of host and intensity of infection in the eels in Lake Vortsjarv. There appeared to be a pronounced variation in prevalences of infected eels (from 3.7% to 100%) between the four investigated lakes. However, in Lake Vortsjarv, the prevalence of adult *A. crassus* infection remained stable (mean about 65%) for many years. The average number of nematode per infected eel (mean intensity) ranged from 12.6 ± 2.5 in 1993 to 4.0 ± 0.6 in 1999 in Lake Vortsjarv, while it was significantly higher (P <0.0001) in the period 1992–1998 compared to 1999–2002 and 2008. The mean number of parasites per swimbladder was not related to eel length and no statistical difference was found in the condition factor of infected and non-infected eels. Although under normal environmental conditions *A. crassus* has not caused serious disease problems to eels in the study area, high intensity of parasite infection may contribute to eel kills due to oxygen deficiency in winter under the ice in Lake Vortsjarv (Kangur *et al*., 2010).

Portugal

No national programme to monitor parasites or pathogens. In a study conducted in 2008 in five brackish water systems (Aveiro Lagoon, Óbidos lagoon, Tagus estuary, Santo André Lagoon and Mira estuary) it was concluded that *A. crassus* was spread in all the surveyed systems except in Óbidos lagoon, which was probably related to the higher salinity observed in this lagoon, similarly to what happens in one sampling site (Barreiro) (Neto *et al.*, 2010) located in the lower part of the Tagus estuary. Prevalence values ranged from 0 to 100 % and intensity values ranging from 0.4 to 5.8 (unpublished data). More recently, within the DCF programme, the parasite was found in the swimbladder of seven among the 404 eels examined for the Óbidos Lagoon. The low prevalence found (1.73%) reinforces the idea that the infection rate is very low in areas with higher salinity, as it is the case in this lagoon. The presence of the parasite had already been reported for the River Minho (Antunes, 1999) and River Mondego (Domingos, 2003), which suggests the parasite is probably widespread in Portugal. The map (Figure 5.6) shows the locations where this parasite has been reported so far.

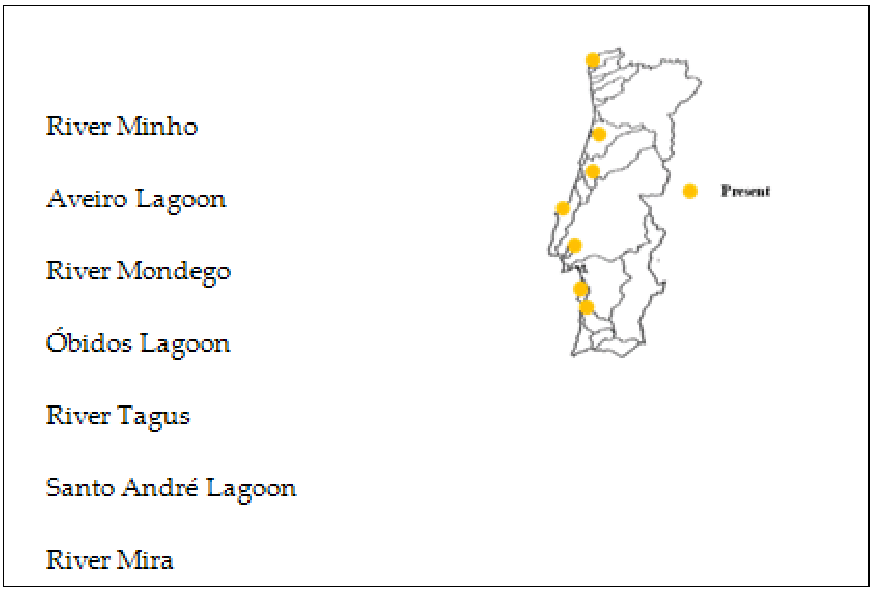


Figure 5-6. Locations in Portugal where the presence of *Anguillicoloides* has been reported so far (Portugal CR).

Spain

New data were reported on parasites and pathogens in Spanish Mediterranean basins and Asturias. These studies reported detailed data on life stages L3 and L4, pre-adult and adult stages, but here the data are presented as total load of parasites in individual eels for studies in Mediterranean region and pre-adult and adult stages for Asturias rivers.

There is a new study in the Mar Menor Lagoon (Murcia) where the prevalence of *A.crassus* has been analysed in 2010 and 2011, resulting in a very low prevalence of this parasite (2.3% in 168 eels analysed). A total of 109 eels were collected between November 2008 and March 2009 and adult worms were recovered from the swimbladders of infected eels (Martinez-Carrasco *et al.*, 2011). The detected prevalence in this case was 7.34%.

The prevalence of other infectious diseases has been reported for the Albufera lake in El Palmar (C. Valenciana) (Bandin, pers. comm. 2010; Esteve and Alcaide, 2010; Muñoz *et al.*, 2009.) and in the Mar Menor Lagoon (Muñoz *et al.*, 2009).

Infection by the parasitic nematode *A. crassus* in a wild riverine stock of European eel, *A.anguilla* (L.), in a near pristine river was investigated. This study highlighted the presence of the parasite and completion of the whole life cycle in eels from the Rıo Esva. Infection levels by *A. crassus* were high at three sites between the mid river to the estuary and also varied among seasons. Condition of eels was lower at upstream sites compared with downstream locations. Although high-quality, environmental conditions in the Rıo Esva may buffer the effects of *A. crassus* on eels, potential impacts and limiting factors for the parasite are discussed. (Costa-Dias *et al*., 2010).

In a study on the *Edwardsiella tarda* reservoirs in Albufera lake, as well as Edwardsiellosis distribution on eels regarding of water physico-chemical parameters, the bacteria was recovered only from the 7,41% water samples and its isolation was related with a high water temperature >20ºC. In addition, percentages of *E.tarda*-positive fish (40–84%) during the warm period (water temperature >20ºC) were also significantly high in comparison with those detected during the cold period (<7.4%). Moreover this 2008 study again remarks that Edwardsiellosis disease is more prevalent in younger eels (25–48 cm) than in silver ones.

England & Wales

A. crassus is now considered ubiquitous throughout England and Wales (Nigel Hewlett, Environment Agency National Fisheries Laboratory, pers. comm.). There is no routine and/or coordinated monitoring of the incidence of parasites or pathogens in eels sampled in England and Wales. Those applying for a licence to move or stock eels in England and Wales must submit a health check of a sample of the fish, which includes a check on parasites and pathogens, but there are few such applications. Eel herpesvirus (HVA) was detected from mortality samples in 2009 and 2010 at:

1) Cromwell Carp Fishery, Nottinghamshire (NGR: SK7968262232)–Following an eel specific mortality at the fishery in 2009, one yellow eel (two yellow eels sampled in total; 755–824 mm) tested positive for the virus *Herpesvirus anguillae*. A moderate infection of adult *A. crassus* (nine nematodes present) was also detected in the swimbladder of one eel.

2) Goltho Lake Fishery, Lincolnshire (NGR: TF1165377083)–Following an eel specific mortality at the fishery in 2010, both dead and live yellow eels were examined (485–883 mm). These tested positive for HVA. Low level infections of adult *A. crassus* were recorded in the swimbladders of the three live eels (eel 1 = 2; eel 2 = 2; eel 3 = 1 adult *A. crassus*). Heavy infections of a larval nematode, *Daniconema anguillae,* were also noted in the fins.

3) An eel specific mortality was observed at Cliffe Pools (near NGR: TQ71977730) during June 2010. The incident was short-lived and the EA were unable to get a sample to examine.

No larval *A. crassus* present and no significant parasite burdens or signs of clinical disease were recorded in glass eels sourced from the River Severn (UK Glass Eels Ltd), one each year dating back to 2009.

Northern Ireland

NI Eastern RBD

A. crassus has been recorded from eels examined in this RBD for the first time in 2010 (N = 52, prevalence 30% mean intensity <one worm per infected eel).

NI Northwestern International RBD

A. crassus was first recorded in the swimbladders of eels in Ireland during an extensive fykenet survey of the Erne system in July 1998. A new record for *A. crassus* in a separate catchment within this RBD (the Foyle) was found in 2008 in one eel.

NI Neagh-Bann RBD

A. crassus was found in Lough Neagh yellow and silver eels for the first time in 2003, and its spread has been monitored via the analysis of a total of 2203 yellow and 800 silver eels from 2003 to 2010. By 2005 prevalence had reached a peak of 93% of yellow eels and 100% of silver eels. But by 2008 the prevalence of *A. crassus* had fallen in both yellow and silver eels and was recorded as 67.3% and 86%, respectively, whilst in 2009 it had fallen to 53.6 and 81%, respectively. In 2010 these infection parameters continued to fall for yellow eels, reaching 48.8% however prevalence in silver eels had risen slightly to 80.7%.

Denmark

The swimbladder parasite *A. crassus* is widely distributed throughout both brackish and freshwaters in Denmark. Monitoring of the parasite takes place on a yearly basis at three locations starting in 1987 and 1988. Sampling takes place during autumn. At Isefjord sampling failed in 2010 due to early ice cover preventing fisheries activity. The number of *A. crassus* infected eels (prevalence) is relatively constant at all three locations.

Table 5-3. ***A. crassus*** **monitoring data for 2010.**

Location	Salinity ppt	Coordinates	Year	Total	Infected	Prevalence	Intensity
				N	n	%	n
Arresø	0	55.59N;11.57E	2010	100	60	60	4.1333
Isefjord	18	55.50N;11.50E	2010	0	-	-	-
Ringk. Fj	5–10	55.55N;08.20E	2010	104	67	64.4	5.2089

Morocco

Epidemiological data of the swimbladder nematode *A. crassus* in Moroccan rivers was initially described by El Hilali *et al.* (1996); Lachheb (1997); Kheyyali *et al.* (1999); El Hilali *et al*. (2005); Wariaghli (2006); Zouhir (2006) and Loukili and Belghyti (2007).

The way of introduction of *A. crassus* is still unknown, because Morocco has never imported live eels but only exports them. This parasite is still spreading over all Moroccan eel fishing areas. The prevalence of the swimbladder *A. crassus* is still spreading in Moroccan waters, but within sites there is a trend for stabilization or even a decrease in prevalence values. Figure 5.7, shows the mean of prevalence, intensity and abundance of eels (yellow and silver eels) caught in Sebou estuary between 2004 and 2009.

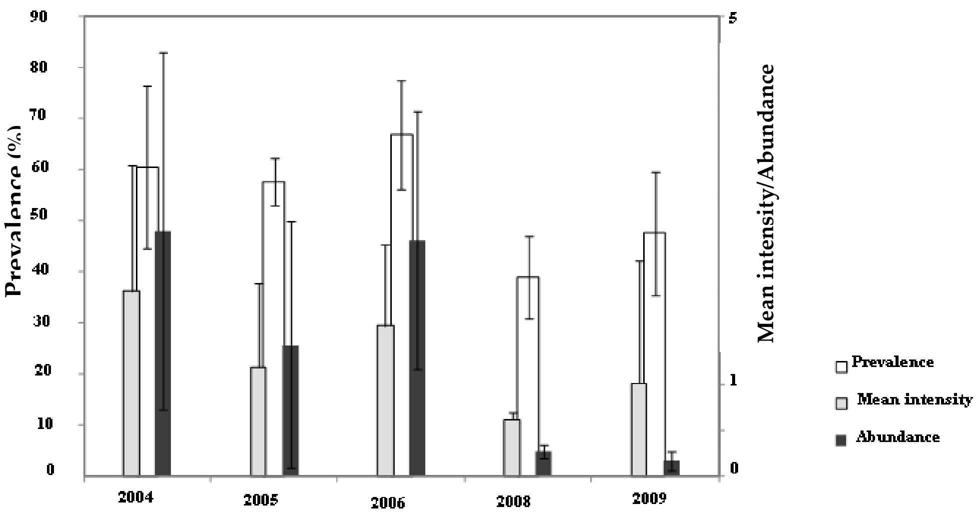


Figure 5-7. Prevalence (%), mean intensity ± SD and abundance ± SD of *A. crassus* for eels caught in Sebou estuary 2004–2009 (Morocco CR).

5.3 Advances in understanding processes related to the impact of contaminants and diseases on eel

5.3.1 Introduction

Several studies have recently been initiated to study the degree and the effects of pollution on the eel, resulting in an increasing quantity of information that demonstrates the negative impact of pollution on eel.

These advances in understanding the effects of contaminants on the eel have been reviewed recently by Geeraerts *et al.* (2010), Elie and Gerard (2009) and by WGEEL 2009 (ICES 2009). For example, there were reports on a negative correlation between TEQ levels in females and survival time in embryos (Palstra *et al.* (2006); a negative effect of cadmium on sexual maturation of female silver eels and on spawning migration by altering the lipid accumulation process (Pierron *et al.* (2008); and on the alteration of transoceanic spawning migration through PCBs (van Ginneken *et al.* (2009)). But this information has been thoroughly reviewed during earlier WGEel session (ICES 2009).

Many gaps in our knowledge remain, especially concerning the impacts (dose-effect relationships) of contaminants and diseases on migration and reproduction success of the European eel. Some new knowledge is awaited from ongoing international projects such as Eeliad (www.eeliad.com) and Pro-eel (http://www.pro-eel.eu).

5.3.1.1 Diseases

A 21 year study of *A. crassus* took place in Upper Lake Constance (ULC; Bernies *et al.*, 2011). The study serves as an invasion record from invasion to steady state. It includes two extensive surveys, one in 1991 during the initial parasite invasion phase and the second in 2006 when the infection was well established. *A. crassus* was first recorded in *A. anguilla* in ULC in 1989. Prevalence reached 60% in 1992 and remained at this level until 2007. In 2008, prevalence decreased to 48%. Infection intensity peaked in 1993 at a mean value of 16 adult parasites per host fish. Around 90% of all *A. anguilla* examined displayed swimbladder lesions, with a significant trend to increasing severity over time. Moreover, heavy swimbladder lesions were seen in ca. 10% of A. anguilla ready to migrate to their spawning habitat. The growth and survival rates of *A. anguilla* during their continental phase were not noticeably altered in infected fish, but damage to the swimbladder probably impairs migration potential and thus the subsequent breeding success of the oceanic phase.

A study from Clevestam *et al*. (2011) results in the hypothesis that a large proportion of female silver eel from the Baltic Sea catchment area will have inadequate or suboptimal energy reserves for successful migration and reproduction. For 378 female silver eels individual net fat reserves after migration and reproductive investments were calculated. The study concludes that a combination of body size and distance to the spawning area in the Sargasso Sea are crucial elements for migration success. An increase in the costs of migration due to heavy infection with *Anguillicoloides crassus* was also evaluated in an additional scenario showing that 26% of the eels had completely depleted all fat reserves.

The gross pathological and histopathological changes associated with parasitic infection in the European eel were investigated by Abulmonem *et al*. (2010). A total of 65 eels collected from three sampling localities in Eastern Delta, Egypt were examined over the period of January–May 2008. The fish were subjected to standard procedures for parasitological and pathological examinations. Overall, 22 (33.8%) of the 65 fish examined were found to have parasitic infections. The eels harboured a total of six parasite species; among them, *A. crassus* was the most prevalent species (10.7%), followed by the Monogenea *Pseudodactylogyrus anguillae* (7.7%) and *Dactylogyrus* species (6.1%), the ciliate *Trichodinella epizootica* (4.6%), the Myxozoa *Myxidium giardi* (3.1%), and the cestode *Proteocephalus macrocephalus* (1.5%). Affected fish showed varying levels of tissue damage and pathological alterations including mild to severe degenerative, necrotic, and inflammatory changes in the affected organs.

It is widely assumed that the likelihood of invasion decreases with increased species richness in the recipient community. However, the invasion paradox supports a negative and a positive relationship between native biodiversity and the success of an invader. Here, we show that for a host–parasite system (*A. anguilla* as host and *A. crassus* as parasitic invader), invasion increases with native micro- and macroparasitic species richness. In fact, about 30% of the *A. crassus* intensity in eels could be explained by the number of both micro- and macroparasite species. It was recommended that researchers incorporate native parasite richness as a risk factor in epidemiological models of *A. crassus* (Martínez-Carrasco *et al*., 2011).

A viral eel study was conducted by Haenen *et al*. (2009). This study shows that EVEX, EVE, and HVA are found at several geographic places in wild and farmed European eel. Transport of clinically healthy but virus-infected elvers and eels may cause introduction of eel viruses into the virus uninfected waters with eels, both fisheries and aquaculture. No serious mortalities are known in wild eel populations due to eel viruses, but wild diseased eels are difficult to trace, particularly migrating silver eels. Related to the fact that migrating silver eels are naturally stressed during their spawning migration, and as a result immunosuppressed, with the assumption, that they swim for some weeks in ambient water temperatures for virus infections, the eels might get viral disease in these waterbodies. To what extent this disease would occur and threaten the wild eel population and recruitment depends on the immune status of the eels, the water temperature, and the pathogenicity of the particular virus strain and impacts on the spawning stock are still not known and will be difficult to estimate (Haenen *et al*., 2009).

To investigate diseases of European silver eels *Anguilla anguilla* in the Netherlands, in November–December 2004 twelve silver eels, and in August–December 2005 80 eels were caught in downstream parts (rivers) of the River Rhine and in Lake Ijsselmeer (Haenen *et al*., 2010). In the pilot study of 2004 in the River Rhine and Lake IJsselmeer respectively, most eels showed aspecific fin haemorrhages, some had ectoparasites, nearly none had parasites in the intestine, half of the groups had *A. crassus* in their swimbladder, only few had *Trypanosoma* in their blood, and no primary virus or bacterial infections were found, although one eel from Lake IJsselmeer was positive for anguillid herpesvirus 1 (AngHV-1, former Herpesvirus anguillae, HVA) in the PCR test only. The blood of all six and 2/6 of the eels respectively was considered abnormal, and the eels had a proper condition. In 2005, in 50 eels from the River Rhine and 30 from Lake IJsselmeer respectively, again aspecific fin haemorrhages were often seen, some of the eels had ectoparasites and parasites in the intestine, most eels had *A. crassus* in their swimbladder, 32% and 53% had Trypanosomas in their blood, from 44% and 13% of the eels AngHV-1 was isolated, and 44% and 27% were tested positive by PCR, with a peak in August, 10% of both groups of eels had an internal bacterial infection, mostly due to *Aeromonas* spp. The blood of about half of the eel was considered abnormal, but the eels had a proper condition. It was concluded, that the silver eels of this study had a proper Fulton condition factor (values 2.00–2.26), with aspecific fin haemorrhages, often were *Trypanosoma*, A. *crassus* and AngHV-1-infected, dependent on the season, and often showed an abnormal haematology. *A. crassus* causes injuries and is a chronic stress factor, more than the other parasites, which were mostly found less prevalent. In fact, lymphocytosis was directly related to *A. crassus* infection. Stress and injuries by *A. crassus* might induce disease through the presence of virus (AngHV-1), relevant in the health status of the silver eels during their spawning migration if ambient water temperatures would enhance a clinical infection of AngHV-1 disease. Moreover this virus might potentially decrease the survival of the silver eels by itself, because spawning migration to the Sargasso Sea takes wild eels to temperate/tropical areas in which the clinical infection by AngHV-1 is surely enhanced (Haenen *et al*., 2010).

5.3.1.2 Contaminants

A preliminary study of PCB contamination carried out on different fish from the Gironde estuary (southwest France) showed a relatively high level of contamination of eel muscles. In order to characterize the contamination level of PCBs and PBDEs (PolyBrominated Diphenyl-Ethers) more than 240 eels were collected during the years 2004–2005, from glass eels to silver eels. Individuals were grouped according to length and localization sites. The results showed a low contamination level of glass eels: respectively 28 ± 11 ng g_1 dw for PCBs and 5 ± 3 ng g_1 dw for PBDEs. The contamination level increased from glass eels to silver eels up to 3399 ng g_1 dw of PCBs for the most contaminated silver eel. These results raise concerns for local people who regularly eat eels caught in the Gironde estuary (Tapie *et al*., 2011).

5.3.1.3 New research projects started

In Belgium, a FNRS–FRFC study has been started in the Walloon Region to study the effects of hazardous substances on the nervous system of the eel ("**Integrated study of the impacts of pollutants on the nervous system of the European eel, *Anguilla anguilla***" by Jean-François Rees, Jean-Pierre Thomé, Cathy Debier, Marc Ylieff, Patrick Kestemont, Frédéric Silvestre). This study postulates that pollutants found to the European eel exercise effects on the nervous system of fish and affect in particular their olfaction. The objective thus is to study *in vitro* and *in vivo* the effects of sublethal concentrations of pollutants on the nervous system of the European eel by integrating several additional approaches (proteomic, transcriptomic, biochemical and behavioural) to bind the cellular and behavioural effects. The results will allow verification of the possibility that the neurological effects of pollutants can play a significant role in the regression of the European eel populations.

Another new research project named IMMORTEEL started recently (IMpacts of Metallic and ORganic contaminations of the systems Gironde and St Laurent on two threatened species, the European and American eels.). A main objective of this project is to test the possibility of using recent tools of molecular biology, DNA microarray as well as markers of genetic polymorphism (specifically Single Nucleotide Polymorphism (SNPs)), to detect, asses and characterize the effects of chronic/environmental exposure to metal and organic contaminants mixtures on natural fish populations. This project will be also developed in the perspective to acquire new insights into the mechanisms of toxicity of those contaminants prevailing in situ in aquatic organisms. For this research, using the new generation (Titanium) of the 454 sequencing technology, we propose to develop such tools for non-model, but ecologically and economically relevant endangered fish species, the European and American eels (*A. anguilla* and *A. rostrata*, respectively).

5.4 Assessment of the quality of local eel stocks

Eel quality (contaminants, parasites and general condition indices) is being monitored in a number of European countries with a substantial eel population. The relevance of these quality data for the conservation of the eel stock, is dependent of linking quality indices to quantitative data on eel stocks in corresponding water sheds (e.g. in terms of production of silver eel and SSB). Ultimately, monitored "quality indices" should indicate potential effects on reproduction success.

Toxicity studies with other species, including fish, show clearly that above certain threshold levels negative effects at the individual level can occur.

A study on brown trout alludes to aspects of maternal transfer of contaminants to eggs and fry in teleost fish. The data suggest a decoupling between lipid content and organohalogen concentrations for anadromous brown trout. Calculations of overall mass balance of the PHCs using only the transfer of lipid between the various life stages might not be an appropriate way. Within the study, the observed lipid and contaminant transfer between different life stages were not always identical: The concentrations of PCBs, DDTs, and PBDEs (ng/g lipid weight) were about 15 times lower in the eggs compared to the muscle of their mother on a lipid weight basis (e.g. 823 ng PCB/g lwvs.12 565 ng PCB/g lw, respectively) (Svendson *et al*., 2007).

Threshold levels may vary up to a factor of 100 between species, and there are at the moment insufficient data indicating the sensitivity of eel. Therefore, as of yet, there is no definite answer on how, and to what extent, reproduction success is affected by eel quality.

However, when scientific data become available, showing the effect levels of contaminants on eel reproduction, the EEQD database will be a valuable tool to support incorporation of the quality of eel in the assessment of effective spawning–stock biomass estimates.

- On basis of the above, the eel quality database should be updated continuously with relevant data from Europe, covering all the major eel production areas.

- Data from stock assessment need to be coupled with eel quality data.
- Scientific data on effects of contaminants/diseases on eel reproduction could then be used to further help the development of eel quality classes, and to categorize the European eel stocks into high and low potential spawners.

Assuming that a relationship between eel quality and reproduction success will be obtained, for management purposes it is essential to understand the quality of eels present in European RBDs in order to evaluate the reproductive potential of the silver eels leaving those systems and to compare eel quality between systems. Due to the absence of accurate threshold levels of contaminants and parasites, comparing the effects of different 'quality' pressures might not be appropriate. Given the need to obtain this estimate of overall quality, WGEEL 2010 began the development of an Eel Quality Index. In our approach we used only a set of the apparently most important pressure parameters, and where sufficient data were available in the EEQD.

Quality classes with boundaries were available in literature for about 30 contaminants (Belpaire and Goemans, 2007). We selected Sum ICES 7 PCBs, Sum DDTs and Cadmium as important parameters (as suggested from the review by Geeraerts and Belpaire, 2010; Pierron *et al.*, 2008, van Ginneken *et al.*, 2009). With respect to diseases, *Anguillicoloides* and viruses seem also to have an impact (Palstra *et al.*, 2007; van Ginneken *et al.*, 2005). See WGEel 2010 for description of methodology.

In a new paper (Lefebvre *et al.*, 2011) indicated two alternative indices for quantifying the gross pathology of the swimbladder of eels infected with the nematode *A. crassus*. The Length Ratio Index (LRI), performed better than the Swimbladder Degenerative Index (SDI), in three of four predefined criteria of decision. The authors recommend the use of the LRI as it can be recorded on live specimens with radio-imagery (non invasive method).

Table 5-4. Boundary values of the quality classes for a series of selected contaminants (From WGEEL, 2010).

Class EQI value	Not impacted 4	Slightly impacted 3	Impacted 2	Strongly impacted 1
Cadmium (ng/g BW)	<5	5–<12,6	12,6–<31,7	≥31,7
Sum PCBs (ng/g BW)	<73	73–<183	183–<460	≥460
Sum DDTs (ng/g BW)	<40	40–<101	101–<254	≥254
A. crassus	not infected	/	/	Infected
EVEX	not present	/	/	Present
HERPES Virus	not present	/	/	Present

The classification above is based on the data available. In some countries, data in eel is monitored in a specific size class (e.g. The Netherlands 30–40 cm) whereas data from other countries is based on the eels that were caught, without size restrictions. As contaminant levels (especially of organic contaminants) tend to rise with length (that is, fat content) (Hoogenboom *et al.*, 2007) monitoring of different size classes between countries may affect the outcome of the classification. This illustrates the need for standardization and harmonization of the methods used during eel quality assessments.

Fat content, and related to condition indices, is considered an important parameter, as a minimum amount of fat is required for the migration and production of eggs. There is a strong relation between maturation stage of the eel and fat levels. Yellow eel have generally lower fat contents which increases with age maturation eel. Maturation of male eels starts at lower lengths than that of females, so incorporating fat levels in the quality rating requires both knowledge of the maturation status of the monitored eels as well as sex.

Threshold values for dioxins?

The difficulty of setting thresholds for a certain contaminant is illustrated e.g. for dioxins. Dioxins are highly reprotoxic and an increased number of data are becoming available (overview of current levels in Figures 5.8–5.10). Furthermore an EU regulation has been issued setting maximum allowed levels in foodstuffs. It is evident that there is a very large range of total-TEQ values that impair reproduction. Elonen *et al.* (1998) compared toxicity of TCDD (dioxin)to seven freshwater species. NOECs, LOEC as low as 175–270 ng/kg for lake herring, 424–2000 zebrafish resp. The LC50 (eggs) ranged from 539 to 2610 ng/kg. Lake trout was found more sensitive (Spitsbergen *et al.*, 1991) at 40 ng TCDD in eggs significant mortality occurs. The LOELand NOEC of these studies do vary significantly, this shows clearly that between species there can be very large differences in sensitivity.

Steevens *et al.* (2005) suggested tissue residue-based toxicity benchmarks distributions rather than single-point estimates (ICES 2010).

Palstra *et al.* (2006) suggested that dioxin-like contaminants (including some PCBs) are capable of "devastating effects" on the development and survival of eel embryos. They observed a correlation between embryo survival time and TEQ levels in the gonads implying TEQ-induced teratogenic effects. Palstra *et al.* (2006) reported these disrupting effects occurring at levels below 4 pg WHO1998 TEQ kg^{-1} gonad. While the lipid reserves are depleted during migration, contaminants are released into the blood (and may damage reproductive organs and affect embryogenesis (Geeraerts and Belpaire, 2010). This suggests that such contaminants may have contributed directly to the observed decline in populations (Van Ginneken *et al.*, 2009). However the data presented in Palstra *et al.*, 2006 need to be confirmed by further evidence through sound experimental set up.

Another factor of influence is the uncertainties in quantification of the *Parental transfer of contaminants.* Parental transfer of PCBs and other pollutants into eggs has been shown in many studies. A one to one transfer of organic contaminants like PCBs on lipid weight can be expected if one assumes that all parts of the fish are in equilibrium. However, in literature there is little consensus about the level, both higher and lower (Svendsen, 2007) than the 1:1 ratio have been reported. This indicates that eggs and muscle lipids may not always be in equilibrium or that other processes influence the composition. Next to this, selective transfer of lower chlorinated PCB congeners to eggs was observed with herbivore mudskippers by Nakata *et al.* (2002). Preliminary data with eggs and tissue of mother eel (PRO-EEL project) suggest that PCB incorporation may be on a one-to-one ratio on lipid base.

Walker *et al.* (1994) measured translocation of 2,3,7,8-TetraCDD from adult female trout to oocytes and assessed mortality in fry. If we compare the recent Belgian data (Geeraerts *et al.*, 2011; Figure 5.10) for 2,3,7,8-TetraCDD values in eel muscle and use their conversion factors for oocytes, this gives levels typically in the range of 0.01 to 0.15 pg g−1 in eggs (with one exception of 2.2 for one site (the Handzamevaart)), which is not high enough to induce mortality in trout fry (Walker *et al*., 1994).

In a Japanese study it was determined that about 20% of the dioxins in adult female crucian carp were transferred to the eggs (Kajiwara *et al*., 2007). Applying this conversion rate to eel, by calculating the mass of eggs which could be produced by using all available lipids through a conversion factor of 1.7 g eggs g−1 fat (as used in van Ginneken and van den Thillart, 2000), ΣPCDD/Fs+DL PCBs levels in eggs would range between 1.4 and 593 pg WHO1998 TEQ g− 1 (mean 42.0 pg WHO1998 TEQ g−1), which, compared to the Palstra *et al*. (2006) benchmark of 4 (2 pg) pgWHO1998 TEQ g−1, suggests that in 79% of the sites, levels are high enough to induce disrupting effects in eel eggs. As arguably, the semelparous eel will use a larger proportion of her body lipids to form eggs, compared to an iteroparous species such as the crucian carp, these data may be an underestimation (Geeraerts *et al*., 2011).

Considering the uncertainties associated with comparing with the aforementioned studies, sound conclusions on the impact of field levels of dioxin related compounds on the stock are, at the time being, difficult to draw, underlining the need for the set up of well designed dose–effect experimental studies (Geeraerts *et al*., 2011).

5.5 Fisheries closure as a human health measure due to contamination

Several MSs provided contaminant data recently measured in eels from their national waterbodies for inclusion in the EEQD. Several countries start to measure dioxin levels in order to compare the levels with the EU consumption limits, and to protect heath of eel consumers. Figure 5.8 shows the levels of dioxins in Germany (river Rhine and lake Constance) (Wahl *et al*., 2011), Figure 5.9 presents dioxin data in eels from Belgium (Geeraerts *et al*., 2011), while Figure 5.10 is comparing the dioxin data from several countries as available in the EEQD. As can be seen through the comparison of the levels with maximum consumption levels (maximum level PCDD/Fs =4 pg WHO_{1998} TEQ g^{-1} fresh weight and maximum level PCDD/Fs and DL-PCBs =12 pg WHO_{1998} TEQ g^{-1} fresh weight), eel levels do exceed the legal limits in a significant proportion of the cases.

Those high dioxin (but also PCB levels) have been the basis for a closure or a restriction in the eel (or fish in general) fisheries. During the last years (2010–2011) fisheries restrictions/bans have been issued for an increasing numbers of waterbodies.

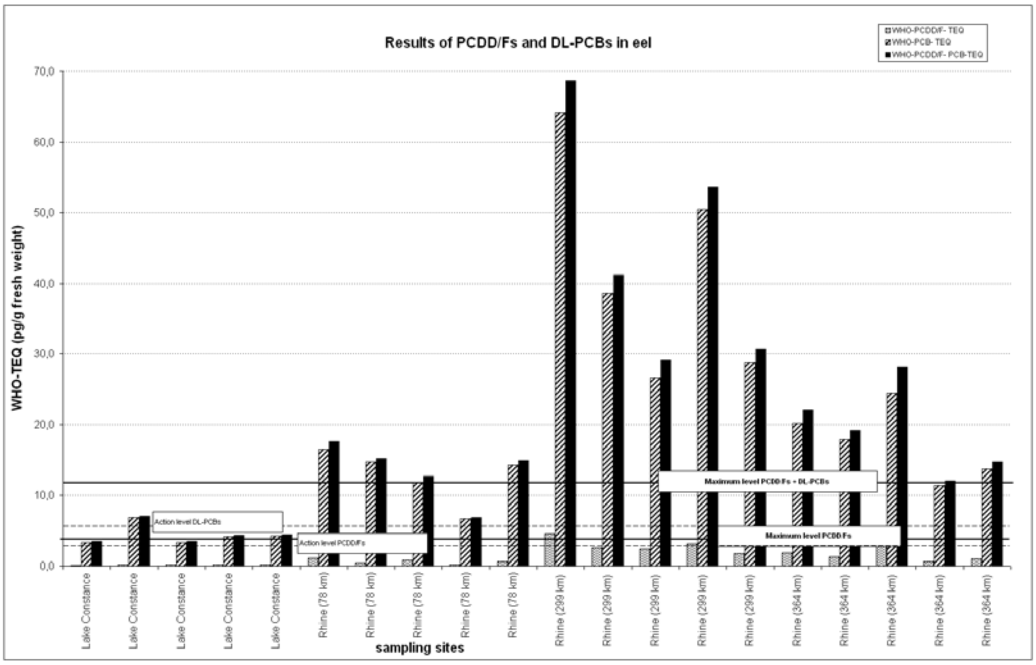


Figure 5-8. Levels of PCDD/Fs and dioxin-like PCBs in eel samples from lake Constance and from the river Rhine in Baden-Württemberg (Wahl *et al.*, 2011).

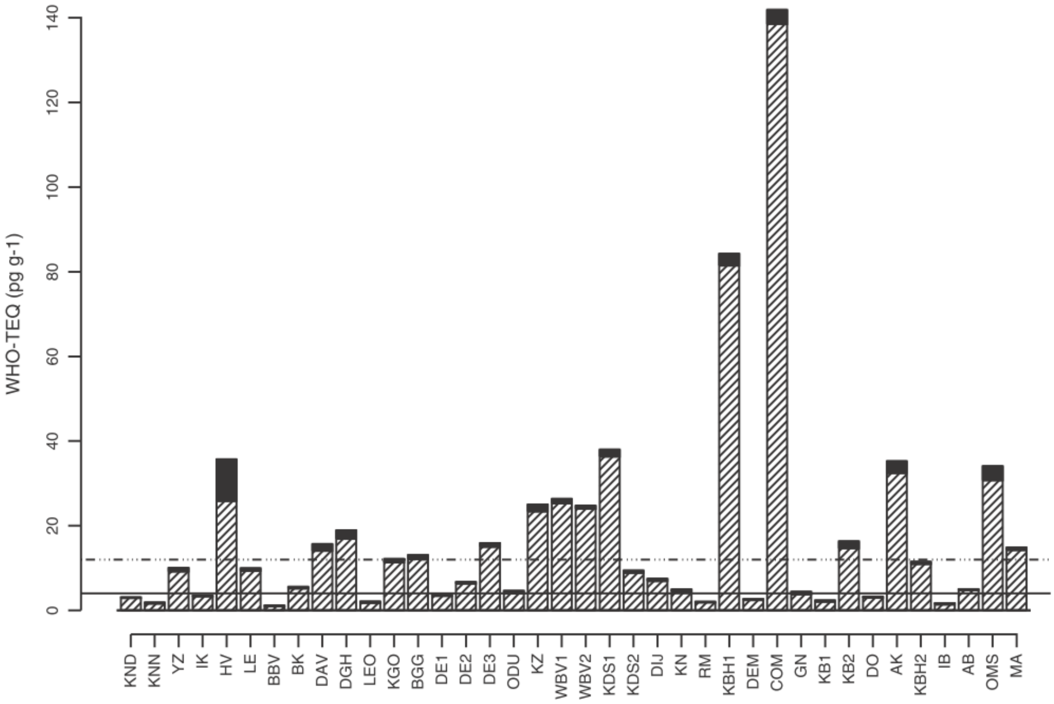


Figure 5-9. Concentrations of DL-PCBs-TEQ (white striped) and PCDD/Fs-TEQ (black) in eel muscle tissue from pool samples in Flanders. For comparison, the permitted maximum levels of the EC Regulation (EC, 2006) are drawn parallel to the X-axis: (—) maximum level PCDD/Fs =4 pg WHO_{1998} TEQ g^{-1} fresh weight, (–···–) maximum level PCDD/Fs and DL-PCBs =12 pg WHO_{1998} TEQ g^{-1} fresh weight (Geeraerts *et al.*, 2011).

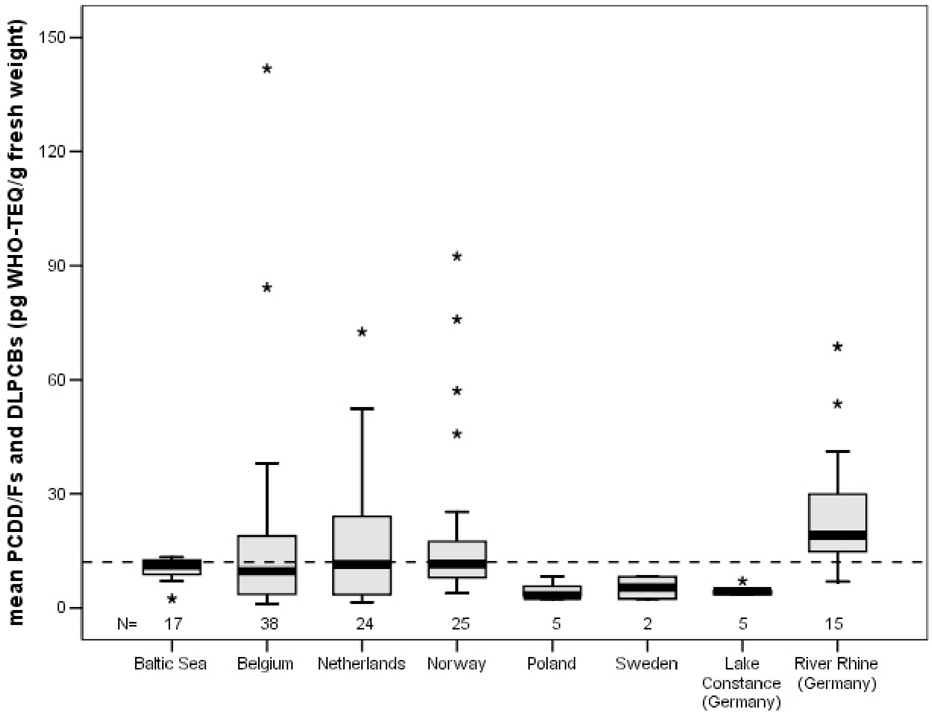


Figure 5-10. Mean concentration of PCDD/Fs and DLPCBs (pg WHO-TEQ/g) in European eel muscle as reported recently by several European countries and included in the European Eel Quality Database (Baltic Sea (Germany and Denmark): Karl *et al.*, 2010; Belgium: Geeraerts *et al.*, 2011; The Netherlands: Data Rikilt/IMARES; Norway: Julshammen and Frantzen 2009; Poland: Szlinder-Richert *et al.*, 2010; Sweden: SLV (Swedish National Food Administration); Germany (Lake Constance and River Rhine: CVUA (Chemisches und Veterinäruntersuchungsämt) Freiburg. The number of measured samples is indicated (N). The maximum consumption level of PCDD/Fs and DL-PCBs (12 pg g^{-1} fresh weight) as set EC by (2006) is represented (- -). (European Commission, 2006).

In Germany fishing eels in some contaminated river stretches was already prohibited for management reasons before exceeded consumption levels (PCDD/Fs, dlPCB) became known (i.e. Upper Rhine). In other affected areas, where fishing eels was only restricted (Lower Rhine, Elbe, Weser, Maas, Ems), trade of eels was banned and advice against consumption of eels was given (ML, 2011).

In the **Netherlands** there are professional fisheries on Rivers Rhine, Meuse and Schelde, constituting a network of dividing and joining river branches. Since 1 April 2011 the eel fishery on the main rivers has been closed due to high levels of pollutants in eel (Figure 5.11). This closure has affected ca. 50 fishing companies, catching 170 tonnes of eel in 2010, which represents roughly one third of the annual eel landings in The Netherlands.

For details on the closure, visit the following website; http://www.rijksoverheid.nl/ministeries/eleni/nieuws/2011/03/31/vangstverbod-paling-en-wolhandkrab-vanaf-1-april-van-kracht.html.

Figure 5-11. Overview of the areas closed for eel and Chinese mitten crab fishery as of 1 April 2011 (Source Ministry of Economic Affairs, Agriculture & Innovation of Netherlands).

In **France** a national programme on PCBs including eel sampling has been in place since 2008. On the foot of these samples, some fisheries bans were enforced (Figure 5.12) and since October 2010 further bans have been taken, the latest being for the Tarn-et-Garone area (August 2011).

In **Belgium** (Wallonia) eel fishing is prohibited in all waters due to high levels of PCBs since 2006. In some other areas (i.e. Flanders) it is recommended not to consume eels.

WGEEL recommends an assessment take place to estimate the possible contribution to the silver eel escapement coming from these closures and an evaluation of the quality of the silver eels being produced.

