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STUDIES AND PILOT PROJECTS FOR CARRYING OUT THE COMMON FISHERIES POLICY

LOT 2: Pilot projects to estimate potential and actual escapement of silver eel

for

The European Commission

Directorate-General for Maritime Affairs and Fisheries

FINAL REPORT

NOTE THAT THE RESULTS OF APPLICATION OF MODELS TO DATASETS HERE ARE FOR ILLUSTRATIVE PURPOSES ONLY AND MUST NOT BE CONSTRUED AS STOCK ASSESSMENTS

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Intellectual Property of data sets and models

The precarious state of the eel stock has given rise to several national/regional initiatives to document local eel stocks and to develop modelling tools. We gratefully acknowledge that data and models have been made available to this project, but recognise that their publication within this report must not breach intellectual property rights, nor interfere with original intentions and applications. Consequently, only limited technical detail is presented here.

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Chapter 1. Executive Summary

The European eel is widely distributed throughout over 90,000 km^2 of inland, estuarine and coastal waters in Europe and parts of northern Africa (Moriarty & Dekker, 1997). Estimates at the glass eel stage indicate that recruitment across Europe fell in the 1980s to about 10% of former levels, and further to 1-5% since 2000 (ICES, 2008). ICES therefore advised that the stock is outside safe biological limits and that current fisheries are not sustainable (ICES, 1999). The status of the stock has not changed and remains critical (ICES, 2010).

The European Commission has initiated an Eel Recovery Plan (Council Regulation No 1100/2007, hereafter the Regulation) to protect and restore the European eel stock to sustainable levels of adult abundance and glass eel recruitment. The essentially local nature of eel stocks means that responsibility for the attainment of this objective largely resides with national governments, with individual river basins as the primary management units. Each Member State is required to establish national Eel Management Plans (EMPs). The objective of these plans is to permit with high probability the escapement to the sea of at least 40% of the biomass of silver eel relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock.

The assessment of local stocks and impacts of anthropogenic factors is a complex issue for eel, given the considerable diversity in environment, biological, fishery-related factors, large spatial coverage, and differences among the monitoring schemes and available data found throughout Europe. The ultimate aim of this project, therefore, is to provide EU eel scientists and managers with a comprehensive knowledge of the techniques most suitable for the assessment of their local eel stocks, and thereby to support the conservation and management of eel through the Eel Management Plan process.

There are a variety of approaches available to assess silver eel production and escapement, which can be categorised into those technical measures that can be used to directly determine actual silver eel escapement by catching and/or counting silver eels, proxy indicators based on knowledge of yellow eel populations, and those that are based on model predictions and extrapolations.

Given the practical and logistical difficulties associated with methods relying solely on capture of silver eels, not least the ability to catch the eels in a manner that is representative of the entire run, there are relatively few places across Europe where this method can be adopted. Although yellow eel surveys are more widespread, surveys have a significant resource requirement and therefore numbers and distribution of surveys is often limited. When one considers the requirements for a suite of methods appropriate to the diverse range of habitats across Europe, therefore, we chose to focus on modelling approaches within this project. Four models were identified as being used in Eel Management Plans and/or immediately available and with ongoing support and development: Demographic model of the Camargue (DemCam); Eel Density Analysis 2.0 (EDA); German Eel Model (GEM); and, Scenario-based Model of Eel Production II (SMEP II). These have each been summarised and described in detail within this report.

DemCam is a stage-, age-, and length-structured model that provides a detailed description of the status of the eel stock in a homogeneous water body, considering the main aspects (both natural and anthropogenic) that affect eel population dynamics. The model requires annual indicators of recruitment (time series or index), fishing impact (mortality rates) and biological parameters (sex ratio of silver eels, natural mortality of juveniles (density dependent) and adults (density independent), growth rates and size and age at silvering. These can either be directly assessed for the studied population (when data are available) or taken from the literature. The results consist of annual estimates of biomass and number of eels in catches, and yellow and silver eel stock by age, length, sex and maturation structure under different management scenarios, such as stocking, fishing regulations, and/or different environmental conditions. DemCam is programmed in MatLab, and therefore can only be used by someone experienced in programming in this language. However, a user-friendly interface is being developed.

EDA is a framework of eel density analyses rather than a fixed, end-user model, which can be applied at River Basin District, Eel Management Unit or even national scales. It operates on a geolocalized river network database and relates yellow eel densities to environmental variables, including anthropogenic impacts, extrapolated from survey sites to the river basin. The predicted yellow eel stock is converted to a potential silver eel escapement using a user-defined conversion rate. The model requires data on the presence/absence and densities of yellow eel at sites throughout the river network, typically derived from scientific surveys (e.g. electro-fishing surveys), and environmental data describing the distance of each site from sea and source, the temperature in each segment of the river network, the mean rainfall, the elevation, slope and stream order (Strahler and Shreve). The anthropogenic impacts are described as the obstacle pressure (cumulative number of dams and their passability), the land use, and fisheries. The model results are the yellow eel density in each reach of river network, the overall yellow eel stock abundance and a potential silver eel escapement under pristine and actual conditions. EDA was developed with the R software and with the PostgreSQL/PostGIS. The software packages are open source so are freely available, but using EDA requires a working knowledge of these software packages.

GEM is an age-based model working on a single spatial unit. The model starts with an estimate of the numbers of eel per age group in the population, and then estimates the number of eels of each age group which leave the system for various reasons (natural mortality, fisheries, predation, turbines, etc) each year, along with updates for recruitment and stocking. The model requires data on the annual catches by fisheries and predators, as weight or numbers of eels, the numbers per age group of eel recruiting or stocked each year, and the annual mortality (%) of silver eels due to hydropower plants and water abstractions. Counts or length distributions are converted to age profiles based on survey data and knowledge of growth rates, and the mean weight of eel per age group is estimated from numbers using user-defined weight-length relationship. The results are presented as annual estimates of population size, fishery catches, catch by predators, mortality by other natural reasons, and silver eel escapement, all expressed as numbers per cohort. GEM runs in MS-Excel® so can be run by anyone with a standard Windows® pc and knowledge of spreadsheets.

SMEP II is an age-, sex- and length-structured model which is applied at the river basin scale. It simulates the biological characteristics of the eel population (growth, natural mortality, sexual differentiation, silvering and dispersal) and a number of potential anthropogenic influences on that population (fishing, turbines, barriers, stocking). It accounts for density dependent effects on biological processes, as well as the effects of habitat structure and quality. It tracks changes in undifferentiated, yellow (male and female) and silver (male and female) eels every year and for each reach in the catchment. The model can be used to project the population forward from a predetermined starting condition or estimate the starting conditions that could lead to a given population size or structure. As a projection tool, the user may vary anthropogenic influences and levels of recruitment in order to create 'what-if' management scenarios, relative to given reference points. The model must at least have information describing the eel life history processes (either site-specific or from literature), the size and structure of the river basin, and the quantity of annual recruitment. Where data are available, either for historic or present conditions, these can be used to characterise the yellow eel population, impacts on production (e.g. fishing or turbines), inputs such as stocking events, and changes in the available area and quality of habitat. Model results are provided both as numbers and biomass of eel, per sex and life stage (elver, yellow, silver), river reach and year; and length frequency distributions. The model runs in MS-DOS® on any Windows® pc, using .csv files for input and results files, so can be run by anyone with knowledge of spreadsheets. One of the key aims and novel approaches of this project was to test and compare the accuracy and precision of the presently available assessment models in their predictions of silver eel escapement under various scenarios, as a means to aid scientists and managers in the selection and application of models most appropriate to particular management situations. Although several assessment models have been developed and applied to a variety eel datasets elsewhere, such formal comparisons had not been achieved before this project.

We have examined a number of candidate data sets from river basin districts (eel management units) across the continental productive range of the European eel, including several each from the North Sea-Baltic, Atlantic and Mediterranean regions. We selected 8 candidate data sets encompassing these three regions, which we considered presented elements that were 'rich' and 'poor' in terms of the eel data they provided, and which broadly represented the variety of eel-producing habitats and anthropogenic impacts found across Europe.

The River Elbe Basin District (Germany: North Sea/Baltic region) provides data for a large, single catchment area with time series data from 1985 onwards documenting eel recruitment and stocking inputs, growth rates and age-length-weight relationships, and losses associated with commercial and recreational fisheries, cormorant predation and passage at hydropower plants. The availability of these time series of data meant that the Elbe was considered as data-rich for this project.

The Anglian River Basin District (UK: North Sea/Baltic region) provided a long time series of eel density and biomass measures from electro-fishing surveys for some river, but individual length and weight data were only available for the most recent 5 years. In the absence of information on recruitment, yellow eel production and silver eel escapement from the same river, the basins of this district were considered relatively data-poor for this project.

The West Coast River Basin District (Sweden: North Sea/Baltic region) is the only basin district considered for this project that includes a significant proportion of open coastal waters. Eel data were available from marine fishery catches, and estimates were also available for mortality associated with eel passing hydropower dams in freshwater. However, as the eel data were limited to catch sizes and associated biological information about the eels in the catches from the saline parts of the district, it was considered data-poor for this project. Indeed, none of the four models could be applied to this RBD because of its open-water nature, and therefore it does not feature in this report.

The Western River Basin District (Ireland: Atlantic region) provides data for a whole EMU and also for some data rich catchments within the EMU such as the Corrib system and the small, commercially unexploited Burrishoole river basin where a 40+ year time series of complete silver eel counts and measures, along with scientific surveys of the rivers and lakes upstream meant that we considered this data set to be data-rich for this project. The WRBD also introduces the complexity of eel data collection and assessments in catchments where eel habitat is dominated by lakes.

The Brittany Eel Management Unit (France: Atlantic region) provided data from a comprehensive series of electro-fishing surveys for eel, time series data on substantial glass and yellow eel fisheries, and a full GIS database mapping the locations and impacts of obstacles to eel dispersal. Therefore, this data set was considered relatively ‘data-rich’ for this project.

The Basque River Basin District (Spain: Atlantic region) provided a time series of glass eel catches since 2003, abundance and biometric data for yellow and silver eel caught during electro-fishing surveys since 1988, and an inventory of dams and hydropower facilities. The River Oria was the site of pilot studies for the Indicang project and therefore fishery-independent data on glass eel and yellow eel recruitment were available from 2005, along with yellow eel density data throughout the river basin from eel-specific electro-fishing surveys since 2004, and estimates of silver eel production from 2007. Therefore, the Oria was considered as ‘data-rich’ for this project.

The Rhone Eel Management Unit (France: Mediterranean region) includes three compartments, the main river basin, the many river basins flowing directly to the sea, and a series of Mediterranean lagoons. Glass eel fishing is forbidden throughout the area, but large yellow and silver eel fisheries occur in the lagoons and River Rhone. There are large numbers of hydropower dams on the rivers, which have a significant impact on silver eel escapement. As with the Brittany EMU, eel presence/absence and density data were available from electro-fishing surveys throughout the Rhone EMU. However, the lack of accompanying quantitative biological information about eel production, or about the human impacts on this production, meant that this data set was considered relatively ‘data-poor’ for this project.

The Sardinian Eel Management Unit (Italy: Mediterranean region) is characterised by a series of lagoons that produce eel. Some eel data were available on age and size of recruits (but not counts), yellow and silver eels, the length distribution and sex ratio of silver eel samples, and rates of natural and fisheries-related mortality. Information describing the spatial dimensions of the eel habitats was also available. However, all these data were relatively sparse and therefore this data set was considered ‘data-poor’ for the project.

Although some of these data sets summarised above were considered to provide elements that were rich in eel data, the comprehensive test of the accuracy and precision of assessment models requires data that provides a complete knowledge of those factors controlling the life history and production of the eel. Furthermore, it requires this knowledge over a time series of sufficient length and with sufficient changes in impacts to represent all likely assessment and management scenarios found in reality. None of the candidate data sets from around Europe includes sufficient data on all these aspects to allow comprehensive testing of approaches to accurately predict or identify all of the assessment (potential target) and management (compliance assessment) scenarios. Therefore, we developed a new, virtual data set (CREPE) encompassing the present understanding of biological processes, management contexts, and the typical availability and quality of survey data in river networks. This data set provided a unique opportunity to test the performance of the models against a truly data-rich scenario.

We applied the models to this artificial, data-rich resource and examined their performance in terms of the accuracy and precision of their predictions of silver eel production/escapement under conditions of high and stable recruitment (synonymous with the 'pristine') and in response to decadal reductions in recruitment (the 'present'), both in terms of the predicted absolute values and in their ability to match the trends in production/escapement over the time series of changing recruitment.

The model testing procedures were conducted in two phases. In the first phase, input data were derived from the document describing CREPE and from the data files provided for each model. The silver eel escapement biomass predictions by the four models were compared to those from CREPE, which had been withheld from users to create a "blind" test. This phase simulated a "data not quite rich" scenario, but probably the best that could be expected in the 'real' world.

In the second phase, the input data were adjusted or 'tuned' to improve the fit between the predictions of silver eel biomass and the actual results from CREPE. 'Tuning' is the systematic revision of model parameter values to produce a model output as close to desired as possible. This illustrates the ideal data-rich situation from which to examine the potential accuracy and precision of the assessment model. However, as such a truly data-rich situation does not exist in the real world, the results represent the best possible but not necessarily what managers should expect in other situations.

The models did not perform well under the Phase 1 test conditions, failing to correctly estimate pristine or current escapement. Three of the models generally overestimated silver eel escapement: SMEP II by a factor of 5 to during the period of high recruitment and 14 to 18 during low recruitment; GEM by a factor of nearly 40 during the high recruitment period and about 10 times during the low recruitment period; and, DemCam by a factor of about 3. In contrast, EDA underestimated silver eel production by a factor of about 6.

However, all four models did succeed in predicting the general trends in escapement.

When comparing the results of the four models from the first phase of test applications, we identified three features of the CREPE dataset which could explain the some of the difficulties encountered by the assessment models during Phase 1 of testing. First, the eel distribution in CREPE environment was concentrated in the downstream part of the catchments, which complicated the application of the single-compartment assessment models GEM and DemCam. In contrast, the multiple compartment approaches of SMEP II and EDA were able to 'handle' this spatial complexity. Second, the growth rates of CREPE eels were relatively high, corresponding to a situation in north of the Gulf of Biscay, and the lognormal variability used in the growth rate calculation led to exceptionally high growth rates for some fish. Third, the mortality rate used in the CREPE dataset was particularly high, and especially so for glass eel, probably too high to be captured and represented by classical approaches in assessment models. Note however that those applying

DemCam, GEM and SMEP II all identified that this early mortality rate was much higher than could be expected from the CREPE information. This feature did not affect the EDA application because it used data for the older yellow eels.

Knowledge of these features of CREPE significantly improved the performed of the assessment models in the second phase of testing, showing a clear convergence towards the CREPE output.

Although DemCam overestimated pristine escapement by a factor 2, it predicted escapement accounting for human impacts to within, on average, 9% at high and 13% at low recruitments. SMEP II also predicted silver eel escapements that were, on average, within 9% of pristine and 16% of 'current' escapement from CREPE under high recruitment conditions. However, SMEP II underestimated silver eel escapement under low recruitment, being on average only 32% of the CREPE result, probably because the whole time series was modelled using the mortality rate curve associated with high densities. EDA overestimated pristine escapement by a factor of 3, but was much better at estimating escapement under 'current' conditions when taking human impacts into account, producing results within, on average, 24% of CREPE for the high recruitment period, and within 57% under the low recruitment conditions. The GEM results were also considerably improved over those from Phase 1, but still overestimated escapement by a factor of about 10 at high recruitment, and about 3 at low recruitment.

The ratio of current/pristine is the primary reference target of the Eel Regulation (EC 1100/2007) – the 40% or 0.4. This ratio predicted by SMEP II during the period of high recruitment was very close to that of CREPE. In contrast, EDA and DemCam both underestimated this ratio considerably during this period, but this is probably because both these models derived pristine biomass as the maximum potential biomass from an excess of recruitment, rather than escapement under a high but not necessarily maximum recruitment described for CREPE. The second application of GEM did not include an estimate of pristine escapement so this ratio could not be calculated for this model.

Given this overestimate of pristine biomass, it is hardly surprising that DemCam and EDA underestimate the ratio of pristine/current silver eel escapement during the period of low recruitment. SMEP II also underestimates this ratio, but this is probably because a high rate of natural mortality was applied in SMEP II and this meant it underestimated the current escapement during this period. This underestimation during the period of low recruitment is to err on the side of caution for the assessment and management of the eel stock as it would probably lead to extra management measures to protect and recover the stock. However, there are potentially social and economic consequences of this cautious approach by inducing management actions which are possibly more than severe than required to achieve the target.

That proves that the four models can each predict an escapement close to the CREPE value, providing there is access to sufficient input and tuning information such as in the virtual fully-data-rich situation of CREPE. However, the application of these (and other models) in the real world should not be so optimistic when most of the time only default values describing eel life processes (as Bevacqua *et al.* 2011 mortality rates) are available. Natural mortality data are rare, especially river-specific, and therefore most model applications will use the average values developed by Bevacqua *et al.* (2011). In the absence of better, more site-specific data, those values derived on the basis of Bevacqua *et al.* may be an improvement from using a single default value but the outputs should still be treated with caution. Applicants and assessors must recognise this limitation of the data and the models. To some extent, in the absence of site-specific data, it makes sense to standardise life history parameters across models and across regions so that we are at least all working to the same set of rules.

Accuracy is a measure of how close the predicted result was to the actual result. The Phase 2 results showed that three of the four models can be quite accurate when given sufficient input and tuning data, reaching average levels within 9% (DemCam, SMEP II), 24% (EDA). GEM was less accurate, a best overestimating silver eel biomass by, on average, a factor of about 3. This possibly suggests that GEM was less flexible than the other models to eel production values that were far outside those of its development data set from the River Elbe. A new CREPE data set based on eel production characteristics from northern Europe would probably have produced more accurate results from GEM. It will be useful in the future to use CREPE to simulate several new datasets corresponding to different biological hypotheses of eel population functioning and different management scenarios in order to test the robustness of the assessment model and their interest in a management process, but this was not practicable within the POSE project.

All four models presented results for silver eel escapement biomass to the nearest kg or even g, so in strictest terms, the models could all be considered to report results with high precision (the detail of the answer). However, although reporting to the nearest kg may seem an attractive quality of any model, it is actually a potentially misleading result, giving the perhaps false impression that the result has a high degree of confidence. A much more meaningful test of precision, which gives a proper measure of the confidence associated with the result, is the measure of the uncertainty associated with this result, e.g. we are 95% sure that the actual result lies between X and Y kg. None of the model results reported results with any degree of uncertainty, and therefore their precision could not be determined.

In addition to these tests of the virtual data-rich scenario, we applied the models to the real data set that came closest to being data-rich (the Western River Basin District, Ireland) to examine their performance under 'real' conditions. Comparisons between the model predictions and the 'known' outputs of Burrishoole silver eel production revealed that the different modelling approaches did not converge to a single conclusion in terms of their accuracy in predicting silver eel production. The predicted results from the models were far from the actual reality under some conditions, but more accurate under other conditions.

In order to provide an illustrative guide to local stock assessment procedures suited to the various habitats from which silver eel can escape, we described the manner in which these assessment models could be applied to the a number of additional 'real' scenarios from across the regions of Europe to further illustrate how the models could be modified to suit local conditions.

This work identified that each of the models is suitable for application to a different character set of eel data and scenarios. Given the broad range of assessment data, impacts and management scenarios that may occur in eel management units across Europe, there are a vast number of possible combinations, making it impracticable to list these and assess each of the four models for their suitability. As an alternative, therefore, we have considered the broad types of scenarios to which the assessment models are probably best suited. This guide will assist managers in identifying the model or models that should be most appropriate for their specific circumstances.

DemCam and GEM are similar in that they are typically applied to a single spatial unit. As a consequence, they are best applied to those areas where the eel production processes are not expected to vary much throughout the management unit. DemCam was developed to model eel production in lagoons, so clearly it is best suited to being applied to such environments. GEM was developed for the River Elbe, but on the basis of a series of production process values being representative across the entire river network.

SMEP II is an eel life history model in the same manner as DemCam and GEM, albeit that they each only model the life history from recruitment to spawner escapement. However, SMEP II specifically incorporates the spatial complexities of a river basin, including any network of rivers, lakes, estuaries and lagoons. Although the descriptions of growth, natural mortality, sex differentiation and silvering are common throughout the river basin, the ability to model the dispersal of eel, the effects of density dependence, and to localize impacts allows a more complex and spatially explicit assessment.

DemCam, GEM and SMEP II all require information describing these biological processes. Although each model can use information taken from the literature, the results from our phase 1 tests of the CREPE data set highlight the risks this poses of introducing additional uncertainty in the model results and the potential pitfalls in assuming that information taken from other rivers, districts or regions is representative of the eel population in question. Local knowledge of growth and natural mortality rates, and of recruitment levels, appears particularly important and emphasizes the importance of collecting local field data.

The EDA model adopts a very different approach to the other three models, at its core relying on identifying relationships between yellow eel densities and habitat characteristics, extrapolating these across the area in question, accounting for any losses due to impacts, and applying conversion rates to produce results for silver eels. Two key consequences of this approach are that EDA is far less reliant on local knowledge of eel production processes, and that it is best applied at much larger spatial scales than the other models – typically at RBD / EMU scale.

This ability of EDA to produce results at the default spatial scale of the Eel Management Plans is no doubt appealing. The other three models have been applied at much smaller management units – even though the GEM was developed for the Elbe river basin district, this was in effect a single, albeit very large, river basin. As such, DemCam, GEM and SMEP II are best applied to “index” rivers and the results then extrapolated to other rivers, etc within the River Basin District. On the other hand, EDA requires a substantial distribution of eel density and habitat data from across the management unit, so is best applied to those areas with comprehensive national eel survey programmes.

Natural mortality data are rare, especially river-specific, and therefore most model applications will use the average values developed by Bevacqua *et al.* (2011). In the absence of better, more site-specific data, those values derived on the basis of Bevacqua *et al.* may be an improvement from using a single default value but the outputs should still be treated with caution. Applicants and assessors must recognise this limitation of the data and the models. To some extent, in the absence of site-specific data, it makes sense to standardise life history parameters across models and across regions so that we are at least all working to the same set of rules. A gap identified which will require attention in the future is the lack of assessment methodology(s) for quantifying eel production in large water bodies (e.g. lakes, large rivers, large estuaries, coastal waters). The ICES SGAESAW began the process of developing assessment methods for marine eels in 2007, with a synthesis of knowledge, but the next steps to actually develop methods and indicators have not been taken yet. The resource limits of POSE prevented this project from exploring this topic further.

Our work reveals that typical eel assessment models require a substantial knowledge of the local eel population if they are to provide good results. However, there are without doubt very large numbers and areas of habitats across Europe that are producing eels, or at least have the potential to do so, but for which there is an absolute lack of any knowledge of the eel stock. Extrapolation of knowledge gained from the data rich monitored catchments to catchments where little or no eel data exists is therefore required. A Bayesian framework was developed for estimating silver eel production for sites across Europe where absolutely no prior knowledge of eel was available, by linking models of growth rate, survivorship and recruitment. The growth rate modelling appeared robust but there were weaknesses in the survivorship and production modelling. The production model estimates did not appear to respond adequately to the higher growth rates. Lack of data (recruitment, production) and maybe lack of appropriate explanatory variables were identified as factors warranting further consideration and development before the Bayesian framework would be suitable for general applications.

The main body of the report concludes with a consideration of the suitability of the assessment models, illustrated by an analysis of their Strengths, Weaknesses, Opportunities and Threats (SWOT analysis), and makes recommendations for further developments in these models and assessment approaches in light of the results of this project.

A series of annexes are also available which provide additional information including more detailed descriptions of the existing assessment models examined in the project, the new virtual, data-rich data set and associated model development, the structure of a database developed to facilitate complex data exchange between data providers and modellers, the development of the Bayesian framework to predict eel growth and production in the absence of any eel data, the Minutes from the various Project Meetings, and a List of names and address details for those who are first points of contact for the models and data sets used in this project.

In conclusion, POSE has provided a standardised, benchmarked suite of assessment methodologies that can feed into the reporting requirements for the EU Regulation and facilitate assessments of the international stock. Our work identified gaps and sensitivities in the models and their approach to stock assessments. Understanding these sensitivities means we are better informed about the modelling processes, and managers who employ the models as part of the EMP reporting process avoid unnecessary pitfalls.

A critical lesson learned during the project has been that it takes a lot of time to evolve the application of a model to a dataset to produce a confident result, and that is providing that the appropriate tuning data are available. Without these tuning data, the model application can be achieved quicker, but the results must be treated with considerable caution! Modelling fisheries data can be time consuming and this extensive time needs to be built into the whole process of stock assessment, particularly with respect to management and reporting of eel under the Regulation. Likewise, we caution against the blind use of model outputs without ground truthing.

We identified several crucial data requirements in each of the models, and these data are rare in the real world. The anticipated developments in the Data Collection Framework (DCF) sampling requirements for eel should fulfil many of the data requirements for local modelling that were identified during this project. The data collection must be coordinated and conducted at appropriate spatial and temporal scales, and embrace both fisheries and non-fisheries sources of data. The inclusion of non-fisheries related data collection under and agreed programme of surveys will close data gaps and improve our ability to undertake good quality assessments.

POSE developed CREPE which can be used as a framework to provide a series of baseline data sets benchmark test these and other models developed in the future. Countries adapting existing models or developing new models should benchmark test the model against CREPE. This will maximise the opportunity for successful reporting to the EU in future years and reduce the threat of uncoordinated assessment outputs. The original version of CREPE produced for POSE created a data set of virtual eel with characteristics closest to the southern parts of the eel's range, and this caused some difficulties for those models that had been developed for more northerly eel populations. Different scenarios (biological, management) will have to be included in different versions of CREPE in the future.

POSE also developed a database structure for eel (DBEEL) in order to facilitate the collation and dissemination of standard data. This structure could be adopted at the national or international level to support the coordinated assessment and management of *Anguilla anguilla*, and the intercalibrations requiring exchanges of eel data. The requirement for such a database has already been raised by the EIFAAC/ICES WGEEL. However, management of the database is a substantial task also requiring quality control measures.

The international coordination of data exchange and reporting, already in place for other species, will support both local and international assessments and reporting. The DBEEL developed in the POSE project provides a cost-effective and practical solution to eel data management and exchange, but this database needs a home, management and a formal data exchange and quality assurance procedure.

However, international coordination of data collection is lacking and in its absence, benchmarking and quality assurance (planning stage) and control (ongoing) of the local stock assessments is difficult to say the least. It should also be acknowledged that some data sets are coming to an end because of reductions or closure of fisheries or other economic factors. Our model testing in POSE highlights the great importance of historic data and time series which are fundamental to deriving the historic eel production values required by the Eel Regulation. Clearly, time series data collections should be protected, and new time series commenced, especially of recruitment and silver eel escapement.

Chapter 2. Introduction

The European eel, *Anguilla anguilla*, is a catadromous species. Neither spawning adult eels nor eggs have ever been observed in the wild, but it is assumed that spawning takes place in the Sargasso Sea area of the Atlantic Ocean, since larvae (leptocephali) of progressively larger size are found as one moves from the Sargasso Sea to European continental shelf waters (Tesch, 2003). At the shelf edge, around 5,000 km from the Sargasso, the laterally flattened leptocephali transform into rounded, transparent glass eels that migrate into coastal waters and estuaries and, as increasingly pigmented elvers, move into lagoons, rivers, lakes and streams. Following immigration into continental waters, the yellow eel growth stage lasts for 5 to 50+ years, during which the eels may occupy fresh water, estuarine or inshore marine areas (Aria *et al*., 2006). Sex differentiation occurs when the eels are partly grown, though the mechanism is not fully understood and appears to depend on local stock density (Davey & Jellyman, 2005), as may be the rate at which yellow eels migrate through catchments (Feunteun *et al.,* 2003). At the end of the continental growing period, the eels begin to mature into silver eels and return from the coast to the Atlantic Ocean, when the females are usually much larger and may be twice as old as males. At this stage, they return to spawn in the Atlantic Ocean. The biology of the returning silver eel in ocean waters is completely unknown. Consequently, the target for restoration of the spawning stock is formulated in terms of the escapement of silver eels from continental waters towards the ocean.

The European eel, is widely distributed throughout over 90,000 km^2 of inland, estuarine and coastal waters in Europe and parts of northern Africa (Moriarty & Dekker, 1997). Most recent evidence suggests a single panmictic population throughout its range (van Ginneken & Maes 2005; Palm *et al*. 2009; Pukolar *et al.* 2009). More than 25,000 people were thought to rely in part at least on eel fisheries for their income in the late 1990s (Moriarty & Dekker, 1997). Estimates at the glass eel stage indicate that recruitment across Europe fell in the 1980s to about 10% of former levels, and further to 1-5% since 2000 (ICES, 2008). ICES therefore advised that the stock is outside safe biological limits and that current fisheries are not sustainable (ICES, 1999).

The European Commission has initiated an Eel Recovery Plan (Council Regulation No 1100/2007, hereafter the Regulation) to protect and restore the European eel stock to sustainable levels of adult abundance and glass eel recruitment. The essentially local nature of eel stocks means that responsibility for the attainment of this objective largely resides with national governments, with individual river basins as the primary management units. Each Member State is required to establish national Eel Management Plans (EMPs). These plans aim to achieve an escapement of silver eel to the spawning population that equals or exceeds 40% of the potential biomass that would be produced under conditions with no anthropogenic disturbance due to fishing, water quality or barriers to migration.

Eel Management Plans were submitted to the EU in late 2008 and early 2009. Evaluations by DG MARE and a panel of experts from ICES continued through 2009 and early 2010 but most EMPs have now been approved for implementation. Review of the EMPs for the partner Nations, along with the evaluation of the draft plans from 16 nations published by ICES in November 2009, confirms the range of assessment and management situations encountered across Europe, and corresponding information needs and anticipated requests for scientific advice. Dekker (2000a) analysed data on recruitment, stock and fisheries to determine how they vary geographically across its range in European waters, noting that the "stock is characterised on the one hand by a large area of distribution and large-scale, long lasting fluctuations in stock, and on the other hand by the partitioning of the stock between small-scale scattered waterbodies, which support small-scale fisheries under local management". Anthropogenic impacts include fisheries, turbines, pollution, lack of water or oxygen, non-native pathogens, habitat loss or disconnect (resulting in density-dependent mortalities).

This project addresses the requirements of the European Commission in concluding a service contract for Studies and Pilot projects for carrying out the common fisheries policy Lot 2: *Pilot projects to estimate potential and actual escapement of silver eel* specified in the Call for Tenders MARE/2008/04 "*Studies and Pilot projects for carrying out the common fisheries policy*".

Assessment of local stocks and impact of anthropogenic factors is a complex issue for eel, given the considerable diversity in environment, biological, fishery-related factors, the large spatial coverage, and differences among the monitoring schemes and available data found throughout Europe. The ultimate aim of this project, therefore, is to provide EU eel scientists and managers with a comprehensive knowledge of the techniques most suitable for the assessment of their local eel stocks, and thereby to support the conservation and management of eel through the Eel Management Plan process.

The original project specification set out eight Terms of Reference (ToR), as follows:

(1) Select six river basins from which silver can escape as part of their spawning migration, with basins representative of the various (silver) eel habitats throughout the range of European eel within EU territory, and covering the North and Baltic Seas, the Mediterranean Sea and Atlantic Arc;

(2) Categorization of these river basins into three groups: data-rich; data-poor; and no-data;

(3) Development and/or elaboration of various methods which can be applied to these data groups to determine, as accurately as possible, the potential silver eel escapement (that would have existed) in the absence of anthropogenic influences.

(4) Development and/or elaboration of various methods which can be applied to these data groups to determine, as accurately as possible, the actual silver eel escapement rate.

(5) Development of coherent local stock assessment procedures suited to the various habitats from which silver eel can escape.

(6) A thorough description of population models used to assess the eel stock in compliance with the recovery target (*vis*. 40% of the potential escapement in the absence of anthropogenic influences). Description of improvements made to these models to adapt them to the biology, ecology, chemistry and geography of the specific eel habitat to which they are applied.

(7) Indication of the precision and accuracy of the population models used.

(8) In addition, guidelines or manuals of best practice should be established for methodologies for assessment of the status of the eel population, impact of fisheries and other anthropogenic impacts.

The project has analysed the assessment models available for the scientific assessment of eel stocks against the EU Regulation Limit (i.e. 40% of pristine silver eel escapement) to establish best practice in using these methodologies for the assessment of the status of eel stocks and current knowledge to inform their management and future monitoring. In addition, the project developed a Bayesian model framework for the pan-European analysis of eel growth and production against environmental descriptors, with the aim of producing a method to estimate eel production from areas where absolutely nothing is known about the local eel population.

The report is primarily aimed at those scientists and managers who are actively involved in the eel management process. Therefore, rather than being structured to report sequentially against the ToRs, this report has been structured to guide the reader (scientist) in comparing the assessment model(s) and approach(es) relevant and most appropriate to their eel management unit – as such, this report constitutes a guide to best practice for the assessment of the eel stock from a productive area, with the specific focus on the management target of the European Eel Regulation, i.e. the ratio of present to historic silver eel escapement biomass (ToR 8). The remaining chapters provide the following information, with references to ToR shown in parentheses:

Chapter 3 describes and considers the variety of approaches available to assess silver eel production and escapement, and introduces the assessment models considered during this project (ToR 3, 4, 8);

Chapter 4 provides an assessment of the accuracy and precision of these assessment models when tested against data-rich scenarios (ToR 7), and includes: a discussion of the information and knowledge required to conduct such tests of accuracy and precision; a description of the development of an ideal data-rich scenario based on a virtual eel population, the application of the assessment models to this virtual data set, consideration of the results and the modifications made to the models to adapt them to the eel production dynamics of the virtual data set (ToR 6); and, a description of a 'real' data set that comes closest to providing a data-rich scenario (ToR 1, 2), and the application of the assessment models to this real almost-rich scenario and consideration of the results (ToR 3, 4);

Chapter 5 provides further illustrations of model applications to other data sets which were considered to be data-poor for a variety of reasons, highlighting how applications are modified to suit local conditions (data, etc) (ToR 1 to 6, 8);

Chapter 6 describes the development and testing of a new method by which to assess eel production in the complete absence of eel data (ToR 3, 4, 5);

Chapter 7 concludes the main part of the report with a consideration of the suitability of the assessment models, illustrated with an analysis of their Strengths, Weaknesses, Opportunities and Threats (SWOT analysis), and makes recommendations for further developments in these models and assessment approaches in light of the results of this project;

Chapter 8 lists the References cited in the report.

A series of annexes provide additional information including more detailed descriptions of the existing assessment models examined in the project (A1 to A4) (ToR 6), the new virtual, data-rich data set and associated model development (B), the structure of the DataBase EEL developed to facilitate complex data exchange between data providers and modellers (C), the development of a

Bayesian framework to predict eel growth and production in the absence of any eel data (D), the Minutes from the various Project Meetings (E), a list of contact details associated with each of the models and data sets used in this project (F), and the SQL programming code for the DBEEL structure (G).

Chapter 3. Methods available to assess silver eel production and escapement

Here we consider the methods available to determine the potential, in the absence of anthropogenic impacts, and actual escapement of silver eel. We first describe the technical measures that can be used to directly determine actual silver eel escapement, and then summarise the main features of model-based approaches that were available at the start of the project (note that we are not aware of any new model-based approaches since the project started).

Note that we refer to silver eel escapement here as the amount of eel that have successfully circumvented all of the potential anthropogenic impacts in continental freshwaters, estuaries and coastal waters (or fresh and saline waters) on their emigration to the oceanic spawning ground. As 'pristine' conditions by necessity constitute an absence of any anthropogenic impacts, the production of emigrating silver eels equals the escapement. For clarity therefore we refer to escapement under pristine conditions as 'production'.

Note also that on first examination of the task of stock assessment, it might appear that time-series data on spawner emigration and glass eel recruitment are the only information essential to assessment of the stock. This view, however, ignores the reality of the current and probable future situation for eel. Only if recruitment were to recover rapidly following measures to increase spawning stock, resulting in confidence that recovery is underway, would these two data items suffice. Such a rapid recovery is an unlikely scenario, given that our ability to increase spawning stock escapement significantly will be limited for at least an eel generation, as a consequence of the past 15 to 25 years of low recruitment yet to feed through to spawner emigration. It is quite probable, therefore, that recruitment will continue to decline for some time, and it will be almost unavoidable that silver eel escapement will also decline considerably further for at least some years. The effectiveness of protection and restoration measures taken under the EU Eel Regulation will therefore have to be judged on a relative scale: the relative improvement of survival from recruit to silver eel, that is: quantification of anthropogenic mortality in the continental stage. This will require the use of "standard" fish stock assessment tools, including cohort-models, length-based assessments, etc. At this moment, experience with applications of this type of models is extremely limited (Vollestad & Jonsson 1988; Dekker 2011, in prep.).

3.1. Methods based on catching or counting silver eels

There are several means by which silver eel escapement can be estimated (at least) directly from catching or counting eels. The EIFAC/ICES Working Group on Eels reviewed these methods in 2008 (ICES, 2008). The following develops from this review, and adds consideration of the major practical issues associated with deploying these methods at geographical scales appropriate for basin district or national assessments.

Traps

Wolf traps, or related systems, or use of winged nets deployed for research purposes can provide precise estimates of migrating eel population dynamics and under some circumstances all silver eels can be counted and weighed. However, this is usually only possible in smaller river systems where discharge patterns allow for silver eel trapping throughout the migration season. Examples of this type of silver eel escapement estimation include the studies undertaken on the Norwegian River Imsa (Vollestad & Jonsson, 1988), the French Rivers Frémur (Feunteun *et al.*, 2000) and Oir (Acou *et al.,* 2009), the Burrishoole (Poole *et al.,* 1990).

There are several issues with applying this method for eel stock assessment that mean it is not widely suitable. There are exceptionally high resource requirements associated with installing and maintaining the traps. Given that the trap is required to fish the entire river width, there are likely to be relatively few suitable sites within RBDs. Full and accurate measures of silver eel escapement require that the trap operates throughout the entire period when emigrating fish are passing the site, and that they are all captured. However, the capture efficiency of the trap may often be reduced by varying flow conditions. Further and given the considerable size range of silver eels in some basins which may vary from 35 to 100+ cm length, the trap design may not be suitable to catch eels across the whole size range – i.e. is size selective. This is often the case with commercial gears (see below), especially where the fishery is controlled by a minimum size limit for the catch. In such circumstances, the catch may not accurately represent the full run.

Where trap efficiency is not 100%, mark-recapture methods can be employed to estimate capture efficiency of the trap. The proportion of marked eels recaptured provides an estimate of the capture efficiency of the fishery. The catch is then raised by this efficiency to estimate the size of the run. A comprehensive measure of capture efficiency would incorporate the varying effects of river condition and fish size. Note therefore that M/R requires a model-based approach to raise the catch to the whole population based on estimates of capture efficiencies: all the methods require some form of model-based approach to raise catches in account of fishery selectivity/efficiency and/or accounting for downstream parts of the basin.

Fisheries-based

Commercial silver eel fisheries can, depending on their location and scale, provide good opportunities for direct estimation of the numbers and biomass of silver eels escaping from hydrosystems, provided that it is possible to determine the efficiency (proportion of run or local stock that is captured) of the eel capture systems involved (see above). Examples of such investigations, of population dynamics and seasonal patterns of seaward migrating eels, include those undertaken on the River Loire, River Shannon and Corrib, River Bann (Lough Neagh outlet), the River Imsa, the Baltic basin and the St Lawrence. Catch and effort data from closely monitored fisheries in enclosed waterbodies such as Lake IJsselmeer (Netherlands) and Lough Neagh (Northern Ireland) allow detailed assessments of eel production. However, such large and discrete eel fisheries constitute only about 5% of the continental fisheries, with the remainder consisting of very small and disparate fisheries in comparison (Dekker, 2000a).

As with scientific monitoring studies, difficulties can occur when the fishing season does not cover the full migration period or when there is significant eel production downstream of the fishery area. Use of mark/recapture (M/R) methods for estimation of fishery capture efficiency allows for estimation of the numbers and biomass of migrating eels at the fishing sites. This can involve use of a variety of tags and marks (see Concerted Action for Tagging of Fish: www.hafro.is/catag). Experimental fisheries could be established in data poor areas and used to improve fishery monitoring methodologies. (Vollestad and Jonsson, 1988; Feunteun *et al*., 2000; Allen *et al*., 2006; WGEEL Baltic sea; and McCarthy *et al*., 2008).

Fish Counters

Counters and various acoustic technologies can allow for the estimation of silver eel escapement in locations where eel capture is not possible. McCarthy *et al*. (2008) used hydroacoustic methods to investigate variations in numbers of silver eels migrating downstream in the headrace canal of the Ardnacrusha hydropower plant in the River Shannon, Ireland. Resistivity counters have been trialled for counting emigrating silver eel in the UK (J. Hateley, pers. comm.), as have high-frequency multi-beam sonar (Didson) in the UK and the Netherlands (J. Hateley & W. Dekker*, pers. comm.). The Didson may not be suitable for deployment in rivers >15 m width if a full width count is required, and the main constraint at sites of appropriate dimensions is that the site must have a suitable profile with minimum or little shadowing of the beam.

* http://www.imares.wur.nl/NL/onderzoek/faciliteiten/didson/

Such eel counts, and linked data on size frequencies of the migrating eels, are only possible in locations where other fish species (with target strengths in the same range as the silver eels) are not also migrating downstream at the same time as eels. Work is in progress in Ireland, UK, Poland, Sweden and other European countries that should lead to improved sampling protocols and to more widespread use of this method for estimation of eel escapement rates.

General issues with catch, count and proxy approaches

Few fisheries or in-river traps are operated at the very downstream extreme of the study basin, and therefore they miss any silver eel produced from the habitats further downstream. This is especially a problem when the study basin includes the estuary or even coastal waters. Given the practical and logistical difficulties associated with methods relying solely on capture of silver eels, not least the ability to catch the eels in a manner that is representative of the entire run, there are relatively few places across Europe where this method can be adopted. When one considers the requirements for a suite of methods appropriate to the diverse range of habitats across Europe, therefore, we are focussing on modelling approaches for this project.

A further limitation of these direct approaches, as it relates to this project and European assessment and management of eel, is their inability to provide a measure of potential 'pristine' silver eel production in the absence of data from the appropriate historic period. Although such historic data exist and have been used for a small number of river basins, e.g. Burrishoole (Ireland), Neagh/Bann (UK N. Ireland), the approach cannot be used to back-calculate from present to historic production. Thus, while a new direct approach might be deployed in a river basin, it can only provide an estimate of silver eel production from now onwards, assuming constant conditions. In the absence of historic data, therefore, we are reliant on models of eel production to estimate pristine levels.

3.2. Methods based on yellow eel proxies

The use of proxy indicators from sedentary eels and habitat population models is another approach that has been applied to estimate silver eel escapement (Feunteun *et al.*, 2000; Aprahamian *et al.*, 2007; Lobon-Cervia & Iglesias, 2008). In many river systems, surveys are commonly conducted to characterize the sedentary 'yellow eel' component of the local stock. Mark-recapture or other more locally adequate methods could be used to estimate density of yellow and pre-migrant silver eels. A number of morphological characteristics have been identified that indicate pre-migrant status of eel, i.e. that they should be expected to emigrate as silver eels in the next migrant season (Feunteun *et al.,* 2000; Durif *et al.,* 2005). It is possible therefore to estimate silver eel production from a water course based on the numbers of such 'pre-migrant' eels (Feunteun *et al*., 2000; Acou *et al.,* 2009, Durif *et al.*, 2009).

The approach introduces two main sources of uncertainty in any estimate of silver eel production. First, if it is conducted in only one year it assumes that all eels classed as pre-migrants will actually become silver eels in the following migration season. Rather, evidence suggests that some pre-migrants may not emigrate in the year of marking (E. Feunteun, pers. comm.), and that studies using this method should continue sampling for these marked eels over a number of subsequent years.

The second assumption is that the eels sampled during the surveys are representative of the eel population across the river basin, such that the survey results can be raised to the system, typically according to the relative wetted areas. These procedures should nevertheless be standardized so that methodologies used can provide representative estimates of silver eel production, e.g. sampling at the beginning of the migratory season (late summer in southern latitudes and middle summer in northern latitudes). Several habitat types representative of each catchment should be evaluated in order to be able to extrapolate for the whole catchment and include it in habitat population models. Eel mortality rates need to be determined throughout the river basin including the estuary as well as fresh-water habitat.

Acou *et al.* (2009) estimated silver eel production from two coastal river systems of western France, the Frémur and Oir. In the Frémur, 29 surveys covered about 2.3% of the wetted area of fluvial habitat, and four sites each in two still waters, and up to 32 sections of the Oir, accounting for 8% of fluvial habitat but only along a 7.5 km length of river.

Obtaining population density estimates for yellow eels in large water bodies including still waters is often difficult or impossible. Studies suggest that eels are often confined to shoreline margins of still waters because of the presence of cover and food (Jellyman & Chisnall, 1999; Schulze *et al.,* 2004), though this is a topic that has received relatively little study. Whilst that presence of eels has also been recorded along the shoreline margins of many lake systems throughout Ireland (Poole, 1994; Moriarty, 1996; Matthews *et al.,* 2001; Rosell *et al.,* 2005), these findings are commonly associated with seasonality, given that the shallow waters warm up quickest thereby promoting eel feeding behaviour in these regions. However, commercial fishing experience and scientific survey data have revealed that as water temperatures begin to rise throughout the summer months, eels are more commonly found in the deeper (>9 m) waters (Matthews *et al.,* 2003; Allen *et al.,* 2006; R. Poole, pers. comm.). Nevertheless, extrapolation of fluvial densities across the entire surface area of still waters may overestimate eel production from some still waters. Conversely, assuming that only a 2.5 m wide shoreline strip of fluvial habitats produced eels and thus that eels were absent from about 95% of the fluvial wetted area, as for the Frémur (Acou *et al.,* 2009) is likely to grossly underestimate eel production in many waters.

The surveys have a significant resource requirement and therefore numbers and distribution of surveys is often limited. To date we are unaware of any study testing the number and distribution of surveys against the accuracy of their representation of the actual yellow eel population in a river system. Statistical methods are available to aid sampling design (e.g. power analysis), but these must be incorporated with spatial information on habitat diversity and distributions in order to develop statistically robust stratified sampling programmes.

3.3. Model-based approaches to estimate potential and actual silver eel escapement

The level of complexity that characterizes the life cycle of eel populations makes the simulation of its dynamics particularly challenging. Studies have already focused on the development of population dynamics models for several eel species including the American eel (*Anguilla rostrata*) (Reid, 2001), shortfin (*A. australis*) and longfin eel (*A. dieffenbachia*) (Francis & Jellyman, 1999), and European eel (*A. anguilla*). The European project SLIME (Dekker *et al.,* 2006) reviewed developments in quantitative modelling of European eel populations and tested different models in light of the management target proposed by the EC. More recently, ICES (2008) provided short descriptions of several additional modelling approaches.

The number and diversity of models developed for Anguillid species is considerable, starting with the age-structured and life table models of Sparre (1979) and Rossi (1979), respectively, and to date including input-output, stochastic and/or spatially distributed demographic, VPA-like, and multi-stage stock-recruitment models, and covering a single life stage (glass eel) to the global stock. De Leo *et al*. (2009) have provided a representative summary of the features of many of the models that have been used over the years to describe the dynamics of eels and predict the status of the stock. Because of the complex life-cycle of eels the range of information needed to describe this and the complexity of the model that could be used is high. Similarly, the range of information needed to estimate unexploited and current stock sizes and escapement is considerable. This includes standard type of stock assessment inputs/estimates such as recruitment levels, catch data but also less common factors such as stage specific stock estimates and indices of habitat quality.

This project has focused only on those models that have been applied in EMPs and/or are under continuing development, compared to those models that are not, as far as we can establish, being used in EMPs or subject to further developments. These models, listed in alphabetical order, are as follows:

Demographic model of the Camargue (DemCam)
Eel Density Analysis 2.0 (EDA)
German Eel Model (GEM)
Scenario-based Model of Eel Production II (SMEP II)

There are considerable differences between these models in terms of their level of complexity, data requirements, real cases in which they can be applied, etc. Knowledge of these differences is very important in order to identify the right model to apply to quantify potential and actual eel production and silver escapement, depending on eel population characteristics and data situations. Our work on testing and comparing these models is described in Chapter 4, but first we provide summary descriptions of the models. More detailed descriptions of the models are provided in annexes A1 to A4 at the end of this report.

3.3.1. Demographic model of the Camargue (DemCam)

Model approach and processes

DemCam is a stage-, age-, and length-structured model that provides a detailed description of the status of the eel stock in a homogeneous water body, considering the main aspects (both natural and anthropogenic) that affect eel population dynamics. A general formulation makes it suitable to describe the demography of other different eel stocks, provided that a sufficient number of data are available for parameter calibration. The model is designed to simulate the condition of the stock in actual, pristine and future conditions under different scenarios.

The model is deterministic with an annual time step, using density dependent juvenile mortality, growth of undifferentiated, male and female eels, fishing mortality and length-dependent maturation.

The model evaluates the consequences of fisheries, restocking, maturation, growth and natural mortality on the yellow and silver eel population and it explicitly account for the dynamics of glass eels to capture the effects of anthropogenic and other factors on this part of the population.

Data requirements

The model requires annual indicators of recruitment and fishing effort and biological parameters, either directly assessed for the studied population (when data are available) or taken from literature. These required parameters are: Annual recruitment (time series or index); Sex ratio (at silver stage or at 30 cm); Density dependent juvenile mortality (back calculated from historical maximum); Sex specific body growth (otolith, age at silvering); Sexual maturation (silver size); adult mortality (literature, or know); Fishing mortality (know, estimated).

Further model developments will focus on improving the description of sex differentiation in small yellow eels and producing mark-recapture estimates of yellow and silver eel abundance. The model also needs to be calibrated against length distributions of yellow and silver eel catches, to allow the ability to carry out sensitivity analyses on parameters and outputs, and to bootstrap input data.

Model Outputs

The output of the model is number of eels in a given time at a given age, size, sex and maturation stage. Based on this information, the user can estimate the number of migrating silver eels for any given time.

The model defines the eel stock and the harvest structured by age, length, sex and maturation stage (yellow or silver) on an annual basis. The output consists of biomass and number of eels in catches, and yellow and silver eel stock by age, length, sex and maturation structure under different management scenarios, such as stocking, fishing regulations, and/or different environmental conditions.

Software Implementation

DEMCAM is programmed in MatLab, and therefore can only be used by someone experienced in programming in this language. However, a user-friendly data interface is being developed.

3.3.2. Eel Density Analysis (EDA 2.0): A statistic model to assess European eel (Anguilla anguilla) escapement in a river network

Model approach and processes

This is a framework of eel density analyses rather than a fixed, end-user model. It relates yellow eel densities to environmental variables, including anthropogenic impacts, and is extrapolated from survey sites to the river basin. The predicted yellow eel stock is then converted to a silver eel escapement, using a conversion rate.

The modelling tool is based on a geolocalized river network database to predict yellow eel densities and silver eel escapement. There are six main steps in the model application:

Relate observed yellow eel presence/absence and densities to descriptor parameters: sampling methods, environmental conditions (distance to the sea, relative distance, temperature, Strahler stream order, elevation and slope), anthropogenic conditions (obstacles, fisheries) and time (year trends);

Extrapolate yellow eel density in each river stretch by applying the statistical model calibrated in step 1;

Calculate the overall yellow eel stock abundance by multiplying these densities by the surface of each stretch and summing them;

Estimate a potential silver eel escapement of each stretch by converting yellow to silver eel abundance using a fixed conversion rate;

Calculate effective escapement by reducing potential escapement with silver eel mortalities during downstream migration

Sum the effective escapement from all the stretches to give estimate at EMU scale

It is also possible to give an estimate of the pristine escapement by running the EDA model with anthropogenic conditions artificially set to zero and time variable data sets before 1980.

Data requirements

The model needs information on the yellow eel population, the environmental characteristics, and the anthropogenic impacts on eel production.

The necessary data on the eel population are the presence/absence and densities of yellow eel, typically derived from scientific surveys (e.g. electro-fishing surveys).

The environmental data are the distances to the sea and source, and the relative distance (between sea limit and the more upstream source), the temperature in each segment of the river network, the mean rainfall, the elevation, slope and stream order (Strahler and Shreve). EDA is designed to be applied at the Eel Management Unit. It uses the CCM v2.1 (Catchment Characterisation and Modelling) a European hydrographical databases (Vogt *et al.,* 2007, http://ccm.jrc.ec.europa.eu/) to derive the environmental descriptors. The CCM2 database includes a hierarchical set of river stretches and catchments based on the Strahler order, a lake layer and structured hydrological feature codes based on the Pfafstetter system (De Jager *et al.,* 2010). The primary catchment is referred to the drainage area; this is the smallest entity in this hierarchy and is drained by CCM river stretch.

The anthropogenic impacts are described as the obstacle pressure (cumulative number of dams and their passability), the land use, and fisheries.

Model Outputs

The outputs of the model are the yellow eel density in each reach of river network, and the overall yellow eel stock abundance and a potential silver eel escapement at pristine and actual conditions. The biomass and number of yellow eel and eel escapement are optional.

Software implementation

The model was developed within the R software and with PostgreSQL/PostGis (open-source software). Statistical models are calculated with a Generalized Additive Model (Hastie & Tibshirani, 1990) using the 'gam' library in the R software to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. This semi-parametric extension of generalised linear models is flexible and allows combination of both linear and complex additive responses within the same model. The best model is selected by the Akaike's Information Criterion (AIC) and Kappa coefficient when presence-absence models were used. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/).

3.3.3. German Eel Model (GEM)

Model approach and processes

The German eel model was developed specifically for describing the dynamics of the eel population of the River Elbe system, especially for estimating the escapement of silver eel between 2005 and 2007. The age-based model is data driven and was adapted to the available data series-estimated relationships. The model treats the productive area as a single unit, so does not take into account spatial aspects like different habitat patterns, area dependent growth, etc. Nor does it account for the potential effects of density on eel production processes such as growth and mortality rates.

The model is based on the structure of the Virtual Population Analysis (VPA), but the GEM works in the opposite direction. The initial population in number, by age group, at the beginning of year 1 is estimated. Then the model estimates the number of eels of each age group which leave the system for various reasons (natural mortality, fisheries, predation, turbines, etc) in the same year. The population at the beginning of the following year is then estimated based on the remaining population, and the numbers of immigrating elvers and restocked eels.

The following parameters are assumed to be stable during the total model period:

Growth and weight-length relationship

Relative age distribution of eel eaten by cormorants

Relative age distribution of silvering eel

Mean weight of eel in the stomach of cormorants

Relative age distribution of immigrating eels

The natural mortality is split up into two components: the effect of cormorants and the remaining natural mortality. It is assumed that the age distributions of eel caught in fisheries and those eaten by predators are similar to the age distribution of the stock. Also, that turbines and pumping stations only impact silver eels.

Note that, for the Elbe data set at least, analyses have shown that the size and the relative age distribution of the initial year has relative low effects concerning the year t_x if the period between the initial year and the year t_x is more than 18 years. Thus, GEM requires a 'training' data set of at least 18 years.

Data requirements

The following input data are required for the model:

Catch in kg by fishermen and angler per year

Number of restocked eel by age group and year

Number of immigrating elvers by age group and year (if data are not available for each year, estimates based on international time series can be used)

Catch in kg by cormorants per year

Mortality of silver eels due to hydropower plants and water removals in % per year

Input data are provided as numbers per cohort, with various counts or length distributions converted to age profiles based on survey data. The model requires descriptions concerning the weight-length relationship and the growth of eel to estimate the mean weight of eel by age group and to transform length based estimates into age based estimates.

The relative age distribution of eel captured by cormorants is required for estimating the total number of eel consumed by cormorants. It is assumed that the proportion of eel in the food of cormorants is dependent on the density of eel.

The model also contains the option to add stochastic noise to the input data which is normally distributed with a mean of zero and a given variance. If the variance is realistic this option can be used for estimating the confidence intervals of the escaping silver eels by means of bootstrap methods.

Outputs

Model outputs are population size, catch by fishermen, catch by anglers, catch by cormorants, mortality by other natural reasons, and silver eel escapement, all expressed as numbers per cohort.

Software implementation

The model was developed in MS-Excel® to facilitate its application to other river systems in Germany. It is easy to substitute assumptions concerning the stability of parameters over the total model period by yearly based estimates.

3.3.4. Scenario-based Model of Eel Populations (SMEP II)

Model approach and processes

SMEP II is a software package developed to model the dynamics and exploitation of eel populations (Aprahamian *et al.,* 2007). It is based on the scale of a river basin, and simulates the freshwater phase of eel production. It consists of 3 main components:

1) a population dynamics model that simulates both the biological characteristics of the eel population and a number of potential anthropogenic influences on that population (we will refer to it as the population dynamics model). Biological processes modelled include growth, natural mortality, sexual differentiation, maturation (silvering) and migration within the basin. Anthropogenic influences include environmental and habitat quality, fishing practices, barriers to migration and stocking;

2) a GIS tool that helps the user determine the spatial structure that they want to use for their calculation and prepare the input files for the population dynamics model to run; and,

3) a statistical model that is used to estimate some of the parameters of the population dynamics model.

The population dynamics model used is a length-based model that describes the dynamics of a population of eels for the duration of its stay in the river basin. The model is also sex-, stage-, and area-specific and accounts for density dependent effects, and habitat structure and quality. Therefore, it tracks changes in undifferentiated, yellow (male and female) and silver (male and female) eels within four seasons and for each reach in the catchment. The model does not make any assumptions about the dynamics of eels that have migrated from the river back to the sea (i.e. on silver eels once they exit the basin). Since only partial simulation of the population dynamics is possible (the salt water phase of population's life is not simulated), processes such as recruitment cannot be modelled explicitly and therefore, information about them needs to be provided externally((or estimated).

Model outputs are provided both as numbers and biomass of eel, per sex and life stage, river reach and year; and length frequency distributions. SMEP II is designed to provide time series or equilibrium outputs for each reach and summarised for the catchment for: undifferentiated eels, male and female yellow and silver eels: in terms of numbers, density and biomass, length-frequency distributions, sex ratios, and predicted catch numbers and biomass. The model can be used to project the population forward from a predetermined starting condition or estimate the starting conditions that could lead to a given population size or structure. As a projection tool, the user may vary anthropogenic influences and levels of recruitment in order to create 'what-if' management scenarios, relative to the given reference point.

Data Requirements

SMEP II must at least have information describing the eel life history processes, and the size and structure of the river basin and the level of annual recruitment, in order to predict potential production under 'pristine', constant conditions. Where data are available, either for historic or present conditions, these can also be applied to characterise the yellow eel population (in the past or present), impacts on escapement (e.g. fishing or turbines), inputs such as stocking events, and changes in the available area and quality of habitat. These additional data allow the user to set the model to simulate escapement under various conditions (past, present and future), and to alter the effects of impacts and inputs in order to examine their relative influence on escapement.

The biological processes that apply to the life cycle of eels in the study river (i.e. growth, sex differentiation, natural mortality and silvering) are defined by the user from river-specific information, or can be parameterised according to values from neighbouring rivers or from the scientific literature.

Recruitment of eel is described according to the length of recruits (mean and standard deviation), the maximum number of recruits in any year, and a time series of recruitment as an index of that maximum.

The model also needs information to describe the effect of density on the dynamics of eels if such effects are to be taken into account in the calculations. This is characterised according to the level of eel density biomass at or above which the density dependent variations in biological processes take a strong effect.

In terms of the spatial component of the model, the user defines the number and topography of the reaches, their length and wetted area, as well as information about obstacles that might constrain the movements of eels between the reaches. The user also sets the speed at which eels move between reaches.

Where anthropogenic impacts are to be included in the model, fishing is described as catch by stage (glass, yellow, silver eels) and assigned to specific reaches seasons and years, and turbine mortality is described as the proportion of eel that are killed when passing a turbine.

If stocking is to be simulated, this is described in terms of the number and length distribution (mean length and range) of stocked eels, the reach where they are stocked, and the season and year when the stocking takes place.

Model Output

SMEP II reports the results of simulations in a series of .csv files that provide, for each reach in every year: the density and biomass of undifferentiated, male and female yellow eels; numbers and weight of emigrating male and female silver eels; the proportion of females; and the numbers and weight of 'catch'.

End-of-run files provides summaries of density and biomass of undifferentiated, males and female yellow eels, biomass of male and female silver eels, and 'catch' (numbers and biomass) of undifferentiated, yellow and silver eels, and the length frequency of eels, stages and sexes in each reach.

Software implementation

The model has been developed using FORTRAN 95, with ArcGIS for automation of map data inputs. Input data and model outputs are provided in .csv files. The model runs on any PC running MS-DOS.

Chapter 4. Testing the accuracy and precision of models to predict silver eel escapement from data-rich scenarios

One of the key aims and novel approaches of this project was to test and compare the accuracy and precision of the various assessment models currently available in their predictions of silver eel escapement under various scenarios, as a means to aid in the selection and application of models most appropriate to particular management situations. Although several assessment models have been developed and applied to a variety eel datasets elsewhere, such formal comparisons had not been achieved before this project.

We have examined a number of candidate data sets from across the continental productive range of the European eel, including several each from the North Sea-Baltic, Atlantic and Mediterranean regions. We selected 8 candidate data sets encompassing these three regions (Figure 4.1). These datasets are described in detail in Chapter 5.

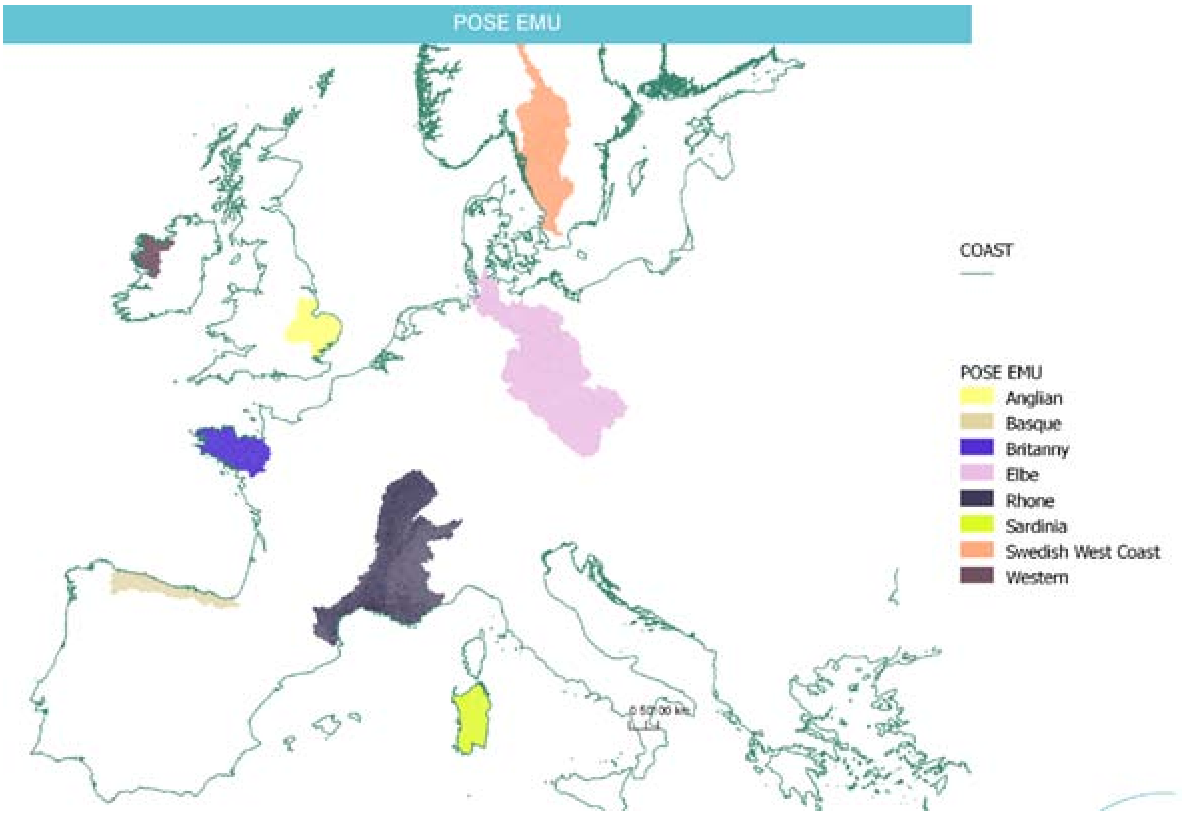


Figure 4.1. Map of Europe showing the locations of the candidate eel data sets considered in the POSE project.

Data were collated and formatted in a single DataBase Eel (DBEEL) to facilitate the delivery of standard data to each of the models. The structure of DBEEL is described in Annex C.

The Basque (Spain), Brittany (France), Elbe (Germany) and Western (Ireland) data sets were considered to be 'rich' in some form of eel data, while the remainder were considered to be 'poor' but all provided some quantity of eel data and therefore were not classified as 'no eel data'.

Therefore, an additional 12 river and marine sites were selected as 'no eel data' sites. The locations and characteristics of these sites, and their use in model testing, are described in Chapter 5.

However, the comprehensive test of the accuracy and precision of assessment models requires data that provides a complete knowledge of those factors controlling the life history and production of the eel. Furthermore, it requires this knowledge over a time series of sufficient length and with sufficient changes in impacts to represent all likely assessment and management scenarios found in reality.

None of the candidate data sets from around Europe includes sufficient data on all these aspects to allow comprehensive testing of approaches to accurately predict or identify all of the assessment (potential target) and management (compliance assessment) scenarios.

The eel data for the Burrishoole River (Ireland) comes closest to this high requirement. This data set provides a 40 year time series of silver eel escapement measured in number, biomass and sex ratio, along with scientific surveys of yellow eels from several lakes (fyke nets) and tributaries (electric fishing), and GIS-based habitat assessment. However, there is no quantitative estimate of recruitment for any year or time series. Furthermore, this eel population is not subject to any fishing pressure or other anthropogenic impacts, so it is not possible to use this dataset to test any approach in predicting the effects of fishing pressure or other anthropogenic impacts on the difference between potential and actual escapement.

Therefore, the model testing required the construction of a new dataset encompassing the present understanding of biological processes, management contexts, data availability and quality in river networks.

In this chapter, we describe the development of this artificial dataset, the application of the models to this dataset, and then to the Burrishoole dataset as a further illustration of application to 'near data rich in reality', and consider the results of these applications in terms of the accuracy and precision of the models.

4.1. Development of the new test dataset – the Constructed Reality for Eel Population Exploration (CREPE)

Here we provide a summary description of the development of CREPE, to explain the features and model outputs pertinent to the testing of the assessment models (see below). A more detailed description of the model and its development is provided in Annex B.

4.1.1. General features of the model

The CREPE model has been designed to incorporate present understanding of biological processes, management contexts, data availability and quality in river networks. Since this virtual approach provides the ideal data-rich situation, as all assessment/management scenarios are programmed into the data and therefore completely understood, the datasets produced by CREPE constitute benchmarks against which all models can be tested. Furthermore, at the conclusion of POSE, these benchmark datasets can be made available to become a standard to test other models developed in the future.

The aim of this model is not to simulate an actual study case but rather to verify and test our knowledge of eel dynamics in river basins. The intention is to capture the main characteristics of the eel dynamics while escaping criticisms based on regional particularities. However, it is based on review of the scientific literature on European eel, in order to be biologically meaningful and therefore to be acceptable to eel ecologists and managers. The simulation lasted 150 years. The time step is the season (i.e. 4 time steps per year) and the year starts with winter.

CREPE model was implemented in object-oriented language JAVA by using the modelling framework SimAquaLife (Dumoulin 2007).

4.1.2. Characteristics of the river networks

The modelled space corresponds to a simple eel management unit (EMU) composed of five random river networks (river basins), including saline waters and lakes (Figure 4.2), and the characteristics of the land draining into the water bodies. The oceanic phase of the life cycle of the eel was not modelled.

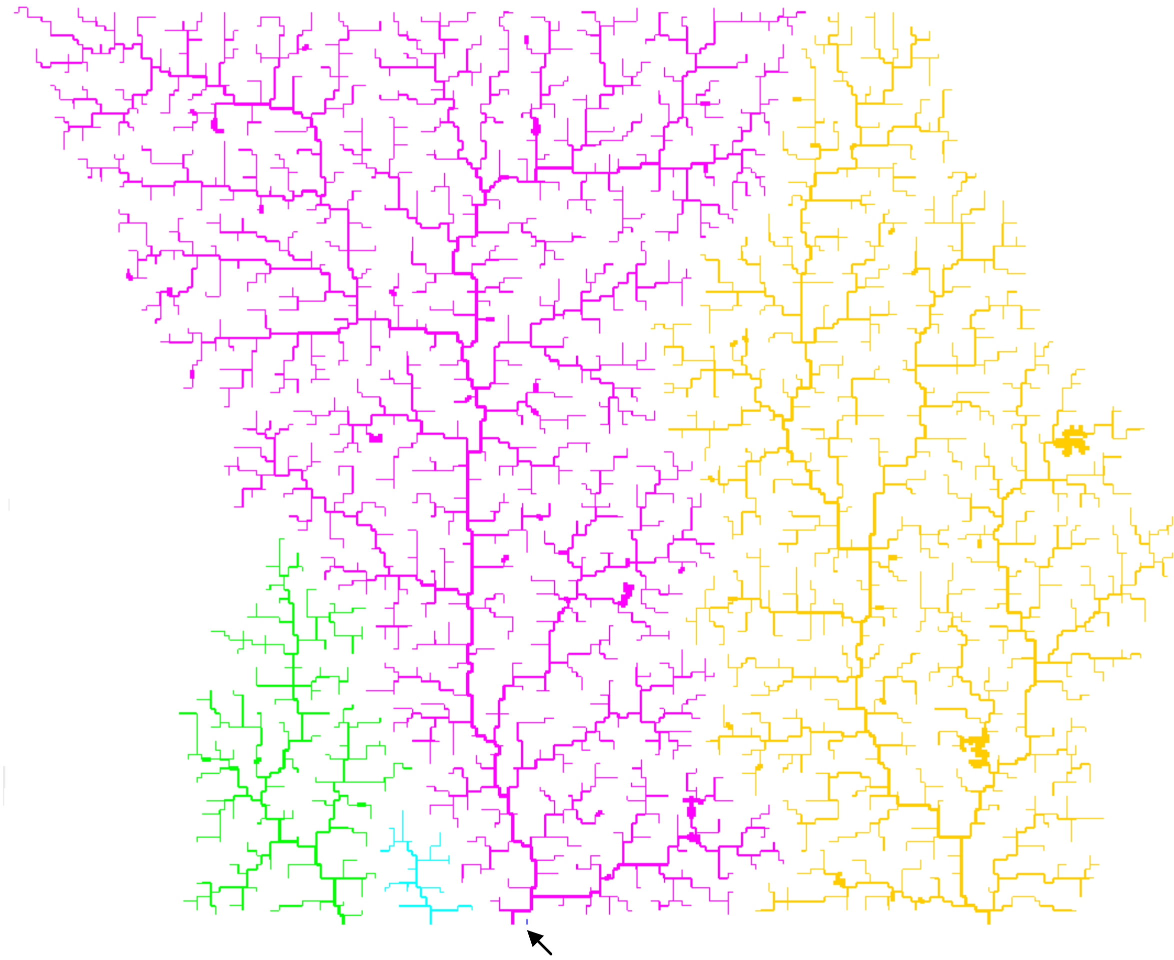


Figure 4.2. Virtual river networks used in CREPE simulation. Each river basin is shown in a different colour, and the arrow indicates the 1 reach river network (black). The thicker regions of the network identify the locations and relative size of the lakes.

The five basins have 1, 96, 740, 3789 and 5275 reaches, respectively. Each reach is characterised by the length and wetted width, the distance to the sea (corresponding to the most downstream reach), distance to the source (i.e. the top of the most upstream reach), and the Shreve index (i.e. the number of sources located upstream of a given reach) and Strahler order (Strahler 1957).

Carrying capacity for eels

The carrying capacity for eel in any reach is arbitrary fixed at 50 kg ha^{-1} for all reaches. This value corresponds to a mean value of observed biomass in a catchment and not to maximum biomass measured in downstream reaches close to 250 kg ha^{-1}. This parameter is used in the “mortality of the standing stock” and “sex determinism” processes.

Temperature time series

In order to take into account annual variations in biological processes due to varying environmental conditions, we simulated a monthly time series of temperature (air temperature as proxy for water temperature: Erickson & Stefan 2000).

Obstacles to migration

We fixed the density of obstacles at 0.01 obstacles per km^2, i.e. 400 obstacles were created in the simulated eel management unit. The locations were chosen randomly among the river reaches (Figure 4.3). We considered that 25% of these obstacles are equipped with turbines and then induced mortality during the downstream passage. An obstacle is characterized by probabilities of the eel passing downstream or upstream, and the mortality induced when fish pass downstream or upstream. Each obstacle is associated with a list of records which summarize the fish counting at the dam.

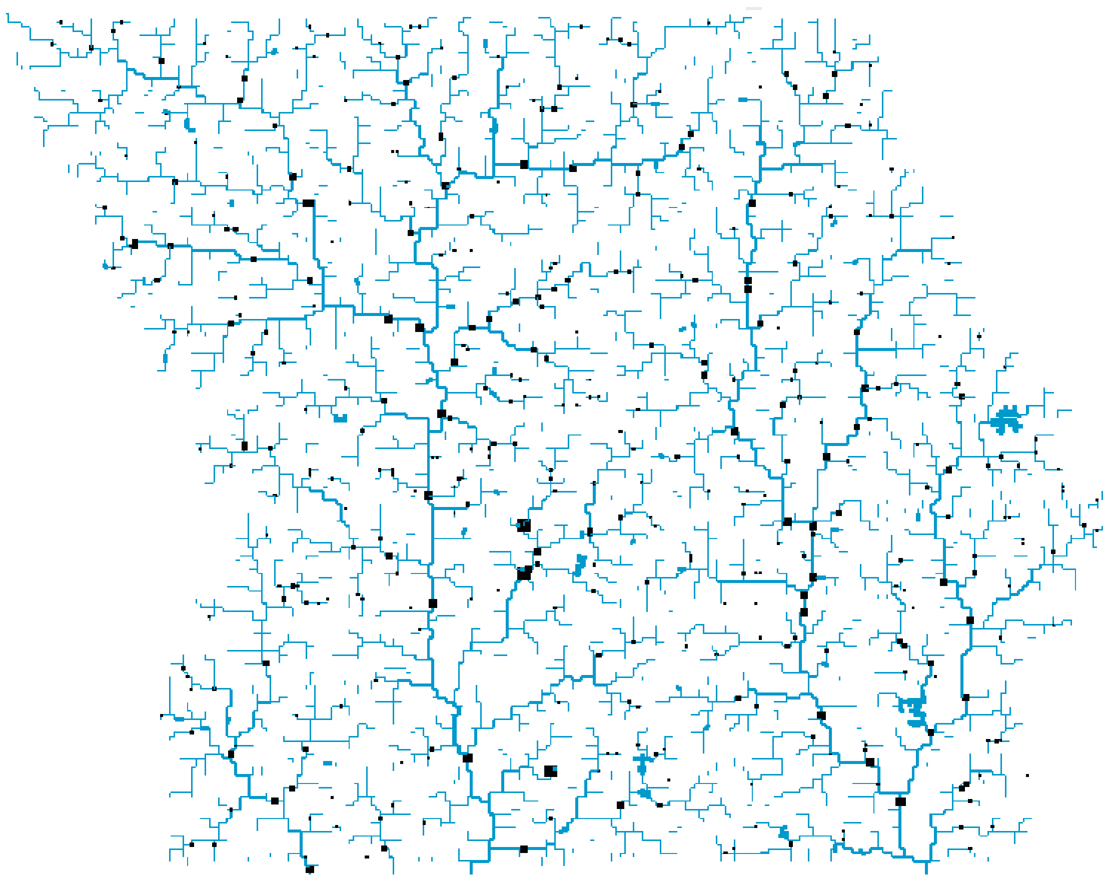


Figure 4.3. Location of dams (black squares) in the river networks

4.1.3. Life history of the Eel

Glass eel recruitment

The time series of glass eel recruitment in the simulated EMU is split into several time periods

the initial phase for populating and stabilizing the model (50 years)

a period with a constant level of glass eel recruitment (25 years)

a period of decreasing recruitment – the 'crash' (30 years)

another period of stable recruitment (20 years)

a period of increasing recruitment (25 years)

The nominal annual recruitment before the crash, i.e. in periods 1 and 2, was set at 10 million glass eels. However, year-to-year variation in glass eel recruitment was introduced into the time series by applying a log-normal stochasticity with standard deviation of 0.3, corresponding to the year-to-year variation in the observed time series of glass eel recruitment in Europe outside North Sea between 1980 and 2000 (ICES, 2011).

The decline in recruitment during the 'crash' is simulated according to an exponential coefficient of decrease set to 0.0961, based on the observed glass eel recruitment data in Europe except North

Sea during 1980-2000 (ICES, 2011). The increase in recruitment, the 'recovery', is simulated according to an exponential coefficient of increase fixed at half the coefficient of decrease.

Growth

The annual growth rate of a super eel combines an intrinsic growth rate, an age effect, a year effect (through the sum of monthly temperatures above 13°C) and a habitat effect (Daverat *et al.,* submitted). Daverat *et al.* (submitted) described habitat according to depth, relative distance to the sea and salinity. The growth rate of eels in a lake is probably lower than in a nearby river because the home ranges in lakes are much larger than in rivers (Minns, 1995). However, there are very few data on which to model such relative growth rates, so in our approach we simply considered that the growth rate of eels is one third less in lake and 10% higher in saline habitats than the rate of eels in rivers.

We assumed no differences in growth rates between undifferentiated, male and female eels, and that there was no density dependent effect on growth rates.

We used an allometric relationship between length (in cm) and weight (in gram):

$$W_i = aL_i^{\,b}$$

Parameters a and b were averaged over the relationships found in literature with a geometric mean for a and an arithmetic mean b.

Natural mortality

The glass eel natural mortality coefficient M_{glass} is fixed at 4.81 year^{-1} as estimated in Lambert (2008). This figure leads to a mortality rate of 70% during a migration season (a quarter of the year). To ensure compatibility with glass eel fishing mortality, the natural mortality is converted into km^{-1} by considering the length of the path to the settlement destination.

The natural mortality rates for elvers and yellow eels were derived from Bevacqua *et al.* (2011), assuming an exponential mortality coefficient M_i (year^{-1}) scaled with body mass W_i (g) and annual average of water temperature T (K). As Bevacqua *et al.* (2011) provided mortality estimates for Low, Average and High densities, we fitted a logistic function to predict a_M supposing that high density corresponds to 90% of carrying capacity saturation, intermediate density to 50 % and low density to 10 %. Bevacqua *et al.* (2011) did not provide any information for undifferentiated eels. Therefore, we supposed a higher value of 50.0 for the minimum of $\log(a_M)$.

Sexual differentiation and sex determinism

We considered that sexual differentiation, as the change from undifferentiated to male or female eel, depends on length but not age (Melià *et al.,* 2006) and takes place during the summer. We assumed that the proportion of fish that have differentiated before length L follows a gamma cumulative distribution function of length, with function values taken Melià *et al.* (2006), i.e. mean length at differentiation of 20.4 cm with standard deviation of 3.8 cm.

Sex determinism in European eels, i.e. becoming male or female, is considered to occur during development in response to environmental factors (Wiberg, 1983). Classically, female production is supposed to be favoured at low population densities (Davey & Jellyman 2005). The probability for an undifferentiated eel to become male was calculated with a logistic function according to the relative saturation of the reach's carrying capacity.

Silvering

Silvering, as a change from yellow stage to silver stage, is supposed to depend only on length (Vollestad 1992) and takes place during winter (Durif *et al.,* 2005). We assumed that the probability that a fish silvers before length L follows a gamma cumulative distribution function of length with mean and standard deviation different between the sexes. We fixed the minimum lengths at which eels silvered to 30 cm since silver eels below this length are rare (Vollestad, 1992). We fixed the mean silvering length parameter of female and male eels to the mean length of silver eels reported by Vollestad (1992), i.e. 62 cm and 41 cm, respectively.

Movements

In CREPE, yellow eel movements (upstream or downstream) occur during spring. After glass eel arrival (which implicitly includes an advective component within the tidal reaches), displacements are simulated as non-orientated movements among contiguous reaches. These movements are characterised by a diffusivity coefficient δ which depends on eel age a. This diffusivity coefficient is represented by a power function $\delta = 230a^{-0.8544}$, after Ibbotson *et al.* (2002).

The impact of obstacles is simulated after computation of the destination reach, i.e. after the simulation of all the basic displacements, to avoid multiple passing over the obstacle induced by a strictly Brownian movement. At each obstacle k encountered, a Bernoulli distribution with probability for upstream movement or probability for downstream movement determines the passing success in the appropriate direction. When the probability is less than the critical value the movement stops.

At the end of winter, silver eels emigrated immediately to the outlet of the catchment, then are removed from the population and counted in the main output of the model. On their way out, they experience the impact of any silver eel fishery and / or obstacles in each reach they pass through.

4.1.4. Fishing pressure

In the first year of the model run, fishermen divided up the river basin amongst themselves so that no spatial overlap between fishermen occurs, i.e. each reach is fished by only one fisherman. A fisherman can fish several reaches but only within one catchment. With 20 fishermen in the longest river network (default value), 48 fishermen are created in the simulated EMU. Each fisherman was able to target glass eels, yellow eels and silver eels.

Glass eel and silver eel fishing mortalities are simulated during the corresponding movement processes: "glass eel arrival" and "catadromous migration". The yellow eel fishing mortalities are simulated during the specific "mortality" process of yellow eels.

The fishing efforts were kept constant at a low level for the first 10 years to avoid closing fisheries when eel abundances were still low. After that period, the fishing efforts were optimised until the model reached quasi-stable efforts. Fishing pressure (effort and location for each stage) was then modified every year based on an artificial simulation of the behaviour of the fishermen in response to an annually updated cost-benefit analysis. For example, fishing effort on glass eel above the tidal influence quickly reduced to zero because of the absence of glass eel in such areas.

4.1.5. Behaviour of Scientists

The simulation of the scientists is based on the virtual ecologist concept (Berger *et al.,* 1999). They are used as the interface between the CREPE virtual word and the stock assessment models, i.e. the scientists are the source of data that are used in the application of the stock assessment models. Three scientists were created: the electro-fishing scientist, the fishery scientist and the obstacle scientist

Electro-fishing scientist

The aim of the electro-fishing scientist is to perform electro-fishing operations in a selection of reaches and to analyse the length distribution of yellow eels. We fixed the density of electro-fishing stations to 0.005 stations per km^2 as this is similar to the value observed in the French electro-fishing survey. In order to distribute the electro-fishing stations in the parts of the river basin where electro-fishing would be practical and where eels were likely to be found, a total of 200 stations were randomly distributed in reaches where the depth is lower than 2 m, the distance to sea is less than 150 km and the distance to source is greater than 5 km (Figure 4.4). The length of each fishing station was fixed at 100 m, and the width was either the width of reach for those less than 10 m wide, or to 10 m for wider larger reaches. The fishing operations took place during autumn.

We defined the reliability (combination of accuracy and precision of sampling and estimation methods) of an electro-fishing estimate of local population size as the ratio between the estimate based on the electro-fishing operation and the actual number of fish present in the fishing station.

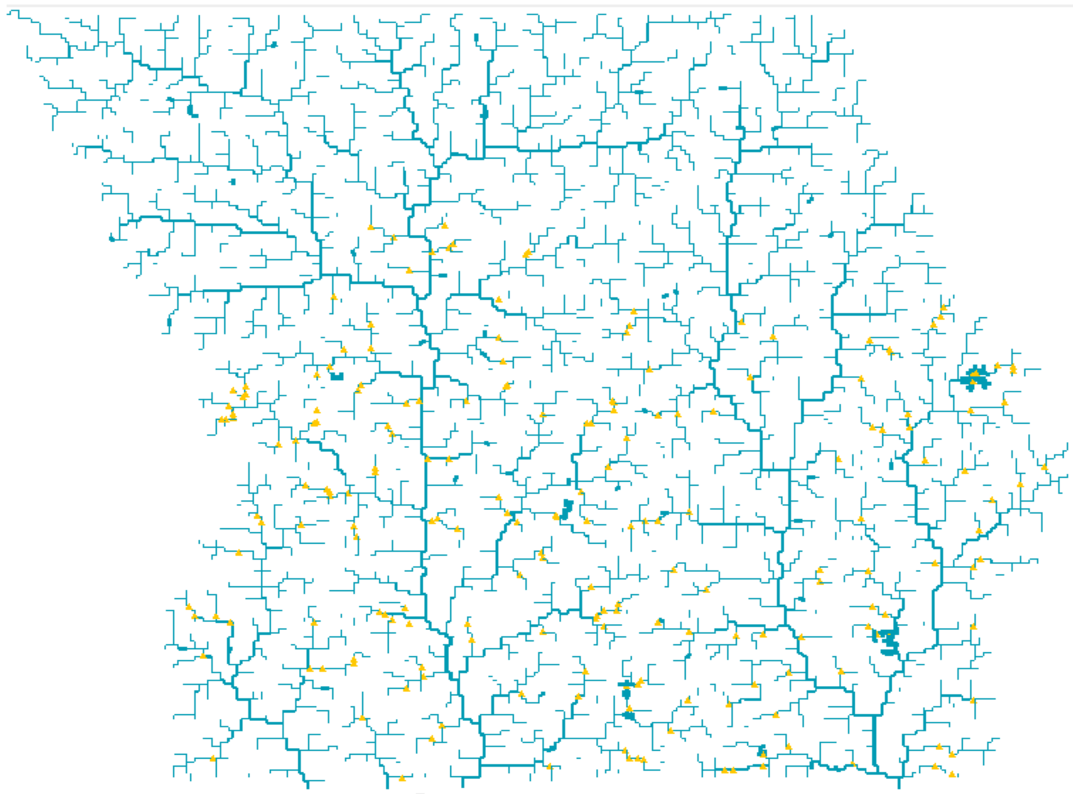


Figure 4.4. Location of electro-fishing stations (orange triangle) in the river networks

The estimate derived from the electro-fishing catch of the number of fish present in a station is then the amount of eel in a reach times the product of reliability and the ratio between the fishing station surface and the reach surface.

No selectivity according to fish length was taken into account in this electro-fishing operation simulation.

With the selected parameters, the proportion of electro-fishing operations with presence of eel was around 15% during periods of high abundance of eel (first years of simulation) and around 5% during periods of eel scarcity. These values are less than the 30% observed in the French survey during the last 20 years.

Fishery scientist

The fishery scientist gathers information from fishery records produced during the “glass eel arrival”, “standing stock mortality” and “catadromous migration” processes. The scientist calculates the total catch and effort for each fisherman on an annual basis. He also collects biological information (length, weight, gender, stage) of fish for a sub-sample of the catches.

Obstacle scientist

The obstacle scientist follows the passage of eels over the dams by analysing the records produced during the simulation of the impacts of the obstacle. He is not used for POSE simulations.

4.1.6. CREPE model run

The flow chart in Figure 4.5 illustrates the scheduling of the processes in CREPE simulation at each time step. Figure 4.6 shows more precisely the succession of biological processes for a super-eel during the four seasons of a year.

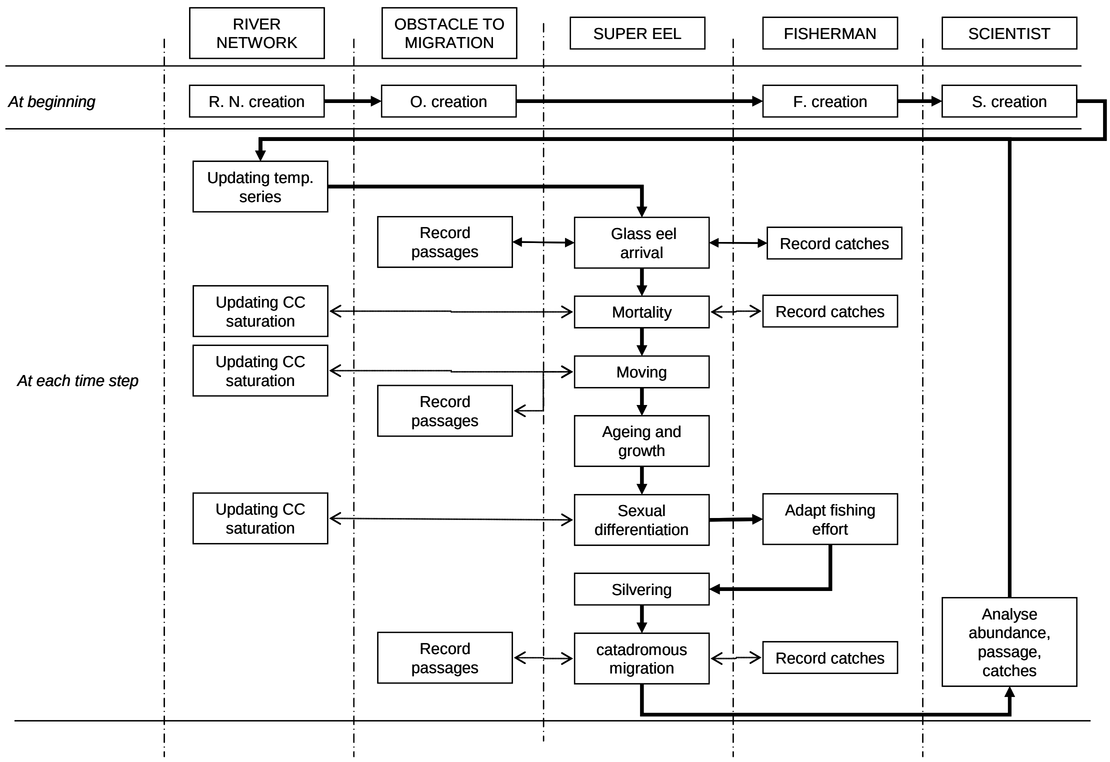


Figure 4.5. Flow chart of processes in CREPE simulation

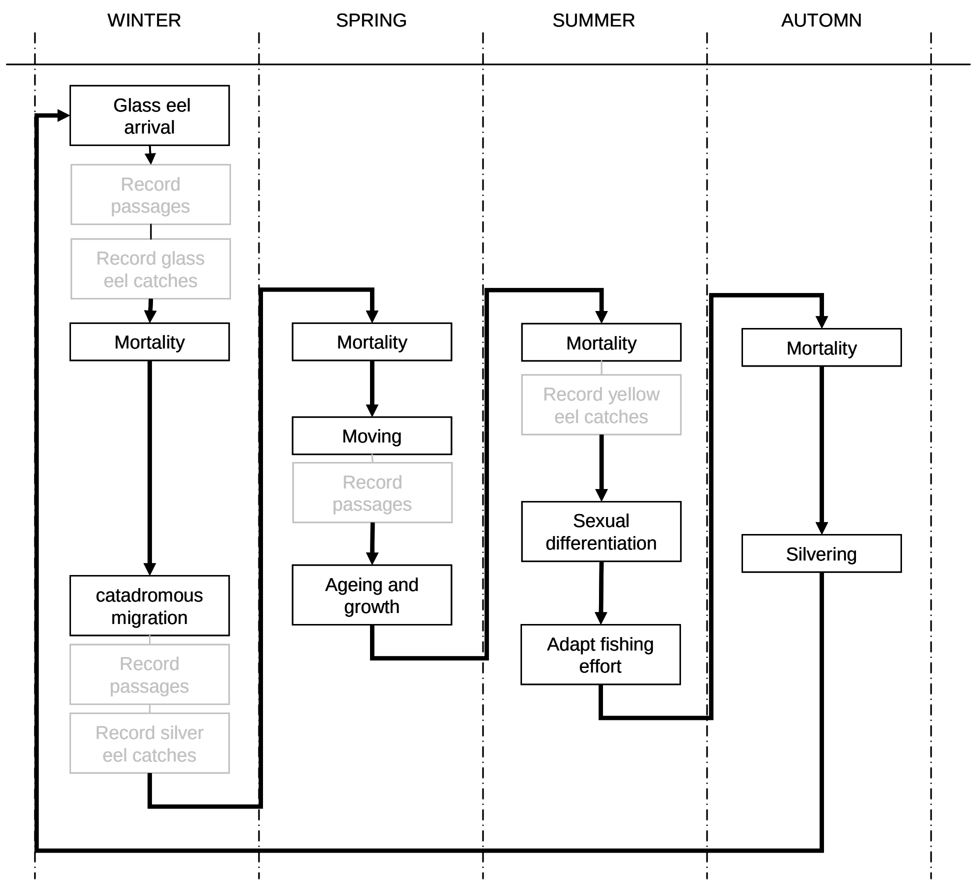


Figure 4.6. Flow chart for super-eel processes during a year of simulation (4 time steps)

4.1.7. Simulation for POSE model application

In POSE project, the aim of CREPE simulations was to produce a test dataset from a truly "data-rich" scenario, with which the other models in POSE could be tested in terms of the accuracy and precision of their estimates of the biomass and mortality indicators.

Two related time series were produced by CREPE, both reflecting a single time series of recruitment over 150 years: a period of stable and high recruitment followed by a decline and then an increase in recruitment. The "pristine" results provided the annual silver eel escapement that would have been expected in the absence of human impacts. The "current" results provided escapement from the same recruitment but impacted by the mortalities caused by fishing and turbines. The ratio of "current" escapement as a proportion of "pristine" provides the measure required for assessing local production of silver eel against the "40% target" of the Eel Regulation (EC 1100/2007). Note however that we use the term pristine here to represent best potential silver eel production under a high, but not necessarily maximum, level of recruitment and in the absence of human impacts. In other situations, "pristine" may be considered as that arising from maximum recruitment.

The same simulation was run for all POSE models to ensure results comparison. The outputs have been adapted to suit the particular requirements of each of these models. These adaptations are described in the following sections.

EDA model

Two files were written by the electro-fishing scientist for the EDA application. The first (EDASurvey.csv) corresponds to the results of the fishing operations (characteristics of electro-fishing stations and number of fish by 15 cm length classes) and is intended to calibrate EDA. The second (reachSurvey.csv) gathers the characteristics of the river networks (characteristic for all reaches) and is used by EDA to extrapolate eel density to the whole EMU.

By comparison with reality, the CREPE simulation is

poor in reach description (no elevation and no land cover) but these factors, presently not integrated in the CREPE processes, will not be good explanatory variables of yellow eel densities in this virtual world,

rich in the knowledge of the impact of dams because the average mortality during downstream migration is well known for all the reaches, and

optimistic since the electro-fishing surveys were 'conducted' at the same locations throughout the time series, and with constant capture efficiency.

SMEP II

Although SMEP II does allow the user to define a very complicated branching network of river reaches, this is very time consuming in terms of preparing the data for the model and in exploring the results. Furthermore, there are typically few electro-fishing surveys throughout the river network in reality, and the results of these surveys are extrapolated to the nearby 'empty' reaches. Therefore, in 'real' applications of SMEP II, the network of river reaches is usually compressed into a smaller number of zones containing one or more electro-fishing surveys, fisheries or other human impacts.

To this end the CREPE simulated data were gathered by stretches which correspond to the collection of reaches fished by one fisherman (therefore within a single catchment). Catchments were simplified to a linear succession of stretches since the list of reaches fished by a fisherman is determined according to the distance to sea.

Four files were produced by the electro-fishing scientist and the fishery scientist:

stretchSurvey.csv summarizes the characteristics of the stretches

electrofishingSurvey.csv gives the results of electro-fishing operations (the same survey as the one for EDA application)

fisherySurvey.csv presents the catch and effort for the fisherman of each stretch, and

fisheryBiologicalSurvey.csv summarizes the biological results from the fishery survey

The SMEP application also requires:

the trend in recruitment (arrival.csv)

the evolution or the year effect (temperatureSerie.csv)

DemCam

As DemCam is functionally similar to SMEP II, albeit for a single spatial zone, the same datasets are suitable.

GEM

The GEM model is very different from the others tested in POSE, such that in reality there are very few datasets to which it could be applied outside of the original German data and the Burrishoole in Ireland. However, as the CREPE output provides comprehensive training data on time series of recruitment and silver eel escapement, and on eel life history processes, the CREPE outputs are suitable for which to test GEM.

Final comparison

To measure the reliability, accuracy and precision of the four models considered in the POSE project, the time series of silver eel escapement series was also recorded from the CREPE simulation, but not provided to the modellers in POSE.

4.2. Description of the application of the models to the CREPE data

Here we describe how the EDA, GEM, SMEP II and DemCam models were applied to the CREPE data set.

ToR 6 required the “description of improvements made to models to adapt them to the biology, ecology, chemistry and geography of the specific eel habitat to which they are applied”. It was neither necessary nor appropriate to alter the manner in which the models functioned, i.e. to improve the models, in order to apply them to the various eel data sets. However, they were all modified on a case-by-case basis by the application (input) of eel data specific to each data set. The descriptions of the various model applications, in this chapter and the next, detail the manner in which each model was parameterised (modified) to suit the data set, including the analysis and processing of input data where appropriate, and report and discuss the results of these applications.

The model testing procedures were conducted in two phases. In the first phase, input data were derived from the description of CREPE (section 4.1 and Annex B) and from the data files provided for each model. The silver eel escapement biomass predictions by the four models were compared to those from CREPE, which had been withheld from users to create a “blind” test. This phase simulated a “data not quite rich” scenario, but probably the best that could be expected in the ‘real’ world.

In the second phase, the input data were adjusted or ‘tuned’ to improve the fit between the predictions of silver eel biomass and the actual results from CREPE. ‘Tuning’ is the systematic revision of model parameter values to produce a model output as close to desired as possible. This illustrates the ideal data-rich situation from which to examine the potential accuracy and precision of the assessment model. However, as such a truly data-rich situation does not exist in the real world, the results represent the best possible but not necessarily what managers should expect in other situations. Note that as the models vary considerably in input data requirements and in outputs, we have not attempted to report their application to the various data sets in a standard format, here or in Chapter 5.

4.2.1. Application of EDA to the CREPE data

Introduction

EDA (Eel Density Analysis) is a framework of eel density analyses rather than an end-user model. EDA first relates eel densities to environmental variables, including anthropogenic impacts and then extrapolates from survey sites to the EMU. The predicted yellow eel stock is then converted to a silver eel escapement, using a conversion rate.

The modelling tool is based on a geo-localized river network database and design to predict yellow eel densities and silver eel escapement.

There are six main steps in the model application:

Relate observed yellow eel presence/absence and densities to descriptor parameters

Extrapolate yellow eel density in each river stretch by applying the statistical model calibrated in step 1

Calculate the overall yellow eel stock abundance by multiplying these densities by the surface of each stretch

Estimate a potential silver eel escapement of each stretch by converting yellow to silver eel abundance with a 5% rate

Calculate effective escapement by reducing potential escapement with mortalities during downstream migration

Sum the effective escapement from all the stretches to give estimate at EMU scale

Input data

The model requires data about abundance of yellow eel. The data from CREPE were extracted from the electro-fishing surveys (see Description for CREPE model, Annex B), yielding data from 200 sites sampled during the autumn of years 51 to 149 and resulting in 19800 discrete data points (Figures 4.7 and 4.8). The density (d) was calculated for all electrofishing operations and expressed in number/100m².

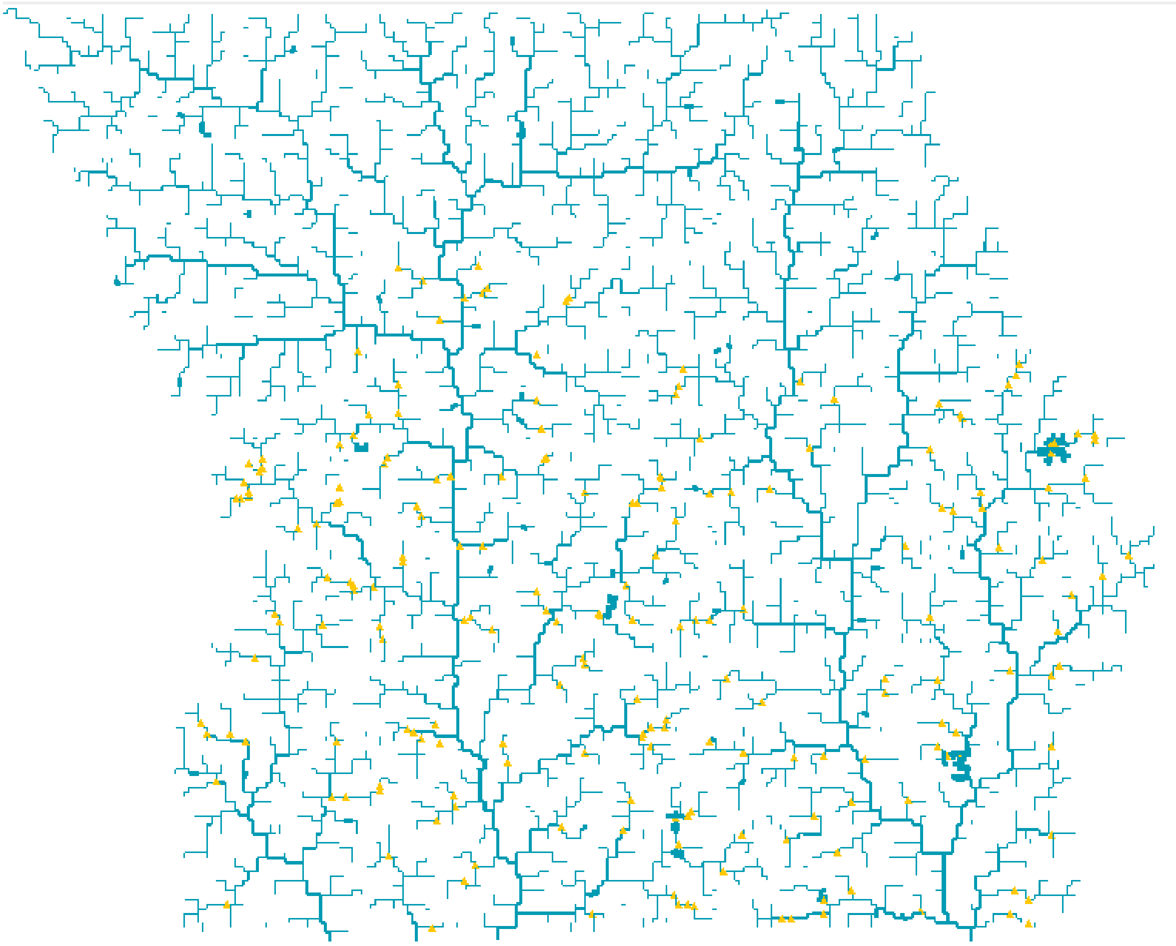


Figure 4.7. Sampling sites (in yellow) in the CREPE catchment on the river network (in blue).

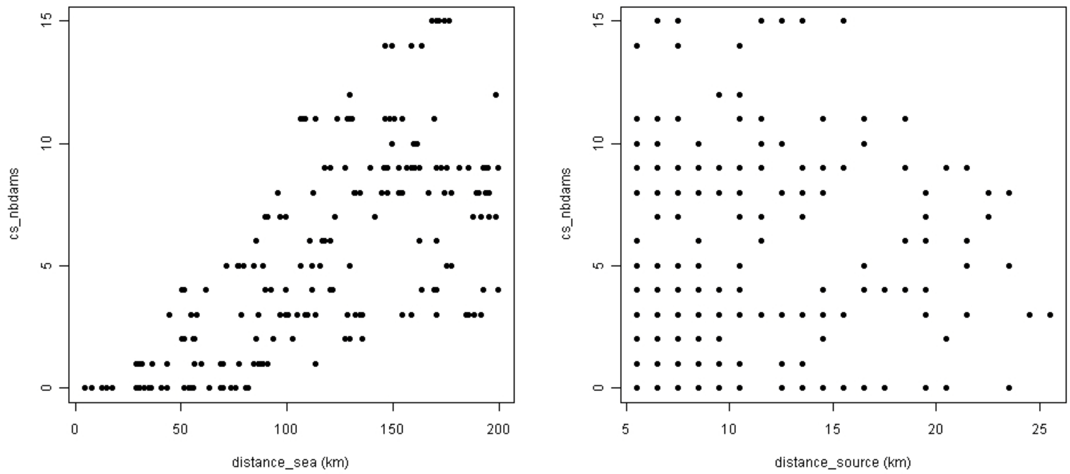


Figure 4.8. Sampling sites per distance from the sea or distance from the source and the cumulative number of dams.

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CREPE data set provided information for each survey site on the distances to sea and source, relative distance, the area of land drainage upstream, and the number of dams downstream of the site. The cumulative number of dams from sea to site was used to characterise the obstacle pressure to upstream migrating eels. In total there were 400 dams distributed throughout the CREPE EMU.

A combination of these 6 variables (year and a set of 5 variables) was tested. Figures 4.9 and 4.10 show that there are several variables with tightly correlated predictors. Therefore, according to the threshold of 0.5 for ρ^2, the following variables should be grouped:

relative_distance, distance_sea and cs_nbdams

up_area, distance_source and relative_distance

The combination of these different groups provided 20 models to be tested.

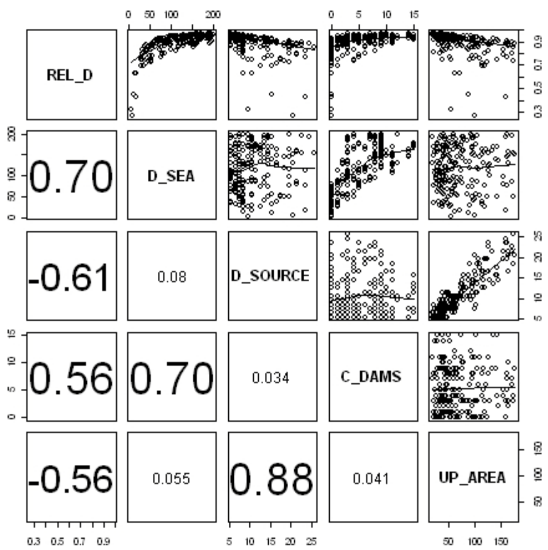


Figure 4.9. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, C_DAMS: cumulative number of dams, UP_AREA: upstream catchment area. The font size of the cross-correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

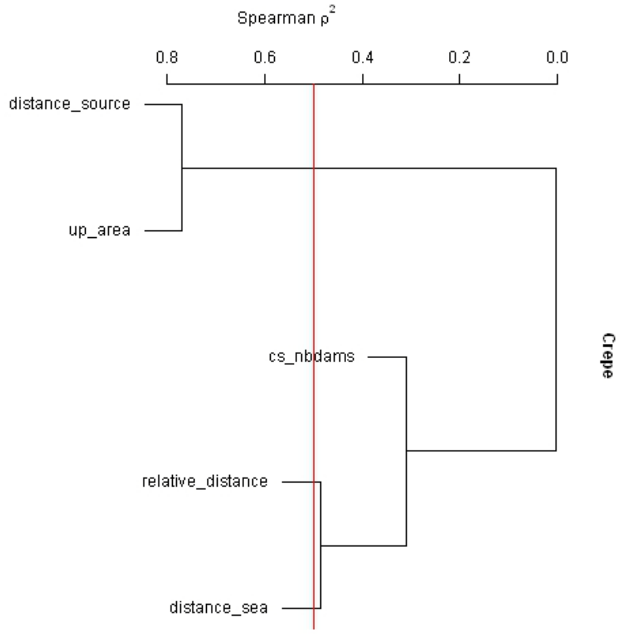


Figure 4.10. Dendrogram obtained by hierarchical cluster analysis of 19 candidate predictors for CREPE dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ^2=0.5.

Model Testing Procedure

We calibrated statistical models with a Generalized Additive Model (Hastie and Tibshirani, 1990) to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/). EDA model is based on a delta-gamma model (Stefánsson, 1996) which combines two generalized additive model (GAM). There are three steps of modelling:

a presence/absence model (delta model) based on a GAM with a binomial distribution and a logit link to determine the probability of a positive catch;

a density model with the positive data (gamma model) using a GAM with a gamma distribution and logarithm link to determine the level of positive catch; and,

the multiplication of the two previous models (delta-gamma model).

The GAMs were computed with the library gam (Hastie, 2010) with a cubic spline smoother (3 degree of freedom) for each environmental variable. The delta-gamma (ΔΓ) generalized additive models explain a large portion of the variability in eel abundance data, as there are many occasions where densities are null.

Delta (Presence-absence) model

Model results are summarized in Table 4.1. The best density model is selected by the Akaike's Information Criterion (AIC) with a lower AIC indicating a better fit (Akaiïke, 1974, Sakamoto *et al.,* 1986). The AIC function for the CREPE analyses indicates a better fit of the response variable at the year of electrofishing sample with the distance to the source and the distance to the sea. GAM explained 31% of the deviance of the abundance of the yellow eel. The three explanatory variables are significant (Table 4.2): year, distance from the sea and distance from the source.

Table 4.1. Model selection results using Akaike's information criterion (AIC) for presence-absence model analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

model	year	distance_sea	distance_source	relative_distance	cs_nbdams	up_area	AIC s=3
1	x	x	x				10795.4
2	x	x				x	10824.4
3	x	x					10860.3
4		x	x				11057.4
5			x			x	11085.7
6		x					11120.5
7	x			x			12234.2
8				x			12479.0
9	x				x	x	12760.5
10	x		x		x		12780.3

Model Goodness of fit (delta model)

The best presence-absence model selected (a binomial model with a logit link and a cubic spline smoother – s=3) is:

d != 0 ~ s(annee,3)+s(distance_source,3)+s(distance_sea,3)

```
matrice confusion

         observed

predicted    1     0

        1  1122   851

        0  1551 16276

correctly predicted 0.88

present correctly predicted 0.42

absents correctly predicted 0.95

kappa with a threshold=0.4

      Kappa   Kappa.sd

1 0.4160392 0.00975904
```

Table 4.2. Table of effects, DF for terms and Chi-squares for non-parametric effects

```
                      Df Npar Df Npar Chisq    P(Chi)

(Intercept)            1

s(annee, 3)            1       2     90.686 < 2.2e-16 ***

s(distance_source, 3)  1       2     23.209 9.123e-06 ***

s(distance_sea, 3)     1       2     66.984 2.887e-15 ***

---
```

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

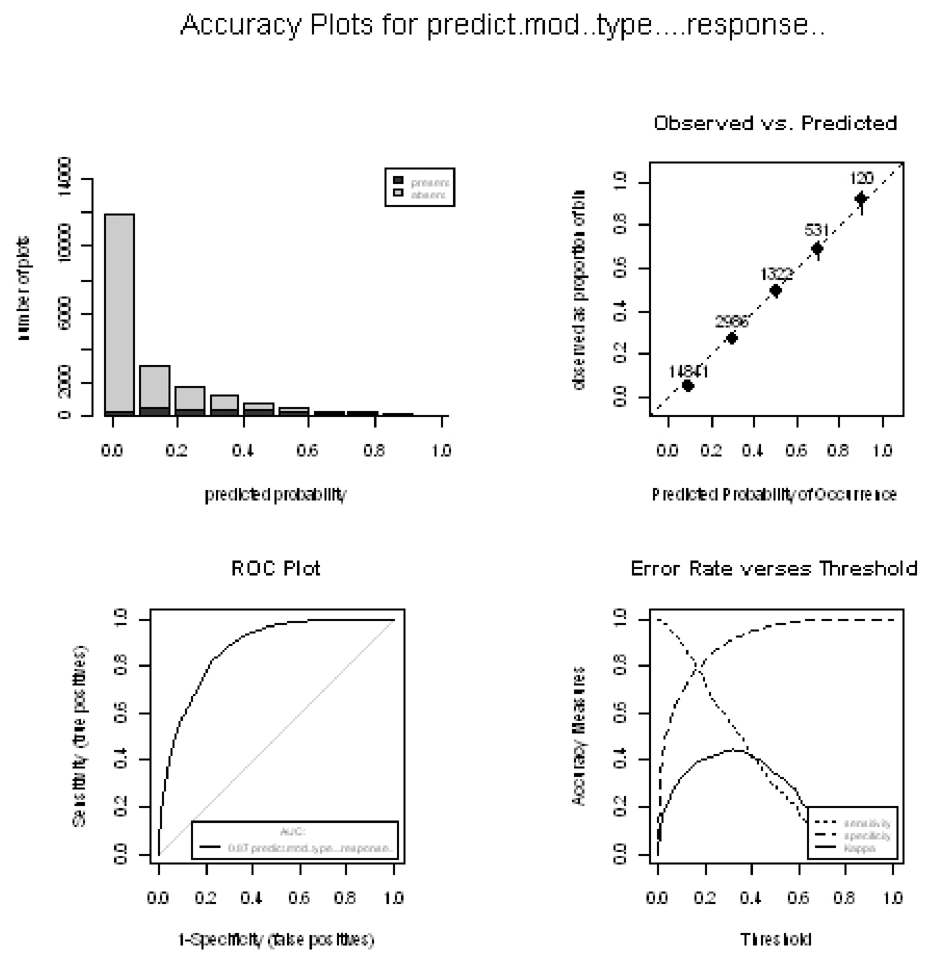


Figure 4.11. Model quality and threshold selection graphs for presence-absence model selected with a histogram plot (upper left), a calibration plot (upper right), a ROC plot with the associated Area Under the Curve (AUC) (lower left), and an error rate verses threshold plot (lower right).

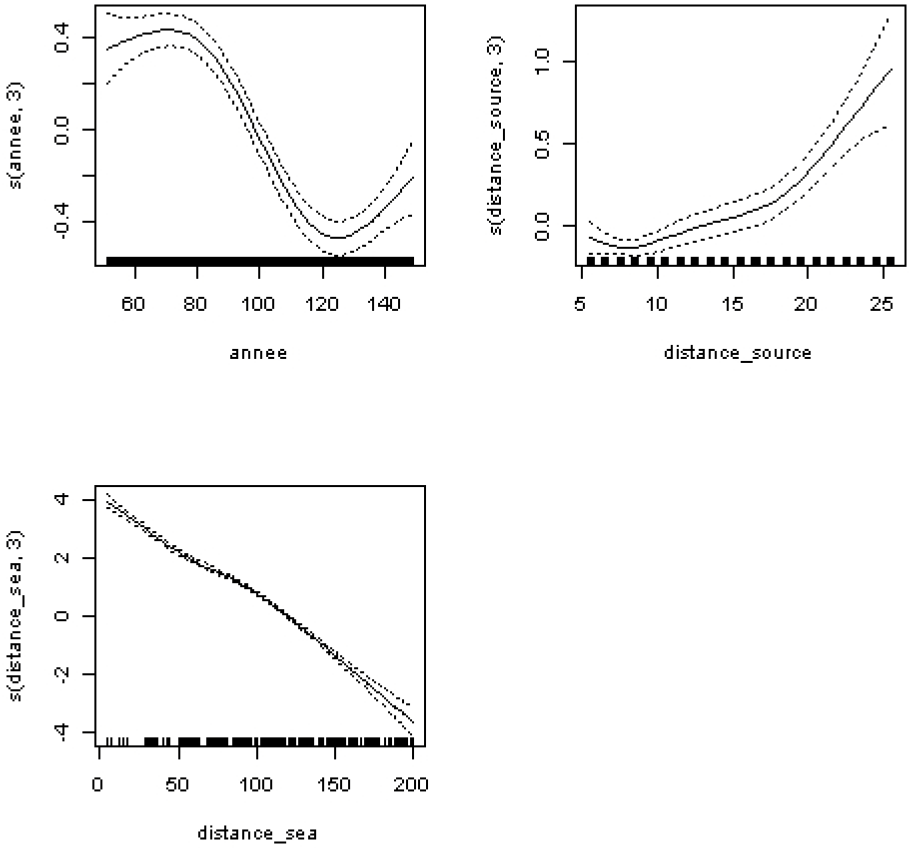


Figure 4.12. Response curves of each variable included in the generalized additive model (GAM) for the presence/absence model for CREPE EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Density model

The eel density model is applied only to those sites where eel were caught, i.e. sites with positive values (>0). It is a gam with a log link. The model results are summarized in Table 4.3. The AIC function indicates a better fit of the response variable at the year of electrofishing sample with the distance from the sea and the upstream catchment area (up_area). GAM explained 41% of the deviance of the abundance of the yellow eel.

The effects of the three explanatory variables in the model (year, up_area and distance_sea) are significant (Table 4.4). The Spearman rank correlation between the observed values and the fitted values is statistically significant ($\rho = 0.527$).

Table 4.3. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

model	year	distance_sea	distance_source	relative_distance	cs_nbdams	up_area	AIC s=3
1	x	x				x	50208.5
2	x	x	x				50317
3	x				x	x	50340.1
4	x		x		x		50353.3
5	x				x		50364.2
6	x	x					50373.8
7	x			x			50582.7
8	x					x	50815.4
9	x		x				50857.5
10	x						50867.6

Table 4.4. Table of effects, DF for terms and Chi-squares for non-parametric effects

	Df	Npar Df	Npar F	Pr(F)	
(Intercept)	1				
s(annee, 3)	1	2	149.971	< 2.2e-16	***
s(up_area, 3)	1	2	17.049	4.399e-08	***
s(distance_sea, 3)	1	2	24.902	1.930e-11	***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

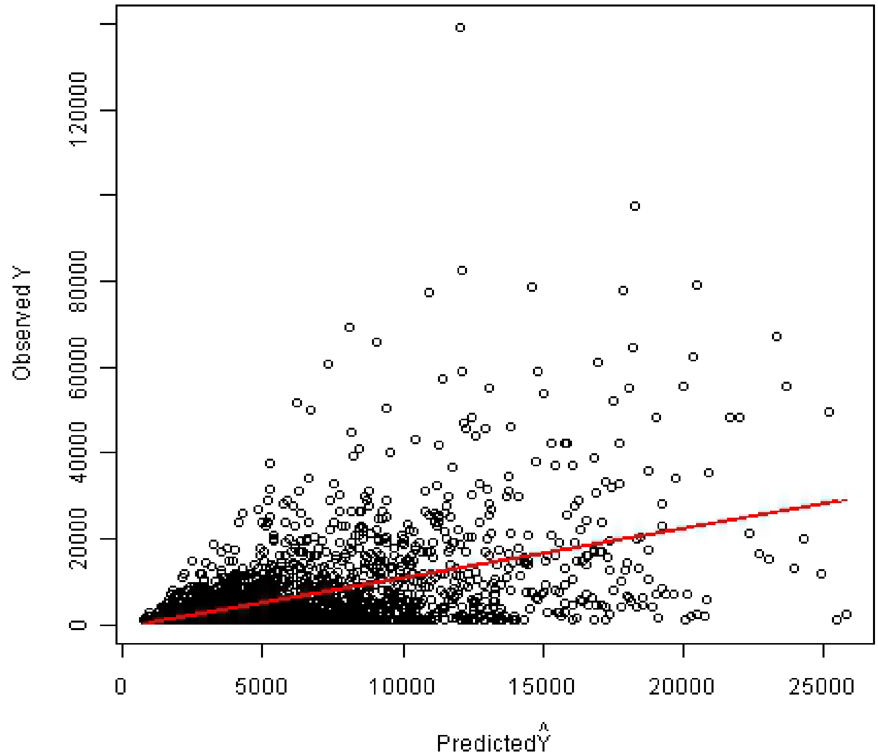


Figure 4.13. Observed vs. predicted regression scatter plot.