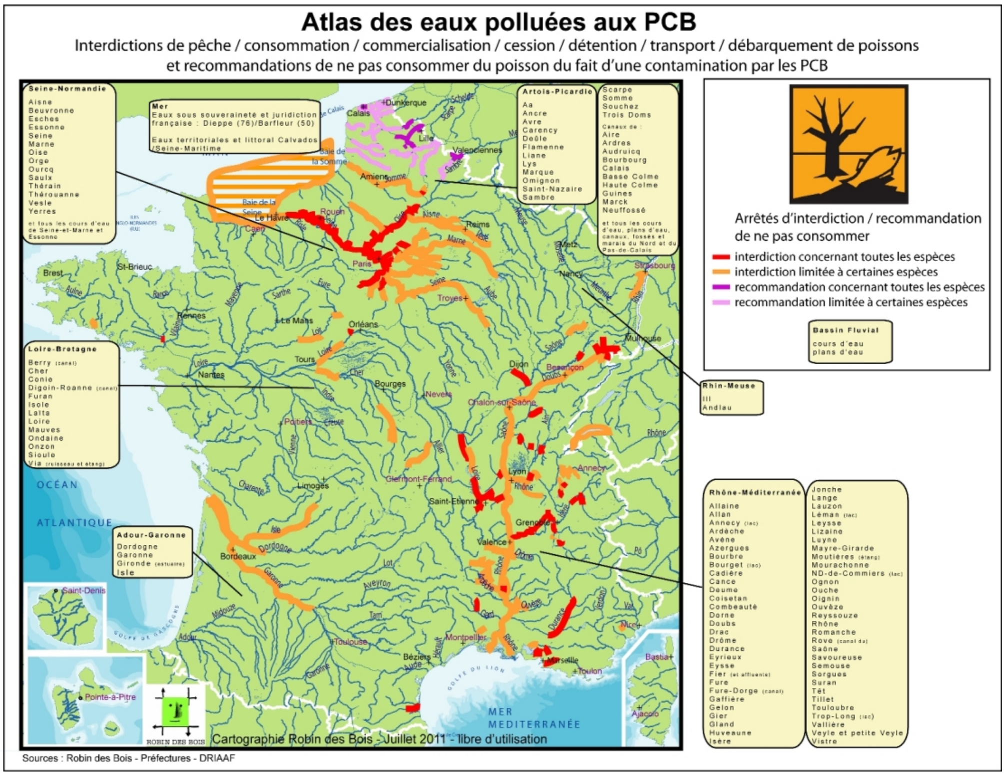


Figure 5-12. Fisheries bans due to PCB levels in France in July 2011. In red, fisheries bans for all species, orange fisheries ban limited to certain species, purple recommendation to not consume fish of all species and in lilac recommendation not to consume fish limited to certain species (Source: http://www.robindesbois.org/PCB/PCB_peche/restrictions_peche.html).

Figure 5-13. Map of Europe showing waterbodies where eel fisheries restrictions have been issued following the detection of high levels of contaminants in eel.

5.6 Eel kills due to contamination or diseases

Sudden eel mortalities caused by acute pollution or sudden disease outbreaks are known to appear regularly, but their impact on the whole stock is unclear.

5.6.1 Contaminants

Fish kills on a local scale often result from inadvertent management of land users (e.g. spill) but severe fish kills often result from accidental discharge or leakage on industrial sites producing or processing pesticides or other chemical compounds. In 1986 for example, incidents with atrazine were reported at Ciba Geigi-Bazel, with pesticides at BASF-Ludwigshafen, with chlorobenzol at Hoechst-Frankfurt (in the River Main), with methanol at Bayer-Leverkusen, with disinfectants at Bayer-Krefeld-Uerdingen, and with ethylene glycol at BASF. Fire-fighting water used to extinguish a fire in an agrochemical warehouse of the Basel chemical company Sandoz, contaminated by a variety of pesticides entered in the Rhine on November 1, 1986. Approximately 6–22 tons of pesticides are estimated to have been discharged into the river (about 1–3% of the inventory). This caused extensive pollution of the river due to pesticides and insecticides, including mercury-based and zinc-based pesticides. Levels of mercury in the Dutch section of the river were reported to be three times the normal limits. As a consequence half a million eels (ca. 200 tonnes) were killed, and the eel population was affected for years up to 650 km downstream (Christou, 2000).

Following Bálint *et al*. (1997) deltamethrin (the active ingredient of the insecticide K-OTHRIN 1 ULV) contributed to the severe eel devastation that occurred in Lake Balaton in 1991 and 1995, killing respectively 300 and 30 tons of eels. It seems that when eel kills occur, it is very hard to correlate these mortalities with precise chemical factors, because of the complexity of the pollution load (including a variety of contaminants which may interact) in many polluted areas (Anon., 1987).

In Belgium, in 2007, 25 tonne of fish (among which numerous eels) were killed in the River Meuse due to the discharge of 64 kg chloropyriphos and 12 kg cypermethrin, two components of pesticides (Geeraerts and Belpaire, 2010).

In Ireland periodic fish kills are recorded that include eel. In the past such fish kills were more frequent and associated with deoxygenation/eutrophication but there have also been some specific toxic effluent cases such as one described in a publication (Moriarty and Nixon, 1990) for a tributary of the River Nore where afterwards the river was treated with rotenone to remove polluted fish from the food chain.

In Sweden, some problems occurred with eel along the South Coast (Hanöbukten) in 2011. Reasons have not been verified nor was the mortality quantified. Fishermen complained and accused industries of polluting the water with unknown substances.

5.6.2 Diseases

Vast eel kills were documented in Canada in June and July 1993 (Lake Saint-François). Mortality estimates report several tens of thousands of individuals. The cause of the mortality was probably a viral disease, possibly in conjunction with other disease agents. Similar earlier eel kills have been reported earlier for the Lake Saint-Pierre and along the Saint-Laurence River in 1973, with mortalities estimated between 100 and 250 tonnes (Desrochers and Letendre, 1994).

On July 2011 an eel kill was observed in Lake Alauksts (Latvia). River Daugava, which is in the Baltic Seawater system runs into this lake. The lake was restocked with glass eel between the years 1935–1995. Local municipality staff estimated the eel mortality at nearly one tonne which is around one eel/20 m^2. The cause of the kill was not known as histochemical and microbiological analysis of three live eels caught in the lake had the acceptable number of *Anguillicoloides* and *Aeromonas*, an acceptable level of heavy metals and pesticides.

In summary, significant number of eels may be lost due to sudden mortalities caused by pollution and/or disease outbreaks. In some cases these mortalities are very high and gain media attention. However in many cases full description of the extent and exact cause of these eel kills is not available. The potential impact of these eel kills on the whole eel stock is therefore difficult to quantify. We recommend that member countries keep track of those incidents and report data in their country reports.

5.7 Conclusions to eel quality

- Data on contaminant and disease levels in eel have been made available to the EEQD. However, still, data are far from sufficient to allow Europe wide analysis of the quality of the stock.
- *Anguillicoloides crassus* has further spread (e.g. in Scotland and Ireland) and is now quasi omnipresent over Europe. In many places the parasitic infection rates seems to stabilize.
- There is a need for a better harmonization/standardization of eel quality assessments. Data are not always accessible.
- Several scientific papers suggest that the levels of particular hazardous substances are so high that an effect on reproduction is likely to occur, but scientific evidence (dose/response studies) is still not available. There is research need for better knowledge to enable quantifying the effects of parasites, diseases, and contaminants on migration and reproduction success, and to further develop the Eel Quality Index.
- High dioxin and/or PCB levels have been the basis for a closure or a restriction in the eel (or fish in general) fisheries. During the last years (2010–2011) fisheries restrictions/bans have been issued for an increasing numbers of water bodies. WGEEL suggest that the eels protected by these measures are in general of lower quality, and hence, their contribution to the restoration of the stock might be limited.

5.8 Recommendations for eel quality

The Working Group recommends that:

Recommendation	For follow up by:
1. In their annual country reports, member countries include data on the occurrence of sudden eel kills due to pollution or disease outbreaks. Information such as water body, water type, location, water surface, year, date, cause of death, and the estimated quantities of dead eels (and other fish) involved should be included	EU Countries, WGEEL Participants
2. In their annual country reports, member countries include a list of areas or water bodies where fisheries restrictions have been issued as a result of contaminant levels measured in eel (or other fish) exceeding human consumption safety limits	EU Countries, WGEEL Participants
3. Data of MS about contamination of eels raised for food regulatory reasons or for WFD should generally be made accessible for WGEEL (i.e. transfer of data to EEQD)	EU Countries, WGEEL Participants
4. Although the impact of contaminants and diseases on effective spawner escapment still remains unknown, regional eel management should generally refer to eel quality aspects like contamination and diseases, i.e include observed or measured impairments of eels in reports	EU Member States
5. Eel Quality index remains to be further developed for a better assessment of the overall status of eel quality over river basins. Eel Quality Assessments are to be linked to the quantitative assessments of effective spawner escapement in the EMUs	WGEEL, Belpaire
6. Research resulting in a better understanding of the eel's sensitivity towards parasites, diseases, and contaminants under field conditions, with respect to reproduction, should be supported. When the effects of stressfactors can be quantified a better, clear decision about the importance of "eel-quality" in eel management can be made	EU Funding
7. the direct impact of fisheries closure for human health's sake on stock restoration should be evaluated, i.e. what is the quantity and quality of eels affected by these measures, and to what extent do they contribute to the stock, considering their low quality?	WGEEL

5.8.1 References included in the EEQD during the 2011 WGEel session

Bergersen, R. and Klemetsen, A. 1988. Freshwater eel *Anguilla anguilla* (L.) from North Norway, with emphasis on occurrence, food, age and downstream migration. Nordic Journal of Freshwater Research 64: 54–66.

CVUA, Germany (Chemisches und Veterinäruntersuchungsämt) Freiburg, Germany (Bissierstr. 5, 79114 Freiburg).

Davidsen, J.G., Finstad, B., Økland, F., Thorstad, E.B., Mo, T.A. and Rikardsen, A.H. 2011. Early marine migration of European silver eel (*Anguilla anguilla*) in Northern Norway. - Journal of Fish Biology 78: 1390–1404.

DCF, Sweden.

Derek Evans. AFBI (Agri-Food & Biosciences Institute), UK.

Elvira Viuda Albacete. (Servicio de Pesca y Acuicultura, Región de Murcia), Spain.

Fisheries Technical Services. Environment Agency, UK.

Fisheries Technical Services. Environment Agency, UK; Cardiff University, UK.

Geeraerts C., Focant J-F., Eppe G., De Pauw E., Belpaire C. 2011. Reproduction of European eel jeopardised by high levels of dioxins and dioxin-like PCBs? Science of the Total Environment 409: 4039–4047.

Imares/Rikilt data from The Netherlands monitoring programme of contaminants in eel.

Janis Birzaks. Latvia.

Julshamn, K. and Frantzen, S. 2009. Miljøgifter i fisk og fiskevarer - en rapport om dioksiner og dioksinlignende PCB, polybromerte flammehemmere og tungmetaller i oljer, makrell, ål og Svolværpostei. Nasjonalt institutt for ernærings- og sjømatforskning (NIFES), Årsrapport 2008.

K Bodles. Northern Ireland, PhD.

Kelly, F., Harrison, A., Connor, L., Matson, R., Morrissey, E., Feeney, R., Wogerbauer, C., O'Callaghan, R. and Rocks, K. 2011. Sampling Fish for the Water Framework Directive – Summary Report 2010. Inland Fisheries Ireland.

Knutzen, J., Becher, G., Biseth, A., Bjerkeng, B., Brevik, E.M., Green, N.W., Schlabach, M. and Skåre, J.U. 1999. Overvåking av miljøgifter i fisk og skalldyr fra Grenlandsfjordene 1997. NIVA Rapport 4065-99: 1–195.

Martínez-Carrasco, C., Ruiz De Ybanez, R., Penalver, J., Mayo-Hernandez, E., Garcia-Ayala, A., Munoz, P. Prevalence of *Anguillicoloides crassus* (Nematoda, Dracunculoidea) in wild European eels (*Anguilla anguilla* L.) from Mar Menor lagoon (Western Mediterranean, Spain). Revue Méd. Vét., 2011, 162, 3, 154–158.

Næs, K., Knutzen, J., Håvardstun, J., Oug, E., Moy, F., Lie, M.C., Knutsen, J.A. and Wiborg, M.L. 2002. Miljøgiftundersøkelser i havner i Telemark, Vestfold, Akershus og Østfold 1999. PAH, PCB, tungmetaller og TBT i sedimenter og organismer. Statlig program for forurensingsovervåking. Overvåkingsrapport 849/02 TA-nr. 1885/2002. Norsk institutt for vannforskning (Norwegian Institute for Nature Research, 109 pp.

O'Leary, C.; Becerra Jurado, G.; Cruikshanks, R. and Gargan, P. Report on the National Eel Monitoring Programme. 2010.

SLV Sweden (National Food Administration, Box 622, SE-751 26 Uppsala).

Sölab stocking study, Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Freshwater Research.

Szlinder-Richert J., Usydus Z- and Pelczarski W. 2010. Organochlorine pollutants in European eel *(Anguilla anguilla* L.) from Poland. Chemosphere 80: 93–99.

6 Glass eel resources and stocking

Chapter 6 contains a review of the latest information on stocking of eel, with particular reference to its efficacy in producing spawner output. This consists of a review of WGEEL and other analyses to date, an assessment of current movements of eel for stocking and a new proposed model to assist in decision-making when stocking exercises are undertaken. Chapter 6 addresses Terms of Reference:

f) respond to specific requests in support of the eel stock recovery Regulation, as necessary;

g) report on improvements to the scientific basis for advice on the management of European and American eel;

Additional ToR:

1) review the glass eel catches for the past two years,

assess quantities	caught in the commercial fishery
	exported to Asia
	used in stocking
	used in aquaculture for consumption
	consumed direct
	mortalities

compare with the EMP commitments to stocking;

2) review the latest information on stocking, including previous reviews by WGEEL, new scientific reports/publications on the uses for stocking, the success of stocking as a method of fisheries support and/or as a means of increasing silver eel escapement and spawner biomass.

and has links to:

c) develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures.

6.1 Introduction

This task was set up for WGEEL 2011 to review the latest information on stocking and its effectiveness. The starting point for this section is to examine previous reviews by WGEEL, and add new scientific reports and other publications on issues relating to stocking and translocation of eel.

The important new focus was to examine what is known about the effectiveness of stocking in increasing silver eel escapement and spawner production, as compared to the traditional use of stocking as a support to fisheries.

Along with a review of the quantities of glass eel being traded in Europe and the amount set aside and used for stocking, this section introduces a new model, [TRANSLOCEEL] working towards predicting the outcome of stocking in terms of net benefit or loss. This is described and initially tested in a variety of scenarios. The Scenarios tested model the losses associated with stocking including fishing losses, transport and/or on-growing losses and anthropogenic mortality at different growth-rate/sex ratio scenarios.

This Chapter also discusses work in progress and known to WGEEL, anticipating where new information is likely to emerge in the near future.

6.2 Summary review of previous Working Group reps and advice

Stocking has been dealt with at all ICES/EIFAC meetings since 1999 (ICES, 2001) with a focus on the amount of stocked eel being used throughout Europe and the risk issues associated with stocking.

Due to the low recruitment and possible lack of spawners (ICES, 2004), restocking has been proposed as a measure to help restore the population. The argument was to avoid the risk of depensation of the stock. Restocking was proposed as a potential precautionary measure with a view to maximize spawner output from existing glass eel stock.

Several essential preconditions have been mentioned, first that demonstrable surplus should exist in some local (donor) glass eel stocks and that anthropogenic mortality in the recipient areas is minimized. The potential risks involved have been discussed (ICES, 2001, 2008). Some of the issues were:

- Movement of fish from donor to recipient waters involves a risk of altering the genetic structure of eel populations in recipient waters.
- The risk of spreading of disease and parasites when eels are moved from one area to another and this may be concentrated in quarantine stations or when seed stock are ongrown in eel farms.
- A further consideration is the observation that the sex ratio of eels varies according to stock density in a catchment. The factors involved in sex determination and the optimum sex ratio of the spawners are unknown and hence stocking represents a risk; however a deliberate manipulation of sex ratio may be seen as an advantage rather than a risk.
- There are some theoretical views and practical study results which suggest that survival to successful spawning of transported eel could be lowered or even non-existent. Studies of eel migration in the Swedish Baltic have suggested possible difficulties encountered by stocked eels when trying to find their way out of the Baltic (Westin, 2003; (ICES, 2006).

6.2.1 Outcome of stocking

WGEEL reports (2006, 2007, 2008 and 2009) have commented extensively on stocking theory and practical approaches to stocking based on manuals and reports (Williams and Aprahamian, 2004; Symonds, 2006; Williams and Threader, 2007).

The outcome of stocking has been evaluated in 2006, 2008 and 2009 WGEEL reports and it was clear from local studies that stocking has enhanced the yellow and silver eel stocks in a number of water bodies. These include several Danish, German, Swedish and Estonian Lakes, Lough Neagh in Northern Ireland as well as Danish streams and marine areas.

Studies of stocked ongrown eel reviewed by WGEEL indicated that the performance of stocked material in the yellow eel phase cannot be assumed to be as good as that of natural immigrants, but also conversely that it often falls within the ranges of best and worst observations of performance of wild stock.

Despite the stocking experiments reported in the above mentioned countries, there is clearly insufficient quantitative data from targeted studies of the performance of stocked eel in open wild environments, and more studies would help considerably in formulating advice. In particular, there is a lack of information on the outcome of previous stocking exercises in terms of the survival of stocked material through to eventual escapement and successful spawning.

Previous WGEEL reports concluded that there was potential for a positive output from stocking but crucially the question still remained as to whether stocked eels are able to find the way back to the spawning grounds in the Sargasso Sea.

The recommendations given were:

1) Stocking should be optimized to support the spawning stock, not to support fisheries. Stocking should preferentially go to areas likely to maximize high quality (non-polluted) silver eel escapement.(2006 onward);
2) Health issues. A protocol for screening stocked stocks should be put in place as soon as possible. Purposely infected eels in aquaculture with pathogens (viruses, etc.) should not be used for stocking purposes.(ICES 2008);
3) Due to uncertainties regarding the outcome of stocking (ICES 2009) recommended that all stocking activity be designed to include traceability of eel into later life stages. The best means of ensuring conclusive traceability is by using batch or other marking methods. OTC alizarin and strontium have all been used successfully to date on glass eel, PIT, CWTs, while other tags are available for larger stages;
4) In order to address the total absence of data on potential ocean migration of silver eel derived from stocking, future tracking studies (similar or successors to EELIAD) should include the ocean tracking of silver eel known to have been derived from stocked material.(2010);
5) Given the current low and declining availability of glass eel for stocking, and the stipulation in the eel regulation for an increasing proportion of glass eel to be made available for stocking, it is essential to optimize the quality and survival of the glass eel destined for stocking. A best practice manual on capture, storage and transport of glass eel is urgently required;
6) It is reasonable to assume that the degree of handling and intervention between glass eel and stocking strongly influences the outcome, and that best stocking practice is that which mirrors the local wild component.

6.2.2 Examination of relevant new material

6.2.2.1 Genetics

ICES Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) elaborated recommendations for eel conservation targets including restocking activities considering conservation genetic aspects in 2004 and 2007. Until 2004, no firm conclusions were arrived at with regard to a clear genetic structuring of the European eel. However the WGAGFM recommended that the precautionary principle be adopted to protect, as of yet, unresolved or potential genetic variability, and as a consequence the transfer of glass eels between basins should be avoided (ICES WGAGFM 2004).

Subsequently, Dannewitz *et al*. (2005) and Maes and Volkeart (2006) provided comprehensive reviews of the population genetics of the European eel.

While Daemen *et al*. (2001), Wirth and Bernatchez (2001) and Maes and Volckaert (2002) independently detected some genetic structure indicative of isolation by distance or a significant correlation between genetic distance and temporal distance among recruitment waves (Maes *et al*., 2006), Dannewitz *et al*. (2005) concluded that European eels sampled along the coasts of Europe and Africa most probably belong to a single spatially homogeneous, panmictic population. However, in light of emerging information suggesting putative stock structure of European eel WGAGFM recommended from the genetic viewpoint that glass eels, elvers and other life-history stages should not be trans-located between river basins for restocking purposes, or if seen as indispensable to avoid an imminent collapse of specific river stocks, where possible the translocation should be done within geographically proximate areas e.g. within the Mediterranean basin, within the North Sea, or within the Baltic Sea (ICES WGAGFM 2007).

Palm *et al*. (2009) again found slight temporal variation between cohorts of adult eels but no geographical differentiation.

Recently, Als *et al*. (2011) published a comprehensive population genetic investigation including for the first time samples of larvae from the spawning area in the Sargasso Sea, along with glass eel samples from continental foraging areas. The results suggest a random arrival of adult eels in the spawning area and subsequent random distribution of larvae across the European and North African coast, providing strong evidence of panmixia in both the Sargasso Sea and across all continental samples of European eel. However, the authors explicitly point to the possibility of within-generation local selection acting on genes in linkage disequilibrium as an explanation of the weak clinal patterns interpreted as genetic structure in previous studies and recommend further clarification by population genomics analyses aimed at identifying genes under selection in continental and Sargasso Sea samples.

For American eel, the hypothesis of panmixia was also confirmed (Bernatchez *et al*., 2011).

6.2.2.2 Spread of diseases and parasites

While the impact of parasites, particularly the infection with the swimbladder nematode *Anguillicoloides* crassus was investigated in greater detail (Kennedy, 2007), eel viruses have received less attention. Various viruses have been isolated from European eel, including the rhabdoviruses eel virus America and eel virus Euopean-X (EVEX), the birnavirus infectious pancreatic necrosis virus as well as a herpesvirus, Herpesvirus anguillae (HVA) (Sano *et al*., 1977; Jørgensen *et al*., 1994; Davids *et al*., 1999; van Nieuwstadt *et al*., 2001; van Ginneken *et al*., 2004; van Ginneken *et al*., 2005). Among these, EVEX and HVA have received most attention. While some authors (Davids *et al*., 1999; Lehmann *et al*., 2005) consider HVA as the most significant viral threat due to documented losses in aquaculture as well as in the wild under certain environmental conditions (Scheinert and Baath, 2004), proven negative impacts caused by EVEX are rare and basically restricted to one publication by van Ginneken *et al*. (2005), showing that European eels infected with EVEX-virus suffered from hematocrit decrease related to distance during simulated migration in large swim tunnels, developed hemorrhage, anemia and died after 1000–1500 km migration.

Recent investigations on HVA and EVEX infections of glass eels retained for restocking programmes showed the presence of both viruses at yet unknown, but obviously source-dependent infection rates. Deliberate infection with HVA is reported as a practice to avoid uncontrolled disease outbreaks in aquaculture, including also the ongrowing of glass eel for subsequent restocking. However, the impacts of the anthropogenic spread of viral diseases via restocking for the wild stock are unknown and should be avoided.

6.2.2.3 Growth of stocked eels

A literature review of the growth and survival of restocked eels in comparison to natural immigrants, and the orientation, navigation and migration of silver eels of restocked origin was made in 2010, including grey literature as far as it was available (Wickström, Clevestam, Sjöberg and Dekker, unpublished ms–Annex 7). There are more current original papers relevant with respect to growth and migration performance of stocked eels.

Stocking young eel as a means to increase fisheries catches of mainly yellow eel has proven to be successful in the past (e.g. Moriarty and Dekker, 1997; White and Knights, 1997; Rosell, 1999; former WGEEL reports). Currently, Psuty and Bohdan (2008) confirmed this for Eel stocking and fishing in the Vistula Lagoon of the Baltic Sea. From the Swedish review it can be concluded, that most of the studies carried out are giving evidence that restocked eels do survive and grow in a comparable way to natural immigrants. The choice of restocking material (glass eel or bootlace) seems not to influence growth, but there is a new study on the way to address this question explicitly for freshwater lakes of different productivity. Comparing restocked and natural eels in Lithuania did not show differences in growth. Lin *et al*. (2007) did not find growth differences between naturally recruited and stocked European eel in freshwater lakes and brackish lagoons in Lithuania except a higher length-at-age of stocked eels at ages 5 to 8 years. For American eel, Verreault *et al*. (2010) revealed a faster growth of stocked eel compared to naturally migrating counterparts in the St Lawrence river.

Concerning possible site-specific effects, Cote *et al*. (2009) observed significant growth differences in *A. rostrata* glass eel of two different origins which had been reared under similar conditions in salt and freshwater aquaria for 70 days. At the same time, glass eel of both origins grew faster in salt water than in freshwater. This also applies to *A. anguilla* as shown for different life stages by a number of studies (e.g. Melia *et al*., 2006; Edeline *et al*., 2005).

Contrary to most studies showing an equal growth of stocked eel and natural recruits, Tzeng presented a paper on stocked vs. natural eels in the Baltic Sea, namely from Latvia (Tzeng *et al*., 2009) indicating a slower growth of stocked eels. In this study they categorized sampled eels from three inland waterbodies into stocked and naturally recruited from the life-history trajectories found when analysing strontium-calcium ratios in the otoliths. Their results indicate a slower growth rate in stocked eels from two of the three habitats studied. However, they suggest that the differences found between wild and stocked eels might be influenced by the productivity where the eels were grown most of their lives which may not be reflected by the site of catch.

Except for length and weight gain, comparisons of stocked and naturally recruited ongrowing eel concerning other growth and fitness parameters are rare. Tzeng *et al*. (2009) investigated the habitat preferences and recapture rates between wild and cultured Japanese eels stocked in a coastal lagoon. There were no obvious differences between eel of the two origins and both stayed mainly in freshwater, i.e. neither in the river nor in the fully marine environment. A study on fat and energy content of European eel derived from coastal waters in the North and Baltic Sea as well as from freshwaters in that region has been conducted. Results will be made available in the near future. New studies to compare survival rates, behaviour, habitat choice and other parameters between stocked and naturally recruited European eel are not known to be underway but are required in order to improve the basis for the assessment of risks associated to stocking practices.

6.2.2.4 Sex differentiation and maturation

Sex differentiation in eel is not likely to take place before the fully pigmented young eel stage and at a body length of >15 cm. This has to be taken into consideration, when assessing sex ratios in stocked eel. A number of sites have been stocked with pre-grown eels from either aquaculture farms or natural waterbodies particularly over the last years. In those cases, the sex of at least a proportion of the stocking material might have been fixed before stocking and analyses of sex ratios in such populations would not be suitable to judge on sex ratios after stocking (Wickström *et al.*, 1996).

For the American eel, (Pratt and Threader, 2011) assessed gender for 13 sexually differentiated eel in the St Lawrence river previously stocked as glass eels and found five of them being males. This is noteworthy because previously only females had been detected in this watershed. The authors assume this is due to either the long holding time of glass eel of several months before stocking and/or density effects in the recipient area.

For the European eel, (Pedersen, 2010) reported after stocking of elvers into a brackish water lagoon a sex ratio of 1:2 (M:F) in catches of those eels in the yellow stage. Referring to sex ratios of natural immigrants in this waterbody, there was no information available.

With respect to silvering, there is some evidence that stocked eels silver and start their descent to sea in a comparable way as natural immigrants. Pedersen, 2010 found previously stocked European eel in a Danish brackish water lagoon to start their travel to sea as silver eel alongside with natural immigrants. Verreault *et al*. (2010) made a similar observation for *A. rostrata* in St Lawrence tributary, where previously stocked eel where found to silver and descend into the estuary on the same route in a comparable manner than wild eels.

6.2.2.5 Migration

As a general finding concerning the contribution of eels from freshwater habitats to the spawning stock, Tsukamoto *et al*. (2010) caught a silver *A. japonica* ready to spawn alongside three spawners originating in brackish and salt water at the ridge of the Marianna trench. This is the first evidence that eels which grew up in freshwater are contributing to the spawning stock at least in *A.japonica*.

Despite several investigations using otolith chemistry and other means to quantify the contribution of restocked eels to the spawning run, no firm conclusions can be drawn. A rough estimate based on the stocking figures and subsequent silver eel run in the Baltic came to the conclusion that the observed percentage of silver eels from restocking origin is reasonably in agreement with expectation: there is no reason to believe they all fail, nor to consider restocking a panacea. The parallel development of stocking and escapement also indicates that the fitness of the stocked and naturally recruited eels is similar.

On tagging experiments in Roskilde Fjord (which has a long and complex connection to the Kattegat) with natural eels and eels of restocked origin, Pedersen (2009) and Pedersen (pers. comm.) wrote: "In autumn 2004 and 2005 silver eels of stocked (N=143) and wild (N=450) origin were Carlin-tagged and released in the bottom of the fjord. The result was a higher recapture rate of wild eels N= 126 (28%) compared to stocked eels N= 27 (19%) but the difference is not statistically different ($\chi2$ test, P=0.12). Independent of eel origin (wild and stocked), both eel types were caught in the same proportion in the southern part of the fjord (56%; wild N=71; farmed N=14) and in the northern part of the fjord (44% wild N=55; farmed N=10), indicating that the stocked eels migrate toward the outlet of the fjord together with the wild silver eels ($\chi2$ test, P=0.94).

Since 2006, 869 eels of varying size and sexual maturity have been tagged in Estonia, upstream of hydroelectric power plants in Narva. All eels in the area above the power plant were considered to be of restocked origin. A total of 93 eels are recaptured, mostly in the lakes they were marked. The few recaptures (7%) that were made outside the immediate vicinity of the tagging area were all made in a direction towards the outlet of the Baltic Sea, in the Sound (Järvalt *et al*., 2010).

In the Vistula lagoon in Poland's Baltic coast, there are indications that silver eels, probably of stocked origin, do not orient towards the lagoon outlet in the northern, Russian part of the lagoon, but initially migrate westward where there is no connection to the sea (Wilkońska and Psuty, 2008). The authors interpret this phenomenon as stocked eels may not find the outlet to the lagoon and in this context they refer to the hypotheses of Westin (1998).

For the American eel, it is known that stocked eels are migrating out of the St Lawrence Systems at an earlier age, 4–6 years old, than their wild counterpart, 20–25 years old, (Verreault *et al*., 2010.), possibly associated with their origin or increased growth rates (Pratt and Threader, 2011). Although the exact numbers of stocked eels leaving the system is still unknown, it may be significantly higher than currently thought because silver eel fishing gears in the St Lawrence River and estuary are not designed to capture the smaller silvering eels. What is known is that the smaller stocked eels can at least initiate the spawning migration.

Regarding the doubt about the ability of stocked eels to navigate properly during the migration, this is based solely on the experiments made by Westin in the 1990s. Statistical analysis of a large historical material of eel tagging in the Baltic came to the conclusion that although some impacts might be present of the increased stocking during the post-war period, these will be hidden by all historical changes such as fluctuating fishing pressure, which precludes an unambiguous analysis. Several other small-scale tagging studies have been reported that both support and don't support a difference in navigation ability. None of those studies contain statistically significant results.

So far, the only comparison of migration behaviour outside the continental shelf between stocked and non-stocked eels is the EELIAD project described below.

A working document, Westerberg and Sjöberg, 2011, describes an experiment which was part of the EELIAD project. The objective was to investigate possible behavioural differences between silver eels of translocated and naturally immigrated origin. Eels that with a high probability were naturally recruited were taken from the Enningdal river and eels probably stocked as glass eel imported from Severn were taken from river Ätran, high up in the river, above more than ten dams without eel ladders. In each group 15 eels were tagged internally with coded acoustic tags and data storage tags and a further ten with external data storage tags.

Both groups were released innermost in a fjord on the Swedish west coast, where the acoustic tags were to be monitored with moored receivers at three transects during the subsequent migration to the open sea. The coverage of the monitoring arrays was not 100% so the absolute number of eels passing the different transects could not be used in a comparison. Instead the delay to the first recording at the innermost transect and the mean migration speed between subsequent transects were compared. No statistically significant difference is found between the stocked and naturally recruited eels.

Two data storage tags with a reasonably long record of active migration have so far been recovered. The two eels had different recruitment history; one naturally immigrated and one translocated and stocked as glass eel. They show very similar behaviour, with a diurnal depth cycle; deep in daytime and shallower during the night. Both follow the Norwegian trench at average depth >200 m. Both migrate at about 30 km/day. This behaviour is consistent with what has been found from data storage tagged silver eels leaving the Baltic. A migration route along the Norwegian Trench is also shown by scientific bottom-trawl surveys.

The overall conclusion is that this experiment shows no evidence of a difference in migration behaviour between stocked and naturally recruited eels.

6.2.2.6 Magnetic compass senses

New work (Durif, submitted and pers. comm.) indicates that European [Yellow] eel have the ability to detect, remember and orientate movements according to magnetic fields of equivalent strengths to the Earths field. This leads to a conclusion that magnetic orientation is likely to be part of the suite of senses that eel use in migration from growing areas to spawning area as silvers. It is not yet known whether or not there is an “imprint” or map laid down by eel in earlier life stages immigrating coastal and freshwater growing areas which informs later navigation on emigration to spawn. Were that the case, there would be a possibility of disorientation leading to reduced emigration success as a result of translocation but this has yet to be demonstrated either way.

6.2.3 A re-assessment of the risks in translocating eel

6.2.3.1 How does new data or information change the advice on the potential of stocking and translocation to impact?

To our current understanding, European and American Eels are panmictic species. New data on growth, sex differentiation, maturation, silvering and migration to the continental shelf have not shown any major differences between stocked and wild eels. From this point of view stocking has the general potential to produce outmigrating silver eel. On the other hand to support a recovery of a panmictic stock, stocking has to give a net benefit to the whole population. The new information available on risks gives no new insights regarding this aspect.

6.2.4 Ongoing studies–work noted as in progress

6.2.4.1 Genetics

For American eel, the hypothesis of panmixia was confirmed (Bernatchez *et al*., 2011). Regardless, there is a suggestion of phenotypic differences between regions (C. Cote, unpublished). If so, then regional eel management may be required including the decision on where to obtain fish for stocking and where to stock. It will be important to determine if low density wild stock phenotypes will be displaced/replaced by the translocated stock.

6.2.4.2 Growth and condition

An assessment of body condition between wild and stocked American silver eels (R. Threader, G. Verreault and M. McNiven) is currently underway. Another technique used to evaluate spawner success may be measured through the comparative analysis of the occurrence of oocyte atresia (C. Couillard; Threader and Groman) between wild and stocked silvers and as measured at various distances along their migration route (Threader). Also, size may have an impact on their swimming energetics for oceanic migration, as was shown for the European Eel by Clevestam *et al.*, 2011. An overall assessment of body condition between the various life stages of American eel is currently underway (Threader, pers. comm.) as a prelude to evaluating stocking success.

6.2.4.3 Sex ratios

It is known that the stocking programme in Canada is producing males of the American eel with a preliminary estimate of a 50:50 ratio. Currently, an evaluation of sex ratios of young wild eels migrating upstream measured at the R.H. Saunders eel ladder will be compared to the ratio of stocked eels measured through the stocking effectiveness monitoring programme.

6.2.4.4 Migration

A follow up of the EELIAD experiment with stocked and naturally recruited eels is planned for the migration season 2011. The same experimental design will be used but the tagging will be made with both Microwave Telemetry satellite tags and internal programmable data storage tags to increase the return of data from the oceanic phase of the migration.

6.2.5 Risk assessments in stocking of eel and precautionary approaches

The influence of some potential risk factors on the eventual production of spawning eel from trans-located stock is not known. These factors are listed in this chapter, and some have been revisited several times in previous reports. WGEEL has advised before that a precautionary approach should be applied in assessing risk when the outcome stocking is uncertain.

Definitions of the precautionary approach as used in relation to the fisheries generally start from the standpoint of *not* ***delaying*** *action* [to protect stocks] *where there is uncertainty that the action will succeed*.

For example, from the 1992 United Nations Conference on Environment and Development, Rio de Janeiro 1992, "the Rio Declaration":

In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, ***lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures*** *to prevent environmental degradation.*

From the OSPAR convention, specifically relating to fisheries management, this has given rise to: "…the A lack of full scientific evidence must not postpone action to protect the marine environment. The principle anticipates that delaying action would in the longer term prove more costly to society and nature and would compromise the needs of future generations”.

This interpretation does not translate easily to the discussions surrounding transfer and stocking of eel, where the interpretation tends to: *do not* ***take action*** *(stocking) where there are uncertainties* [over whether or not this will result in viable spawners]. This reflects a general common usage of the precautionary approach which states that “the burden of proof that an action is not harmful falls upon those taking the action” (Wikipedia).

A significant problem for stocking under this interpretation is that there can emerge a long list of “what if” scenarios of unknown potential factors with at least the potential to cause failure of spawner production, each requiring lengthy investigation before stocking plans can be adjusted (or not) according to the latest perceived risk. Inevitably, not all new proposed risk factors will turn out to be significant problems.

WGEel, having an overall view, must start from the consensus that eel is a panmictic single-stock, and that restocking is only acceptable as a means to assist overall stock recovery, rather than one to maintain fisheries. Therefore, WGEEL attempted to approach the issue interpreted in line with this scientific point of view, and with the ultimate goal of ensuring net benefit from restocking activity for the whole stock.

There are, however, widely diverging and as yet irreconcilable views, within and outside WGEEL, of the meaning of a precautionary approach in the decision framework surrounding stocking of eel. At one end of the spectrum are managers faced with a risk assessment to stock for the first time, seeing the stocking as a new action. On the other hand, where stock transfers to maintain [fishable] stocks have been the *status quo* for decades, some see the cessation of stocking as the action to be risk assessed, with stocking the *status quo*.

The following is a summary of stocking risk assessment and precautionary approaches:

- WGEEL is not in a position to recommend a general prescription or proscription of stocking. As stocking is permitted and occurring under the terms of the EU Eel Regulation, we can only give guidelines on how to minimize risk. This has been set out in detail in previous working group reports;
- There is no standard agreed interpretation of a precautionary approach which can be applied universally to stocking of eel; indeed there are many different and often conflicting positions taken. This arises from the fact that the precautionary approach is applied differently by fisheries policy-makers and managers in different countries, reflecting the diversity of problems faced.

What is now needed for assessment of risk associated with stocking

Thus, what is needed is a means to conduct realistic assessment of the risk associated with the factors on the list of potential problems. Given that translocation and stocking already takes place and has taken place over decades, there ought to be already in place the opportunity for practical scientific assessments on existing populations. Instead of simply advocating a “do nothing until proven safe” approach, the existing and ongoing translocated stocks could be seen as the test bed for practical investigation.

Where there are still uncertainties, and new uncertainties

Summary of evidence emerging from existing translocation

There is now ample evidence that translocated and stocked eel will, in productive environments, produce eel which grow to yellow and silver, and will attempt to migrate. An extensive review of this subject was carried out by Wickstrom *et al*. (2010) and made available to WGEEL 2011. The review found that while there are some suspected examples where emigration is less successful than wild recruits, there are many more cases where trans-located eel appear to have successful emigration behaviour mirroring the natural to the furthest point at which naturally recruited eel have been observed. The latest data storage tag data on some eel originating in wholly stocked populations and tracked out from the Baltic (H. Westerberg, working paper submitted to WGEEL 2011) have shown that eels of stock origin can have the same oceanic migration behaviour and the diel vertical migration found by satellite and data storage tags in the eeliad programme. There remains a problem that no eel, let alone one originating in translocated stock, has yet been followed to the spawning grounds.

This absence of data on the ultimate fate of either stocked eel or natural immigrants leads inevitably to the recommendation that further targeted studies are needed to complete the picture. The EELIAD project, tracking the ocean migration of silver eels to their spawning grounds, is a key example of such a project. It is important, in the context of assessing the value and impact of stock transfers, that future work in eeliad and subsequent programmes, specifically includes eel known to be derived from stock transfers. Work planned on the Swedish west coast, linked to eeliad, in the next two years is already intending to tag eels known to come from both stocked and unstocked groups.

It is also obvious that unless stocked batches of eel can be tracked through remaining stages of the life cycle to spawning, no further information will be gained to fill in this gap in our knowledge. Therefore it is a strong recommendation of this WG that all stocked eel should be marked and thereby separable from wild eel in subsequent sampling. This should be done using an internationally approved and coordinated method.

6.3 Data on glass eel, landings and trade

6.3.1 Introduction

Article 12 of the Regulation states "No later than 1 July 2009, Member States shall: take the measures necessary to identify the origin and ensure the traceability of all live eels imported or exported from their territory".

ToR

1) Assess quantities of glass eel caught and their fate:
- caught in the commercial fishery
- exported to Asia
- internal trade between EU Countries
- used in stocking
- used in aquaculture for consumption
- consumed direct

- mortalities

2) Assess where possible “movement through” countries and match up import/exports;

3) compare with the commitments to stocking in the EMP (use stocking data supplied in ICES review Table).

Definitions

In this task we want to be sure that we only obtain data on **stocking** that is the practice of adding fish [glass eel] to a waterbody from another source, to supplement existing populations or to create a population where none exists.

It is not where glass eel are caught and transported around an obstruction, we have termed this **assisted migration**.

6.3.2 Trade analysis

The trade analysis was conducted using two sources of data;

1) data from the EuroStat database for France, UK, Spain, Germany, Netherlands, Denmark, Sweden and Greece and

2) data direct from glass eel dealers for the UK. These data consist of the date of each shipment, its destination, the amount, its fate (for stocking, aquaculture, consumption) and from which eel management unit the glass eel were obtained. In addition the European Commission requested member countries to identify the total amount of glass eel harvested, the amount imported and exported by life stage and for any information on seizures (and has been referred to as EC data in this report).

The EuroStat database query was for the period November 2010–May 2011 and undertaken in July 2011. In the EuroStat database there is no separation by life stage and the distinction between glass eel and yellow eel was made according to the methods in Briand *et al*. (2008). The EuroStat database has several limitations when dealing with glass eel. Sometimes the nature of the exports is not clear and must be assumed from its price, while at others some glass eels may be included in a consignment of yellow eels, and the proportion of glass eels must be estimated from price. Furthermore all data in EuroStat are rounded to the nearest 100 kg, while much trading of glass eel takes place in smaller quantities (see Table 6.2). Additional data on imports and exports were obtained from Country Reports. No information was available from the following countries: AT, BG, CY, CZ, HU, LT, LU, MT, RO, SL and SK.

The trade value can be different for import and export data, which reflects the fact that the trade value appearing in the statistics is the one that is declared by enterprises within the country. For example, Spanish middlemen might buy glass eel directly from the French fishers and this trade may only be reported as an import and not as an export from France. Thus we used maximum reported trade from either side of a border as the ‘real’ export value.

The best estimate of the total catch of glass eel in 2011 was 40 791 kg (Table 6.1) of which 40 066 kg were exported. The estimate of 3059 kg for Spain is a minimum figure and is based on the EuroStat export figure for Spain, it does not include the quantity of glass eel caught in Spain and stocked, sent to aquaculture or consumed within Spain itself. For France the estimate of catch is the total export figures from EuroStat (33 700 kg) together with those stocked in France (733 kg). Portugal has a glass eel fishery operating on the river Minho, however no catch data for 2011 were available.

Table 6-1. The amount of glass eel caught and exported in 2011.

Country	Total catch (kg)	Total exported (kg)
UK*	3682	2917
France**	34 433	33 700
Spain***	3059	3059
Portugal	No Data Available	No Data Available
Morocco****	390	0

***Data from dealers and Country Reports;**

****Data from EuroStat and Country Report;**

*****Data from EuroStat;**

******Data from Country Report.**

6.3.2.1 Destination of the catch by country

The destination of the catch from the France, UK and Spain is shown in Table 6.2 and in Figures 6.1, 6.2 and 6.3. For France and Spain the data are from the EuroStat database, while for UK the data are direct from the glass eel dealers with additional data from the country reports. There was no export of glass eel from Morocco.

Of the glass eel exported from France to UK a portion was reexported to other countries within Europe. Table 6.3 shows the final destination of glass eel caught in UK, France, Spain and Morocco and includes those glass eel that were stocked in the country of origin.

From the total quantity of glass eel exported (39 676 kg) it was possible to identify the origin of 70.6% (28 002 kg) through EuroStat figures and from Country Reports. Of the remaining 11 674 kg the destination may be to those countries identified as "Unknown" in Table 6.2 and possibly additional amounts to those countries mentioned in Table 6.3 but not identified in their country report.

Finland stocked 306 000 individuals in 2011 (250 kg). As these were imported from Sweden and not directly from the donor nations a value of zero has been recorded in Table 6.3. Similarly for Belgium, the 40 kg of UK caught glass eel were imported via the Netherlands.

Table 6–2. The direct destination of glass eel caught in UK, France and Spain in 2011.

	Quantity exported (kg)		
Destination Country	UK	France	Spain
Belgium	40.0$	120.0	
Czech Rep	30.0		
Denmark	515.0	4000.0	
Estonia	306.5		
Germany	882.0	3400.0	1300.0
Greece	411.0	400.0	200.0
Latvia	100.0		
Netherlands	593.0	5200.0	300.0
Poland		80.0	
Slovakia	79.5		
Spain		4691.0	
Sweden			
UK		4200.0	
Hong Kong		4.8	1200.0
Unknown*		11 655.1	58.7

$ From Belgium Country Report

*** AT, BG, CY, CZ, HU, LT, LU, MT, RO, SL, SK**

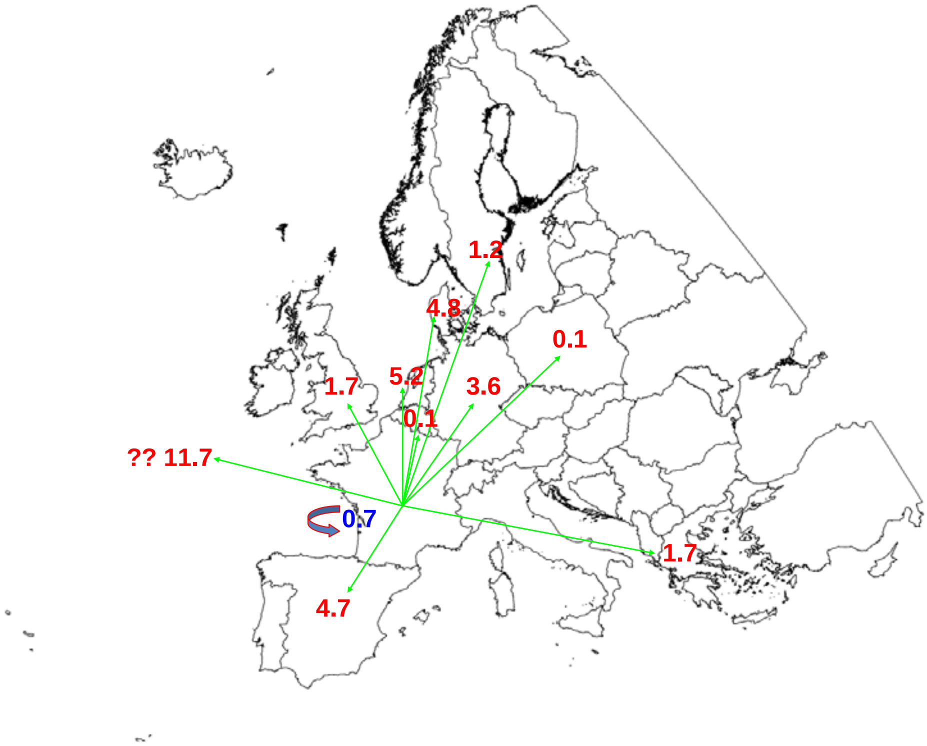


Figure 6-1. Destination and quantity of glass eels landed in France (values in tonnes). 0.7 tonnes were sold for use within France. The destination of 11.7 tonnes of French glass eels was unclear, but thought to be divided between one or more of the European countries with no associated value in the map).

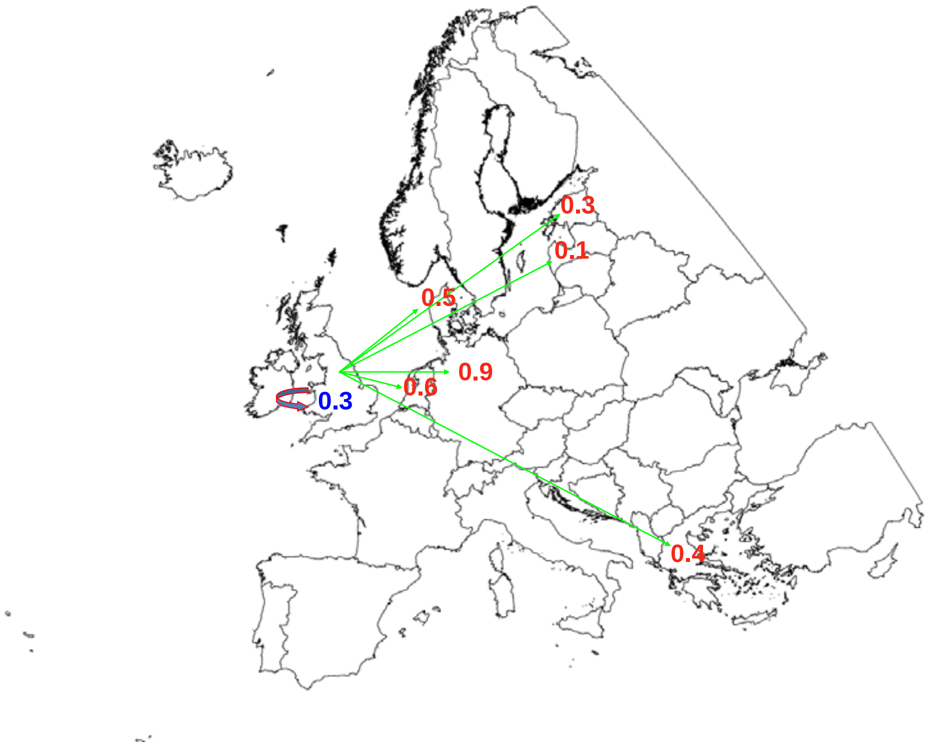


Figure 6-2. Destination and quantity of glass eels landed in the UK (values in tonnes). 0.3 tonnes were used within the UK.

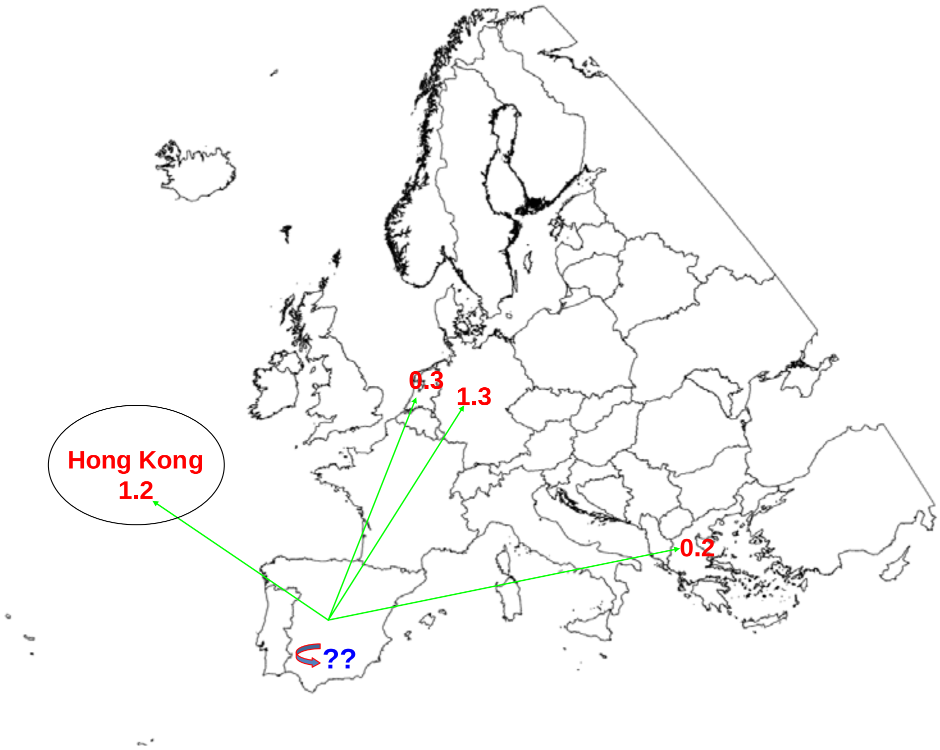


Figure 6-3. Destination and quantity of glass eels landed in Spain (values in tonnes). The export of 1.2 tonnes to Hong Kong was reported by Spanish authorities as "angulones" (see below). The quantity of glass eels used within Spain itself is unknown.

Table 6-3. Destination of glass eel caught in 2011, by country of origin.

	Quantity exported (kg) by Country of Origin				
Destination Country	UK	France	Spain	Morocco	Total
Austria	0.0	n/a	n/a	0.0	n/a
Belgium[1]	0.0	120.0	0.0	0.0	120.0
Bulgaria	0.0	n/a	n/a	0.0	n/a
Cyprus	0.0	n/a	n/a	0.0	n/a
Czech Rep	30.0	n/a	n/a	0.0	n/a
Denmark	515.0	4750.0	0.0	0.0	5265.0
Estonia	306.5	0.0	0.0	0.0	306.5
France	0.0	733.0	0.0	0.0	733.0
Germany	882.0	3550.0	1300.0	0.0	5732.0
Greece	411.0	1712.0	200.0	0.0	2323.0
Finland[2]	0.0	0.0	0.0	0.0	0.0
Hungary	0.0	n/a	n/a	0.0	n/a
Ireland	0.0	0.0	0.0	0.0	0.0
Italy	0.0	0.0	0.0	0.0	0.0
Latvia	100.0	0.0	0.0	0.0	0.0
Lithuania	0.0	n/a	n/a	0.0	n/a
Luxembourg	0.0	n/a	n/a	0.0	n/a
Malta	0.0	n/a	n/a	0.0	n/a
Morocco	0.0	0.0	0.0	390.0	390.0
Netherlands	593.0	5200.0	300.0	0.0	6093.0
Norway	0.0	0.0	0.0	0.0	0.0
Poland	0.0	80.0	0.0	0.0	80.0
Romania	0.0	n/a	n/a	0.0	n/a
Slovakia	79.5	n/a	n/a	0.0	n/a
Slovenia	0.0	n/a	n/a	0.0	n/a
Spain	0.0	4691.0	n/a	0.0	n/a
Sweden	0.0	1200.0	0.0	0.0	1200.0
UK	332.3	714.0	0.0	0.0	1046.3
Hong Kong	0.0	4.8	1200.0	0.0	1204.8
Unknown*	0.0	11 655.1	58.7	0.0	11 713.8

[1] Belgium stocked 40 kg of glass eel that were caught in the UK, but sent via the Netherlands; (Belgium Country Report); n/a indicates no information available.

[2] Finland stocked 250 kg (306 000 ind.) obtained from Sweden.

6.3.2.2 Data audit and anomalies

There was insufficient corroborative data to conduct a rigorous audit.

For the UK it was possible to explore variation between several datasets regarding the export of glass eel (Figure 6.4). Correspondence between the 2011 EuroStat analysis (see Briand *et al.*, (2008) for methods) and the EC data are high, but not exact (4.16 vs. 4.08 tonnes respective total exports), because the latter, (although also based on the EuroStat data), were derived using the assumption that all eel exports categorized as “live” were glass eels. The EuroStat database query was for the period November 2010–May 2011 and though the database for some countries (France and Spain) includes exports in May 2011 those for UK only include exports up to and including March 2011. Thus some of the discrepancies for the UK may reflect that the data has yet to be entered into the EuroStat database. Similarly the EC data for the UK extend only until March 2011. By contrast, data provided to EC by Spain does not appear to tally with either the EuroStat analysis, or with the data provided in the country report, and thus likely has an independent origin. The EC dataset has no data from France for 2011. The UK country report data detailing values obtained directly from dealers (including reexported eels) indicate higher exports (total exports = 7.02 tonnes), probably because it also includes export data from April and May 2011.

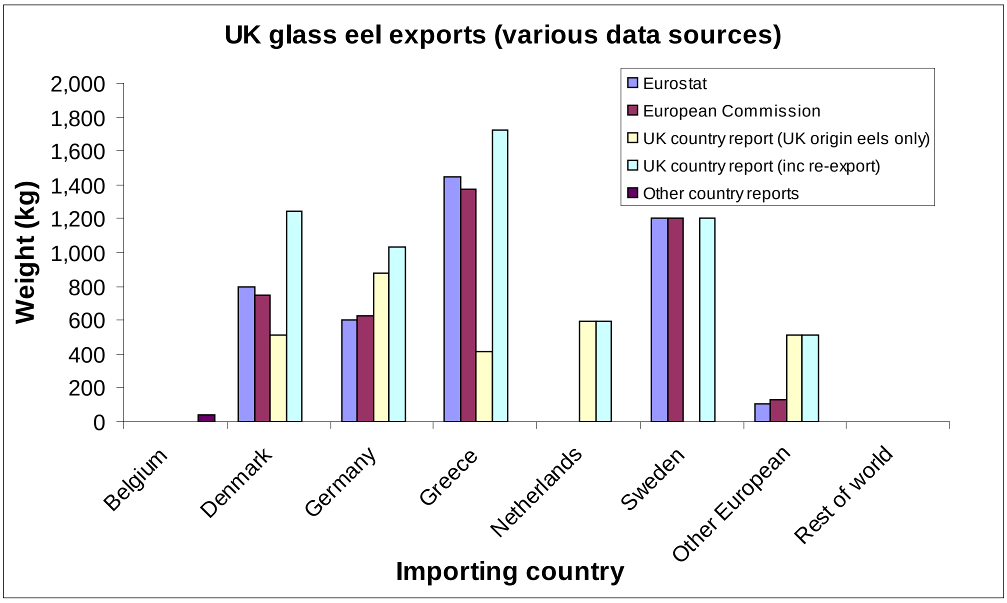


Figure 6-4. Comparison of UK exports of glass eel in 2011 from various data sources: EuroStat data analysis; EC statistics (UK entry till March only); UK country report (UK origin eels only, December 2010–May 2011); UK CR (including re-exports, December 2010–May 2011); according to the individual country reports/questionnaires.

Some specific anomalies were noted and for the Netherlands in particular there was a marked difference between the information obtained direct from the dealers and that obtained from the export data from UK. The EuroStat database indicated no glass eel exported to the Netherlands while information from dealers indicated a total of 593 kg were exported from the UK (Table 6.4, Figure 6.4). For those countries the imports are >100 kg the EuroStat figures underestimate the information from the dealers for Denmark, Germany and Greece but not for Sweden (perhaps because of missing data for April and May in the EuroStat data). There is a requirement to determine whether this lack of correspondence between the datasets is real or reflects the fact that the EuroStat database was not complete at the time of analysis. Accordingly, we reanalysed the UK trade data (in Country report), seeking (an arbitrary) cut-off date that best matched the EuroStat/European Commission data. Figure 6.5 shows the three datasets using 6th April 2011 as the cut-off date, and indicates a high correspondence between the three datasets. This strongly suggests that the disparity between UK trade and EuroStat/European Commission data in Figure 6.4 results the incompleteness of the EuroStat and European Commission data at the time of analysis. Even closer correspondence could be generated using slightly different cut-off dates for individual countries. The close similarity between UK trade data and the

EuroStat data strongly support the validity of the latter as a means of estimating total exports of European glass eel.

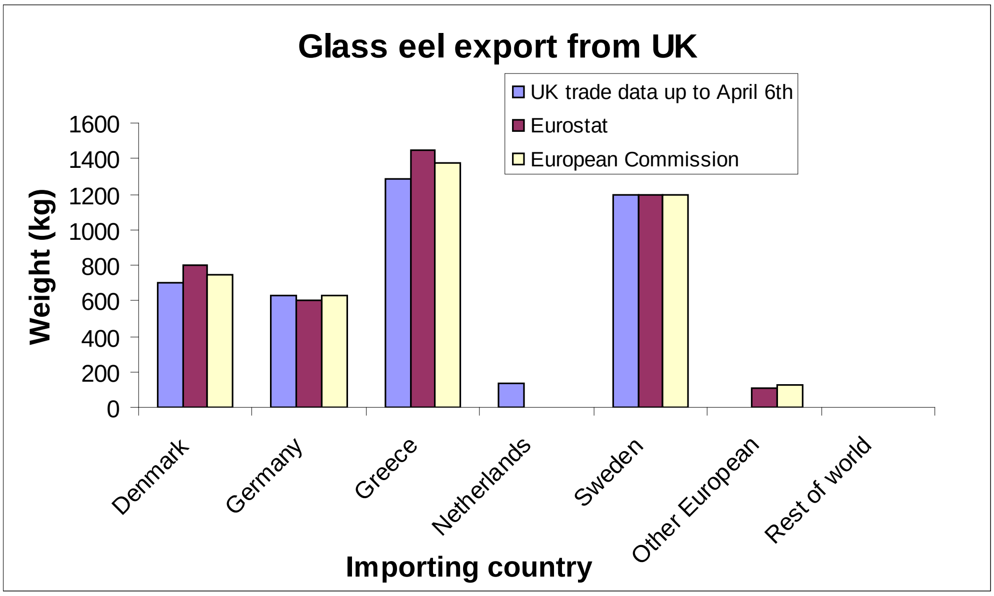


Figure 6-5. Close correspondence between EuroStat, European Commission and UK trade data (when applying a cut-off date of 6th April to the latter).

Table 6-4. A comparison between the dealers import returns from within the Netherlands with export data to the Netherlands.

Country of Origin	Netherlands (dealers returns)	Export data
UK	256.0	593.0
France	4838.0	5200.0
Spain	1890.0	300.0
Unknown	10.0	
Total	6994.0	6093.0

These differences between the various datasets are only evident where the various sources are available. For countries other than the UK there was little scope for scrutinizing anomalies between several datasets. For Latvia the Country Report indicates an import of 100 kg from the Czech Republic, which has no glass eel fishery, so these must have been imported from elsewhere, mostly likely the UK (Table 6.3). The UK export data indicates an export of 100 kg to Latvia and 30 kg to the Czech Republic.

In some cases there is reason to doubt the accuracy of the data, even where no other data source was available for cross-checking. For Spain the EuroStat database indicates there is an export of high value going from Spain to Hong Kong in May 2011. According to the CITES trade data sent to the EC, Spain reports a 2011 export of 1200 kg of "Angulones" (slow-growers), pre-Convention specimens with origin Portugal, so it is assumed that the permit was issued before the cut off date for trade in live pre-Convention specimens (not permitted after 1st April 2011), and the trade was reported later in Customs (Vicki Crook, pers. comm.).

6.3.3 Fate of glass eel

The amount of glass eel that was stocked, used for aquaculture or consumed, together with the proportion where the fate could not be identified is shown in Table 6.5. Of the total quantity of glass eel imported 11.8% were stocked, 30.5% were used in aquaculture, whilst the fate of the vast majority (57.7%) remains unknown (Table 6.5).

6.3.4 Trend in the price of glass eel

Prices are corrected for inflation using the price index for France. The glass eel prices show an exponential rise from around €5 in the 1960s to a maximum of €500 per kilogramme in 2005 (Table 6.6). The high price in 1969 corresponds to the onset of Japanese buying on the French market.

Table 6-5. The fate of glass eel by country.

	Quantity (kg)				
Destination Country	Total	Stocked	Aquaculture	Consumed	Not known
Austria	n/a				
Belgium	160.0	160.0	0.0	0.0	0.0
Bulgaria	n/a				
Cyprus	n/a				
Czech Rep	n/a	30.0			
Denmark	5265.0		1265.0		4000.0
Estonia	306.5	208.4	98.1	0.0	0.0
France	733.0	733.0	0.0	0.0	0.0
Germany	5732.0	661.0	371.0		4700.0
Greece	2323.0		1723.0		600.0
Finland[1]	250.0	250.0	0.0	0.0	0.0
Hungary	n/a				
Ireland	0.0	0.0	0.0	0.0	0.0
Italy	0.0	0.0	0.0	0.0	0.0
Latvia	100.0	100.0	0.0	0.0	0.0
Lithuania	n/a				
Luxembourg	n/a				
Malta	n/a				
Morocco	390.0	0.0	390.0	0.0	0.0
Netherlands[2]	6053.0	173.0	5880.0	0.0	0.0
Norway	0.0	0.0	0.0	0.0	0.0
Poland	80.0	80.0	0.0	0.0	0.0
Romania	n/a				
Slovakia	79.5	79.5			
Slovenia	n/a				
Spain	n/a				
Sweden1	950.0	798.0	152.0	0.0	0.0
UK	1046.3	1042.3	4.0	0.0	0.0
Hong Kong	1204.8		1204.8		
Unknown*	11 713.8				11 713.8

[1] **For Sweden the estimate was calculated:- (1200–250 (sent to Finland)) = 950 kg; (1200*3000) = 3 600 000; 3 600 000–306 000 (sent to Finland) = 329 400; 2 766 166 (no. stocked in Sweden) / 329 400 = 84%; 950*0.84 = 798 kg.**

[2] **40 kg of glass eel sent to the Netherlands was exported to Belgium.**

Table 6-6. Trend in glass eel trade price (€) computed from various sources. Prices corrected for inflation using price index in France.

Year	French custom	French trader	Asturian (Spain) Market	EuroStat France	EuroStat Spain	EuroStat UK	Average price
1961		6.6					6.6
1962		4.0					4.0
1963		3.4					3.4
1964		10.0					10.0
1965		7.1					7.1
1966		9.5					9.5
1967		12.3					12.3
1968		8.2					8.2
1969	1056.8	13.4					535.1
1970	68.4	13.3					40.9
1971		20.9					20.9
1972	76.6	25.2					50.9
1973		33.1					33.1
1974		20.5					20.5
1975	41.9	21.6					31.8
1976	45.5	14.5					30.0
1977	41.0	18.8					29.9
1978	42.5	19.0					30.8
1979							
1980	24.3						24.3
1981							
1982	42.8						42.8
1983	50.9	42.7	56.6				50.1
1984	32.9	28.6	59.1				40.2
1985	49.8	37.2	69.7				52.2
1986		49.0	81.8				65.4
1987	63.0		43.1				53.1
1988	59.4	54.1	90.9				68.1
1989	108.0	110.5	127.9				115.5
1990	108.7	119.9	134.6				121.1
1991	94.2	108.9	135.6				112.9
1992	162.3		110.7				136.5
1993	155.8	86.0	97.1				113.0
1994	177.3	109.1	95.8				127.4
1995	134.8	94.1	90.2		163.2		120.6
1996	202.6	199.0	148.4	206.4	185.8	193.4	189.3
1997	246.5	366.2	224.3	260.6	247.0	344.8	281.6
1998	297.3	267.3	250.9	295.6	313.6	294.9	286.6
1999	212.9	270.3	173.7	208.1	214.2	267.8	224.5
2000	226.4	207.0	226.8	216.3	254.7	254.6	231.0
2001	331.0	358.5	261.2	267.4	306.7	304.1	304.8

Year	French custom	French trader	Asturian (Spain) Market	EuroStat France	EuroStat Spain	EuroStat UK	Average price
2002	247.0	252.3	231.4	220.4	230.8	202.8	230.8
2003	235.0	254.1	215.6	236.7	199.2	226.1	227.8
2004	496.7	452.8	431.7	423.5	282.4	229.9	386.2
2005	857.2	873.2	562.7	648.7	308.7	531.1	630.3
2006	432.3		373.7	370.3	297.4	404.4	375.6
2007				499.2	343.5	265.0	369.2
2008				316.3	281.9		299.1
2009				344.5	146.6	408.1	299.7
2010				584.3	322.7	338.7	415.2
2011				351.5	228.0	431.3	336.9

6.3.5 Amount of glass eel stocked by country and wrt EMP target

In 2008, twelve countries proposed the use of stocking in their management plans to enhance eel populations (ICES, 2008), between 2009 and 2011 stocking of glass eel was undertaken in 4–7 countries (Table 6.7). Of the various countries which stocked glass eel only, Sweden and Latvia achieved their target. The most common reason for a country being unable to achieve its stocking target was the high price of glass eels which over the last three years has averaged between €300–415 per kg (Table 6.6).

ICES identified ~40 t yr^{-1} of glass eels were needed to meet EMP requirements, which approximates to the best estimate of the total annual European catch for 2011 (Table 6.7). However, at least 30% of the 2011 catch was used in aquaculture (Table 6.5) and it has only been possible to identify 12% of the total catch being used for stocking. It has not been possible to identify the fate of the remaining 58%.

6.3.6 Conclusions to glass eel trade

The usefulness of the glass eel trade data incorporated in the EuroStat and EC figures is itself questionable, because both include re-exports and thus elevate the apparent total trade in glass eel in a non-predictable way. Data from the UK (see Country Report) for exports of glass eel solely of UK origin indicate a total trade of only 2.92 tonnes), less than half the figure which includes re-exports.

The indications are that there is no consistent approach to ensuring traceability of live eel, and in particular glass eel, across Europe as required by the Regulation. For those countries which catch glass eel the UK comes closest to achieving this where for each batch of glass eel exported the date, amount, destination, fate and origin are recorded, and the data made available to the regulatory authority. For France and Spain no system of traceability exists. Similarly for those countries importing glass eel there is no system by which the fate or amount of glass eel can be effectively traced.

Table 6-7. The quantity of glass eel purchased with EMP target in brackets, the percentage of the EMP target achieved, the percentage of glass eel purchased used for stocking and the quantity of glass eel harvested from the years 2009–2011, by country.

	Purchased (kg) (EMP TARGET)			Target achieved (%)			% used for stocking			glass eel harvest (kg)		
country	2009	2010	2011	2009	2010	2011	2009	2010	2011	2009	2010	2011
Austria	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Belgium	152	143 (500)	160 (500)	n/a	28.6	32.0	100	100	100	0	0	0
Bulgaria	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Cyprus	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Czech Rep	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Denmark	no data	4443 (4000)	no data	no data	1.3	no data	no data	1.2	no data	0	0	0
Estonia	750	750	750	18.5	8.0	27.8	50	33.3	68	0	0	0
France	no data	No data	733 (?)	no data	Not achieved	Not achieved	no data	no data	100	42 800	40 700	34 433**
Germany	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Greece	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Finland	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0	0	0
Hungary	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Ireland	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0	0	0
Italy	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Latvia	no data	No data	100 (100)	no data	no data	100	no data	no data	100	0	0	0
Lithuania	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Luxembourg	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Malta	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Morocco	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	837[1]	3413[1]	390

country	Purchased (kg) (EMP TARGET)			Target achieved (%)			% used for stocking			glass eel harvest (kg)		
	2009	2010	2011	2009	2010	2011	2009	2010	2011	2009	2010	2011
Netherlands	no data	No data	6994 (550)	no data	no data	49	no data	no data	2.6	0	0	0
Norway	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0	0	0
Poland	80 (4000)	80 (4000)	80 (4000)	2.0	2.0	2.0	100	100	100	0	0	0
Portugal	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	576	947.2	no data
Romania	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Slovakia	no data	No data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Spain	no data	No data	no data	no data	no data	no data	no data	no data	no data	2255.9	5458.2	2765*
Sweden	205 (833)	870 (833)	1200 (833)	17.2	100	100	70	82	84 [3]	0	0	0
UK	217 (2054)	1385 (2054)	1046 (2054)	10.6	67.4	50.9	100	100	99.6	420	1889	3642

[1] From China and Hong Kong Custom data (Vicki Crook, pers. comm). [2] see Table 6.1; [3] see Table 6.5.

6.4 Determining net benefit

6.4.1 Stocking model TranslocEEL

The overall objective of the model is to assess the net benefit (or loss) of stocking to the population. To achieve this, survival data from both source and supplied areas are needed. The projected spawner output, calculated with stocking or in the "do nothing" option are calculated and compared. To the benefit of intra EU stocking, this could allow the derivation of transfer authorization similar to the non detrimental finding used by ICES for trade of eel outside EU.

If glass eel recruitment falls any further, the glass eel fishing and transport may no longer appear as non-detrimental.

6.4.2 Baseline of the model

TranslocEel is an expansion of Aström and Dekker's model (2007) and the SED model (Lambert, 2008) to multiple spatial compartments. It is based on the combination of mortality lines (one in each compartment) and a single stock–recruitment relationship (panmixia). A full description of the model is presented in Annex 7.

The mortality lines are based on off-line estimations of mortality in the historical situation (ca. 1980–2005). For all compartments, the lifespan mortality for females from glass eel to silver is used. In addition, for the donor compartment, additional information concerning glass eel mortality is needed. In the recipient compartments, the mortality of stocked fish is adjusted relative to mortality of wild fish.

The ecological features of each compartment (age at silvering for female, proportion of undifferentiated eel that become female, fecundity, capacity for female to reach the Sargasso) and the proportion of glass eel arriving in each compartment are first defined. The stock–recruitment relationship is then fitted to mimic the observed European recruitment trend (Aström and Dekker, 2007).

With the TranslocEel mathematical formulation, it is possible to define the global threshold to which the weighted summation of survivals in each compartment must comply to halt a decline of recruitment and avoid a population crash. The local Aström and Dekker (2007) thresholds (based on local parameters) comply with this global condition.

This model also allows the user to define whether a compartment is functioning as a source (the number of females that are escaping to spawn will result in a glass eel return that produces the same number of female silver eel) or sink (the opposite).

The main assumptions of TranslocEel model are:

- The proportion of males is not a limiting factor for the population dynamics.
- There is no density-dependant regulation of mortality or sex ratio determination.
- The proportion of recruitment arriving in each compartment is constant over time.

The outputs of the model are a comparison of five management options where 1000 glass eel can either be stocked into various compartments or left *in situ*. These options are evaluated after several generations (15 and 50 years):

a) stocked into the West compartment (which is also always the donor compartment);
b) stocked into the North compartment;
c) stocked into the South compartment;
d) a stocking ban with a corresponding mortality reduction;
e) a stocking ban without a corresponding mortality reduction (i.e. glass eel fishing still continuing, but the captured fish are not used for stocking).

A corresponding level of mortality was assigned to each of these stocking/no stocking options, and combined to make model scenarios. The possible levels of mortality (ΣA -the sum of anthropogenic mortality over the lifetime of an eel) which could be used for any scenario include:

1) Historical conditions (ΣA historical);
2) Reductions in mortality that correspond to the local threshold exemplified in Aström and Dekker's model (2007) which lead to a global stabilization of eel stocks ad prevents a crash (ΣA local threshold);
3) Reductions in mortality that lead to each stocking compartment acting as a source rather than a sink (ΣA source threshold);
4) Reductions in mortality that lead to a recovery (quantified as ΣA local threshold/2);
5) Reductions in mortality that correspond to a crash (quantified as (ΣA historical + ΣA local threshold)/2).

6.4.3 Choice of scenarios

The following scenarios (numbers refer to the mortality level in the W-N-S compartments respectively as defined above) were chosen to exemplify different combinations of recovery programmes and levels of mortality reduction:

1) 1-1-1. The scenario where all compartments are operating with historical levels of anthropogenic mortality (this can be considered the baseline scenario).
2) 2-2-2. The scenario where all three compartments (West, North and South) are at the threshold level proposed by (Åström and Dekker, 2007), i.e. the lowest level required to halt the decline and avoid a global crash.
3) 5-2-2. The scenario where the West compartment does not reach the local threshold level proposed by (Åström and Dekker, 2007), but rather falls midway between the local threshold and historical levels of anthropogenic mortalities. North and South compartments are at the local thresholds in this scenario.
4) 2-2-5. The scenario where the South compartment does not reach the local threshold level proposed by (Åström and Dekker, 2007), but rather falls midway between the local threshold and historical levels of anthropogenic mortalities. North and West compartments are at the local thresholds in this scenario.

6.4.4 Calibration (choice of parameters)

The mortality factors selected for the simulations are presented in Table 6.8. The mortality coefficients for West compartment come from calibrated and expert estimates of all anthropogenic impacts in rivers on the French Atlantic Coast (Lambert, 2008). The coefficients for the North are those published by Dekker (2000) for "elsewhere than Bay of Biscay" in his procrustean model. It is based on catch from the North of Europe and Mediterranean zone with a lifespan corresponding to northern latitudes. The lifespan mortality in the South compartment is the ratio of current escaping biomass ($B_{current}$) to the best achievable escaping biomass (B_{best}) for the French "Rhône Méditerranée" eel management unit (WGEEL 2010). This rough estimate is higher than the value calibrated for the specific Camargue lagoon within this zone (0.47) but is likely to be more realistic for the compartment in general. To calculate the potential reduction of glass eel fishery mortality induced by a stocking ban, we used the figure of 60%, which represents the amount of glass eel caught that are dedicated to stocking as fixed in the EU regulation for 2013.

The fate of the stocked fish is adjusted to take into account a post-fishing, transport and quarantine mortality of 25%. We used 20% post fishing mortality as a mean value between mortality induced by push net and hand net (Briand *et al.*, 2009), an additional 5% mortality during transport and quarantine. Currently, there is no clear evidence of a difference in mortality between wild and stocked fish in the same environment and, therefore, 100% adjustment factors were used for the three compartments.

Table 6-8. Mortality parameters used in simulations.

	Compartment		
	WEST	NORTH	SOUTH
sigma M	2.2781	2.2400	0.8400
sigma A historical	1.8348	3.2400	2.7992
duration of glass eel stage	0.25		
F for glass eel	3.3195		
M+H for glass eel	4.8852		
proportion of glass eel dedicated to stocking	60%		
post fishing, transport and quarantine mortality rate	25%	25%	25%
adjustment factor of anthropogenic mortality for stocked fish	100%	100%	100%

Table 6.9 presents parameters summarizing the main ecological characteristic in the three compartments and in the ocean. Oceanic journey for leptocephalus was fixed at two years (Bonhommeau *et al.*, 2010). The exponential trend of recruitment corresponded to the result of the glass eel time-series analysis (see Chapter 2.1.2). The proportion of recruitment arriving in each compartment was based on a free interpretation of leptocephali distribution of 20° longitude established by Bonhommeau *et al.* (2010). The continental lifespan was fixed to be in accordance with the previous mortality estimates and the life parameter table (WGEEL 2010).

Data were collated by WGEEL (ICES 2010a) concerning the length and age of silver eel escaping to migrate across Europe. From these data, we chose three index countries from which to extract length data. For the west compartment, we use length data collated for France (average length of females at silvering = 672.5 mm). Sweden was used for the north compartment (average length of females at silvering = 727.8 mm) and lengths from Italy were used to represent the south compartment (average length of females at silvering = 596 mm) (ICES 2010).

Specific fecundity was calculated with the weight-at-length relationship of Melia *et al.* (2006) and the fecundity relationship relative to weight of Andrello *et al.* (2011).

Clevestam *et al*. (2011) have found that silver eels below a certain length will consume that much energy (fat) during their long migration that they probably are not capable of spawning successfully when arriving at the Sargasso Sea. From a combination of size of silver eels leaving the Baltic Sea and the long distance, they concluded that about 20% will arrive with a net content of fat of 0% or lower. By applying their data derived from the North compartment to the corresponding length and distance data for the West and the South compartments (from WGEEL 2010) in a proportional way, we concluded that 18% of silver eel from the West, and 28% from the South compartments are also probably unable to spawn successfully by the time they reach the Sargasso sea. Specific fecundity and capacity to reach the Sargasso were standardized relative to the values calculated for the West compartment. Therefore silver eel numbers are expressed in terms of 'West females'.

The last parameter, the proportion of undifferentiated eel that become female, were initially set as West (0.5), North (0.9) and South (0.1), but after discussion with WGEEL members, it was felt that 0.1 was too low for the south, as the proportion of females in the Mediterranean lagoons is currently more than 90%. It also became apparent that it was going to be difficult to fix this parameter. Instead, we collated the sex ratio of silvers escaping from France (60% female), Sweden (90%) and Italy (50% female) for the period 1980–2000. We then worked back to undifferentiated eel using the continental lifespan of males and females and the lifetime mortality of each (Eq. 5, Annex 8). This resulted in values of 81%, 98% and 73% for the west, north and south respectively for the proportion of undifferentiated eel that become female.

Table 6-9. Ecological parameters used in simulations.

	Compartment		
	WEST	NORTH	SOUTH
exponential decrease in recruitment (year-1)		0.0982	
duration of ocean migration (year)		2	
proportion of recruitment arriving in each compartment	0.5	0.1	0.4
continental lifespan for female (year)	9	16	6
proportion of undifferentiated eel that become female	81%	98%	73%
Length at maturity for female (mm)	672	727	596
Relative specific fecundity according to West	100%	129%	68%
Relative capacity to reach the Sargasso according to West	100%	98%	88%

The main outputs of the model calibration are presented in Table 6.10. The net reproductive output was estimated to 30.5 glass eel per West silver eel escaping, lower than the value of 121 glass eel per silver eel found by Bonhommeau *et al*. (2009) and the value of 149 glass eel per silver of Andrello *et al*. (2011), both based on a steady state hypothesis. The Aström and Dekker threshold to prevent a recruitment collapse corresponds to a reduction of the anthropogenic mortality to 59%, 55% and 28% in West, North and South relative to historical conditions. The historical mortality has to be reduced to 49%, 57% and 37% so that the West, North and South compartments become sources for the population.

Table 6-10. Main outputs of the model calibration.

	Compartment		
	WEST	NORTH	SOUTH
Net reproductive output (glass eel per West silver eel)		30.5	
∑A (Aström and Dekker) threshold	0.7546	1.4724	2.0136
∑A source threshold	0.9352	1.3850	1.7567

6.4.5 Results

1) Scenario 1-1-1 represents historical conditions, where anthropogenic mortality is high in all compartments. In this case, stocking of 1000 glass eel has no long-term benefit in any compartment (levels at 15 years are already lower than the original 1000, and by 50 years, the numbers of fish resulting from the stocking are practically zero) (Figure 6.6). All compartments operate as sinks in this scenario, and none are able to produce a stable return of glass eel over the time period under consideration (50 years). By way of comparison, the option of not removing the fish in the first place does not lead to a sustainable return either, whether or not the corresponding fishing mortality is reduced. In the case where stocking is banned, and the corresponding fishing mortality is eliminated, the theoretical 1000 fish also become a non-sustainable entity, but over a longer time period than when the 1000 fish are stocked elsewhere, or when the corresponding fishing mortality is not reduced (left hand bar). This scenario provides a baseline against which other scenarios can be judged, as this represents the historical norm in the case of stocking.

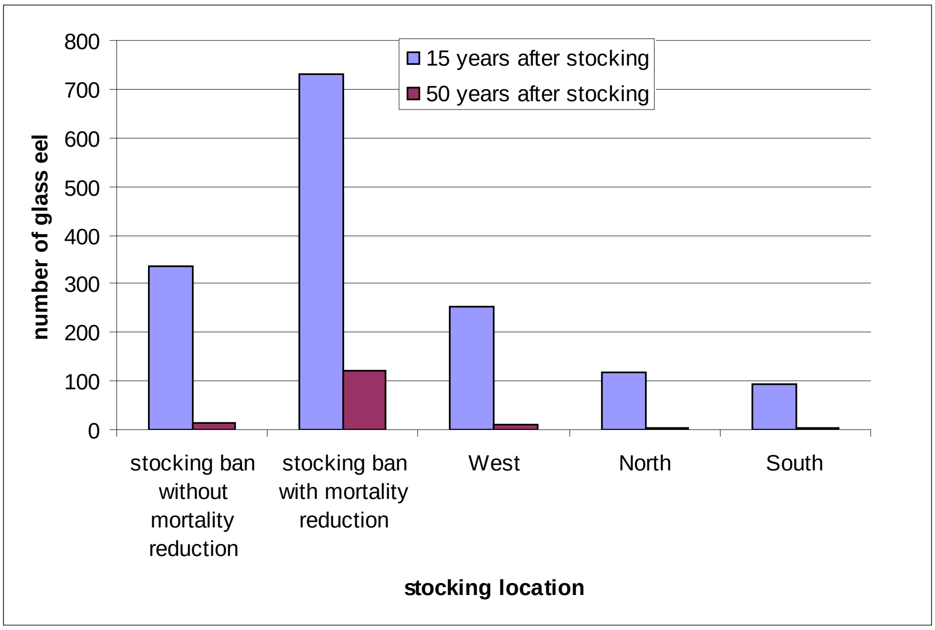


Figure 6-6. Outputs for the Scenario (1-1-1) representing historical conditions.

2) The second scenario (2-2-2), represents a situation where conservation measures have been implemented across Europe, and all compartments are experiencing low enough anthropogenic mortality that the global population will not crash; i.e. they are stable in the long term, and recruitment is not declining (in reference to the Åström and Dekker (2007) local threshold). In this scenario, no stocking with a corresponding reduction in fishing mortality is by far the best measure both in the short (15 years) and long run (50 years) as glass eel return will increase in the west compartment (Figure 6.7). Where a stocking ban is implemented but the corresponding fishing mortality is not reduced, the glass eel return stabilizes over 50 years at a relatively high level, but does not increase. Stocking the 1000 glass eel into an area in the west other than the donor catchment produces a stable return, at a level just above 1000 fish. Stocking in the remaining compartments also produces a stable return, but at lower levels. In this scenario, the West compartment acts as a source while the rest are sinks.