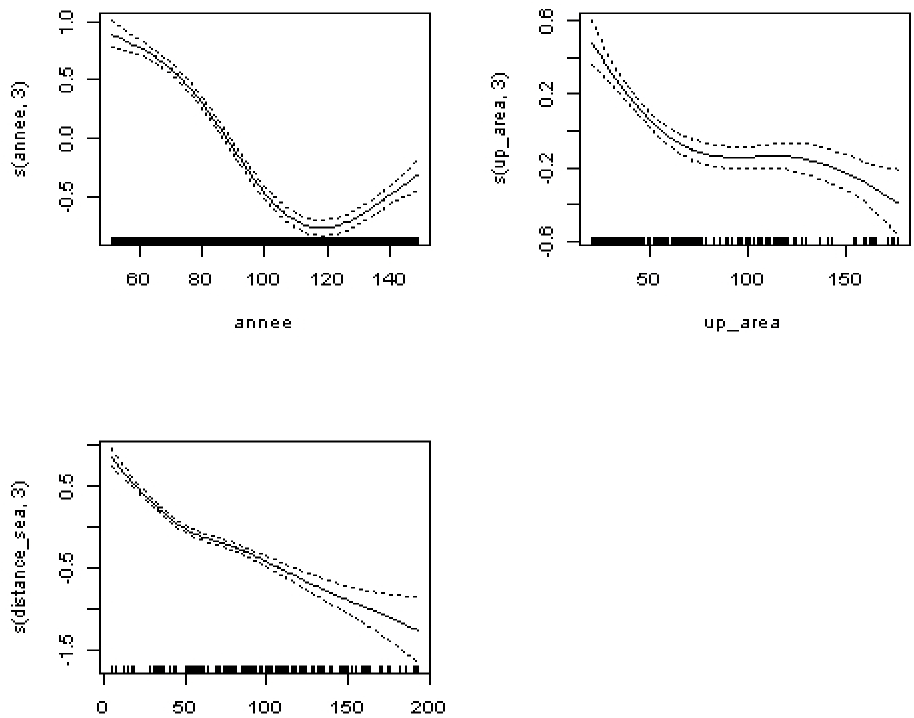


Figure 4.14. Response curves of each variable included in the generalized additive model (GAM) for the density model for CREPE EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

The best density model selected is: d ~ s(annee,3)+ s(distance_sea,3)+s(up_area,3.

EDA predictions for CREPE

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel in a stretch is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch by a yellow-to-silver eel conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou, 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al.* (2000) estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA, a constant conversion rate of 5% was applied, based on the assumption of no density-dependent influences on biological processes.

This potential escapement in number ($N_{current}$) is then converted into biomass using the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for CREPE.

The best achievable escapements (B_{best}) can be calculated by raising the current biomass artificially forced to zero anthropogenic impacts (no dam, land use mortality to "no impact" and silver eel catch to 0), and accounting for the silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historic level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment.

The model parameters used to evaluate $B_{current}$, B_{best} and $B_{pristine}$ are given in Table 4.5. Table 4.6 summarises the EDA estimates of the total numbers of yellow and silver eels in the CREPE EMU, along with estimates of $B_{current}$, B_{best} and B_0. Figure 4.15 presents the predicted time series of silver eel escapement during years 51 to 149 in the CREPE data set.

Table 4.5. Data input for silver eel estimation for CREPE dataset.
With $Y_{glass}(t=100-\tau)$, $Y_{yellow}(t=100-\tau+\lambda_{yellow})$ and $Y_{silver}(t=100)$ in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\bar{w}_{glass}$ (g)	$\bar{w}_{yellow}$ (g)	$\bar{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel	5.939	4.886 (in t=100)	0.448 (in t=100)	206.54 (in t=100)	448.04 (in t=100)	41 (in t=94)	1408 (in t=98)	1254 (in t=100)

during ¼ year and 0.1386 after (Lambert, 2008)								

Table 4.6. Summary of EDA predictions of yellow and silver eel production from the CREPE EMU in the final year of the data set, and for historic B_0

Model output	Value
Total water surface (km²)	179.18
Average number of yellow eel per 100 m²	$16.012\ 10^{-3}$
Average number of silver eel per 100 m²	$8.01\ 10^{-4}$
Total number of yellow eel	28 690
$N_{current}$: Total number of silver eel	1 434.52
$B_{current}$ (kg)	642.59
$\mathbf{B_{best}}$ (kg)	644.66
B_0 (kg)	2656.6

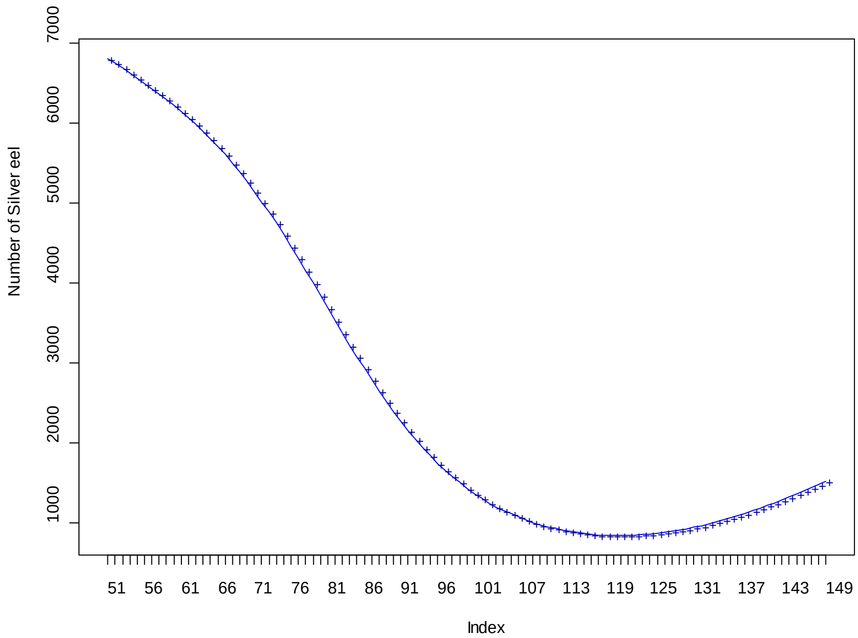


Figure 4.15. Evolution of $N_{current}$ (total number of silver eel) for the CREPE data set during the period from year 51 to year 149

The comparison between the model output of EDA and CREPE shows a bias in the EDA predictions corresponding to an underestimation of the biomass (half of this bias is probably due to an underestimation of the wetted area of the catchment). The general trend is predicted by EDA model but the beginning and the end of a period of stable escapement are difficult to detect. The underestimated yellow eel production is the consequence of a lack of sampling in the largest and most downstream reaching, the sampling stations are not homogeneously distributed but this probably reflects reality. Furthermore, a correct estimation of the wetted area is required; we used the formula given by McGinnity *et al.* (*in press*) to predict river width of a reach and to calculate the wetted area but this formula cannot be used with lakes and the CREPE dataset included lakes with no way to list them.

Changes made during the tuning process for the application of EDA to the CREPE dataset

A new CREPE dataset was used during the tuning process, providing the time series of silver eel escapement withheld during the first application phase and reducing the bias detected in the first application to enhance the EDA predictions.

The model parameters used to evaluate $B_{current}$, B_{best} and $B_{pristine}$ were changed (Table 4.7) because mean weight of yellow eel $\overline{w}_{glass}$ and silver eel $\overline{w}_{silver}$ show strong variations between years, and anthropogenic mortalities on glass eel Y_{glass}, yellow eel Y_{yellow} and silver eel Y_{silver} varied from year to year. To calculate the potential escapement in number ($N_{current}$), the time series of mean weight per year of each life stage ($\overline{w}_{glass}$, $\overline{w}_{yellow}$, $\overline{w}_{silver}$) have been used; these chronological series derived from the biological survey of silver eel fishery provided for CREPE. The current escapement is finally calculated by subtracting these time series of silver eel catches (and not the mean silver eel catches for the whole period) to this potential escapement in biomass. In the present application, mortalities induced by turbines are not included. The best achievable escapements (B_{best}) is calculated by raising the current biomass artificially forced to zero anthropogenic impacts (no dam, land use mortality to "no impact" and silver eel catch to 0), and accounting for the silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010). In this second application phase, the time series anthropogenic mortalities on glass eel and yellow eel have been used. Furthermore, to reduce the bias from the estimation of the wetted area, we also used directly the wetted area given in the new CREPE dataset.

Table 4.8 summarises the EDA estimates of the total numbers of yellow and silver eels in the CREPE EMU, along with estimates of $B_{current}$, B_{best} and B_0. Figure 4.16 presents the predicted time series of silver eel escapement during years 51 to 149 in the new CREPE data set.

Table 4.7. Data input for silver eel estimation for CREPE dataset.
With Y_{glass}(t= $year - \tau$), Y_{yellow}(t= $year - \tau + \lambda_{yellow}$) and Y_{silver}(t= $year$) in kg from year 51 to year 149.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	6.16	range from 1 to 5.68 (4.857 in t=100)	0.448	range from 15.8 to 271.4 (210.54 in t=100)	range from 109.6 to 603.8 (477.11 in t=100)	range from 46.7 to 4489 (393 in t=94)	range from 5.9 to 3923 (1409 in t=98)	range from 0 to 3622 (1000 in t=100)

Table 4.8. Example of EDA predictions of yellow and silver eel production from the CREPE EMU in year 100, and for historic biomass B_0.

Model output	Value
Total water surface (km²)	407.80
Average number of yellow eel per 100 m²	0.011

Average number of silver eel per 100 m²	5.4 10^{-4}
Total number of yellow eel	44056
$N_{current}$: Total number of silver eel	2203
$B_{current}$ (kg)	157
B_{best} (kg)	8254
B_0 (kg)	67624

Points to consider regarding the model application

The use of the default silvering rate of 5% leads to a negative escapement (Figure 4.16), i.e. the potential escapement is lower than the silver eel yield for the year 102. A rate of 10% or 15 % solves the problem and no more negative values are calculated. However a deeper exploration of the CREPE results indicates an average silvering rate of about 5%, but the abundance of the standing stock in CREPE is 1.7 times higher than that calculated by EDA. The explanation of this difference could be an underestimation of eel densities in the downstream, deep water area of the river basin, where electro-fishing operations are not feasible and therefore where densities are extrapolated from the upstream river stretches. A silvering rate of 10% (the double of the true value) was used to overcome this problem (Figure 4.17).

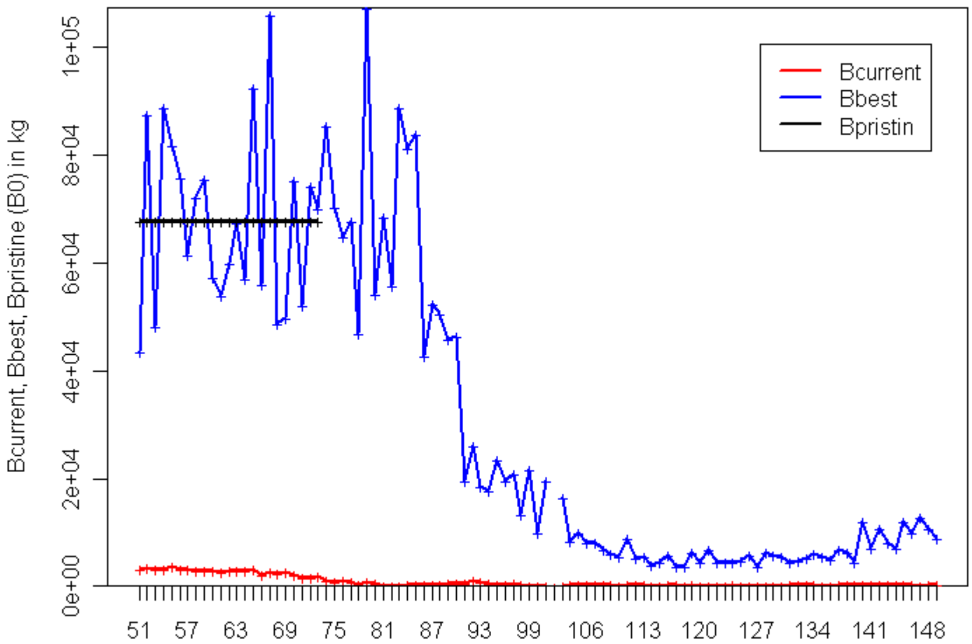


Figure 4.16. Evolution of $B_{current}$, B_{best} for the CREPE data set during the period from year 51 to year 149; $B_{pristine}$ is calculated from year 51 to year 74 with default value of silvering rate (5%).

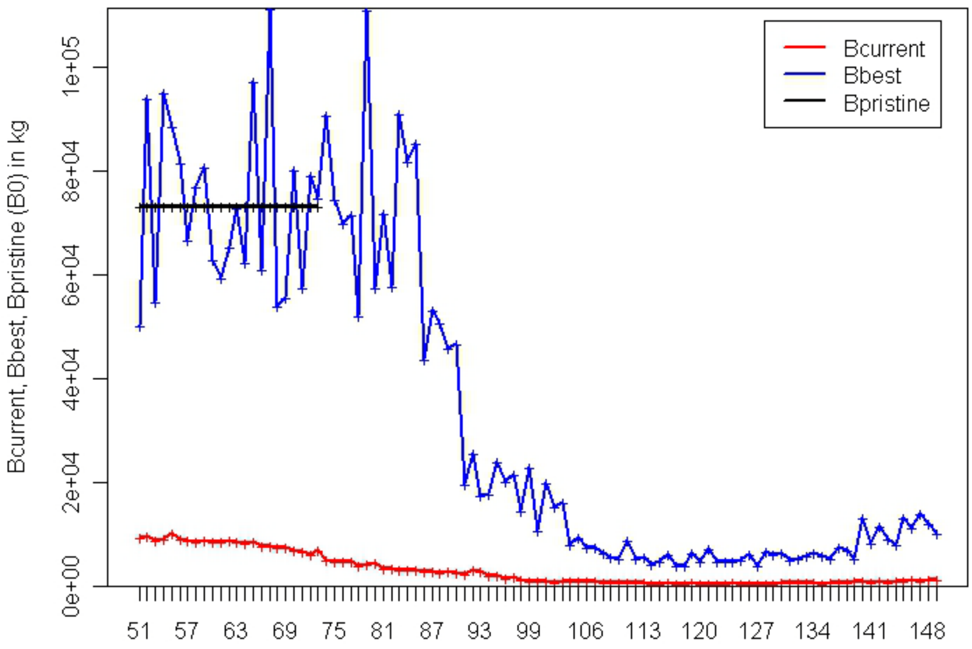


Figure 4.17. Evolution of $B_{current}$, B_{best} for the CREPE data set during the period from year 51 to year 149; $B_{pristine}$ is calculated from year 51 to year 74 with a value of silvering rate of 10%.

With this correction the EDA current biomass is in the same order of magnitude as the escapement simulated by CREPE. The general trend is respected even if the stable period is difficult to identify (Figure 4.18). The degree of smoothness of model terms could have been increased (with a degree of smoother s = 4, 5, or larger) to fit more closely to the CREPE current biomass. As a conclusion, combined delta-gamma generalized additive models are useful in the analysis and explained a considerable portion of the variability in the current biomass, but a fairly large number of degrees of freedom is required to describe the proper behaviour of the data.

On the contrary, the pristine biomass is clearly overestimated by a factor of about 3. The ratio B current / B pristine is very pessimistic. This is precautionary, but social and economic impacts are likely to be unacceptable. This overestimation is the consequence of no density dependence regulation hypothesis in EDA that leads to underestimate the natural mortality in high densities like in pristine conditions.

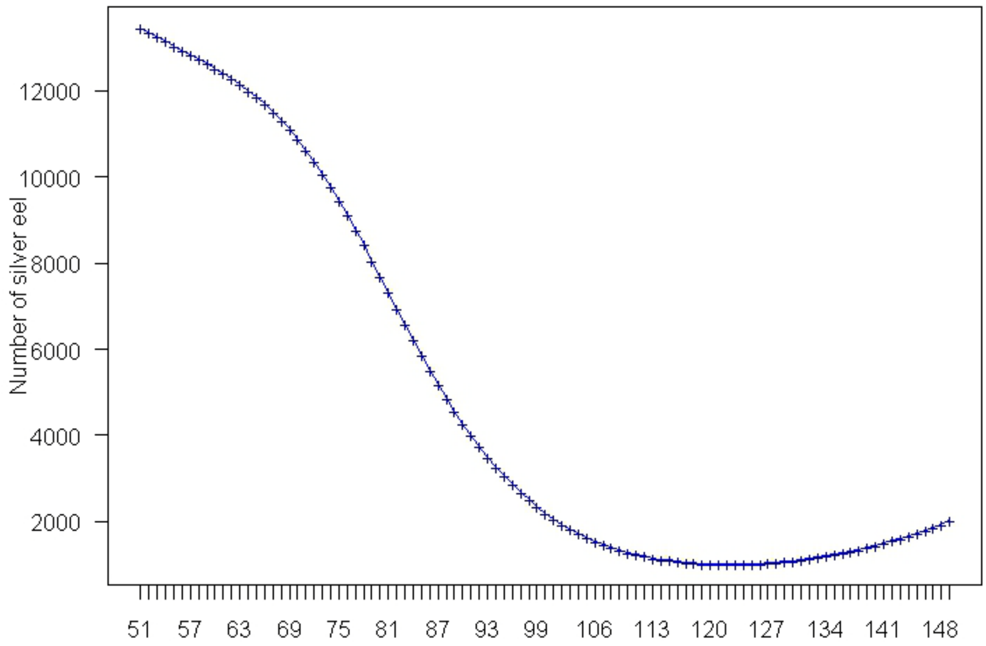


Figure 4.18. Evolution of potential silver eel escapement in number ($N_{current}$) for the CREPE dataset during the period from year 51 to year 149

4.2.2. Application of GEM to the CREPE data set

The aim of the application was to test the precision and accuracy of the adapted GEM in estimating the silver eel escapement from the CREPE data set, based on the period from years 49 to 149 in the virtual, data-rich scenario. The following input data were derived from the CREPE report (Annex B) and analysis of the CREPE output data files supplied by the virtual electro-fishing and fishery scientists (see above).

The total wetted area of the EMU was 990,100 ha;
The absolute recruitment per year was available for the total period;
Eels were simulated at ages 0 to 20 years;
The natural mortality of male and female eel was estimated based on the method described by Bevacqua et al. (2011), where it was assumed that the eel density is low and that the mean water temperature is 12.6 °C;
The weight length relationship, and growth rates of male and female eel were estimated by means of the electro-fishing and fishery data, where:

Weight – length relationship: $W = 0.00000092\ L^{3.18}$
Growth of males: $L = 52.7 * (1 - e^{(-0.398*(Age + 0.166)})$
Growth of females: $L = 69.6 * (1 - e^{(-0.366*(Age - 0.09)})$

The proportion of silver eel by length and sex was estimated based on the description of CREPE;
Catch in kg per year was given for glass, yellow and silver eels, and the length-based selectivity of fishing gears was defined as described in Annex B; and,
The mortality rate for silver eels passing turbines was assumed to be 25% per annum throughout the time series.

The GEM was developed to describe the dynamic of eel population in the river Elbe system. The use of the model for the CREPE dataset required the following modifications.

Whereas a constant proportion of male silver eel (5%) was used in the original GEM application to the Elbe data, a variable proportion of male eels was simulated in the CREPE application, albeit with a maximum proportion of 0.5. Therefore, models for male and female eel were separately prepared but linked by different parameters – Tables 4.9 and 4.10 present the age-based parameter values for male and female eels, respectively. The apparent increase in length between glass eel and age 1 elvers was particularly high (13 to 23 cm). Eels in Europe show a wide range growth curves (e.g. Berg, 1985; Vøllestad & Jonsson, 1992; Simon, 2007).

The separated models took into account the sex-specific growth of male and female eel, and the different development from yellow eel to silver eel. In addition, the impacts of recreational angling and cormorant predation were switched off, because these factors were not reported for the CREPE data set.

Table 4.9. Age based data for male CREPE eel applied in GEM

Age	Mean weight per age [g]	Mean length per age [cm]	Nat. Mortality low density	Relative age distribution first year	Relative catchability by age group	Relative proportion of silver eel
0	0.4	7.0	22.3	1.000	0.00	0.00
1	48.3	30.5	13.2	1.000	0.18	0.00
2	95.8	37.8	9.3	0.696	0.94	0.05
3	141.3	42.7	7.3	0.598	1.00	0.31
4	179	46.0	5.9	0.493	1.00	0.68
5	207.9	48.2	4.9	0.381	1.00	0.87
6	229	49.7	4.3	0.252	1.00	0.94
7	243.9	50.7	3.8	0.172	1.00	0.96
8	254.3	51.4	3.4	0.113	1.00	0.97
9	261.5	51.8	3.1	0.071	1.00	0.99
10	266.4	52.1	2.9	0.051	1.00	0.99
11	269.7	52.3	2.8	0.036	1.00	0.98
12	271.9	52.5	2.6	0.024	1.00	0.98
13	273.4	52.6	2.5	0.017	1.00	0.99
14	274.4	52.6	2.3	0.011	1.00	0.99
15	275.1	52.7	2.3	0.007	1.00	0.99
16	275.6	52.7	2.2	0.004	1.00	1.00
17	275.9	52.7	2.1	0.002	1.00	1.00
18	276.1	52.7	2.1	0.001	1.00	1.00
19	276.2	52.7	2.0	0.001	1.00	1.00
20	276.3	52.7	1.9	0.000	1.00	1.00

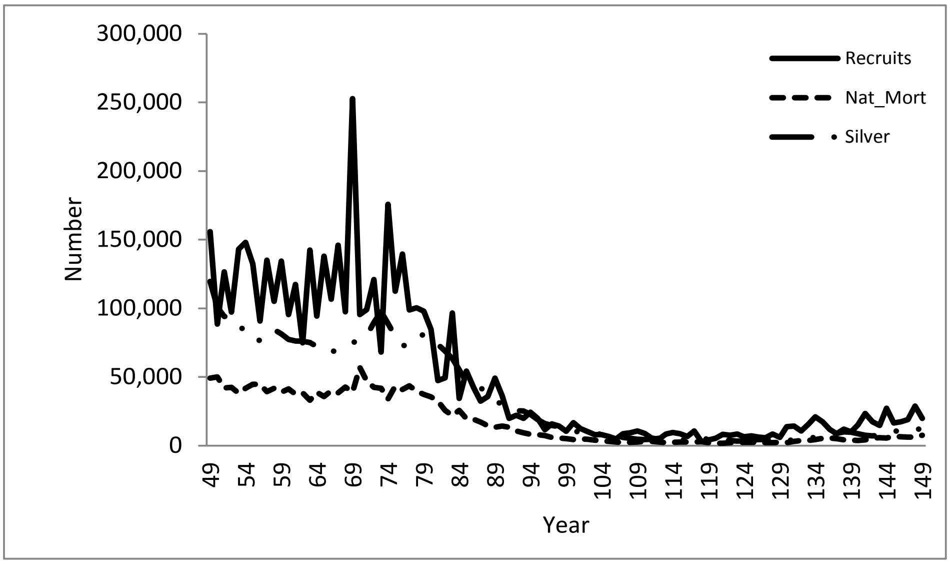
Table 4.10. Age based data for female CREPE eel applied in GEM

Age	Mean weight per age [g]	Mean length per age [cm]	Nat. Mortality low density [%]	relative age distribution first year	Relative catchability by age group	Relative proportion of silver eel
0	0.4	7.0	34.7	1.000	0.00	0.00
1	12	19.7	23.2	1.000	0.25	0.00
2	74.9	35.0	16.7	0.696	0.99	0.01
3	174.2	45.6	13.2	0.598	1.00	0.07
4	280.5	53.0	10.8	0.493	1.00	0.20
5	375.9	58.1	9.1	0.381	1.00	0.39
6	453.8	61.6	7.7	0.252	1.00	0.53
7	513.9	64.1	6.7	0.172	1.00	0.63
8	558.7	65.8	6.0	0.113	1.00	0.71
9	591.3	67.0	5.4	0.071	1.00	0.74
10	614.6	67.8	4.9	0.051	1.00	0.77
11	631.1	68.4	4.6	0.036	1.00	0.78
12	642.8	68.8	4.3	0.024	1.00	0.78
13	650.9	69.0	4.0	0.017	1.00	0.80
14	656.6	69.2	3.7	0.011	1.00	0.80
15	660.6	69.3	3.5	0.007	1.00	0.81
16	663.4	69.4	3.3	0.004	1.00	0.81
17	665.3	69.5	3.1	0.002	1.00	0.81
18	666.6	69.5	3.0	0.001	1.00	0.82
19	667.6	69.6	2.9	0.001	1.00	0.82
20	668.2	69.6	2.8	0.000	1.00	0.82

Phase 1 results

The development of the recruits in number, the number of natural dead eel and the number of escaping silver eel are presented in Figure 4.19 by year and sex. The weight of escaping silver eel in kg per year is given in Figure 4.20. It must be pointed out that the results of the first 20 years have higher uncertainty because GEM uses this period to stabilise parameters. The model simulates the population of the first year. The effect of the first year concerning the population of the subsequent years decreased with increasing period and can be neglected after 20 years (corresponding to the maximum age).

Male eels



Female eels

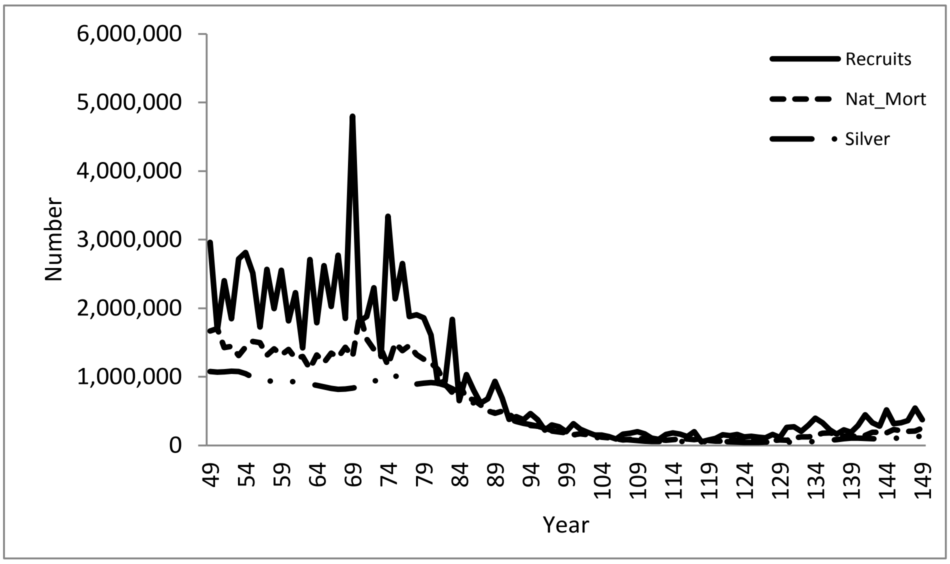


Figure 4.19. Recruits in number (Recruits), the number of natural dead eel (Nat_Mort) and the number of escaping silver eel (Silver) by year and sex from CREPE years 49 to 149. The upper chart presents the results for male eels and the lower chart presents the results for female eels.

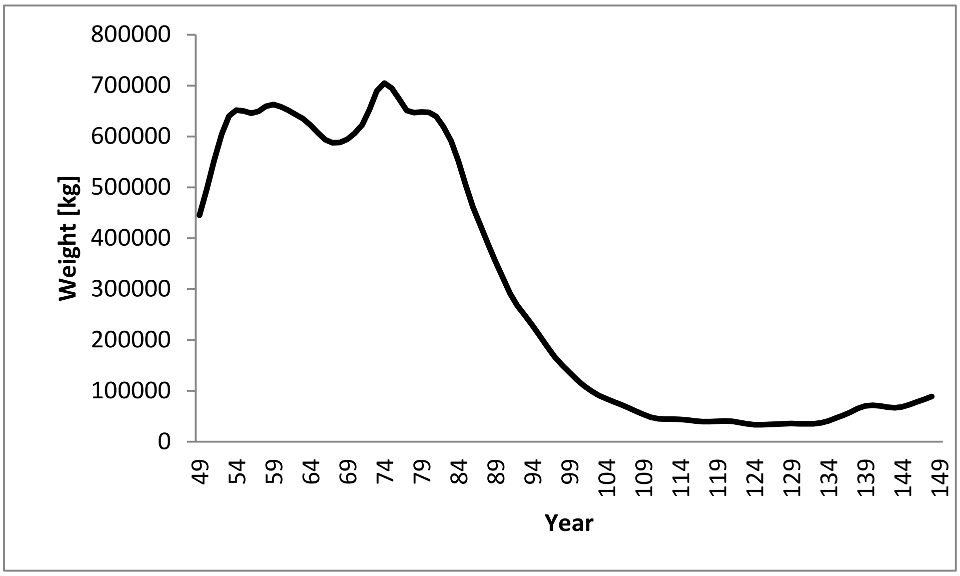


Figure 4.20. The total weight in kg of escaping silver from CREPE years 49 to 149, based on the initial parameter set derived from the CREPE Description and the data provided by the 'scientists'.

During the first phase of testing, it was not possible to conduct a comprehensive analysis of the CREPE database, nor for discussions between the data provider and modeller to optimize model parameterisation because of time and resource constraints. Therefore, GEM was run with several different parameter sets to examine their effects on the estimated output of silver eels. The following variations were explored:

The natural mortality was estimated based on low density and a mean temperature of 12.6 °C (Bevacqua et al., 2011). In addition, it was assumed that the proportion of female elvers is 95% and the mortality of escaping silver eel by obstacles was zero.

The natural mortality was estimated based on high density and a mean temperature of 12.6 °C. In addition, it was assumed that the proportion of female elvers is 95% and the mortality of escaping silver eel by obstacles was zero.

The natural mortality was estimated based on high density and a mean temperature of 12.6 °C. In addition, it was assumed that the proportion of female elvers is 50% and the mortality of escaping silver eel by obstacles was 0%.

The natural mortality was estimated based on high density and a mean temperature of 12.6 °C. In addition, it was assumed that the proportion of female elvers is 50 % and the mortality of escaping silver eel by obstacles was 80%.

The development of the weight of escaping silver eel in kg for the different versions of the model parameters are presented in Figure 4.21. The substantial differences between the estimates of silver eel biomass, especially during the periods of relatively high recruitment, illustrate a very strong dependence of the model estimates on the model parameters. All versions clearly show different phases of the recruitment (high and low). Figure 4.21 further illustrates that the estimates of the period of high recruitment are especially dependent on the model parameters because the difference in the weights of escaping silver eel were high for this period. Clearly, optimization of the model parameters is required by a discussion of the data provider and the modeller.

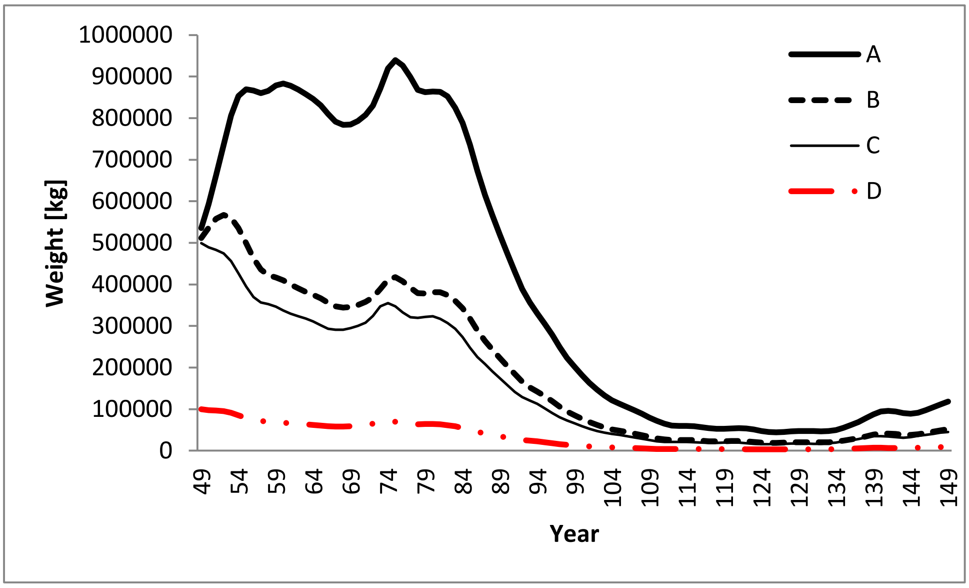


Figure 4.21. Weight of escaping silver eel in kg for four different scenarios (A: natural mortality based on low density and a mean temperature of 12.6 °C, proportion of female elvers is 95 %, no mortality of escaping silver eel by obstacles, B: natural mortality based on high density and a mean temperature of 12.6 °C, proportion of female elvers is 95 %, no mortality of escaping silver eel by obstacles, C: natural mortality based on high density and a mean temperature of 12.6 °C, proportion of female elvers is 50 %, no mortality of escaping silver eel by obstacles, C: natural mortality based on high density and a mean temperature of 12.6 °D, proportion of female elvers is 50 %, 80 % mortality of escaping silver eel by obstacles)

The maximum and the minimum weights of escaping silver eel over the total period were estimated in order to facilitate comparisons between the results of GEM and CREPE. The metrics presented in Table 4.11 show that the variation of the model parameters results in significant changes of the estimated weight of escaping silver eel. For example, the minimum value of the escaping silver eel biomass for a natural mortality based on high density, 50% of female elvers and a mortality of migrating silver eel of 80% (3143 kg) is in the range of the estimates of CREPE. In contrast, the maximum values for the same model parameters are about 7.5 times higher. That means that GEM overestimates the eel stock during the phase of high recruitment.

Table 4.11. The GEM predictions of the maximum and the minimum weight of silver eel escaping from the CREPE EMU over the total period, under varying levels of natural and turbine-induced mortalities, and with varying sex ratio of elvers (represented by the proportion of females)

Density for estimating natural mortality	Proportion of female elvers	Mortality by obstacles	Minimum weight of escaping silver eel in kg	Maximum weight of escaping silver eel in kg
Low density	95 %	0 %	44465	939759
High density	95 %	0 %	18898	567380
High density	50 %	0 %	15716	499254
High density	50 %	20 %	3143	99851

The temporal developments of different data of the CREPE database which are important for the GEM model were evaluated to find an explanation for the range of observed models estimates. The maximum and minimum values of settled elvers and catches of glass, yellow and silver eels were estimated to explore the relationships between the parameters for the periods of high and low recruitment. The period from year 49 to 149 was used because the data of the earlier years were used to establish the eel stock in the catchment and therefore included some unexpected high variation of the data. To minimize the effect of single years on the data, the time series were smoothed with following equation:

$y(t) = (x(t-2) + 2 * x(t-1) + 4 * x(t) + 2 * x(t+1) + x(t+2))/10.$

The development of the number of settled arrivals, the catch of glass eel in kg and the catch of yellow and silver eel by fishermen in kg are presented in Figures 4.22 to 4.25.

The quotients (maximum / minimum) of arriving elvers, and catches of yellow and silver eels were all quite different. The quotient of arriving elvers was 91 (32 for smoothed time series); the quotient for catch of yellow eel was 7.7 (7.2 for smoothed time series); and the corresponding quotient for the catch of silver eel was 13.0 (11.0 for smoothed time series). The strong difference between the quotients of the arriving elvers and the catch suggests strong density dependent effects between the arrival of elvers and the catches of yellow and silver eel. The catches of yellow and silver eel by fishermen further indicates high patchiness of the distribution of eel because a larger proportion of fishermen did not capture yellow or silver eel (see CREPE database).

The adaptation of models to such strong changes and density dependent changes of the simulated eel stock requires considerable time for a deep study of the available data of the CREPE database and a model adaptation for optimizing the parameters based on an intensive discussion between the data provider and the modeller.

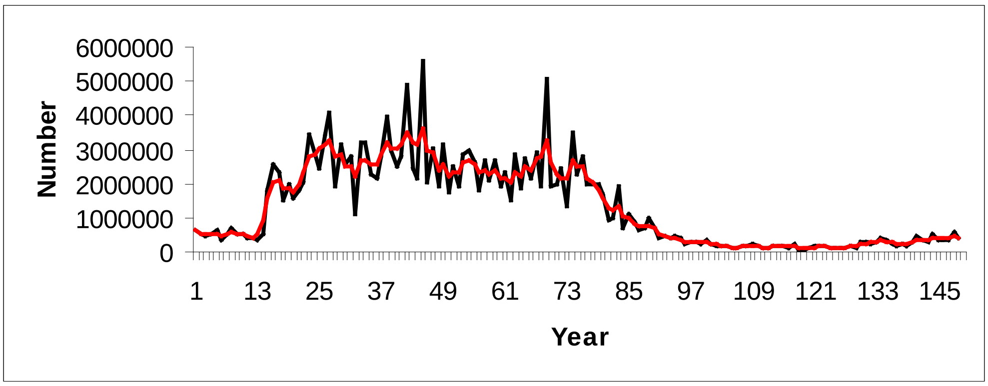


Figure 4.22. Settled arrival in number (black: original data, red: smoothed data)

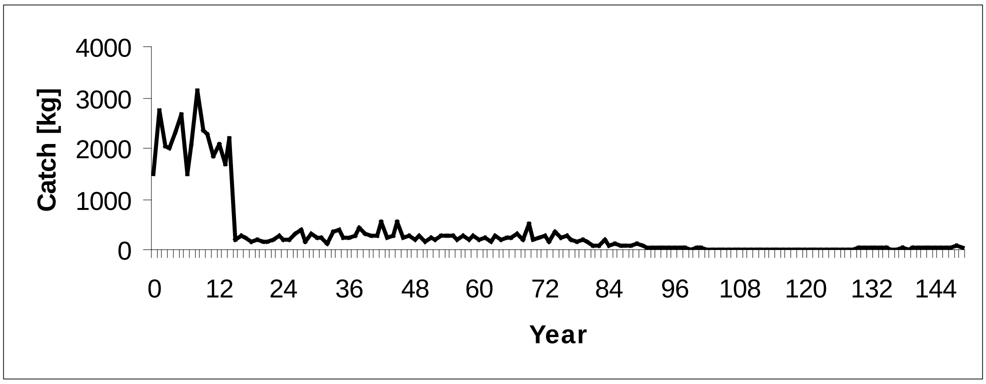


Figure 4.23. Catch of glass eel in kg

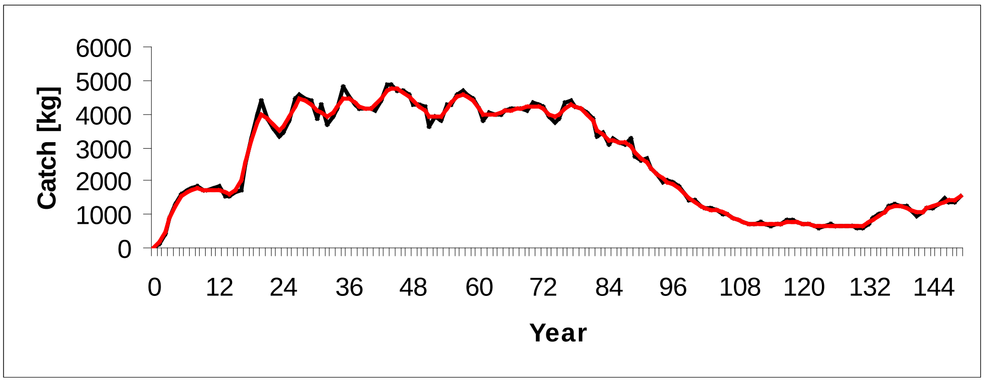


Figure 4.24. Catch of yellow eel in kg by fishermen (black: original data, red: smoothed data)

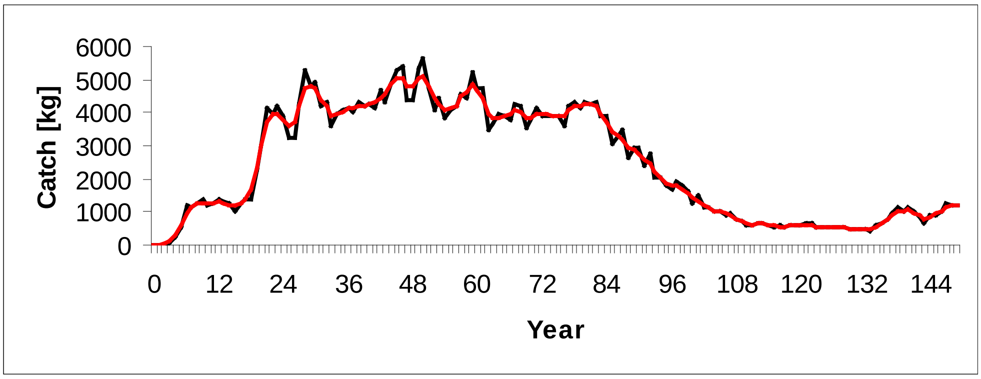


Figure 4.25. Catch of silver eel in kg by fishermen (black: original data, red: smoothed data)

Changes of the model parameters used by GEM, especially the maximum proportion of male and female silver eel, significantly influenced the annual weight of escaping silver eel. A decrease in the maximum proportion of male and female silver eel from 0.5 to 0.3 resulted in a decrease in the weight of escaping silver eel of between 3 % and 10 %. Variations in the relative proportion of silver eel by sex (see Tables 4.9 and 4.10) also resulted in significant changes to the total weight of escaping silver eel per year.

The assumed effect of available obstacles concerning the survival of the migrating silver eel is probably underestimated and need more detailed analyses. The model treated the CREPE dataset system as one unit and differences in stock characteristics of the different catchments were not taken into account. The analyses of the spatial distribution of the obstacles in combination with a more detailed calculation of the mortality of silver eel due to the obstacle will improve the quality of the estimates.

Phase 2 results

Additional runs of the GEM with the data of CREPE with variations of the model parameters resulted in a lower difference between estimates of the weight of the escaping silver eel of GEM and CREPE, and indicated that the remaining differences are probably caused by differences in the levels of natural mortality applied by CREPE and GEM. Variations of GEM with an increased natural mortality dependent of the different stock levels suggested that the natural mortality in CREPE was 4 times higher than in GEM for the high stock level, and about 2.5 times higher for the low stock level.

The differences of natural mortality can be affected by following:

Both models estimate the natural mortality based on Bevacqua *et al.* (2011). GEM used the published estimates for three density levels, whereas CREPE modified the published algorithm to estimate natural mortality for continuous variations of eel density. We tested the effect of these different approaches by estimating natural mortality for the three density levels based on both methods, and for both sexes.

Figure 4.26 shows the natural mortality by length for both sexes and the three density levels. Strong differences of natural mortality were found for all three levels of density for male eel. In all cases the estimates of CREPE were higher. In contrast, the estimates of GEM for female eel were higher than the estimates by CREPE. These differences of the natural mortality significantly influence the estimation of the weight of escaping silver eel, especially for the period of high stock size where male eels are dominant because densities are high relative to the defined carrying capacity of eel of 50 kg per ha.

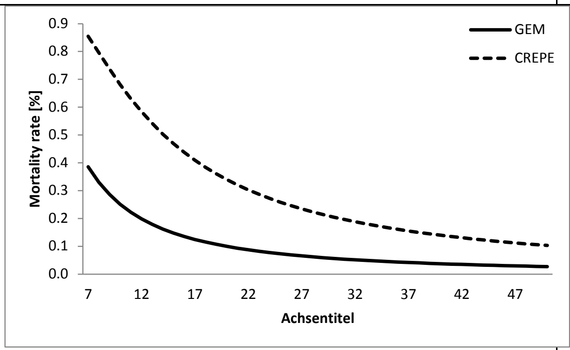
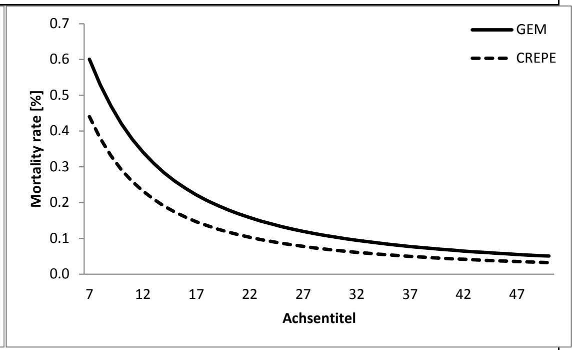
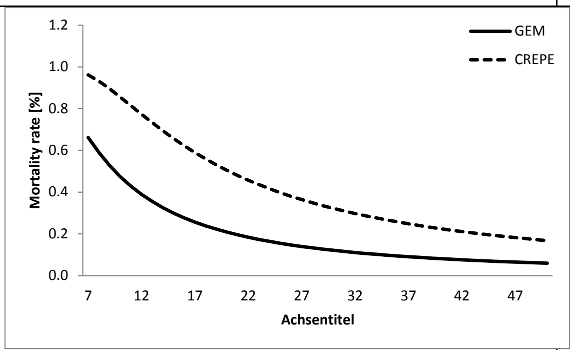
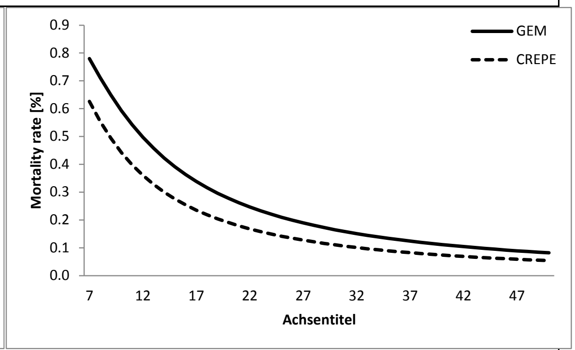
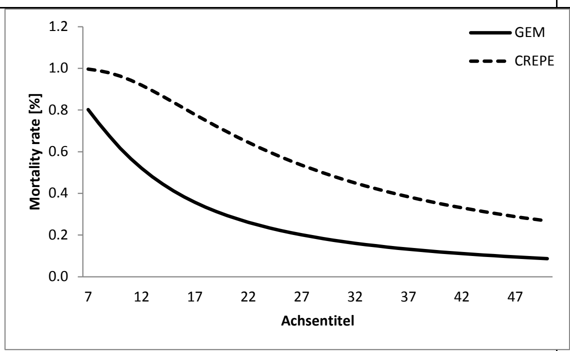
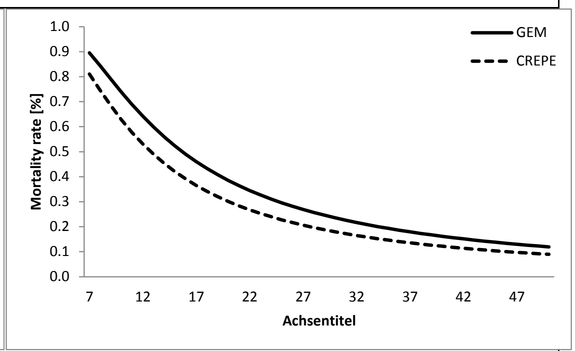
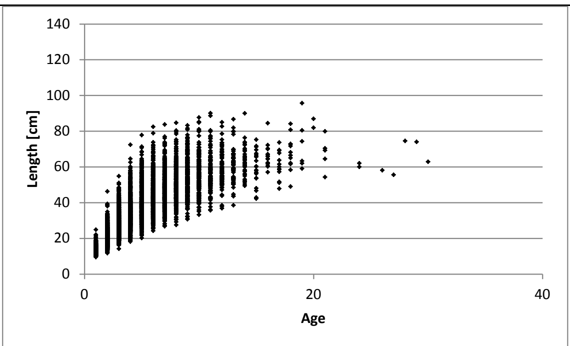
Density	Male	Female
Low		
Medium		
High		

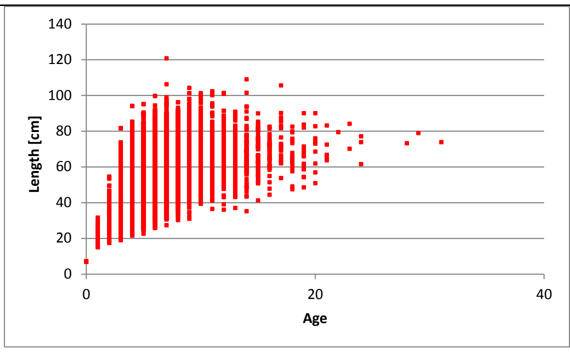
Figure 4.26. Mortality rate in [%] per year by length for different density levels and both sexes which are used in the CREPE and GEM models

CREPE and GEM used different models for the growth per year of eel. CREPE used a constant growth of individuals which was chosen as constant for the total life span. The constant growth varied between 2.9 cm per year and 16.1 cm per year based on data of age group 7 of the samples of fishery given in the CREPE database.

In addition, the length range of the same year class was different for the data based on the samples of the electro fishing surveys and the data of the commercial fishery. Figure 4.27 shows the age length data of both sources where the data of all years are summarized. GEM used the Bertalanffy growth function (BGF) for describing the relation between the observed age – length data. The parameter k of the von Bertalanffy growth function strongly varied dependent on the data used for the estimation, especially, the k value of the BGF varied in a wide range. Table 4.12 presents the parameters of the BGF by sex with the lowest and highest k values. Figure 4.28 illustrates the growth for the different k values by sex. The different growth also influence the natural mortality because the BGF is used by GEM to transfer length based data in to age base data. The natural mortality of eel with lower growth is higher because the eel stay longer in the length range where the natural mortality is high. With increasing length and age the natural mortality decrease



Age – length relation based on all samples of electro fishing surveys



Age – length relation based on all samples of commercial fishery

Figure 4.27. Age – length data from CREPE database based on electro fishing surveys and biological samples of the fishery

Table 4.12. Parameters of the Bertalanffy growth function for the lowest and highest k values by sex based on the CREPE database

	Male (low k value)	Male (high k value)	Female (low k value)	Female (high k value)
L(inf)	50.0	52.75	67.75	69.64
k	0.21	0.40	0.17	0.31
t_0	-0.45	-0.17	-0.35	0.09

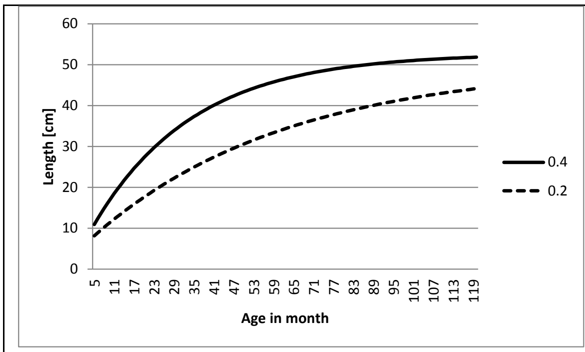
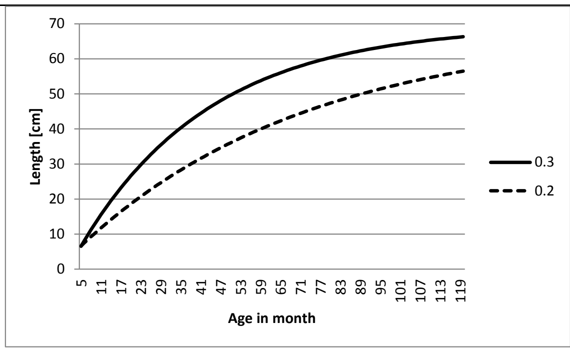
	
Male eel	Female eel

Figure 4.28. Bertalanffy growth functions based on the parameters with lowest and highest k value for both sexes

To evaluate the possible effects of the two parameters mentioned above new runs of GEM were realized where the mortality of CREPE was used and the effects of both growth functions were evaluated.

Following four versions of input data were used:

A) Use of the GEM data for the natural mortality (high density, 12.6 °C) and BGF with high k value

B) Use of the GEM data for the natural mortality (high density, 12.6 °C) and BGF with low k value

C) Use of the CREPE data for the natural mortality (high density, 12.6 °C) and BGF with high k value

D) Use of the CREPE data for the natural mortality (high density, 12.6 °C) and BGF with low k value

Figure 4.29 presents the temporal development of the weight of escaping silver eel for the input parameters A) and C). The weight of escaping silver eel was lower than for the input parameters of A) with a factor between 0.39 and 0.43, but, the weight of escaping silver eel was highest than the results of CREPE. Negative stocks were estimated if the input parameters of version B) or D) are used indicating high sensitivity of the model estimates concerning the natural mortality and growth. The negative stock estimates showed that the natural mortality was too high.

The mean natural mortality used in CREPE model is probably higher than the estimates for high density in the CREPE model as used above and lower the estimates based on the growth of eel where the low k value of BGF was used.

The high spatial patchiness of eel in CREPE in combination with the CREPE model option to use natural mortality above the level of highest density according to Bevacqua et al. (2011) can be the reason for the smaller weight of escaping silver eel calculated by CREPE.

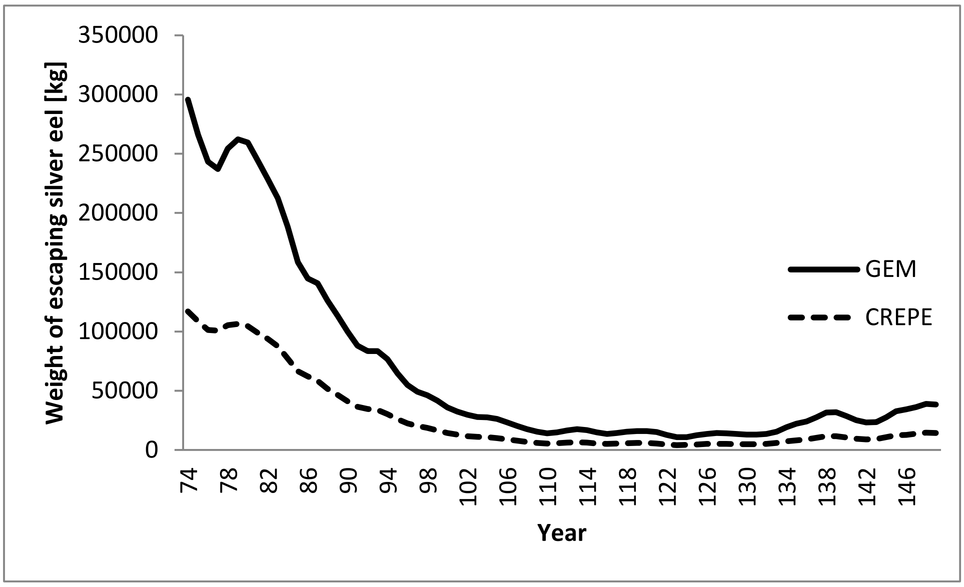


Figure 4.29. Weight of escaping silver eel based on the natural mortality used by GEM based on the literature (GEM) and based on the realization of the natural mortality realized in the CREPE dataset.

The differences between the description of CREPE and the realization of CREPE could not be detected during the model adaptations, especially, because effects of the model realization varied in dependence of the stock development. Therefore, adaptations of GEM are required to describe the development of eel with strong variations of the stock dynamics as recruitment, natural mortality etc.

The experience of the model adaptation to the CREPE dataset clearly supported the experience based on the model adaptations to the real datasets that enough time must be planned for this process (halve year and longer) and intensive corporation between the data provider and model developer is a very important, sensitive and time consuming process. It requires more time than it was available for the adaptation of GEM and other models.

Conclusions

The used of the natural mortality based on the CREPE data results in a strong decrease of the weight of escaping silver eel. Nevertheless, strong differences between the estimates of GEM and CREPE were observed for the first period with high stock level. However, for the period of low stock level, the estimates of both models are close together.

The GEM can be used to describe the eel stock of the CREPE data, but requires detailed discussions between the data providers and the modellers to optimize the results. Also, the incorporation of density dependent growth and mortality as well as the incorporation of spatial effects might also improve the quality of the model results.

4.2.3. Application of SMEP II to the CREPE data set

SMEP II must at least have information describing the size and structure of the river basin, the eel life history processes (growth, natural mortality, sex differentiation and silvering), the level of annual recruitment, and a spatial limit to production (~carrying capacity), in order to predict potential production under 'pristine' or unspoilt conditions. Predictions accounting for the impacts of human factors such as fisheries and turbines (negative impacts) or stocking (positive impact) also require descriptions of these impacts in terms of the numbers or rate of eels lost or gained, and the locations of the impacts within the river basin.

Here we first describe the derivation of the input data used in applying SMEP II to the CREPE data set.

The model testing procedure was conducted in two phases. In the first phase, input data were derived from the description of CREPE and from the data files provided for SMEP II. The silver eel escapement biomass predictions by SMEP II were compared to those from CREPE, which had been withheld from the SMEP II users to create a "blind" test. This phase simulated a "data not quite rich" scenario, but probably the best that could be expected in the 'real' world.

In the second phase, the input data were adjusted or 'tuned' to improve the fit between the predictions of silver eel biomass and the actual results from CREPE. 'Tuning' is the systematic revision of model parameter values to produce a model output as close to desired as possible. This illustrates the ideal data-rich situation from which to examine the potential accuracy and precision of the assessment model. However, as such a truly data-rich situation does not exist in the real world, the results represent the best possible but not necessarily what managers should expect in other situations.

Phase 1

Input data

Spatial description of the river basin

The virtual river basin district created by CREPE contains five discrete river networks. As these networks are discrete, SMEP II cannot be applied to them all simultaneously. Therefore, we applied SMEP II to the river network with the greatest number of zones (catchment 39862).

SMEP II allows the user to define a very complicated branching network of river reaches, but this is very time consuming to prepare the input data and to explore the results. Also, there are rarely more than a few electro-fishing surveys throughout an entire river network, and the results of these surveys are often extrapolated to neighbouring 'empty' reaches. Therefore, the network of river reaches in SMEP II is usually compressed into a small number of zones containing one or more electro-fishing surveys, fisheries or other human impacts.

For this zonal compression, the CREPE data were gathered into stretches corresponding to the areas fished by one fisherman within a single catchment. Catchments were simplified to a linear succession of stretches since the list of reaches fished by a fisherman is determined according to the distance to sea.

Reaches are described in SMEP II according to river length and width (both in km), and the model converts these to estimate wetted areas (km^2). The CREPE data gave reach length (km) and wetted area (km^2), so width was back-calculated from wetted area / length. The reaches were 17 or 18 km long and average widths varied from about 0.084 to 1.392 km. Note that the 'actual' widths in the CREPE river network would, in many cases, be much narrower than these averages but the procedure to collapse branches to a few zones and conserve reach lengths means that the reaches appear rather wide in order to preserve the appropriate wetted areas.

Reaches were characterised according to their connectivity to other reaches, and the distance from the bottom of each reach to the lowest point in the basin.