3) In the third scenario (5-2-2), reductions in mortality have been implemented in the South and North compartments in line with the local threshold described in Åström and Dekker (2007), but the West compartment continues to experience a level of mortality midway between this local threshold and historical levels. In this case, none of the stocking options produces a stabilization of the glass eel return (Figure 6.8). In this scenario, the resulting decrease in glass eel return is also apparent in the cases where no stocking has taken place, irrespective of whether the corresponding fishing mortality has reduced. The reduction in post-fishing and/or transport mortality means that the non stocking options lead to a slightly slower decrease than any of the stocking.

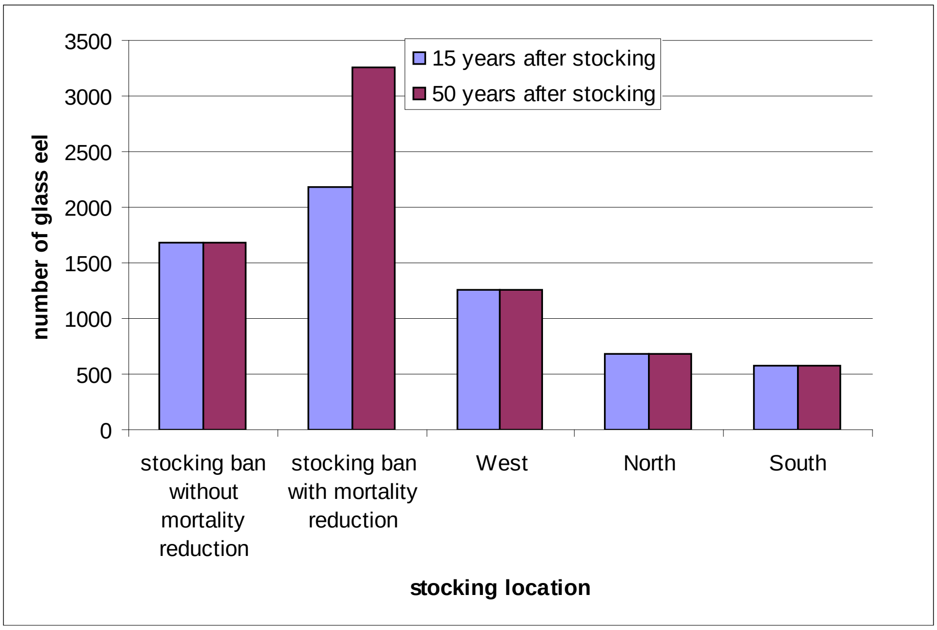


Figure 6-7. Outputs for the scenario (2-2-2), representing a situation where conservation measures have been implemented across Europe, and all compartments are experiencing low enough anthropogenic mortality that the global population will not crash.

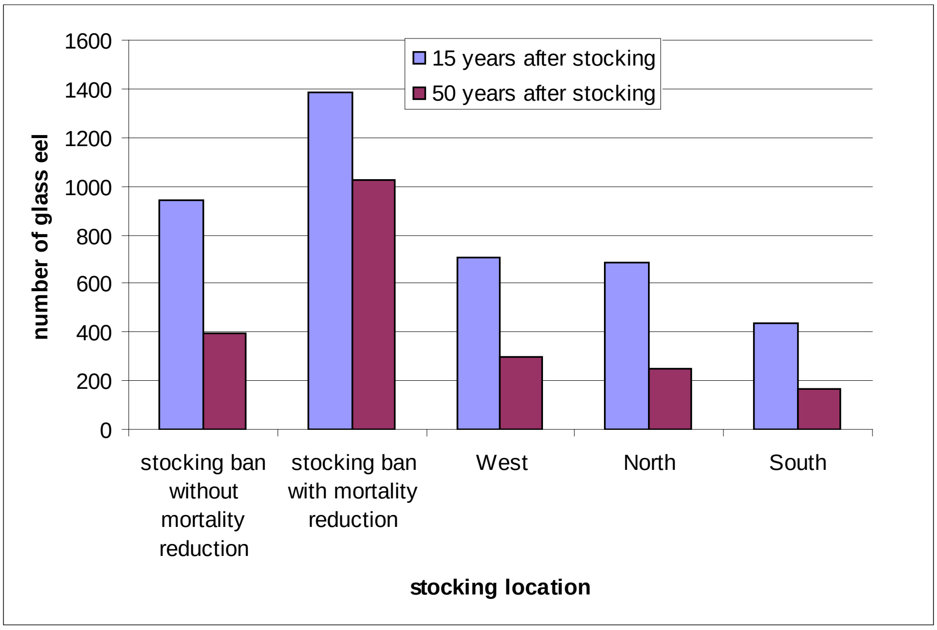


Figure 6-8. Outputs for the scenario (5-2-2) representing reductions in mortality have been implemented in the South and North compartments in line with the local threshold described in Åström and Dekker (2007), but the West compartment continues to experience a level of mortality midway between this local threshold and historical levels.

4) The last scenario is similar to the third scenario in that one of the compartments (in this case the South) has only implemented a partial reduction in mortality, and has not reached the reduction required to meet the local threshold. With the South below the threshold (Figure 6.9) the no stocking option which includes a corresponding decrease in fishing mortality, avoids a crash and improves the stock, in contrast to the third scenario. The difference between the outcomes (in the case of stocking ban with mortality reduction) of this scenario (the south still has relatively high levels of mortality) and the third scenario (where the west has relatively high levels of mortality) is worth noting, and results from the relative difference in mortality between levels 2 and 5 in the two compartments. Reducing mortality from level 5 to level 2 in the south compartment produces a relatively larger positive global impact than the same reduction in west compartment, which is why in the fourth scenario, the no stocking option results in a stabilization of the stock. In contrast, reducing the mortality from level 5 to level 2 in the west is not enough to result in a sustainable increase in stock.

Figure 6-9. Outputs for the scenario which is similar to the third scenario in that one of the compartments (in this case the South) has only implemented a partial reduction in mortality, and has not reached the reduction required to meet the local threshold.

6.4.6 Evaluation of the modelling approach

After the preliminary runs, it was clear that this model is very sensitive to parameter calibration. For example a lower value of anthropogenic mortality in the South compartment in historical conditions inverts the picture for 2-2-2 scenario simulation, and the South become a source for population dynamics instead of the West. A sensitivity analysis is required to prioritize the parameters, of which most require improvement in calibration. It will also be necessary to strive for more consistency in the methodology used to evaluate mortality in different compartments.

6.4.7 Modelling-conclusions

- The fish resulting from stocking one compartment disperse relatively quickly to the other compartments.
- For the four scenarios described above, the only situation which results in an increase numbers of glass eel produced in the long term is when the glass eel are left *in situ*, and the corresponding mortality is reduced. All other situations lead, at best, to a stabilization of the population.
- When comparing stocking locations, the outcome is always better when the glass eel are stocked in a source compartment rather than a sink.

6.4.8 Stocking-conclusions, recommendations and advice

- WGEEL, having an overall view, must start from the consensus that eel is a panmictic single-stock, and that restocking is only acceptable as a means to assist overall stock recovery, rather than one to maintain fisheries. Therefore, WGEEL attempted to approach the issue interpreted in line with this scientific point of view, and with the ultimate goal of ensuring net benefit from restocking activity for the whole stock.

- WGEEL is not in a position to recommend a general prescription or proscription of stocking. As stocking is permitted and occurring under the terms of the EU eel regulation, we can only give guidelines on how to minimize risk. This has been set out in detail in previous working group reports.
- There is no standard agreed interpretation of a precautionary approach which can be applied universally to stocking of eel, indeed there are many different and often conflicting positions taken. This arises from the fact that the precautionary approach is applied differently by fisheries policy-makers and managers in different countries, reflecting the diversity of problems faced.

Modelling section

- The results of the model (using the current parameters outlined above) indicate that stocking should always be carried out with caution as it generally does not make the most efficient use of the glass eel cohort, when compared with the option of leaving them *in situ* with a corresponding reduction in fishing mortality.
- The most efficient use of stocking is when anthropogenic mortality is reduced to levels which ensure that the global population does not crash.
- Stocking is more efficient if the recipient compartment is a source rather than a sink (but the identification of source or sink compartments is based on a thorough understanding of the global dynamics of the eel population).

Recommendation	For follow up by:
1. WGEEL recommends that, in the absence of an agreed standard definition, risk assessment methods are applied to stocking decisions as robustly as local data permits	Member States
2. It is recommended that all countries put in place a traceability system to meet the requirements of Article 12 of the Regulation. Essential elements allowing traceability and permitting cross-checking between countries can be identified: for each batch of glass eel exported, the date, the amount, the price, the destination EMU and fate (stocking/aquaculture/consumption), and the EMU of origin need to be recorded and made available to the appropriate regulatory authority	Member States, EU, CITES
3. It is possible for stocked compartments to produce increased number of glass eel relative to the no-stocking option, but only when post fishing and transport mortality of the stocked fish is outweighed by increased survival of stocked fish relative to their wild congeners in the donor basin. We recommend that the model be used to test for situations where this might apply and assess whether it is likely	WGEEL

6.5 Chapter 6-overall conclusions

Linking the three elements of this work area, WGEEL 2011 concludes that:

Declared glass eel total catch from fisheries in 2011 was approximately equal to the current requirements for stocking listed in eel management plans submitted under the EU regulation.

The best estimate of WGEEL is that glass eel fisheries in 2011 distributed 12% of their total catch to restocking, 30% to Aquaculture. The fate of the remainder is unknown. There are insufficient traceability systems in place to improve upon this level of detail. This poor data reporting makes it difficult to provide accurate advice on the fate of glass eel and the proportions and mortalities of glass eel set aside and used for stocking.

Giving priority to the recovery of the European stock, the objective of any stocking exercise should be to maximize net benefit to the stock as a whole until clear signs of recovery. However, stocking with an element of fishery support, combined with maintaining some spawner escapement, is not excluded in the EU Regulation. Given the current assessment of the overall stock, stocking, where it occurs, should be in conjunction with reductions in fisheries (yellow and silver) mortality and other direct mortalities (e.g. turbine, pumping stations) affecting the stocked eels. Stocking should not be seen as a substitute for reducing mortality, but as an additional measure.

The Regulation contains an obligation that up to 60% of the catch of eel less than 12 cm is used for stocking. WGEEL 2011 makes the management recommendation that this 60% should be stocked in areas where anthropogenic mortality is minimal and environmental quality is high. Those wishing to stock to support fisheries or to mitigate against other anthropogenic mortalities should draw on stock from the remaining 40% allocated by management to other uses.

The burden of proof that stocking will generate net benefit in terms of spawner escapement rests with those taking the stocking action. Prior to stocking, or for continuing existing stocking, a risk assessment should be conducted, taking into account fishing, holding, transport and post-stocking mortalities and other factors such as disease and parasite transfers. WGEEL 2011 offer the TRANSLOCEEL model as a framework for assisting with such risk assessments. The best available parameters or estimates of mortality in source and supplied areas should be used as inputs to the model.

Preliminary results of the latest tracking studies show, that as far as we can track any eel, coastal and oceanic migration routes and behaviour patterns of eel of stocked origin are indistinguishable from those derived from naturally immigrated recruits.

7 Marine Strategy Framework Directive

Chapter 7 addresses five new ToRs inserted into the WGEEL ToRs during the intersessional period in order to support the advisory process on the MSFD.

h) Identify elements of the EGs work that may help determine status for the eleven Descriptors set out in the Commission Decision (available at http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2010:232:0014:0024:EN:PDF;

i) Provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status;

j) take note of and comment on the Report of the Workshop on the Science for area-based management: Coastal and Marine Spatial Planning in Practice (WKCMSP) http://www.ices.dk/reports/SSGHIE/2011/WKCMSP11.pdf;

k) provide information that could be used in setting pressure indicators that would complement biodiversity indicators currently being developed by the Strategic Initiative on Biodiversity Advice and Science (SIBAS). Particular consideration should be given to assessing the impacts of very large renewable energy plans with a view to identifying/predicting potentially catastrophic outcomes;

l) identify spatially resolved data, for e.g. spawning grounds, fishery activity, habitats, etc.

7.1 Introduction

To comply with the obligation to implement the Marine Strategy Framework Directive (MSFD), ACOM has recently inserted a task to all Expert Groups, asking them to address new ToRs on MSFD. Term of Reference tasked the WGEEL to identify elements that may help determine status for the eleven Descriptors set out in the Commission Decision of 1 September 2010 on ***Criteria and methodological standards on good environmental status of marine waters***, and provide views on what good environmental status (GES) might be for those descriptors. To respond, at least in part, to this request, the WG has debated the subject and the possibility/adequacy of providing those elements indicating the descriptors, the criteria as well as the indicators related to them, which could be used in the case of the European eel. The need to set targets for those indicators in order to assess progress towards GES in a near future was also taken into consideration while selecting the descriptors. Information on regions/subregions where data exists was gathered whenever possible/available.

7.2 The eel and the MSFD

The MSFD requires that each Member State develops a marine strategy for its marine waters which, despite being specific to its own waters, reflects the overall perspective of the marine region or subregion concerned. These marine strategies should culminate in the execution of programmes of measures designed to achieve or maintain good environmental status. The definition of criteria and the selection of indicators to assess GES is an important step in the monitoring process of the implementation of this directive. Hence, the Commission Decision (1 September 2010) set the criteria and methodological standards on good environmental status as a starting point for the development of coherent approaches in the preparatory stages of marine strategies, including the determination of characteristics of good environmental status and the establishment of a comprehensive set of environmental targets, to be associated in a coherent and coordinated manner in the framework of the requirement of regional cooperation (Directive 2008/56/EC).

The need to address new ToRs on MSFD gave rise to a debate on the subject and the collection of information available for each marine region and/or subregion considered in the MSFD during the present meeting of the WGEEL.

The first question raised in this debate was: ***Can we consider the eel as a good candidate species for assessing progress towards GES of marine waters?***

This question raised a discussion on how to cope with the biological aspects of the European eel, such as long distance migration; panmictic population; diadromous species with a wide distribution range; complex life cycle with several life phases, not completely understood; late sex differentiation; and sex related differences in growth, and yet be able to assess environmental status at each marine region/subregion. Compared to other marine species, the European eel might not be the best candidate for assessing progress towards GES of marine waters because good biological indicators to evaluate the ecological status of a specific area should preferably use more sedentary species with a short lifespan, provided good knowledge of their life cycle exists, which is not the case for the eel. Nevertheless, the analysis of the descriptors listed in the MSFD and the evaluation of the available information on the species for those descriptors allowed the WGEEL to consider the species for assessing GES of marine waters.

7.3 Selection of descriptors and indicators

To proceed with the request from ACOM, the WGEEL conducted an analysis on the ***Criteria and methodological standards on good environmental status of marine waters*** (Commission Decision, 1 September 2010) during the meeting, to reply to the following questions:

- Which descriptors can we select?
- Which indicators are the most appropriate?
- In which marine regions or subregions can they be applied/does information exist?

The descriptors selected by the WG included, in a first approach, the following: **D1**-Biological Diversity is maintained; **D3**-Populations of all commercially exploited fish and shellfish are within safe biological limits; **D9**-Contaminants in fish and other seafood for human consumption do not exceed levels established; and **D11**-Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment. However, when trying to set indicators for each descriptor selected it was decided to exclude D11 due to difficulties associated with assessing the impact of renewable energy on the eel. Despite the existence of scientific evidence showing that eels can use electromagnetism for orientation when migrating to reproduce (Oehman *et al.*, 2007), currently it is not possible to quantify the impact of electromagnetic fields on marine species.

The results of the work developed to address new ToRs on MSFD during this session of the WG are presented in Table 7.1. For descriptor 1 there is a list of the available information for the marine regions/subregions. As can be observed the Baltic Sea and the Great North Sea are the marine subregions where there is more information.

As for **D9**, it is highly recommended that assessment and monitoring of environmental status within the scope of MSFD is done exclusively for yellow eels caught in the marine region/subregion concerned to avoid catching animals that are migrating to the sea after having spent most of their life in river basins draining to those regions, which would render the evaluation of the ecological quality of an area a difficult or impossible task.

7.4 ToR l. Spatially resolved data

Currently there is no spatially resolved database of eel catch, fisheries or survey data that is available relevant to MFSD. Recruitment time-series data are spatially linked, usually to either an estuary, or a fixed point at or near the tidal interface of a river (see Chapter 2.).

The Workshop on Baltic Eel has made initial attempts to collate spatial fisheries data for the whole of the Baltic Region (see Chapter 1.4).

It is envisaged by WGEEL, through improved DCF for eel and some planned data exchange and reporting under the EU Regulation (commencing in 2012) that an eel database will be established and this may support the analysis of spatially resolved eel data. The lack of marine fisheries for eel, outside the Baltic and Skagerrak/Kattegat areas, make it difficult to improve on collection of data on eel in the marine areas.

7.5 Conclusions/recommendations on MFSD

The European eel is classified as threatened by OSPAR and HELCOM conventions. This species can be used as a candidate species to evaluate the status of marine regions in the implementation of the MSFD. However, because it is a diadromous species with a wide distribution and with a life cycle that is not completely understood as such some caution has to be taken when selecting the phase of their life (glass eel, yellow eel or silver eel) to assess the environmental status of marine regions as they can incorporate pressures occurring in all the habitats they occupy during their lifespan.

Despite the existence of other (sedentary) species that can be used as better indicators of certain pressures in the marine regions it can be concluded that the European eel can be used to assess the environmental status for descriptors D1 and D3 and D9, especially for the Baltic Sea and the North Sea, as shown in Table 7.1. It is however, recommended that the choice of the life stage is made in accordance with the proposal presented in the same table.

Table 7-1. Descriptors, criteria and indicators selected on MSFD. Available data for each marine region/subregion was registered whenever possible.

			BALTIC SEA	NORTHEAST ATLANTIC OCEAN			MEDITERRANEAN SEA
Descriptors	Criteria	Indicators	Baltic	Greater North Sea	Celtic seas	Remaining Atlantic area	Mediterranean
D1	Species distribution	Distributional range	Yes	Yes	No	??	??
Biological diversity (Species level)		Distributional pattern	Yes	Yes	No	??	??
		Area covered by the species	No	Yes	No	??	??
	Population size	Abundance and/or biomass	No	Yes	No	??	??
	Population condition	Demographic characteristics	Yes	Yes	No	??	??
		Genetic structure	Yes	Yes	No	??	??
D1	Habitat distribution	Distributional range	No	Yes	Yes	??	??
Biological diversity (Habitat level)		Distributional pattern	Incomplete	Yes	Yes	??	??
	Habitat extent	Habitat area	No	No	No	??	??
		Habitat volume	Not relevant	Not relevant	Not relevant	Not relevant	Not relevant
	Habitat condition	Condition typical species & communities	Yes	Yes	No	??	??
		Relative abundance and/or biomass	Incomplete	Incomplete	No	??	??
		Physical hydrological & chemical condition	Yes	Yes	No	??	??
D1	Ecosystem structure	Composition & relative % of ecosystem components	Yes	Yes	No	??	??
Biological diversity (Ecosystem level)							

			BALTIC SEA	NORTHEAST ATLANTIC OCEAN			MEDITERRANEAN SEA
Descriptors	**Criteria**	**Indicators**	Baltic	Greater North Sea	Celtic seas	Remaining Atlantic area	Mediterranean
D3	Level of pressure fishing activity	Fishing mortality					
Fish and shellfish exploitation		Ratio catch/biomass					
	Reproductive capacity	SSB					
		Biomass indices					
D9 (Y)	Quantification of contaminants	Levels of contaminants detected & number of contaminants exceeding regulatory levels					
Contaminants in fish for human consumption		Frequency of levels exceeded					

(Y) To apply only to yellow eels.

?? No information could be obtained from the participants in the meeting.

8 Research needs

1) International Stock Assessment of the Eel Stock in support of the EU Regulation for Eel Stock Recovery and CITES trade restrictions:

Mortality based indicators and reference points routinely refer to mortality levels assessed in (the most) recent years. ICES (2011) noted that the actual spawner escapement will lag behind, because cohorts contributing to recent spawner escapement have experienced earlier mortality levels before. As a consequence, stock indicators based on assessed mortalities do not match with those based on measured spawner escapement. There is therefore, a need for both biomass and mortality reference points.

Biomass/density assessment

- An international calibration and standardization of eel standing stock estimates. Calibration between e-fishing streams, cpue in lakes, estuaries, and other large waterbodies; standardization and intercalibration between methods. Links to DCF, WFD and EU Regulation.
- Development of assessment methods for eels in large waterbodies (e.g. lakes, estuaries, open sea). Links to SGAESAW, DCF, WFD & EU Regulation.
- An EU-wide approach to assessing stocking and determining net benefit to the stock. Links to EU traceability, CITES, EU Regulation and ICES advice.
- Assess whether density-dependent influences (DD) on eel population dynamics occur at the local level and whether DD will play a role at the continental scale in the decline/recovery of the eel stock.

Mortality assessment

- The stock response to implemented management actions, in terms of silver biomass, will be slow and difficult to monitor. There is a need for developing methods for quantifying anthropogenic mortalities and their sum 'lifetime mortality' and estimating same across Europe. Links to DCF, WFD, EU Regulation.
- Determine impacts of different contaminants and pathogens on the 'quality' of silver eels (individuals and stock) in terms of their potential for spawning migration, fecundity and fertility. This will require sensitivity thresholds to be quantified for eel and for these to be applied at the local stock level in conjunction with the European Eel Quality Database. Links to EU Regulation & eel stock recovery.

2) Socio-economics of the small-scale fisheries, with reference to eel.

Fisheries on all continental life stages are found throughout the distribution area. Impacts vary from almost nil to heavy overexploitation. The EU Regulation delegates the processes of assessing and managing the fisheries to the Member States, and quantification of fishing mortalities is foreseen by 2012. With the implementation of the management plans and the decline in the stock, a progressive restriction or collapse of local small-scale fisheries is foreseen. This change will come to the detriment of culture and heritage (e.g. fishing techniques, skills, gastronomy).

There is a need to establish a project to examine and document the eel fisheries, their extent (past & present), socio-economic status and future as follows:

- The cultural, historic and socio-economic setting interferes with straightforward fish stock management, in the sense that individual fishers and their organizations seek to balance objectives. Because of the small scale of eel fisheries, these aspects so far have hardly been monitored, and the interaction with stock management not foreseen. Quantification of historical case studies, understanding the processes and interactions involved is required.
- Establish the previous status of the eel fishery in the various regions and river basins with a **historical** approach that would permit a reference situation on a social, economic and ecological point of view: what we had, what we have lost (what is the loss), what could be the objective of restoration.
- Ensure that the continuation and development of an eel fishery meets with the conditions of sustainable development and exploitation.

These need to use basic biological and socio-economic indicators (see Indicang). For practical reasons, the historical approach generally only uses these basic indicators particularly in socio-economic (population, production, turnover) but the project could involve research with sociology and economic indicators that require the application of methods geared to these fields.

9 References

Abulmonem or Abdelmonem, A.A., Metwally, M.M., Hussein, H.S., Hany, Elsheikha H.M. 2010. Gross and microscopic pathological changes associated with parasitic infection in European eel (*Anguilla anguilla*, Linnaeus 1758). Parasitol Res (2010) 106:463–469.

Adam, B., Bosse, R., Dumont, U., Hadderingh, R., Joergensen, L., Kalusa, B., Lehmann, G., Pischel, R. and Schwevers, U. 2005. Fish protection technologies and downstream fishways. Dimensioning, design, effectiveness inspection. DWA German Association for Water, Wastewater and Waste, Hennef, Germany.

Als, T.D, Hansen, M.M., Maes, G.E., Castonguay, M., Riemann, L., Aarestrup, K., Munk, P., Sparholt, H., Hanel, R. and L. Bernatchez. 2011. All roads lead to home: Panmixia of European eel in the Sargasso Sea. Molecular Ecology 20 (7): 1333–1346.

Andersen, S. M., Teilmann, J., Harders, P. B., Hansen, E. H., and Hjøllund, D. 2007. Diet of harbour seals and great cormorants in Limfjord, Denmark: interspecific competition and interaction with fishery. ICES Journal of Marine Science 64: 1235–1245.

Andrello, M., Bevacqua, D., Maes, G. E. and De Leo, G. A. 2011. An integrated genetic-demographic model to unravel the origin of genetic structure in European eel (*Anguilla anguilla* L.). *Evolutionary Applications.* 4 (4): 517–533.

Anonymous. 1987. (French) Rapport du comité d'experts sur la pollution transfrontière du Rhin. La Gazette Officielle de la Pêche 920–921, p 27, 925–926, p 55.

Antunes C. 1999. *Anguillicola* infestation of eel population from the Rio Minho (North of Portugal). ICES-EIFAC, 20-24 September, Silkeborg, Denmark.

APEM. 2007. River Dee fish entrainment study. APEM Scientific Report UU 886. APEM Aquatic Scientists, Heaton Mersey, Stockport. 188 pp.

Aström, M. and Dekker, W. 2007. When will the eel recover? A full life-cycle model. *ICES Journal of Marine Science.* 64 (7): 1491–1498.

Balint T, Ferenczy J, Katai F, Kiss I, Kraczer L, Kufcsak O, Lang G, Polyhos C, Szabo I, Szegletes T, Nemcsok J. 1997. Similarities and differences between the massive eel (*Anguilla anguilla* L) devastations that occurred in Lake Balaton in 1991 and 1995. Ecotox Environ Safe 37(1): 17–23.

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

Beaulaton, L. and Briand, C. 2007. Effect of management measures on glass eel escapement. ICES Journal of Marine Science, 64: 1402–1413.

Behrmann-Godel, J. and Eckmann, R. 2003. A preliminary telemetry study of the migration of silver European eel (*Anguilla anguilla* L.) in the River Mosel, Germany. Ecology of Freshwater Fish 12: 196–202.

Belpaire, C., Goemans, G. 2007. The European eel *Anguilla anguilla*, a rapporteur of the chemical status for the Water Framework Directive? 12 years of monitoring in Flanders. Vie et Milieu – Life and Environment, 57, 235–252.

Belpaire, C., Geeraerts, C., Evans, D., Ciccotti, E., Poole, R. 2011a. The European eel quality database: towards a pan-European monitoring of eel quality. Environ Monitor Assess 2011.

Belpaire, C., Geeraerts, C., Roosens, L., Neels, H., Covaci, A. 2011b. What can we learn from monitoring PCBs in the European eel? A Belgian experience. Environment International 37: 354–364.

Berg, R. 1986. Fish passage through Kaplan turbines at a power plant on the River Neckar and subsequent eel injuries. Vie Milieu 36: 307–310.

Bernatchez, L, C. Coté, M. Castonguay. 2011. Genetic structure of the American eel with emphasis on the St Lawrence river basin. Great Lakes Fishery Commission. Project Completion Report. 41 p.

Bernies, D., Brinker, A., and Daugschies, A. 2011. An invasion record for the swimbladder nematode *Anguillicoloides crassus* in European eel *Anguilla anguilla* in a deep coldmonomictic lake, from invasion to steady state Journal of Fish Biology (2011) 79, 726–746.

Bevacqua D., Gatto M., De Leo G. A. and Melià. P. 2009. Size selectivity of fyke nets for European eel *Anguilla anguilla*. Journal of Fish Biology 74 (9): 2178–2186.

Bevacqua D., Melià, P., DeLeo, G. A. and Gatto M. 2011. Intraspecific scaling of natural mortality in fish: the paradigmatic case of the European eel. Oecologia. 165: 333–339.

Birzaks, J. 2011. Report on the eel stock and fishery in Latvia 2010/2011. (Country Report for the WGEel 2011).

Bonhommeau, S. 2009. Estimates of the mortality and the duration of the trans-Atlantic migration of European eel *Anguilla anguilla* leptocephali using a particle tracking model, Journal of Fish Biology 74, (9): 1891–1914.

Bonhommeau, S., Castonguay, M., Rivot, E., Sabatié, R. and Le Pape, O. 2010. The duration of migration of Atlantic Anguilla larvae. *Fish and Fisheries.* 11 (3): 289–306.

Borgsteede, F.H.M., Haenen, O. L. M, De Bree, J., Lisitsina, O. I. 1999. Parasitic infections of European eel (*Anguilla anguilla* L.) in the Netherlands. Helminthologia 36, 251–260.

Boubée, J., Jellyman, D. and Sinclair, C. 2008. Eel protection measures within the Manapouri hydro-electric power scheme, South Island, New Zealand. Hydrobiologia 609: 71–82.

Brämick, U. 2007. Ergebnisse von Schadensabschätzungen in Binnengewässern am Beispiel des Aal. BFN Schriftenreihe 204, 145–155.

Briand C., Bonhommeau S., Beaulaton L. and Castelnaud G. 2008. An appraisal of historical glass eel fisheries and markets: landings, trade routes and future prospect for management. In The Institute of Fisheries Management Annual Conference 2007 (ed. C. Moriarty), Westport, Ireland.

Briand, C., Sauvaget, B., Beaulaton, L., Girard, P., Véron, V. and Fatin, D. 2009. Push net fisheries are responsible for injuries and post fishing mortalities in glass eel (*Anguilla anguilla*) Technical note IAV.16 p.

Calles, O. and Bergdahl, D. 2009. Ålens nedströmspassage av vattenkraftverk. - Före og efter åtgärd. Forskningsrapport, Karlstad University Studies 2009:19, 41 pps. (In Swedish).

Calles, O., Olsson, I.C., Comoglio, C., Kemp, P.S., Blunden, L., Schmitz, M. and Greenberg, L.A. 2010. Size-dependent mortality of migratory silver eels at a hydropower plant, and implications for escapement to the sea. Freshwater Biology 55: 2167–2180.

Carpentier, A., Marion, L., Paillisson, J.-M., Acou, A. and Feunteun, E. 2009. Effects of commercial fishing and predation by cormorants on the *Anguilla anguilla* stock of a shallow eutrophic lake Journal of Fish Biology. 74: 2132–2138.

Carr, J.W. and Whoriskey, F.G. 2008. Migration of silver American eels past a hydroelectric dam and through a coastal zone. Fisheries Management and Ecology 15: 393–400.

Carss, N. and Ekins, G. 2002. Further European integration: mixed subspecies colonies of great cormorants *Phalacrocorax carbo* in Britain – colony establishment, diet, and implications for fisheries management. ARDEA 90 (1): 23–41.

Castelnaud G. and Beaulaton L. 2008. Indicateurs d'abondance et de pression issus des pêcheries. Chapitre 6 in L'anguille Européenne, indices d'abondance et de colonisation, Adam G., Feunteun E., Prouzet P., Rigaud C. (Coord.), Projet Européen Intereg Indicang. Edition QUAE. pp. 189–220.

Christou M.D. 2000. Substances dangerous to the environment in the context of Council Directive 96/82/EC. Report by Technical Working Group 7 EUR 19651 EN. pp 43.

Churchward, A. S. 1996. Overview of the Severn fishery, England. EIFAC (FAO) Working Party on Eel, September 1996 in IJmuiden, Holland.

Ciccotti, E. 2005. Interactions between capture fisheries and aquaculture: the case of the eel (*Anguilla anguilla* L., 1758). In: "Interactions between Capture Fisheries and Aquaculture: a methodological perspective?, Cataudella S., Massa F. & D. Crosetti Eds, Studies and Reviews, General Fisheries Commission for the Mediterranean. N. 78, Rome, FAO, 2005: 190–203.

Clevestam, P. and Wickström, H. 2008. Rädda ålen och ålfisket! -Ett nationellt bidrag till en Europeisk förvaltningsplan. Slutrapport från pilotprojekt till Fonden för fiskets utveckling Dnr: 231-1156-04, 42 (in Swedish).

Clevestam, P., Ogonowski, M., Sjöberg, N. B. and Wickström, H. 2011. Too short to spawn? Implications of small body size and swimming distance on successful migration and maturation of the European eel *Anguilla anguilla*. Journal of Fish Biology 78(4): 1073–1089.

Consolo, M., Utmar, P., Roppa, F., Sponza, S. 2009. Interactions between fish resources and Cormorants Phalacrocorax carbo in the Grado and Marano lagoon (NE Italy). Acrocephalus 30, 17–23.

Costa-Dias S., Dias E., Lobón-Ceviá J., Antunes C. and Coimbra J. 2010. Infection by *Anguillicoloides crassus* in a riverine stock of European eel, *Anguilla anguilla*. Fisheries Management and Ecology, 2010.

Coté, C. L., Castonguay, M., Verreault, G. and Bernatchez, L. 2009. Differential effects of origin and salinity rearing conditions on growth of glass eels of the American eel *Anguilla rostrata*: implications for stocking programmes. Journal of Fish Biology 74: 1934–1948.

CVUA. 2010. Report - Chemisches und Veterinäruntersuchungsamt Freiburg, Germany 2010. Dioxine, PCB und weitere Schadstoffe in (Wild-)Fischen aus Binnengewässern in Baden-Württemberg. 27 p. http://www.cvuas.de.

Daemen E, Cross T, Ollevier F, Volckaert FAM. 2001. Analysis of the genetic structure of European eel (*Anguilla anguilla*) using microsatellite DNA and mtDNA markers. Mar. Biol. 139:755–764.

Dannewitz, J., Maes, G.E., Johansson, L., Wickström, H., Volchaert, F.A.M., and Järvi, T. 2005. Panmixia in the European eel: a matter of time? Proc. Royal Society of London, 272: 1129–1137.

Daverat *et al*. In press. One century of eel growth: temperature and simple habitat characteristics effects. Ecology of Freshwater Fish.

Davidse A., Haenen O.L.M., Dijkstra S.G., van Nieuwstadt A.P., van der Vorst T.J.K., Wagenaar F. and Wellenberg G.J. 1999. First isolation of herpesvirus of eel (*Herpesvirus anguillae*) in diseased European eel (*Anguilla anguilla*) in Europe. Bulletin of the European Association of Fish Pathologists 19, 137–141.

Davidsen, J. G., Finstad, B., Økland, F., Thorstad, E. B., Mo, T. A. & Rikardsen, A. H. 2011. Early marine migration of European silver eel (*Anguilla anguilla*) in Northern Norway. Journal of Fish Biology 78: 1390–1404.

Dekker, W. 2000. A Procrustean assessment of the European eel stock. ICES Journal of Marine Science. 57 (4): 938–947.

Dekker W. 2000. The fractal geometry of the European eel stock. ICES J. Mar. Sci., 57, 109–121.

Dekker W. 2002. Monitoring of glass eel recruitment: thematic overview. Institute for fisheries research RIVO, IJmuidden, The Netherlands.

Dekker W. 2003c. Did lack of spawners cause the collapse of the European eel, *Anguilla anguilla*? Fisheries Management and Ecology 10: 365–376.

Dekker W. 2009. A conceptual management framework for the restoration of the declining European eel stock. Pages 3–19 in J.M. Casselman & D.K. Cairns, editors. Eels at the Edge: science, status, and conservation concerns. American Fisheries Society, Symposium 58, Bethesda, Maryland.

Dekker W. 2010. Ex-post evaluation of eel stock management: a methodology under construction. IMARES report C056/10, 67 pp.

Denayer, B., Belpaire, C. 1992. Effekten van pompgemalen op vispopulaties. Schade aan witvis en stelling aan de werking van een vijzelpompgemaal. Rapport Ministerie van de Vlaamse Gemeenschap, Instituut voor Bosbouw en Wildbeheer nr. IBW.Wb.V.R.92.10.

Desrochers, D. and Letendre, M. 1994. *Anguilla rostrata*, evaluation de la mortalité d'anguille dans le haut Saint-Laurent en 1993. Ministrère de l'Environnement et de la Faune. Rapport de travaux 06-29. 38 p + 2 annexes.

Desrochers, D. 1995. Suivi de la migration de l'anguille d'Amérique *Anguilla rostrata* au complexe Beauharnois, 1994. p. 107. Saint-Bernard-de-Lacolle, Que: Rapport préparé pour Hydro-Québec par Milieu & Associés Inc.

Dias, E. 2007. Estudo da dieta do Corvo-marinho-de-faces-brancas (Linnaeus, 1758) no Estuário do Rio Minh (NO-Portugal). Master Thesis, University of Porto, Portugal.

Dieperink, C. 1995. Depredation of commercial and recreational fisheries in a Danish fjord by cormorants (*Phalacrocorax carbo sinensis*) Shaw. *Fisheries Management and Ecology* 2:197–207.

Dirksen, S., Boudewijn, T.J., Noordhuis, R., and Marteijn, E.C.L. 1995. Cormorants *Phalacrocorax carbo sinensis* in shallow eutrophic freshwater lakes: prey choice and fish consumption in the non breeding period and effects of large scale fish removal. ARDEA 83: 167–184.

Doherty, D. and McCarthy, K. 1997. The population dynamics, foraging activities and diet of great cormorants (*Phalacrocorax carbo* L.) in the vicinity of an Irish hydroelectricity generating station. Suppl. Ric. Biol. Selvaggina XXVI: 133–143.

Domingos I. 2003. A enguia-europeia, *Anguilla anguilla* (L., 1758), na bacia hidrográfica do Rio Mondego.[The European eel (*Anguilla anguilla* (L.1758) in the Mondego River catchment]. PhD thesis, Universidade de Lisboa.

Durif, C., Elie, P., Gosset, C., Rives, J. and Travade, F. 2003. Behavioural study of downstream migrating eels by radio-telemetry at a small hydroelectric power plant. American Fisheries Society Symposium 33: 343–356.

Edeline, E., Dufour, S. and Elie, P. 2005. Role of glass eel salinity preference in the control of habitat selection and growth plasticity in *Anguilla anguilla*. Marine Ecology Progress Series. 304: 191–199.

El-Hilali M., Yahyaoui A., Sadak A., Maachi M. and Taghy Z. 1996. Premières données épidémiologiques sur l'anguillicolose au Maroc. Bull. Fr. Pêche et Piscicult., 340 : 57–60.

El-Hilali M., Yahyaoui A. and Chetto N. 2005. Etude de l'infestation des anguilles (*Anguilla anguilla*) par le parasite (*Anguillicola crassus*) dans l'estuaire du Sebou au nord-ouest du Maroc. Bull. Inst. Sci., Rabat, sect. Sci. Vie, 26–27 : 39–400.

Elie, P., Girard, P. 2009. Effets des micropolluants et des organismes pathogènes chez l'Anguille européenne *Anguilla anguilla* L. 1758, Cemagref: 121.

Elonen, G. E., R. L. Spehar, *et al*. 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to seven freshwater fish species during early life-stage development. Environmental Toxicology and Chemistry **17**(3): 472–483.

Engström, H. 2001. Long term effects of cormorant predation on fish communities and fishery in a freshwater lake. Ecography 24: 127–138.

Environment Agency. 2011. Screening at intakes and outfalls: measures to protect eel. The eel manual – GEHO0411BTQD-E-E. Environment Agency, Bristol. 125 pp.

Esteve C. and Alcaide E. 2010. Temporal distribution of *Edwardsiella tarda* disease on wild eel population from Albufera lake. VIII Aquatic Microbiology meeting of the "Sociedad Española de Microbiología" Vigo (Spain) September 2010.

European Commission. 1979. Council Directive 79/409/EEC on the conservation of Wild Birds. EC: Brussels.

European Commission. Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. OffJ Eur Union 2006: 5–24.

EU. 2007. Council Regulation (EC) No 1100/2007 of 18 September 2007 establishing measures for the recovery of the stock of European eel. Official Journal of the European Union L 248/17.

Ferrante, M., C., Clausi, M. T., Meli. R., Fusco, G., Naccari, C., Lucisano, A. 2010. Polychlorinated biphenyls and organochlorine pesticides in European eel (*Anguilla anguilla*) from the Garigliano River (Campania region, Italy) Chemosphere 78 (2010) 709–716.

Feunteun. 2002.

Frear, P. A. and Axford, S. N. 1991. Impingement and mortality of fish associated with the River Ouse abstraction scheme. Fisheries Science Report 62/91. National Rivers Authority, Yorkshire Region. 34 pp.

Food and Agriculture Organization of the United Nations. 1995. Precautionary Approach to Fisheries. Part 1: Guidelines on the Precautionary Approach to Capture Fisheries and Species Introductions. FAO Fisheries Technical Paper No. 350/1, URL: http://www.fao.org/DOCREP/003/V8045E/V8045E00.HTM.

Geeraerts, G., Belpaire, C. 2010. A review of the effects of contaminants on European eel. Ecotoxicology 19, 239–266.

Geeraerts C., Focant J-F., Eppe G., De Pauw E., Belpaire C. 2011. Reproduction of European eel jeopardised by high levels of dioxins and dioxin-like PCBs? Science of the Total Environment 409: 4039–4047.

Germonpré, E., Denayer, B., Belpaire, C., Ollevier, F. 1994. Inventarisatie van pompgemalen in het Vlaamse gewest en preliminair onderzoek naar de schade van diverse pomptypes op vissen na gedwongen blootstelling. Onderzoek uitgevoerd door AMINAL en de Katholieke Universiteit Leuven.

Germonpre E. 1995. Inventarisatie en effekt van pompgemalen voor het visbestand in Vlaanderen. KUL, thesis, Leuven, 1993.

Gomes P. and Larinier M. 2008. Dommages subis par les anguilles lors de leur passage au travers des turbines Kaplan. Etablissement de formules prédictives.

Gosset, C., Travade, F., Durif, C., Rives, J. and Elie, P. 2005. Tests of two types of bypass for downstream migration of eels at a small hydroelectric power plant. River Research and Applications 21: 1095–1105.

Gravato C., Guimarães L., Santos J., Faria M. and Alves A. 2010. Comparative study about the effects of pollution on glass and yellow eels (*Anguilla anguilla*) from the estuaries of Minho, Lima and Douro Rivers. Ecotoxicology and Environmental Safety, 73:524–533.

Hadderingh, R. H. 1979. Fish intake mortality at power stations the problem and its remedy. Aquatic Ecology Volume 13, Numbers 2–3, 83–93, DOI: 10.1007/BF02284734.

Hadderingh, R. H., Van Aerssen, G. H. F. M., Groeneveld, L., Jenner, H. A. and Van der Stoep, J. W. 1983. Fish impingement at power stations situated along the rivers Rhine and Meuse in the Netherlands. Hydrobiological Bulletin 17: 129–141.

Hadderingh, R.H. & Bakker, H.D. 1998. Fish mortality due to passage through hydroelectric power stations on the Meuse and Vecht Rivers. I Jungwirth, M., Schmutz, S. & Weiss, S. (eds.) Fish Migration and Bypasses. Fishing New Books, Oxford.

Hadderingh RH and Bruijs MCM. 2002. Hydroelectric power stations and fish migration. Tribune de l'eau 55(19-620/5-6: 79–87.

Haenen, O.L.M., 1995. *Anguillicola crassus* (Nematoda, Dracunculoidea) infections of European eel (*Anguilla anguilla*) in the Netherlands: epidemiology, pathogenesis and pathobiology. Thesis with hypotheses, LU Wageningen, 127 pag.

Haenen, O.L.M. and Davidse, A. 2001. First isolation and pathogenicity studies with *Pseudomonas anguilliseptica* from diseased European eel *Anguilla anguilla* (L.) in The Netherlands. Aquaculture 196: 27–36.

Haenen, O.L.M., van Ginneken, V.J., Engelsma, M.Y. and van den Thillart G.E.E.J.M. 2009. Impact of eel viruses on recruitment of European eel. Chapter 16 in: Spawning Migration of the European Eel. Reproduction Index, a Useful Tool for Conservation Management Ed. Vd Thillart, Dufour and Rankin. Fish and Fisheries Series, 480p: 387–400. Springer, Germany, ISBN978-1-4020-9094-3 (Print) 978-1-4020-9095-0 (Online).

Haenen, O.L.M., J. Lehmann, M.Y. Engelsma, F.-J. Stürenberg, I. Roozenburg, S. Kerkhoff, and J. Klein Breteler. 2010. The health status of European silver eels, *Anguilla anguilla,* in the Dutch River Rhine watershed and Lake IJsselmeer. Aquaculture 309: 15–24.

Hald P. 2007. Skarvernes Fødevalg ved Hirsholmene i årene 2001–2003. http://www.sns.dk/publikat/2001/hirsholmen_skarv_2001_2003.pdf.

Haro, A., Castro-Santos, T. and Boubée, J. 2000. Behaviour and passage of silver-phase American eels, *Anguilla rostrata* (LeSueur), at a small hydroelectric facility. Dana 12: 33–42.

Helfman, G.S., Facey, D.E., Stanton Hales, L. and Bozeman, E.L. 1987. Reproductive ecology of the American eel. Am. Fish. Soc. Symp. 1, 42–56.

Holzner, M. 1999. Vermeidung von Fischschäden im Kraftwerksbereich. Schriftenreihe Bayr. Landesfischereiverb. 1: 224 pp.

Hoogenboom L.A.P., van Klaveren J.D., Baars A.J., van Leeuwen F.X.R., Hoogerbrugge R., van Leeuwen S.P.J., De Boer J. 2001. Scenario studies on maximum levels for dioxins, dibenzofurans and dioxin-like PCBs in fish. RIVM report 639102 023 project 639102, RIVM, IJmuiden, NL. 30 pp.

Hoogenboom, L. A. P., M.J.J. Kotterman, M. Hoek-van Nieuwenhuizen, M.K. van der Lee, W.A. Traag. 2007. Onderzoek naar dioxines, dioxineachtige PCB's en indicator-PCB's in paling uit Nederlandse binnenwateren 2006. RIKILT IMARES Rapport 2007.003.

Ibbotson, A., Smith, J., Scarlett, P. and Aprhamian, M. 2002. Colonisation of freshwater habitats by the European eel *Anguilla anguilla*. *Freshwater Biology*, **47**, 1696–1706.

ICES. 2000. Report of the EIFAC/ICES Working Group on Eels, Silkeborg, Denmark, 20–24 September 1999. ICES C.M. 2000/ACFM:6.

ICES. 2001. International Council for the Exploration of the Sea. Report of ICES/EIFAC Working Group on Eels. ICES C.M. 2001/ACFM:03.

ICES. 2002. International Council for the Exploration of the Sea. Report of ICES/EIFAC Working Group on Eels. ICES C.M. 2002/ACFM:03.

ICES. 2004. Report of ICES/EIFAC Working Group on Eels, 7–11 October 2003, Sukarrieta, Spain. ICES CM 2004/ACFM:09.

ICES. 2004. Report of ICES Advisory Committee on Fishery Management and Advisory Committee on Ecosystems. ICES Advice, Vol 1, Nº 2. 1544 pp.

ICES. 2006. Report of the 2006 Session of the Joint EIFAC/ICES Working Group on Eels. CM2006/ACFM, 16: 352p.

ICES. 2007. Report of the 2007 Session of the Joint EIFAC/ICES Working Group on Eels. ICES Advisory Committee on Fishery Management. ICES CM 2007/ACFM:23.

ICES. 2008. The report of the 2008 Session of the Joint EIFAC/ICES Working Group on Eels, September 2008; *ICES CM 2008/ACOM:15.192pp* and country reports.

ICES. 2009. The report of the 2009 Session of the Joint EIFAC/ICES Working Group on Eels, September 2009; *ICES CM 2009/ACOM:15.117pp* and country reports.

ICES. 2010a. The report of the 2010 Session of the Joint EIFAC/ICES Working Group on Eels, September 2010; *ICES CM 2009/ACOM:18.198pp* and country reports.

ICES. 2010b. Report of the Study Group on International Post-Evaluation on Eels (SGIPEE), 10–12 May 2010, Vincennes, France. ICES CM 2010/SSGEF:20. 42 pp.

ICES. 2011. Report of the Study Group on International Post-Evaluation on Eels (SGIPEE), 24–27 May 2011, London, UK; ICES CM 2011/SGEF:13. 39pp.

ICES WGAGFM. 2004. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture 2004.

ICES WGAGFM. 2007. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture 2007.

INBO Fishing mortality caused by pumping stations; focus on European Eel, presentation.

Jansen, H.M., Winter, H.V., Bruijs, M.C.M. and Polman, J.G. 2007. Just go with the flow? Route selection and mortality during downstream migration of silver eels in relation to river discharge. ICES Journal of Marine Science 64: 1437–1443. Rhapport GHAAPPE RA08.05. pp 70.

Järvalt, A.; Kask, M.; Krause,T., Palm, A.; Tambets, M.; Sendek, D. 2010. Potential Downstream Escapement of European Eel From Lake Peipsi Basin. 2010 (467, 6), 1–11. http://balwois.com/balwois/administration/full_paper/ffp-1789.pdf.

Jepsen, N., Reinhard K., Per, S., Bregnballe, T. 2010. The use of coded wire tags to estimate cormorant predation on fish stocks in an estuary. Marine and Freshwater Research, 2010, 61: 320–329.

Jørgensen P., Castric J., Hill B., Ljungberg O. and de Kinkelin P. 1994. The occurrence of virus-infections in elvers and eels (*Anguilla anguilla*) in Europe with particular reference to VHSV and IHNV. Aquaculture 123, 11–19.

Julshamn, K., and Frantzen, S. 2009. Miljøgifter i fisk og fiskevarer - en rapport om dioksiner og dioksinlignende PCB, polybromerte flammehemmere og tungmetaller i oljer, makrell, ål og Svolværpostei. Årsrapport 2008. Miljøgifter i fisk og fiskevarer.

Jürgens, M., Johnson, A., Chaemfa, C., Jones, K., Hughes, D. 2009. The organic chemical contamination of eels in the lower Thames in 2007. CEH Wallingford, Lancaster Environment Centre, Lancaster University.

Kajiwara, Y., Kashiwagi, N., Kadokami, K. Nationwide study of dioxins in the freshwater fish *Carassius auratus* (gibelio) *langsdorfii* (crucian carp) in Japan: Concentrations and estimation of source contribution ratios. Chemosphere 69 (2007) 1177–1187.

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

Karl H., Bladt A., Rottler H., Ludwings R. and Mathar W. 2010. Temporal trends of PCDD, PCDF, and PCB levels in musce meat of herring from different fishing grounds of the Baltic Sea and actual data of different fish species from the Western Baltic sea. Chemosphere 78: 106–112.

Keller, T. 1995. Food of cormorants *Phalacrocorax carbo sinensis* wintering in Bavaria, Southern GermanyARDEA 83(1): 85–82.

Kemp, P. S.; Russon, I. J.; Waterson, B.; O'Hanley, J. and Pess, G. R. 2008. Recommendations for a "Coarse-Resolution Rapid-Assessment" Methodology to assess barriers to fish migration, and associated prioritization tools. Appendixes: Barriers to fish Migration Survey: Assessment Criteria and Procedure. Extended Field Manual. SNIFFER. WFD111.

Kennedy C.R. 2007. The pathogenic helminth parasites of eels. Journal of Fish Diseases 30, 319–334.

Kennish, M. J. 1992. Ecology of estuaries: anthropogenic effects. Boca Raton, FL: CRC Press.

Kheyyali D., Lachheb K., Yahyaoui A. and Hossaini-Hilali J. 1999. Status of European Eel infestation by the nematode *Anguillicola crassus* in aquatic ecosystems in Morocco. Actes Inst. Agron. Vet., 19: 177–180.

Kibel P. 2008. Archimedes Screw Turbine Fisheries Assessment. Phase II: Eels and Kelts.

Kloppmann, M. 1999. Untersuchungen zur Effizienz der Fischscheuchanlage der Müllverwertung Rügenberger Damm, Zwischenbericht. Auftraggeber: Müllverwertung Rügenberger Damm GmbH & Co. KG/.

Knights, B. and White, E.M. 1998. Enhancing immigration and recruitment of eels: the use of passes and associated trapping systems. Fisheries Management and Ecology 5: 459–471.

Knösche, R. 2005. Untersuchung zur Entwicklung der Fischerei im Land Brandenburg unter Beachtung der Kormoranbestände und Entwicklung eines Monitorings. Abschlussbericht, pp.:121.

Knutzen J, Brevik E, Green N, Biseth A, Egaas E, Schlabach M, Skare JU. 1998. (Norwegian) Overvaking av miljøgifter i fisk og skalldyr fra Grenlandsfjordene 1996. (Monitoring of micropollutants in fish and shellfish from the Grenland fjords (S. Norway 1996). OR-3834. Norsk institutt for vannforskning (NIVA), pp 150.

Knutzen J, Becher G, Biseth A, Bjerkeng B, Brevik E, Green N, Schlabach M, Skare JU. 1999. (Norwegian) Overvaking av miljøgifter i fisk og skalldyr fra Grenlandsfjordene 1997. (Monitoring of micropollutants in fish and shellfish from the Grenlandsfjords (S. Norway) 1997. OR-4065. Norsk institutt for vannforskning (NIVA), pp 195.

Knutzen J, Bjerkeng B, Green NW, Kringstad A, Schlabach M, Skare JU. 2001. (Norwegian) Overvaking av miljøgifter i fisk og skalldyr fra Grenlandsfjordene (Monitoring of micropollutants in fish and shellfish from Grenlandfjordene) 2000. OR-4452. Norsk institutt for vannforskning (NIVA), pp 230.

Kohl, F. 2010. How many Cormorants in Europe? Documentation of EAA - European Anglers Alliance. Issue 01.2 EN.

Krieg, H-J., Oesmann, S., Stiller, G. and Jacobi, A. M. 2010. Literaturstudie zu den Auswirkungen von Kühlwasserentnahme und –einleitung auf das aquatische Milieu des Elbeästuars – unter besonderer Berücksichtigung von Biomasseschädigungen des Phytoplanktons, des Zooplanktons und der Fischeier und Fischlarven sowie die Folgen auf den Sauerstoffhaushalt. HUuG Tangstedt. 133 pp.

Kruitwagen, G., Klinge, M. 2007. Monitoring van stroomafwaartse migratie van vis bij gemaal Katwijk. Rapport Witteveen+Bos in opdracht van het Hoogheemraadschap van Rijnland.

Kruitwagen, G., Klinge, M. 2008a. Monitoring van stroomafwaartse migratie van vis bij de gemalen Halfweg, Spaarndam en Gouda. Rapport Witteveen+Bos in opdracht van het Hoogheemraadschapvan Rijnland.

Kruitwagen, G., Klinge, M. 2008b. Sterfte van schieraal door gemaal IJmuiden, onderzoeksjaar 2008. Rapport Witteveen+Bos in opdracht van Rijkswaterstaat Noord-Holland.

Kruitwagen, G., Klinge, M. 2008c. Stroomafwaartse passage van vis via gemaal Schoute. Rapport Witteveen+Bos in opdracht van de 's-Gravenhaagse Hengelsport Vereniging.

Kunst JM, Spaargaren B, Vriese T, Kroes M, Rutjes C, van der Pouw Kraan E, Jonker RR. 2008. Gemalen of vermalen worden; onderzoek naar visvriendelijkheid van gemalen. I&M-99065369-MK, pp.70.

Lachheb K. 1997. Contribution à l'étude de l'anguillicolose chez l'anguille européenne au Maroc. Thèse 3ème Cycle, Inst. Agron. Vét. Hassan II, Rabat. 91p.

Lambert, P. 2008. Évaluation des effets possibles de différents niveaux de réduction des impacts sur le temps de restauration du stock d'anguille européenne Rapport ONEMA-Cemagref. 15 p.

Lambert, P., Verreault, G., Lévesque, B., Tremblay, V., Dutil, J.-D. and Dumont P. 2011. Détermination de l'impact des barrages sur l'accès de l'anguille d'Amérique (*Anguilla rostrata*) aux habitats d'eau douce et établissement de priorités pour des gains en habitat. Rapp. tech. can. sci. halieut. aquat. 2921: x +43 p.

Larsson, P., Hamrin, S. and Okla, L. 1990. Fat content as a factor inducing migratory behaviour in the eel (*Anguilla anguilla* L) to the Sargasso Sea. Naturwissenschaften 77, 488–490.

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

Leonards P.E.G., Dulfer W.J., Evers E.H.G., van de Guchte K. 2005. Inventarisatie en evaluatie dioxinen in het Nederlandse aquatische milieu: status 2005. C061/05. 43 pp.

Lehmann J., Stürenberg F.J., Mock D. and Feldhaus A. 2005. Untersuchungen auf virale Infektionen bei Aalen. Fischer und Teichwirt 6, 205–206.

Limburg, K. E., Wickström, H., Svedäng, H., Elfman, M. and Kristiansson, P. 2003. Do Stocked Freshwater Eels Migrate? Evidence from the Baltic Suggests "Yes". In Biology, Management and Protection of Catadromous Eels, ed. Dixon, D. A. (American Fisheries Society, Symposium 33, Bethesda, Maryland), pp. 275–284.

Lin, Y., -J. L. Lozys, J., -C. Shiao, Y., Iizuka and W., -N. Tzeng. 2007. Growth differences between naturally recruited and stocked European eel *Anguilla anguilla* from different habitats in Lithuania. Journal of Fish Biology 71: 1773–1787.

Lorentsen, S.H., Gremillet, D., G.H. Nymoen. 2004. Annual variation in diet of breeding great cormorants: Does it reflect varying recruitment of gadoids? Waterbirds 27(2): 161–169.

Loukili A. and Belghyti D. 2007. The dynamics of the nematode *Anguillicola crassus*, Kuvahara 1974 in eel *Anguilla anguilla* (L. 1758) in the Sebou estuary (Morocco). Parasitol. Res 100: 683–686.

Maes, J., Taillieu, A., Van Damme, P. A., Cottenie, K. and Ollevier, F. 1998. Seasonal patterns in the fish and crustacean community of a turbid temperate estuary (Zeeschelde Estuary, Belgium) Estuarine, Coastal and Shelf Science 47: 143–151.

Maes, G.E., and Volckaert, F.A.M. 2002. Clinal genetic variation and isolation by distance in the European eel *Anguilla Anguilla* (L.). Biol. J. Linn. Soc. 77: 509–521.

Maes, J., Turnpenny, A. W. H., Lambert, D. R., Nedwell, J. R., Parmentier, A. and Ollevier, F. 2004. Field evaluation of a sound system to reduce estuarine fish intake rates at a power plant cooling water inlet. Journal of Fish Biology 64: 938–946.

Maes, E.G. Pujolar, J.M., Raeymaekers, C., Joost, D. J. and Volckaert, F. 2006. Microsatellite conservation and Bayesian individual assignment in four Anguilla species. Marine Ecology Progress Series 319: 251–261.

Maes, E. G., Pujolar, J. M., Hellemans, B., Volckaert, F. A. M. 2006. Evidence for isolation by time in the European eel (*Anguilla anguilla* L.). Molecular ecology 15. 2095–2107.

Kruitwagen, G., Klinge, M., Manshanden, G.A.M. 2006. Opening eerste vispassage in een gemaal.H2O 39 (9): 19.

Marine Stewardship Council. 2002. MSC Principles and Criteria for Sustainable Fishing, URL: http://www.msc.org/documents/msc-standards/MSC environmental standard for sustainable fishing.pdf.

Martínez-Carrasco, C., Ruiz De Ybanez, R., Penalver, J., Mayo-Hernandez, E., Garcia-Ayala, A., Munoz, P. Prevalence of *Anguillicoloides crassus* (Nematoda, Dracunculoidea) in wild European eels (*Anguilla anguilla* L.) from Mar Menor lagoon (Western Mediterranean, Spain). Revue Méd. Vét., (2011), 162, 3, 154–158.

Martínez-Carrasco, C., Serrano, E., Ruiz de Ybáñez, R., Peñalver, J., García, J., García-Ayala, A., Morand, S., Muñoz, P. The European eel—the swim bladder–nematode system provides a new view of the invasion paradox. Parasitol Res (2011) 108:1501–1506.

Melià, P., Bevacqua, D., Crivelli, A. J., De Leo, G. A., Panfili, J. and Gatto, M. 2006. Age and growth of *Anguilla anguilla* in the Camargue lagoons. *Journal of Fish Biology*. 68 (3): 876-890.

ML Niedersachsen. 2011. Flussfisch-Monitoring zur Ermittlung der Schadstoffbelastung in Fischen aus Niedersachsen. Niedersächsisches Ministerium für Ernährung, Landwirtschaft, Verbraucherschutz und Landesentwicklung, www.ml.niedersachsen.de.

Möller, H., Lüchtenberg, H. and Sprengel, G. 1991. Rückführung der am Einlaufrechen des Kernkraftwerkes Brunsbüttel zurückgehaltenen Fische in die Elbe. Unveröffentlichter Bericht, Auftraggeber: Kernkraftwerk Brunsbüttel GmbH, 94 S.

Monein-Langle, D. 1985. Morphologie et Physiologie digestives de la Civelle D'*Anguilla anguilla* (Linnaeus, 1758) en Phase prealimentaire dans les Conditions naturelles et en Regime thermique particulier. PhD Thesis, Académie de Montpellier, Universite de Perpignan – Laboratoire de Biologie Marine. 143 pp.

Montén, E. 1985. Fisk och turbiner. Om fiskars möjligheter att oskadda passera genem kraftverksturbiner. Vattenfall, Stockholm. 116 s.

Moriarty, C. and Nixon, E. 1990. Eel Biomass in a Small Lowland Stream. Internationale Revue der gesamten Hydrobiologie und Hydrographie, 75: 781–784. doi: 10.1002/iroh.19900750612.

Moriarty, C., and Dekker, W. 1997. Management of the European eel. Second report of the EC concerted action AIR-A94–1939. Marine Institute, Dublin.

Muñoz, P., Ruiz de Ybañez, R., Martínez-Carrasco, C., Peñalver, J., García J.A.,Cabrera, García-Ayala A. 2009. Health status of wild european eels (*Anguilla anguilla* L.) from two ecosystems of the western mediterranean: preliminary results. 14 EAFP International Conference. Praga September 2009.

Næs. K. *et al*. Miljǿgiftundersǿkelse i havner i Telemark, Vestfold, Akershus of Ǿstfold 1999. PAH, PCB, tungmetaller of TBT i sedimenter of arganismer. Statlig program for forurensningoverǻking. Overvǻkingsrapport 849/02 TA-nr. 1885/2002.

Nagel, F., Kammann, U., Wagner, C. and Hanel, R. 2011. Metabolites of polycyclic aromatic hydrocarbons (PAHs) in bile as biomarkers in European eel (*Anguilla anguilla*) from German rivers. Arch. Environ. Contam. Toxicol. DOI 10.1007/s00244-011-9693-8.

Nakata, H., Y. Sakai, *et al*. 2002. Growth-dependent and species-specific accumulation of polychlorinated biphenyls (PCBs) in tidal flat organisms collected from the Ariake Sea, Japan. Archives of Environmental Contamination and Toxicology **42**(2): 222–228.

Neto *et al*. 2010. Epidemiology and pathology of *Anguillicoloides crassus* in European eel *Anguilla anguilla* from the Tagus estuary (Portugal). Diseases of Aquatic Organisms.

Neto, A.F., Passos, D., Costa, J.L., Costa, M.J., Caçador, I., Pereira, M.E., Duarte, A.C., Pacheco, M. and Domingos, I. 2011. Accumulation of metals in *Anguilla anguilla* from the Tagus estuary and relationship to environmental contamination. Journal of Applied Ichthyology doi: 10.1111/j.143-0426.2011.01814.x.

NIMO (Niagara Mohawk Power corporation). 1995. Fish entrainment and mortality study, final report, Oswego River project, FERC No. 2474. Prepared by Kleinschmidt Associates.

NIMO (Niagara Mohawk Power corporation). 1996. Fish entrainment and mortality study, final report, Lower Raquette River project, FERC No. 2430. Prepared by Kleinschmidt Associates.

NYPA (New York Power Authority). 1998. Draft final report. Estimation of survival of American eel after passage through a turbine at the St Lawrence-FDR power project, New York. Prepared by Normendeau Associates, Inc. and J.R. Skalski.

Oehman, M. C., Sigray, P. and Westerberg, H., "Offshore Windmills and the Effects of Electromagnetic Fields on Fish," Ambio 36 (2007): 630–633.

Ontario Water Power Association. 2010.: Best management practices. Guide for American eel and waterpower in Ontario. 148 pp.

Palm, S., Dannewitz, J., Prestegaard, T., Wickstrom, H. 2009. Panmixia in European eel revisited: no genetic difference between maturing adults from southern and northern Europe. Heredity, 103, 82–89.

Palstra A.P., van Ginneken V.J.T., Murk A.J., van den Thillart G.E.E.J.M. 2006. Are dioxin-like contaminants responsible for the eel (*Anguilla anguilla*) drama? Naturwissenschaften 93(3): 145–148.

Palstra, A. P., Heppener, D. F. M., van Ginneken, V. J. T., Szekely, C. and van den Thillart, G. E. E. J. M. 2007. Swimming performance of silver eels is severely impaired by the swim-bladder parasite *Anguillicola crassus*. Journal of Experimental Marine Biology and Ecology, 352, 244–256.

Parsons J., Vickers K.U. and Warden Y. 1977. Relationship between elver recruitment and changes in sex ratio of silver eels *Anguilla anguilla* L., migrating from Lough Neagh, Northern Ireland. Journal of Fish Biology 10, 211–229.

Passos D.M. 2008. Concentração de metais pesados na enguia europeia, *Anguilla anguilla* (Linnaeus, 1758), em estuários e lagoas costeiras de Portugal [Heavy metal concentration in the European eel, *Anguilla anguilla* (Linnaeus, 1758), in Portuguese estuaries and coastal lagoons]. Licenciateship in Biology, Universidade de Aveiro.

Patrick, P. H., and R. S. Mckinley. 1987. Field evaluation of a Hidrostal pump for live transfer of American eels at a hydroelectric facility. North American Journal of Fisheries Management 7: 303–305.

Pedersen, M.I. 2009. Does stocking of Danish Lowland Streams with elvers increase European eel populations? *American Fisheries Scociety Symposium*, **58**; 149–156.

Pedersen, M. I. 2010. Effektvurdering af aleudsaetninger i Roskilde Fjord. DTU Aqua-rapport nr. 230; 2010. 31 Seiten.

Pierron F., Baudrimont M., Dufour S., Elie P., Bossy A., Baloche S., Mesmer-Dudons N., Gonzalez P., Bourdineaud J.P., Massabuau J.C. 2008. How cadmium could compromise the completion of the European eel's reproductive migration, Environmental Science and Technology, 42 (12), 4607–4612.

Poole, W.R., J.D. Reynolds and C. Moriarty. 1990. Observations on the silver eel migrations of the Burrishoole River system, Ireland, 1959 to 1988. Int. Rev. gesamt. Hydrobiol. 75: 807–816.

Pratt, C. T. and Threader, W. R. 2011. Preliminary Evaluation of a Large-Scale American Eel Conservation Stocking Experiment. North American Journal of Fisheries Management 31: 619-628.

Psuty, I. AND Bohdan, D. 2008. The effectivness of glass eel stocking in the Vistula Lagoon, Poland. Acta Ichthyologica et Piscatoria, 38 (2); 103–111.

Rathcke, P.-C. 1996. Untersuchungen über Art und Menge der Fische im Rechengut des Kraftwerkes Moorburg. Auftraggeber: Hamburgische Elektricitäts-Werke AG.

Rauck, G. 1980. Mengen und Arten vernichteter Fische und Krebstiere an den Rechen des Einlaufbauwerkes im Kernkraftwerk Brunsbüttel, sowie Testversuche zur Reaktion von Fischen auf die Elektroscheuchanlage auf der Basis von dort anfallenden Fischproben. Bundesforschungsanstalt für Fischerei, Veröffentlichungen des Instituts für Küsten- und Binnenfischerei Hamburg, 71/1980, 22 S.

Riemersma, P. and G.J.M. Wintermans. 2005. Optimalisatie vismigratie Den Deel, najaarsonderzoek. Onderzoek naar mogelijkheden inzet scheepvaartsluis ter bevordering van vismigratie bij gemaal Den Deel. Grondmij Noord bv en Wintermans Ecologenbureau.

RMC. 1995. Report on studies to evaluate American eel passage. Prepared by RMC Environmental Services for Allegheny Power Service Corporation. RMC Environmental Services: Drumore, Pennsylvania. 76p.

Roqueplo, C., Lambert, P., Gonthier, P. and Mayer, N. 2000. Estimation de la mortalité des civelles de la Gironde après leur passage dans le circuit de refroidissement de la Centrale nucléaire du Blayais Etude Cemagref, Groupement de Bordeaux, n° 58. 56 pp.

Roosens, L., Geeraerts, C., Belpaire, C., Van Pelt, I., Neels, H., Covaci, A. 2010. Spatial variations in the levels and isomeric patterns of PBDEs and HBCDs in the European eel in Flanders, Environ. Int. 36, 415–423.

Tesch, F.W. 2003. *The Eel* (JE Thorpe, Ed.). Blackwell Publishing, Oxford.

Rosell, R., Evans, D. and Allen, M. 2005. The eel fishery in Lough Neagh, Northern Ireland –an example of sustainable management? Fisheries Management and Ecology, 2005, 12, 377–385.

Ryan, A. 2010. A report on the Assessment of Obstructions to Eel Migration, using the Nore as a case study. Unpublished MSc project Report, NUI Maynooth; 64pp.

Sano, T., Nishimura, T., Okamoto, N.,and Fukuda, H. 1977. Studies on viral diseases of Japanese fishes.7: Rhabdovirus isolated from European eel, *Anguilla anguilla*. Bulletin of the Japanese Society of Scientific Fisheries, 43 (5); 491–495.

Santoul, F., Hougas, J.B., Green, A.J. and Mastrorillo, S. 2004. Diet of great cormorants (*Phalacrocorax carbo sinensis*) wintering in Malause (Southwest France). Ach, Hydrobiol. 160(2): 281–287.

Schafer, M. 1982. Dispersion and feeding ecology of wintering cormorants *Phalacrocorax carbo*. PhD thesis, University of Aberdeen.

Scheinert, P. and Baath C. 2004. Das Aalherpesvirus- Eine neue Bedrohung der Aal-bestande. Fischer und Teichwirt 6: 692–693.

Simon, J. 2011. Wenn Jäger zu Gejagten werden – Nahrungsuntersuchungen an Kormoranen in der Havel bei Potsdam. Fischer & Teichwirt 01/2011: 6–9.

Sonnesen P. 2007. Skarvens prædation omkring Ringkøbing Fjord – en undersøgelse af sammenhænge mellem fødevalg og fiskebestandenes sammensætning. pp. 76.

Spah, H. 2001. Fishery biological opinion of the fish compatibility of the patented hydraulic screw from Ritz Atro. Bielfield, Germany.

Spitsbergen JM, Walker MK, Olson JR, Peterson RE. Pathological alterations in early life stages of lake trout, Salvelinus namaycush, exposed to 2,3,7,8-tetrachlorodibenzo-pdioxinas fertilized eggs. Aquat Toxicol 1991 ;19:41–72.

Stachel, B., Kühn, T. Reincke, H., Schröter-Kermani, C. Uhlig, S. 2006. Statistical evaluation with dioxin patterns in different fish species from River Elbe, Germany. Fresenius Envoron Bull 15: 1624–31.

Steevens, J. A., M. R. Reiss, *et al*. 2005. "A methodology for deriving tissue residue benchmarks for aquatic biota: A case study for fish exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin and equivalents." Integrated Environmental Assessment and Management **1**(2): 142–151.

STOWA. 2010. Worden vissen in de maling genomen? STOWA rapportnummer 2010-21, pp. 48.

Suter, W. 1997. Roach rules: Shoaling fish are a constant factor in the diet of cormorants *Phalacrocorax carbo in Switzerland*. ARDEA 85(1): 9–27.

Svärdson, G. 1976. The decline of the Baltic eel population. Inst. Freshw. Res. 55: 136–143.

Svendsen, T.C., Vorkamp, K., Frederiksen, M. , Rønsholdt, B. and Frier, J.O. 2007. Body burdens of persistent halogenated compounds during different development stages of anadromous brown trout (*Salmo trutta*). Environ Sci Technol. 2007 Sep 1;41(17):5980–5.

Symonds, J.E. 2007. The American eel (*Anguilla rostrata*): stock enhancement review. The Huntsman Marine Science Centre. 135 p.

Szlinder-Richert J., Usydus Z- & Pelczarski W. 2010. Organochlorine pollutants in European eel (*Anguilla anguilla* L.) from Poland. Chemosphere 80: 93–99.

Tabouret, H., *et al*. 2011. Heavy metals and organochlorinated compounds in the European eel (*Anguilla anguilla*) from the Adour estuary and associated wetlands (France). J. Environ. Monit., 13, 1446.

Tapie N., *et al*. PBDE and PCB contamination of eels from the Gironde estuary: From glass eels to silver eels. Chemosphere (2011).

Tesch FW. 2003. The eel. Blackwell Publishing; 408pp.

Travade, F., Larinier, M., Subra, S., Gomes, P. and De-Oliveira, E. 2010. Behaviour and passage of European silver eels (*Anguilla anguilla*) at a small hydropower plant during their downstream migration. Knowledge and Management of Aquatic Ecosystems 398, 01.

Tremblay, V., Cossette, C., Dutil, J.-D., Verreault G., and Dumont P. 2011. Assessment of upstream and downstream passability for eel at dams. Can. Tech. Rep. Fish. Aquat. Sci. 2912: x + 73p.

Tsukamoto, K., Chow, S., Otake, T., Kurogi, H., Mochioka, N., Miller, M. J., Aoyama, J., Kimura, S., Watanabe, S., Yoshinaga, T., Shinoda, A., Kuroki, M., Oya, M., Watanabe, T., hata, K., Ijiri, S., Kazeto, Y., Nomura, K. and Tanaka, H. 2011. Oceanic spawning ecology of freshwater eels in the western North Pacific. Nature Communications 2:179. DOI: 10.1038/ncomms1174.

Turnpenny, A., Bromley, R. Coyle, S. & Hawley, K. 2008. AMP 4 Lower Thames water intakes investigation. Jacobs UK Ltd. 175 pp.

Tzeng, W.-N. 2009. Migratory behaviour and habitat use of Japanese eel in the estuary as revealed by both conventional mark-recapture method and otolith elemental signature. Summary presented at ICES Saline SG.

Tzeng, W.-N. 2009. Naturally-recruited and restocked eels in Lativia: discrimination and growth comparison. Summary presented at ICES Saline SG.

Ubl, C. 2004. Untersuchungen zum Nahrungspektrum des Kormorans im Bereich des Greifswalder Boddens. Fischerei & Fischmarkt 4(2): 32–38.

van Banning, P. and Haenen, O. 1990. Effects of the swimbladder nematode *Anguillicola crassus* in wild and farmed eel, *Anguilla anguilla*. In: Pathology in marine science. F. O. Perkins and T. C. Cheng (Eds). Academic Press, San Diego, CA (USA), pp. 317–330.

van Ginneken VJT, van den Thillart GEEJM. 2000. Eel fat stores are enough to reach the Sargasso. Natura 2000; 403:156–7.

van Ginneken V., Haenen O., Coldenhoff K., Willemze R., Antonissen E., van Tulden P., Dijkstra S., Wagenaar F. and van den Thillart G. 2004. Presence of virus infections in eel populations from various geographic areas. Bulletin of the European Association of Fish Pathologists 24, 270–274.

van Ginneken V., Ballieux T.B., Willemzer R., Coldenhoff K., Lentjes E., Antonissen E., Haenen O. and van den Thillart G. 2005. Haematology patterns of migrating European eels and the role of EVEX virus. Comparative Biochemistry and Physiology Part C 140, 97–102.

van Ginneken, V., Palstra, A., Leonards, P., Nieveen, M., van den Berg, H., Flik, G., Spanings, T., Niemantsverdriet, P., van den Thillart, G., and Murk, A. 2009. PCBs and the energy cost of migration in the European eel (*Anguilla anguilla* L.). Aquat. Toxicol. 92, 213–220.

van Nieuwstadt A.P., Dijkstra S.G. and Haenen O.L.M. 2001. Persistence of herpesvirus of eel Herpesvirus anguillae in farmed European eel *Anguilla anguilla*. Diseases of Aquatic Organisms 45, 103–107.

van Rijn S. and M.R. van Eerden. 2001. Aalscholvers in het IJsselmeergebied: concurrent of graadmeter? [Cormorants in the IJsselmeer area: competitor or indicator?] RIZA rapport 2001.058.

van Rijn S. 2005. Monitoring Aalscholvers in het IJsselmeergebiedVoortgangsverslag 2005. RIZA werkdocument 2005.

Veldkamp, R. 1995. Diet of cormorants *Phalacrocorax carbo sinensis* at Wanneperveen, the Netherlands, with special reference to Bream *Abramis brama*. ARDEA 83(1): 43–55.

Verreault, G. and Dumont, P. 2002. An estimation of American eel escapement from the upper St Lawrence River and Lake Ontario in 1996 and 1997. American Fisheries Society Symposium 33: 243–251.

Verreault, G., P. Dumont, J. Dussureault and R. Tardif. 2010. First record of migrating silver American eels (*Anguilla rostrata*) in the St Lawrence Estuary originating from a stocking program. Journal of Great Lakes Research 36 (2010) 794–797.

Vøllestad, L. 1992. Geographic variation in age and length at metamorphosis of maturing European eel: environmental effects and phenotypic plasticity. Journal of Animal Ecology, 61, 41–48.

von Raben, K. 1964. Regarding the problem of mutilations of fishes by hydraulic turbines. Fisheries Research Board of Canada Translation Series No. 448: 1–12.

Vriese, F.T., Hop, J., Kampen, J. 2010. Gemalen en schade aan vis. Visadvies BV, Nieuwegein. Projectnummer VA2009_33, pp. 145.

Wahl, K., Malisch, R., Kotz, A., Bitomsky, N., Kypke, K., Adamovic, K., Gerteisen, I., Leswal, S., Schächtele, J., Tritschler, R., Winterhalter, H. 2011. PCDD/Fs, PCBs and further contaminants in fresh water fish from waterbodies in Baden-Württemberg, Germany. Extended abstract presented Dioxin 2011, 21–25 August, Brussels (Belgium).

Wakabayashi and Egusa. 1972. Preliminary experiments on environmental factors influencing the prevalence of columnaris disease. Fish Pathology 7, 58–63.

Walker M.K., R.E. Peterson. 1994. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to brook trout (*Salvelinus fontinalis*) during early development. Environmental Toxicology and Chemistry Vol 13, no. 5 pp. 817–820.

Walker MK, Cook PM, Batterman AR, Butterworth BC, Berini C, Libal JJ. 1994. Translocation of 2,3,7,8-Tetrachlorodibenzo-P-dioxin from adult female lake trout (*Salvelinus namaycush*) to oocytes — effects on early-life stage development and sac fry survival. Can J Fish Aquat Sci;51:1410–9.

Wariaghli F. 2006. L'Anguille *Anguilla anguilla* L. 1758 : Contamination métallique au bas Sebou et à Moulay Bousselham et biologie et croissance à l'oued Martil. Mém. DESA. Univ. Mohamed V-Agdal, Fac. Sciences, Rabat. 82p.

Warke, G. M. A. and Day, K. R. 1995. The feeding ecology of cormorants in Northern Ireland. Ardea.

Watene, E.M., Boubée, J.A.T. and Haro, A. 2003. Downstream movement of mature eels in a hydroelectric reservoir in New Zealand. American Fisheries Society Symposium 33: 295–305.

West, B., Cabot, D. and Greer-Walker, M. 1975. The food of the cormorant at some breeding colonies in Ireland. Proc. R. Ir. Acad., 75, 285–304.

Westin, L. 1998. The spawning migration of European silver eel (*Anguilla anguilla* L.) with particular reference to stocked eel in the Baltic. Fisheries Research (Amsterdam) 38: 257–270.

Westin, L. 2003. Migration failure in stocked eels *Anguilla anguilla*. Marine Ecology Progress Series, 254: 307–311.

White, E. M. and Knights, B. 1997. Environmental factors affecting migration of the European eel in the Rivers Severn and Avon, England", Journal of Fish Biology 50: 1104–1116.

WHO. 2000. Consultation on assessment of the health risk of dioxins; re-evaluation of the tolerable daily intake (TDI): executive summary. Food Additives and Contaminants Part A 17(4): 223–240.

Wickström, H., Westin L., Clevestam P. 1996. The Biological and economical yield from a long-term stocking experiment. Ecology of Freshwater Fish 5: 140–147.

Wickström, H., Dekker, W. and Andersson, J. 2011. Report on the eel stock and fishery in Sweden 2010/11 (Country Report for the WG Eel 2011).

Wickstrom, H., Clevestam, P., Sjoberg, N.B., and Dekker, W. 2010. Kunskapsgenomgång utsatt ål och dess lekvandring (Migration patterns and orientation in restocked eels – a scientific review). Avdelningen för forskning och utveckling, Sötvattenslaboratoriet, Stockholm, Sweden. Internal report. Translated from Swedish by W Dekker as working paper for WGEEL 2011.

Wilkońska, H. and I. Psuty. 2008. Using a side-selective fyke net barrier to research fish assemblages in the transitional and transboundary waters of the Vistula Lagoon, southern Baltic. J. Appl. Ichthyol. 24: 650–657.

Williams, B. and Aprahamian, M.W. 2004. Management Guidelines for the Stocking of Eel and Elver (*Anguilla anguilla* L.). Environment Agency, Bristol.

Williams, B. and Threader, R. 2007. A review of the proceedings and outcomes of the workshop on the American eel (*Anguilla rostrata*) stocking in Canadian waters. Montréal, Canada, March 26 and 27, 2007. 37p.

Winter, H.V., Jansen, H.M. and Bruijs, M.C.M. 2006. Assessing the impact of hydropower and fisheries on downstream migrating silver eel, *Anguilla anguilla*. Ecology of Freshwater Fish 15: 221–228.

Winter, H.V., Jansen, H.M. and Breukelaar, A.W. 2007. Silver eel mortality during downstream migration in the River Meuse, from a population perspective. ICES Journal of Marine Science 64: 1444–1449.

Wirth T, Bernatchez L. 2001. Genetic evidence against panmixia in the European eel. Nature 409:1037–1040.

Witteveen and Bos. 2010a. RW1774-2 Sterfte van schieraal door gemaal IJmuiden, onderzoeksjaar 2009.

Witteveen and Bos. 2010b. MDBK34-1 Monitoring van vismigratie bij gemaal J.L. Hoogland en de Johan Friso-sluis.

Wlasow, T., Gomulka, P., Martyniak, A., Boron, S., Hliwa, P., Terelecki, J. and Szymanska, U. 1998. *Anguillicola crassus* larvae in cormorant's prey fish in Vistula Lagoon, Poland. Bulletin Francais de la Peˆche et de la Pisciculture 349, 223 7.

Wright, N. M. 1986. The Diet of Cormorant fledglings on Sheep Island, Co. Antrim. BSc thesis, University of Ulster.

Zouhir I. 2006. Contribution à l'étude de l'éco-biologie de l'anguille européenne (*Anguilla anguilla*, L., 1758) au niveau de l'estuaire du Sebou. Mém. DESA., Univ. Mohammed V-Agdal, Fac. Sci., Rabat. 83p.

Žydelis, R. and Kontautas, A. 2008. Piscivouros birds as top predators and fishery competitors in the lagoon systems. Hydrobiologica 611: 45–54.