Growth rates

Growth rates were derived from the length and age data provided for all the eels caught in electro-fishing surveys. A simple linear trend line was fitted through the data, assuming the intercept at 7 cm to reflect the size at transformation from leptocephalus to glass eel. This suggested an average annual growth rate of 50 mm.

Growth rates are fixed throughout the period of investigation, i.e. assuming no change in influence of environment. Growth rates were not moderated by changes in environment, such as inter-annual variation in monthly temperatures, because this is not possible in SMEP II.

Baseline growth rate was not varied between habitats in the SMEP II simulations because this feature is not possible in the present version, whereas such rates in CREPE varied between saline and freshwater environments, and between rivers and lakes. SMEP II uses an index of environmental quality to moderate the local biomass threshold, whereas CREPE uses a similar index to moderate growth rate.

Growth was applied to eels in the 3^{rd} time step in every year, i.e. summer.

The **length / weight** relationship is reported in the CREPE description, with constants a = 0.00091842 and b = 3.1813, for length in cm and weight in g. As SMEP II models length in mm, the constant (a) was converted using a rule from www.FishBase to derive the 'A' value appropriate for length (mm): 'A' = 0.00000060498.

Natural Mortality

According to the Crepe description above, natural mortality of glass eel arriving from the sea at 70 mm length was set at a rate of $4.81.y^{-1}$ for the first season (time step) after arrival (Lambert, 2008), which corresponds to a mortality rate of 70% during the season of immigration. This very high rate not only takes account of mortality because of predation and disease, but also because of the presumed failure of some glass eel to 'settle'. Thereafter, the rates were set according to the model of Bevacqua *et al.* (2011), which estimates natural mortality based on water temperature, eel density and fish size. The average annual temperature of 12.56 °C was estimated from the 'TemperatureSeries' dataset provided for CREPE.

The Bevacqua *et al.* (2011) model estimates natural mortality of an eel of given weight, whereas SMEP II applies natural mortality to eels of given length. Therefore, the length/weight relationship (above) was used to define weight-at-length and hence length-based mortality rates for SMEP II. Also, the Bevacqua *et al.* (2011) model reports separate mortality rates for males and females, but SMEP II applies the same rate to both sexes, so we have used the mean value for males and females combined.

The Bevacqua *et al.* (2011) model estimates mortality rates at low, average and high densities, though the boundaries of these densities are not defined by the authors, presumably because they will vary with local environmental conditions. Three mortality rate input files were created for SMEP II, according to these low, average or high density conditions (Table 4.13). However, mortality rates are fixed throughout the time period of any SMEP II simulation, so different conditions were used for different phases of the recruitment time series (see below).

Note that because the mortality rates are fixed in SMEP II, the results may be insensitive to substantial changes in recruitment during the simulation. Ideally with this approach, that simulations ought to be constrained to periods when density does not change much, e.g. at times of stable high or low recruitment.

Table 4.13. Annual mortality rates of CREPE eels at various length classes, estimated under low, average and high density conditions. Length classes correspond to the glass eel + years growth (120 mm), additive annual growth to 520 mm, and then 100 mm intervals thereafter. The rate to 120 mm covers the first year after glass eel arrival, assuming an average growth rate of 50 mm per annum.

Length class (mm)		Annual mortality rate	
	Low density	Average density	High density
0 – 80	2.56	2.68	2.82
80 – 120	0.30	0.56	0.83
120 – 170	0.18	0.34	0.50
170 – 220	0.12	0.23	0.34
220 – 270	0.09	0.17	0.25
270 – 320	0.07	0.13	0.2
320 – 370	0.06	0.11	0.16
370 – 420	0.05	0.09	0.13
420 – 470	0.04	0.08	0.11
470 – 520	0.04	0.07	0.10
520 – 600	0.03	0.05	0.08
600 – 700	0.02	0.04	0.06
700 – 800	0.02	0.03	0.05
800 – 900	0.02	0.03	0.04
900 – 1000	0.01	0.03	0.04

Sex differentiation

The parameters describing the probability that an undifferentiated eel is expected to differentiate at given length, and whether to become male or female, were derived from the information provided in the CREPE description. The critical length of differentiation was set in SMEP II at 200 mm.

The length above which all eels would be female was set as 600 mm because 99% of male eels were expected to silver and emigrate at lengths below this value, according to the function shown described in the CREPE report (see Annex B).

Silvering

SMEP II requires a description of the probability of a male or female eel becoming silver and emigrating, depending on the length of the eel. The lengths at which 50% of male and female eels underwent the silvering process were set at 410 and 620 mm, respectively.

Carrying Capacity

The carrying capacity value was used in CREPE to moderate density dependent effects on natural mortality and sex determinism. In SMEP II, a similar parameter called the Maximum Biomass is used to moderate natural mortality, sex determinism and movement rate.

The carrying capacity in the original CREPE model was set as 50 kg per hectare. This value corresponds to a mean value of biomasses observed across the CREPE river basin. However, the Maximum Biomass applied in SMEP II represents the maximum biomass likely to occur within the river basin – for CREPE this was 250 kg ha^{-1} (2500g per $100m^2$).

As there was some uncertainty as to which was the more appropriate parameter value to apply in SMEP II, tests were run with both values.

Recruitment

The CREPE data set was simulated using a 150 year time series of recruitment, with relative levels of glass eel arrival split into 5 time periods:

1. the initial phase for populating and stabilizing the model (around 50 years)
2. a period with a constant level of glass eel arrival/recruitment (25 years)
3. a period of decreasing arrival – the 'crash' (30 years)
4. another period of stable arrival (20 years)
5. a period of increasing arrival / recruitment (25 years)

The nominal annual amount of glass eel arriving at the sea edge of the CREPE basin under high recruitment conditions was set at 10 million for the basin as a whole, but varied from year to year to more realistically simulate natural variations.

The quantity of eels arriving at the mouth of each river network was proportional to the catchment area of river network relative to that of the whole basin. The catchment area of network '39862' was 21,336 km^2, which is 53.34% of the total wetted area of the basin (40,000 km^2). The time series of glass eel arrivals was adjusted accordingly (Figure 4.30). However, because the stochastic variation in arrivals allowed the number to be greater than 5,334,000 (53.34% of 10,000,000), the index of recruitment for SMEP II constructed in relation to a maximum of 12.2 million, so that the recruitment index time series consisted of values between 0 and 1.

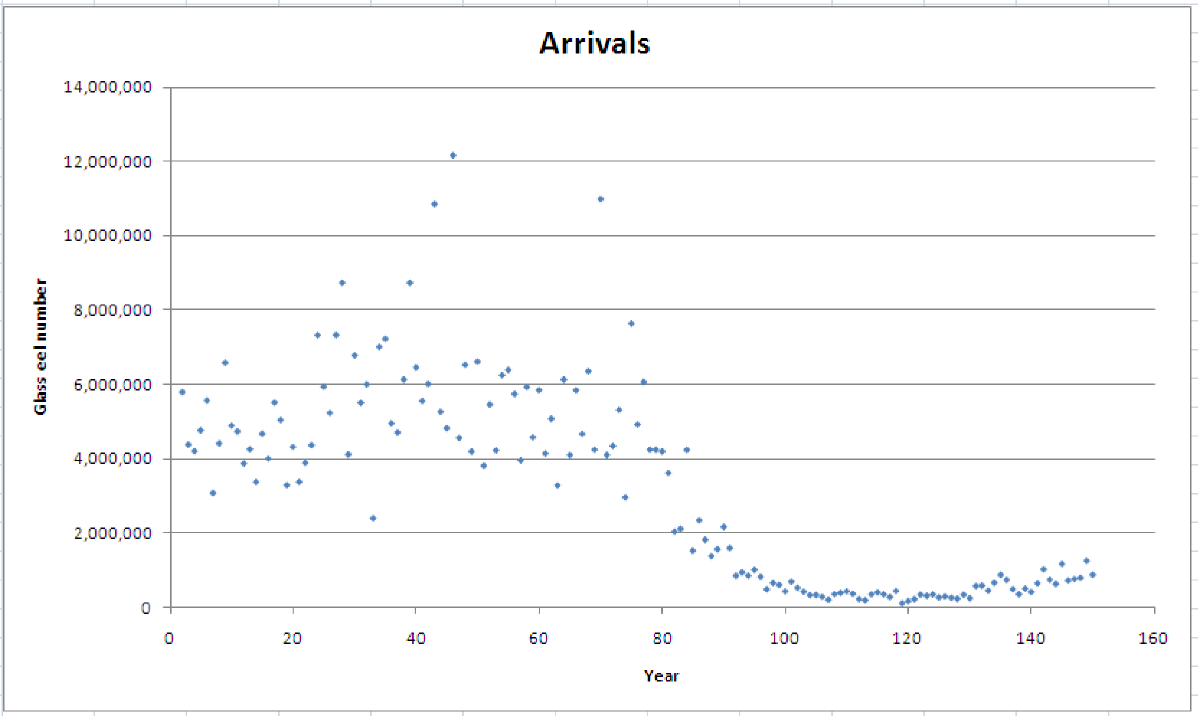


Figure 4.30. Number of glass eel arriving at the seaward end of river network 39862 per annum for the 150 year time series. Note that the first 50 years of data were used to 'train' the dataset and are not used in the test procedures.

The average length of recruits was set at 70 mm as this was the standard length of glass eel caught in the CREPE fisheries. SMEP II also requires detail of the variation in the length of recruits, although such variation was not applied in CREPE. As variation in size of recruits is a requirement in SMEP II, we set the lower boundary to 60 mm and the upper boundary to 80 mm.

Movement rates

In SMEP II, the dispersal of eels throughout the river network is achieved by a combination of a general tendency to move upstream, along with a directional movement driven by the local densities in the 'donor' and 'recipient' reaches.

SMEP II requires the user to define the distances that undifferentiated and differentiated yellow eels will travel in any year. In CREPE, diffusivity of elvers was set so that they could in theory disperse throughout the entire river length. However, the electro-fishing survey data revealed that eels were only found up to about 200 km from the sea (reach 9 out of 20).

In SMEP II, eel movements only occur in autumn (time step 3), but the rate is set as an annual rate. If migration is 100 km from year to year, then the rate set in SMEP II has to be 4* the observed rate to achieve the same distance travelled over time. Therefore, the migration speed of undifferentiated eels was set to 480 km.yr^{-1}, to allow for undifferentiated eels to travel at least 100 km in their first year. The speed for differentiated eels was set at 200 km.yr^{-1}, giving them the opportunity to move between neighbouring reaches in any year.

Anthropogenic Impacts

Fisheries

CREPE included fisheries occurred for glass, yellow and silver eels. In river 39862, the glass eel fishery operated in reaches 1 to 6 for the first 15 years but thereafter only in reach 4. The yellow eel fishery operated at various times in reaches 1 to 11 over the first 15 years, then only in reaches 1 to 6 to year 106, and reaches 1 to 5 thereafter. The silver eel fishery operated at various times in reaches 1 to 9 over the first 15 years, but thereafter only in reaches 1 to 4. As the first 50 years of the time series (the training dataset) were excluded, only the glass eel fishery in reach 4, the yellow eel fishery in reaches 1 to 6, and the silver eel fishery in reaches 1 to 4 were modelled in SMEP II.

For SMEP II, catches were described as number by stage (glass, yellow, silver), according to reach, season and year. Catch weights (kg) from the CREPE files were converted to estimates of numbers of eels using estimates of the average weight of each stage derived from surveys of fishing data. The average weight of glass eel (0.4488 g) was the same throughout the network and time series. The average weights of yellow and silver eels varied between reaches and over the time series, so reach/year-specific conversions were applied. During the first 14 years of the time series, survey data of numbers and weight of yellow and silver eel catches were missing for some reaches (presumably simulating a period of learning and organisation for the fishery scientists). Although data for these early years were not used in the model tests, we assigned average eel weights from neighbouring reaches and years in order to complete the dataset.

In SMEP II, glass, yellow and silver eel fisheries were fixed to the 2nd, 3rd and 4th time steps, i.e. spring, summer and autumn, respectively.

Obstacles

The impact of each obstacle was characterized in CREPE by the mortality rate (%) associated with passage upstream or downstream. However, the CREPE description (Annex B) implied that only the downstream mortality at turbines was simulated.

Where more than one turbine location occurred within a SMEP II zone, the mortality rates were combined to provide a total turbine mortality rate for passage through the zone. The mortality was fixed to the lowermost point of each zone, and the rate did not vary over time.

These mortalities were not restricted to downstream migrating silver eels, but were applied to all eels moving downstream past a turbine.

In general, mortality rates due to the presence of turbines in reaches were low: 0 to 1% for most, 1-2% for three reaches (15-14, 13-12, 9-8), but the loss from 4 to 3 was 19.9%.

Phase 1 results

The aim of these SMEP II applications was to test the precision and accuracy of the model in estimating historic and present silver eel escapement biomass (kg). As the first 50 years of the CREPE data set were used to allow CREPE to reach a kind of stable condition, the historic period for comparison with SMEP II was the period of high recruitment from year 50 to 75. The "present" period might be considered to be the end of the CREPE time series (year 150), but in reality (in 2011) the European eel recruitment time series is still at a very low level. Therefore, we used SMEP II to predict the 'present-day' silver eel escapement during the periods of low and increasing recruitment.

The use of a fixed set of length-based mortality rates in SMEP II, with only the choice of setting rates suitable to high, average or low densities, means it is not appropriate to apply SMEP II to the entire time series of recruitment and impacts in a single application, and then analyse the results for the appropriate time periods to estimate escapement under high recruitment, and low recruitment after the crash. The solution was to run a series of simulations for different sections of the time series, applying the high, average or low mortality rates when appropriate.

First, we applied SMEP II to the first 75 years of CREPE when recruitment was high, using mortality rates according to the High density scenario. Then we projected forwards through the period of declining recruitment, using the Average mortality rates. Finally, we predicted through the period of low and then increasing (recovering) recruitment using the Low mortality rates.

Figure 4.31 shows the CREPE recruitment time series from year 50 to 150, and the SMEP II predictions of silver eel biomass for periods of high, low and increasing recruitment, with the model constrained by 2500g (red) and 500g (blue) biomass threshold conditions. The broken lines represent the predictions in the absence of human impacts, and the solid lines represent predictions taking into account these impacts.

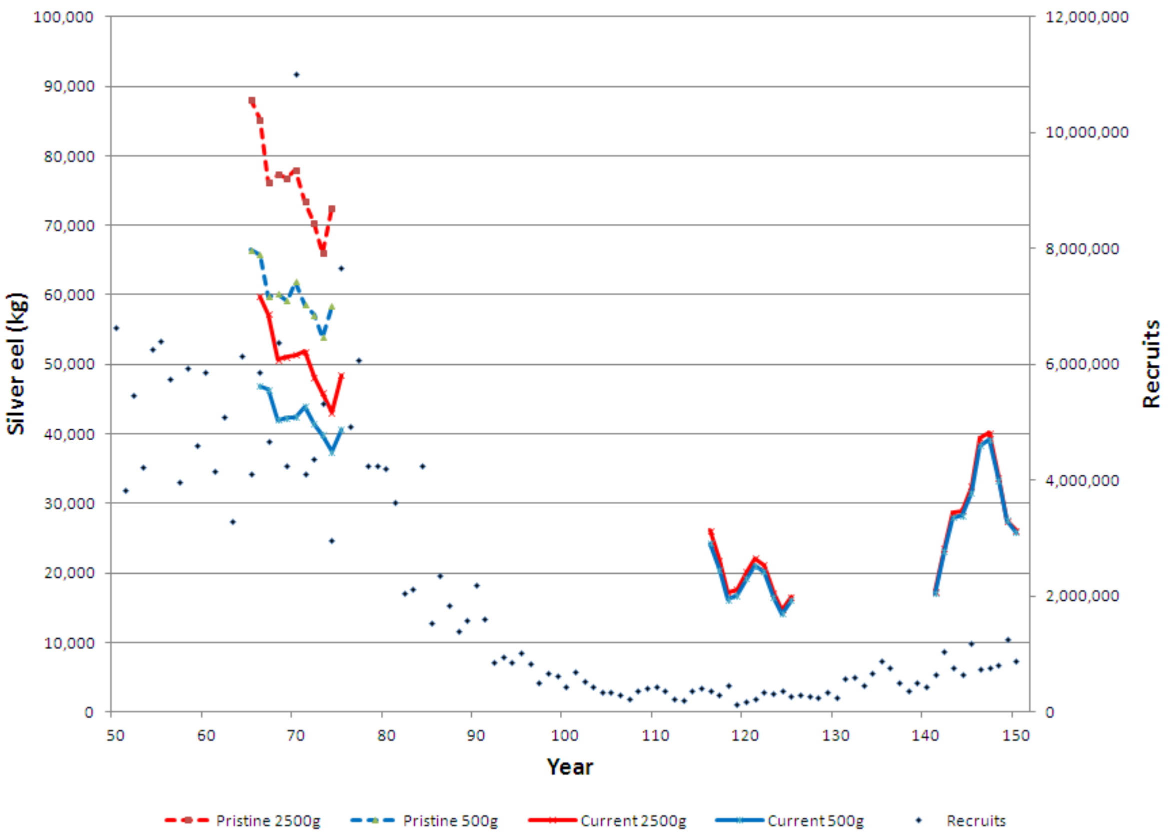


Figure 4.31. Ten year 'snapshots' of SMEP II model predictions of silver eel escapement biomass, compared to the recruitment time series from CREPE.

The recruitment time series in CREPE varied from year to year, as did fishery catches, and hence silver eel escapement also varied from year to year even during the period of high and relatively stable recruitment. During the period of high recruitment – the historic period - SMEP II predicted 'pristine' annual silver eel escapements of about 55,000 to 88,000 kg, depending on previous recruitment levels and on the biomass thresholds applied by the model. Taking the human impacts into account for the same period, SMEP II predicted annual escapement biomasses of about 47,000 to 60,000 kg.

Some 50 years later in the time series, after the recruitment had declined to about 2% of the historic high and 'stabilised' at this low level, SMEP II predicted annual silver eel escapement biomasses of about 15,000 to 25,000 kg when taking human impacts into account.

As recruitment began to recover in the last decades of the time series, SMEP II predicted escapement increasing from about 18,000 to 40,000 kg.

The average weights of male and female silver eels were within 1g between scenarios: 670 to 671 g for females, 206 to 207 g for males. Therefore, the differences between the escapement predictions of the two biomass threshold conditions during the period of high recruitment were mainly due to differences in the relative proportions of male and female silver eels. The stronger effect of density on gender choice resulting from the 500 g biomass threshold meant that SMEP II predicted a lower proportion of female (heavier) silver eels under this scenario. The sex ratio produced under the 500 g condition was much closer to that observed in the CREPE data for silver eel catches, suggesting that this condition more closely represented the original CREPE conditions.

Note that the results for the two biomass threshold conditions were very close together during the low and increasing recruitment periods. This is to be expected because density dependent effects on eel production would be limited during these periods of low recruitment.

These average weights are a little higher than those reported from the CREPE silver eel fishery during the entire time series, which were 536 and 193 g for females and males, respectively.

In addition to the results shown here, SMEP II also produces results for biomass and numbers of female and male silver eels, allowing the derivation of sex ratio, and the numbers and weight of eel lost to fishing and turbines. It also reports results for yellow eels, showing numbers, density and length frequency per reach for every year of the modelled time series.

Phase 2 - 'Tuning' SMEP II for the CREPE data set

Discussion with the CREPE data provider and comparison of the phase 1 predictions of silver eel escapement biomass with those produced by CREPE revealed that SMEP II had significantly overestimated silver eel escapement biomass – by around a factor of 5 to 9 under high recruitment, and 10 to 18 times under low and recovering recruitment. Therefore, we engaged in a second phase to adjust (tune) the SMEP II model conditions to produce a results data set as close as possible to the CREPE test data set.

Two new CREPE data sets provided silver eel escapement numbers in scenarios where human impacts were or were not applied. We modelled SMEP II to the silver eel production results from the new CREPE 'pristine' scenario data set including knowledge of silver eel escapement, based on the same recruitment trend as before, but in the absence of any impacts of fishing or turbines. However, we also analysed some the data resulting from the new CREPE 'impacts' data set, where this provided parameter values that could not be provided from the 'pristine' data set, e.g. eel measures from fishery catches.

The following section only describes the changes that we made to some model parameters that were altered during the tuning process.

Growth rates

The first application of SMEP II produced silver eels that were much older than those produced by CREPE – average age of female and male silver eels were 13 & 9 vs 6 & 4 years, respectively. As the slivering process was driven by length of the eel, and the silver eel produced by both models were of similar length, the different mean ages meant that the CREPE eels were growing much faster than the SMEP II eels.

The growth rate function used in the original application of SMEP II underestimated growth rate for two reasons. First, the growth function was derived from the length-at-age measures for yellow eels caught during the electrofishing surveys. This dataset was biased towards the slower growing eels because those with the fastest growth rates quickly became silver eels and 'left' the population, and therefore were less likely to be caught in surveys of resident eels. Second, the simplistic fit of a linear growth curve and selection of a lifetime average growth rate underestimated growth rate in young eels and overestimated it in older eels (Figure 4.32). Therefore, the SMEP II tuning application used a new growth function based on a non-linear von Bertalanffy fit of measures of yellow and silver eels (Figure 4.33).

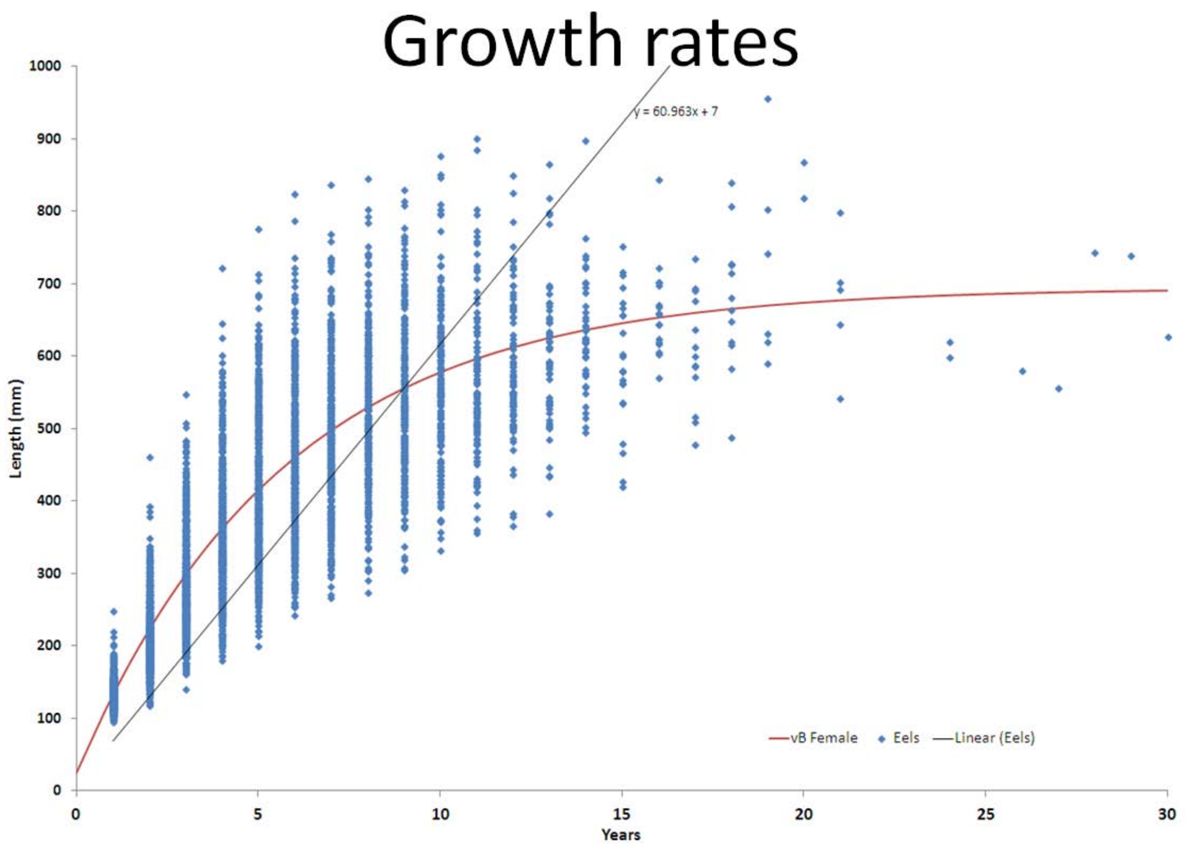


Figure 4.32. Linear and non-linear regression lines fitted to length-at-age data for a select group of CREPE eels. In comparison with the non-linear trend based on a von Bertalanffy curve, the linear trend underestimates growth rate at ages up to 9 years old, and overestimates growth rate at older ages.

As the new Pristine scenario had no fisheries and therefore no silver eel length measures were available, growth rates were derived from the length and age data provided for all the eels (55,788 eels) caught in the fisheries and electro-fishing surveys in river 39862, under the IMPACTS scenario.

The mean values of *von Bertalanffy* growth model coefficients L_∞ and k were estimated by fitting non-linear regression to the length-at-age data using the IGOR+ software package. L_∞ = 699.548 mm; k = 0.290.

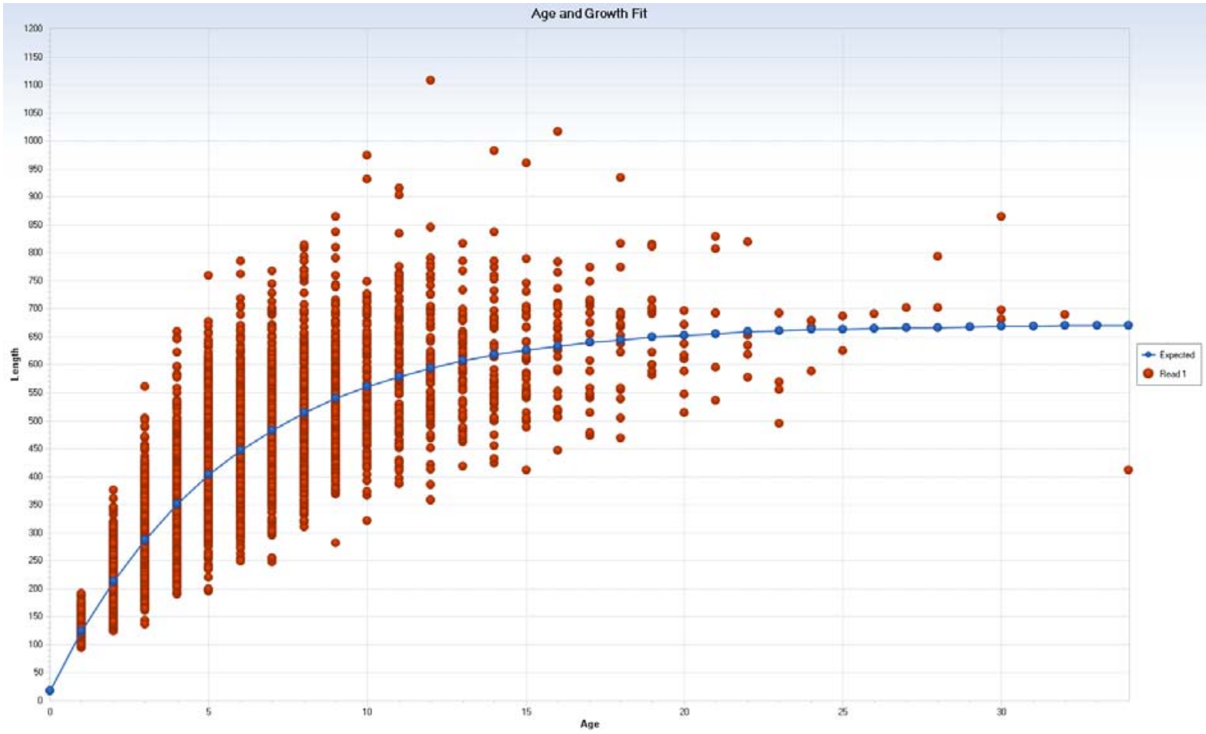


Figure 4.33. Length-at-age for CREPE eels captured in electrofishing surveys and by fisheries. The curve (blue line) represents the average growth function of the von Bertalanffy form fitted by non-linear regression.

The VBF coefficients for the electro-fishing data from the new 'pristine' and 'impacts' datasets varied a little (L_∞ 671.864mm vs 736.64mm; k 0.178 vs 0.150), but plotting the two curves on the same chart showed that this difference was only apparent in eels above age 10 and length 550mm.

The **length / weight** relationship was tuned to the length and weight measures of the eels caught in fisheries and electrofishing surveys in the new 'impacts' scenario (6e-7, 3.1917).

Carrying Capacity

The carrying capacity in the original CREPE model was set as 50 kg per hectare (500g per 100m^2) as defined in the CREPE description. This value corresponds to a mean value of biomasses observed across the CREPE river basin. This value was tuned in SMEP II to deliver a silver eel production with the appropriate sex ratio.

Recruitment

We used the new glass eel arrivals data for river basin 39862 (Figure 4.34) to construct the 'recruitment index', expressed relative to 9 million eels, the maximum number of glass eel arriving at the bottom of the river basin throughout the time series. In CREPE, only about 30% of glass eels survived to become settled elvers (see Natural Mortality section for further details). Figure 4.34 also illustrates the time series of settled elvers deriving from these glass eel.

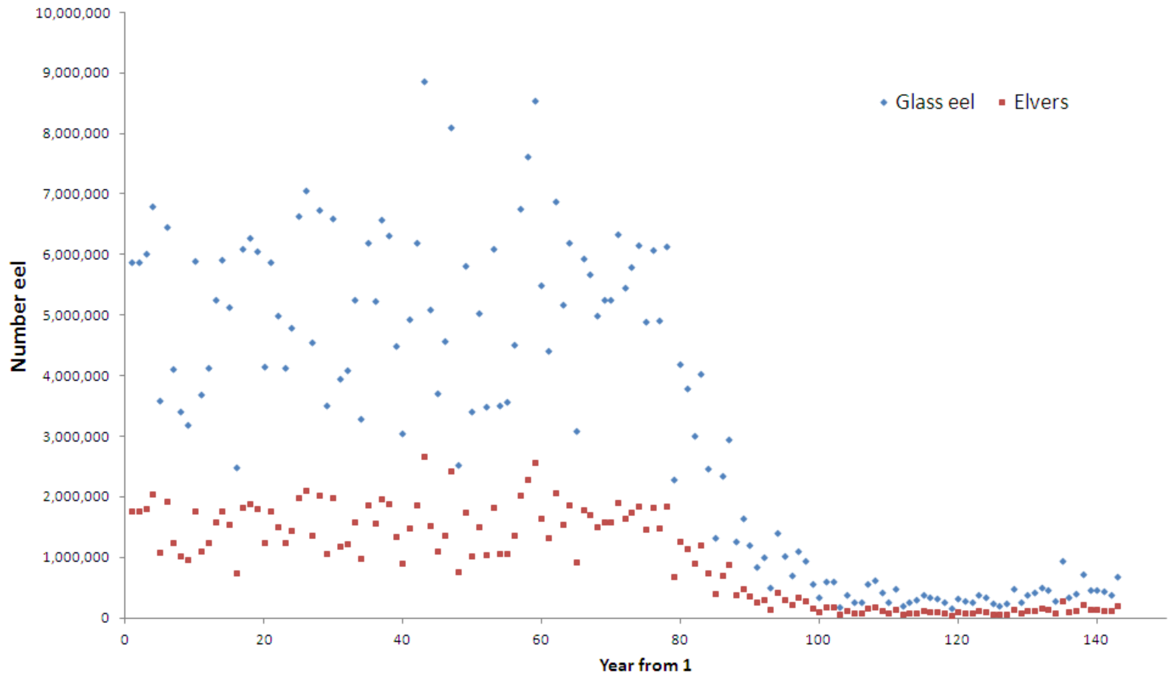


Figure 4.34. Number of glass eel (blue diamonds) arriving at the seaward end of river network 39862 per annum for the 143 year time series, and number of settled elvers (red squares) deriving from these arrivals.

As before, the average length of recruits was set at 70 mm, to represent glass eel recruits as detailed in the CREPE Description.

Natural Mortality

The CREPE data set was constructed with mortality rates of settled elvers and older eels set according to the model of Bevacqua *et al.* (2011). However, an exceptionally high mortality rate was set for glass eel on their arrival from the sea. Tests revealed that this initial mortality rate had the strongest influence on the silver eel results, and tuning focussed on identifying the most appropriate value.

Silvering

The length (but not age, see above) of silver eel produced by SMEP II in the first tests were similar to those from the CREPE catches. However, we fine tuned the inflection parameters for the male and female silvering probability curves to find the values that resulted in mean lengths very close to those estimated from the silver eel catch fishery data provided as part of the Impacts dataset (no fishery and therefore no silver eel data available under Pristine conditions). The best parameter values were 345 mm for males and 590 mm for females.

Anthropogenic Impacts

The purpose of this test series was to compare the outputs of SMEP II with those of CREPE in the absence of anthropogenic impacts, so none were modelled.

Phase 2 Results

Once again, the aim of these SMEP II applications was to test the precision and accuracy of the model in estimating historic and present silver eel escapement from a data-rich scenario.

Silver eel escapement in the absence of human impacts

The numbers of silver eel produced under SMEP II modelling with various natural mortality rates for the 0-80 mm "glass eel" was compared with the numbers reported in the CREPE data. A rate of 8.9 per year was found to give a close approximation of the CREPE results (Figure 4.35). The results from this mortality rate were then used to explore other characteristics of the eel produced by SMEP II in comparison with those in the CREPE results for the 'pristine' scenario.

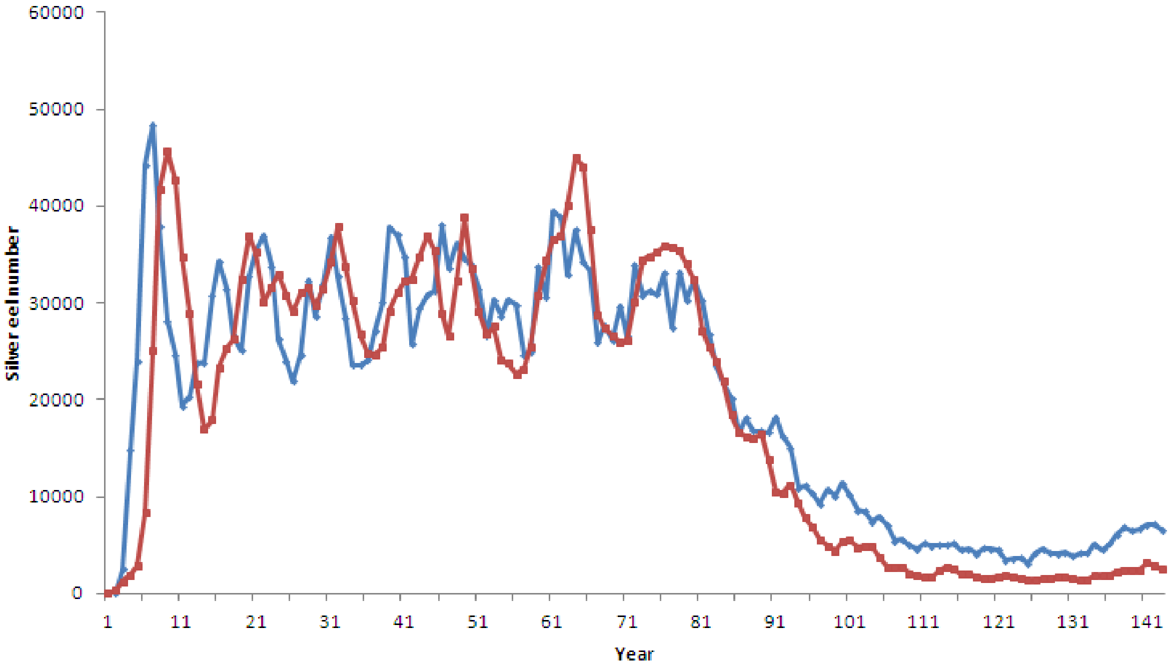


Figure 4.35. Time series of silver eel production (numbers) from CREPE (blue line) and from SMEP II simulation (red line).

Size and age of silver eels

The 'pristine' escapement data file from Crepe reported numbers but not biomass of silver eel so it was not possible to derive average weights of male and female silver eels. Also, as there was no fishing in the 'pristine' version of CREPE, there were no silver eel catch data from which to characterise the size and age of the silver eels. Our alternative was to examine the fishery catch data from the 'impacts' version of CREPE data. We assumed that the fisheries for glass, yellow and silver eel had no effect on the size range of silver eels, and that the silver eel catch was representative of the silver eel production. Both assumptions might be flawed, but time constraints prevented a full analysis of the 'impacts' production necessary to test these assumptions.

The SMEP II reports provided median lengths of female and male silver eels in the final year of the simulation. SMEP II does not report the ages of silver eels, but these were inferred by aligning median length on the VBF growth curve used by the model. Neither does SMEP II report the weight range of silver eels, but it does report silver eel production in numbers and biomass, so these were used to estimate average weights of female and male silver eels. SMEP II produced silver eels that were similar in length, age and weight to those in the Crepe data, confirming that the silvering parameters in SMEP II were very close to those applied in Crepe (Table 4.14).

Table 4.14. Mean length, age and weight of female and male silver eels reported by CREPE and SMEP II for the final year of the analysis of the Pristine data set.

		Length (mm)	Age (y)	Weight (g)
Crepe	Females	629	6.6	552
	Males	465	4.4	187
SMEP II	Females	629	7.5	504
	Males	463	4	199

Silver eel escapement accounting for human impacts

The SMEP II conditions selected from the tuning process above, were then used to repeat the model application against the original CREPE data sets of silver eel biomass, to provide the results necessary to test the accuracy and precision of SMEP II to predict historic and present escapement biomass.

Figure 4.36 present the results for silver eel escapement biomass predicted by SMEP II for scenarios where human impacts are present or absent (only to year 75). As the tuning process used a single series of mortality at length parameters, this was applied again here and meant that the results could be presented for the complete 150 year time series. Once again, however, we ignore the first 50 years of results because this was the period of model stabilisation.

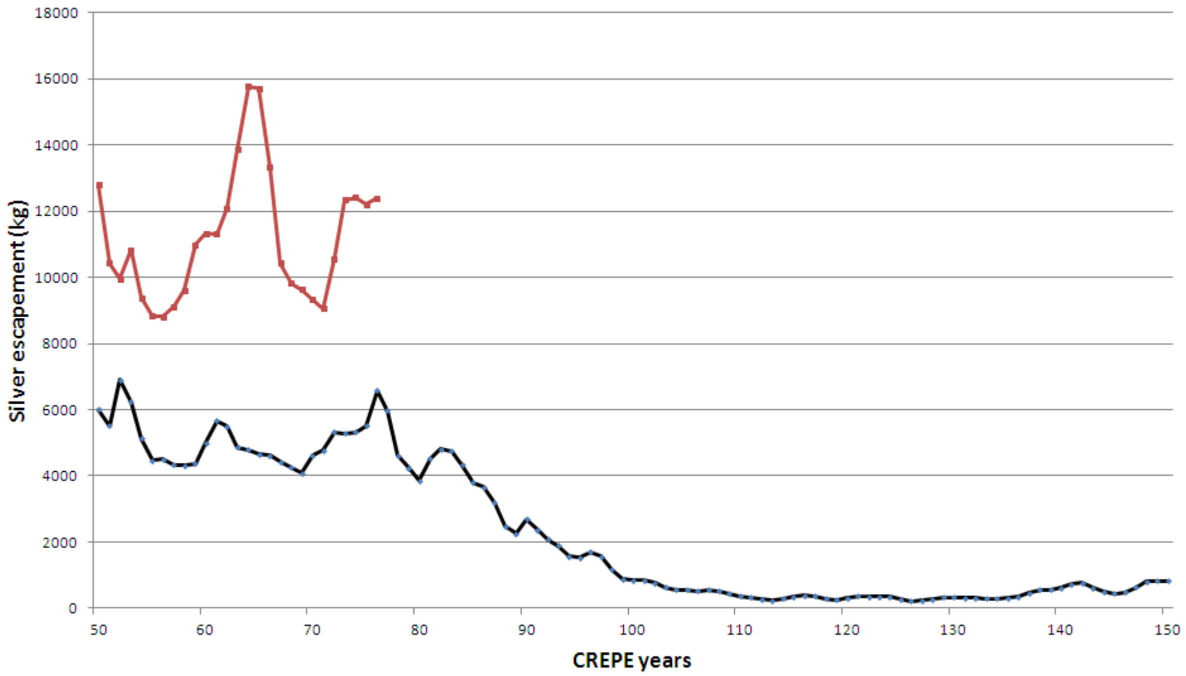


Figure 4.36. Time series of silver eel escapement biomass (kg) predicted by the 'tuned' SMEP II for river 39862 under scenarios where human impacts are present (black) or absent (red).

SMEP II predicts the 'pristine' silver eel escapement from river 39862 under high recruitment would have been about 9,000 to 15,000 kg, whereas the escapement during the same period but including the effects of human impacts would have been about 4,000 to 7,000 kg. During the period of declining, low and then increasing recruitment, which was only modelled including human impacts, SMEP II predicted the escapement biomass would fall to just a few hundred kg and only slowly begin to increase above to reach about 1,000 kg by the end of the time series.

Conclusions

The improvement in results predictions shown in Phase 2 demonstrate that it is possible to tune SMEP II to produce results very close to those of the CREPE dataset. This tuning revealed that SMEP II was most sensitive to rates natural of mortality and particularly those of the youngest eels. This may be due in part because the rates applied in CREPE were especially high for glass eel to elver stage, but it highlights the importance of having good data. Natural mortality data are rare, especially river-specific, and therefore most model applications will use the average values developed by Bevacqua *et al.* (2011). In the absence of better, more site-specific data, those values derived on the basis of Bevacqua *et al.* may be an improvement from using a single default value but the outputs should still be treated with caution. Applicants and assessors must recognise this limitation of the data and the models. To some extent, in the absence of site-specific data, it makes sense to standardise life history parameters across models and across regions so that we are at least all working to the same set of rules.

4.2.4. Application of DemCam to the CREPE data set

Simulations were run for 149 years with two different mortality scenarios varying the density level (high and intermediate) of the relationship used in Bevacqua *et al.* (2011). The first application of DemCam to CREPE was affected by a bug in the length-weight relationship with a propagation of the error to the not performing results.

Input data for the model were derived as follows:

Life history parameters

The *a* and *b* constants for the allometric length (cm) /weight (g) relationship were set as $a = 8.34\text{x}10^{-4}$, $b = 3.17$ from Bevacqua *et al.* (2011).

Assuming that we are dealing with an Atlantic eel stock, the body-growth (Von Bertalanffy's curves for each sexual stage) and sexual maturation parameters were taken from Andrello *et al.,* (2011). Accordingly, the average ages of male and female silver eels were set as 8 and 13 years, respectively. The average lengths of male and female silver eels were set as 398 and 658 mm, respectively.

Recruitment

As recruitment input, the glass eel recruitment data (149 years) was used after having subtracted the glass eel harvest. Pristine condition was assumed as that with the recruitment such that the settlement was equal to S_{max} (settlement carrying capacity) without any fishing pressure.

Carrying capacity

In the second application, the carrying capacity S_{MAX} for was estimated to be equal to 430 elvers per hectare, assuming a high density natural mortality, and equal to 171 elvers per hectare assuming an intermediate density (as first application conditions). These values were obtained by fitting the fishermen's harvest through a non-linear parameter calibration that showed a better fitting when high mortality is used (Figure 4.36).

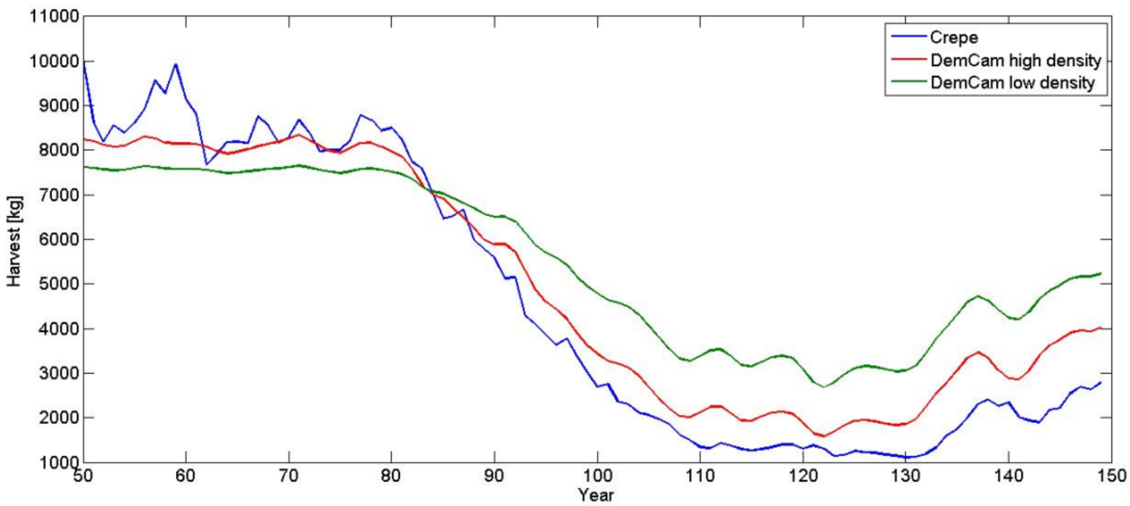


Figure 4.36. Time series of fishermen's harvest from CREPE (blue line) and from DemCam simulations (red line: high density; green line: intermediate density).

Environment

The simulations ware run on a single compartment with a wetted area of 8,140 hectares, which was the surface of habitat downstream of the first dam (~20% of the whole habitat). The annual average water temperature was calculated from the temperature series as 12.6°C.

Impacts

Fishing mortality for yellow eels and silver eels was computed as $F = q \times E \times \varphi$. Fishing effort E was set using a per hectare fishing effort (fixed monthly effort) commonly observed in exploited lagoons (two peaks: spring and fall). Hence, the actual fishing effort applied was equal to approximately 40,000 gear days, which is approximately the effort executed by 10 fishermen, each using 13 fishing gears and operating 300 days per year.

Gear selectivity φ, varying from 0 to 1, was calculated as in Bevacqua *et al.* (2009) by assuming a knot to knot mesh size of 12 mm in order to obtain the same minimum landing size as in CREPE (i.e. ca 30 cm).

The catchability of silver eel was set at 2.8×10^{-7}gear^{-1}day^{-2}, and that of yellow eel was set at 2.3×10^{-7}gear^{-1}day^{-2}, based on per-hectare estimates for other environments (Bevacqua, unpublished data).

We hence obtained, for $\varphi = 1$, $F_{yellow} = 0.3$ yr^{-1} and $F_{silver} = 0.37$ yr^{-1}, that means an annual removal of 26% of yellow eels and 31% of silver eels.

Results

The first application of the DemCam to CREPE data was affect by an error that leads to an overestimation of the standing and the migrating stock. The process of 'calibrating' DemCam to Crepe data, in the second application, was performed under two different hypotheses. The first calibration was run using the intermediate density natural mortality (as first application conditions) that leads to a very low carrying capacity. In this scenario the model reproduces a similar trend of Crepe results, but it initially underestimates and finally overestimates the migrating stock (Figure 4.37, green line) with a silver eel production that ranges between 1.1 and 0.4 kg/ha. This is due to the higher mortality used in CREPE model for undifferentiated eels. In order to incorporate this knowledge, a second calibration process was run using a high density natural mortality. Under this hypothesis the model improved the fitting of fishermen's harvest increasing its performances also in the dynamics of standing and migrating stock with a silver eel escapement that varies between 1.4 kg/ha, with high recruitment, and 0.3 kg/ha, with low recruitment (Figure 4.37, red line).

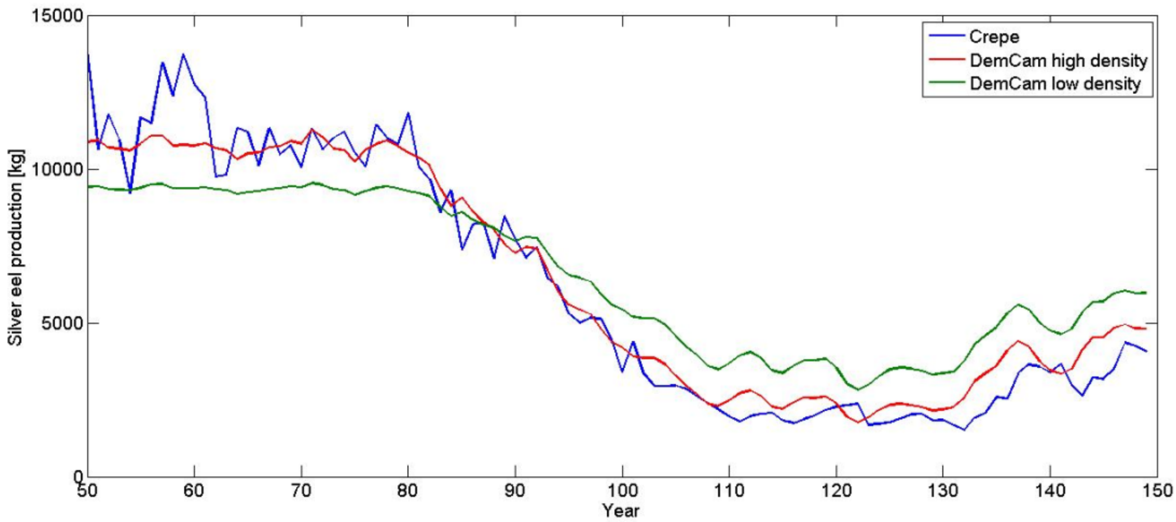


Figure 4.37. Time series of silver eel production from CREPE (blue line) and from DemCam simulations (red line: high density; green line: intermediate density)

4.3 Comparison of EDA, GEM and SMEP II outputs with CREPE virtual dataset

4.3.1. First applications of the assessment models on the CREPE dataset (Phase 1)

The comparison of silver eel escapement simulated by CREPE and first calculated by SMEP II is presented in Figure 4.38. The results are extrapolated to the whole CREPE eel management unit according to the relative catchment area of the basin considered by SMEP. This comparison shows that SMEP II generally overestimated the silver eel production. This overestimate was by a factor of 5 to 9 during the period of high, stable recruitment, but increased to a factor of 10 to 18 during the periods of rapidly changing or low recruitment.

The comparison between CREPE and EDA (Figure 4.39) shows a bias in the EDA predictions corresponding to an underestimate of 6 times (half of this bias probably due to an underestimate of the wetted area of the catchment). The general trend is predicted by EDA but periods of stable escapement (induced by stable recruitment) are difficult to detect. The underestimated yellow eel production is the consequence of a lack of sampling in the largest and most downstream reaching, and this probably reflects reality. Also, a correct estimation of the wetted area requires that data used to predict river width are also based on a random sampling of the river network and not only the river widths at electrofishing sites. Both biases have been put forward in the different EMU where EDA was tested and are seen as very useful to know if we wish to try to enhance EDA prediction in the future.

The silver eel escapement calculated by GEM overestimates that of CREPE by a factor of nearly 40 during the initial period of stable high recruitment, and a factor of about 10 during the latter period of declining and then increasing recruitment, therefore exaggerating the decreasing of escapement (Figure 4.40).

DemCam succeeded in reproducing the global trend of escapement but calculated an escapement about 3 times higher than the CREPE value, and slightly more during the period of low recruitment (Figure 4.41).

Overall, all four models failed to correctly estimate pristine escapement under the Phase 1 test conditions. Therefore, the ratio between pristine and current biomasses, one of the three key points for post-evaluation of management, gave results that should be used with caution (Figure 4.42). SMEP II is "optimistic" with high value of Bcurrent/B_0 and a passage under the threshold of 40% not before year 115 (instead of year 85) and an improvement above the limit already for year 112. EDA is wrong at the beginning of the series (before year 80), but it gives a passage under the 40 % threshold in year 92, not so far from the CREPE value. DemCam overestimates the pristine escapement. In consequence, the Bcurrent/B_0 is pessimistic during the high recruitment. It fits the CREPE value very well after year 100.

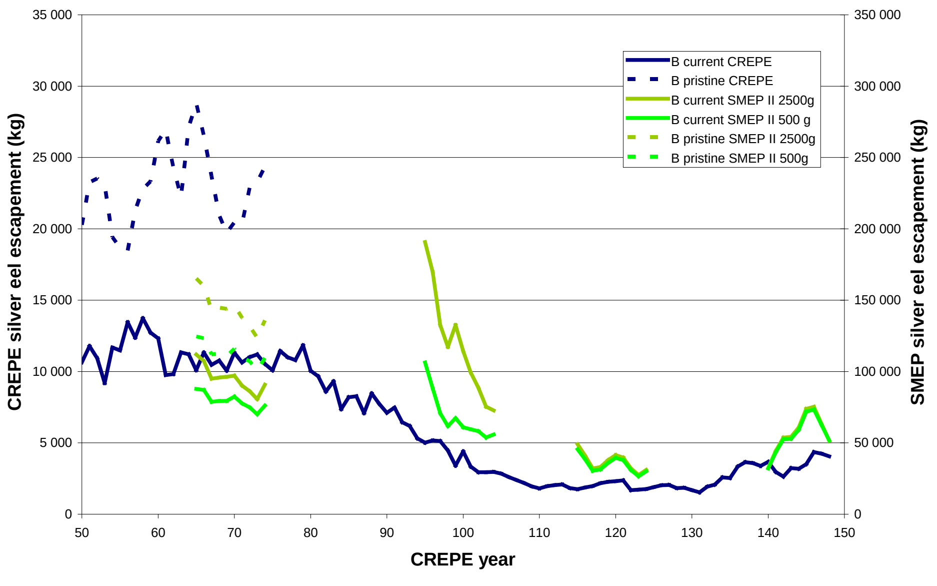


Figure 4.38. Evolution of silver eel escapement simulated by CREPE and calculated by SMEP II in current and pristine conditions

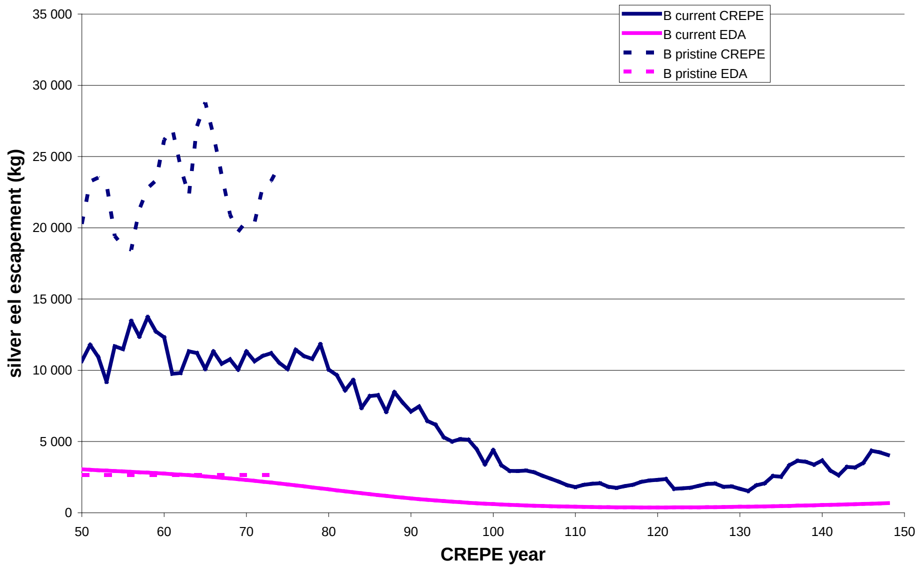


Figure 4.39. Evolution of silver eel escapement simulated by CREPE and calculated by EDA in current and pristine conditions

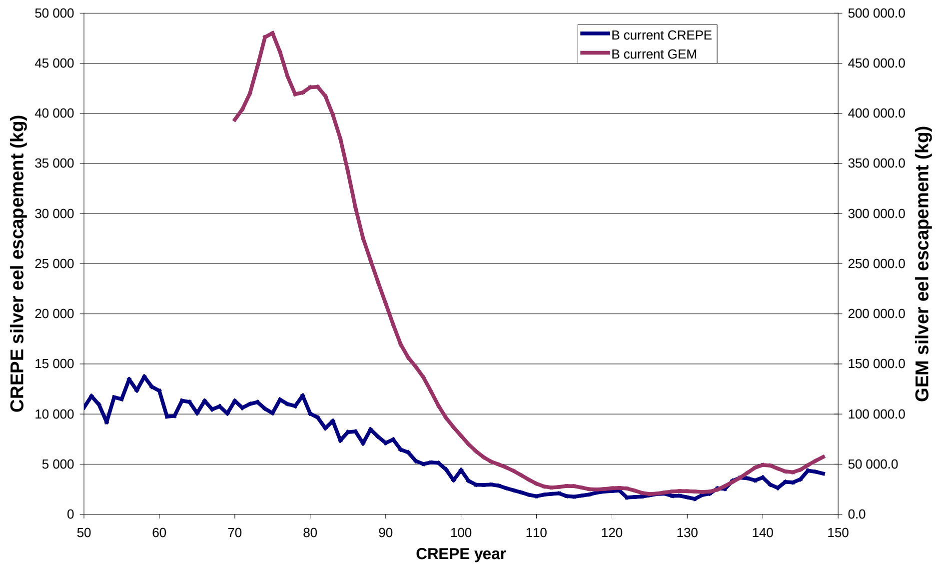


Figure 4.40. Evolution of silver eel escapement simulated by CREPE and calculated by GEM in current conditions

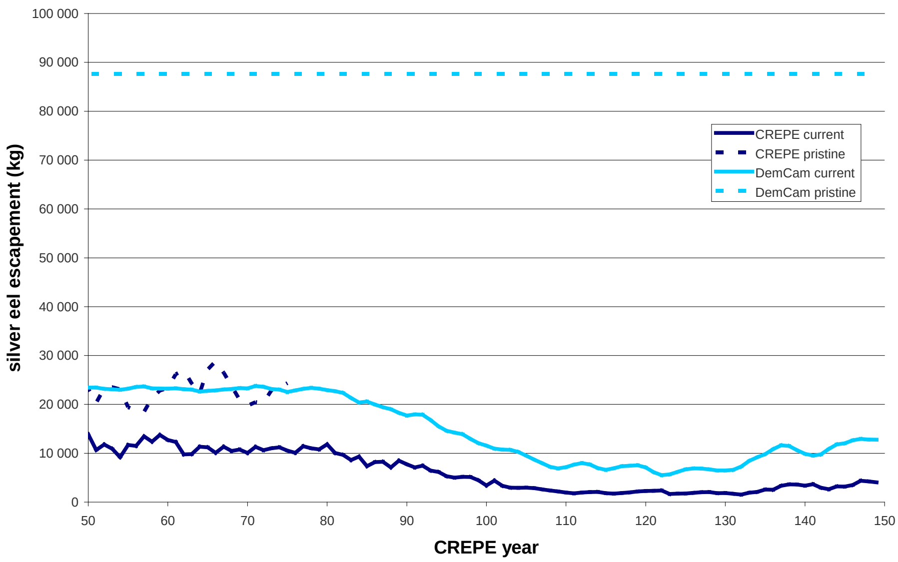


Figure 4.41. Evolution of silver eel escapement simulated by CREPE and calculated by DemCam in current and pristine conditions

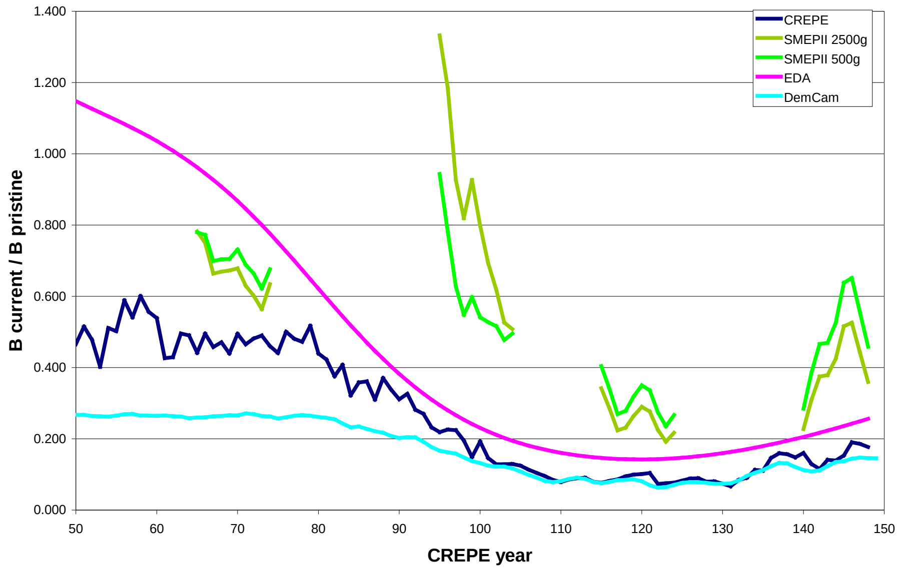


Figure 4.42. Evolution of ratio between current and pristine escaping biomasses simulated by CREPE and calculated by DemCam, EDA and SMEP II.

4.3.2. Phase 2 in assessment model application

The second step corresponded to a phase of further exchange between CREPE data provider and modellers. The aim was to explain the difference between CREPE output and results of assessment models.

Evaluation of the CREPE dataset

When comparing the results of the four models from the first series of test applications, we identify three specifics or originalities in the CREPE dataset which could explain the difficulties encountered.

First, the eel distribution in CREPE environment was concentrated in the downstream part of the catchments. This spatial structure is observed in actual basins but could be exaggerated by the strictly diffuse colonisation for yellow eel in CREPE and the use of super-individuals as computer coding technics. This downstream concentration complicated the application of single-compartment assessment models as GEM and DemCam. In contrast, the multiple compartment approach of SMEP II was able to 'handle' this spatial complexity.

Second, the growth rates of CREPE eels were relatively high since they corresponded to a situation in north of the Gulf of Biscay. More problematic is the lognormal variability used in the growth rate calculation that led to exceptionally high growth rates for some fish (Figure 4.43).

Furthermore, the growth rates derived from biological data from electro fishing surveys gave results similar to the original growth rates programmed in CREPE (Figure 4.44). However, the lengths at age data from the yellow eel fishery survey showed higher rates for younger eels due to the selectivity of the gear. Therefore, models that only used the fishery data to derive growth rates would inevitably introduce errors in growth rates.

Figure 4.43. Theoretical variability growth rate for 1000 fish with CREPE parameters.

Figure 4.44. Length at age in CREPE electro-fishing survey dataset.

Third, the mortality rate used in the CREPE dataset is high, probably too high to be caught by classical approach in assessment models (Figures 4.45 to 4.47). This overestimation is a consequence of (i) the amplification by two successive exponentials of the small fitting errors in interpretation of Bevacqua *et al.* (2011) results (see Figure B6 in annex B), (ii) a choice of a too high value for undifferentiated eel that dramatically collapsed the cohort abundance in the few first years of life. This point could be considered outside of eel ecology features but also highlighted the need of procedure to identify extreme cases that do not correspond to simple application with default parameters of the assessment model. The parameterisation of this mortality process should probably be corrected for the next simulations. Note however that those applying DemCam, GEM and SMEP II all identified that this early mortality rate was much higher than could be expected from the CREPE information.

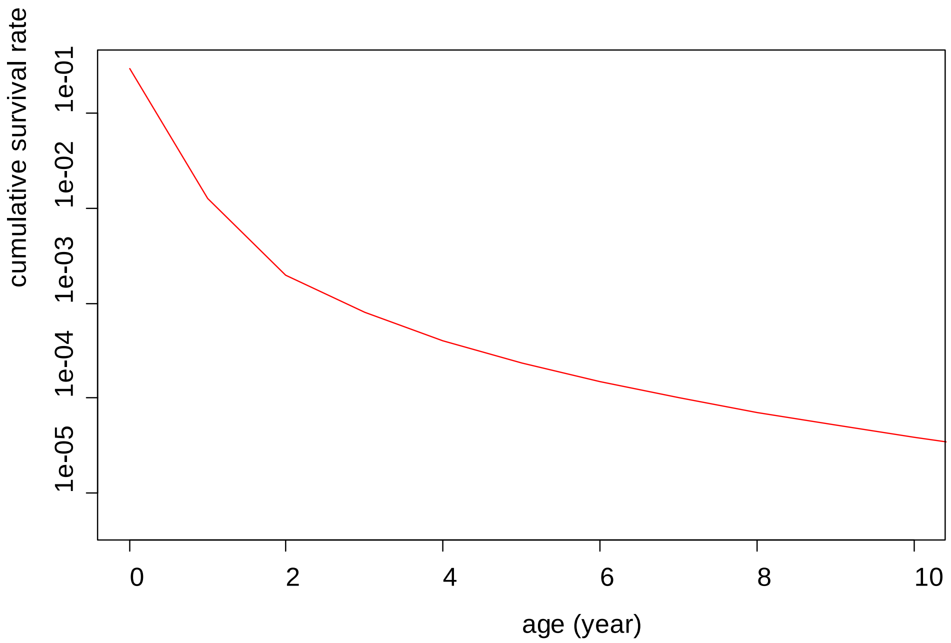


Figure 4.45. Comparison of cumulative survival rate (in logarithm scale) between Bevacqua et al. (2011) model and CREPE simulation in high density condition (saturation of the carrying capacity of 90 % in CREPE).

Figure 4.46. Comparison of cumulative survival rate (in logarithm scale) between Bevacqua et al. (2011) model and CREPE simulation in medium density condition (saturation of the carrying capacity of 50 % in CREPE).

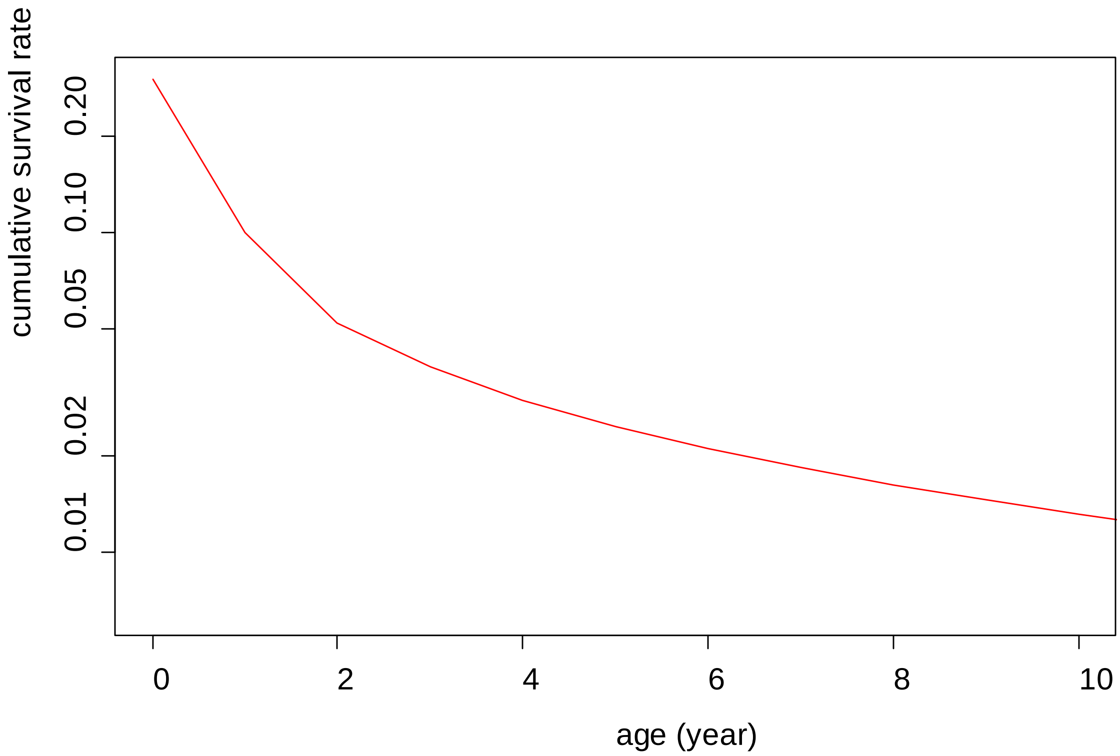


Figure 4.47. Comparison of cumulative survival rate (in logarithm scale) between Bevacqua et al. (2010) model and CREPE simulation in low density condition (saturation of the carrying capacity of 10 % in CREPE)

Results of the Phase 2 application of the assessment models to CREPE data

Knowledge of these features of CREPE significantly improved the performance of the assessment models in the second phase of testing under some conditions, showing a clear convergence towards the CREPE output (Figure 4.48 to Figure 4.52).

Although DemCam overestimated pristine escapement by a factor 2, it predicted escapement accounting for human impacts to within, on average, 9% at high and 13% at low recruitments (Figure 4.48). SMEP II also predicted silver eel escapements that were, on average, within 9% of pristine and 16% of 'current' escapement from CREPE under high recruitment conditions (Figure 4.49). However, SMEP II underestimated silver eel escapement under low recruitment, being on average only 32% of the CREPE result. This was probably because the whole time series was modelled using the mortality rate curve associated with high densities. A low density mortality rate curve might have been more appropriate for the period of low recruitment, but conversely would have likely resulted in an overestimate of escapement when recruitment was high. EDA overestimated pristine escapement by a factor of 3, but was much better at estimating escapement under 'current' conditions when taking human impacts into account: producing results within, on average, 24% of CREPE for the high recruitment period, and within 57% under the low recruitment conditions (Figure 4.50). The GEM results were also considerably improved over those from Phase 1, but still overestimated escapement by a factor of about 10 at high recruitment, and about 3 at low recruitment (Figure 4.51).

The ratio of current/pristine is the primary reference target of the Eel Regulation (EC 1100/2007) – the 40% or 0.4. This ratio predicted by SMEP II during the period of high recruitment was very close to that of CREPE (Figure 4.52). In contrast, EDA and DemCam both underestimated this ratio considerably during this period (Figure 4.52), but this is probably because both these models derived pristine biomass as the maximum potential biomass from an excess of recruitment, rather than escapement under a high but not necessarily maximum recruitment described for CREPE. The second application of GEM did not include an estimate of pristine escapement so this ratio could not be calculated for this model.

Given this overestimate of pristine biomass, it is hardly surprising that DemCam and EDA underestimate the ratio of pristine/current silver eel escapement during the period of low recruitment. SMEP II also underestimates this ratio, but this is probably because a high rate of natural mortality was applied in SMEP II and this meant it underestimated the current escapement during this period. This underestimation during the period of low recruitment is to err on the side of caution for the assessment and management of the eel stock as it would probably lead to extra management measures to protect and recover the stock. However, there are potentially social and economic consequences of this cautious approach by inducing management actions which are possibly more than severe than required to achieve the target.

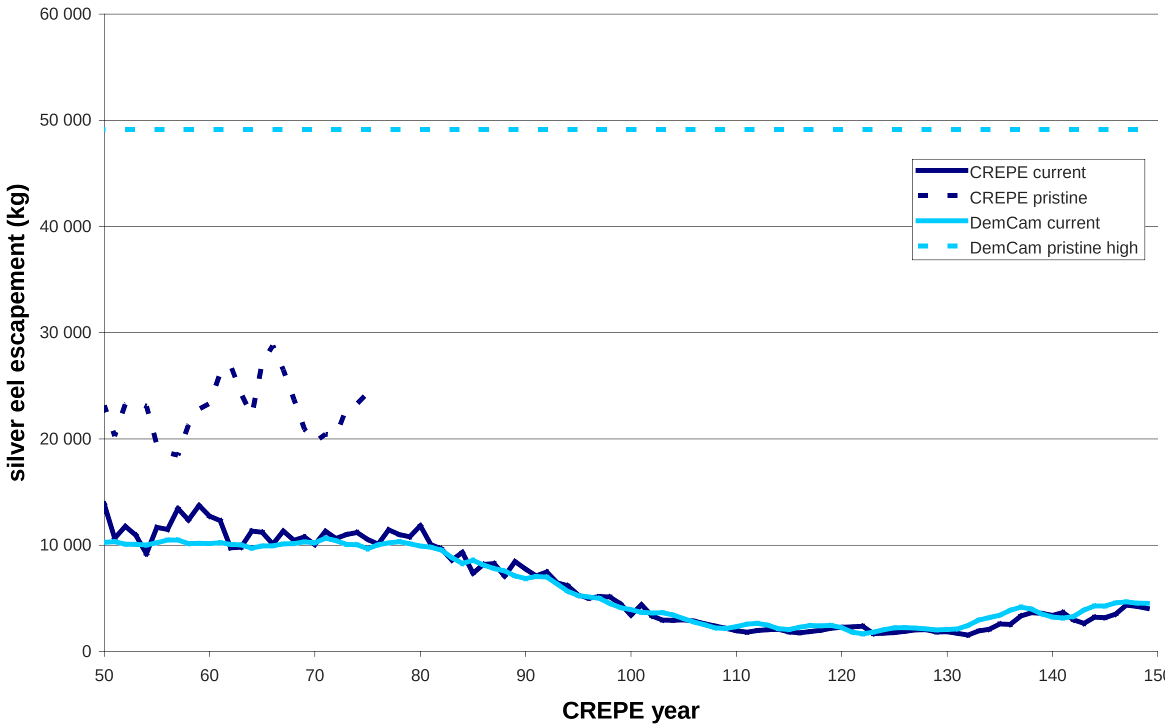


Figure 4.48. Evolution of silver eel escapement simulated by CREPE and calculated by DemCam in current and pristine conditions after the second step of calibration

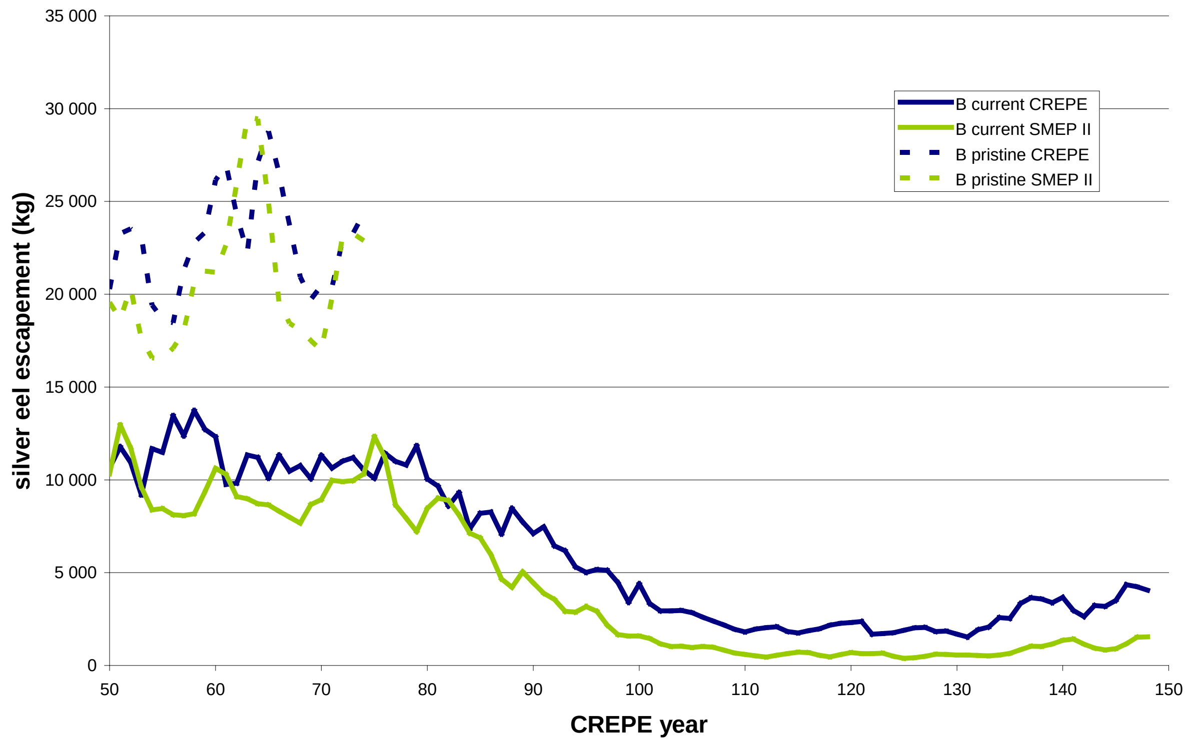


Figure 4.49. Evolution of silver eel escapement simulated by CREPE and calculated by SMEP II in current and pristine conditions after the second step of calibration

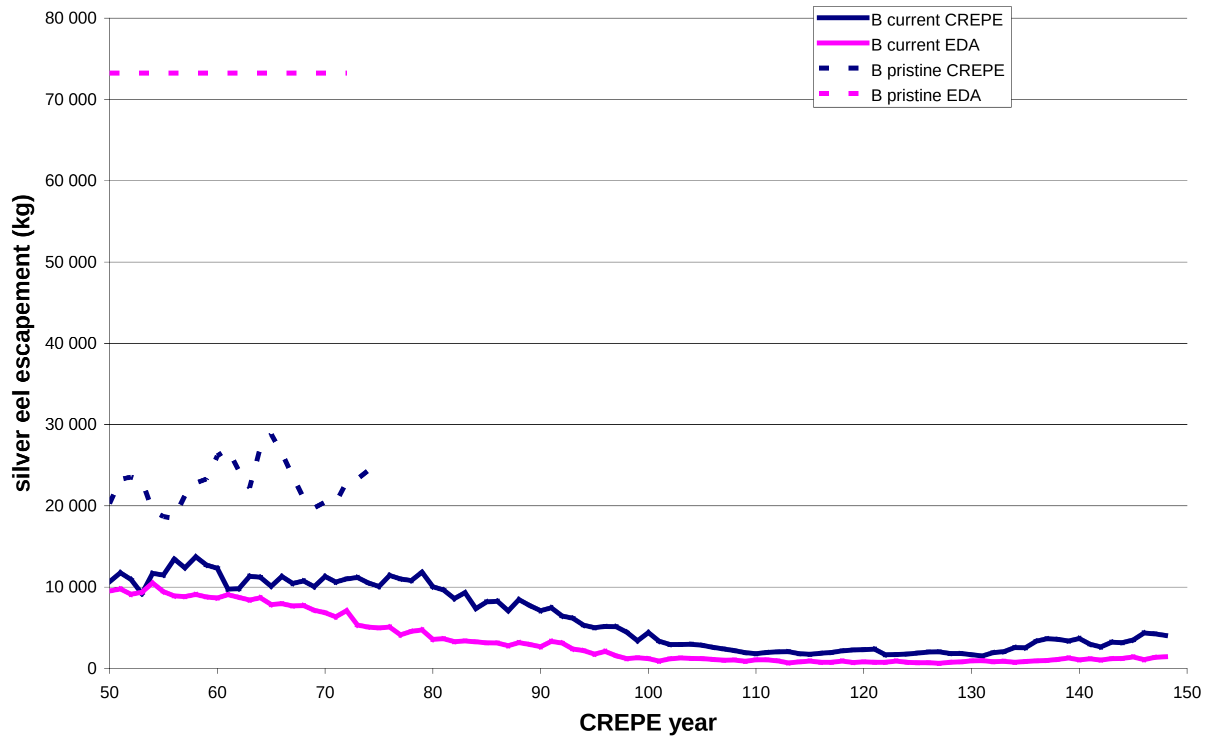


Figure 4.50. Evolution of silver eel escapement simulated by CREPE and calculated by EDA in current and pristine conditions after the second step of calibration

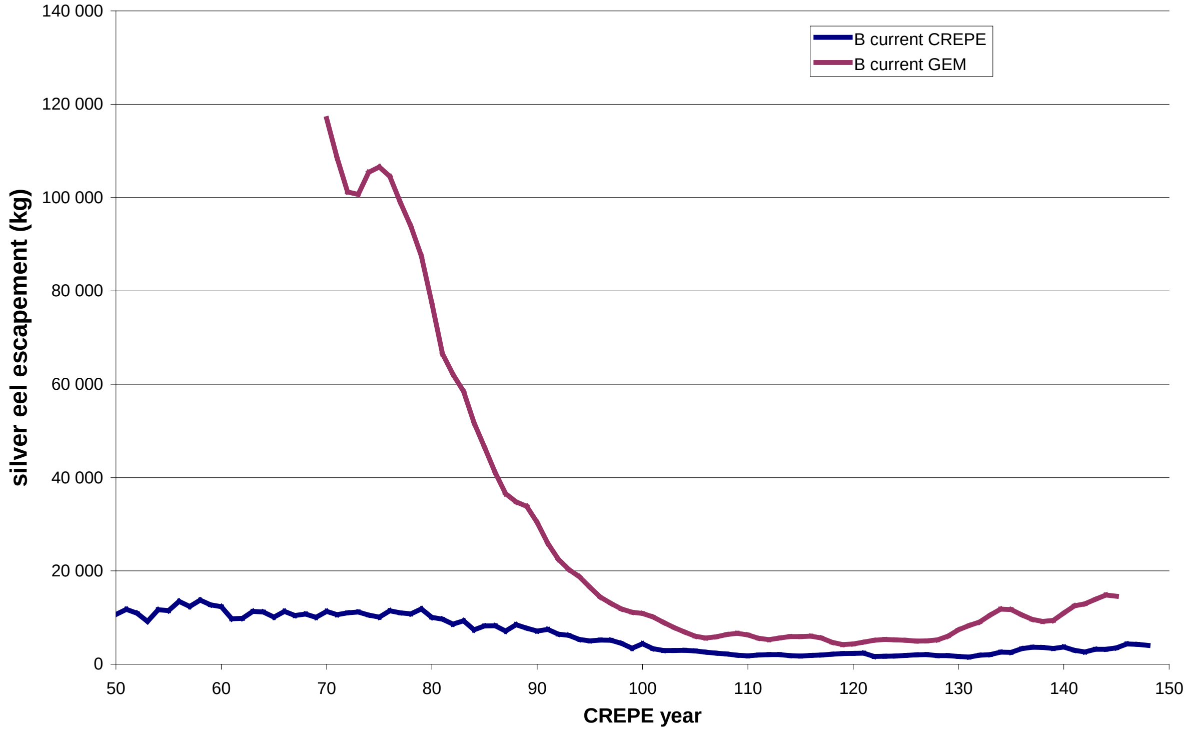


Figure 4.51. Evolution of silver eel escapement simulated by CREPE and calculated by GEM in current conditions after the second step of calibration

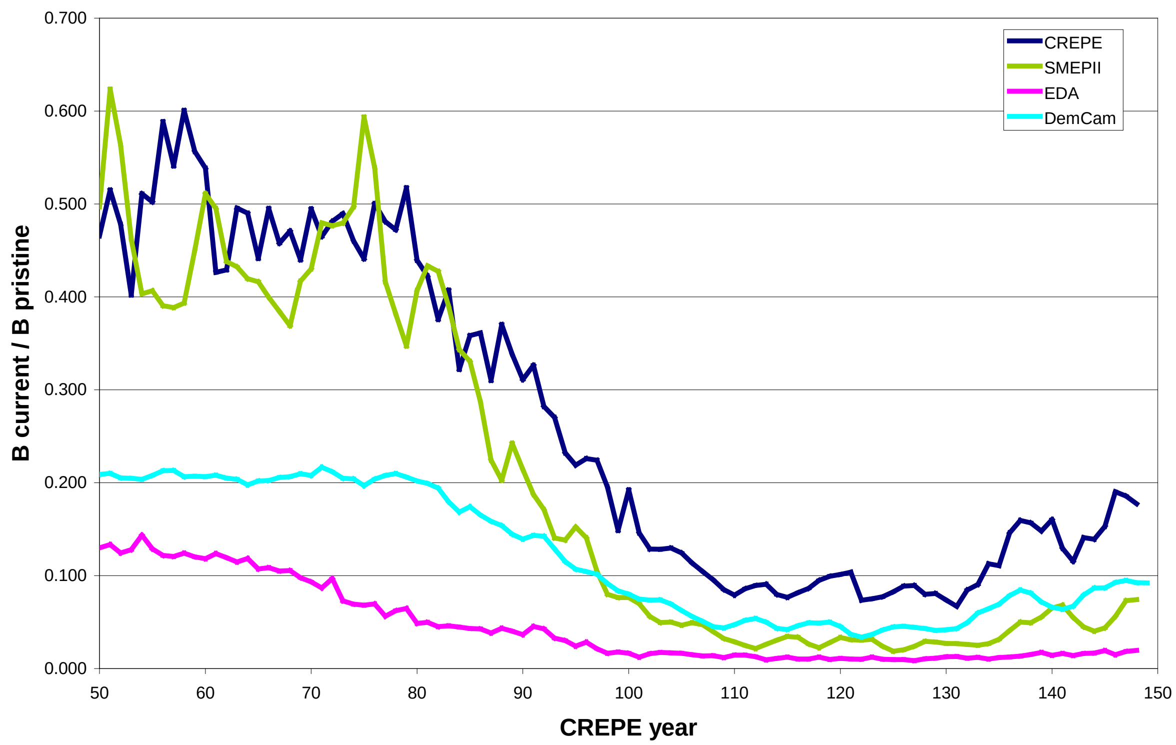


Figure 4.52. Evolution of ratio between current and pristine escaping biomasses simulated by CREPE and calculated by SMEP II and EDA after the second step of calibration.

These results prove that the four models each have the potential to calculate an escapement close to the CREPE value providing the information available in the virtual fully- data -rich situation as in CREPE environment. However, the transposition in the real world should not be so optimistic when most of the time only default values (as Bevacqua *et al.* 2011 mortality rates) are available.

Natural mortality data are rare, especially river-specific, and therefore most model applications will use the average values developed by Bevacqua *et al.* (2011). In the absence of better, more site-specific data, those values derived on the basis of Bevacqua *et al.* may be an improvement from using a single default value but the outputs should still be treated with caution. Applicants and assessors must recognise this limitation of the data and the models. To some extent, in the absence of site-specific data, it makes sense to standardise life history parameters across models and across regions so that we are at least all working to the same set of rules.

In addition to the obvious requirement to represent local natural mortality and growth rates, accurately quantified time series of recruitment were important to DemCam, GEM and SMEP II. This observation supports recommendations from WGEEL that existing recruitment time series should be protected and new recruitment sampling programmes should be implemented across the eel-producing range.

It is interesting to note that some of the model applications introduced biological processes (natural mortality with compensation for settled eels) or differences (growth between male and female) that did not exist in the CREPE simulation. That proves (again) that different models with different and not necessary consistent hypotheses can produce similar outputs. The consequences for management should be considered – in the real world there may not be on.

4.3.3 What are the accuracy and precision of each model?

Accuracy is a measure of how close the predicted result was to the actual result. The Phase 2 results showed that three of the four models can be quite accurate when given sufficient input and tuning data, reaching average levels within 9% (DemCam, SMEP II), 24% (EDA). GEM was less accurate, a best overestimating silver eel biomass by, on average, a factor of about 3. This possibly suggests that GEM was less flexible than the other models to eel production values that were far outside those of its development data set from the River Elbe. A new CREPE data set based on eel production characteristics from northern Europe would probably have produced more accurate results from GEM. It will be useful in the future to use CREPE to simulate several new datasets corresponding to different biological hypotheses of eel population functioning and different management scenarios in order to test the robustness of the assessment model and their interest in a management process, but this was not practicable within the POSE project.

The classical definition of the precision of a measure is the refinement of the measurement, typically in terms of the number of digits expressed. All four models presented results for silver eel escapement biomass to the nearest kg or even g, so in strictest terms, the models could all be considered to report results with high precision (the detail of the answer). However, although reporting to the nearest kg may seem an attractive quality of any model, it is a potentially misleading result, giving the perhaps false impression that the result has a high degree of confidence. A much more meaningful test of precision, which gives a proper measure of the confidence associated with the result, is the measure of the uncertainty associated with this result, e.g. we are 95% sure that the actual result lies between X and Y kg. None of the model results reported results with any degree of uncertainty, and therefore their precision could not be determined.

4.4. Description of the Burrishoole dataset

The Burrishoole data from the Western River Basin District of Ireland provided the ‘real’ scenario closest to ‘rich’ because of the count and measure of emigrating silver eel since the early 1970s.

The Burrishoole freshwater catchment (Figure 4.53) has a total area of 89.5 km^2. It contains 45 km of small shallow streams and three main lakes, brackish Lough Furnace (141 ha) and two freshwater lakes, Lough Feeagh (410 ha) and Bunaveela Lough (46 ha). The catchment is composed primarily of dalriadan schists, gneiss and quartzites and the poor buffering capacity of the catchment leads to low alkalinity values particularly in the western streams and loughs. Land use changes in the last 60 years, including the commercial planting of coniferous forestry and sheep grazing has lead to significant peat erosion in the catchment, and with resulting nutrient enrichment of receiving waters (Dalton *et al.,* 2010). Exploitation of eels does not occur on the Burrishoole catchment.

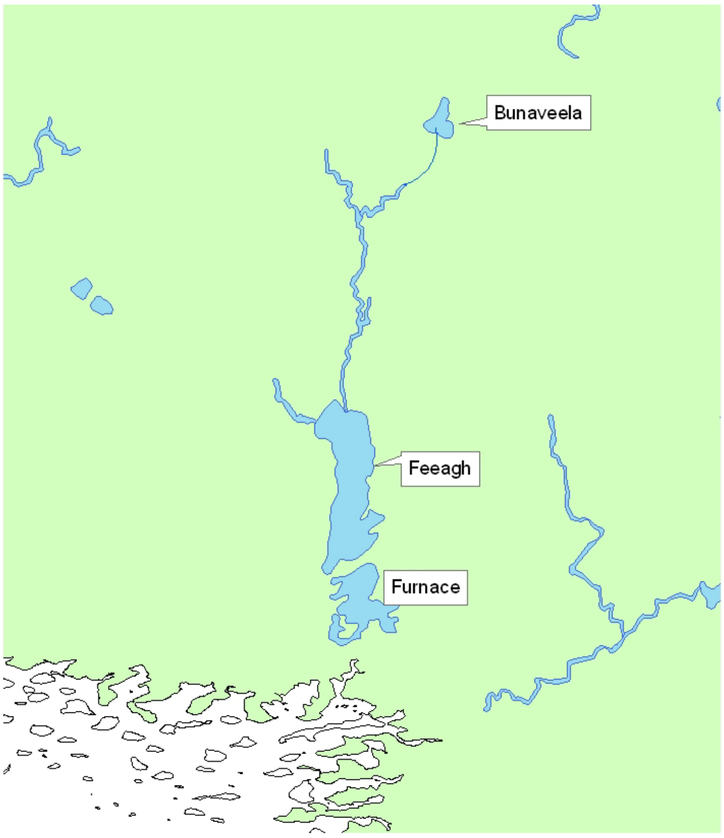


Figure 4.53. Map of the Burrishoole catchment. Note that this map illustrates the data available from the CCM data set and therefore does not show the lower order streams, of which there are many in this catchment

Little quantitative data on elver migrations are available. However, a total of 62.9 kg of elvers were trapped in 1980 and this weight was estimated to be only a fraction of the total run. The catches in 1982 were extremely poor and continued to be low for the remainder of the 1980's: 1.2 kg in 1987 and 14.6 kg in 1988, reflecting a reduction in catch of between 70 and 95% (Poole, 1994).

A complete yellow (brown) eel survey dataset exists for the Burrishoole from 1986 to 1989 with additional information for 1973 and post 1990. There is some evidence that eel in Lough Feeagh (freshwater) have been increasing in size, and maybe in density. The increase in size may be due to falling recruitment. The change in density is more difficult to explain, but the 1973 information is based on a "one-off" survey. Smaller eels in 1973 would also have been less well represented in fyke net catch.

Recaptures over an 18 year period have verified age and growth determinations based on otolith readings.

Data available for the Burrishoole include:

- Total annual silver eel escapement from freshwater since 1970, total numbers, sex ratio, sizes
- Length, weight, age, sex for yellow eels in all six lakes and the estuary in 1987 and 1988
- Fyke net CPUE, and mark recapture data for 1987 and 1988 in the lakes
- Electrofishing data for stream sites since 1987
- Age and growth data for yellow eels and silver eels, 1987 & 1988, and from 1990.
- Some age data and CPUE from the 1960s and 1970s
- CPUE, length and age data from 2005-2009 to compare with 1987 & 1988

Annual monitoring of silver eel takes place in the Burrishoole catchment. It is the only catchment in Ireland that gives a complete assessment of silver eel production (Poole *et al*., 1990; Poole *et al*., 1996).

Counts of silver eel between 1971 (when records began) and 1982 averaged 4,400. They dropped to 2,200 between 1983 and 1989 and increased again to above 3,000 in the 1990s (Figure 4.54). The average weight of the eels in the run has been steadily increasing from 95 g in the early 1970s to 215 g in the 1990s (Figure 4.54). The increase in average weight has been caused in part by a change from a predominantly male sex ratio to more than 60% females (Poole *et al*., 1990) and in part by an increase in the mean size of females from 52 cm in 1987 to 55 cm in 2005 without any change in their mean age (Table 4.15). Both factors may indicate a decrease in yellow eel population density. Burrishoole eels are generally considered relatively old and slow growing, typical of oligotrophic Irish waters. Growth rates in the more productive waters in Ireland are generally faster than observed in Burrishoole.

However, there are no quantitative estimates of recruitment for any year or time series, and only a semi-quantitative estimate of trend over the 1980s.

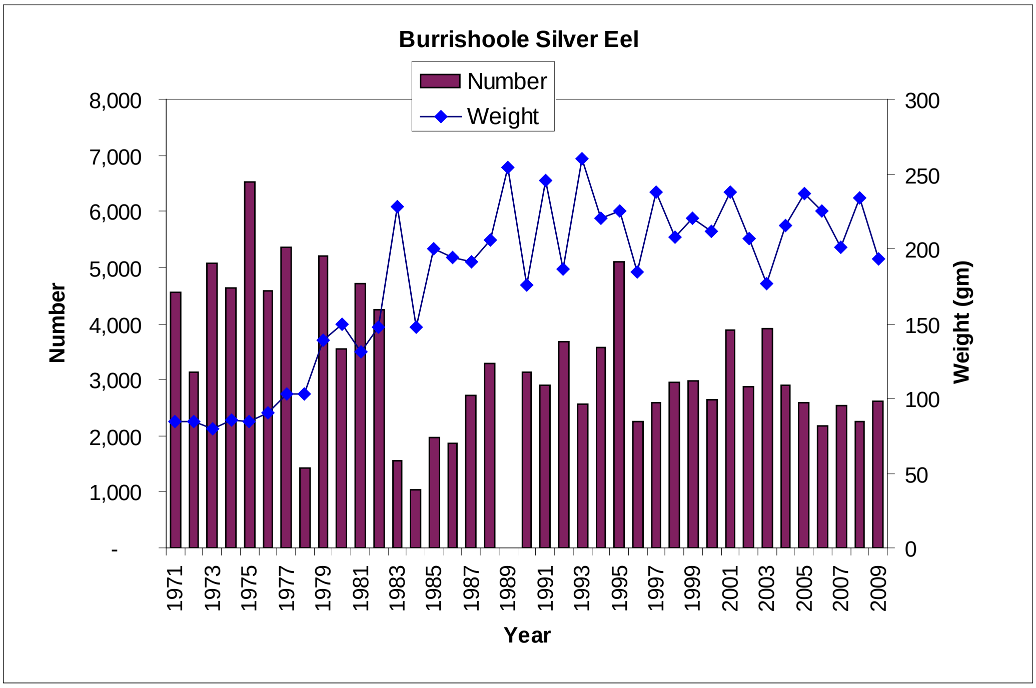


Figure 4.55. Annual silver eel catch in the Burrishoole System for 1971 to 2009.

Table 4.15. Summary statistics for the Burrishoole silver eel census showing pre 1980 and post 1995 silver eel numbers, biomass and production figures. Also included are the average number of females and average biomass of females for the same periods.

Silver Eel	1971-1980	1995-2008
Average count	4409	2808
Biomass (kg)	436	602
Production (kg/ha)	0.9	1.3
Number of females	1626	1932
Biomass of Females	329	529

4.5. Application of the models to the Burrishoole dataset

The DemCam and SMEP II models were applied to the Burrishoole dataset to test whether or not they could be parameterised (tuned) in such a way as to predict a silver eel production time series similar to that observed (i.e. test their accuracy), and to identify those parameters (data requirements) that had the greatest influence in their accuracy. As such, the application of these models required a series of iterations and associated discussions between the modellers and data providers in order to identify the most appropriate parameters.

As EDA is most suited to the application to entire RBDs or EMUs, it was not applied specifically to the Burrishoole dataset. The application of EDA to the broader, Western River Basin District is described in Chapter 5. However, the area-specific production rate for silver eel in across the Western River

Basin District has been extrapolated to the wetted area of the Burrishoole, to provide an estimate of present day silver eel production, in order to compare with the results of DemCam and SMEP II. Note however that EDA does not at present take into account lakes or lower order streams, both of which may have eel production characteristics quite different from those river areas that can be effectively surveyed for eels. As such, this extrapolation is difficult and must be viewed with great caution.

GEM was not applied to the Burrishoole dataset but rather, was applied to the neighbouring Corrib basin. This application and the results are described in Chapter 5.

4.5.1. Application of (DEMCAM) to the Burrishoole data set

Simulations were run from 1900 to 2010. Input data for the model were derived as follows:

Life history parameters

The a and b constants for the allometric length (cm) /weight (g) relationship were set as a = 8.34×10^{-4}, b = 3.17.

The average age of male and female silver eels were set as 17.5 and 26 years, respectively. The average lengths of male and female silver eels were set as 357 and 468 mm, respectively.

Recruitment

There are no quantitative or time series data on recruitment of eel to the Burrishoole river basin. In 1980, a trap caught 63 kg of glass eel migrating upstream, but there is no information to suggest what proportion of the total recruitment was represented by this catch. We arbitrarily assumed therefore that pristine glass eel recruitment was equal to about 79 kg, which is equivalent to about 500 individuals per hectare.

Anecdotal evidence suggests that the trend in the Shannon closely reflects the trend in WRBD rivers (Russell Poole, pers. comm.). Records for the Shannon catch (Ardnacrusha) began in 1977 and peaked at 6700 kg in 1979. A time series index has been produced according to the Shannon data, on the assumption that recruitment was at a constant high before 1980, and then followed the index of the Shannon catch for more recent years.

The average weight of glass eel recruits was set as 0.33 g, on advice from Russell Poole.

Carrying capacity

The carrying capacity for elvers was set at 1000 elvers per hectare, after a series of trials revealed this level to produce sex ratios and silver eel production values close to those observed in the Burrishoole.

Environment

The simulation was run on a single compartment with a wetted area of 474 hectares, which was the wetted area of habitat upstream of the silver eel traps (McGinnity *et al*., in press). The annual average water temperature was set at 10.5 C, on advice from Elvira de Eyto.

Impacts

No fishing or other anthropogenic impacts affect the Burrishoole eel population so none were modelled in DEMCAM. Similarly, no stocking has taken place.

Results

The results of the DEMCAM simulations are presented in Figures 4.56 to 4.58, which illustrate the predicted time series of glass eel recruitment, silver eel production and yellow eel standing stock.

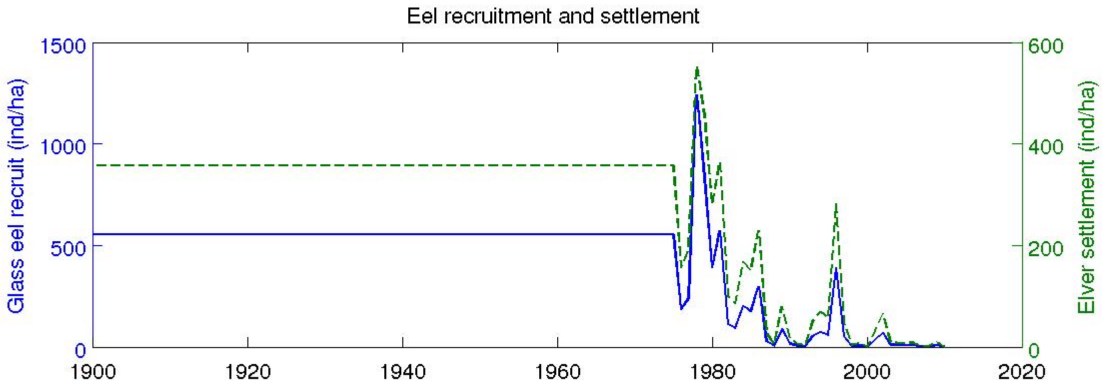


Figure 4.56. DemCam simulation of glass eel recruitment and elver settlement for the Burrishoole. Note that the two y-axes show different scales, though both present individuals per hectare.

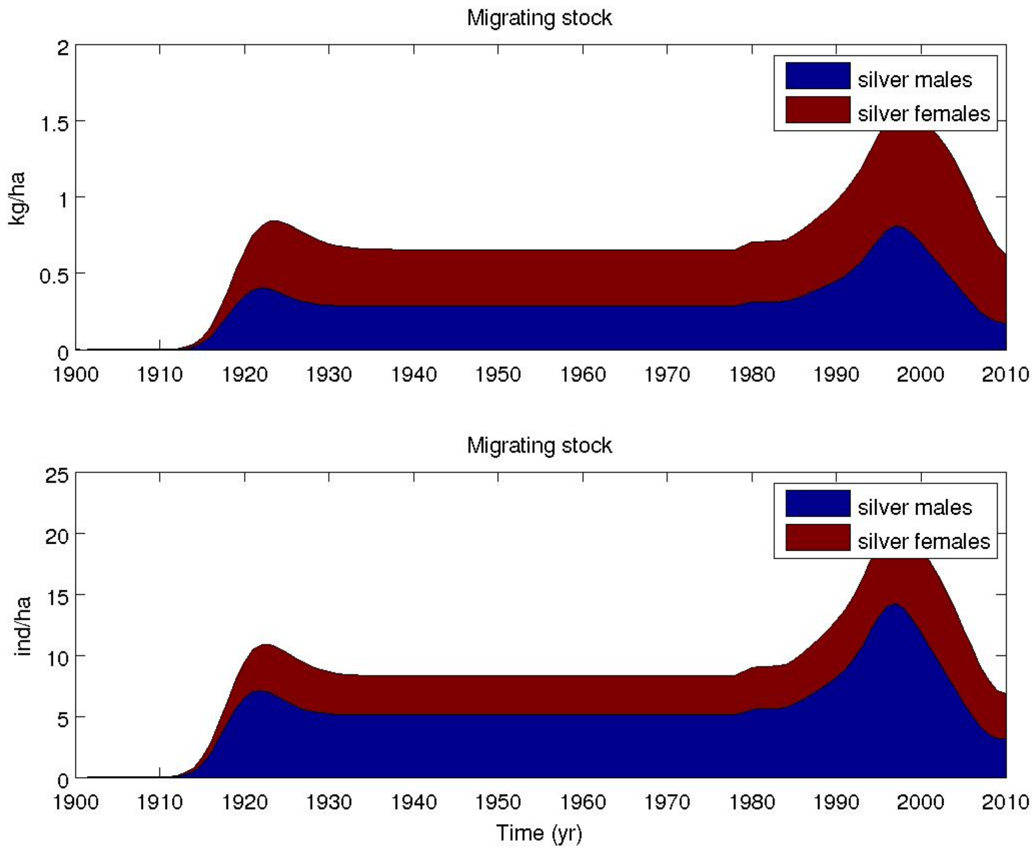


Figure 4.57. DemCam simulation of silver eel production in the Burrishoole. The top chart shows silver eel production for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

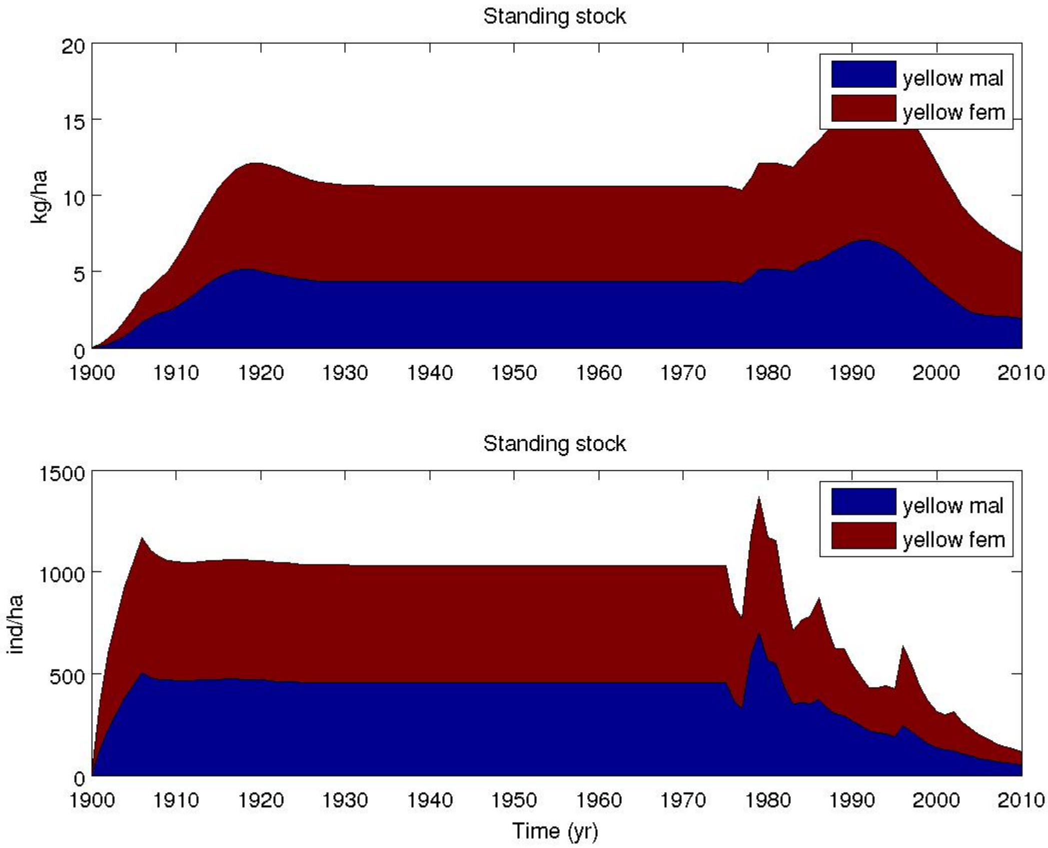


Figure 4.58. DemCam simulation of standing stock of yellow eel in the Burrishoole. The top chart shows yellow eel stock for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

The effect of the decrease in glass eel recruitment (Figure 4.56) is initially masked in the silver eel output biomass (Figure 4.57) by density dependent compensation effects which result in lower levels of natural mortality and in sex ratio skewed towards female silver eels. However, after a predicted peak in silver eel escapement in 2000 as a result of the high recruitment in the early 1980s, the yellow eel standing stock (Fig 4.58) and silver eel production rapidly decline to the point in 2010 where the predicted silver eel production biomass (0.62 kg/ha) is very close to that predicted for the pre-1970 period.

Considerations for the model application

The process of 'tuning' DemCam to the eel data from the Burrishoole revealed that the simulations were sensitive to the parameterisation of recruitment, both in terms of pristine level and time series, and the selection of an appropriate carrying capacity.

4.5.2. Application of SMEP II to the Burrishoole data

Input data for the SMEP II were derived as follows:

Growth Rates

Growth rates derived from annual increment data provided by Russell Poole (RP). Rates for undifferentiated eels (Undiff) derived from the mean of all 1st yr mean increments from FW samples (28.6 mm). Rates for male and female yellow eels derived from mean increments from same samples (16.3 mm). Means rounded to nearest 10 mm to suit length class requirements, so undifferentiated growth modelled as 30 mm per annum, and male and female yellow growths modelled as 20 mm per annum.

Length weight constants were supplied by RP from Burrishoole lake eel data. These constants were originally derived for length (cm) and weight (g), whereas SMEP II models length in mm. Therefore, the constant (a) was converted using a rule from www.FishBase to derive the 'a' value appropriate for length (mm). A = 0.00000239; b = 2.949.

Carrying Capacity

The **Maximum Biomass per unit area** parameters is required by SMEP II to set the threshold for the effects of density dependence, based on the assumption that density dependence has a significant effect on life processes at biomasses approaching and exceeding this threshold. Data from electro-fishing surveys only provided measures of the lengths of eels, so their weights were estimated by applying the Length-Weight relationship provided by RP for the eels from Burrishoole lakes. These weights were summed and then converted to a biomass per 100m^2 based on the wetted area of each survey location. The maximum biomass in the rivers was 843 g, which was derived from the 1991 catch of the Stream B site (code MI0106). Note however that these electrofishing surveys were only possible in the shallower parts of the rivers, and therefore no quantitative estimates of eel density were available for the deeper parts of the rivers or for the lakes.

Note that unlike the density estimates used below, that were raised from catch to local population based on catch depletion analysis, these biomass estimates were based on the eel catch alone. We chose not to assign lengths to the 'extra' eels in the local population estimates, though this could have been achieved by assuming they were of average length for the catch.

An alternative approach based on directly measured biomass data would have been to use the maximum biomass per 100m^2 measured in WFD surveys for the WRBD. None of these surveys took place in the Burrishoole basin so we would have to assume that whatever factors limit maximum biomass per unit area ought to be similar across the RBD. The maximum biomass per unit area reported in the WFD rivers surveys in 2008 or 2009 was 492 g per 100m^2. One issue with these data, however, is that they are from recent years and local densities might be lower than true maxima if they were affected by the decline in recruitment. The Burrishoole electrofishing data date back as far as 1991, so the first approach was favoured.

Recruitment

A time series of annual numbers of eel recruiting to the basin is required in order to predict changes in production over time. However, there are no data on recruitment of eel to the Burrishoole river basin. In 1980, a trap caught 63 kg of glass eel migrating upstream, but there is no information to suggest what proportion of the total recruitment was represented by this catch. The catch reduced by between 70 and 95% between 1980 and 1987/88 and is thought to have remained low ever since. At 3000 glass eel per kg, this 1980 catch represents 189,000 glass eels.

Anecdotal evidence suggests that the trend in the Shannon closely reflects the trend in WRBD rivers (Russell Poole, pers. comm.). Records for the Shannon catch (Ardnacrusha) began in 1977 and peaked at 6700 kg in 1979. At 3000 glass eel per kg, the 6700 kg represents 20,100,000 glass eels. A time series index has been produced from 1977 to 2010, as a proportion of this 1979 level (Figure 4.59).

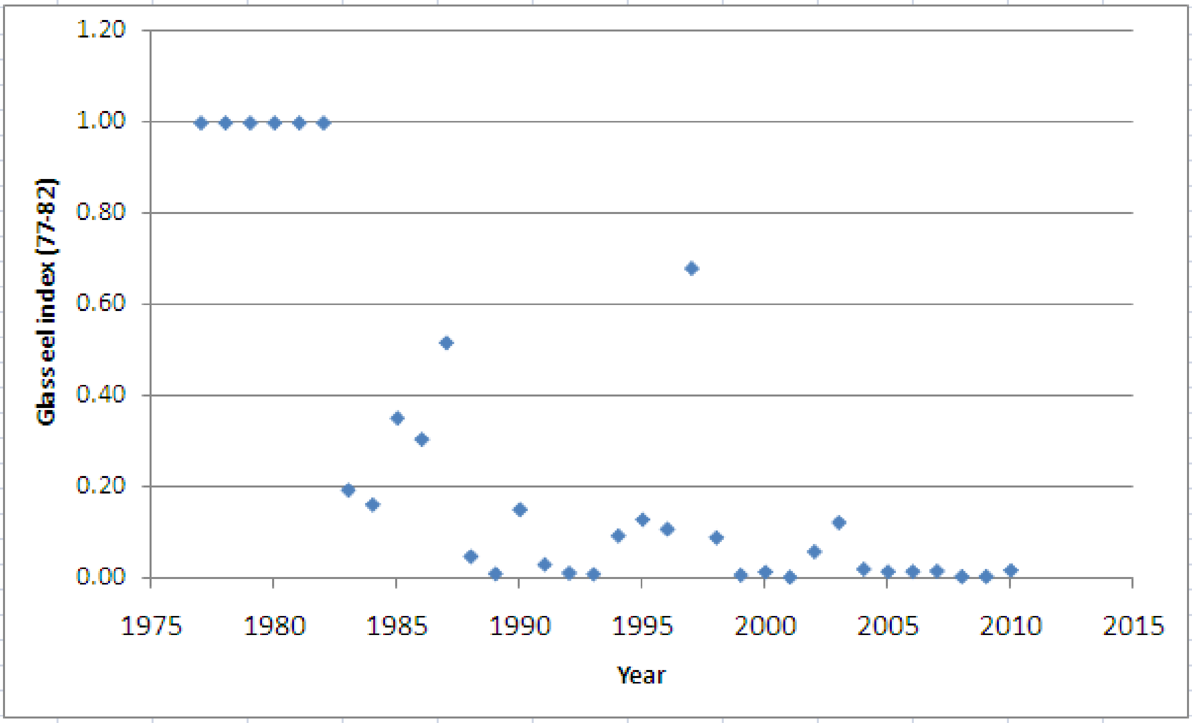


Figure 4.59. Scientific surveys of glass eel recruitment to the Shannon river basin, expressed as an index of the average catch from 1977 to 1982, just prior to the start of the EU-wide rapid decline in glass eel indices.

There is no scientific evidence to suggest that relative recruitment might be proportional to the wetted area of the river basin. However, the Shannon has a total wetted area of 42,465 ha, whereas the Burrishoole has a total wetted area of about 474 ha, or about 1.1% that of the Shannon. As the 1980 glass eel catch in the Burrishoole was close to (about 1.4%) the catch in the Shannon in that year (4500 kg), we will start exploring the effects of recruitment on the basis that the Burrishoole recruitment follows the trend of that for the catch in the Shannon. As the 1980 catch in the Shannon was 67% of the peak catch (from 1979), we will start the modelling by assuming that the maximum recruitment to the Burrishoole was 280,000.

There is no information available to determine the proportion of the recruitment represented by the catches in either of these time series. As such, they can only provide indications of the minimum glass eel recruitment to two rivers of very different size that flow to the sea along the west coast of Ireland.

The average length of recruits was set at 70 mm (range 50-100 mm), to represent glass eel recruits. Small yellow eels may also recruit from the tidal waters but there are no data on these so they have not been considered in the model application.

Sex differentiation

SMEP II requires a description of the length at which an undifferentiated eel is expected to become male or female, the probability of it becoming male or female under differing conditions of density, and the length above which all eel with no defined sex should be treated as females – this is only relevant where yellow eel survey data are used to describe the starting population. The user is required to define the length at differentiation and the length above which all undefined eels should be treated as females. By default, the shape of the curve describing how the proportion of females relates to density as a proportion of the local carrying capacity, and the density above which all undifferentiated eels become male, are provided in this file, although the user can alter these if local data are available.

In the absence of direct measures for the Burrishoole eels, the length at differentiation was set as 200 mm based on advice from RP (the smallest male recorded was 241 mm, and the smallest female 284 mm, but undifferentiated eels have been recorded at lengths up to 345 mm). The length above which all eels were to be assigned to females in the initialisation of the population was set as 460 mm because the longest male yellow or silver eel was 454 mm.

Natural Mortality

Lambert (2008) reported an estimated mortality for recruiting glass eel of 4.81 per year, or approximately 70% over three months. As with the CREPE modelling, we applied this rate for the first 3 months after recruitment. Thereafter, natural mortality rates were estimated using the relationship published by Bevacqua *et al.* (2011), which derives natural mortality based on water temperature, eel density and fish size. Mean water temperature for the river basin was supplied by Elvira de Eyto (EdE) and reference weights of 80, 110, 140, 170, 200, 240, 280, 320, 360, 400, 500, 600, 800 and 1000 mm eels were derived from the Length/weight relationship (see above) (Table 4.16). As the Bevacqua *et al.* (2011) model reports separate mortality rates for males and females, but SMEP II applies the same rate to both sexes, we have used the mean value for males and females combined, and the Low density range of rates, as recommended by data providers.

Table 4.16. Length-based rates of natural mortality for Burrishoole eels, derived from the Bevacqua et al. (2011) model.

Length class (mm)		Annual mortality rate	
	Low density	Average density	High density
0 – 80	0.31	0.58	0.87
80 – 110	0.20	0.38	0.56
110 – 140	0.15	0.27	0.41
140 – 170	0.11	0.21	0.31
170 – 200	0.09	0.17	0.25
200 – 240	0.07	0.13	0.20
240 – 280	0.06	0.11	0.16
280 – 320	0.05	0.09	0.13
320 – 360	0.04	0.08	0.11
360 – 400	0.04	0.07	0.10
400 – 500	0.03	0.05	0.07
500 – 600	0.02	0.04	0.06
600 – 800	0.01	0.03	0.04
800 – 1000	0.01	0.02	0.03

Silvering

SMEP II requires that the mean lengths at which male and female eels become silver eels are defined. The mean lengths at silvering for males (360 mm) and females (540 mm) were derived from the length distributions of silver eel catches, and as provided by EdE to the DemCam model.

Movement Rates

The annual movement rates of undifferentiated (50 km.yr^{-1}) and yellow (10 km.yr^{-1}) eels as they are dispersed upstream in the model simulations were derived from data provided by EdE on the maximum distances upstream where eels of 200 mm were found in electrofishing surveys. Note that the rates applied in SMEP II are higher than one might expect but this is because the rates are input as a distance travelled per year but SMEP II applies ¼ of this rate in the one season when movement is applied in the simulation. If movements were applied in more seasons, e.g. spring and summer, then SMEP II would apply ½ the rate, and the user would adjust the input rate accordingly.

Spatial description of the river basin

The river basin was described according to detailed river maps from the Irish GIS (Figure 4.60).