Annex 1: List of participants

Name	Address	Phone/Fax	E-mail
Jan Andersson	Swedish University of Agricultural Sciences Dept. of Aquatic Resources Institute of Coastal Research Simpevarp, 100 SE-572 95 Figeholm Sweden	Phone: +46 104 784112	jan.andersson@slu.se
Miran Aprahamian	Environment Agency Richard Fairclough House Knutsford Road WA4 1HT Warrington UK	Phone: +44 192 5542713 Fax: +44 192 5415961 Mobile: +44 768 905933	miran.aprahamian@environment-agency.gov.uk
Aizkorri Aranburu	AZTI - Tecnalia / Itsas Ikerketa Saila Txatxarramendi z/g 48395-Sukarrieta Bizkaia Spain	Phone: +34 94 657 4000 (ext. 459) Phone: +34 94 657 2555	aaranburu@azti.es
Laurent Beaulaton	Conseil Supérieur de la Pêche Immeuble Le Péricentre 16, avenue Louison Bobet FR-94132 Fontenay-sous-Bois Cedex France	Phone: +33 145 143634 Mobile: +33 681 4752 71	Laurent.beaulaton@onema.fr
Claude Belpaire	Research Institute for Nature and Forest (INBO) Duboislaan 14 1560 Groenendaal-Hoeilaart Belgium	Phone: +32 265 80411 Mobile: + 32 475 678992	Claude.Belpaire@inbo.be
Daniele Bevacqua	Università degli Studi di Parma Dipartimento di Scienze Ambientali Via Università 12 143100 Parma Italy	Phone: +39 333 3024144 Fax: +39 521 905402	daniele.bevacqua@gmail.com

Name	Address	Phone/Fax	E-mail
Janis Birzaks	Institute for Food Safety, Animal Health and Environment (BIOR) 8 Daugavgrivas Str. LV-1048 Riga Latvia	Phone: +37 176 12 536 Fax: +37 176 16 946	janis.birzaks@bior.gov.lv
Uwe Brämick	Institute of Inland Fisheries, Potsdam Im Königswald 2 14469 Potsdam Germany	Phone: +49 332 01 4060	uwe.braemick@ifb-potsdam.de
Cédric Briand Co-Chair	Institution d'Amenagement de la Vilaine 56 130 La Roche Bernard France	Phone: +33 999 08844	cedric.briand@lavilaine.com
Fabrizio Capoccioni	Univeristà di Roma Tor Vergata Dipertimento d'Biologie Vie delle Recirce Scientific 1 00133 Rome Italy	Phone: +39 347 0852870 Fax: +39 067 2595852	fabrizio.capoccioni@uniromae.it
Marie-Noëlle de Casamajor	Ifremer Laboratorie Halieutique d'Aquitaine UFR Sciences et Techniques 1 Allee du parc Montaray 64600 Anglet France	Phone: +33 229 008581 Fax: +33 229 008552	Marie.Noelle.De.Casamajor@ifremer.fr
Gérard Castelnaud	CEMAGREF 50 Avenue de Verdun 33612 Cestas France	Phone: +33 557 890803 Fax: +33 557 890801	Gerard.Castelnaud@cemagref.fr
Eleonora Ciccotti	Univeristà di Roma Tor Vergata Dipertimento d'Biologie Vie delle Recirce Scientific 1 00133 Rome Italy	Phone: +39 067 2595969 Fax: +39 067 2595965	ciccotti@uniroma2.it

Name	Address	Phone/Fax	E-mail
Fernando Cobo Gradin	University of Santiago de Compostela Praza do Obradoiro, s/n 15782 Santiago de Compostela Spain	Phone: +34 986 565640 Fax: +34 986 503543	Fernando.cobo@usc.es
Willem Dekker	Swedish University of Agricultural Sciences Department of Aquatic Resources Institute of Freshwater Research, Drottningholm Stångholmsvägen 2 SE-178 93 Drottningholm Sweden	Phone: +46 10 478 4248 Fax: +46 10 478 4269	Willem.Dekker@slu.se
Isabel Domingos	Instituto de Oceanografia, FCUL (Faculty of Sciences, University of Lisbon) Campo Grande 1749-016 Lisbon Portugal	Phone: +35 121 7500970 Fax: +35 121 7500009	idomingos@fc.ul.pt
Malte Dorow	Institute for Fisheries, Sate Research Center for Agriculture and Fishery Fischerweg 408 D-18069 Rostock Germany	Phone: +49 381 8113403 Fax: +49 381 8113407	m.dorow@lfa.mvnet.de
Caroline Durif	Institute of Marine Research-Austevoll 5392 Storebø Norway	Phone: +47 458 47514	caroline.durif@imr.no
Derek Evans	Agri-food and Biosciences Institute 18a Newforge Lane BT9 5PX Belfast UK	Phone: +44 289 0255551	derek.evans@afbini.gov.uk
Elvira de Eyto	Marine Institute Furnace Newport Co. Mayo Ireland	Phone: +353 984 2300 Fax: +353 984 2340	elvira.deeyto@marine.ie

Name	Address	Phone/Fax	E-mail
Jason Godfrey	Marine Scotland Freshwater Laboratory Faskally Pitlochry Perthshire PH15 5LB Scotland UK	Phone: +44 122 4294444	j.d.godfrey@marlab.ac.uk
Martin de Graaf	Wageningen IMARES PO Box 68 NL-1970 AB IJmuiden Netherlands	Phone: +31 317 486826	martin.degraaf@wur.nl
Reinhold Hanel	vTI-Institute of Fisheries & Ecology Forestry and Fisheries Palmaille 9 22767 Hamburg Germany	Phone: +49 403 8905290	reinhold.hanel@vti.bund.de
Ain Jarvalt	Centre for Limnology Estonian University of Life Sciences Kreutzwaldi 4 EE-51014 Tartu Estonia	Phone: +372 745 4544 Fax: +372 745 4546	ain.jarvalt@emu.ee
Michiel Kotterman	Wageningen IMARES PO Box 68 NL-1970 AB IJmuiden Netherlands	Phone: + 31 317 487132	michiel.kotterman@wur.nl
Tagried Kurwie	Mahurangi Technical Institute Warkworth New Zealand PO Box 414	Phone: +64 942 58934 Fax: +64 942 58926	t.kurwie@btinternet.com
Patrick Lambert	CEMAGREF 50 avenue de Verdun F-33612 Cestas cedex France	Phone: +33 557 890809 Fax: +33 557 890801	patrick.lambert@cemagref.fr
Kieran McCarthy	National University of Ireland, Galway University Road Galway Ireland	Phone: +353 914 92333 Fax: +353 914 95426	tk.mccarthy@nuigalway.ie

Name	Address	Phone/Fax	E-mail
Ciara O'Leary	Inland Fisheries Ireland Swords Business Campus Swords Co. Dublin Ireland	Phone: +353 188 42600 Fax: +353 183 60060	ciara.oleary@fisheriesireland.ie
Tomasz Nermer	National Marine Fisheries Research Institute ul. Kollataja 1 PL-81-332 Gdynia Poland	Phone: +48 587 356211	nermer@mir.gdynia.pl
Michael Ingemann Pedersen	DTU Aqua - National Institute of Aquatic Resources Department of Inland Fisheries Vejlsøvej 39 DK-8600 Silkeborg Denmark	Phone: +45 892 13128	mip@aqua.dtu.dk
Russell Poole Chair	Marine Institute Marine Institute Catchment Research Facility Furnace Newport Co. Mayo Ireland	Phone: + 353 984 2300 Fax: +353 984 2340	russell.poole@marine.ie
Stanislaw Robak	Inland Fisheries Institute ul. Oczapowskiego 10 10-719 Olsztyn 5 Poland	Phone: +48 895 240171 Fax: +48 895 240505	robak@infish.com.pl
Robert Rosell	Agri-food and Biosciences Institute 18a Newforge Lane BT9 5PX Belfast UK	Phone: 028 902 55506 Fax: 028 902 55004	robert.rosell@afbini.gov.uk

Name	Address	Phone/Fax	E-mail
Karin Schindehütte	Ministry for Climate Protection, Environment, Agriculture, Nature Conservation and Consumer Protection of the State of North Rhine-Westphalia Schwannstr. 3 40476 Düsseldorf Germany	Phone: +49 211 4566780 Fax: +49 211 4566947 In Lissabon: +49 157 73355437	karin.schindehuette@mkulnv.de
Maria Jose Servia Garcia	Universidad de la Coruña Rúa Maestranza 9 15001 A Coruña Spain	Phone: +34 986 565640 Fax: +34 986 503543	mservia@udc.es
Marcello Schiavina	DSA - Dipartimento di Scienze Ambientali Università degli Studi di Parma & DEI Politecnico Di Milano via Ponzio 34/5 Milano I-20133 Italy	phone: +39 022 3999676	m.schiavina@gmail.com
Eva B. Thorstad	Norwegian Institute for Nature Research (NINA) N-7485 Trondheim Norway	Phone: +47 738 01400 Fax: +47 738 01401 Mobile; +47 916 61130	eva.thorstad@nina.no
Ron Threader	Ontario Power Generation Ottawa/St. Lawrence Plant Group 2 Innovation Dr. Renfrew Ontario K7V 4H4 Canaada	Phone: +613 433 9673 Fax: +613 433 9679	ron.threader@opg.com
Guy Verrault	Ministere des Ressources naturelles et de la Faune du Quebec 186 Fraser G5R 1C8 Rivière-du-Loup Quebec Canada	Phone: +1 418 8628213 ext. 306 Fax: +1 418 8628176	guy.verreault@mrnf.gouv.qc.ca

Name	Address	Phone/Fax	E-mail
Alan M. Walker	Centre for Environment, Fisheries and Aquaculture Science Lowestoft Laboratory Pakefield Road NR33 0HT Lowestoft Suffolk UK	Phone: +44 150 2524351 Fax: +44 150 2526351	alan.walker@cefas.co.uk
Håkan Westerberg	Paskvagen 39 42653 Vistra Frolunda Sweden	Phone: +46 705 269956	hakan.westerberg@slu.se
Håkan Wickström	Swedish University of Agricultural Sciences Dept. of Aquatic Resources Institute of Freshwater Research Stångholmsvägen 2 SE-178 93 Drottningholm Sweden	Phone: +46 104 784246	hakan.wickstrom@slu.se
Klaus Wysujack	Johann Heinrich von Thünen-Institute Institute for Fishery Ecology Palmaille 9 D-22767 Hamburg Germany	Phone: +49 410 27086013	klaus.wysujack@vti.bund.de
Ahmed Yahyaoui	Laboratory of Zoology Faculty of Science Mohammed V-Agdal University B.P. 1014 Rabat Morocco	Phone: +212 5377 75461	a.yahyaoui@fsr.ac.ma

Annex 2: Agenda

Agenda for Joint EIFAAC/ICES WGEEL 2011-Lisbon

Sunday 4th September Afternoon

Meeting of task leaders in the afternoon; 17:00–19.30

Monday 5th September

9.00	Get organized
9.30–10.00	Welcome RP
	Local Welcome/info: I. Domingos & Professor M. José Costa
10.00–10.15	Intro to Working Group, ToR, EIFAAC, etc. RP
10.15–11.15	Coffee
11.15–11.30	Task 1 - introduced by Briand/Beaulaton
11.30–11.45	Task 2 - introduced by Aprahamian
11.45–12.00	Task 3 - introduced by Rosell
12.00–12.15	Task 4 - introduced by Lambert
12.15–12.30	Task 5 - introduced by Belpaire
12.30–12.45	Task 6 - introduced by de Graaf
12.45–13.00	Task 7 & SGIPEE report; introduced by Dekker
13.00–13.30	BaltEel Workshop and Questions so far
13.30–14.30	Lunch
14.30–14.45	Task 8; introduced by Castelnaud
14.45–15.00	Task 9; introduced by Domingos
	Coffee
15.00–17.30	Break-out into Subgroups
17.30–18.30	Plenary; plan of attack, gaps, etc.

Tuesday Subgroups breakout

	Coffee available @ 11.00 and 16.00
18.00	Subgroup/task leaders coordination meeting

Wednesday Subgroups breakout

	Coffee available @ 11.00 and 16.00
14.00–15.30	Plenary

Draft Advice Session in pm (17.00); subgroup leaders in Neptune's Room

Thursday

09.00	Draft conclusions and recommendations draft 1.
11.00	Coffee
15:30–18.00	Producing draft report [**DEADLINE 18:00**]
16.00	Coffee

pm Circulate draft advice and report for comment

Friday

9.00–11.00	Discuss and agree Recommendations & Conclusions, and agree advice
	Coffee available @ 11.00 and 16.00
11.00–17:00	Review Report.

Annex 3: WGEEL Terms of Reference for the next meeting

2011/2/ACOM18 The **Joint EIFAAC/ICES Working Group on Eels** [WGEEL] (Chaired by Russell Poole, Ireland and Cedric Briand*, France, will meet in to be announced, to be confirmed 2012, to:

1) assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans; examine criteria for defining a recovery;
2) develop and test methods to post-evaluate effects of management actions at the stock-wide level (in conjunction with SGIPEE), including quality assurance checking of Eel Management Unit biomass estimates;
3) develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures; test data scenarios at the local level;
4) provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods;
5) review and develop approaches to quantifying the effects of eel quality on stock dynamics and integrating these into stock assessments; develop references points for evaluating impacts on eel;
6) respond to specific requests in support of the eel stock recovery Regulation, as necessary; and
7) report on improvements to the scientific basis for advice on the management of European and American eel.

Material and data for the meeting must be available to the Group no later than 14 days prior to the starting date.

WGEEL will report by date XXX for the attention of ACOM, WGRECORDS, SGEF and EIFAAC.

Annex 4: Tables from Chapter 2

Due to the size of the tables in Annex 4, it is not possible to reproduce them completely here: Data resides with WG/ICES and can be requested from ICES or a Group Member.

Table 2.1. Series information for recruitment time-series.

id	Series name	omment on the locatio	x	y	tylcode	area	country	river	location	samplingtype	unit	life stage	shortname	long name
1	IYFS scientific estimate		5	57	Recruit	North s	Sweden		IYFS/IBT	scientific estima	Index	glass eel	YFS1	glass eel scientific estimate
41	IYFS2 scientific estim	Skagerrak-Kattegat			Recruit	North s	Sweden		IYFS/IBT	scientific estima	Index	glass eel	YFS2	glass eel scientific estimate
2	Ringhals scientific survey		12.07	57.15	Recruit	North s	Sweden	Katteg	Ringhals	scientific estima	Index	glass eel	Ring	glass eel scientific estimate
3	Viskan Sluices trapping all		12.07	57.12	Recruit	North s	Sweden	Viskan	Sluices	trapping all	Kg	glass eel + y	Visk	glass eel + yellow eel trapping all
4	Bann Coleraine trappi	River Bann flowing	-6.42	55.12	Recruit	British	Northe	Bann	Colerain	trapping partial	Kg	glass eel + y	Bann	glass eel + yellow eel trapping partial
5	Erne Ballyshannon trapping all		-8.15	54.3	Recruit	British	Ireland	Erne	Ballysha	trapping all	Kg	glass eel + y	Erne	glass eel + yellow eel trapping all
6	Shannon Ardcrusha trapping all		-8.36	52.42	Recruit	British	Ireland	Shann	Ardcrush	trapping all	Kg	glass eel + y	ShaA	glass eel + yellow eel trapping all
45	River Feale	Not influenced by v	-9.63	52.47	Recruit	Atlanti	Ireland	Feale		trapping all	Kg	glass eel	Feal	glass eel trapping all
46	River Maigue	not influenced by w	-8.78	52.65	Recruit	Atlanti	Ireland	Maigue		trapping all	Kg	glass eel	Maig	glass eel trapping all
47	River Igh	not influenced by w	-9.43	52.83	Recruit	Atlanti	Ireland	Igh		trapping all	Kg	glass eel	Ig	glass eel trapping all
7	Severn EA commercial catch		-2.56	51.67	Recruit	British	UK	Severn	EA	commercial cat	t	glass eel	SeEA	glass eel commercial catch
8	Severn HMRC comm	Severn River.	-2.56	51.67	Recruit	British	UK	Severn	HMRC	commercial cat	Kg	glass eel	SeHM	glass eel commercial catch
9	Vidaa Højer sluice commercial catch		8.4	55.58	Recruit	North s	Denma	Vidaa	Højer slu	commercial cat	Kg	glass eel	Vida	glass eel commercial catch
10	Ems Herbrum comme	Herbrum is a weir	7.2	53.02	Recruit	North s	Germa	Ems	Herbrum	commercial cat	Kg	glass eel	Ems	glass eel commercial catch
11	Lauwersoog scientific estimate		6.19	53.4	Recruit	North s	Netherlands		Lauwers	scientific estima	nb/h	glass eel	Lauw	glass eel scientific estimate
12	Rhine DenOever scientific estimate		5.25	52.4	Recruit	North s	Netherl	Rhine	DenOeve	scientific estima	Index	glass eel	RhDO	glass eel scientific estimate
13	Rhine Ijmuiden scientific estimate		4.6	52.46	Recruit	North s	Netherl	Rhine	Ijmuiden	scientific estima	Index	glass eel	RhIj	glass eel scientific estimate
14	Katwijk scientific estimate		4.4	52.21	Recruit	North s	Netherlands		Katwijk	scientific estima	Index	glass eel	Katw	glass eel scientific estimate
15	Stellendam scientific e	to be updated	4.03	51.83	Recruit	North s	Netherlands		Stellenda	scientific estima	Index	glass eel	Stel	glass eel scientific estimate
16	Ijzer Nieuwpoort scien	The Nieuwpoort st	2.45	51.08	Recruit	North s	Belgiur	Ijzer	Nieuwpo	scientific estima	Kg	glass eel	Yser	glass eel scientific estimate
17	Vilaine Arzal trapping all		-2.39	47.5	Recruit	Atlanti	France	Vilaine	Arzal	trapping all	t	glass eel	Vil	glass eel trapping all
18	Loire Estuary commercial catch		-1.96	47.28	Recruit	Atlanti	France	Loire	Estuary	commercial cat	Kg	glass eel	Loi	glass eel commercial catch
19	Sèvres Niortaise Estuary commercial CPI		-1.07	46.31	Recruit	Atlanti	France	Sèvres	Estuary	commercial CP	cpue	glass eel	SevN	glass eel commercial CPUE
20	Gironde Estuary (catch) commercial catcl		-0.8	45.4	Recruit	Atlanti	France	Girond	Estuary (	commercial cat	t	glass eel	GiTC	glass eel commercial catch
21	Gironde Estuary (CPUE) commercial CPL		-0.8	45.4	Recruit	Atlanti	France	Girond	Estuary (	commercial CP	cpue	glass eel	GiCP	glass eel commercial CPUE
42	Gironde scientific esti	The Gironde surve	-0.8	45.4	Recruit	Atlanti	France	Girond	Scientific	scientific estima	Index	glass eel	GiSc	glass eel scientific estimate
22	Adour Estuary (catch) commercial catch		-1.44	43.48	Recruit	Atlanti	France	Adour	Estuary (	commercial cat	t	glass eel	AdTC	glass eel commercial catch
23	Adour Estuary (CPUE) commercial CPUE		-1.44	43.48	Recruit	Atlanti	France	Adour	Estuary (	commercial CP	cpue	glass eel	AdCP	glass eel commercial CPUE
24	Ion Estuary commerci	Until the 70's only l	-6.08	43.55	Recruit	Atlanti	Spain	Ion	Estuary	commercial cat	Kg	glass eel	Io	glass eel commercial catch
25	Albufera de Valencia c	beware data year n-	-0.3	39.31	Recruit	Medite	Spain	Albufe	Albufera	commercial cat	Kg	glass eel	Albu	glass eel commercial catch
26	Minho spanish part cc	beware data year n-	-8.81	41.9	Recruit	Atlanti	Spain	Minho	spanish p	commercial cat	Kg	glass eel	MiSp	glass eel commercial catch
27	Minho portugese part	beware data year n-	-8.81	41.9	Recruit	Atlanti	Portug	Minho	portuges	commercial cat	Kg	glass eel	MiPo	glass eel commercial catch
43	Ebro delta lagoons	beware data year n-	0.76	40.72	Recruit	Medite	Spain	Ebro d	river delt	commercial cat	Kg	glass eel	Ebro	glass eel commercial catch
44	Albufera de Valencia c	beware data year n-	-0.3	39.31	Recruit	Medite	Spain	Albufera lagoon		commercial CP	cpue	glass eel	AlCP	glass eel commercial CPUE
28	Tiber Fiumara Grande commercial catch		12.14	41.44	Recruit	Medite	Italy	Tiber	Fiumara	commercial cat	t	glass eel	Tibe	glass eel commercial catch
29	Imsa Near Sandnes trapping all		5.59	58.54	Recruit	North s	Norway	Imsa	Near Sar	trapping all	Number	glass eel	Imsa	glass eel trapping all
30	Dalälven trapping all	370 mm average si	16.6	60.18	Recruit	Baltic	Sweden	Dalälve	?	trapping all	Kg	yellow eel	Dala	yellow eel trapping all
31	Motala Ström trappin	Northest series av	16.17	58.58	Recruit	Baltic	Sweden	Motala	Ström	trapping all	Kg	yellow eel	Mota	yellow eel trapping all
32	Mörrumsån trapping a	Baltic	14.66	56.4	Recruit	Baltic	Sweden	Mörrumsån		trapping all	Kg	yellow eel	Morr	yellow eel trapping all
33	Kävlingeån trapping al	Baltic entrance	13.1	55.78	Recruit	Baltic	Sweden	Kävling	?	trapping all	Kg	yellow eel	Kavl	yellow eel trapping all
34	Rönne Å trapping all	Baltic entrance	14.7	55.1	Recruit	North s	Sweden	Rönne Å		trapping all	Kg	yellow eel	Ronn	yellow eel trapping all
35	Lagan trapping all	Baltic			Recruit	North s	Sweden	Lagan		trapping all	Kg	yellow eel	Laga	yellow eel trapping all
36	Göta Älv trapping all	The Göta Älv trap	12.12	58.02	Recruit	North s	Sweden	Göta Älv		trapping all	Kg	yellow eel	Gota	yellow eel trapping all
37	Shannon Parteen trapping partial		-8.62	52.6	Recruit	British	Ireland	Shann	Parteen	trapping partial	Kg	yellow eel	ShaP	yellow eel trapping partial
38	Guden Å Tange trappi	hydropower	9.6	56.35	Recruit	North s	Denma	Guden	Tange	trapping all	Kg	yellow eel	Gude	yellow eel trapping all
39	Harte trapping all	The Harte hydropo	9.41	55.51	Recruit	Baltic	Denma	Harte	?	trapping all	Kg	yellow eel	Hart	yellow eel trapping all
40	Meuse Lixhe dam trap	On the Meuse, the	5.67	50.75	Recruit	North s	Belgiur	Meuse	Lixhe dar	trapping partial	Kg	yellow eel	Meus	yellow eel trapping partial
50	Bresle	Channel			Recruit	Atlanti	France	Bresle	Bresle riv	trapping all	Number	yellow eel	Bres	yellow eel trapping all

Table 2.2. Recruitment raw data. These data remain the property of the Country of origin and should not be used without the permission of the data provider.

Anguilla anguilla

A. anguilla	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*
North sea	North sea	North sea	North sea	British Isle	British Isle	British Isle	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	British Isle	British Isle
Sweden	Sweden	Sweden	Sweden	Northern Ireland	Ireland	Ireland	Ireland	Ireland	Ireland	UK	UK
Index	Index	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg
IYFS scientific estimate	IYFS2 scientific estimate	Ringhals scientific survey	Viskan Sluices trapping all	Bann Coleraine trapping partial	Erne Ballyshannon trapping all	Shannon Ardcrusha trapping all	River Feale	River Maigue	River Inagh	Severn EA commercial catch reports	Severn HMRC nett trade export
Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel

A. anguilla	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*
North sea	North sea	North sea	North sea	North sea	North sea	North sea	North sea	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean
Denmark	Germany	Netherlands	Netherlands	Netherlands	Netherlands	Netherlands	Belgium	France	France	France	France
Kg	Kg	Index	Index	Index	Index	Index	Kg	Kg	Kg	cpue	t
Vidaa Højer sluice commercial catch	Ems Herbrum commercial catch	Lauwersoog scientific estimate	Rhine DenOever scientific estimate	Rhine Ijmuiden scientific estimate	Katwijk scientific estimate	Stellendam scientific estimate	Ijzer Nieuwpoort scientific estimate	Vilaine Arzal trapping all	Loire Estuary commercial catch	Sèvres Niortaise Estuary commercial CPUE	Gironde Estuary (catch) commercial catch
Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel

A. anguilla	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*
Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Atlantic Ocean	Mediterranean Sea	Mediterranean Sea	Mediterranean Sea	Mediterranean Sea	North sea
France	France	France	France	Spain	Spain	Portugal	Spain	Spain	Spain	Italy	Norway
cpue	Index	t	cpue	Kg	Kg	Kg	Kg	Kg	cpue	t	Number
Gironde Estuary (CPUE) commercial CPUE	Gironde scientific estimate	Adour Estuary (catch) commercial catch	Adour Estuary (CPUE) commercial CPUE	Nalon Estuary commercial catch	Minho spanish part commercial catch	Minho portugese part commercial catch	Ebro delta lagoons	Albufera de Valencia commercial catch	Albufera de Valencia commercial CPUE	Tiber Fiumara Grande commercial catch	Imsa Near Sandnes trapping all
Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Glass eel	Yellow eel

A. anguilla	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*	*A. anguilla*
Baltic	Baltic	Baltic	Baltic	North sea	North sea	North sea	British Isle	North sea	North sea	North sea	North sea
Sweden	Sweden	Sweden	Sweden	Sweden	Sweden	Sweden	Ireland	Denmark	Denmark	Belgium	France
Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Kg	Number
Dalälven trapping all	Motala Ström trapping all	Mörrumsån trapping all	Kävlingeån trapping all	Rönne Å trapping all	Lagan trapping all	Göta Älv trapping all	Shannon Parteen trapping partial	Guden Å Tange trapping all	Harte trapping all	Meuse Lixhe dam trapping partial	Bresle river
Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel

Angilla rostrata

A. rostrata	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*
North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic
Canada	Canada	Canada	Canada	Canada	Canada	Canada	Canada
Number	Number	Number	Number	cpue	Number	ycsi	(eels/100m²)
Mouses-Saunders ladd	Beauharnois west ladd	Beauharnois east ladd	Chambly ladd	Bay of Quinte trawling (CPUE)	Lake Ontario electrofishing	Sud-Ouest River (YCSI)	**Atlantic Provinces Restigouche River electrofishing**
Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel

A. rostrata	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*	*A. rostrata*
North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic	North Atlantic
Canada	Canada	Canada	Canada	Canada	Canada	Canada	Canada
(eels/100m²)	(eels/100m²)	(eels/100m²)	(eels/100m²)	(eels/100m²)	number	number	number
Atlantic Provinces Miramichi River electrofishing	**Atlantic Provinces Nashwaak River electrofishing**	**Atlantic Provinces Big Salmon River electrofishing**	**Atlantic Provinces St. Marys River electrofishing**	**Atlantic Provinces LaHave River electrofishing**	**Newfoundland Campbellton R. Fence**	**Newfoundland Conne R. Fence**	**Newfoundland W. Arm Brook Fence**
Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel	Yellow eel

*** Data resides with the WG/ICES and can be requested from ICES or a Working Group member.**

Contact the Country Report Author for permission before using for analysis.

Table 2.3. GLM predictions for selected years according to the area in percentage of mean [1960–1979].

	Glass eel		Yellow eel
	Elsewhere Europe	North sea	Europe
1950	0.594	0.230	1.84
1951	0.497	0.297	2.34
1952	0.351	1.060	2.34
1953	0.496	0.922	3.78
1954	0.682	1.535	1.82
1955	0.437	1.532	2.79
1956	0.483	1.121	1.34
1957	0.593	0.622	1.49
1958	0.434	1.235	1.51
1959	0.691	1.516	3.15
1960	1.277	1.853	1.60
1961	1.093	1.094	1.69
1962	1.411	1.687	1.64
1963	1.720	2.092	1.39
1964	0.914	1.046	0.55
1965	1.242	0.805	1.02
1966	0.771	0.806	1.42
1967	0.785	0.910	0.92
1968	1.312	1.120	1.57
1969	0.568	0.760	1.05
1970	0.971	0.886	0.51
1971	0.551	0.580	0.56
1972	0.533	0.860	1.03
1973	0.594	0.417	1.25
1974	0.904	1.070	0.58
1975	0.686	0.511	1.09
1976	1.114	0.992	0.34
1977	1.027	0.790	0.65
1978	1.129	0.924	0.61
1979	1.397	0.796	0.53
1980	1.180	0.687	0.89
1981	0.906	0.540	0.36
1982	1.022	0.335	0.47
1983	0.498	0.268	0.42
1984	0.576	0.094	0.31
1985	0.534	0.098	0.61
1986	0.357	0.092	0.40
1987	0.648	0.105	0.44
1988	0.656	0.091	0.57
1989	0.471	0.045	0.34
1990	0.389	0.146	0.27
1991	0.181	0.028	0.40
1992	0.249	0.063	0.19

	Glass eel		Yellow eel
	Elsewhere Europe	North sea	Europe
1993	0.284	0.065	0.12
1994	0.300	0.074	0.54
1995	0.321	0.050	0.12
1996	0.283	0.052	0.10
1997	0.358	0.044	0.20
1998	0.206	0.030	0.13
1999	0.236	0.057	0.21
2000	0.199	0.046	0.16
2001	0.098	0.009	0.17
2002	0.146	0.027	0.36
2003	0.123	0.025	0.18
2004	0.079	0.007	0.23
2005	0.099	0.016	0.07
2006	0.070	0.005	0.12
2007	0.070	0.013	0.19
2008	0.058	0.006	0.08
2009	0.042	0.010	0.07
2010	0.048	0.008	0.13
2011	0.042	0.006	0.13
2006–2011	0.052	0.009	0.09

Table 2.5. Total landings (all life stages) from 2011 Country Reports, except note Finland, Latvia, Lithuania, Netherlands, Portugal, Spain, France and UK (see Table notes at bottom of table). Norway (NO), Sweden (SE), Finland (FI), Estonia (EE), Latvia (LV), Lithuania (LT), Poland (PL), Germany (DE), the Netherlands (NL), Belgium (BE), UK (UK), Ireland (IE), France (FR) and Spain (ES), Portugal (PT) and Italy (I).

	NO	SE	FI □	EE	LV □	LT □	PL	DE	DK	NL ●	BE	GB √	IE	FR Δ	ED ●	PT #	I
1945	102	1664							4169	2668							
1946	167	1512			1				4269	3492							
1947	268	1910			10	8			4784	4502							
1948	293	1862			10	14			4386	4799							
1949	214	1899			11	21			4492	3873					9		
1950	282	2188			14	29			4500	4152					4		
1951	312	1929			13	32			4400	3661					92		
1952	178	1598			14	39			3900	3978					102		
1953	371	2378			30	80			4300	3157					97		
1954	327	2106			24	147	609		3800	2085					112		
1955	451	2651			47	163	732		4800	1651					117		
1956	293	1533			26	131	656		3700	1817					124		
1957	430	2225			25	168	616		3600	2509					97		
1958	437	1751			27	149	635		3300	2674					128		
1959	409	2789			30	155	566	84	4000	3413					120		
1960	430	1646			44	165	733	51	4723	2999					125		
1961	449	2066			50	139	640	48	3875	2452					125		
1962	356	1908			46	155	663	67	3907	1443					119		
1963	503	2071			64	260	762	55	3928	1618					115		
1964	440	2288			43	225	884	56	3282	2068					108		
1965	523	1802			41	125	682	56	3197	2268		566			97		
1966	510	1969			43	238	804	68	3690	2339		618			126		
1967	491	1617			46	153	906	92	3436	2524		570			133		

	NO	SE	FI □	EE	LV □	LT □	PL	DE	DK	NL ●	BE	GB √	IE	FR Δ	ED ●	PT #	I
1968	569	1808			34	165	943	103	4218	2209		587			140		
1969	522	1675			43	134	935	302	3624	2389		607			127		2469
1970	422	1309			29	118	847	238	3309	1111		754			146		2300
1971	415	1391			29	124	722	255	3195	853		844			166		2113
1972	422	1204			25	126	696	239	3229	857		634			109		1997
1973	409	1212			27	120	636	257	3455	823		725			91		588 *
1974	368	1034			20	86	796	224	2814	840		767			100		2122
1975	407	1399			19	114	793	226	3225	1000		764			110		2886
1976	386	935	28		24	88	803	205	2876	1172		627			142		2596
1977	352	989	63		16	68	903	214	2323	783		692			89		2390
1978	347	1076	77		18	70	946	163	2335	719		825			137		2172
1979	374	956	77		21	57	912	158	1826	530		1206			90		2354
1980	387	1112	79		9	45	1221	140	2141	664		1110			102		2198
1981	369	887	39		10	27	1018	131	2087	722		1139			90		2270
1982	385	1161	38		12	28	1033	166	2378	842		1189			146		2025
1983	324	1173	38		9	23	822	155	2003	937		1136			71		2013
1984	310	1073	28		12	27	831	114	1745	691		1257			98		2050
1985	352	1140	28		18	29	1010	477	1519	679		1035			100		2135
1986	272	943	28		19	32	982	405	1552	721		926		2462	63		2134
1987	282	897	19		25	20	872	359	1189	538		1006		2720	84		2265
1988	513	1162			15	23	923	364	1759	425		1110		2816	55		2027
1989	313	952			13	21	752	379	1582	526		1172		2266	46	14	1243
1990	336	942			13	19	697	374	1568	472		1014		2170	37	13	1088
1991	323	1084			14	16	580	335	1366	573		1058		1925	35	23	1097
1992	372	1180			17	12	584	322	1342	548		915		1585	40	30	1084
1993	340	1210		59	19	10	495	250	1023	293		857		1736	41	34	782

	NO	SE	FI □	EE	LV □	LT □	PL	DE	DK	NL ●	BE	GB √	IE	FR Δ	ED ●	PT #	I
1994	472	1553		47	19	12	531	246	1140	330		1077		1694	34	27	771
1995	454	1205		45	38	9	507	242	840	354		1312		1832	49	24	1047
1996	353	1134		55	24	9	499	220	718	300		1246		1562	61	26	953
1997	467	1382		59	25	11	384	263	758	285		1190		1537	61	25	727
1998	331	645		44	30	17	397	28	557	323		943		1345	79	23	666
1999	447	734		65	26	18	406	38	687	332		963		1253	91	23	634
2000	281	561		67	17	11	305	36	600	363		702		1200	85	22	588
2001	304	543		65	15	12	296	141	671	371		742	98	1103	149	15	520
2002	311	633	0	50	19	13	236	130	582	353		650	123		157	27	415
2003	240	565	1	49	11	12	204	125	625	279		574	111		142	11	446
2004	237	551	0	39	11	16	148	117	531	245		634	136		110	9	379
2005	249	628	0	36	11	22	284	108	520	234		545	101		126	7	75 *
2006	293	670	1	33	8		257	87	581	230		408	133		114	10	56 *
2007	194	568	1	31	10		244	317	526	130		427	114	698	152	11	277
2008	211	495	1	30	13		227	398	457	122		397	125	657	79	7	56*
2009	69	388	2	22	5		156	446	467	275		458	0		99	8	280
2010	32	417	2	19	9		178	313	422	502	0	434	0	781	76	11	249
2011													0		85	2	

□ From 2008 CR, Country not present in 2009

• Partial, for area (Neth) or life stage (Spain)

*** Only freshwater**

√ From 2008 CR, data source unknown

Δ Partial, discontinued

#Coastal yellow eel landings only (Portugal)

Table 2.7. Stocking of glass eel. Numbers of glass eels (in millions) stocked in Sweden (SE), Finland (FI), Estonia (EE), Latvia (LV), Lithuania (LT), Poland (PL), Germany (DE), the Netherlands (NL), Belgium (BE), Northern Ireland (NI), France (FR), Spain (ES) and Canada (CAN - *A. rostrata*).

	SE	FI	EE	LV	LT	PL	DE	NL	BE	N.Irl.	FR	ES	Total	CAN – A.R
1927				0.3										
1928					0.1									
1929					0.2									
1930														
1931				0.4	0.2									
1932					0.2									
1933				0.3	0.2									
1934					0.3									
1935				0.2	0.6									
1936					0.3									
1937				0.3	0.3									
1938					0.4									
1939				0.2	0.1									
1940														
1941														
1942														
1943														
1944														
1945													0	
1946								7.3					7.3	
1947								7.6					7.6	
1948								1.9					1.9	
1949								11					10.5	
1950								5.1					5.1	
1951								10					10.2	
1952						18		17					34.5	
1953						26	2.2	22					49.6	
1954						27	0	11					37.1	
1955						31	10	17					57.5	
1956			0		0.3	21	4.8	23					49.4	
1957						25	1.1	19					44.8	
1958						35	5.7	17					57.6	
1959						53	11	20					83.3	
1960			1	3.2	2.3	64	14	21					105.3	
1961						65	7.6	21					93.7	
1962			1	1.9	2	62	14	20					100.3	
1963				1.5	1	42	20	23					87.8	
1964			0	0.9	2.4	39	12	20					74.4	
1965			1	0.4	2.1	40	28	23					93.3	
1966		1.1			0.7	69	22	8.9					101.6	
1967		3.9		1	0.5	74	23	6.9					109.3	
1968		2.8	1	3.7	3	17	25	17					69.7	

	SE	FI	EE	LV	LT	PL	DE	NL	BE	N.Irl.	FR	ES	Total	CAN – A.R
1969					0	2	19	2.7					23.9	
1970			1	1.8	2.8	24	28	19					75.6	
1971					1.6	17	24	17					60.3	
1972			0	1.6	0.3	22	32	16					71.1	
1973					1.4	62	19	14					96	
1974			2		1.8	71	24	24					122.7	
1975					2.2	70	19	14					105.2	
1976			3	0.6	1	68	32	18					121.7	
1977			2	0.5	1.4	77	38	26					145.2	
1978		3.7	3		2.7	73	39	28					148.8	
1979					0.8	74	39	31					144.65	
1980			1		1.8	53	40	25					120.5	
1981			3	1.8	3	61	26	22					116.4	
1982			3		4.6	64	31	17					119.4	
1983			3	1.5	3.7	25	25	14					72.1	
1984			2			49	32	17		4			103.1	
1985			2	1.5	1.6	36	6	12		10.9			70.52	
1986			3		2.6	54	24	11		17.8			111.61	
1987			3	0.3		57	26	7.9		13.8			107.55	
1988				2.2		16	27	8.4		6.32			59.42	
1989						5.9	14	6.8					27	
1990	0.7	0.1				8.6	17	6.1					32.2	
1991	0.3	0.1	2			1.7	3.2	1.9					9.2	
1992	0.3	0.1	3			14	6.5	3.5		2.36			29.06	
1993	0.6	0.1				11	8.6	3.8	0.8				24.5	
1994	1.7	0.1	2		0.1	12	9.5	6.2	0.5	2.32			34.52	
1995	1.5	0.2		0.6	1	24	6.6	4.8	0.5	2.06			40.96	
1996	2.4	0.1	1		0.4	2.8	0.8	1.8	0.5	0.1		0.1	10.37	
1997	2.5	0.1	1			5.1	1	2.3	0.4	0.21		0.1	12.58	
1998	2.1	0.1	1		0.1	2.5	0.4	2.5		0.05		0.1	8.36	
1999	2.3	0.1	2	0.3		4	0.6	2.9	0.8	3.6		0.2	17.02	
2000	1.4	0.1	1			3.1	0.3	2.8		0.45		0.1	9.23	
2001	0.8	0.1				0.7	0.3	0.9	0.2			0	3	
2002	1.7	0.1		0.2			0.3	1.6		3.02		0	6.94	
2003	0.8				0.4	0.5	0.1	1.6	0.3	4.1		0.1	7.89	
2004	1.3	0.1				2.3	0.2	0.3		1.28		0.1	5.5	
2005	1	0.1		0.1			0.6	0.1		2.16			4.05	0.6
2006	1.1	0.1		0				0.6	0.3	0.99			3.08	1.2
2007	1	0.1		0			1.6	0.2	0	3		0	5.98	0.9
2008	1.4	0.2							0.3	1.28			3.17	2.7
2009	0.8	0.1					0.2	0.3	0.4	0.65			2.37	1.3
2010	1.9	0.2					3.7	2.7	0.4	3	1	0	12.82	
2011		0.3	1	0.1			1.7	0.8	0.4	3.1	2	0.1	9.32	

Table 2.8. Stocking of young yellow eel. Numbers of young yellow eels (in millions) stocked in Sweden (SE), Finland (FI), Estonia (EE), Latvia (LV), Lithuania (LT), Poland (PL), Germany (DE), Denmark (DK) the Netherlands (NL), Belgium (BE), and Spain (ES).

	SE	FI	EE	LV	LT	PL	DE	DK	NL	BE	ES	Total
1947									1.6			1.6
1948									2			2
1949									1.4			1.4
1950							0.9		1.6			2.5
1951							0.9		1.3			2.2
1952							0.6		1.2			1.8
1953							1.5		0.8			2.3
1954							1.1		0.7			1.8
1955							1.2		0.9			2.1
1956							1.3		0.7			2
1957							1.3		0.8			2.1
1958							1.9		0.8			2.7
1959							1.9		0.7			2.6
1960							0.8		0.4			1.2
1961		0		1			1.8		0.6			3.5
1962		0		0.7			0.8		0.4			2
1963				0.4			0.7		0.1			1.2
1964		0		0.4			0.8		0.3			1.6
1965		0		0.3			1		0.5			1.9
1966		0					1.3		1.1			2.5
1967				0.8			0.9		1.2			2.9
1968							1.4		1			2.4
1969							1.4					1.4
1970				0.4			0.7		0.2			1.3
1971							0.6		0.3			0.9
1972							1.9		0.4			2.3
1973						0.2	2.7		0.5			3.4
1974							2.4		0.5			2.9
1975							2.9		0.5			3.4
1976				0.3			2.4		0.5			3.2
1977						0.1	2.7		0.6			3.4
1978							3.3		0.8			4.1
1979		0					1.5		0.8			2.4
1980							1		1			2
1981							2.7		0.7			3.4
1982				0.3		0.1	2.3		0.7			3.4
1983				0.4		2.3	2.3		0.7			5.7
1984						0.3	1.7		0.7			2.7
1985						0.5	1.1		0.8			2.4
1986						0.2	0.4		0.7			1.3
1987							0.3	1.58	0.4			2.28
1988			0.2	0.8		0.1	0.2	0.75	0.3			2.35

	SE	FI	EE	LV	LT	PL	DE	DK	NL	BE	ES	Total
1989						0.7	0.2	0.42	0.1		0.06	1.48
1990	0.8					1	0.4	3.47			0.03	5.7
1991	0.9					0.1	0.5	3.06			0.06	4.62
1992	1.1					0.1	0.4	3.86			0.06	5.52
1993	1						0.7	3.96	0.2	0.2	0.17	6.23
1994	1				0.1	0.1	0.8	7.4		0.1	0.12	9.62
1995	0.9		0.2				0.8	8.44		0.1	0.22	10.66
1996	1.1					0.5	1.1	4.6	0.2	0.1	0.1	7.7
1997	1.1					1.1	2.2	2.53	0.4	0.1	0.14	7.57
1998	0.9				0.1	0.6	1.7	2.98	0.6	0.1	0.09	7.07
1999	1				0.1	0.5	2.4	4.12	1.2	0.04	0.04	9.4
2000	0.67					0.8	3.3	3.83	1		0.05	9.65
2001	0.44		0.44			0.6	2.4	1.7	0.1		0.06	5.74
2002	0.26		0.36	0.2		0.6	2.4	2.43	0.1	0.01	0.04	6.4
2003	0.27		0.54			0.50	2.60	2.24	0.10	0.01	0.06	6.32
2004	0.18		0.44		0.10	0.50	2.20	0.75	0.10	0.01	0.06	4.34
2005	0.07		0.37			0.70	2.10	0.30		0.01	0.12	3.67
2006	0.003		0.38			1.10	5.50	1.60				8.58
2007	0.03		0.33			0.90	9.10	0.83			0.02	11.21
2008	0.12		0.19			1.00		0.75	0.23		0.04	1.58
2009	0.02		0.42			1.40	4.76	0.81	0.30		0.02	7.73
2010			0.21			1.40	3.84	1.55	0.10		0.01	7.11
2011						2.60	2.70	1.56	4.50			11.36

Annex 5: Occurrence of European eel, *Anguilla anguilla*, in scientific bottom-trawl surveys

Håkan Westerberg
Guldborg Søvik
Henrik Sparholt
Stein Tveite

Summary

The distribution in space and time of eels in the catches from three bottom-trawl surveys in the North Sea, Skagerrak, Kattegat and the Baltic have been analysed. The surveys (ICES BITS and NS-IBTS surveys and the Norwegian Pandalus survey, PA) are mainly conducted in the first or fourth quarters and cover the following time periods:

	Quarter 1		Quarter 4	
Survey	From	To	From	To
BITS	1991	2011	1991	2010
NS-IBTS	1978	2011	1991	1994
PA	2006	2011	1984	2005

The total numbers of hauls in the analysis are:

	Quarter	
Survey	1	4
BITS	4562	3024
NS-IBTS	12 222	1079
PA	489	2038
Total	17 273	6141

The mean catch per trawl hour was approximately 0.01 eels in all surveys during the first quarter. In the fourth quarter the cpue was more than 1 in the Baltic and 0.1 in the NS-IBTS.