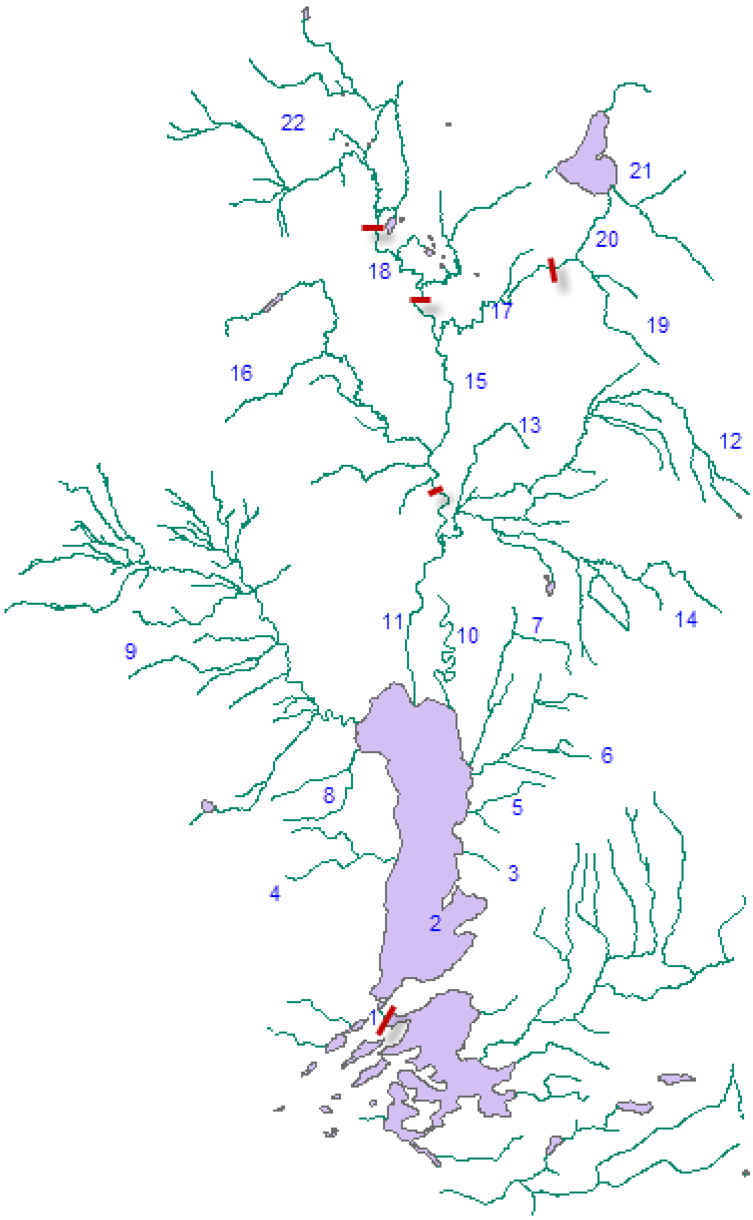


Figure 4.60. Schematic map of the Burrishoole river basin. The distribution of reaches used in the SMEP II simulations are labelled 1 to 22.

Only the freshwater part of the basin was defined in the model, shown in Figure 4.60, as the lowermost red bar.

In the absence of a complete understanding of the yellow eel densities throughout the river basin, there is no benefit in fully mapping the complexity of branching structure. Therefore, reaches were defined according to discrete tributaries and on the basis of the distribution of electrofishing sites. Twenty two reaches were defined as shown Figure 4.60.

Reaches are described in SMEP II according to river length and width, and the model uses these to estimate wetted area. The total length of river in each reach was extracted from the Irish rivers GIS. The GIS included channel widths for various sub-sections within each reach, so we calculated the mean of these widths to derive a representative width for the reach. The resultant wetted area was combined with the wetted area of any lakes in the reach (extracted from GIS) to estimate the total wetted area of each reach. This procedure resulted in a total wetted area of about 470.7 ha (21.4 ha of rivers and 449.3 ha of lakes) which is close to the actual estimate provided by RP (474 ha).

However, the procedure to combine together all the river lengths within each reach while conserving mean width is likely to result in a reach that is far longer than in reality, in effect stretching out the river basin. This stretching may cause issues in simulating the upstream movement of eel over time based on fixed rates of movement, especially in larger basins, since it will take much longer for modelled eels to arrive at the upper areas of any reach than would be expected in reality. Therefore, we applied a further procedure to conserve reach length and wetted area at the expense of reach width. Reach length was adjusted down to the maximum length measured on the map between the bottom and top of the reach, and 'mean' width adjusted to retain the same wetted area. In effect, we created a simplified map of the Burrishoole which spread over a distance similar to reality but with relatively wide streams and rivers.

One exception to this approach was reach 18, where the length was increased from 1836 to 4521 m (less than 6311 total length in reach) in order that reach 22 was further from the sea than reaches 20 and 21 on the eastern upper branch. This exception was only to avoid a possible conflict within the programming where a reach with a higher number label is closer to the sea than another reach.

Reaches were further characterised according to their connectivity to other reaches (Figure 4.61) and the distance from the bottom of each reach to the lowest point in the basin which was estimated based on the distances measured between the top and bottom of each reach.

Reach Definitions:

Length (km)	Width (km)	Reach label (description)
0.126	0.013	Reach 1 (starting at tidal limit)
6	0.658333333	Reach 2
0.549	0.0008	Reach 3
1.746	0.001503213	Reach 4
1.172	0.001129693	Reach 5
2.329	0.002473056	6
3.242	0.001363155	7
1.796	0.001417851	8
7.703	0.009937161	9
2.979	0.0008	10
3.039	0.008	11
7.381	0.00243806	12
2.024	0.0008	13
4.969	0.006162133	14
3.314	0.00716	15
5.216	0.004981141	16
2.501	0.001810736	17
4.521	0.003780739	18
2.73	0.001169484	19
1.126	0.00239	20
4.035	0.113258984	21
3.713	0.013500156	22

Reach Number	distance in km
1	0
2	0.126
3	2.126
4	2.626
5	3.126
6	3.626
7	4.126
8	4.626
9	5.126
10	5.626
11	6.126
12	8.566
13	8.7
14	9
15	9.165
16	9.5
17	12
18	12.479
19	14.501
20	15.179
21	16.305
22	17

Figure 4.61. Reach Definitions and Distance from Sea input files for SMEP II.

Lough Feeagh dominates the lower part of the river basin, and the wetted area overall. There are no data available on the density of eel in this Lough, as the only surveys conducted to date have been fyke net surveys which provide estimates of catch per unit effort (CPUE) but these cannot be converted to an area-specific measure.

Anthropogenic Impacts

Obstacles

There are no obstacles in the river basins that are thought to pose a significant barrier to the upstream movement of eels.

Fisheries and Turbines

There have been no commercial fisheries for eel, nor are there any turbines which cause mortalities, so none were included in the simulations. Data derived from Burrishoole scientific surveys for comparison with SMEP II outputs.

Indications of silver eel production

Counts and weight measures of the silver eel production from the freshwater compartment of the Burrishoole river basin were provided from 1971 onwards. The proportion of females amongst the silver eels was estimated by dissection and visual inspection of the gonads in 1985 to 1988, and by estimation from length distributions backed up by dissection of random sub-samples in latter years.

The numbers of silver eel have clearly declined during the time series, from around 4409 in the 1970s to around 2796 in the 2000s (Figure 4.62). At the same time, however, the biomass of silver eel has varied considerably from year to year, to the extent that there is no obvious trend (Figure 4.62).

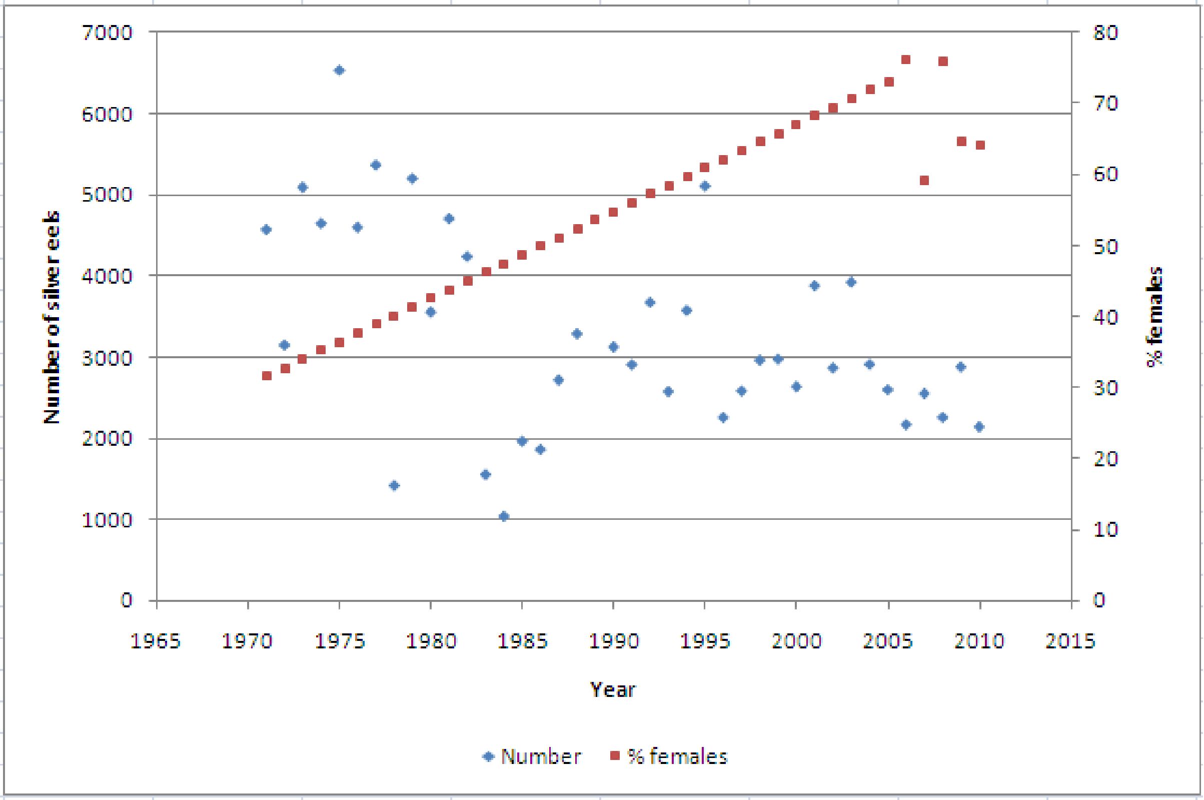


Figure 4.62. Numbers of silver eels caught escaping from the freshwater compartment of the Burrishoole river basin from 1971 onwards (blue diamonds), and the percentage of females (red squares) in the escapement during the same time period.

The reason for this difference in escapement patterns is clearly shown by the pattern of sex ratio across the time series. The sex ratio has changed from a predominance of males in the 1970s (30-40% females) to a predominance of females from around 1990 onwards, exceeding 70% by 2005 (Figures 4.62 and 4.63). Thus, there are fewer silver eels being produced now than 30 years ago, but overall, the eels that are being produced are on average heavier so the total biomass has changed little.

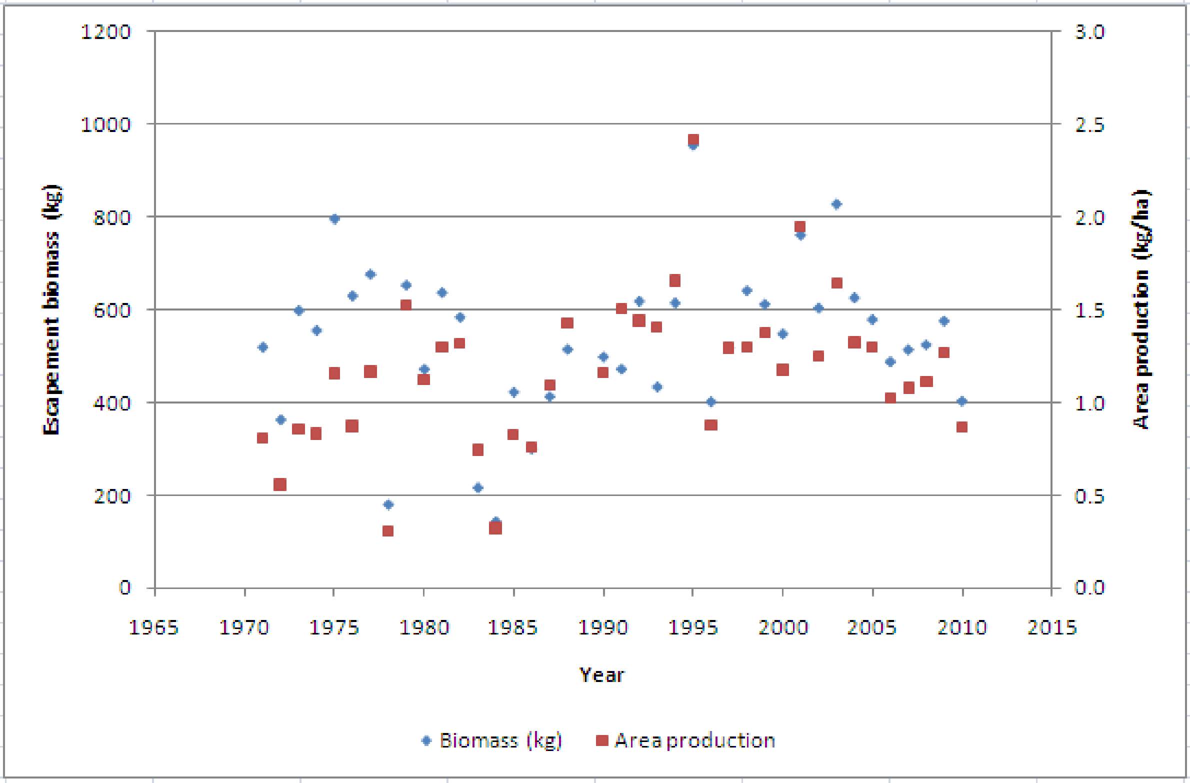


Figure 4.63. Biomass (kg) of silver eels caught escaping from the freshwater compartment of the Burrishoole river basin from 1971 onwards (blue diamonds), and the production expressed as kg per hectare (kg/ha) during the same time period (red squares).

The average weight of male silver eels has remained very similar throughout the time series, increasing only slightly from 82 g in the 1970s to 84 g in the 2000s. In contrast, the average weight of female silver eels has increased considerably, from 196 g in the 1970s to 267 g in the 2000s.

Indications of the yellow eel population

Where information on the yellow eel population in the river basin is available, this can be used to test how well the model represents some aspects of eel production. Data from a small number of electro-fishing surveys conducted in some reaches were available from 1991 onwards (not all years), and these provided estimates of local density of eels as numbers per 100 m^2. These data include catches of zero eels in some sites in some years. EdE confirmed that these zero catches are correct. Although we assume that eel have full and free access to the entire river basin, we have included these zero catches in the data used to estimate reach-specific densities. Where two or more electrofishing surveys were conducted in the same reach in the same year, the catches and areas fished were combined as if there had only been one survey but over the combined wetted area. An alternative approach would be to calculate average values for the two (or more) surveys combined.

Survey data were available from 17 reaches for some but not all years 1991 to 2009. No sites were surveyed in every year, so we have combined years to get the fullest possible distribution of yellow eel data. The earliest year with most sites surveyed is 1997. All of the sites surveyed in 1998 and 1999 had been surveyed several times in the earlier years. Therefore, we have selected the period 1991 to 1997 to construct the first dataset. The period from 2002 onwards shows a more consistent pattern of surveys compared to the 1990s. On the basis that the first period was 7 years, we have constructed a second dataset based on the surveys from 2003 to 2009.

Average values for densities were calculated by pooling the catches for surveys in all years and comparing against total wetted area surveyed. Table 4.17 presents the mean densities for various reaches, for the two study periods.

Table 4.17. Densities of yellow eels in SMEP II reaches of the Burrishoole river basin, derived from electro-fishing surveys conducted during the periods 1991 to 1997, and 2003 to 2009

Period	1991-1997	2003-2009
Reach Number	Density (eels. 100m^2)	Density (eels. 100m^2)
3	2.65	0.27
5	1.37	0.00
6	3.00	1.94
9	1.37	1.14
11		1.18
12	1.21	0.26
13	1.20	0.21
14	0.48	1.25
15	0.45	0.30
16	1.69	1.51
17	1.20	0.54
18	0.97	1.25
19	1.90	
20	0.43	0.32
21	0.71	0.78
22	1.74	0.65

Results of the application of SMEP II

The aim of these SMEP applications was to identify the best parameter set that produces a SMEP II simulation of the Burrishoole eel stock and silver eel escapement pattern as close as possible to that observed by scientific monitoring programmes.

Silver eel escapement under historic high levels of recruitment (B_0)

This is the biomass of silver eel that should be expected under pristine conditions. The European Eel Regulation sets a limit target for present day production of 40% of B_0. However, as the biomass of silver eel escaping from the freshwater compartment of the river basin has been monitored since the 1970s, before the European-wide crash in recruitment commenced, we treat the silver eel biomasses from the 1970s to 1982 as being representative of historic escapement potential in the absence of anthropogenic impacts.

The numbers of silver eel was around 5000 in the 1970s to around 2500 in the 2000s. The biomass of silver eel has varied considerably from year to year, but the average is around 500 to 600 kg per annum throughout. The sex ratio has changed from 30-40% females in 1970s to 70% by 2005.

Yellow eel densities derived from the electro-fishing surveys ranged from about 3 to 0.5 eels per 100m^2 in the 1990s, and reduced to 2 to 0.3 eels per 100m^2 in the 2000s.

Recruitment was set at 189,000 per year. Mortality rates were initially set according to the HIGH DENSITY values from the Bevacqua *et al.* (2011) model, on the assumption that recruitment was relatively high and therefore density dependent mortality would have been relatively high in the 1970s.

Table 4.18 presents the silver eel outputs for the last 10 years of the simulation, under stable recruitment – note that this is not a prediction based on 10 actual years. Silver eel production is characterised in terms of the total weight (kg) and numbers of eels, the numbers of males and females and the % females, and the average weight of male and female silver eels. SMEP II predicted an output of about 2200 silver eels, with a total weight of about 420 kg. About 42% of these silver eels were females, and the average weights of female and male silver eels were 318 and 104 g, respectively.

Table 4.18. SMEP II predictions of potential silver eel escapement from the Burrishoole river basin, as numbers, weight and sex ratio, for conditions of stable high recruitment

Model Year	Silver eel		Females		Males		% females
	kg	No.	kg	No.	kg	No.	
10	430	2199	297	936	131	1263	0.43
9	429	2198	296	932	131	1266	0.42
8	427	2194	295	926	132	1268	0.42
7	425	2189	293	921	132	1268	0.42
6	424	2183	291	915	132	1267	0.42
5	422	2175	289	910	131	1266	0.42
4	420	2167	288	904	131	1263	0.42
3	418	2160	286	900	131	1260	0.42
2	416	2152	285	896	130	1255	0.42
1	415	2145	284	894	130	1251	0.42

Table 4.19 presents the predicted densities (per $100m^2$) of differentiated yellow eels across the reaches, averaged across the last 10 years of the simulation. Note that we have excluded the undifferentiated eels, those < 200 mm total length, on the assumption that these would be under-represented in the electro-fishing catches. Densities in the lower reaches are predicted to be around 7 eels per $100m^2$ in the lower reaches around Lough Feeagh and the main river, around 5 eels in the middle river reaches, and < 2 eels in the highest reaches.

Table 4.19. SMEP II predictions of yellow eel densities in the 22 reaches of the Burrishoole river basin, for conditions of stable, high recruitment

Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)
1	5.36	9	7.02	17	3.00
2	4.32	10	7.39	18	1.94
3	7.41	11	4.15	19	1.23
4	7.39	12	5.39	20	0.22
5	7.40	13	5.91	21	0.03
6	7.38	14	5.97	22	2.80
7	7.38	15	3.87		
8	7.39	16	5.63		

This first set of parameters produces silver eels of about the correct size and the sex ratio is close to that required, but the quantity of silver eels produced is low while the densities of yellow eels in the reaches are higher than found in the 1990s surveys.

There is the possibility that Feeagh has a relatively high production of silver eels compared to the river reaches, but it is difficult to parameterize SMEP II in such a way to concentrate production in Feeagh without resulting in a silver eel output almost entirely of males – unless the 'rules' for sex differentiation are different in lakes than they are in river environments. Therefore, we have here assumed that Feeagh has the same productive capacity as river reaches, relative to wetted areas.

The yellow eel densities predicted by SMEP II are higher than those found in the surveys, but the surveys were conducted 10 -20 years after the start of the crash in recruitment, so we might expect the densities in the 1970s to have been higher. However, the predicted numbers and weight of silver eels was low compared to those annual trap catches. Therefore, the parameterization needed to be adjusted to increase the relative silver eel production from the same amount of yellow eels.

Reducing the mortality rates across the whole length structure would have resulted in increased densities of undifferentiated eels, and hence a much higher percentage of males in the silver eel outputs. Therefore, we chose to reduce the mortality rates only for those eels longer than 200 mm, as this ought to preserve the sex ratio at the same time as increasing the production of silver eels from a unit amount of yellow eels. We applied the mortality rates associated with AVERAGE DENSITY from the Bevacqua et al. (2010) model for eels longer than 200 mm.

Table 4.20 presents the silver eel outputs for the last 10 years of the simulation, under a stable recruitment but with the new mortality function. SMEP II predicted an output of about 4500 silver eels, with a total weight of about 700 kg. About 26% of these silver eels were females, and the average weights of female and male silver eels were 323 and 105 g, respectively.

Table 4.20. SMEP II predictions of potential silver eel escapement from the Burrishoole river basin, as numbers, weight and sex ratio, for conditions of stable high recruitment but with a reduced mortality for eels longer than 200 mm.

Model Year	Silver eel		Females		Males		% females
	kg	No.	kg	No.	kg	No.	
10	777	4235	501	1577	275	2658	0.37
9	802	4418	510	1603	292	2814	0.36
8	820	4562	512	1608	306	2954	0.35
7	828	4661	508	1591	319	3070	0.34
6	826	4711	496	1552	329	3159	0.33
5	815	4713	478	1493	336	3220	0.32
4	795	4669	454	1416	340	3253	0.30
3	767	4582	425	1323	341	3259	0.29
2	732	4457	393	1219	339	3238	0.27
1	694	4302	359	1113	334	3190	0.26

Table 4.21 presents the predicted densities (per 100m^2) of differentiated yellow eels across the reaches, which were similar to those predicted in the first scenario, at around 7 eels per 100m^2 in the lower reaches around Lough Feeagh and the main river, around 5 eels in the middle river reaches, and < 2 eels in the highest reaches.

Table 4.21. SMEP II predictions of yellow eel densities in the 22 reaches of the Burrishoole river basin, for conditions of stable, high recruitment but with a reduced mortality for eels longer than 200 mm

Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)
1	8.07	9		17	
2	4.99	10		18	
3	7.92	11		19	
4	7.91	12		20	
5	7.92	13		21	
6	7.90	14		22	
7	7.90	15			
8	7.91	16			

As the silver eel production was similar in numbers, weight and sex ratio to that observed in the 1970s trap catches, we used this set of parameters as the starting point to predict forwards and model the effects of changes in recruitment to achieve the observed change in the characteristics of the silver eel trap catches over the 1990s and 2000s.

Silver eel escapement biomass under present conditions ($B_{present}$)

This is the silver eel biomass that escapes to the sea under present conditions. As with B_o, however, $B_{present}$ is already known for the freshwater compartment of the Burrishoole river basin, and the purpose of applying SMEP II is to explore the parameters describing life history processes, and particularly changes in recruitment, that produce SMEP II results similar to those found by scientific surveys.

We applied the recruitment index from the Shannon glass eel catch surveys from 1977 to 2010, with annual catch reported as a proportion of the maximum catch (6700 kg in1979). In the absence of known yellow eel densities for all the reaches at the beginning of the time series, we used the yellow eel predictions from 189,000 recruit simulation to construct the necessary input files describing the yellow eels in all reaches. Also, we started with the same mortality rates as applied above.

We project forwards from 1977 to 2010 (34 years).

Test Data

The silver eel trap catches – the observed data – gave numbers of silver eel of around 2500 in the 2000s, of which about 70% were females. The biomass of silver eel has varied considerably from year to year, but the average is around 500 to 600 kg per annum throughout. Yellow eel densities derived from the electro-fishing surveys ranged from about 2 to 0.3 eels per $100m^2$ in the 2000s.

SMEP II results

As the recruitment time series is dynamic, and predicted silver eel outputs in any year are the result of recruitment and yellow eel production over a number of previous years, simple comparison of means is not appropriate to examine the results of these predictions. Therefore, we consider the results over the last 10 years (~2001 to 2010). As above, Tables 4.22 and 4.23 present the complete results but here we report the range of values during this period of the predictions.

SMEP II predicted that the silver eel production would decline from about 360 kg to 39 kg, and about 1100 to 130 eels over the last 10 years of the simulation. The sex ratio was strongly biased towards females throughout, ranging from 78 to 95%. Yellow eel densities were very low throughout the reaches, ranging from 0.6 eels per 100 m^2 to almost zero.

Table 4.22. SMEP II predictions of silver eel escapement from the Burrishoole river basin, as numbers, weight and sex ratio, for 2001 to 2010, based on a recruitment trend derived from the Shannon glass eel catch time series and assuming natural mortality rates based on those predicted by the Bevacqua et al. (2010) model for average densities

Model Year	Silver eel		Females		Males		% females
	kg	No.	kg	No.	kg	No.	
2001	360	1153	353	1101	6	52	0.95
2002	280	881	275	842	4	39	0.96
2003	219	678	216	653	2	25	0.96
2004	141	420	139	410	1	10	0.98
2005	88	266	87	260	0.7	6	0.98
2006	87	278	85	267	1	10	0.96
2007	100	328	98	313	1	15	0.96
2008	107	350	104	326	2	25	0.93
2009	72	242	68	200	4	42	0.83
2010	694	4302	35	102	3	30	0.26

Table 4.23. SMEP II predictions of yellow eel densities in the 22 reaches of the Burrishoole river basin, for 2001 to 2010, based on a recruitment trend derived from the Shannon glass eel catch time series and assuming natural mortality rates based on those predicted by the Bevacqua et al. (2010) model for average densities.

Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)
1	0.38	9	0.46	17	0.07
2	0.23	10	0.61	18	0.07
3	0.61	11	0.08	19	0.08
4	0.60	12	0.16	20	0.05
5	0.61	13	0.55	21	0.01
6	0.60	14	0.44	22	0.24
7	0.60	15	0.08		
8	0.61	16	0.46		

Taking all these metrics together, and assuming that the Shannon glass eel catch is an appropriate index for recruitment to the Burrishoole, it appears that our first parameterization over-estimated the mortality of yellow eels. We had applied the HIGH to AVERAGE values, but as the recruitment declined very rapidly in the early part of the time series, it is acceptable to apply the LOW rates.

We applied the mortality rates associated with LOW DENSITY from the Bevacqua *et al.* (2011) model for eels of 200 mm and longer – see Tables 4.24 and 4.25. SMEP II predicted that the silver eel production would decline from about 400 kg to 44 kg, and about 1400 to 160 eels over the last 10 years of the simulation. The sex ratio was strongly biased towards females throughout, ranging from 70 to 90%. Yellow eel densities were very low throughout the reaches, ranging from 0.9 eels per 100 m^2 to almost zero.

Table 4.24. SMEP II predictions of silver eel escapement from the Burrishoole river basin, as numbers, weight and sex ratio, for 2001 to 2010, based on a recruitment trend derived from the Shannon glass eel catch time series and assuming natural mortality rates based on those predicted by the Bevacqua et al. (2010) model for low densities

Model Year	Silver eel		Females		Males		% females
	kg	No.	kg	No.	kg	No.	
2001	418	1402	394	1214	23	188	0.87
2002	304	1004	288	867	15	137	0.86
2003	220	711	209	625	10	87	0.88
2004	136	415	131	379	4	35	0.92
2005	82	252	80	235	2	17	0.93
2006	80	260	77	241	2	20	0.92
2007	94	314	91	289	2	25	0.92
2008	103	349	99	310	3	39	0.89
2009	74	264	67	199	6	65	0.75
2010	44	161	38	114	5	47	0.71

Table 4.25. SMEP II predictions of yellow eel densities in the 22 reaches of the Burrishoole river basin, for 2001 to 2010, based on a recruitment trend derived from the Shannon glass eel catch time series and assuming natural mortality rates based on those predicted by the Bevacqua et al. (2010) model for low densities

Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)	Reach Number	Density (eels.100m^2)
1	0.70	9	0.68	17	0.11
2	0.35	10	0.88	18	0.10
3	0.89	11	0.11	19	0.12
4	0.88	12	0.23	20	0.06
5	0.89	13	0.79	21	0.01
6	0.87	14	0.60	22	0.33
7	0.87	15	0.12		
8	0.88	16	0.68		

Conclusion

The predicted silver eel outputs are low compared to those observed recently, although the sex ratio is close to observed. The yellow eel densities are also lower than observed. If we assume that SMEP II models the life history of the Burrishoole eel with reasonable confidence, our results suggest that the recruitment to the Burrishoole was greater than the Shannon Index parameterisation. Further tests will be conducted using other recruitment time series to explore whether these provide a closer fit of SMEP II predictions to the observed silver eel productions. However, additional testing is also required to further explore the parameters used to describe the life history processes, in particular the growth rates of yellow eels which may lower than modelled above, and the lake/river interaction that we do not understand yet.

4.6. Comparison of SMEP, EDA and DemCam predictions for the Burrishoole

Note that results presented above, and the conclusions reported here are provisional because work is continuing during the final months of the project to optimize the parameterisations of the model applications. Therefore, the following may be subject to change before the end of the project.

Considering the most recent period of the actual time series of silver eel escapement from the Burrishoole, under presumed conditions of relatively low recruitment, SMEP II, EDA and DemCam give contrasting results in terms of the escapement of individual eels per unit of wetted area (Figure 4.64). DemCam calculates too high values of escapement and exaggerates the decreasing trend. SMEP II is pessimistic in the absolute values of silver eel escapement but correctly predicts the trend. The single value for EDA calculated for Western RBD (and extrapolated to the Burrishoole) overestimates the actual figure observed in Burrishoole by a factor of 10 to 15.

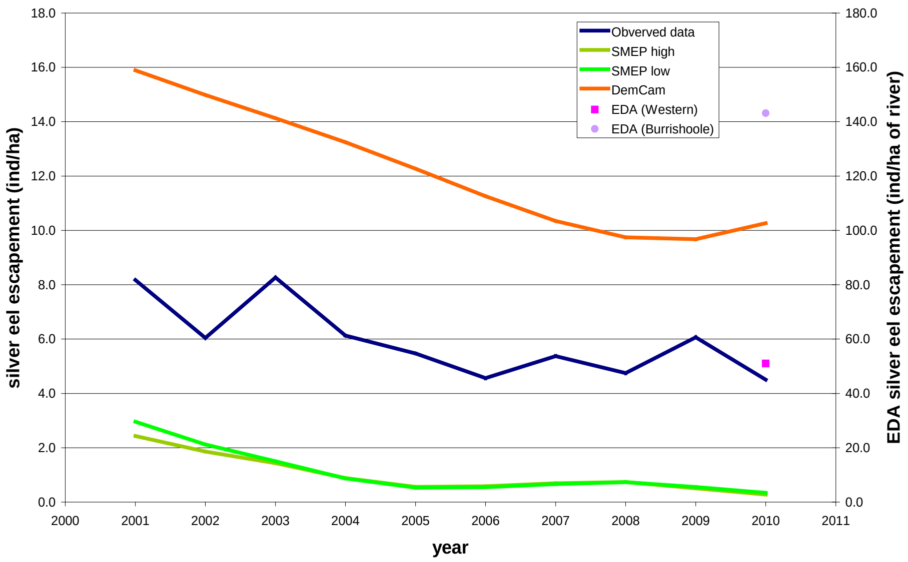


Figure 4.64. Evolution of silver eel escapement observed in Burrishoole basin and calculated by SMEP II, EDA and DemCam

4.7. Conclusions about the assessments of the models for the Burrishoole

SMEP II, EDA and DemCam gave contrasting results of silver eel escapement per unit of wetted area for the Burrishoole data set. DemCam overestimated escapement and exaggerated the decreasing trend. The single value for EDA calculated for Western RBD (and extrapolated to the Burrishoole) overestimated the actual figure observed in Burrishoole by a factor of 10 to 15. SMEP II was pessimistic in the absolute values of silver eel escapement but correctly predicted the trend.

These comparisons between the model predictions and the 'known' outputs of the Burrishoole silver eel production, revealed that the different modelling approaches did not converge to a single conclusion in terms of their accuracy in predicting silver eel production.

Chapter 5. Model applications to other data sets

In this chapter we describe the application of models to a range of eel data sets from the North Sea-Baltic, Atlantic and Mediterranean regions, in order to provide an illustrative guide to local stock assessment procedures suited to the various habitats from which silver eel can escape, and how the models could be modified to suit local conditions. These applications were not intended to be part of the model testing procedures (Chapter 4), but to provide more illustrations of the manner in which models can be applied to a variety of management scenarios.

The chapter concludes with a guide to managers on the selection of models suitable to a variety of assessment scenarios.

5.1. Western River Basin District, Ireland

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Overview

The Western River Basin District (WRBD) covers the area from Galway in the south to Sligo in the north on the Atlantic West coast of Ireland (Figure 5.1). There are 89 catchments in the RBD with over 14,200 km of river, of which at least 64 are probably relevant for eel, with the Corrib, Moy, Ballysadare (Arrow) and Garavogue (Gill) catchments being the largest and most important for eel. The basin area is rich in lakes with a total of 5,638 lakes: 69 (1%) are greater than 50 hectares whereas about 4285 (76%) are less than 1 hectare in size.

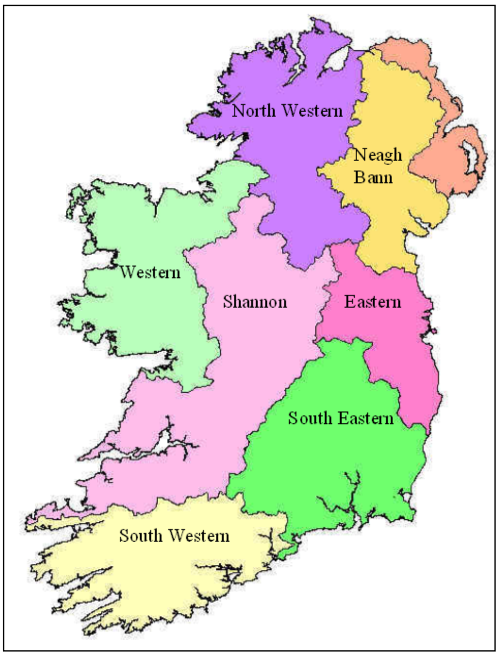


Figure 5.1. Map of Ireland showing the 8 Water Framework Directive River Basin Districts.

There are two main geology types, carboniferous (calcareous) and siliceous (non-calcareous), which largely determine the productivity of the catchments and the growth and biological characteristics of the eel stocks (Figure 5.2). The WRBD is dominated in the eastern part, east of the great western lakes, principally by well drained karst limestone overlain by grassland generally used for agricultural purposes. The western part of the basin is dominated by siliceous non-calcareous acidic geology with wet peaty soils. A full GIS database exists of the catchment characteristics, including a rivers and lakes database, risks and pressures and this is also currently being updated to include fish passage barriers (McGinnity *et al*., 2003). Main land use in the western part is afforestation and subsistence agriculture (grazing). Agriculture comprises 47% with natural areas (forestry, peatland, lakes etc.) comprising 53%. The basin is relatively sparsely populated with urban infrastructure comprising about 0.03% of the basin area.

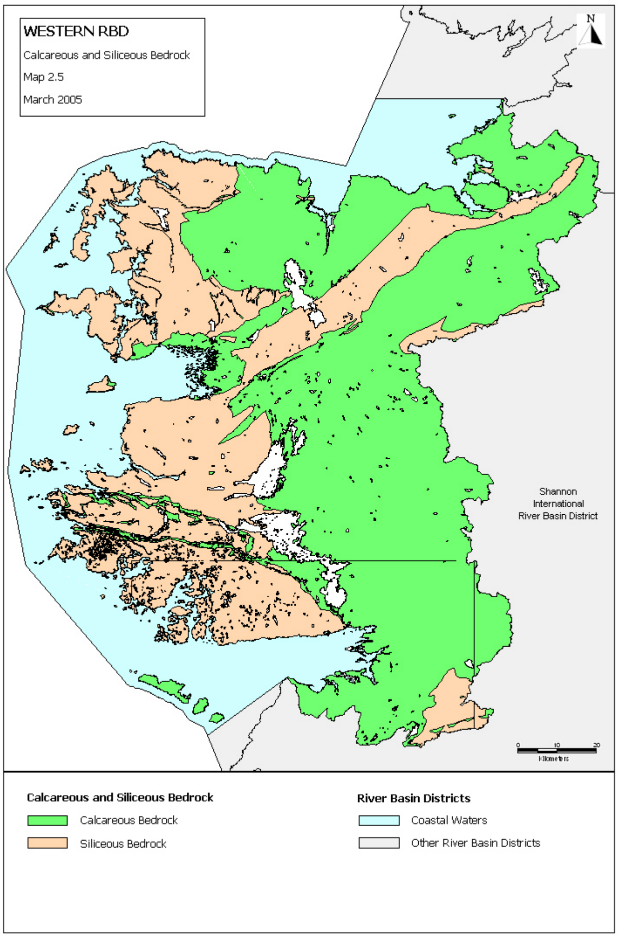


Figure 5.2. Map showing distribution of calcareous and siliceous (non-calcareous) rock types in the Western RBD.

This quantification of the wetted area habitats using the GIS database and the fisheries database should facilitate the modelling of the data required in the POSE project and will also be used for the extrapolations from data rich to data poor catchments.

Stock status

No data-based estimates of truly pristine escapement exist for Irish eel catchments. The potential production of silver eels (in biomass terms) for the WRBD prior to the decline in European-scale recruitment following 1982, and the biomass of silver eels currently escaping from the RBD have been estimated using a habitat-based extrapolation of productivity information from index catchments not necessarily within the RBD. Historical production of silver eels from Irish RBDs was calculated (for freshwaters only) using catch series for four catchments (where the fishery efficiency was estimated) for periods prior to 1980. These data were calibrated using eel growth rates for 17 catchments and a regression model was developed relating production to catchment geology, a proxy for productivity. RBD specific impacts were then imposed on this potential productivity to derive an approximate estimate of current escapement.

Pristine escapement for the WRBD was estimated at 173 tonnes, whereas current escapement is estimated to be approximately 51 tonnes i.e. 30% of pristine.

There is a general paucity of information on glass eel and elver migration in the WRBD. All available information indicates that runs decreased dramatically following 1982. In Galway, eel runs were too low to warrant capture and transfer following 1987. In Burrishoole, the elver run declined by 70-95% from 1980-1987/8.

Yellow eel stocks may reflect the reduction in recruitment. Commercial catch per unit effort (CPUE) has declined considerably at the coastal end of some catchments where recruitment of young eels is normally concentrated. CPUE has remained roughly static for upstream fisheries that rely on immigration of more mature eels. However, the quality of the information makes it difficult to determine trends in CPUE within the WRBD.

There has been a reduction in the silver eel catch in the Galway Fishery on the Corrib River since 1982. This may be an indication of a reducing stock. This and all other eel fisheries have been closed since 2009.

In the unexploited Burrishoole system, the sex ratio of silver eels increased to 60% females by 1990 and female eel size has increased concurrently. The mean age recorded for migrating female silver eels in Burrishoole is approximately 30 years and it is still to be observed as to whether the silver eel output from catchments in kg/ha will be maintained in the next few years.

There is insufficient scientific information available to comment on stocks within transitional waters in the region where fisheries occur.

Impacts to eel production

Commercial and recreational fisheries

There is no glass eel fishery in the WRBD.

Commercial fishing for eel was closed in 2010. Prior to this, the commercial adult eel fishery was highly focused, with the majority occurring on the seven largest waterbodies but also including relatively small scale, scattered and sporadic fisheries on the smaller lakes. In some cases, lakes were only fished for one, or a few, nights. Almost no river fishing took place for yellow (brown) eel. The main gear types for yellow eel were standard fyke nets set in chains of 5 or 10 nets and baited longlines. The capacity and the reported effort of the yellow (brown) eel fishery increased over 2001-2007.

In spite of increasing capacity and effort the yellow (brown) eel catch decreased slightly or at best remained static. Certainly, the proportion of the national yellow (brown) eel catch attributable to the WRBD decreased somewhat over the period 2001-2007. Nevertheless, the WRBD yellow (brown) eel fishery was important in a national context, accounting for 17.8-23.4 tonnes or 23-42% of the contemporary national catch over this period.

Silver eels were fished using fyke nets or stocking-shaped nets called "coghill nets" which are attached to fixed structures in the river flow, often at "eel weirs". The main fisheries for silver eel were historically on the Moy and on the Corrib. In recent years there has been virtually no fishing with coghill nets for silver eel on any fisheries. On the Moy, silver eel were netted at three locations until 2004. Some silver eel are caught in fyke nets fished on Loughs Conn and Cullin and there has been limited exploitation of silver eel in the Garravogue River/Lough Gill and Ballysadare River/Lough Arrow catchments. Overall, the silver eel fisheries capacity and effort remained largely constant from 2001-2007. Catches were returned for a high proportion of the licences issued and most returns were for active licences.

In contrast with the yellow (brown) eel fishery, the silver eel capacity and effort decreased slightly from 2001-2007 and yet its reported yield increased from 9.6-15.6 tonnes over this period. The yield of the WRBD silver eel fishery was highly and increasingly significant in an Irish context accounting for 28% of the 2001 catch and 45% of the 2007 catch.

Until 2004, the commercial fishery reporting system in Ireland was somewhat *ad hoc*, with locally applied data collection. A standard catch reporting system was introduced in 2005 with a standard catch reporting form which included month fished, number of days fished, catch, dealer, mortalities, and undersized catch released. The fishery was reduced in season length in 2009 and has been closed in 2010 as one of the management actions in the Irish Eel Management Plan.

Recreational eel fishing was only carried out by a minority of anglers and there is no legal, or voluntary, declaration of catch which is probably small. Angling for eel is now on a catch and release basis. Recreational net, pot and line eel catches are not differentiated as such and were included in the commercial catch returns. Some "recreational" fishing using fyke nets and baited pots takes place within the region controlled by the WRFB.

Obstacles to migration

Water levels are regulated by means of a weir on the River Corrib at Galway. There are several canals and disused millraces which link the Corrib to the estuary and circumvent the regulating weir. The flows through these are significant during-periods of low discharge over the weir. There is evidence that upstream migration in the Corrib system is difficult and mortality is high (Moriarty, 2001).

In the Ballina district there are no significant natural or man-made barriers to the upstream migration of elvers or escapement of silver eel downstream of Loughs Conn and Cullin. A man-made weir on the Owengarve River (Curry River), an important tributary of the upper Moy, includes a fish pass which, in low water conditions, facilitates the upstream migration of elver.

In the Sligo District, two sets of natural falls on the Ballysadare River present a significant obstruction to the upstream migration of elver, but both have fish passes.

There are two hydro electric installations on the Ballysadare River, each located to one side of the river with a race leading to the turbines. There are no data to suggest that either is responsible for silver eel mortalities. There are no hydro electric installations in either the Ballina or Bangor Districts.

Water quality

In the WRBD the main surface water pressures derive from water abstraction, flow regulation, morphological alterations (drainage and river defence works), and pollution from point sources (from industrial and urban wastewater mainly) and diffuse sources (urbanisation, agriculture, forestry and peat harvesting). The main morphological pressures arise from channelization and dredging impacting bed slope, side slope and flow changes. Impoundments by dams also give rise to some impacts on lakes. Land use such as peat extraction, coniferous forestry and urban impact are also important pressures. Morphological impact for transitional and coastal waters is low. Point source pressures identified for surface waters include waste water treatment plants, storm overflows, integrated pollution control licences and waste treatment plants. The overall vulnerability of surface waters is significant or probably significant in most of the eastern half of the WRBD. The main challenge for water quality is eutrophication, which has been increasing persistently since the 1970s and is probably the most serious environmental pollution problem in Ireland. Poor water quality impacts on the potential of rivers to produce salmon. It is unknown whether similar poor water quality levels have an effect on eel.

Pollution

Persistent organic pollutants levels are relatively low in silver eels caught at the weir in Galway City, and in eels from Corrib and Conn, as are PCBs in eels from Lake Furnace and River Owengarve (Santillo *et al*., 2005) and L. Conn, Corrib and Burrishoole (McHugh *et al*., 2010) but dioxin levels are higher in the Burrishoole eels.

Diseases and parasites

Nationwide, preliminary analysis of information available on the presence of *Anguillicoloides crassus* in different catchments would indicate that approximately 50% of the Irish wetted area is now potentially infected by the parasite and that it continues to spread. Tests of infection of silver eels caught at the Galway weir in Galway City, and for eels from Corrib, Conn and Burrishoole, indicate that *A. crassus* has spread into the Corrib and Moy catchments but not the Burrishoole.

Stocking

Historically, moderate amounts of elvers were translocated through the Corrib catchment until the late 1980s when runs of elvers became prohibitively poor. In the 1990s, stock enhancement by transferring glass from the neighbouring River Erriff was carried out in the catchment. Recent official stocking in the WRBD is low and only occurs in the Moy catchment with 13.5 kg being translocated in 2005. There has been some unofficial stocking of the Corrib system but data are unavailable for these actions.

Data for POSE

In addition to the comprehensive GIS-based habitat inventory for the WRBD, catch data (yellow and silver eel) and some length, weight and age data from the commercial fisheries are available. The historical eel survey data include fyke net survey catches, CPUEs and length and age details. Some electro fishing survey data from rivers are also available. There are no recruitment data and silver eel have only been quantified from one catchment (Burrishoole). Silver eel fisheries have been sampled on the Corrib and Moy systems and comparative historical data exist. Comprehensive data are available for the Burrishoole catchment and recent survey data (fisheries – 2008) and fyke net surveys (2009) are available for Loughs Conn, Cullen, Corrib and Burrishoole.

In addition to the general data available for the entire RBD which are have been used for the application of the EDA (reported below), specific case study data are available for the data-rich Burrishoole (see Chapter 4) and the data-poor Corrib, which have been used for the application of DEMCAM and GEM (see below). The Corrib catchment was selected as it was the most important catchment for the eel fishery in the WRBD. Its character is intermediate between the calcareous western half and the siliceous eastern half of the WRBD.

5.2. Application of EDA to the Western River Basin District

Input data

The model requires data about abundance of yellow eel. These data were extracted from the electro-fishing surveys, yielding data from 88 sites sampled during 1991 to 2009 and resulting in 664 discrete data points (Figures 5.3, 5.4 and 5.5). The density (d) was calculated for all electrofishing operations and expressed in number/100m².

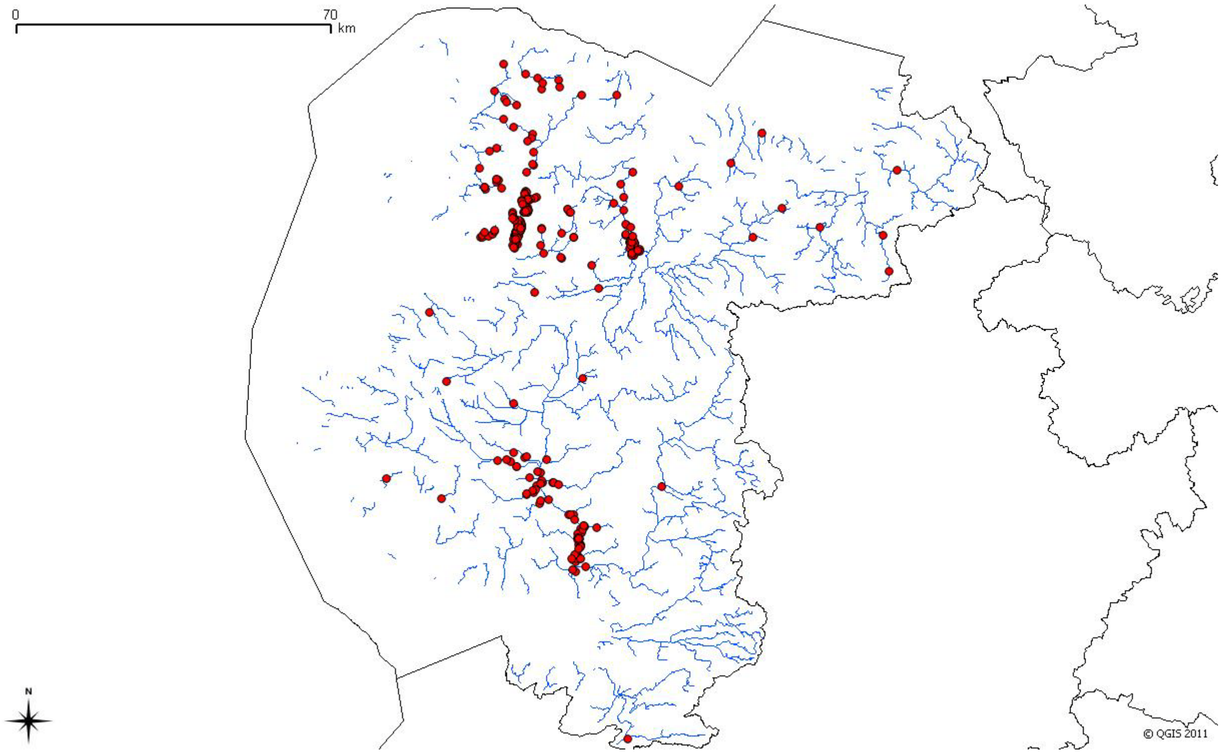


Figure 5.3. Sampling sites (in red) in the Western Emu catchment on the CCM river network (in blue).

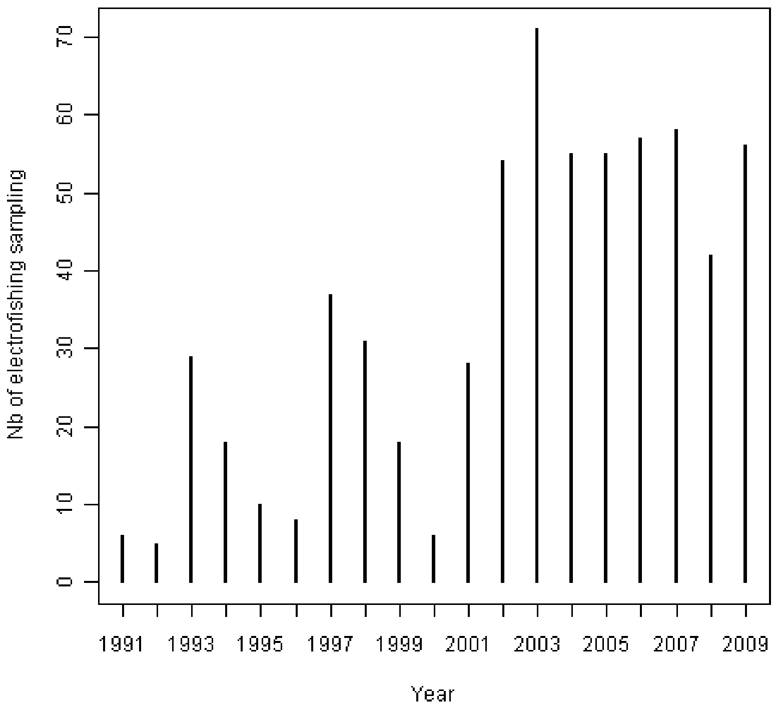


Figure 5.4. Number of electrofishing sampling per year in Western Emu.

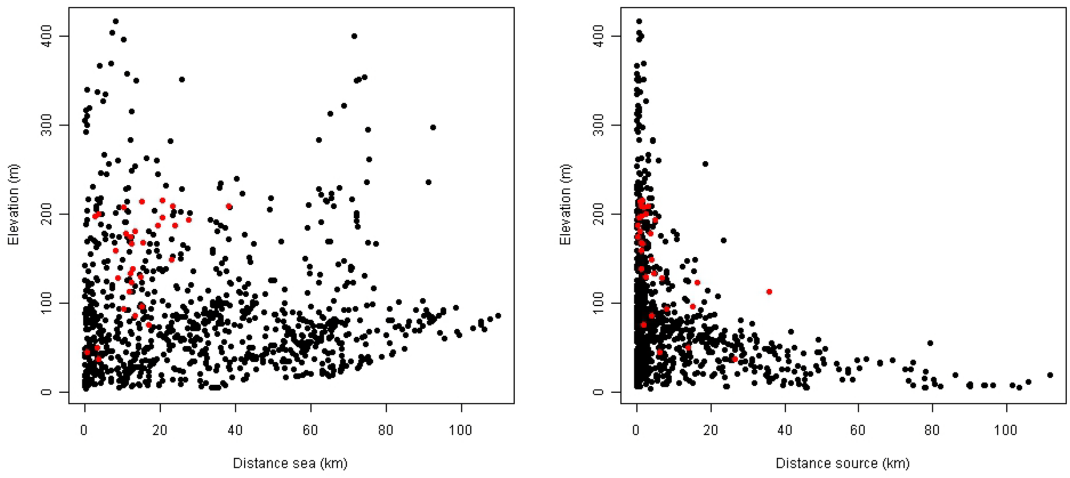


Figure 5.5. Sampling sites (in red) in the Western Emu catchment on the CCM river network (in black).

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CCM data set for the WRBD provided information for each survey site on the distances to sea and source, relative distance, mean slope and elevation, altitudinal gradient, the area of land drainage upstream, mean annual temperature and rainfall, and land use cover (urban, agricultural and no impact). No data on the locations or characteristics of dams were available and therefore this variable could not be tested.

A combination of 14 variables was tested (Figure 5.6). A separate entry was used for the 4 groups of variables described in Figure 5.7 to avoid spurious correlations.

According to the threshold of 0.5 for ρ^2, the following variables should be grouped:

- relative_distance and distance_source
- elevation_mean, temperature_mean and rain_mean
- p_up_agricultural, p_no_impact and p_agricultural

The combination of the different groups results in 480 models to be tested.

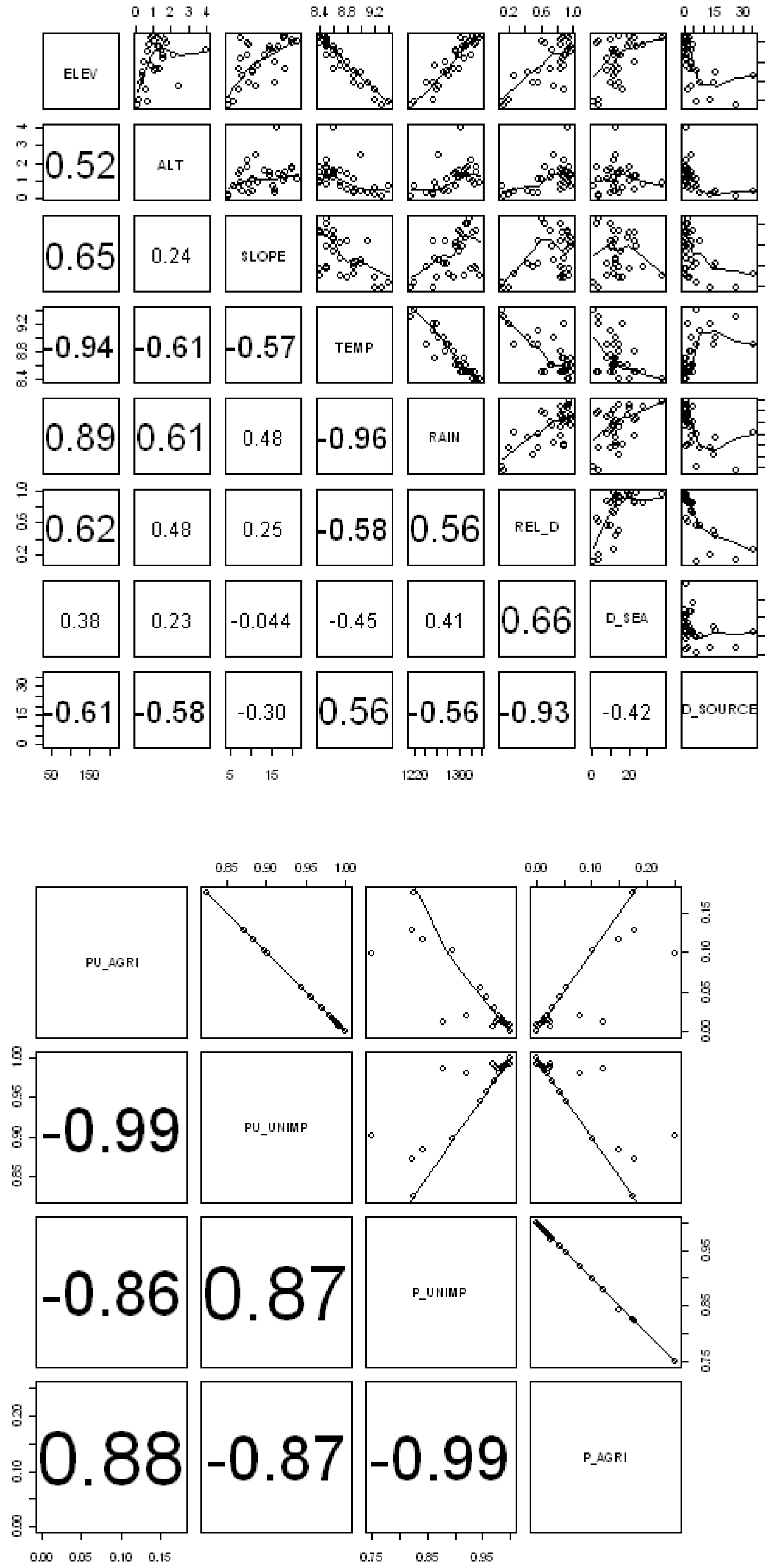


Figure 5.6. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With ELEV: elevation mean, ALT: altitudinal gradient, SLOPE: slope mean, TEMP: temperature mean, RAIN: rain mean, REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, PU_AGRI : upstream percent in agricultural use, PU_UNIMP: upstream percent in unimpact land use (p_up_no_impact), P_UNIMP: local percent in

unimpact land use (p_no_impact), P_AGRI: local percent in agricultural use. The font size of the cross-correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

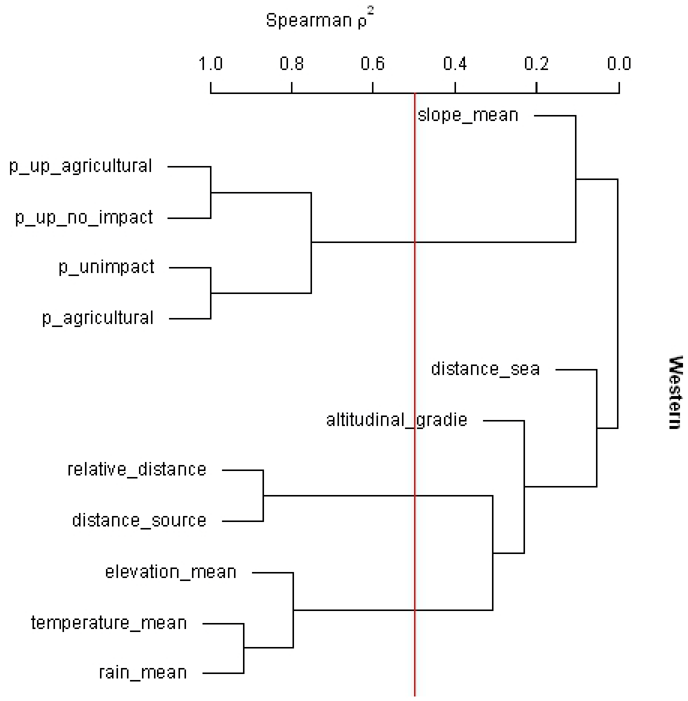


Figure 5.7. Dendrogram obtained by hierarchical cluster analysis of 12 candidate predictors for Western dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ²=0.5.