At least 80% of eels caught in the surveys are migrating silver eels, which are shown by the size distribution and the seasonal variation. There is a clear diurnal variation in the catchability of the eels, which indicates that the eels are pelagic in the night and at the bottom in the day. The migration route from the Baltic seems to be through the Kattegat, Skagerrak and along the Norwegian Trench north to at least 60°N.

It is noteworthy that none of the time-series shows a significant decrease in eel abundance, despite the fact that they cover a time period with a large decrease in recruitment. Indeed most have a positive, although not statistically significant, trend. The mean age of the Baltic eels is approximately 12 years. This means that the BITS time-series 1991 to 2010 corresponds to the recruitment during the period 1979 to 1998. During this time interval the European glass eel recruitment index dropped by 96% in the North Sea area and 85% elsewhere.

The maximum deviation from the mean trend within a 95% confidence limit corresponds to a 78% decrease in the Baltic. The reason why the silver eel abundance does not reflect the decrease in recruitment should be investigated further, taking stocking

of eels and other factors into account. For the North Sea the variability of the data is too large for a simple trend analysis.

The average cpue per ten meter depth interval during the 4th quarter was calculated for the whole 1991–2010 year period in the southwestern area of the Baltic. Combining the area per depth interval with the number of eels per unit area we get an estimate of the total number of eels of 2 800 000. A corresponding calculation for the 1st quarter shows that a total of 38 000 eels remain. The conclusion is that almost 3 million eels must have left this area of the Baltic yearly and presumably migrated out through the Sound and Belts.

The analysis of the BITS, NS-IBTS and PA surveys show that the bottom-trawl data gathered in scientific surveys are valuable also for eel studies. Together they give a relatively complete coverage of the North Sea, Skagerrak–Kattegat and southern Baltic. They provide fishery-independent data on the migration phase of the eel which otherwise are rare. Especially the BITS time-series can provide a valuable index of eel escapement for the whole Baltic drainage area.

Annex 6: The context of eel fisheries and the concerns

Eel has been an important resource throughout Europe and the decline of the stock is a biological, socio-economic and cultural loss (Moriarty and Dekker, 1997; Dekker, 2002; Feunteun, 2002; Ciccotti, 2005; Castelnaud and Beaulaton, 2008).

All eel life stages are targeted by fisheries throughout its European range; however such fisheries reflect local traditions of availability and market or consumption customs. On the Atlantic coasts of France, Spain, Portugal and the Severn estuary in England, where glass eel ascent has been historically plentiful, specific glass eel fisheries have been developed for direct human consumption, stocking inland waters or the provision of seed for aquaculture. Yellow eel have been fished in inland and coastal waters throughout Europe and North Africa, using a variety of fishing gears including lines, nets and traps. Silver eels are intercepted using barriers, nets and traps during their downstream migration and although found throughout Europe, silver eel fisheries are predominantly located along Scandinavian coasts and Mediterranean lagoons.

Large-scale fisheries for eel are rare and account for less than 5% of the total European catch (Dekker, 2002a). The remaining small-scale fisheries distributed throughout Europe and the Mediterranean may be commercial, semi-commercial or recreational. The processing and trade industries are organized into larger size companies and operate on an international scale (Dekker, 2002a).

Since 2002, ICES has pointed out the urgent need for an eel recovery plan, to include measures to reduce exploitation and other mortality of all life stages and restore habitats. This led to the development of the Regulation 1100/2007 which required the implementation in 2009 of Eel Management Plans by EU Member States. Each Plan contains a series of measures aimed at achieving an escapement of 40% of silver eel relative to pristine production.

In order to achieve compliance, many MS have imposed restrictions on eel fishing, or in some cases fishery closures. The information currently available on the effective extent of reduction is not sufficient to get a clear view of the current status of fisheries. The real extent of eel fisheries reduction will therefore constitute a point to be ascertained in 2012, when Member States report to the EU in accordance with Article 9 of the Regulation. The socio-economic consequences of catch reductions, either as a consequence of the diminishing resource, or legal restrictions, needs to be assessed, to obtain a better understanding of the socio-economic and biological consequences of management actions. Furthermore, the strong interactions between eel fisheries, the aquaculture sector, and other market forces, including the plans for a European wide transport of glass eel for stocking, also need to be thoroughly investigated. Finally, the progressive decline of small-scale eel fisheries is creating a loss of the cultural heritage (fishing techniques, skills, gastronomy) for the future generations.

Proposal from WGEEL 2010

Management of fisheries includes economic, social and political issues along with scientific advice on the status of the exploited stocks. Where so-called traditional fisheries are involved, this puts additional pressure on the system as many traditions, practices, techniques and even local gastronomies and recipes may be changed or lost.

Climate change is putting additional pressures on the system with fisheries and fisheries managers have to adapt to cope with changing environment and species abundance and availability. Diadromous species, and their fisheries, are particularly vulnerable in this context.

It is proposed to establish a project to examine and document the eel fisheries, their extent, socio-economic status and future.

- establish the previous status of the eel fishery in the various regions and river basins with a **historical** approach that would permit a reference situation on a social, economic and ecological point of view: what we had, what we have lost (what is the loss), what could be the objective of restoration;
- the maintenance of an eel fishery and obviously the objective of restoration, development of this activity in future as proposed above, need to meet with the conditions of sustainable development.

These need to use basic biological and socio-economic indicators (see Indicang; Adan *et al.*, 2008; Castelnaud and Beaulaton, 2008). For practical reasons, the historical approach generally only uses these basic indicators particularly in socio-economic (population, production, turnover) but the current approach and a prospective approach could involve research with elaborate sociology and economic indicators that require the application of methods geared to these fields.

Additional questions to be tackled might include:

1) Is the species in its fishing area able to accept a fishing pressure and of what magnitude?
2) Are the eel fisheries economically viable, in connection with the fishery context of the fishing area (*mono or multispecies, full or part time activity*)?
3) Are the fishers able to integrate the concept of sustainable development for their activity and participate in a global management process? What are the conditions and consequences?

In order to tackle these questions, point 1 needs fishery biology indicators, point 2 needs economic indicators and point 3 needs sociological indicators.

Points 1 and 2 are connected with the fishery monitoring systems for a part of the information required and the objective of assessment of eel fisheries sustainability give a new interest to the traditional statistics. Such assessment and diagnosis involves the improvement and creation of appropriate monitoring systems which help in their basic theoretical function which is to fill-in local, national and FAO fisheries statistical databases.

Point 3 gives a central role to the fishers in the feasibility of the investigation and the process of evaluation and decision, which will consider the vision of a particular user of the inland water services. This will challenge the scientists and the community concerning the priorities to be taken into account for the management of eel in its socio-politic, economic and ecological context. This would also feed into a review of how the EU Regulation has impacted on individual fishers and local communities.

Annex 7: Migration patterns and orientation in restocked eels–a scientific review

Fiskeriverket
Avdelningen för forskning och utveckling
Sötvattenslaboratoriet
Håkan Wickström
Patrik Clevestam
Niklas B. Sjöberg
Willem Dekker

Kunskapsgenomgång
Datum
2010-03-10 (translated 2011-08-22)

[This is a translation from "**Vandringsmönster och orienteringsförmåga hos utsatt ål – en kunskapsgenomgång**", an unnumbered document dated March 10, 2010].

Mission

We have chosen to structure our task into two steps by summarizing the literature on 1. Growth and survival of restocked eels in comparison to natural immigrants, and 2. Orientation, navigation and migration of silver eels of restocked origin.

We consider the problem involved in the last point to be the central question, and will therefore treat the first point somewhat more superficially, amongst others because there is no controversy on this point between scientists or managers. The latter question is often debated, mostly because knowledge of the ocean migration and spawning stock is still very limited. Thus, this will get most emphasis.

Implementation

The number of publication (peer-reviewed and other published sources) on this subject is so small, that we have chosen to add grey or unpublished literature obtained from colleagues in and outside Europe.

We will not discuss genetic aspects, but refer the reader to Dannewitz *et al.* (2005) and Palm *et al.* (2009); these publications indicate that the eel is panmictic; from this standpoint, there is no argument against transport of eels within its distribution area. Today, most scientists agree that the stock is panmictic (FAO/ICES 2009). Additionally, we will not discuss whether there is a local surplus of eels anywhere, but refer to Bark *et al.* (2007), who consider the situation in England and Wales. Two French eel researchers recently said that there is hardly a local surplus anymore (Briand *et al.*, 2007; Beaulaton, 2008; Briand, 2009; Briand pers. comm.; Beaulaton pers. comm.). The use of glass eel for restocking is regulated by the EU Eel Regulation (EG nr 1100/2007) and indirectly also by the trade restrictions imposed by CITES (CITES 2008).

In some cases, we will cover trap & transport of young eels upstream within their own river too.

Our review will summarize individual publications and discuss specific topics, as follows:

1) The phase between restocking and final departure from the area where it was restocked.
2) The contribution of restocked eels to the silver eel run, in its initial phase.
3) Orientation, navigation and migration.
4) Ocean migration.

Literature review

1) The phase between restocking and final departure from the area where it was restocked

Many scientific studies show that restocked eels survive, grow and mature to the silver eel stage, and, if physically possible, start their outward (ex. Westin, 1990; Wickström *et al.*, 1996; Pedersen, 2000; Wickström, 2001; Clevestam and Wickström, 2008). This holds more or less equally for eels restocked as glass eel, following a period in aquaculture, and bootlace eels obtained from nearby rivers (in Sweden: from the west coast). Silver eels of restocked origin are being caught in substantial quantities in lakes such as Ången and Fardume Träsk, where no natural eel stock occurs. In lake Ången, 17% of the restocked eels were recaptured as silver eel (860 eels), using an emigration-trap (Wickström, 1986a; 2001 and unpublished). In lake Fardume Träsk, more than 7000 silver eels (13%) had left the lake by the year 2000 (Wickström *et al.*, 1996; Wickström, 2001). But in the lakes Götemaren and Frisksjön, very little eels have been caught in the traps, despite good recaptures in earlier surveys in the lakes (Wickström, 1986b; Wickström *et al.*, 1996).

There is a substantial variation between lakes in size and weight of silver eels of restocked origin, their growth and age at silvering (Wickström *et al.*, 1996; Clevestam and Wickström, 2008). The age at silvering is negatively correlated with growth rate (Svedäng *et al.*, 1996; Clevestam and Wickström, 2008). The variation in age can, to some extent, be explained by environmental conditions. Svedäng *et al.* (1996) concluded that habitat differences and temperature determine the growth, and this conclusion is corroborated by Clevestam and Wickström (2008) and Lin *et al.* (2007). Habitat quality is apparently determining growth and age at silvering.

Silver eels of restocked origin, coming from freshwater environments, show a lower silvering index (gonadosomatic index, stomach & intestines index, eye-index, fin index) then emigrating eels in the Baltic outlet (Clevestam and Wickström, 2008). There are strong indications that silver eel leaving freshwater are less mature than those in the Baltic outlet anyhow, which is in line with other studies showing the gradual transition from yellow to silver stage (Durif *et al.*, 2005). What is considered a silver eel in inland lakes and on the Baltic coast is simply not the same as what is caught in the Baltic outlet, but it is not clear if and how this is related to the restocking discussion (Clevestam and Wickström, 2008; Sjöberg *et al.*, 2008).

The choice of restocking material (glass eel or bootlace) does not influence growth, age at silvering or silvering index (Clevestam and Wickström, 2008). However, the combination of chosen seed material and local habitats might affect results. For example, in lake Roxen, glass eel grows faster, attains a lower fat content, lower age and GSI than bootlace eels. However, these results are rather preliminary (Clevestam and Wickström, 2008). Comparing restocked and natural eels in Lithuania (Lin *et al.*, 2007) did not show differences in growth, too. However, it was shown that natural eels are smaller than restocked eels of the same age, which might be the result of energy loss while migrating the long way towards Lithuania (in this study, no maturing eels were studied).

At the meeting of Study Group on Anguillid Eels in Saline Waters (SGAESAW) in Gothenburg in 2009 (ICES 2009), Tzeng reported results from studies done with wild and farmed eel (*Anguilla japonica*) in an estuary in Taiwan. Tzeng could not find any difference between the marked eels of different origin when it came to re-catch and habitat. The results indicated a reduced growth of the cultured eels, but it could also be an effect of different productivity in the nursery areas where eels spent most of the time (Tzeng, 2009).

The knowledge of survival, growth, size and age-at sexual maturity of eels restocked and living in coastal environments is more uncertain than for vulnerable populations of freshwater because of the difficulty to positively identify eels exposed on the coast.

Otolith chemistry studies of eels from seven, for the eel fishery important lakes located relatively far into the country and at least some immigration barriers in Sweden shows that the commercial catch of migrating eels in these lakes is almost completely dominated by the restocked eels (Clevestam and Wickström, 2008). From Lake Balaton in Hungary, where only restocked eels occur, large quantities of silver eel (75 tons) were caught annually in an emigration trap at Siofok on the south side of the lake (Bíró, 1992; 1997). From the Masurian marshes in northeastern Poland Ciepielewski (1976), Moriarty *et al*. (1990) and Robak (2005) report on the catch of emigrating eels originating in major releases made in the area. Also from Lake Võrtsjärv in central Estonia significant catches of silver eels have been reported as a result of the large-scale restocking, that was initiated after the power plant in Narva blocked all immigration to Lakes Peipsi and Võrtsjärv (Järvalt, 2003). Eel farming in the lagoons at the Italian Comacchio, carried out more or less extensively for decades, is also a good example of how eels of different origin (mainly around the Mediterranean), become silver eels caught (in so-called lavorieri) when they emigrate to the Adriatic Sea (De Leo and Gatto, 2001).

Allen *et al*. (2006) and Rosell (2009) have compared the effect of restocking eels of different origins into Lough Neagh in Northern Ireland. Glass eels came from the estuary of the river Bann (41 km from the lake outlet, the river Bann drains Lough Neagh) and more recently (since 1984) also from the Severn in England (425 km from the lake's outlet). Their model shows that the return, in terms of eel catches, on restocking from Severn-origin can be up to three times lower compared to releases based on eels from the Bann estuary, but probably the difference is smaller. Although the results are uncertain, the study suggests that the imported eels from the Severn could have a reduced ability to survive and grow up. The study also shows that the survival rate of restocked eels strongly depends on density: restocking more than 200 glass eels per hectare per year, does not increase eel production in the lake. However, we cannot determine whether the density-dependent mortality affects the eels imported from England and those transported up from the Bann estuary to the same extent (Rosell, pers. comm.).

In an earlier report of WGEEL (FAO/ICES 2006) Rosell wrote "The heavy additional stocking bought in the period 1984 to 1989 shows clearly in maintained cpue in the period 1999 to 2005, tallying with known growth parameters (in Lough Neagh, our comment). Given the known escapement of silver eels from the Lough Neagh system (Rosell *et al*., 2005) it is highly probable that bothering additional stock bought from the Severn (England) and local trap and transport derived glass eel contribute to spawners (FAO/ICES 2006)."

2) The contribution of restocked eels to the spawning run, first phase

Results from studies based on otolith chemistry (Sr/Ca) of emigrating eels coming out of the Baltic Sea (captured at Kullen, in Køge Bay and south of Lolland) shows that the restocked eels are present among the emigrating silver eel (Limburg *et al*., 2003; Clevestam and Wickström, 2008). Limburg *et al*. (2003) studied the proportion of eels of restocking origin to be 27% of which 20% were restocked into freshwater and 7%

were restocked in brackish water. In this study, silver eels were caught at two sites in the Baltic outlet. The proportion of silver eels of restocked origin was higher among eels caught south of Lolland/Denmark than among eels caught at Kullen/Sweden. Limburg used microchemical otolith analysis to identify the growing environment/origin of the silver eels. Two techniques, namely electron (WDS), and nuclear (PIXE) micro-probe were used.

In a large study of migration of eels caught at Kullen and Køge, Clevestam and Wickström (2008) estimated proportion restocked eels to be 21%. The proportion restocked on the coast as juveniles or bootlace was about 10%, and the proportion of eels restocked into freshwater was about 11%. Only 4.5% were classified as restocked and grown up in freshwater. A very large proportion of eels (70%) were not classifiable as to its recruitment background, and the authors emphasize that the uncertainty in the estimates is great. They argue that at least 12% are of restocked origin and that the proportion may well be higher. The group of undoubtedly restocked origin consists of both glass eels and bootlace with significant freshwater growth, and eels restocked into freshwater that went more or less directly into coastal waters, all with very distinct Sr: Ca patterns. Common to the group of eels with uncertain identification of their origin was that they had full or significant growth in brackish waters. Results from Clevestam and Wickström (2008) also show that the qualitative aspects of the restocked silver eels from the Swedish freshwater (measured in eels from seven lakes of importance to the fishery), were not reflected in the eels captured at Kullen and in Køge Bugt that were thought to be of freshwater origin. Eels from Kullen/Køge growing up in freshwater were younger, smaller (shorter/lighter), had higher fat content and maturity differed when compared to the silver eels caught in the lakes.

Why the proportion of eels from freshwater is so low, we can only speculate about, but numerous migration obstacles and mortality in turbines (Montén, 1985) and fishing mortality can all be involved. Knowledge of the contribution of the different nations and different types of water bodies to the total spawning–stock biomass of silver eel (restocked and/or wild) is currently very limited. It has been speculated that eels caught in southern Sweden, in fact, come from the east of the Baltic Sea. Sjöberg *et al*. (2008) have also shown that silver eels caught and marked in Swedish coastal waters in the 2000s are captured in Køge Bugt and other places in the Sound.

Shiao *et al*. (2006) estimate the proportion of yellow eels of restocked origin in the Curonian Lagoon in Lithuania to be 20% of the stock. The lagoon is located near the Baltic Sea and has open access to the Baltic Sea. The study only included yellow eels. It also showed that restocked eels seem to prefer to remain where they were restocked (this conclusion is based on few individuals).

To which extent exposed eel contribute to silver eel escapement from the Baltic Sea was discussed with Dekker (pers. comm., October 2009). Based on a very rough calculation, he noted that the observed proportion of restocked eels in the silver eel run bears a reasonable relationship with the amount of restocked eels, i.e. they contribute to spawner biomass, but that release is not a panacea: "Taking the data from the 2009 WGEEL report, the number of eels (glass eel and bootlace combined) stocked in the Baltic was approximately 10 million in 1990 and approximately 5 million in 2000. Assuming a survival rate of 10% and individual weight of 0.8 kg per silver eel, this boils down to 800 tonnes in 2000 or 400 t in 2010. Commercial catch (combining yellow and silver eel for all Baltic countries, taking 50% for Germany (The Baltic only, leaving out the North Sea)) was in the order of 1500 t, both in 2000 and 2009. Adding a margin for escaping silvers (10–50%), the total production might be in the order of 2000 t. The expected contribution of restocked eels to the silver eel production equals 20–40%. Due to the many uncertainties, we should consider this a ballpark estimate. More in particular, hydropower mortality impacts the freshwater, and thus might reduce the contribution of restocked eels. Bottom line conclusion: The observed percentage of silver eels from restocking origin is reasonably in agreement with expectation: there is no reason to believe they all fail, nor to consider restocking a panacea. It should be noted that the supplement for the silver eels escaping the fishery in reality is 65–90% of the run. Possibly, this high figure is because silver eel catches are included (?).Sjöberg *et al*. (2008) indicate that the percentage of silver eels being caught peaked in the 1960s at an average of 48% and that the current level is about 30%. If the recapture percentage equals the proportion of silver eels caught in the fishery, it gives an estimated total silver eel production of 3000–5000 tonnes. The expected contribution to the silver eel production from restocked origin will then be 8–27%, and the entire range from 8% to 40% (as above) should be considered possible. That range, while covering the observed proportion of affected eels in the silver eel catches, but unfortunately dos not really provide guidance how well restocked eels contribute to spawner escapement compared with natural immigrants.

Results from Limburg *et al*. (2003) and Clevestam and Wickström (2008) show that migrating eels from different backgrounds/life strategy, caught in/around the Sound, differ in various respects (quality parameters). Limburg writes that the fat content in wild tended to be higher than in presumed restocked eels. Clevestam and Wickström, 2008 note that eels with a catadromous life-strategy (assumed natural recruitment) differ in a way that collectively makes them better equipped for migration and spawning than other groups (in contrast to coastal restocking or in freshwater, and unclassified eels). Somewhat simplified, one can say that the catadromous eels are older, bigger, fatter and exhibit a higher degree of maturity when caught on his way out of the Baltic Sea than other groups. However, this may be an effect of the higher age (age is correlated with length) resulting from a catadromous strategy, i.e. of growth-rate/habitat quality (Svedäng *et al*., 1996; Lin *et al*., 2007; Cleve Tribal and Wickström, 2008).

3) Orientation, navigation and migration

Between 1984 to 1992 Westin (1990, 1998, 2003) carried out 14 tagging experiments with approximately 1800 marked silver eels in different set-ups, contrasting glass eel from known origin (from France, restocked in Fardume Träsk on Gotland) and unknown origin (silver eels caught in the Stockholm archipelago and on the coast of Gotland). A large group (20%) of eels in the study was anosmic (no smell). Average recapture rate from these trials was 16% (9% for those of restocking origin and 20% for those with unknown origin (anosmic eels omitted)). Recapture is lower than expected when compared to other taggings made in the Baltic Sea (Sjöberg and Petersson, 2005). The tagging method (using eels of unknown origin and controls) was not conventional, in the sense that eels were caught, tagged and released at the same site, within a few days. Eels in the other twelve experiments were transported by car, boat or helicopter or held in storage for several weeks before tagging and releasing. The author thinks that recaptures, both geographically (in the southwestern Baltic Sea) and in time (with a high degree of overwintering) shows clearly that the eel of restocked origin lacks the necessary experience/imprinting from natural immigration and therefore do not find their way out of the Baltic Sea (Westin, 1990; 1998; 2003).

In a review of silver eel tagging experiments during 1900s, Sjöberg and Peterson (2005) noted a change in emigration pattern. In the early 1970s, fewer eels are caught in the Sound (Öresund) than in the other exits of the Baltic (based on 42 tagging experiments, approximately 7000 marked eels). They further discuss that this may be a result of extensive restocking, but that it may also be natural for Swedish eels to go through all exits from the Baltic Sea and that many factors may be governing the choice.

Statistical analysis of a much larger material of historical tagging, based on about 300 experiments in which 40 000 silver eels tagged (made between 1903 and 2006) and a recapture rate of an average of about 33%, do not show any clear effect on re-catching patterns that can related to the extensive restocking programmes in the Baltic region (Sjöberg *et al.*, 2008). The study indicates that although some impacts might be present, these will be hidden by all historical changes such as fluctuating fishing pressure, which precludes an unambiguous analysis. Results from chemistry analysis of re-captured tagged silver eels indicate that the majority (90%) of the re-captures have grown up in brackish water only (based on four label trials with 707 eels in which 192 recaptures were analysed). Recaptures of brackish water eels require a finer resolution of the otolith chemistry to allow for a more secure identification of the origin. Furthermore, the authors used eels from the natural environments around the Baltic Sea, apparently consisting of both natural migrants and restocked ones. The results show a well-functioning navigation ability in the recaptured migrating silver eels (Sjöberg *et al.*, 2008).

Tagging of eel in lake Mälaren, mostly consisting of restocked origin, shows that recaptures are made mostly within the lake itself, west of the release site, which indicates that something is not right with the migratory behaviour (Sjöberg and Wickström, 2008). However, the few eels caught in places outside the lake, in the Östergötland archipelago, the coast of Skåne, in Kögebukten and the Great Belt, migrated in the right direction towards the Baltic outlet (Sjöberg, unpublished.). The study is continued and data storage tags will indicate whether westward migrating eels have overwintered (Sjöberg and Westerberg, unpublished material). There are also examples where eels migrated west and then turned back and found out towards the outlet to the Baltic. It is too early to say whether overwintering of supposed silver eels in lakes is a natural behaviour or an effect that can be connected to the restocking origin. Older tagging studies in Mälaren have shown overwintering behaviour (Sjöberg and Wickström, 2008) and work is underway to clarify whether these eels were restocked or natural (Sjöberg, pers. comm.). Overwintering in rivers has been observed in natural eels but it is not known what causes this behaviour (Feunteun *et al.*, 2000).

Since 2006, 869 eels of varying size and sexual maturity have been tagged in Estonia, upstream of hydroelectric power plants in Narva. All eels in the area above the power plant were considered to be of restocked origin. A total of 93 eels are recaptured, mostly in the lakes they were marked. The few re-captures (7%) that were made outside the immediate vicinity of the tagging area were all made in a direction towards the outlet of the Baltic Sea, in the Sound (Anonymous, 2008; Järvalt, 2009; Järvalt, pers. comm.).

In the Vistula lagoon in Poland's Baltic coast, there are indications that silver eels, probably of stocked origin, do not orient towards the lagoon outlet in the northern, Russian part of the lagoon, but initially migrate westward where there is no connection to the sea (Wilkońska and Psuty, 2008). The authors interpret this phenomenon as that restocked eels may not find the outlet to the lagoon and in this context they refer the hypotheses of Westin (1998).

On tagging experiments in Roskilde Fjord (which has a long and complex connection to the Kattegat) with natural eels and eels of restocked origin, Pedersen (2009) and Pedersen (pers. comm.) wrote: "In autumn 2004 and 2005 silver eels of stocked (N=143) and wild (N=450) origin were Carlin-tagged and released in the bottom of the fjord. The result was a higher recapture rate of wild eels N= 126 (28%) compared to stocked eels N= 27 (19%) but the difference is not statistically different ($\chi 2$ test, P=0.12). Independent of eel origin (wild and stocked), both eel types were caught in the same proportion in the southern part of the fjord (56%; wild N=71; farmed N=14) and in the northern part of the fjord (44% wild N=55; farmed N=10), indicating that the stocked eels migrate toward the outlet of the fjord together with the wild silver eels ($\chi 2$ test, P=0.94)."

In Canada too, there are studies of silver eels of restocked origin migrating at an apparently normal manner together with natural silver eels towards the open sea. This concerns the American eel, *Anguilla rostrata*. Verrault (submitted and pers. comm.) recaptured six silver eels on the coast of Nova Scotia, in Kamouraska, that were marked as glass eel only four years ago in the Richelieu River (downstream of Lake Champlain) about 500 km upstream. The author interprets this as first evidence that the restocked eels migrate along with naturally recruited eels, at least so far as to the estuary. These recaptured eels had a very rapid growth, which is probably explained by their low age and small size at transformation into silver eels.

Orientation and navigation has been studied by many researchers (e.g. Hanson *et al.*, 1984; Karlsson, 1985; Hanson and Westerberg, 1987; Tesch *et al.*, 1992; Westerberg and Begout-Anras, 2000; Nishi *et al.*, 2004; 2005; Nishi and Kawakura, 2005). During the 2000s, Durif conducted advanced research at the Norwegian Austevoll Research Station. At WGEEL 2006 (FAO/ICES 2006) she reported two magnetic senses that eels may have, namely, an inclination, respectively a polar compass. By knowing the slope and intensity of the earth magnetic field an eel can determine how close the pole is in relation to the equator, i.e. how far north in the Atlantic it is and the compass sense then indicates the direction. Having these two magnetic senses, the eel would theoretically know its position in the northern hemisphere along a north–south gradient, even after a possible relocation, but would not know its east–west position. After that meeting, Durif continued testing eels in manipulated magnetic fields. Her current work and unpublished results indicate that silver eels once they are out in the Atlantic can orient towards lower magnetic field areas, i.e. towards the equator, until the right field strength has been found, and then follow an iso-line towards the Sargasso Sea. However, the two magnetic senses give no clue of how to migrate out of a lake, a fjord and so on, towards the open sea, without additional information (Durif and Skiftesvik, 2007; Durif, pers. Comm.). Generally, migrating animals integrate information from multiple senses (McKeown, 1984).

Nishi and Kawakura (2005) found that the Japanese eel (*Anguilla japonica*) have a magnetic susceptibility already in the glass eel stage. They suggest that the geomagnetic information naturally immigrating glass eels experience, contributes their ability in the silver eel stage to find their way back to the spawning area. Magnetic susceptibility in the larval stage (leptocephali) has not been studied.

The parasite (*Anguillicoloides crassus*) causes damage to the swimbladder of the eel. Eel with severely infected or damaged swimbladders are unlikely to reach the Sargasso Sea (Palstra *et al.*, 2007). It has also been shown that infection rates in the swimbladder is important for the ability to migrate (Sjöberg *et al.*, 2009): eels with a low infection are more successful in their migration and are more capable to carry vertical migration patterns (Westerberg *et al*., 2007), which are probably required to satisfy different physiological requirements during the long spawning migration (Scaion *et al*., 2009). In selecting areas for release of imported elvers, area's suitability investigated in view of the prevalence of swimbladder nematodes should be preferred (Sjöberg *et al*., 2009).

4) The ocean migration

The objective of the project EELIAD is to answer what happens during the ocean phase, spawning migration to the Sargasso Sea, and larval route to Europe. The data generated provide a basis for the preservation of the European eel stock. The project uses a variety of new scientific techniques, such as tracking with satellite transmitters and data storage tags, genetic analysis and advanced mathematical models. Through collaboration with other research projects EELIAD will hopefully help to give an overall picture of the eel's life cycle.

Tagging experiments in EELIAD project began in autumn 2008. Preliminary results from satellite tags show that eels migrate at great depth (Aarestrup *et al*., 2009) and that the way out from Swedish waters appears to be north of the British Isles. A minor objective in EELIAD project is to study whether there is any behavioural difference between eels which immigrated naturally or were restocked. Part of the tagged eels was probably of restocked origin and was thus able to be traced out into the Atlantic. Sweden is actively involved in this project. A significant proportion of the eels tagged in 2008–2009 and 2010 have grown up as yellow eel in Sweden. Unique knowledge has already been produced concerning the migratory behaviour after the eel has left the Baltic Sea. The most remarkable result so far was an eel from Skåne that was shown to migrate all the way north of the Shetlands. Unfortunately, the origin of this eel was unknown.

References

Aarestrup, K., Okland, F., Hansen, M. M., Righton, D., Gargan, P., Castonguay, M., Bernatchez, L., Howey, P., Sparholt, H., Pedersen, M. I. and McKinley, R. S. 2009. Oceanic Spawning Migration of the European Eel (*Anguilla anguilla*). Science 325: 1660.

Allen, M., Rosell, R. and Evans, D. 2006. Predicting catches for the Lough Neagh (Northern Ireland) eel fishery based on stock inputs, effort and environmental variables. Fisheries Management and Ecology 13: 251–260.

Anonymous. 2008. Eel management plan of Estonia. 26 p.

Bark, A., Williams, B. and Knights, B. 2007. Current status and temporal trends in stocks of European eel in England and Wales. ICES Journal of Marine Science 64: 1368–1378.

Beaulaton, L. 2008. Systèmes de suivi des pêches fluvio-estuariennes pour la gestion des espèces : construction des indicateurs halieutiques et évaluation des impacts en Gironde. Doctorat de l'Institut National Polytechnique de Toulouse en Agrosystèmes, Ecosystèmes et Environnement, 348p.

Bíró, P. 1992. Die Geschichte des Aals (*Anguilla anguilla* L.) im Plattensee (Balaton). Österreichs Fischerei, Wissenschaft 45: 197–207.

Bíró, P. 1997. Temporal variation in Lake Balaton and its fish populations. Ecology of Freshwater Fish 6(4): 196–216.

Briand, C. 2009. Dynamique de population et de migration des civelles d'anguille européenne (*Anguilla anguilla*) en estuaire de Vilaine. Thèse de Doctorat Halieutique, Ensa-Rennes Organismes d'accueil : Institut d'Aménagement de la Vilaine ; Pôle halieutique Agrocampus Ouest N° ordre : 2-2009 N° Série : H-76, 208 p.

Briand, C., Castelnaud, G., Beaulaton, L., Casamajor, M.-N., Laffaille, P. Report on the eel stock and fishery in France. 2007. **In**: FAO European Inland Fisheries Advisory Commission; International Council for the Exploration of the Sea. Report of the 2008 session of the Joint EIFAC/ICES Working Group on Eels. Leuven, Belgium, 3–7 September 2008. EIFAC Occasional Paper. No. 43. ICES CM 2009/ACOM:15. Rome, FAO/Copenhagen, ICES. 2009. 192p. (Includes a CD-ROM).

Ciepielewski W. 1976. Rozmiary, plec i wiek wegorza (*Anguilla anguilla* L.) splywajacego z dwoch jezior mazurskich. Rocz. Nauk Rol. 97 (2): 7–24. (The size, sex and age of seaward migrating eel from two Masurian Lakes. **Ur**: Report of the Joint ICES/EIFAC Symposium on Eel Research and Management (Anguilla spp.), Helsinki, Finland, 9–11 June 1976. EIFAC Technical Paper No. 28).

CITES. 2008. Convention on international trade in endangered species of wild fauna and flora. Appendices I, II and III, valid from 1 July 2008. 47 p.

Clevestam, P., Wickström, H. 2008. Rädda ålen och ålfisket! – Ett nationellt bidrag till en europeisk förvaltningsplan.Vetenskaplig slutrapport från pilotprojekt till Fonden för fiskets utveckling (in Swedish) Dnr: 231-1156-04.

Dannewitz, J., Maes, G., Johansson, L., Wickström, H., Volckaert, F. and Järvi, T. 2005. Panmixia in the European eel: a matter of time... Proc. R. Soc. B 272: 1129–1137.

De Leo, G. A., Gatto M. 2001. A stochastic bioeconomic analysis of silver eel fisheries. Ecological Applications 11:281–294.

Durif, C., S. Dufour and P. Elie. 2005. The silvering process of *Anguilla anguilla*: a new classification from the yellow resident to the silver migrating stage. Journal of Fish Biology J. Fish Biol. 66(4): 1025–1043.

Durif C. M. F. and Skiftesvik A. B. 2007. Europeisk ål. In Kyst og havbruk 2007 (eds E. Dahl, P. K. Hansen, T. Haug & Ø. Karlsen), pp. 88–89. Fisken og havet).

E.G. 2007. Förordning (EG) nr 1100/2007 (EUT L 248, 22.9.2007).

FAO/ICES. 2006. Report of the 2006 session of the Joint EIFAC/ICES Working Group on Eels. Rome, 23–27 January 2006. EIFAC Occasional Paper. No. 38, ICES CM 2006/ACFM:16. Rome, FAO/Copenhagen, ICES. 2006. 352p.

FAO/ICES. 2009. Report of the 2009 session of the Joint EIFAC/ICES Working Group on Eels. Göteborg, Sweden, from 7 to 12 September 2009. EIFAC Occasional Paper. No. 45. ICES CM 2009/ACOM:15. Rome, FAO/Copenhagen, ICES. 2010. 540p. (Online).

Feunteun, E., Acou, A., Laffaille, P. and Legault, A. 2000. European eel (*Anguilla anguilla*): prediction of spawner escapement from continental population parameters. Canadian Journal of Fisheries and Aquatic Sciences 57: 1627–1635.

Hanson, M., L. Karlsson, and H. Westerberg. 1984. Magnetic material in European Eel (*Anguilla anguilla* L.). Comp. Biochem. Phys. A: Physiology 77:221–224.

Hanson, M. and H. Westerberg. 1987. Occurrence of magnetic material in teleosts. Comp. Biochem. Phys. A: Physiology 86:169–172.

ICES. 2009. Report of the Study Group on Anguillid Eels in Saline Waters (SGAESAW), 16–18 March 2009, Sackville, Canada; 3–5 September 2009, Gothenburg, Sweden. ICES CM/DFC:06. 183 p.

Järvalt, A. 2003. Võrtsjärve asustatud angerja (*Anguilla anguilla*) kasvu, leviku ulatuse ja tagasipüügi hindamine Peipsi vesikonnas. (Estimation of growth, distribution and recapture of eel (*Anguilla anguilla*) stocked into Lake Võrtsjärv in Lake Peipsi Basin) Keskkonnateenistuse poolt tellitud uurimisprojekti aruanne (Project Report) Tartu, 35 pp.

Järvalt, A. 2009. Stocking results in L. Võrtsjärv, Estonia. **Ur**: FAO/ICES 2009. Report of the 2009 session of the Joint EIFAC/ICES Working Group on Eels. Göteborg, Sweden, from 7 to 12 September 2009. EIFAC Occasional Paper. No. 45. ICES CM 2009/ACOM:15. Rome, FAO/Copenhagen, ICES. 2010. 540p. (Online).

Karlsson, L. 1985. Behavioural responses of European silver eel (*Anguilla anguilla*) to the geomagnetic field. Helgolander Meeresuntersuchungen 39:71–81.

Limburg, K., H. Wickström, H. Svedäng, M. Elfman and P. Kristiansson. 2003. Do Stocked Freshwater Eels Migrate? Evidence from the Baltic Suggests "Yes". American Fisheries Society Symposium 33:275–284.

Lin, Y., -J. L. Lozys, J., -C. Shiao, Y., Iizuka and W., -N. Tzeng. 2007. Growth differences between naturally recruited and stocked European eel *Anguilla anguilla* from different habitats in Lithuania. Journal of Fish Biology 71: 1773–1787.

McKeown, B.A. 1984. Fish Migration. Croom Helm Publishers, London. 300 pages.

Montén, E. 1985. Fish and turbines: fish injuries during passage through power station turbines. Vattenfall, Stockholm,Sweden. 111 p.

Moriarty, C. http://www3.interscience.wiley.com/journal/119384606/abstract - c1M. Bninska and M. Leopold. 1990. Eel, *Anguilla anguilla* L., stock and yield in Polish lakes. Aquaculture Research 21 (3): 347–356.

Nishi, T., Kawamura, G. and Matsumoto, K. 2004. Magnetic sense in the Japanese eel *Anguilla japonica* as determined by conditioning and electrocardiography. Journal of Experimental Biology 207: 2965–2970.

Nishi and T., Kawamura. 2005. *Anguilla japonica* is already magnetosensitive at the glass eel phase. Journal of Fish Biology 67(5): 1213–1224.

Nishi, T., Kawamura, G. and Sannomiya, S. 2005. Anosmic Japanese eels *Anguilla japonica* can no longer detect magnetic fields. Fisheries Science 71: 101–106.

Palm S, Dannewitz J, Prestegaard T, Wickström H. 2009. Panmixia in European eel revisited: no genetic difference between maturing adults from southern and northern Europe. Heredity 103: 82–89.

Palstra, A. P., Heppener, D. F. M., van Ginneken, V. J. T., Szekely, C. and van den Thillart, G. 2007. Swimming performance of silver eels is severely impaired by the swim-bladder parasite *Anguillicola crassus*. Journal of Experimental Marine Biology and Ecology 352: 244–256.

Pedersen, M., I. 2000. Long-term survival and growth of stocked eel, *Anguilla anguilla* (L.), in a small eutrophic lake. Dana 12: 71–76.

Pedersen, M. I. 2009. The effect of stocking the shallow Roskilde Fjord. **Ur**: ICES. 2009. Report of the Study Group on Anguillid Eels in Saline Waters (SGAESAW), 16–18 March 2009, Sackville, Canada; 3–5 September 2009, Gothenburg, Sweden. ICES CM/DFC:06. 183p.

Robak, S. 2005. Sex structure, recruitment rate, and size of maturing specimens of European eel (*Anguilla anguilla* (L.)) in a population from lakes in northeastern Poland. Arch. Pol. Fish. 13 (2): 181–192.

Rosell, R.S., Evans, D, and Allen, M. 2005. The Eel fishery in Lough Neagh, Northern Ireland – An example of sustainable management? Fisheries Management and Ecology 12: 377–385.

Rosell, R. S. 2009. From updated analysis of the Lough Neagh (Northern Ireland) input-output data to 2008. Presented to WGEEL 2009 by R Rosell. **Ur**: FAO European Inland Fisheries Advisory Commission; International Council for the Exploration of the Sea. Report of the 2009 session of the Joint EIFAC/ICES Working Group on Eels. Göteborg, Sweden, from 7 to 12 September 2009. EIFAC Occasional Paper. No. 45. ICES CM 2009/ACOM:15. Rome, FAO/Copenhagen, ICES. 2010. 540p. (Online).

Scaion, D., Belhomme, M. and Sébert, P. 2009. High hydrostatic pressure improves the swimming efficiency of European migrating silver eel. Respiratory Physiology & Neurobiology 165(1): 112–114.

Shiao J.,-C. L. Lozys, Y., Iizuka and W., -N. Tzeng. 2006. Migratory patterns and contribution of stocking to the population of European eel in Lithuanian waters as indicated by otolith Sr:Ca ratios. Journal of Fish Biology 69: 749–769.

Sjöberg, N. B. and Petersson, E. 2005. Tagging experiments with silver eel (*Anguilla anguilla*) in helping to understand the migration in the Baltic. Fiskeriverket Informerar [Finfo], 46 p.

Sjöberg, N. and Wickström, H. 2008. Utvärdering av ålutsättningar –Delrapport år 2008. PM till Naturvårdsverket-Havsmiljöenheten, 6p.

Sjöberg, N. B., Wickstrom, H. and Petersson, E. 2008. Bidrar den omflyttade ålen till lekbeståndet i Sargassohavet? - Blankålsmärkning kan ge svaret. Slutrapport från pilotprojekt till Fonden för fiskets utveckling (in Swedish) Dnr: 231-0721-05, 43p.

Sjöberg, N. B., Petersson, E., Wickström, H. and Hansson, S. 2009. Effects of the swimbladder parasite *Anguillicola crassus* on the migration of European silver eels *Anguilla anguilla* in the Baltic Sea. Journal of Fish Biology 74: 2158–2170.

Svedäng H., E. Neuman and H. Wickström. 1996. Maturation patterns in female European eel: Age and size at the silver eel stage. Journal of Fish Biology 48(3): 342–351.

Tesch, F. W., T. Wendt, and L. Karlsson. 1992. Influence of geomagnetism on the activity and orientation of eel, *Anguilla anguilla*, as evident from laboratory experiment. Aq. Ecol. Freshw. Fish 1:52–60.

Tzeng, W.-N. 2009. Migratory behaviour and habitat use of Japanese eel in the estuary as revealed by both conventional mark-recapture method and otolith elemental signature. **In**: ICES. 2009. Report of the Study Group on Anguillid Eels in Saline Waters (SGAESAW), 16–18 March 2009, Sackville, Canada; 3–5 September 2009, Gothenburg, Sweden. ICES CM/DFC:06. 183 p.

Verreault, G., Dumont, P., Dussureault, J.,Tardif, R. 2010. First record of migrating silver American eels (*Anguilla rostrata*) in the St Lawrence Estuary originating from a stocking program. (Submitted to CJFAS).

Westerberg, H. and M. L. Begout-Anras. 2000. Orientation of silver eel (*Anguilla anguilla*) in a disturbed geomagnetic field. **In**: *Advances in Fish Telemetry.* Proceedings of the Thirdy Conference on Fish Telemetry. Moore, A. and I. Russel, editors. Centre for Environment, Fisheries and Aquaculture Science. Lowestoft. pp. 149–158.

Westerberg, H., Lagenfelt, I. and Svedang, H. 2007. Silver eel migration behaviour in the Baltic. Ices Journal of Marine Science 64: 1457–1462.

Westin, L. 1990. Orientation mechanisms in migrating European silver eel (*Anguilla anguilla*): Temperature and olfaction. Marine Biology 106: 175–179.

Westin, L. 1998. The spawning migration of European silver eel (*Anguilla anguilla* L.) with particular reference to stocked eel in the Baltic. Fisheries Research 38: 257–270.

Westin, L. 2003. Migration failure in stocked eels *Anguilla anguilla*. Marine Ecology Progress Series 254: 307–311.

Wickström; H. 1986a. Sötvattenslaboratoriets ålundersökningar 1977–1985. Information från Sötvattenslaboratoriet, Drottningholm 13. 43p.

Wickström, H. 1986b. Growth of cultured eels stocked in two Swedish lakes. Vie et Milieu. 36(4):273–277.

Wickström, H. 2001. Stocking as a sustainable measure to enhance eel populations. Diss. Stockholm Univ., 2001. 39 p.

Wickström, H, Westin L, Clevestam P. 1996. The Biological and economical yield from a long-term stocking experiment. Ecology of Freshwater Fish 5: 140–147.

Wilkońska, H. and I. Psuty. 2008. Using a side-selective fyke net barrier to research fish assemblages in the transitional and transboundary waters of the Vistula Lagoon, southern Baltic. J. Appl. Ichthyol. 24: 650–657.

Annex 8: TranslocEel model presentation

P. Lambert, CEMAGREF, Bordeaux, France.

Let

- τ_o the duration of the oceanic journey
- τ_w, τ_n and τ_s the continental lifespan of the West, the North and the West compartments, with $\tau_{\max} = \max(\tau_w, \tau_n, \tau_s)$
- ΣM_w, ΣM_n and ΣM_s the lifespan natural mortality for the West, the North and the West compartments
- ΣA_w, ΣA_n and ΣA_s the lifespan anthropogenic mortalities for the West, the North and the West compartments
- ΣA_w^*, ΣA_n^* and ΣA_s^* the lifespan anthropogenic mortalities specific for the stocked eel in the West, the North and the West compartments
- $\Sigma \tilde{A}_w$, $\Sigma \tilde{A}_n$ and $\Sigma \tilde{A}_s$ the lifespan anthropogenic mortalities threshold in the West, in the North and West compartments that lead to stabilize the recruitment (avoid the crash of population)

- α_w, α_n, α_s the proportion of recruitment arriving in the West, the North and the West compartments ($\alpha_w + \alpha_n + \alpha_s = 1$)
- φ'_w, φ'_n, φ'_n the proportion of female for a cohort in the West, the North and the West compartments
- φ_w, φ_n, φ_n the proportion of female corrected by the specific fecundity and the capacity to reach the Sargasso. This factor is standardized relative to the value in the West compartment ($0 \le \varphi_i \le 1$).

- D the exponential decrease of recruitment observed in Europe since 1980
- b the net individual reproductive output (number of glass eel arriving per female silver eel leaving the continent) or the slope of the first part of a hockey stick stock–recruitment relationship

- $R(t)$ the total recruitment the year t, $R_w(t)$, $R_n(t)$ and $R_s(t)$ recruitment arriving in the West, the North and the West compartments the year t

  $R_i(t) = \alpha_i R(t)$

  $R_i(t) = \alpha_i R(t_0) e^{-D(t-t_0)}$

- $S(t)$ the female escapement the year t, $S_w(t)$, $S_n(t)$ and $S_s(t)$ female silver eel escaping from the West, the North and the West compartments the year t

  $S(t) = \sum_{i \in \{w,n,s\}} S_i(t)$

Population dynamics

Baseline of the model "TranslocEel"

TranslocEel is a generalization of Aström and Dekker model {, 2007 #3834} {and SED model, \Lambert, 2008 #4251} to several spatial compartments.

It is based on the combination of mortality lines (one in each compartment) and a single stock–recruitment relationship.

The mortality lines are based on off-line estimations of mortality in historical situation.

Except for the donor compartment for stocking (where glass eel mortality is also needed) the lifespan mortality is used. In recipient compartment, the mortality for stocked fish is adjusted relative to mortality of wild eel.

Then relation stock–recruitment relationship is fitted to mimic the observed European recruitment trend and by considering the regional characteristics of each compartment.

The main assumptions of TranslocEel model are

- Male number is not a limiting factor for the population dynamics.
- There are no density regulations
    - in mortality
    - in sex determinism
- The proportion of recruitment arriving in each compartment is constant over time.

The recruitment trend

TranslocEel model is based on the exponential trend of recruitment observed in Europe, the same for all the compartments.

$$R(t) = R(t_0) e^{-D(t-t_0)} \quad (1)$$

Mortality (replacement line)

The female silver eel silver eel $S_i(t)$ leaving the compartment i depends on the recruitment $R_i(t-\tau_i)$ arriving in this compartment lagged by the lifespan τ_i, on the corrected proportion of female (standardized to West specific fecundity) in that compartment φ_i and the natural (M_i) and anthropogenic (A_i) lifespan mortality in the compartment. The total female escapement is then:

$$S(t) = \sum_{i \in \{w,n,s\}} R_i(t-\tau_i)\varphi_i e^{-(\Sigma M_i + \Sigma A_i)}$$

Considering that the recruitment in one compartment is a proportion of the total recruitment $R_i(t) = \alpha_i R(t)$, we have

$$S(t) = \sum_{i \in \{w,n,s\}} \alpha_i \varphi_i R(t-\tau_i) e^{-(\Sigma M_i + \Sigma A_i)}$$

Recruitments can be calculated with a common time reference $t-\tau_{\max}$ according to equation (1):

$$R(t-\tau_i) = R(t-\tau_{\max})e^{-D((t-\tau_i)-(t-\tau_{\max}))}$$

or

$$R(t-\tau_i) = R(t-\tau_{\max})e^{D(\tau_i-\tau_{\max})}$$

And finally

$$S(t) = R(t-\tau_{\max}) \sum_{i\in\{w,n,s\}} \alpha_i \varphi_i e^{D(\tau_i-\tau_{\max})} e^{-(\Sigma M_i + \Sigma A_i)} \qquad (2)$$

Reproduction (stock–recruitment relationship)

We used a "hockey stick" stock–recruitment relationship:

$$\begin{aligned} R(t+\tau_o) &= bS(t) \text{ for } S(t) < S_{off} \\ &= bS_{off} \quad \text{for } S(t) \geq S_{off} \end{aligned}$$

Where $S(t)$ is the female escapement the year t, $R(t+\tau_o)$ is the associated recruitment after the duration of the oceanic journey τ_o, b is the net individual reproductive output (number of glass eel arriving per female silver eel leaving the continent) and S_{off} is the female silver eel escapement above which recruitment is constant.

Net individual reproductive output determination

$$S(t) = \frac{R(t+\tau_o)}{b}$$

With equation (1) and by reference to $t-\tau_{\max}$

$$S(t) = \frac{R(t-\tau_{\max})e^{-D(\tau_o+\tau_{\max})}}{b}$$

By combining with equation (2) we obtain the equality:

$$\frac{R(t-\tau_{\max})e^{-D(\tau_o+\tau_{\max})}}{b} = R(t-\tau_{\max}) \sum_{i\in\{w,n,s\}} \alpha_i \varphi_i e^{D(\tau_i-\tau_{\max})} e^{-(\Sigma M_i + \Sigma A_i)}$$

That leads to calculate the net individual reproductive output

$$b = \frac{e^{-D(\tau_o+\tau_{\max})}}{\sum_{i\in\{w,n,s\}} \alpha_i \varphi_i e^{D(\tau_i-\tau_{\max})} e^{-(\Sigma M_i + \Sigma A_i)}}$$

Or simply

$$b = \frac{1}{\sum_{i\in\{w,n,s\}} \alpha_i \varphi_i e^{D(\tau_o+\tau_i)-(\Sigma M_i + \Sigma A_i)}} \qquad (3)$$

Calculation of the female silver eel escapement threshold

The threshold of silver eel escapement is calculated as the value corresponding to the historical recruitment R_{hist}:

$$R_{hist} = bS_{off}$$

Then we obtain

$$S_{off} = R_{hist} \sum_{i \in \{w,n,s\}} \alpha_i \varphi_i e^{D(\tau_o + \tau_i) - (\Sigma M_i + \Sigma A_i)}$$

Special case considering a single compartment

In case of dynamics based only on the single compartment i (one of w, n or s) we have

$$b_i = \frac{e^{-D(\tau_o + \tau_i)}}{\varphi_i e^{-(\Sigma M_i + \Sigma A_i)}}$$

or

$$b_i = e^{\Sigma M_i + \Sigma A_i - D(\tau_o + \tau_i)} / \varphi_i \qquad (4)$$

which corresponds to the Aström and Dekker {, 2007 #3834} formula with $\varphi_i = 1$.

We can easily demonstrate

$$b = \frac{1}{\sum_{i \in \{w,n,s\}} \frac{\alpha_i}{b_i}}$$

For the threshold we have

$$S_{off,i} = R_{hist,i} \varphi_i e^{D(\tau_o + \tau_i) - \Sigma M_i - \Sigma A_i}$$

Since $R_{hist,i} = \alpha_i R_{hist}$ we have

$$S_{off} = \sum_{i \in \{w,n,s\}} S_{off,i}$$

Source/sink

The compartment is a source if the number of females that are escaping to spawn will result in a glass eel return that produces the same number of female silver eel:

$$\varphi_i \, \mathrm{e}^{-\Sigma M_i - \Sigma A_i} \geq 1/b$$

Otherwise it is a sink:

$$\varphi_i \, \mathrm{e}^{-\Sigma M_i - \Sigma A_i} < 1/b$$

Sex ratio in silver eel escapement in historical condition

$$S_{f,i}(t) = R_i \left(t - \tau_{f,i}\right) \varphi_i e^{-(M_i + A_i)\tau_{f,i}}$$

$$S_{m,i}\left(t\right)=R_i\left(t-\tau_{m,i}\right)\left(1-\varphi_i\right)e^{-\left(M_i+A_i\right)\tau_{m,i}}$$

We have

$$R_i\left(t-\tau_{m,i}\right)=R_i\left(t-\tau_{f,i}\right)e^{-D\left(\tau_{f,i}-\tau_{m,i}\right)}$$

then

$$S_{m,i}\left(t\right)=R_i\left(t-\tau_{f,i}\right)\left(1-\varphi_i\right)e^{-D\left(\tau_{f,i}-\tau_{m,i}\right)-\left(M_i+A_i\right)\tau_{m,i}}$$

The percentage of female in silver eel escapement is

$$f_{\%i}=\frac{S_{f,i}\left(t\right)}{S_{f,i}\left(t\right)+S_{m,i}\left(t\right)}$$

$$f_{\%i}=\frac{R_i\left(t-\tau_{f,i}\right)\varphi_i e^{-\left(M_i+A_i\right)\tau_{f,i}}}{R_i\left(t-\tau_{f,i}\right)\varphi_i e^{-\left(M_i+A_i\right)\tau_{f,i}}+R_i\left(t-\tau_{f,i}\right)\left(1-\varphi_i\right)e^{-D\left(\tau_{f,i}-\tau_{m,i}\right)-\left(M_i+A_i\right)\tau_{m,i}}}$$

$$f_{\%i}=\frac{\varphi_i}{\varphi_i e^{-\left(M_i+A_i\right)\tau_{f,i}}+\left(1-\varphi_i\right)e^{-D\left(\tau_{f,i}-\tau_{m,i}\right)-\left(M_i+A_i\right)\tau_{m,i}}}$$

$$f_{\%i}=\frac{\varphi_i}{\varphi_i+\left(1-\varphi_i\right)e^{-D\left(\tau_{f,i}-\tau_{m,i}\right)-\left(M_i+A_i\right)\tau_{m,i}+\left(M_i+A_i\right)\tau_{f,i}}}$$

$$f_{\%i}=\frac{\varphi_i}{\varphi_i+\left(1-\varphi_i\right)e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}}$$

$$f_{\%i}\varphi_i+f_{\%i}e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}-f_{\%i}\varphi_i e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}=\varphi_i$$

$$\varphi_i\left(1-f_{\%i}+f_{\%i}e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}\right)=f_{\%i}e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}$$

$$\varphi_i=\frac{f_{\%i}e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}}{\left(1-f_{\%i}+f_{\%i}e^{\left(\Sigma M_i+\Sigma A_i-D\tau_{f,i}\right)\left(1-\tau_{m,i}/\tau_{f,i}\right)}\right)} \quad (5)$$

Condition to avoid population crash

At global scale

With the replacement line

$$S\left(t\right)=f_{R\to S}\left(R\left(t-\tau_w\right),R\left(t-\tau_s\right),R\left(t-\tau_n\right)\right)=R\left(t-\tau_{\max}\right)\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{D\left(\tau_i-\tau_{\max}\right)}e^{-\Sigma M_i-\Sigma A_i}$$

and the stock–recruitment relationship

$$R\left(t+\tau_o\right)=f_{S\to R}\left(S\left(t\right)\right)=bS\left(t\right),$$

the condition to avoid the crash of the population is that the anthropogenic mortality $\tilde{A}_i$ leads to a recruitment produced after a life cycle greater or equal to the initial recruitment:

$$f_{S\to R}\left(f_{R\to S}\left(RR(t-\tau_w),R(t-\tau_s),R(t-\tau_n)\right)\right)\geq R(t+\tau_o)$$

$$bR(t-\tau_{\max})\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{D(\tau_i-\tau_{\max})}e^{-\Sigma M_i-\Sigma A_i}>R(t+\tau_o)$$

At the minimum the recruitment should be constant i.e. $D=0$ and $R(t-\tau_{\max})=R(t+\tau_o)$

Therefore the condition to avoid crash is

$$\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\frac{1}{b} \tag{6}$$

With equation (3) we obtain

$$\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{D(\tau_o+\tau_i)-\Sigma M_i-\Sigma A_i}$$

$$\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>e^{D\tau_o}\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{(D\tau_i-\Sigma M_i-\Sigma A_i)} \tag{7}$$

At compartment scale

Hypothesis 1: Population working only on a single compartment

We consider in that case that spawners escaping from the compartment i contribute solely to the population functioning.

Equation (6) becomes with equation (4)

$$\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\varphi_i e^{-\Sigma M_i-\Sigma A_i+D(\tau_o+\tau_i)}$$

Or simply

$$\Sigma\tilde{A}_i<\Sigma A_i-D(\tau_o+\tau_i) \tag{8}$$

which corresponds to the formula of cumulative mortality threshold used in ICES {, 2009 #5086}.

We can verify that the respect the local condition (8) in each compartment is a sufficient (but not a necessary) condition to avoid population crash.

$$\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\varphi_i e^{-\Sigma M_i-\Sigma A_i+D(\tau_o+\tau_i)} \qquad \forall i\in\{w,n,s\}$$

$$\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma A_i+D(\tau_o+\tau_i)} \qquad \forall i\in\{w,n,s\}$$

$$\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{-\Sigma M_i-\Sigma\tilde{A}_i}>\sum_{i\in\{w,n,s\}}\alpha_i\varphi_i e^{D\tau_o}e^{-\Sigma M_i-\Sigma A_i+D\tau_i}$$

$$\sum_{i\in\{w,n,s\}}\alpha_i e^{-(M+\tilde{A}_i)\tau_i}>e^{D\tau_o}\sum_{i\in\{w,n,s\}}\alpha_i e^{-\Sigma M_i-\Sigma A+D\tau_{ii}}$$

Hypothesis 2: All compartments are working as source for the population

By considering that each compartment should respect the global stock–recruitment, i.e. by using directly b instead b_i, the local condition becomes

$$\varphi_i e^{-\Sigma M_i - \Sigma \tilde{A}_i} > \frac{1}{b}$$
$$\Sigma \tilde{A}_i < \log\left(b\varphi_i\right) - \Sigma M_i \qquad (9)$$

We can verify that the respect of the source condition (9) in each compartment is also a sufficient (but not a necessary) condition to avoid population crash (7).

$$\varphi_i e^{-\Sigma M_i - \Sigma \tilde{A}_i} > \frac{1}{b}$$
$$\alpha_i \varphi_i e^{-\Sigma M_i - \Sigma \tilde{A}_i} > \frac{\alpha_i}{b}$$

$$\sum_{i \in \{w,n,s\}} \alpha_i \varphi_i e^{-\Sigma M_i - \Sigma \tilde{A}_i} > \frac{\sum_{i \in \{w,n,s\}} \alpha_i}{b}$$

$$\sum_{i \in \{w,n,s\}} \alpha_i \varphi_i e^{-\Sigma M_i - \Sigma \tilde{A}_i} > \frac{1}{b}$$

Stocking

Effect of stocking

Let $R_r^*\left(t\right)$ the number of stocked fish the year t in the receptor compartment r ($r \in \{w,n,s\}$)

The spawning stock produced after the first generation in the receptor compartment r is

$$S_r^*\left(t+\tau_r\right) = R_r^*\left(t\right)\varphi_r e^{-\Sigma M_r - \Sigma A_r^*}$$

This produces recruitment in each compartment i equal to

$$R_i^*\left(t+\tau_r+\tau_o\right) = b\alpha_i S_r^*\left(t+\tau_r\right)$$
$$R_i^*\left(t+\tau_r+\tau_o\right) = b\alpha_i R_r^*\left(t\right)\varphi_r e^{-\Sigma M_r - \Sigma A_r^*}$$

The survival stock is then

$$S_i^*\left(t+\tau_r+\tau_o+\tau_i\right) = b\alpha_i R_r^*\left(t\right)\varphi_r\varphi_i e^{-\left(\Sigma M_r + \Sigma A_r^*\right) - \left(\Sigma M_i + \Sigma A_i\right)}$$

This produce recruitment in each compartment j

$$R_i^*\left(t+\tau_r+2\tau_o+\tau_i\right) = b^2\alpha_j\alpha_i R_r^*\left(t\right)\varphi_r\varphi_i e^{-\left(\Sigma M_r + \Sigma A_r^*\right) - \left(\Sigma M_i + \Sigma A_i\right)}$$

And so on

Reduction of mortality due to no stocking

We consider that 60% of capture of glass eel are affected to stock

$$C_{glass,j}(t) = R(t)\alpha_j \frac{F_{glass,j}}{F_{glass,j} + M_{glass,j}} e^{-\left(F_{glass,j}+M_{glass,j}\right)\tau_{glass}}$$

$$0.6C_{glass,j}(t) = R(t)\alpha_j \frac{F'_{glass,j}}{F'_{glass,j} + M_{glass,j}} e^{-\left(F'_{glass,j}+M_{glass,j}\right)\tau_{glass}}$$

Where $F_{glass,j}$ and $F'_{glass,j}$ are glass eel fishery mortality with and without stocking.

$$\frac{F_{glass,j}}{F_{glass,j} + M_{glass,j}} e^{-\left(F_{glass,j}+M_{glass,j}\right)\tau_{glass}} = \frac{1}{0.6}\frac{F'_{glass,j}}{F'_{glass,j} + M_{glass,j}} e^{-\left(F'_{glass,j}+M_{glass,j}\right)\tau_{glass}}$$

$$\frac{F_{glass,j}}{F_{glass,j} + M_{glass,j}} e^{-F_{glass,j}\tau_{glass}} = \frac{1}{0.6}\frac{F'_{glass,j}}{F'_{glass,j} + M_{glass,j}} e^{-F'_{glass,j}\tau_{glass}}$$

Let $F'_{glass,j} = \rho F_{glass,j}$

$$\frac{F_{glass,j}}{F_{glass,j} + M_{glass,j}} e^{-F_{glass,j}\tau_{glass}} = \frac{1}{0.6}\frac{\rho F_{glass,j}}{\rho F_{glass,j} + M_{glass,j}} e^{-\rho F_{glass,j}\tau_{glass}}$$

$$\frac{1}{F_{glass,j} + M_{glass,j}} = \frac{1}{0.6}\frac{\rho}{\rho F_{glass,j} + M_{glass,j}} e^{-(\rho-1)F_{glass,j}\tau_{glass}}$$

$$\rho\left(F_{glass,j} + M_{glass,j}\right) = 0.6\left(\rho F_{glass,j} + M_{glass,j}\right)e^{(\rho-1)F_{glass,j}\tau_{glass}}$$

Annex 9: Technical minutes from the Eel Review Group

- RGEEL
- by correspondence 18–20 October 2011
- Participants: Erkki Ikonen (Chair), Martin Castonguay and Henrik Svedäng (Reviewers), Reinhold Hanel (WG participant) and Russell Poole (WG Chair), Arjan Heinen (Observer) and Henrik Sparholt and Michala Ovens (ICES Secretariat).
- Working Group: WGEEL

General

The RG acknowledges the intense effort expended by the working group to produce the report.

The Review Group considered the following stocks:

- European eel (*Anguilla anguilla* L.).

And the following special requests:

a) Assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans; examine criteria for defining a recovery;

b) Develop and test methods to post-evaluate effects of management actions at the stock-wide level (in conjunction with SGIPEE), including quality assurance checking of Eel Management Unit biomass estimates;

c) Develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures; test data scenarios at the local level;

d) Provide practical advice on the establishment of international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and review data quality issues and develop recommendations on their inclusion, including the impact of the implementation of the eel recovery plan on time-series data and on stock assessment methods;

e) Review and develop approaches to quantifying the effects of eel quality on stock dynamics and integrating these into stock assessments; develop references points for evaluating impacts on eel;

f) Respond to specific requests in support of the eel stock recovery Regulation, as necessary; and

g) Report on improvements to the scientific basis for advice on the management of European and American eel;

h) Identify elements of the EGs workthat may help determine status for the 11 Descriptors set out in the Commission Decision (available at http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2010:232:0014:0024:EN:PDF;

i) Provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status;

j) Take note of and comment on the Report of the Workshop on the Science for area-based management: Coastal and Marine Spatial Planning in Practice (WKCMSP) http://www.ices.dk/reports/SSGHIE/2011/WKCMSP11.pdf;

k) provide information that could be used in setting pressure indicators that would complement biodiversity indicators currently being developed by the Strategic Initiative on Biodiversity Advice and Science (SIBAS). Particular consideration should be given to assessing the impacts of very large renewable energy plans with a view to identifying/predicting potentially catastrophic outcomes;

l) identify spatially resolved data, for e.g. spawning grounds, fishery activity, habitats, etc.

Additions to the ToR

1) Review the glass eel catches for the past two years,

assess quantities	caught in the commercial fishery
	exported to Asia
	used in stocking
	used in aquaculture for consumption
	consumed direct
	mortalities

compare with the EMP commitments to stocking;

2) Review the latest information on stocking, including previous reviews by WGEEL, new scientific reports/publications on the uses for stocking, the success of stocking as a method of fisheries support and/or as a means of increasing silver eel escapement and spawner biomass.

Reviewer 1 Martin Castonguay

I find that the critical state of the resource does not come out as clearly as it should in the advice. Only in the last paragraph, in the section "Comparison with previous assessment and advice", does the reader clearly find out about the critical status of the resource and the urgent need for remedial actions. I would therefore recommend stating this clearly in the first paragraph.

I fully agree that stocking is at best a controversial measure. Stocking may possibly be a good measure to enhance yellow eel fisheries in the short term but it remains to be seen if it results in enhanced escapement of good-quality spawners to the Sargasso Sea. Although the advice is quite clear on this, I was surprised to see that the focus of the e-mail discussions this week were almost exclusively on this issue.

In my opinion, the most important issue may be that the critical state of the resource requires first differentiating between management measures that will generate short-term benefits vs. longer term ones, and second prioritizing measures that will generate short-term benefits. It is not good enough to state that "all measures detrimental to eel production and escapement should be reduced to as close to zero as possible" while you know very well that for some of these detrimental effects, there is no short-term (or even long-term) fix. Reducing pollution or restoring connectivity (removing hydropower dams) is no easy task. What is the probability that such a management plan would succeed, even in the long term? In fact the only parameter you can manipulate in the short term is fisheries, and especially silver eel commercial fisheries. For example, if you were able to cut down silver eel fishing effort by half in a given year, or even better stop all silver eel fisheries over the entire species' range, you would have a quasi immediate benefit on the population (species) which would be detected as soon as two years later, when glass eels produced from this enhanced escapement will return to the continent. In fact, I imagine that the target reference point based on a 40% escapement level of pristine biomass will essentially mean shutting down most if not all fisheries, be they silver or yellow eel fisheries.

I am sure you are all aware of these issues but my point is that the advice should identify and give priority to measures that will result in short-term benefits. This is important because it will show that the eel collapse is not an intractable problem that might provide benefits only decades from now. There are possible solutions that will have medium-term benefits to those primarily impacted (fishers). In other words there is hope, it's a case of short-term pain for medium or long-term gain...

Reviewer 2 Henrik Svedäng

As this is a very extensive report, I have focused on parts of the report that might need more attention or have to be discussed.

2. Data and trends

Temporal trends in recruitment (Figure 2.6): The figure depicts a rather clear-cut decline in yellow eel recruitment over the last 60 years, and this interesting information should be highlighted. Several conclusions might be drawn: There has been an on-going reduction in recruitment over a considerable period of time, for instance, over several climatic oscillation periods.

Simulations of management actions on recruitment (p. 35): Why has the exponential increase rate been set so low (0.05)? Given that other increase rates had been considered, the probability of detecting an enhanced recruitment would have been rather different.

For the time being, the message emanating out from these simulations is that we shouldn't expect any increase in recruitment in many years to come. However, this is something that could be questioned: If the measures of the national EMPs indeed had been implemented, would it not have been reasonable to expect much higher an increase rate?

It is hence recommended that before making any further simulations, the anticipated increase rate should be estimated with regard to what can be expected from the stipulated measures in the national EMPs. In this year's report, it could be added that the arbitrarily chosen increase rate might have been set too low, and that next year's assessment has to be considered in relation to what can be expected from the proclaimed measures vowed in the national EMPs.

Data quality (such as landings in Point 2.4) and assessment

The eel stock status is dire. We can be certain that recruitment has fallen drastically over a long period of time. It should however also be strongly emphasised that our knowledge of the stock is very poor in quantitative terms. Landing data are unreliable. There are no estimates of mortalities made on a regular basis. SSB is unknown. IUU is said to be widespread. The assessment of stock status is thus almost based entirely on recruitment indices.

The precautionary approach in such a case is quite straightforward: the advice should inform the managers that there is less scope for exploitation than it would have been if the information had been in a better shape.

Trends in stocking

An evaluation of the outcome of the stocking with regard to landings is missing. It should therefore be included in the next year's ToR that the WG has to produce a time-series on the proportions of the silver eel landings that are based on stocked eels.

Objectives, targets and reference values (pp. 54–63)

The purpose of this section is problematic. Are really mortality reference points required because "B_{lim} is unachievable, at least for the coming generation" as the argument goes in the report (p. 56)? The presented exercise in maths gives that the sum of all anthropogenic impacts (A) should not exceed a fixed value of 0.92 (i.e. exp(-0.92)=0.4), if the reference point on the number of escaped silver eels is set to 40**%.** According to this newly invented reference system, the exploitation rate might be increased in areas where the current biomass/number of silver eels is below B_{lim}. As stated in **Section 3.6 Recommended reference values**: A current biomass of 1% of B_0 gives a certain opportunity of exploitation/scope for anthropogenic mortality A=0.023 (10% of B_o gives A=0.23, 20% of B_0 gives A=0.46, etc.).**This reference limit on mortality thus sets a scope for exploitation.**

As it is very difficult to estimate B_0, pristine biomass, and we cannot be assured that it will be accurately done everywhere, limits of exploitation should be set as close to zero as possibly also on the scale of single EMPs.

This reference exercise points at the weakness of the EU regulation in having *local* limit reference points in biomass (or number as used in the report). It's an open question whether it is really sensible to argue for a summed human exploitation rate up to 60% everywhere, at the same time as the eel stock as a whole is in such a bad shape? In other words, this procedure of several Blims for one stock is not logical.

This kind of management plan is rather unique for ICES stocks, that ONE stock is divided into different management subunits, where there is possible to fish rather a lot in some places and nothing at all in others. Given the serious situation of European eel, I think we should not argue for any exploitation for the time being. Depensation is global and not a local phenomenon for the stock, of course. In other words, this is an exceptional situation and the question of reference points should not be addressed until we can identify a clear and steady increase in the stock. Until then the stock is well below Blim everywhere.

4. Assessment of quality of the stocks (pp. 64–)

This section identifies a number of factors that are likely to reduce the quality of the affected spawners. This reduction in "spawning quality" can be regarded as a mortality factor, thus reducing the scope for exploitation even further. This argument has however not been put forward in the report.

Glass eel stocking

6.2.2.5 Migration (pp. 132–134)

Stocking is not an option until it can be proved that the navigational abilities of translocated eels are not seriously impaired (as the majority of studies indicate). Stocking or rather translocation of eels has been put forward as a mean to improve the status of the eel stock. This kind of conservation measure relies on two assumption: a) there are local eel surpluses, b) the translocated eel is able to navigate from its new location to the Sargasso Sea. By focusing on the second issue, one could say it is question of spawner quality: what will the contribution to spawning in the Sargasso Sea of the translocated eel in comparison with naturally recruited eels in the same area. It is reasonable to believe that the orientation will be inflicted upon to some extent, so the spawning quality of the translocated eel will always be less than that of the naturally recruited eel.

This reduction in spawning quality will depend on the actual nature of translocation. Moving glass eels from a French estuary up in the very same river system might not inflict so much on the navigational ability of the eels. Moving glass eels from, for instance, the Bristol channel to Northern Ireland will probably have a larger impact on the navigational abilities but still, it seems not completely unreasonable that some eels will find their way to the Sargasso Sea, if they are able to enter the sea eventually. If the French glass eels however are removed to a lake on an island in the Baltic Sea, they seem to have lost their way completely as the well-known studies of late Westin have shown. Similar results have been obtained from another lake entering into the Baltic Sea, Lake Mälaren. A recent tagging study in Lake Mälaren (see Annex 5) showed a very low escapement rate, most of the silver eels were seemingly unable to find their out from the lake, thus lowering the spawner quality value dramatically.

In spite of a review in Annex 5, the report limits the discussion on this important issue to two unfinished and unpublished studies on stocked and natural recruited eel. The preliminary results from these studies are not clear-cut enough to render any special attention: (a) stocked and natural recruited eels swam at the same speed along a stretch of a Swedish fjord, (b) two eels swam along the Norwegian Trench and showed a similar diurnal vertical behaviour. These findings are however used in the report as a striking fact in favour of stocking, as the overall conclusion was stated as: "this experiment shows no evidence for a difference in migration behaviour between stocked and naturally recruited eels". This statement is iterated in both the abstract and summary, and on p. 162 without any modification or discussion.

This statement should be deleted as it is a) neglecting other studies that have shown large differences in navigational abilities between stocked and naturally recruited eels, b) the referred studies are not published and gives no support in any direction as the present observations are not conclusive as they do not address the question to what extent the stocked eels are likely to contribute to SSB in the Sargasso Sea.

6.2.5 Risk assessment in stocking and precautionary approaches (pp. 135–137)

The WG has difficulties in defining precautionary approach to stocking practices and should be helped in doing so.

Determining net benefit (p. 152)

I oppose one of the main assumptions of TranslocEel model: that the proportion of males is not a limiting factor for the population dynamics. Males could be just as important as females for sexual reproduction. We know very little about sexual behaviour of eels, and the lack of males might be just as crucial as having a high population fecundity. As the sex ratio seems to be influenced by density-dependent factors, reductions of abundance in so-called surplus areas by glass eels fishing represents an obvious risk factor that hitherto has been ignored.

Conclusions

The stock status of the European eel is very grave as it can be inferred from various recruitment indices. The assessment is however weak, mostly due to poor data and lack of basic biological knowledge of the European eel. The different EMPs have not been evaluated. The report is very focused on various measures in order to improve the situation. Some of these methods such as translocation and stocking of eels cannot be entirely evaluated due to the lack of scientific data. However, existing peer-reviewed studies rather clearly point at a reduced ability of translocated eels to find their way back to the Sargasso Sea.

Annex 10: Recommendations

From Chapter 2: Data and Trends

Recommendation	For follow up by:
1. From 2008 four glass eel recruitment-series have been stopped in France. This adds up to four series lost elsewhere in Europe. This is only the consequence of changes brought in the catch report compilation. The only data currently available is the EMU and this prevents from doing any analysis at the estuarine level. The recommandation is for the previous details to be reported again in the data collection procedures.	ICES ? DCF, France.
2. A small number (three) of recruitment-series are available in the mediterranean area. Change this.	GFCM DCF
3. Catch and effort data be collected and made available to the working group	DCF
4. The 2001 meeting of WGEEL (ICES 2002) recommended the formation of an international commission that could act as a clearing house for handling and coordinating data collection & storage, stock assessment, management and research. Noting the urgent need to plan and coordinate the data collection and tool development for the 2012 post-evaluation; this recommendation is reiterated. In particular, it is recommended to organize a (series of) workshop(s) in relation to local eel stock monitoring, with a focus on standardization and coordination, preparing for the 2012 post-evaluation, setting the scene for the 2013 international stock. National surveys of eel stocks should now be included in the DCF under the following headings: Recruitment Surveys (Time-Series), internationaly coordinated; Silver Eel Escapement Indices, including biomass estimates; Yellow Eel Stock Surveys, including collection of biological and eel quality data.	ICES, EU, PGCCDBS, PGMED, DCF
5. National data on trend in silver eel and yellow abundance be made available to the working for an analysis next year	WGEEL 2012
6. It is unlikely that we detect a change brought by management actions to the recruitment level and silver eel escapement in the short term. It is of utmost importance to work on anthropogenic mortalities and interim targets.	ICES, EU
7. Establish a project on the Eel Fisheries Resource - a social and economic perspective.	EU, EIFAAC

From Chapter 3: Objectives, targets and reference values

Recommendation	For follow up by:
The EU Eel Regulation set a limit, corresponding to B_{lim} = 40% of B_0. According to the EU Eel Regulation, quantification and implementation of these reference points is up to EU Member States. Because uncertainties may vary from Member State to Member State, no universal values for the precautionary reference points can be provided. Hence, it is recommended that ICES abstains from advising precautious reference points, cautioning for the required extra margin on all reference points instead.	ACOM
As an initial option, it is recommended to set $B_{MSY\text{-}trigger}$ at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ correspondingly.	ACOM
The biomass reference point of B_{lim} = 40% of B_0 corresponds to a lifetime mortality limit of ΣA_{lim} = 0.92, unless strong density-dependence applies. In the latter case, a more complex assessment will be required, and a limit of $\%SPR_{lim}$ = 40% can be applied	ACOM

From Chapter 4: Quantitative assessment of the status of local eel populations

Recommendation	For follow up by:
Lifetime anthropogenic mortality	
Express anthropogenic mortality events in terms of numbers or % eels, and size-based selectivities	Member states
Collect data on your eel numbers, densities and length distributions making use of WFD	Member states
Analyse the fisheries data collected for the Data Collection Framework (DCF) to estimate fishing-based mortality	Member states
Develop the requirements for eel in the DCF and WFD to reflect these data requirements to support the estimation of anthropogenic mortalities and their summation	Member states
Sex Ratios	
collect sex-ratio data of young yellow (<35 cm TL) eel at a fixed location(s) in the lower reaches of a river; provide an estimate of age and density	Member states
Hydropower, Pumping stations, Water intake and other barriers	
conduct an inventory of hydropower, pumping station, water inlet location and characteristics	Member States
undertake studies to quantify the effect of pumping stations as migration barriers for (silver) eel migration undertake; studies to quantify the impact of water intake on eel	Member States
conduct inventory of temporary, permanent, natural and artificial obstacles for eel migration along with estimation of habitat loss for eel above these barriers	Member states
Predation	
Estimate eel predation by cormorants and put in wider ecological context	EIFAAC WG on cormorants

From Chapter 5: Assessment of the Quality of Eel Stocks

Recommendation	For follow up by:
1. In their annual country reports, member countries include data on the occurrence of sudden eel kills due to pollution or disease outbreaks. Information such as water body, water type, location, water surface, year, date, cause of death, and the estimated quantities of dead eels (and other fish) involved should be included	EU Countries, WGEEL Participants
2. In their annual country reports, member countries include a list of areas or water bodies where fisheries restrictions have been issued as a result of contaminant levels measured in eel (or other fish) exceeding human consumption safety limits	EU Countries, WGEEL Participants
3. Data of MS about contamination of eels raised for food regulatory reasons or for WFD should generally be made accessible for WGEEL (i.e. transfer of data to EEQD)	EU Countries, WGEEL Participants
4. Although the impact of contaminants and diseases on effective spawner escapment still remains unknown, regional eel management should generally refer to eel quality aspects like contamination and diseases, i.e include observed or measured impairments of eels in reports	EU Member States
5. Eel Quality index remains to be further developed for a better assessment of the overall status of eel quality over river basins. Eel Quality Assessments are to be linked to the quantitative assessments of effective spawner escapement in the EMUs	WGEEL, Belpaire
6. Research resulting in a better understanding of the eel's sensitivity towards parasites, diseases, and contaminants under field conditions, with respect to reproduction, should be supported. When the effects of stressfactors can be quantified a better, clear decision about the importance of "eel-quality" in eel management can be made	EU Funding
7. the direct impact of fisheries closure for human health's sake on stock restoration should be evaluated, i.e. what is the quantity and quality of eels affected by these measures, and to what extent do they contribute to the stock, considering their low quality?	WGEEL

From Chapter 6: Glass Eel Resources and Stocking

Recommendation	For follow up by:
1. WGEEL recommends that, in the absence of an agreed standard definition, risk assessment methods are applied to stocking decisions as robustly as local data permits	Member States
2. It is recommended that all countries put in place a traceability system to meet the requirements of Article 12 of the Regulation. Essential elements allowing traceability and permitting cross-checking between countries can be identified: for each batch of glass eel exported, the date, the amount, the price, the destination EMU and fate (stocking/aquaculture/consumption), and the EMU of origin need to be recorded and made available to the appropriate regulatory authority	Member States, EU, CITES
3. It is possible for stocked compartments to produce increased number of glass eel relative to the no-stocking option, but only when post fishing and transport mortality of the stocked fish is outweighed by increased survival of stocked fish relative to their wild congeners in the donor basin. We recommend that the model be used to test for situations where this might apply and assess whether it is likely	WGEEL

Annex 11: Country Reports 2010: Eel stock, fisheries and habitat reported by country

In preparation to the Working Group, participants of each country have prepared a Country Report, in which the most recent information on eel stock and fishery are presented. These Country Reports aim at presenting the best information, which does not necessarily coincide with the official status.

Participants from the following countries provided an (updated) report to the 2011 meeting of the Working Group:

- Belgium
- Canada
- Denmark
- Estonia
- Finland*
- France
- Germany
- Ireland
- Italy
- Latvia
- Morocco
- Netherlands
- Norway
- Poland
- Portugal
- Spain
- Sweden
- The United Kingdom of Great Britain and Northern Ireland

* Not present at Working Group

For practical reasons, this report presents the country reports in electronic format only (URL). Available at:

http://www.ices.dk/reports/ACOM/2011/WGEEL/CountryReportsWGEEL_2011.pdf