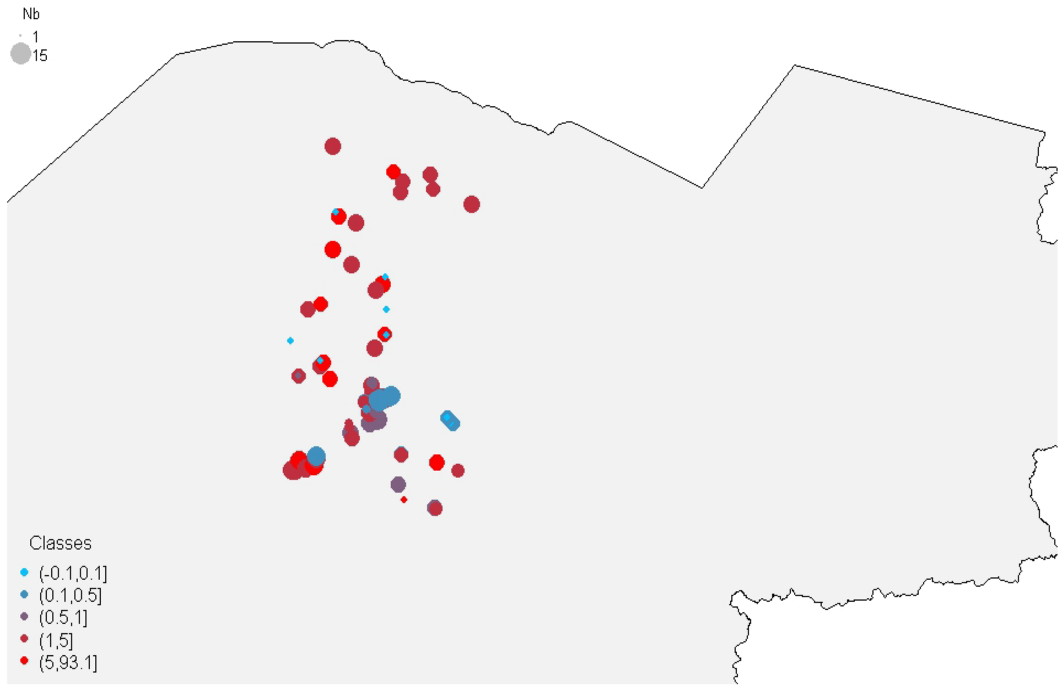


Figure 5.8. Map of density observed densities. Densities in eel m^{-2}. All years.

Model Testing Procedure

We calibrated statistical models with a Generalized Additive Model (Hastie & Tibshirani, 1990) to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/). EDA model is based on a delta-gamma model (Stefánsson, 1996) which combines two generalized additive model (GAM). There are three steps of modelling:

1. a presence/absence model (delta model) based on a GAM with a binomial distribution and a logit link to determine the probability of a positive catch;
2. a density model with the positive data (gamma model) using a GAM with a gamma distribution and logarithm link to determine the level of positive catch; and,
3. the multiplication of the two previous models (delta-gamma model).

The GAMs were computed with the library 'gam' (Hastie, 2010) with a cubic spline smoother (3 degree of freedom) for each environmental variable. The delta-gamma (ΔΓ) generalized additive models explain a large portion of the variability in eel abundance data, as there are many occasions where densities are null.

Presence-absence model (Delta model)

The results for tests of 480 models are summarised in Table 5.1. The best density model is selected by the Akaike's Information Criterion (AIC) with a lower AIC indicating a better fit (Akaïke 1974, Sakamoto et al. 1986). The AIC function indicates a better fit of the response variable at the year of electrofishing sample with mean slope, altitudinal gradient and relative distance (model 1 Table5.1).

For this model the kappa=0.4±0.04 with a threshold = 0.6. GAM explained only 14% of the deviance of the abundance of the yellow eel. Three explanatory variables are significant (Table 5.2): year, slope, altitudinal gradient.

Table 5.1. Model selection results using Akaike's information criterion (AIC) for presence-absence model analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

model	year	month	rain_mean	slope_mean	altitudinal_gradient	relative_distance	p_agricultural	p_no_impact	AIC s=3
1	x			x	x	x			747.6
2	x	x		x	x	x			748.7
3	x			x	x	x		x	751.0
4	x	x		x		x		x	751.0
5	x			x	x	x	x		751.1
6	x	x		x	x	x		x	751.2
7	x	x		x		x	x		751.2
8	x	x		x	x	x	x		751.3
9	x	x	x	x		x			751.9
10	x	x	x	x					752.1

Model Goodness of fit (presence-absence model)

The best presence-absence model selected is:

d != 0 ~ s(annee,3) + s(slope_mean,3) + s(altidudinal_gradient,3) + s(relative_distance,3)

matrice confusion

observed

predicted 1 0

1 306 76

0 107 155

correctly predicted 0.72

present correctly predicted 0.74

absents correctly predicted 0.67

Table 5.2. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar Chisq	P(Chi)	
(Intercept)	1				
s(annee, 3)	1	2	14.9130	0.0005778	***
s(slope_mean, 3)	1	2	26.4848	1.774e-06	***
s(alt_gradie, 3)	1	2	16.1909	0.0003050	***
s(relative_distance, 3)	1	2	3.7213	0.1555901	

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

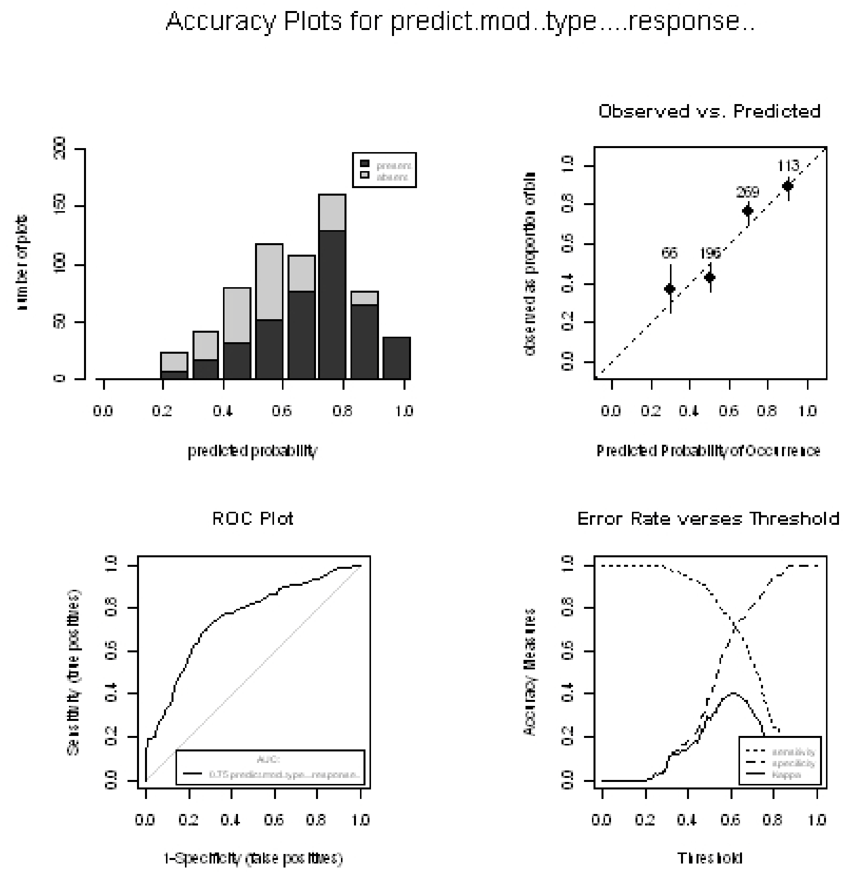


Figure 5.9. Model quality and threshold selection graphs for presence-absence model selected with a histogram plot (upper left), a calibration plot (upper right), a ROC plot with the associated Area Under the Curve (AUC) (lower left), and an error rate verses threshold plot (lower right).

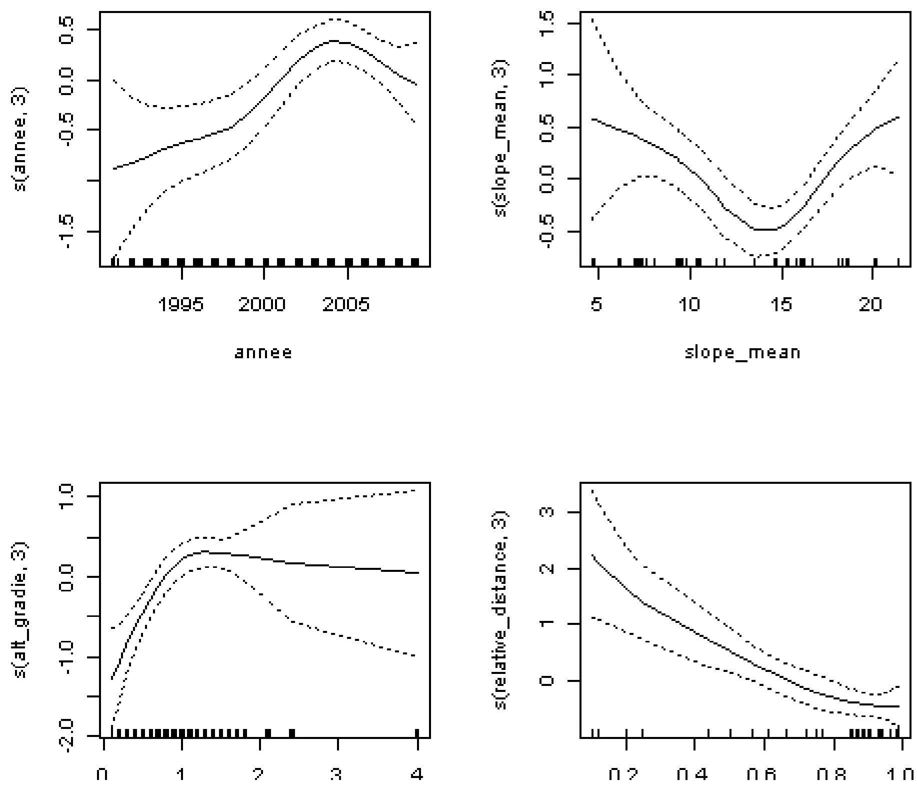


Figure 5.10. Response curves of each variable included in the generalized additive model (GAM) for the presence/absence model for Western EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The residual of the presence absence model are plotted for each year (Figure 5.11). If the results of the fishing operation are in overplot (several year point on top of the other), they are slightly moved around their origin to show on the map. Several operations for the same year will remain in overplot. The residuals correspond to predicted-observed values, hence a blue spot will indicate a predicted value larger than the observed one, and a red point a predicted value lower than the observed one.

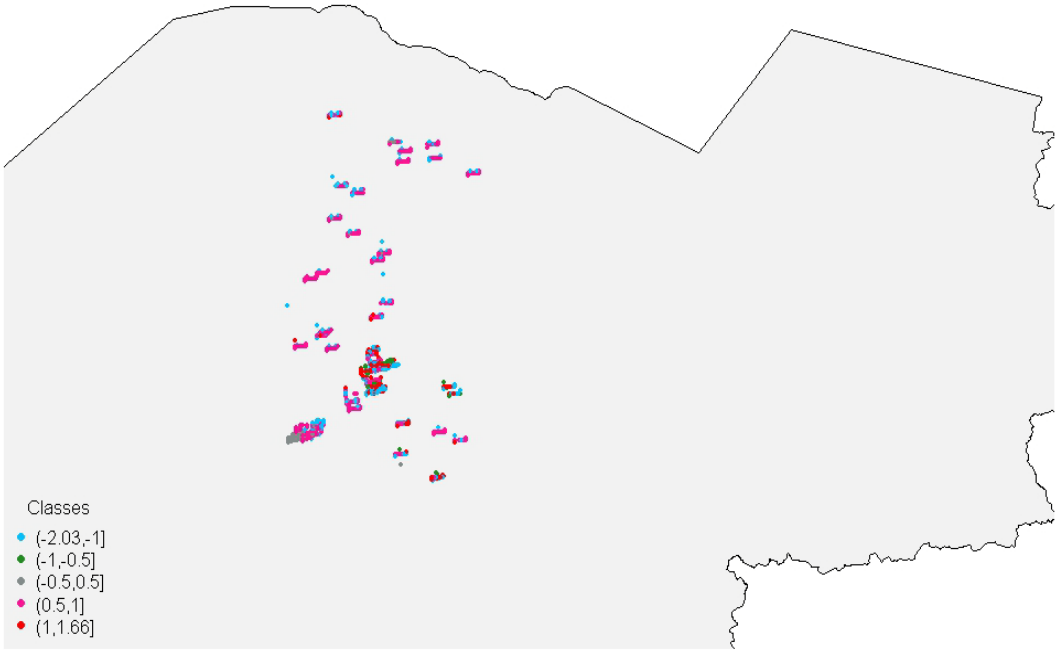


Figure 5.11. Residuals per year: presence-absence model.

In the next graph (Figure 5.12), the residuals are averaged for each station. A blue spot is a station with on average over the years predicted values far larger than the observation. The size of the point is related to the number of electrofishing operation in the station.

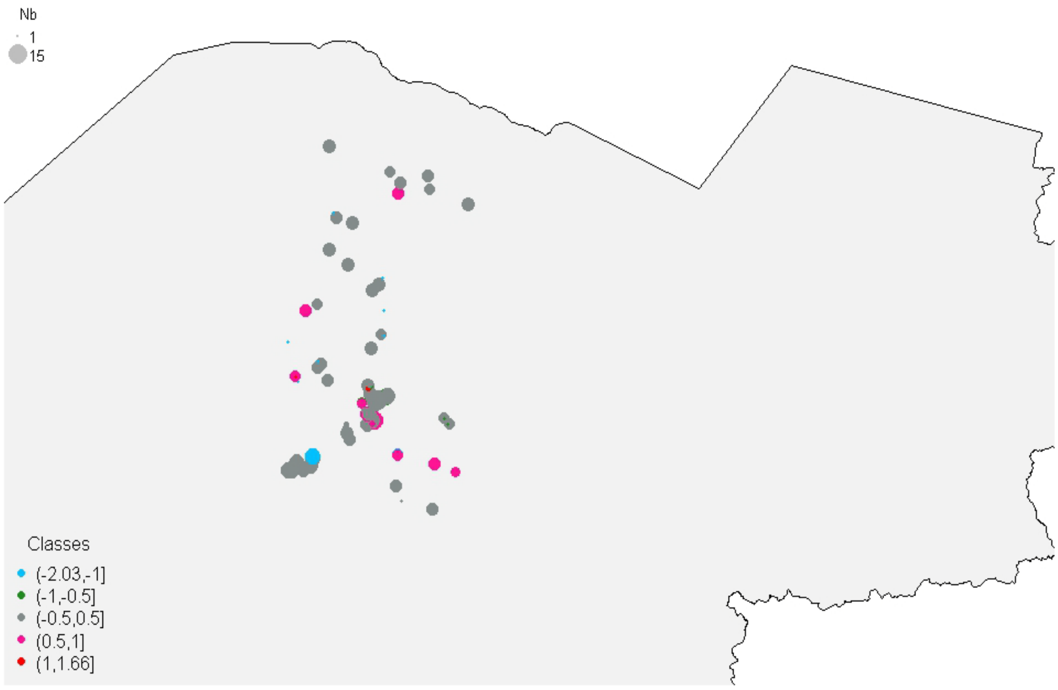
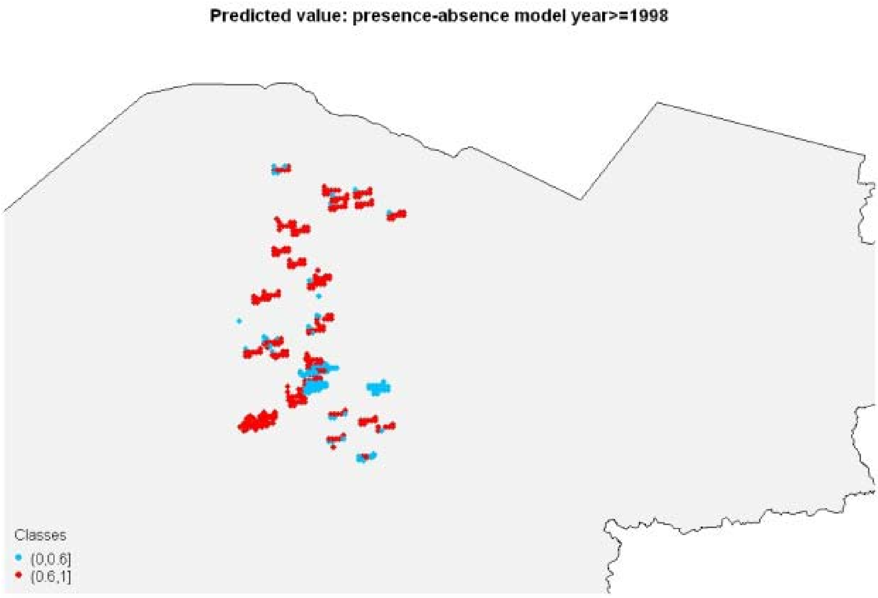


Figure 5.12. Mean Residuals per stations: presence-absence model.

The map of predicted value (Figure 5.13) from the model is computed before and after the break point in the year variable response trend. The threshold chosen for the response is the value giving the best Kappa, i.e. trade-off between the lowest number of wrongly predicted presence (false positive), and wrongly predicted absence (false negative).



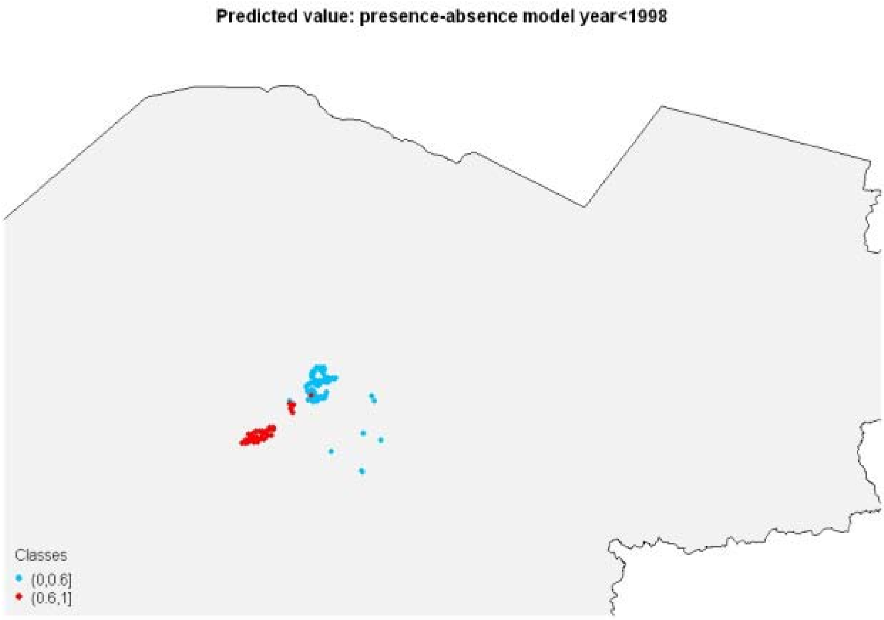


Figure 5.13. Predicted value: presence-absence model year<1998 (upper chart), Predicted value: presence-absence model year≥1998 (lower chart)

Density model

The eel density model is applied only to those sites where eel were caught, i.e. sites with positive values (>0). It is a gam model with a gamma distribution and a logarithm link. 480 models have been tested.

Model results are summarized in Table 5.3. The AIC function indicates a better fit of the response variable at the year and month of electrofishing sample with rain, slope, distance from the sea, distance from the source, and the upstream percent of agricultural land use (p_up_agricultural). GAM explained 37% of the deviance of the abundance of the yellow eel. The effects of the six explanatory variables in the model (year, month, slope, rain, distance sea, distance source, upstream percent of agricultural land use) are significant (Table 5.4). The Spearman rank correlation between the observed values and the fitted values is statistically significant (ρ=0.528, p-value<2.2 10^{-16}, Figure 5.14).

Table 5.3. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

model	year	month	rain_mean	slope_mean	altitudinal_gradient	distance_sea	distance_source	p_agricultural	p_up_agricultural	p_no_impact	p_up_no_impact	AIC s=3
1	x	x	x	x		x	x		x			1813.5
2	x	x	x	x		x	x				x	1813.6
3	x	x	x	x		x	x					1822.3
4	x	x	x	x		x	x	x				1823.7
5	x	x	x	x		x	x			x		1823.7
6	x	x		x		x	x	x				1824.9
7	x	x		x		x	x			x		1824.9
8	x	x	x	x			x		x			1825.1
9	x	x	x	x			x				x	1825.1
10	x	x		x	x	x	x			x		1825.9

Model Goodness of fit

The best density model selected is: d ~ s(year, 3) + s(month, 3) + s(rain_mean, 3) + s(slope_mean, 3) + s(distance_sea, 3) + s(distance_source, 3) +s(p_up_agricultural, 3)

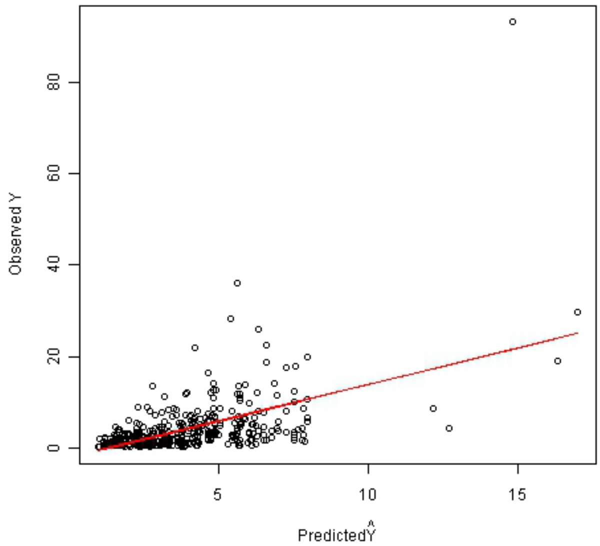


Figure 5.14. Observed vs. predicted regression scatter plot.

Table 5.4. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar F	Pr(F)	
(Intercept)	1				
s(annee, 3)	1	2	16.8163	9.87e-08	***
s(month, 3)	1	2	3.0771	0.0472117	*
s(slope_mean, 3)	1	2	11.8849	9.75e-06	***
s(rain_mean, 3)	1	2	7.2052	0.0008455	***
s(distance_source, 3)	1	2	3.4802	0.0317583	*
s(distance_sea, 3)	1	2	8.2048	0.0003231	***
s(p_up_agricultural, 3)	1	2	6.6250	0.0014809	**

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

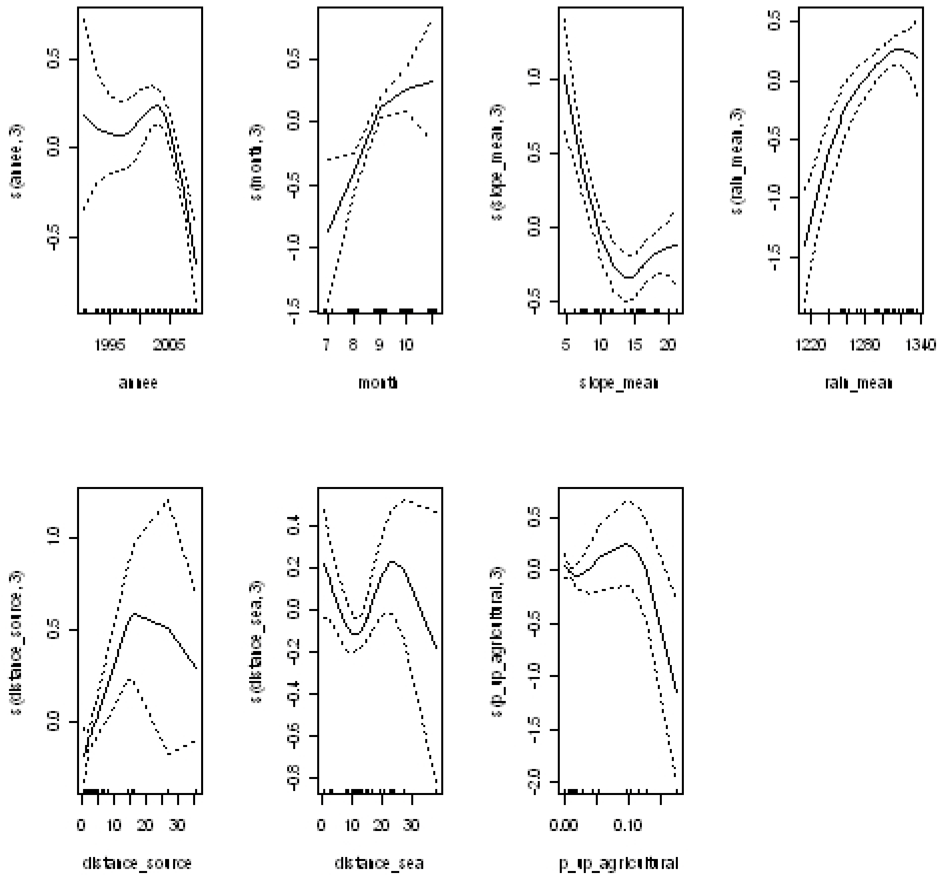


Figure 5.15. Response curves of each variable included in the generalized additive model (GAM) for the density model for Western EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The map below (Figure 5.16) shows the mean residuals for operations where positive densities were observed. The sign is calculated according to predicted – observed. The size of the points gives an indication of how many electrofishing operations occurred at that point and thus of the « weight » of that station in the model.

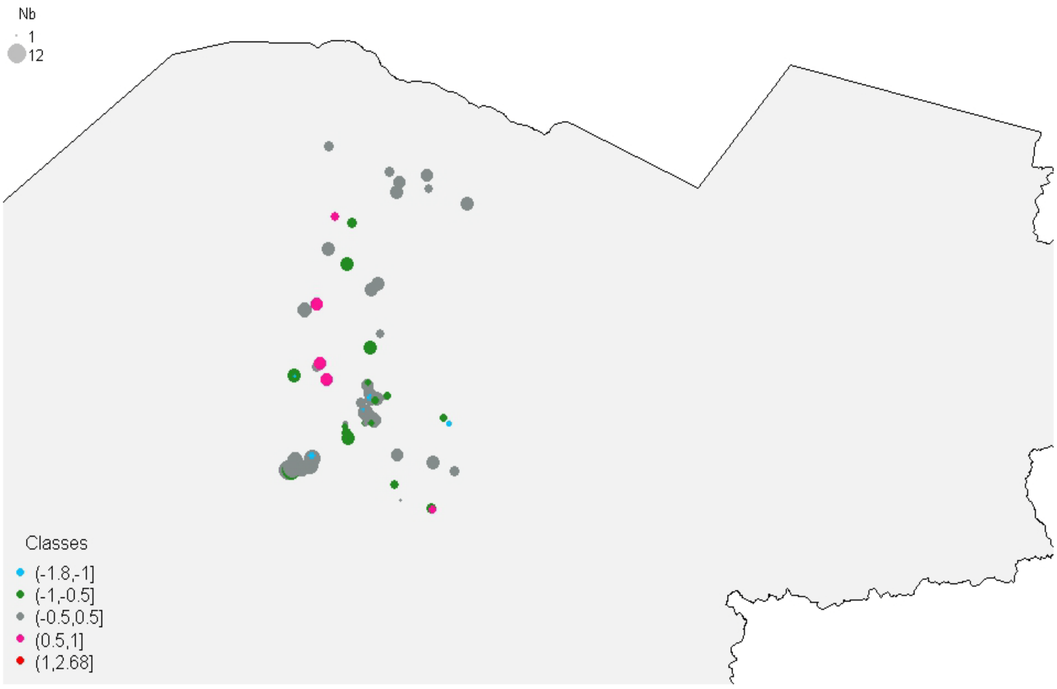


Figure 5.16. Mean residuals per stations: density model.

The next map (Figure 5.17) gives the same information but residuals are « jittered » around the point. There is no clear spatial trend in the presence of large positive or negative residuals.

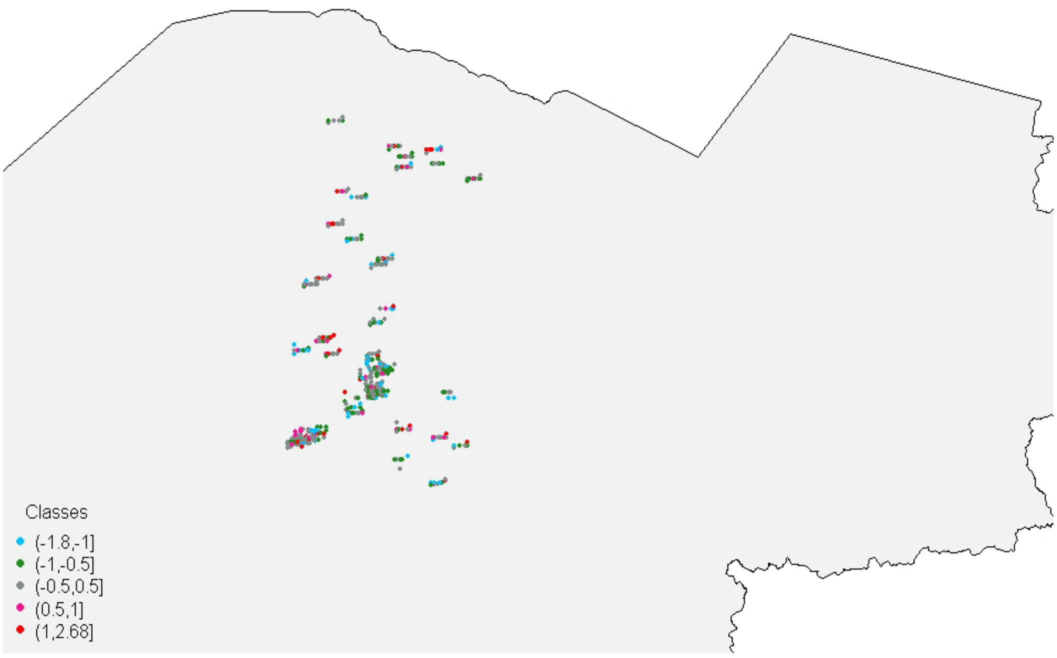


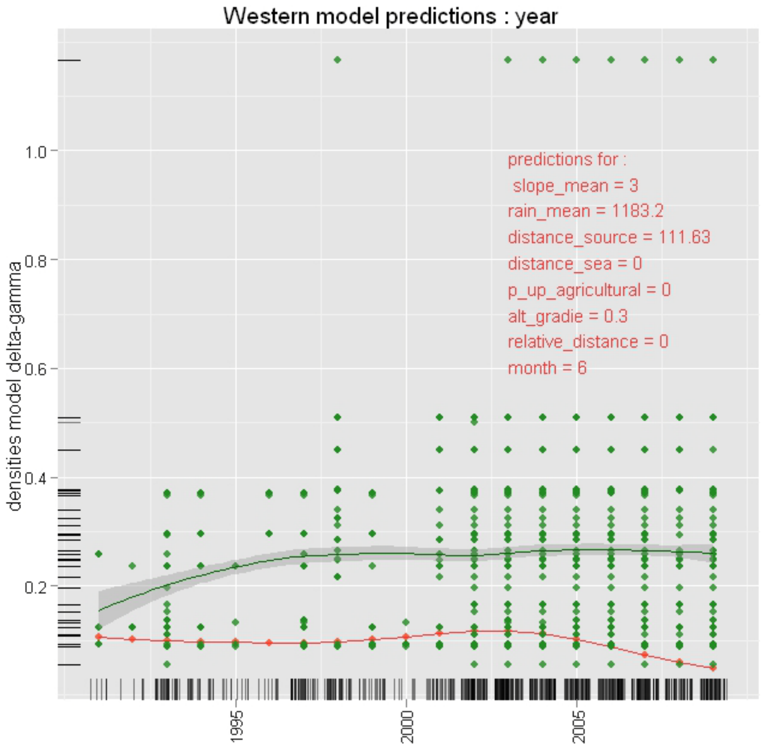
Figure 5.17. Residuals per year: density model.

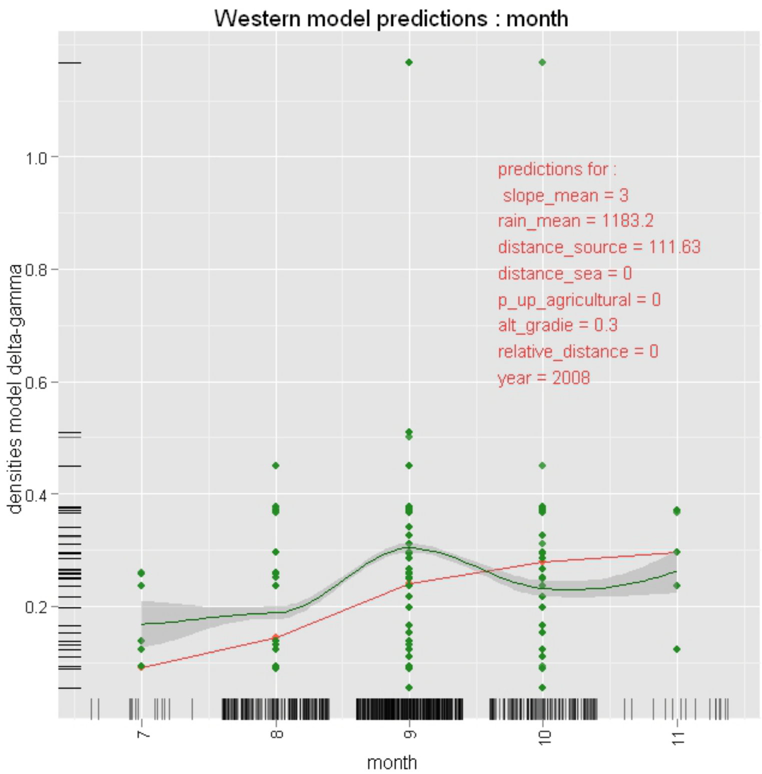
River width

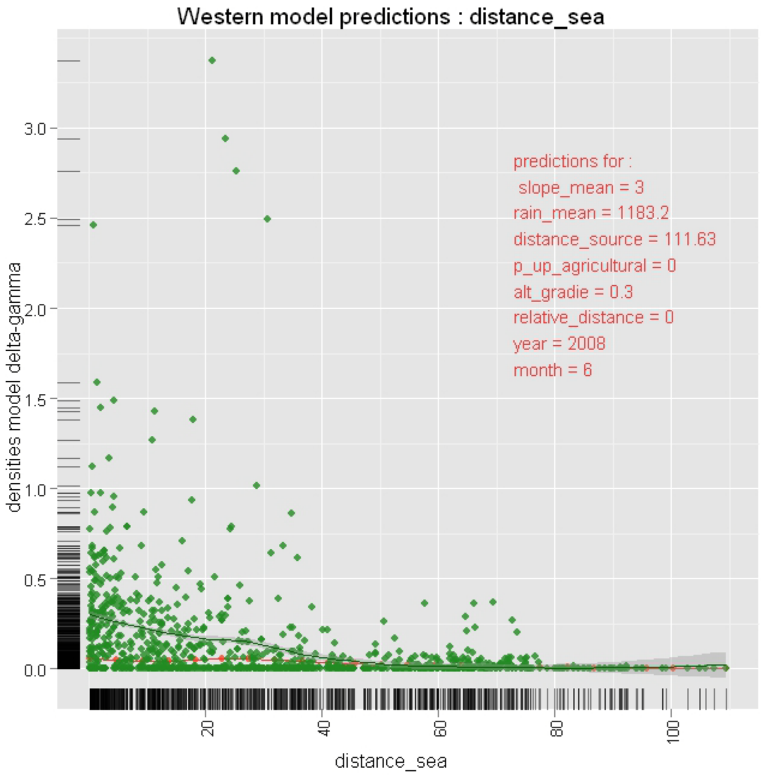
The river width has been calculated from observed width (N=38) of river stretches. A linear model was used to predict river width using the following variables: shreve order, altitudinal gradient and the basin area upstream from the river stretch. Given the paucity of data, no log transformation was applied to the dataset. The model should be based on a larger dataset as the predicted surface area 52.6 km^2 is still larger than area reported by the marine institute 33.42 km^2.

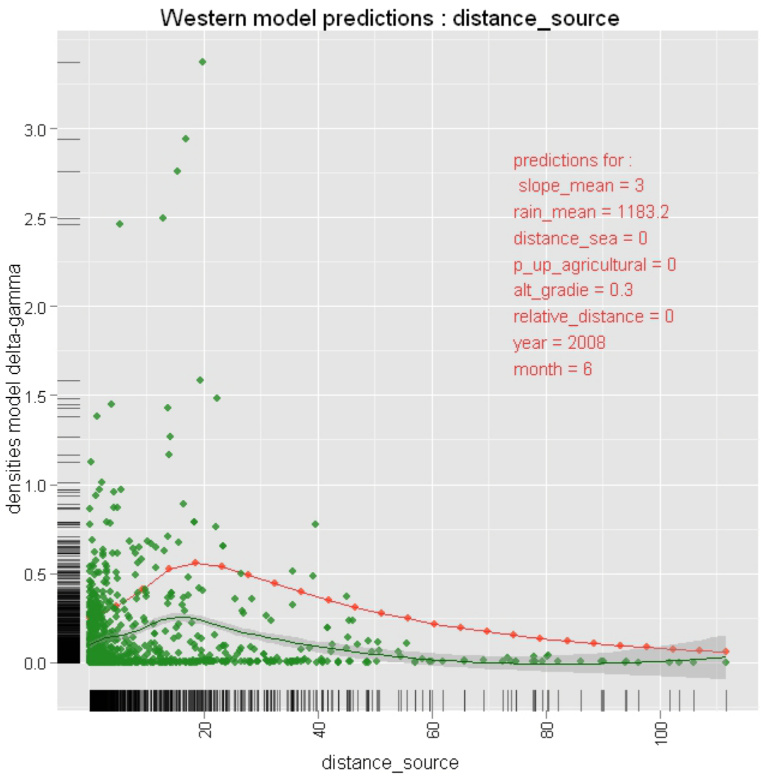
Final model

The final model is the product of the delta model by the gamma model. The delta model is the probability of having a positive density. The gamma model is the level of a density for positive values. The model is used to predict densities on all the river stretches of the EMU. The following series of figures (not numbered) present the trend (red curve) of an effect given that all other parameters are fixed (see figure for the value of fixed parameters) as well as the predicted density of each river stretch of the CCM (green points) and its average along the examined effect (green curve). The rugs on the scale have been moved slightly to indicate the density of observation. The densities for the following graphs are provided in eel.m^{-2}

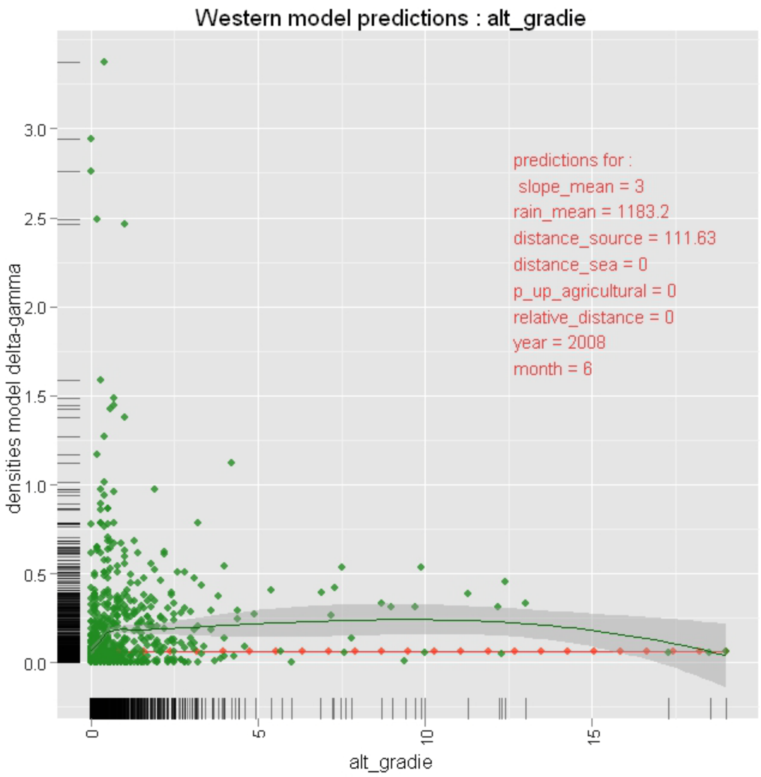

Western model predictions : year
densities model delta-gamma
1.0
0.8
0.6
0.4
0.2
1995
2000
2005
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
month = 6

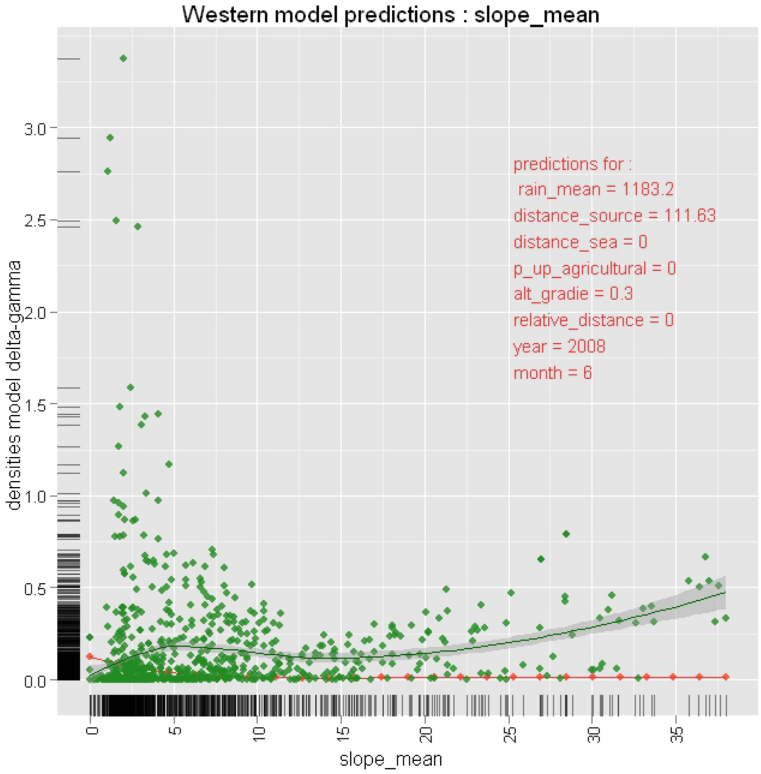

Western model predictions : month
densities model delta-gamma
1.0
0.8
0.6
0.4
0.2
7
8
9
10
11
month
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
year = 2008


Western model predictions : distance_sea
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_source = 111.63
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
year = 2008
month = 6
densities model delta-gamma
distance_sea

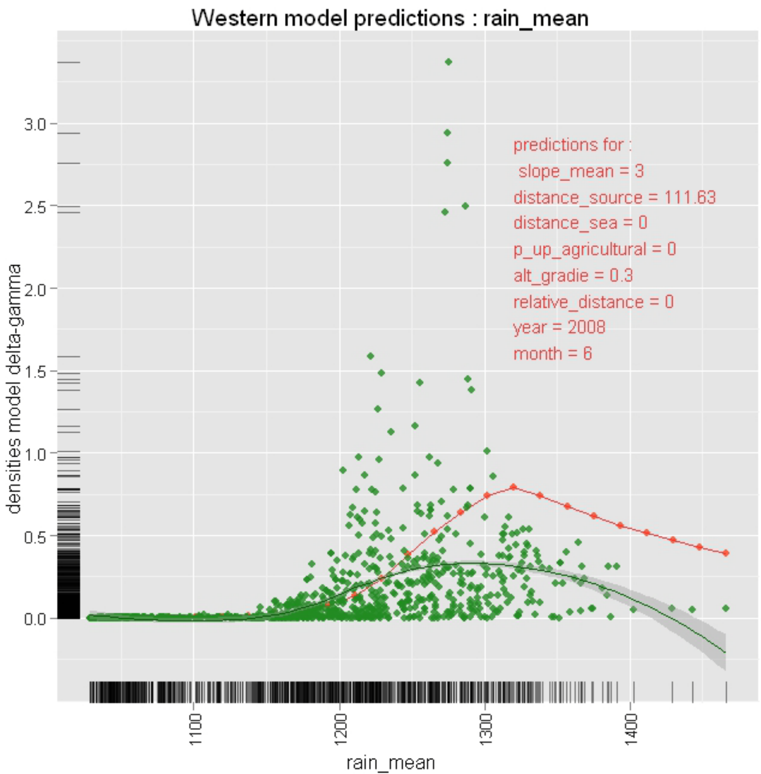

Western model predictions : distance_source
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
year = 2008
month = 6
densities model delta-gamma
distance_source


Western model predictions : relative_distance
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
year = 2008
month = 6
densities model delta-gamma
relative_distance

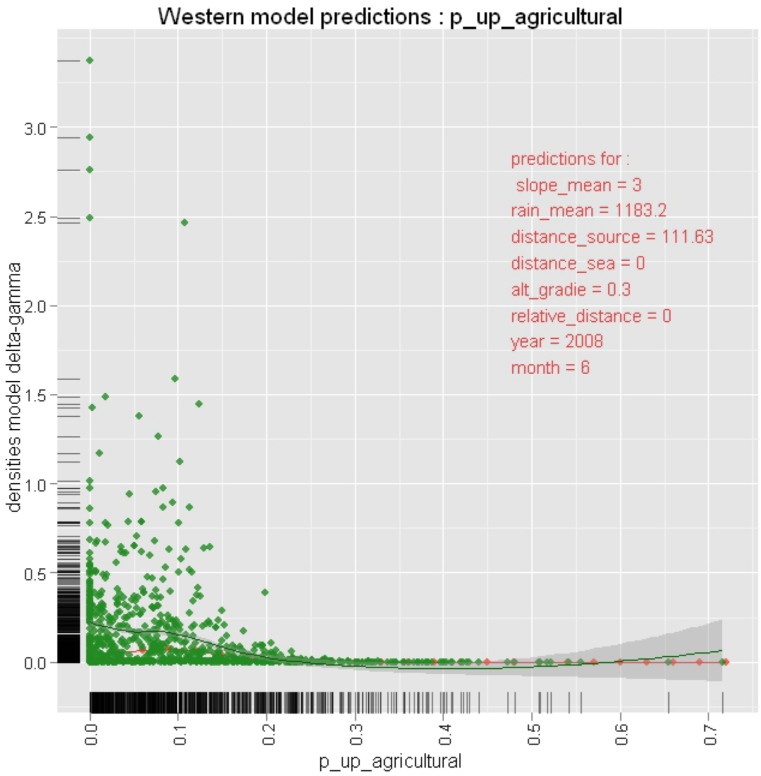

Western model predictions : alt_gradie
predictions for :
slope_mean = 3
rain_mean = 1183.2
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
relative_distance = 0
year = 2008
month = 6
densities model delta-gamma
alt_gradie



Western model predictions : slope_mean
predictions for :
rain_mean = 1183.2
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
year = 2008
month = 6
densities model delta-gamma
slope_mean



Western model predictions : rain_mean
predictions for :
slope_mean = 3
distance_source = 111.63
distance_sea = 0
p_up_agricultural = 0
alt_gradie = 0.3
relative_distance = 0
year = 2008
month = 6
densities model delta-gamma
rain_mean



The following figure (Figure 5.17) gives the yellow eel production along the examined variable. The insert chart in the upper corner also gives the cumulated wetted area along the same variable that supports that production.

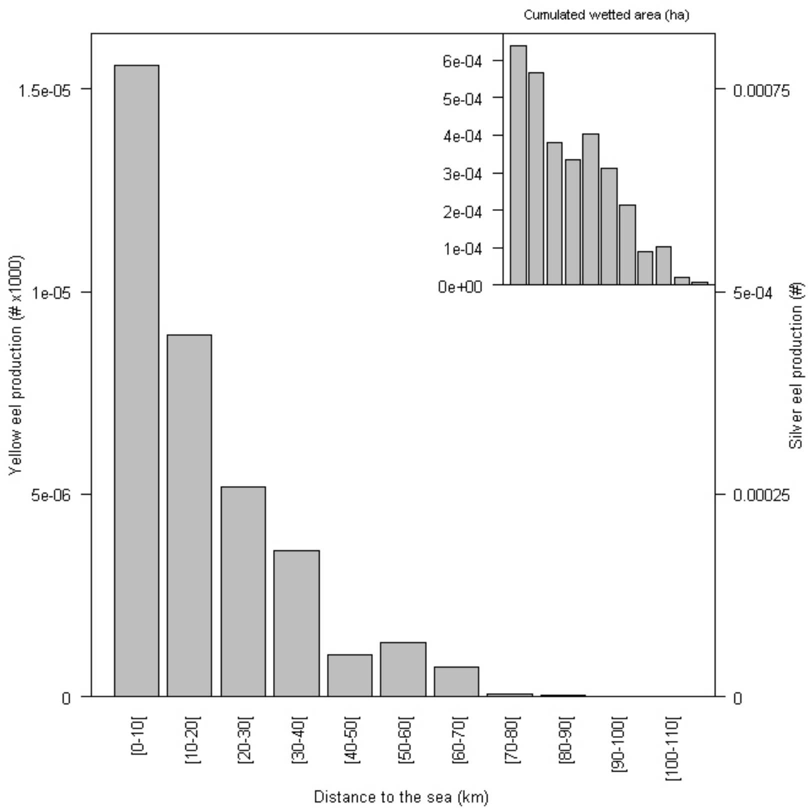


Figure 5.17. Yellow and silver eel production in number of eels along the distance to the sea.

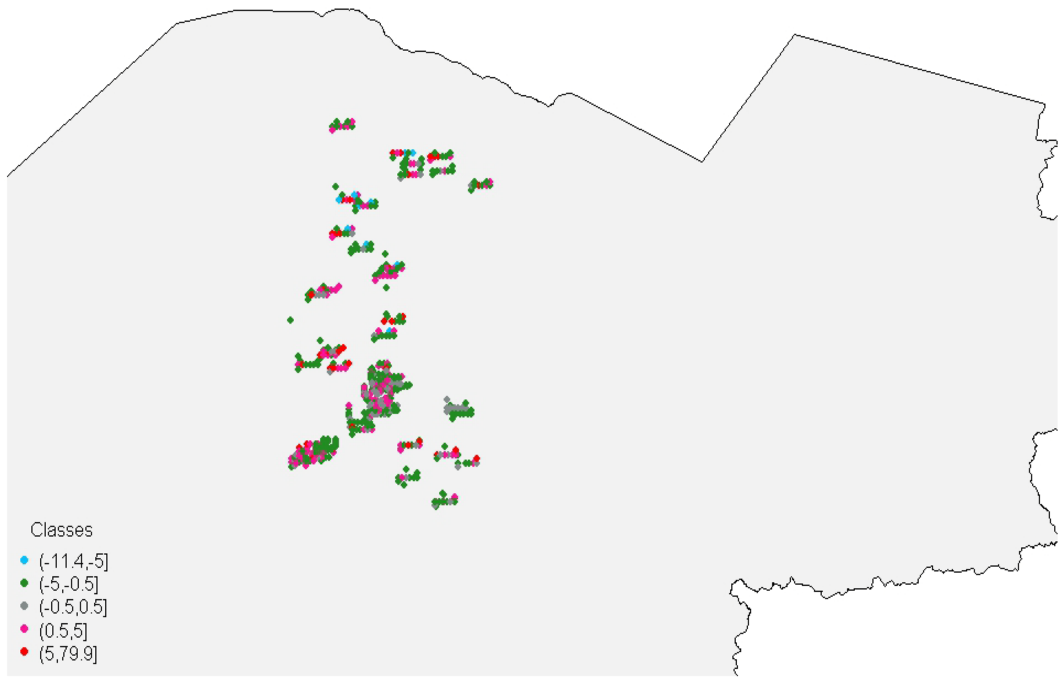


Figure 5.18. Map of residuals (predicted - observed), for the full model. Each year of data a point is shown with a small distance around the point, residuals in eel.m^{-2}.

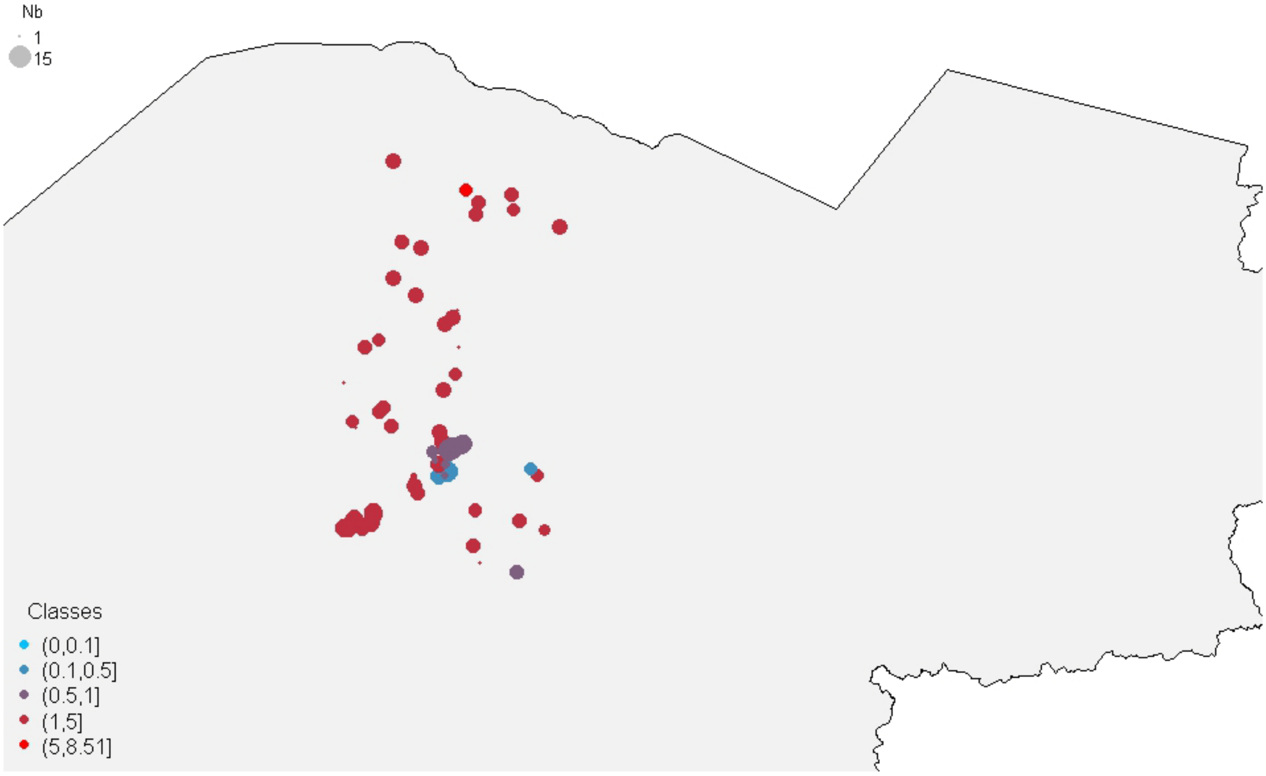


Figure 5.19. EDA predictions for the full model, the size model is given according to the number of electrofishing at that point classes in eel.m^{-2}.

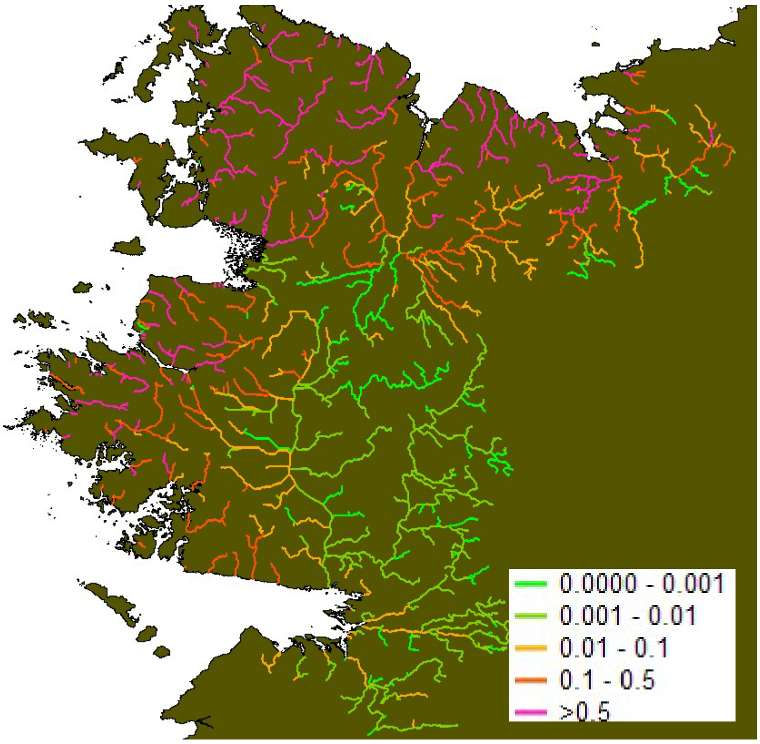


Figure 5.20. EDA predictions for densities (eel.m^{-2})

EDA predictions for the Western River Basin District

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch with a conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al*. (2000) have estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA a constant conversion rate of 5% was chosen as a default value. This constant rate conversion of eel in equivalent silver eel is based on the assumption of density-independent biological processes.

This potential escapement in number in then converted into biomass with the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for WRBD. The current biomass ($B_{current}$) is calculated by subtracting the anthropogenic mortalities on silver eel to the potential escapement in biomass.

Best achievable escapements (B_{best}) can be calculated by the current biomass artificially forced to null anthropogenic impact (no dam, land use mortality to "no impact" and silver eel catch to 0) added with silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historical level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment.

Table 5.5 summarises the EDA estimates of the total numbers of yellow and silver eels in the CREPE EMU.

Table 5.5. Model outputs for EDA simulations of the Western River Basin District.

Model output	Value
Total water surface (km²)	52.6
Average number of yellow eel per 100 m²	10.3
Average number of silver eel per 100 m²	0.51
Total number of yellow eel	5448044
$N_{current}$: Total number of silver eel	272402

Changes made during the tuning process for the application of EDA to the Western River Basin District

In the second application phase changes of units for yellow eel densities have been carried out. To evaluate Bcurrent, Bbest and Bpristine, the mean weight of each life stage (glass eel $\overline{w}_{glass}$, yellow eel $\overline{w}_{yellow}$ and silver eel $\overline{w}_{silver}$) and the anthropogenic mortalities on glass eel Y_{glass} , yellow eel Y_{yellow} and silver eel Y_{silver} have been used. The model parameters used to evaluate Bcurrent, Bbest and Bpristine are given in Table 5.6, along with the EDA estimates of the total numbers of yellow and silver eels in the CREPE EMU.

Table 5.6. Data input for silver eel estimation for the Western River Basin District
With Y_{glass}(t= $2010-\tau$), Y_{yellow}(t= $2010-\tau+\lambda_{yellow}$) and Y_{silver}(t=**2010**) in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	18 as an average for the more productive areas 25-30 for the less productive areas.	13 Averaged from Matthews et al. 2001	0.344 Poole pers. com (data for Jan-March 1988 and 2005)	178	217 (Burrishoole) 300 (Corrib Fishery) 647 (Mask Fishery)	0	35000[1] (2001-2007 average) (20t declared + 15t IUU (Illegal, Unreported and Unregulated fishing) Currently zero	28000[1] (2001-2007 average) (16t declared + 12t IUU) Currently zero

[1] ***assuming that river catches are zero***

Table 5.7. Model outputs for EDA simulations of the Western River Basin District.

Model output	Value
Total water surface (km²)	52.6
Average number of yellow eel per 100 m²	0.103
Average number of silver eel per 100 m²	0.0051
Total number of yellow eel	54480.44
$N_{current}$: Total number of silver eel	2724.02
Bcurrent(t=2010) in kg	1056.9
$\mathbf{B_{best}}=B_{current}$(t=2010,hi,j=Ø,Ysilver(t)=0) in kg	1312.8
$B_0 = \overline{B_{best}\left(t < 1980\right)}$ in kg	2874.71[1]

[1] ***calculated for the earliest year available: 1991***

5.3. The Corrib river basin within the Western River Basin District (Ireland)

Case study details

The Corrib catchment area comprises about 3,000 km^2 and including the three large lakes of Corrib, Mask and Carra (Figure 5.21). The largest of these, Lough Corrib, lies 6 km upstream from the sea. All three lakes are mesotrophic, the lowest total alkalinity being 1.5 mg per litre, although artificial enrichment is becoming a problem.

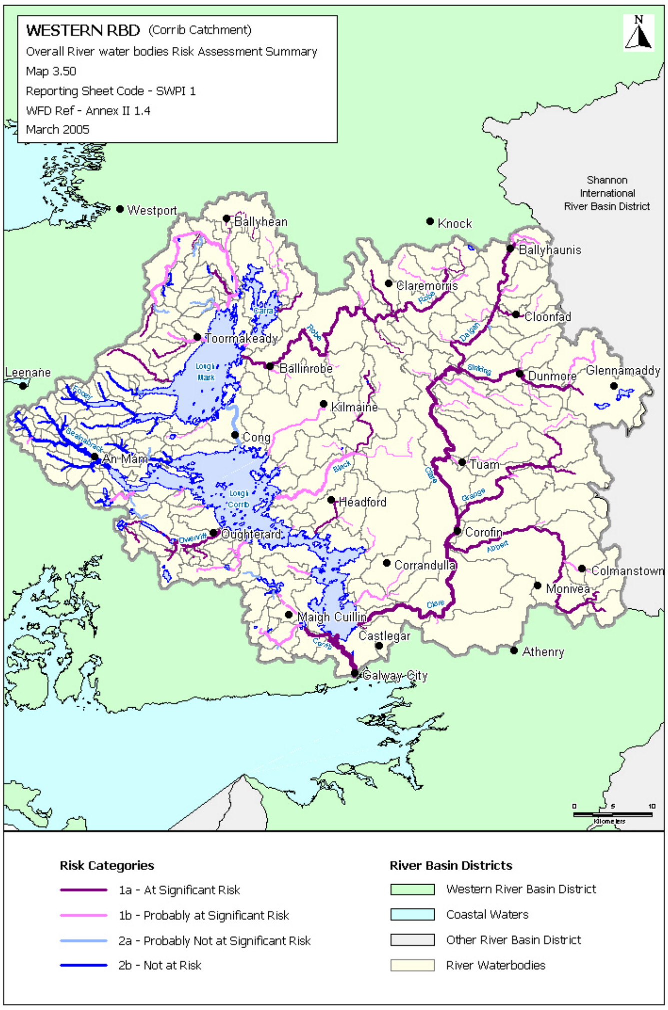


Figure 5.21. The Corrib catchment and summary of the river risk assessment.

The status of the fishery prior to 2010 was one of limited catches, declining stocks and poor recruitment.

Recruitment to the Corrib catchment dramatically reduced after 1984. A study on elver migration into the Corrib catchment was undertaken between 1980 and 1982 (McGovern & McCarthy, 1992). Upstream ascent of elvers into Lough Corrib and throughout the system is difficult. It seems possible that mortalities are high (Moriarty 2001). Galway fishery Annual Reports record that 100 kg of elvers were trapped and transported in 1978, 75 kg in 1984, 5 kg in 1987 but runs were too small to warrant transportation since then.

The yellow (brown) eel stock would appear to have fallen in the lower basins (using CPUE as a proxy) but remained more consistent in upper basins until 2001 (Moriarty, 2001).

Surveys were carried out in the catchment in 1967, 1990 and in 2001 (Moriarty, 2001). In the southern basin of Lough Corrib (nearest the sea) the catch in 1990 was less than half that of 1969 and large eels had become very scarce. CPUE in the northern basin was more or less the same, medium and larger sized eels were more plentiful and smaller eels were fewer over the same time period. Assuming that fyke net CPUE is proportional to the stock density, it would appear that the stock in the southern basin has fallen to less than half that of the 1960s while the northern basin is more or less unchanged (Moriarty, 2001).

All the eels which survive to the silver stage and migrate to the sea have to pass through the River Corrib and the city of Galway. The Galway Fishery comprises a weir with 14 coghill nets that have been fished consistently by the state since 1978. The downward trend in silver eel catch (Figure 5.21) therefore probably reflects the decreasing stock in the greater Corrib catchment and falling silver eel escapement.

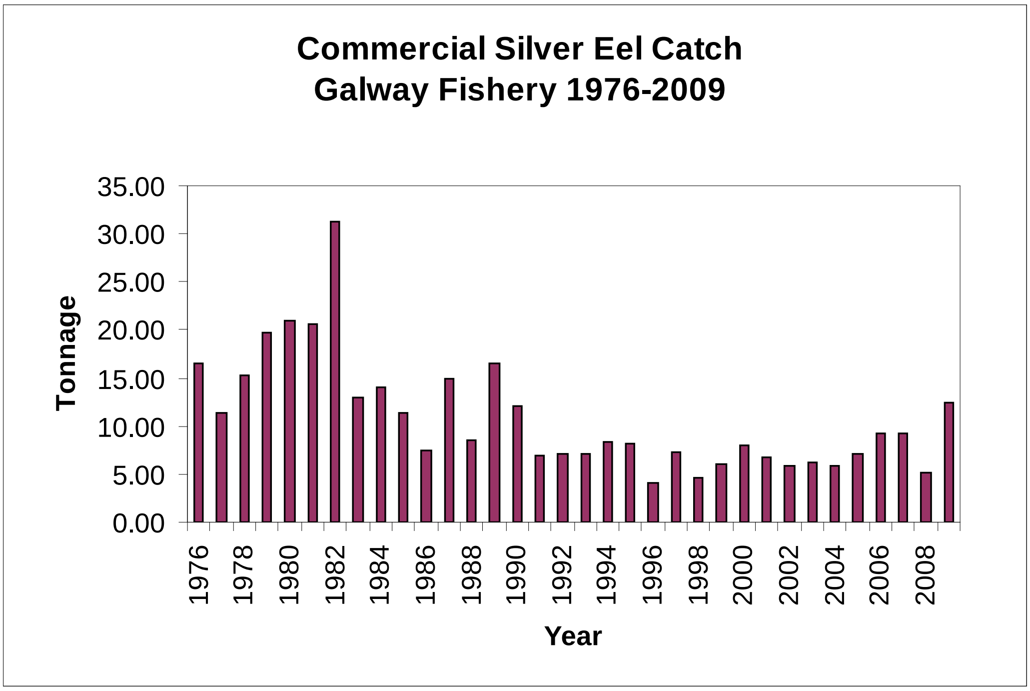


Figure 5.21. Annual silver eel catch (t) in the commercial Galway Fishery, Corrib System, for 1976 to 2009; 2009 data from research fishery.

5.4 Application of DemCam to the Corrib data set

DemCam simulations were run from 1900 to 2010.

Input data

Life history parameters

The *a* and *b* constants for the allometric length (cm) /weight (g) relationship were set as a = 8.34×10^{-4}, b = 3.17.

The average age of male and female silver eels were set as 12 and 20 years, respectively. The average lengths of male and female silver eels were set as 380 and 548 mm, respectively.

Recruitment

There are no quantitative or time series data on recruitment of eel to the Corrib river basin. We therefore referred to the time series of glass eel catches on the nearby Shannon basin (see application to Burrishoole, earlier), and arbitrarily assumed that pristine glass eel recruitment for the period before 1977 was about 9,500 kg per year, which is equivalent to about 1000 eels per hectare.

A time series index has been produced according to the Shannon data, on the assumption that recruitment was at a constant high before 1980, and then followed the index of the Shannon catch for more recent years.

The average weight of glass eel recruits was set as 0.33 g, on advice from Russell Poole.

Carrying capacity

The carrying capacity for elvers was set at 2000 elvers per hectare, after a series of trials revealed this level to produce sex ratios values close to those observed in the silver eel catches.

Environment

The simulation was run on a single compartment with a wetted area of 28,869 hectares, and assuming that the entire wetted area was a homogenous habitat. The annual average water temperature was set at 11.0 C, on advice from Elvira DeEyto.

Impacts

Fishing for yellow eel (in loughs) and silver eel (in the outflow River Corrib) occurred until 2009, when all fishing was closed as part of the national measures in relation to Irish Eel Management Plans. In order to obtain a fishing mortality rate for silver eel similar to those assessed by the data providers (i.e. $F_{silver} = 0.91\ yr^{-1}$ and consequent removal of 35% of the silver eel stock) we set an annual fishing effort equal to 264,000 gear*day, which approximately corresponds to an effort executed by 30 fishermen, each using 27 fishing gears and operating 330 days per year. The catchability of silver eel was set at 0.1, in order to obtain an annual silver eel survival of 0.4 for a given unit of effort, as suggested by Russell Poole. The catchability of yellow eel was set at 0.01, which was chosen on an arbitrary basis, being 1 order of magnitude lower than that for silver eels.

Results

The results of the DemCam simulations are presented in Figures 5.22 to 5.25, which illustrate the predicted time series of glass eel recruitment, silver eel production, yellow eel standing stock and catches of yellow and silver eel.

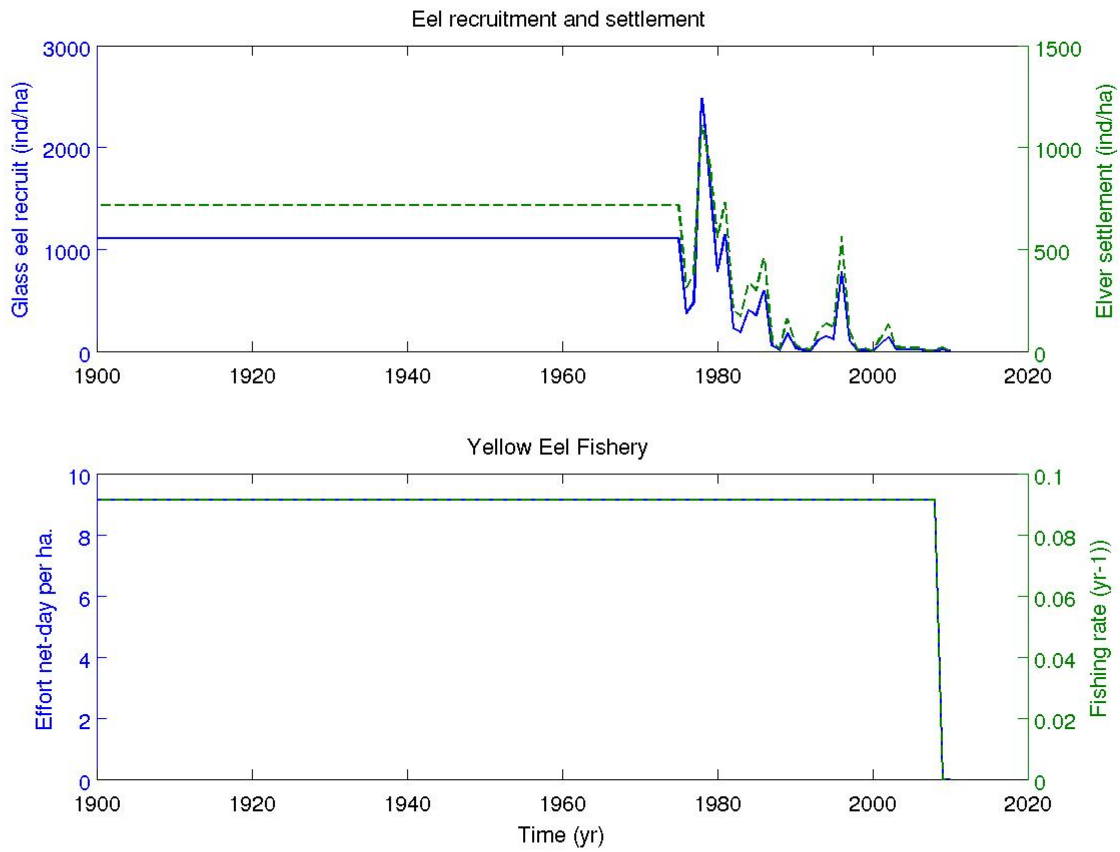


Figure 5.22. DemCamsimulation of glass eel recruitment and elver settlement for the Corrib (upper chart) and yellow eel exploitation rates. Note the different scales on the two y-axes of each chart.

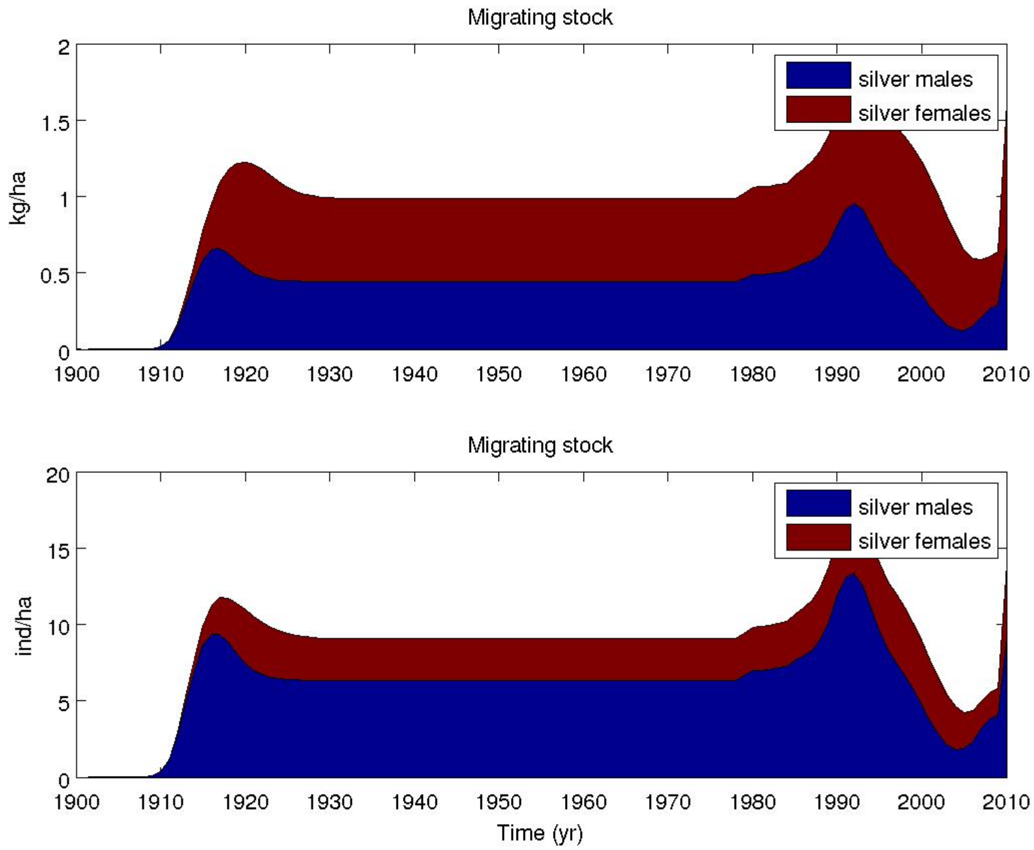


Figure 5.23. DemCam simulation of silver eel production in the Corrib. The top chart shows silver eel production for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

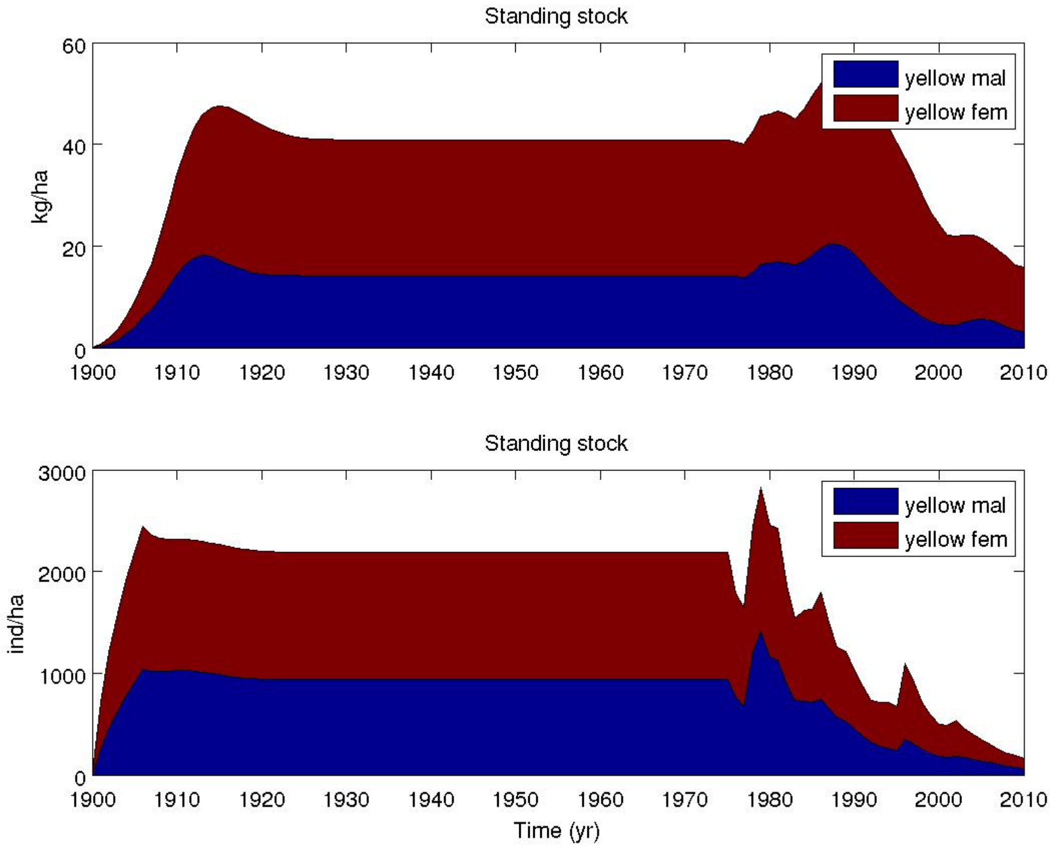


Figure 5.24. DemCam simulation of standing stock of yellow eel in the Corrib. The top chart shows yellow eel stock for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

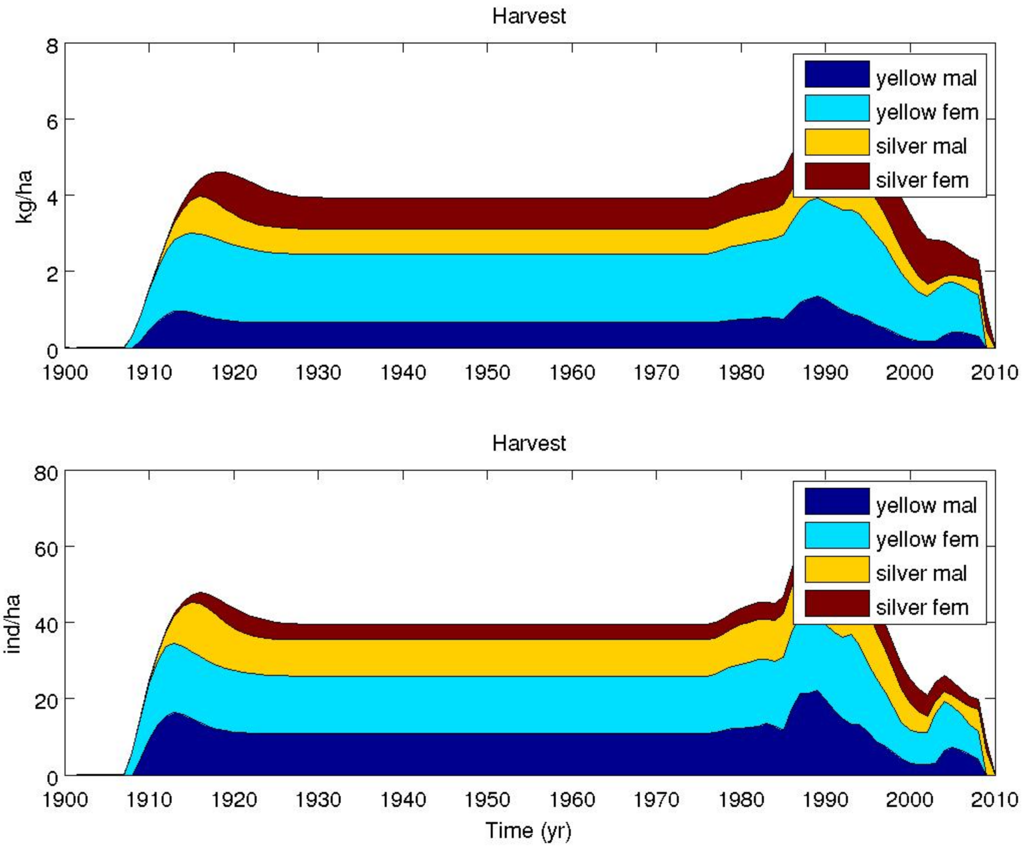


Figure 5.25. DemCam predictions of the catch of male and female yellow and silver eels during the time series, expressed as kg per hectare in the upper chart, and individuals per hectare in the lower chart.

DemCam predicted that the pristine production of silver eels (i.e. production at high recruitment and excluding the effects of the fishery) was about 5 kg per hectare, whereas the escapement under high recruitment would be about 1 kg/ha. The model predicted that silver eel escapement in the 2000s, just prior to the closure of the fishery would have been about 0.7 kg/ha, and that this should have risen to about 1.4 kg/ha after the closure of the fishery.

Considerations for model application

The process of 'tuning' DemCam to the eel data from the Corrib revealed that the simulations were sensitive to the parameterisation of recruitment, both in terms of pristine level and time series, and the selection of an appropriate carrying capacity.

The selection of these values required several iterative applications of the model, and therefore the results of application of the model to other data sets should be carefully considered. That said, however, DemCam has now been applied to a number of eel stocks and a suite of suitable parameters is increasing with every application, which should lead to greater confidence in the results of application to other eel stocks that are similar in characteristics as those tested so far.

5.4. Application of GEM to the Corrib data

The following input values and data were derived from the data provided by Russell Poole and Elvira De Eyto. Data were available for the period from 1977 – 2008.

The total wetted area of the river basin was set at 28,869 ha, which was based on 847 ha of rivers and 28022 ha of lakes.

The model was run with eels of ages 0 to 35 years.

It was assumed that following relations and model parameters were constant for the whole period

Weight – length relationship:	$W = 0.00083\ L^{3.18}$
Growth of males:	$L = 57.2 * (1 - e^{(-0.0823*(Age\ -2.01)})$
Growth of females:	$L = 89.6 * (1 - e^{(-0.045*(Age\ -2.4)})$

The natural mortality of male and female eel was estimated based on the method described by Bevacqua *et al.* (2011), where it was assumed that the eel density is low and that the mean water temperature is 10.3 °C.

Recruitment

Direct estimates of the number of immigrating elvers (recruitment) were not available for the Corrib river basin. However, a recruitment index was available for the nearby River Shannon, to the south of the Corrib, where there is an elver trap at the Ardnacrusha Power Station. In the absence of a recruitment measure for the Corrib, we assumed that recruitment might be similar in characteristic to that in the Shannon, and proportional to the relative wetted area (as a proxy of freshwater outflow) of the two basins. As the wetted area of the Corrib basin is about 68% that of the Shannon basin, we derived a recruitment level and constructed an index of recruitment relative to the annual recruitment of elvers prior to 1977 (Figure 5.26). This may not be a valid approach given the very catchment specific variations of recruitment being observed and remains one of the main weaknesses of the application of GEM to the Corrib data.

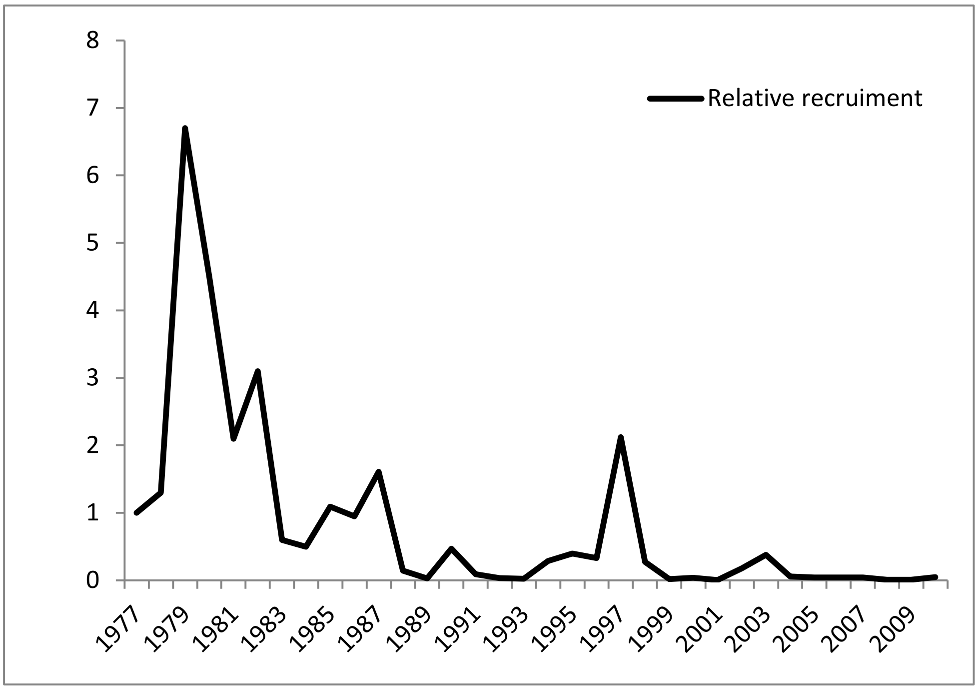


Figure 5.26. Time series of catches of elvers at Ardnacrusha Dam on the River Shannon, expressed as an index of the catch in 1977

The mean length of immigrating elvers was estimated as 7.078 cm and the mean weight as 0.263 g,

Fisheries

There has been no commercial fishing for glass eel.

Traditionally, the yellow eel stocks of the main lakes were exploited using baited long-lines and small summer fyke nets. Angling effort for eel in the area is low and cormorant predation is not thought to be significant. Both commercial and angling exploitation of yellow eels ceased in 2009.

The main silver eel fishery was the Galway Fishery at the outflow of the catchment, which comprised a weir with 14 coghill nets that have been fished consistently by the state since 1978. When the silver eel fishery was closed in 2009, the State continued the Galway fishery as a research effort in 2009. The Galway fishery was closed in 2010. Some silver eel fishing also took place in the catchment, especially at the outflow of Lough Mask, but this was also closed in 2009.

Total catch in kg by area and eel stage were available from 2003 to 2008. Prior to this, the only data available were the silver eel catches from Galway Weir from 1976 to 2002. We used the data from 2003 to 2008 to derive raising factors to estimate the total catch in relation to the catch in Galway Weir. These raising factors were used to estimate the total catch for the river basin between 1976 and 2002.

The selectivity of fishing gears was approximated by logistic function of the length distribution of eel captured in 2008. The growth functions of male and female eel were used to transfer the selectivity by length into selectivity by age.

Length frequencies of captured eel by sex were available for 2008 and 2009 (Figure 5.27). The length frequencies were transferred into relative age frequencies by means of the growth curves. These field data were compared with estimates of the relative age frequencies of the model.

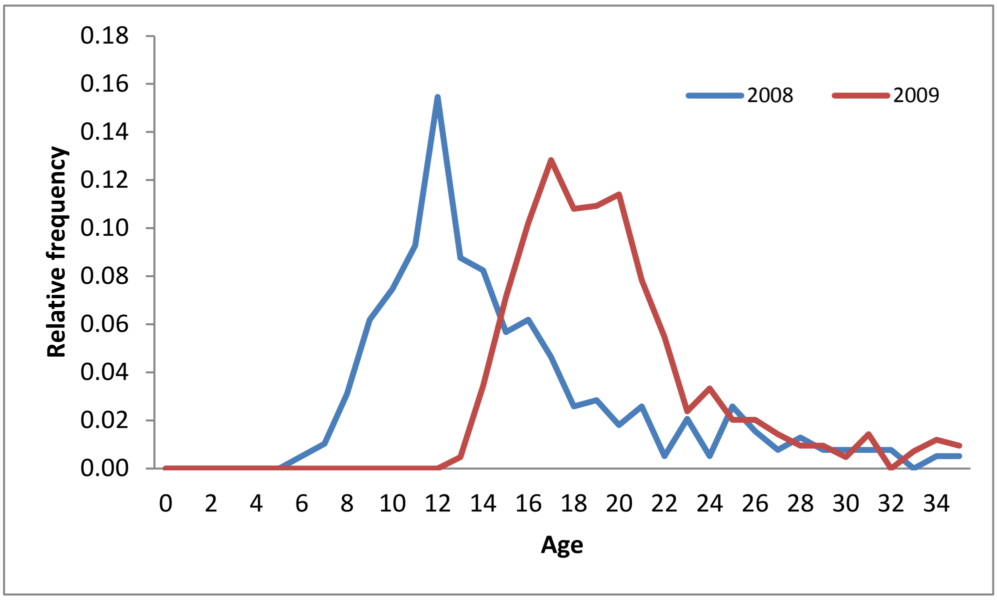


Figure 5.27. Relative age frequencies of captured female yellow eel in 2008 (blue) and captured female silver eel in 2009 (red).

Silvering of eel

The proportion of silver eel by length in the population was described by a logistic function based on combined catches of yellow and silver eels. Figure 5 shows the relative frequency of silver eel by length (blue line). The relatively high proportion of silver eels between 30 and 43 cm were taken to represent the male silver eels, whereas the high proportions about 49 cm were thought to represent the female silver eels. Hence, the relative frequencies of eel larger than 45 cm were used to estimate the parameter of the logistic curve for female silver eels - presented in Figure 5.28 as the red line. These data were again transferred into proportion of silver eel by age by means of the growth function.

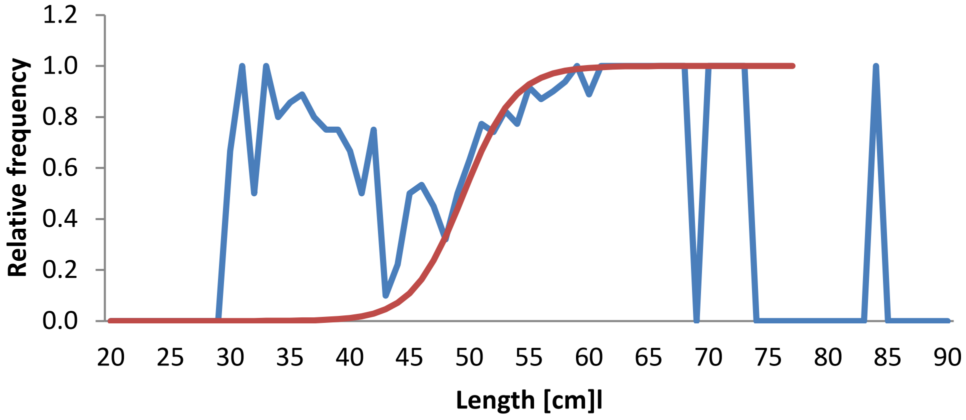


Figure 5.28. Relative frequency of silver eel by length (blue) amongst the combined yellow and silver eel catches, and logistic function for describing the proportion of female silver eel (red)

Model modifications

The GEM was developed to describe the dynamic of eel population in the river Elbe system. The use of the model in the Corrib catchment required the following modifications.

Whereas a constant proportion of male silver eel (5%) was used in the original GEM application to the Elbe data, a variable proportion of male eels was simulated in the application to the Corrib data set, albeit with a maximum proportion of 0.5. Therefore, models for male and female eel were separately prepared but linked by different parameters – Tables 5.6 and 5.7 present the age-based parameter values for male and female eels, respectively. The separated models took into account the sex-specific growth of male and female eel, and the different development from yellow eel to silver eel. In addition, the impacts of recreational angling and cormorant predation were switched off, because these factors were not reported for the CREPE data set.

The following model parameters were used to adapt the model to available field data in 2008 and 2009. It was assumed that the parameters would be similar during the total period from 1977 to 2009.

- Maximum proportion of silver eel (P_{sm} – maximum proportion of male silver eel, P_{sf} – maximum proportion of female silver eel)
- Proportion elvers by sex (P_{em} – proportion of male elvers, P_{ef} – proportion of female elvers, $P_{em} + P_{ef} = 1$)
- Proportion of total catch by sex (P_{cm} – proportion of male catch, P_{cf} – proportion of female catch, $P_{cm} + P_{cf} = 1$)
- Relative age structure of the first year

Table 5.6. Age based data for male Corrib eel applied in GEM

Age	Mean weight per age [g]	Mean length per age [cm]	Nat. Mortality low density	Relative age distribution first year	Relative catchability by age group	Relative proportion of silver eel
0	3	12.55	22.26	1.00	0.00	0.00
1	6	16.08	13.18	0.87	0.00	0.00
2	10	19.33	9.30	0.75	0.00	0.00
3	16	22.32	7.30	0.65	0.00	0.00
4	23	25.08	5.93	0.56	0.00	0.00
5	32	27.61	4.94	0.49	0.00	0.00
6	41	29.95	4.31	0.42	0.00	0.00
7	51	32.10	3.81	0.36	0.01	0.01
8	62	34.09	3.39	0.32	0.03	0.03
9	73	35.91	3.11	0.27	0.08	0.08
10	85	37.59	2.92	0.24	0.16	0.16
11	96	39.14	2.76	0.21	0.29	0.29
12	108	40.57	2.60	0.18	0.47	0.47
13	119	41.88	2.47	0.15	0.66	0.66
14	131	43.09	2.34	0.13	0.81	0.81
15	142	44.21	2.26	0.12	0.88	0.88
16	153	45.23	2.19	0.10	0.92	0.92
17	163	46.18	2.12	0.09	0.95	0.95
18	173	47.05	2.05	0.07	0.97	0.97
19	182	47.85	1.99	0.06	0.98	0.98
20	192	48.59	1.93	0.06	0.99	0.99
21	200	49.27	1.90	0.05	0.99	0.99
22	208	49.90	1.87	0.04	0.99	0.99
23	216	50.47	1.82	0.04	1.00	1.00
24	224	51.01	1.79	0.03	1.00	1.00
25	230	51.50	1.77	0.03	1.00	1.00
26	237	51.95	1.74	0.02	1.00	1.00
27	243	52.36	1.72	0.02	1.00	1.00
28	249	52.74	1.69	0.02	1.00	1.00
29	254	53.10	1.68	0.02	1.00	1.00
30	259	53.42	1.67	0.01	1.00	1.00
31	264	53.72	1.65	0.01	1.00	1.00
32	268	53.99	1.64	0.01	1.00	1.00
33	272	54.25	1.63	0.01	1.00	1.00
34	276	54.48	1.63	0.01	1.00	1.00
35	279	54.70	1.63	0.01	1.00	1.00

Table 5.7. Age based data for female Corrib eel applied in GEM

Age	Mean weight per age [g]	Mean length per age [cm]	Nat. Mortality low density [%]	Relative age distribution first year	Relative catchability by age group	Relative proportion of silver eel
0	1	9.17	34.74	1.00	0.00	0.00
1	3	12.71	23.22	0.87	0.00	0.00
2	6	16.9	16.69	0.75	0.00	0.00
3	10	19.33	13.23	0.65	0.00	0.00
4	16	22.42	10.81	0.56	0.00	0.00
5	24	25.38	9.05	0.49	0.00	0.00
6	34	28.20	7.72	0.42	0.00	0.00
7	45	30.91	6.69	0.36	0.02	0.00
8	59	33.49	5.99	0.32	0.05	0.00
9	74	35.96	5.41	0.27	0.17	0.00
10	90	38.32	4.91	0.24	0.41	0.00
11	108	40.57	4.57	0.21	0.66	0.01
12	127	42.73	4.26	0.18	0.84	0.01
13	148	44.79	3.99	0.15	0.94	0.02
14	170	46.77	3.75	0.13	0.98	0.05
15	192	48.65	3.53	0.12	0.99	0.09
16	216	50.45	3.33	0.10	1.00	0.18
17	240	52.17	3.15	0.09	1.00	0.31
18	265	53.82	3.02	0.07	1.00	0.44
19	291	55.40	2.91	0.06	1.00	0.57
20	316	56.90	2.80	0.06	1.00	0.70
21	343	58.34	2.69	0.05	1.00	0.80
22	369	59.72	2.60	0.04	1.00	0.88
23	395	61.03	2.54	0.04	1.00	0.91
24	422	62.29	2.48	0.03	1.00	0.94
25	448	63.49	2.39	0.03	1.00	0.96
26	475	64.64	2.31	0.02	1.00	0.98
27	501	65.74	2.26	0.02	1.00	0.98
28	527	66.79	2.21	0.02	1.00	0.99
29	552	67.79	2.17	0.02	1.00	0.99
30	578	68.75	2.12	0.01	1.00	0.99
31	602	69.67	2.08	0.01	1.00	1.00
32	627	70.55	2.06	0.01	1.00	1.00
33	651	71.38	2.04	0.01	1.00	1.00
34	674	72.19	1.99	0.01	1.00	1.00
35	697	72.95	1.96	0.01	1.00	1.00

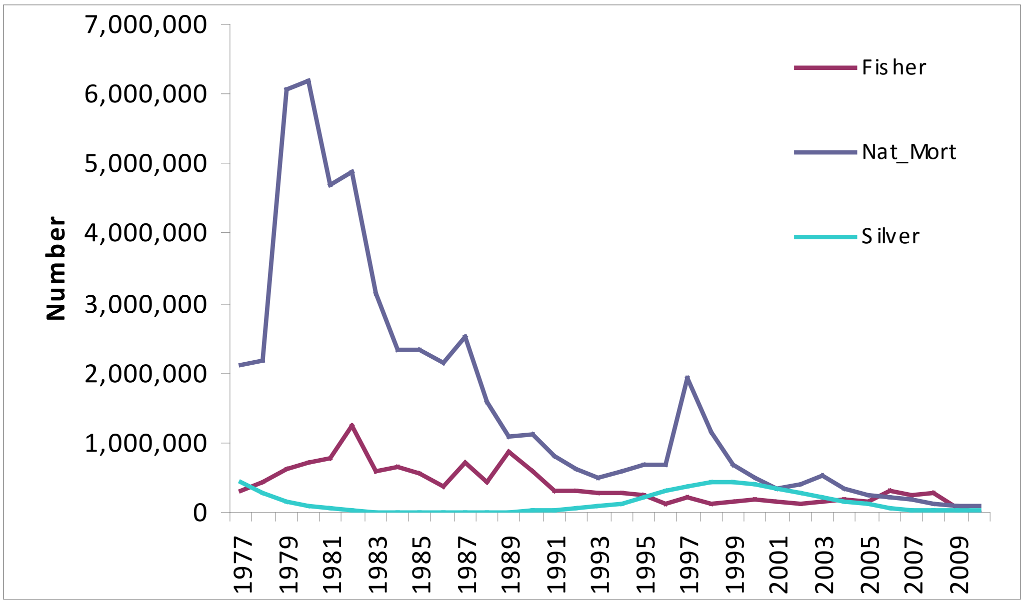
Results

The model parameters were determined by means of the solver in EXCEL to minimize differences between the observed relative age distribution of female yellow eel in 2008 and female silver eel in 2009 and the corresponding estimates of the model. These optimized model parameters are presented in Table 5.8.

Table 5.8. Model parameters of the standard GEM model application to the Corrib data

Proportion of female elvers	0.80
Proportion of female catch	0.90
Maximum proportion of male silver eel	0.25
Maximum proportion of female silver eel	0.47
Nat. mortality	Low density
Factor for the population of the first year	1

The stock dynamics predicted by GEM between 1977 and 2009 are presented in Figure 5.28, and Table 5.9 presents GEM predictions of standing stock, fishery catches and silver eel escapement for the most recent three years of the simulation (2007 to 2009), reported as total numbers and weight, and numbers and weight relative to wetted area of the basin.



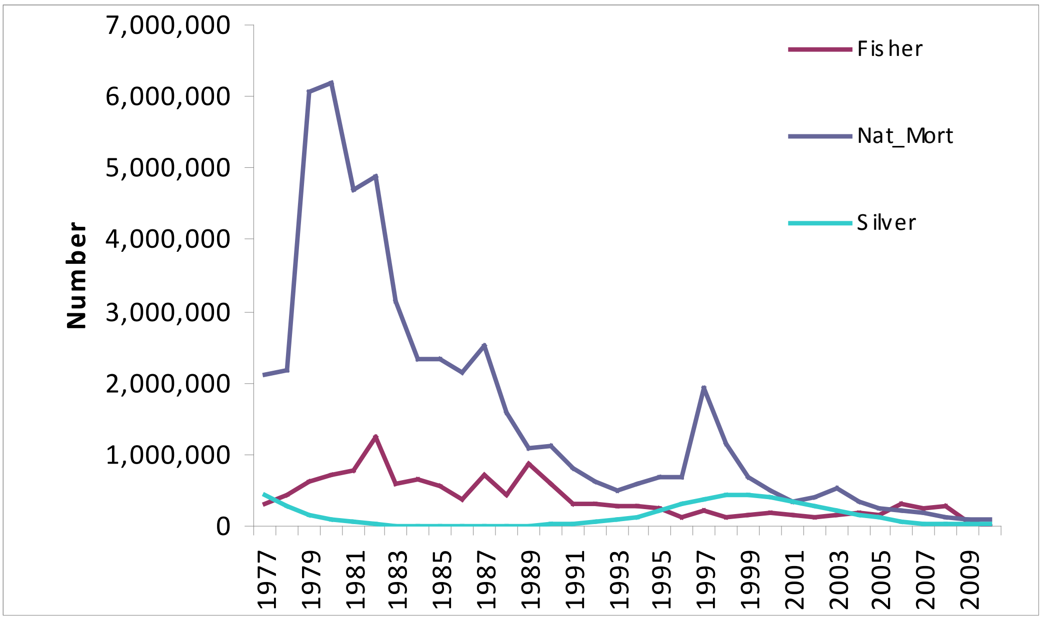


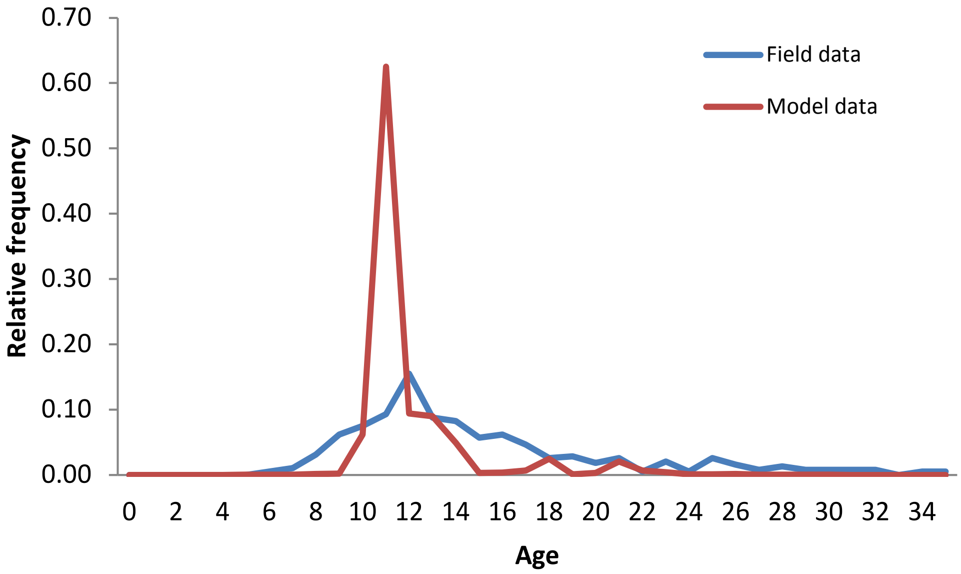
Figure 5.28. Predicted time series of numbers of eels caught by fishermen (Fisher), died of natural mortality (Nat_Mort) and escaping as silver eels (Silver) from 1977 to 2009. The upper chart provides the results for the male eels, and the lower chart provides the results for the female eels.

Table 5.9. Estimates of eel stock parameters in the Corrib catchment between 2007 and 2009

	2007	2008	2009
Weight of total stock in kg per ha	10.0	8.8	7.5
Number of captured silver eel	233421	296967	67269
Number of captured silver eel per ha	8.1	10.3	2.3
Catch of silver eel in kg per ha	1.1	1.3	0.3
Number of escaped silver eel in number	105510	96534	102663
Number of escaped silver eel per ha	3.65	3.34	3.56
Weight of escaped silver eel in kg / ha	0.38	0.20	0.15

The estimated relative age frequencies of female yellow eel in 2008 and female silver eel in 2009 of the model differ from the corresponding field data (Figure 5.29).

The peaks in the relative age distribution of silver eel in 2009 based on the model predictions (Figure 7, lower panel) present the strong year classes, especially the year classes 1994 to 1998 with the very strong year class from 1997. In addition, the year classes from 1979 and older determine the relative age distribution of eel older than 18 years. The slightly stronger year classes from 2002 and 2003 are not yet detected by the fishery by the end of the simulation.



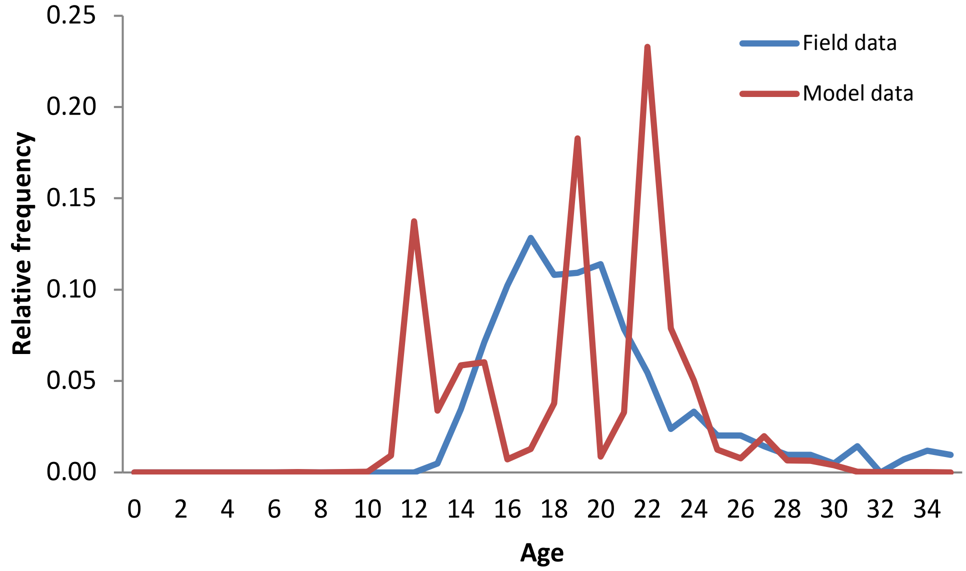


Figure 5.29. Comparison of relative age distributions of field data (observed) and GEM estimates of female yellow eel in 2008 (upper chart) and female silver eel in 2009 (lower chart)

The model estimates of the catch of youngest age groups and the corresponding field data differ in that GEM predicts that the youngest yellow eel caught by the fishery would be age 9, whereas eels as young as 7 years were observed in the fishery catches. In contrast, GEM estimates the youngest captured silver eels in 2009 to be three years younger than those observed in the actual fishery catches.

These differences between the observed and GEM predictions may be the result of various erroneous model assumptions such as the fishing gear selectivity characteristic, the proportion of silver eel by age, the growth curves, etc. However, GEM does not take into account density dependent effects concerning life history processes such as growth, natural mortality, etc. which results in a smoothed relative age frequency in the model predictions.

Mean catches of silver eel per hour in Galway Weir were available from 1998 to 2008. For comparing these field data with estimates of the model, the catch of silver eel in number per ha was estimated from model outputs for the same period. Both estimates show the same temporal development (Figure 5.30).

Figure 5.30. Comparison of the silver eel catch per net from the Galway fishery against the estimated silver eel catch per ha estimated by GEM.

Conclusions

The GEM can be used to describe the eel stock in the Corrib catchment between 2007 and 2009. The estimated stock parameters (see Table 3) were similar to the estimates provided in the Irish Eel Management Plan. However, there are a number of aspects of the model application process and the supply of input data that might be expected to improve the model results.

The model treated the Corrib system as one unit but major differences in eel stock characteristics are observed between the lakes and the rivers, between the lower and upper reaches of Lough Corrib, and between Lough Corrib and Lough Mask further upstream.

The lack of a reliable recruitment figure was a major deficiency in the model application. Thereafter, the main difficulty encountered in applying GEM to the Corrib data set was the uncertainty of applying the input data for the whole time series. Age-based estimates of various parameters could improve the quality of the model estimates.

The sex ratio of the silver eels seems to be causing the model difficulties.

Annual samples for estimating the required model input data like growth function, proportion of silver eel by length / age for both sexes etc improve the quality of the model estimates. The incorporation of density dependent growth and mortality as well as the incorporation of spatial effects might also improve the quality of the model results.

5.5. The Elbe river (Germany)

Contributors: Uwe Brämick (IfB), Erik Fladung ((IfB), Peer Doering-Arjes (IfB)

Overview

The source of the Elbe lies in the Czech Giant Mountains. The mainstem traverses through the Czech Republic and Germany and after 1094 kilometres flows into the North Sea (Figure 5.31). The limit defined in the Water Framework Directive between coastal and transitional waters is to be found at river kilometre 727.7 in Cuxhaven. This is also taken as a basis as the seaward side limit of the Elbe RBD in the context of the Eel Management Plan. Tributaries of the Elbe are concentrated in Germany and the Czech Republic, but also extend for a small part as far as Austria and Poland. More than 99% of the Elbe RBD lies in Germany and the Czech Republic. The shares of Poland and Austria come to less than 1% in total and are located in the source areas of tributaries to the Elbe.

Figure 5.31. Boundaries of the international Elbe RBD (Info-Blatt IKSE No 1 – March 2005)

The Elbe RBD consists of the mainstem, tributaries and standing water connected to the river Elbe and covers a total of 223 540 ha. The areas near the sources of some tributaries are in the salmonid region and do not form part of the eel river basin. The majority of eel habitat in the German part of the RBD covers 201 019 hectares, the majority of which is lakes connected to the river network (136

662 ha). In addition, there are 18 097 ha of flowing waters and 46 260 ha of transitional waters (according to the Water Framework Directive definition).

In accordance with the classification pursuant to Directive 2000/60/EC (EC Water Framework Directive), the mainstem of the Elbe lies in ecoregion 14 (Northern German plain) and is largely to be assigned to Type 20 (sandy river). Up to the Geesthacht damming stage – which from the point of view of the upstream migration of eel is rather undersized, but basically functional – there are no barriers to upstream migration of eel in the mainstem. Within German territory, the mainstem of the Elbe and the important tributaries are the natural region for production of the eel.

Until 1830, the Elbe was structurally in a near-natural state. At the beginning of the upgrading to a waterway in the mid-19th century, the Elbe was then subject to a variety of anthropogenic influences, which together with increasing water pollution led to serious changes in quality and quantity of the fish species community (Bauch, 1958, Petermeier *et al.,* 1996).

In connection with the incipient industrialisation, from 1830, to create the smoothest possible passage, impediments to shipping (e.g. sand and gravel banks) were removed, cut-offs were constructed and dependent waters obstructed. The construction of groynes, bank protection and parallel works, which gathered pace from 1860, served to narrow the channel and to ensure continuous navigability (Kisker, 1926). Extensive construction of dykes resulted in a large-scale reduction in retention areas (Simon, 1994). A large number of dependent waters dried up or were permanently cut off from the river flow as a result of the continuing deepening of the Elbe riverbed caused by the building activities (Knösche, 1998). Whereas the stabilisation of the mainstem is characterised mainly by longitudinal structures, the tributaries, in addition to a large number of channel alignments, were mainly provided with transverse structures. The latter are of considerable importance for the eel population, since they impede or prevent the natural upstream migration of the eel. After 1940, no more major development work took place.

From 1930, with the introduction of industrial and municipal wastewater, water pollution increased, resulting in local fish mortalities and a reduced supply of animal plankton, which also affected the eel stock. Since 1990, a large number of factory closures and increased building of wastewater treatment plants has resulted in an abrupt improvement in the water quality of the Elbe (IKSE, 2000). This has resulted in the continuing recovery in of many typical river fish stocks (Fladung, 2002). At present, the ecological status of the fish fauna in the mainstem of the Elbe is classified as good (ARGE Elbe, 2008).

Stock status

Systematic monitoring of the trend in the eel population in the Elbe RBD has not been carried out to date. According to the yield statistics from commercial fishing, the average catch of eels for consumption in the Elbe RBD has fallen since 1985 from over 500 tonnes to less than 200 tonnes today (Figure 16). As there has been a consistent demand for eel in Germany, we assume that there has been a constant fishing effort. Therefore, the trend shown by Figure 5.32 is very likely to also reflect a reduction in the eel population. On this basis, it is estimated that the current size of the eel population in the Elbe RBD is well below the level of previous decades.

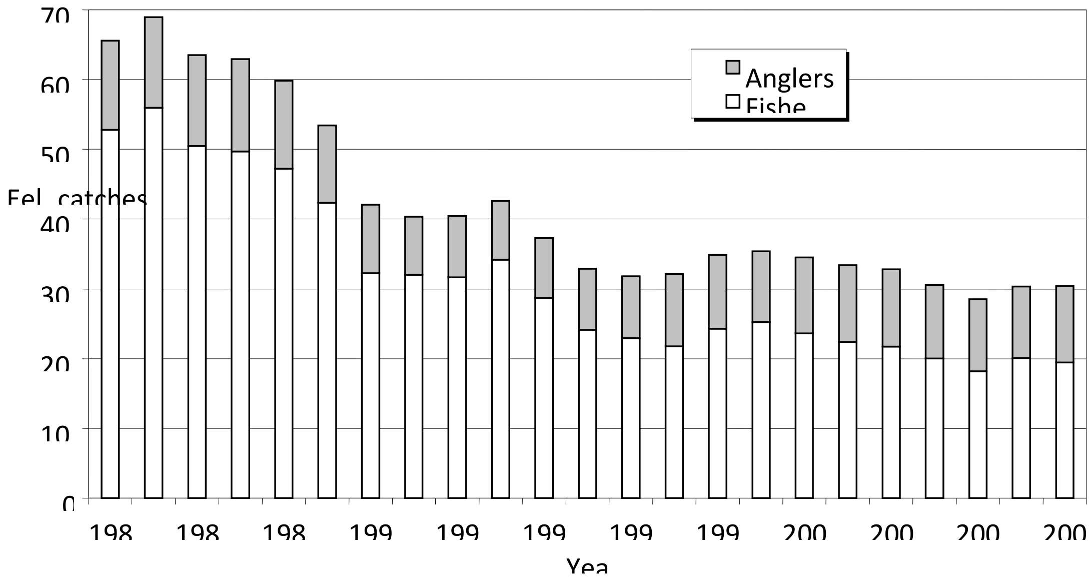


Figure 5.32. Trend in catches in the Elbe RBD during the period 1985-2007 (IfB data)

Directly measured data on the escapement of silver eel and anthropogenic mortalities for the period prior to 1980 are not available. Instead, the reference value was modelled on the basis of data on the current natural upstream migration of eel in the central part of the Elbe (280 km above the estuary (Brämick *et al.*, 2006)), on the trend in the occurrence of ascending eel at river estuaries along the European Atlantic coast (ICES, 2008) and on the natural mortality in European waters (Dekker 1999). For details see the German Eel Model (GEM) in this report. The trend in the recruitment of glass eels in Europe (ICES, 2008) remained relatively stable during the period 1950-1980 and then declined sharply. The fundamental input parameters in the Elbe eel stock model are natural upstream migration, restocking, natural mortality, and mortality caused by cormorants, fishers, anglers, hydroelectric installations.

An annual silver eel escapement of 1,381 tonnes (6.9 kg/hectare) for the period 1975-1980 has been estimated as the reference value.

The same model was used to estimate the present escapement of silver eels, but also accounting for restocking inputs and current anthropogenic mortality factors.

Impacts to eel production

Fisheries

In general, the eel is the most economically important fish species for most of the lake and river fisheries in North Germany and therefore also in the Elbe RBD (Knösche, 2003). According to a study by IfB Potsdam-Sacrow, the eel, for example at the Brandenburg commercial fishery accounts for a 56% share of the market output and is therefore the decisive factor from a business point of view in the yield from catches (Knösche *et al*., 2005). Compared to other fish species, it produces by far the highest selling price and records sustained high demand.

There are about 400 fisheries for eel as a full-time or part-time activity on the Elbe and its tributaries. The number of active anglers within the Elbe RBD can be estimated on the basis of the number of valid fishing permits at approximately 344 000. The focus is on catching yellow and silver eels weighing between 300 and 800 g for marketing or own consumption. The current minimum sizes specific to each Land of between 28 and 45 cm set a lower limit for the eel year groups affected by fishing mortality.

The main fishing gear of commercial fishers for eel are fyke nets (about 11000) which are used in various constructions and sizes from the “Stromreuse” to the “Bunge”. To a minor extent, dip nets (31) and schokkers are also used for targeted eel fishing. Furthermore, in smaller tributaries, stationary traps (38) are also used. Electrical fishing requires a permit and is practised in only a few fishing undertakings (24). Anglers are generally authorised to catch eels with handlines and bobbing.

The catches of commercial fishers have declined from 500 to 550 tonnes in the 1980s to the current 195 tonnes and therefore to 30-40% of the historic catches. Since 1985, the catches by anglers have declined but currently amount to approximately 0.5 kg/hectare per year. The annual catch by anglers using pole-and-line fishing in 2006 amounted in Saxony to 6.1 tonnes of eel (Füllner *et al*., 2008).

A monetary assessment of the significance of eel-fishing for anglers is difficult, however, since the use of the waters for angling is not undertaken from the economic point of view. However, reference is made to the fact that the restocking with eel in the Elbe RBD, which represents a considerable financial expense, is undertaken to a considerable extent with own funds from both commercial and pole-and-line fishing.

Predation

In connection with the sharp increase in the cormorant population since 1990 (1000 to 4500 breeding pairs), the quantities of eel caught by cormorants have risen considerably and are currently estimated for the Elbe RBD at 100 to 130 tonnes per year. For details see also the model (GEM).

Obstacles to migration

A large number of longitudinal and transverse structures have been erected in the Elbe RBD over many centuries. The first dykes in the Elbe were already constructed in the year 1100. Mill dams were documented for the Havel, the largest tributary of the Elbe in 1232. By 1375, a large number of weirs had already been created to control the banked-up water level (Brämick *et al.*, 1998).

The water bodies of Elbe RBD have been classified according to mortality rates caused by hydroelectric installations and summarised in Table 5.10.

Table 5.10. Proportions of the Elbe RBD with different mortality rates caused by hydroelectric installations and cooling-water intakes (reference year 2007)

Mortality rate (%)	Sub-basin area (hectares)
0	129 855
10	559
20	2 307
30	4 999
40	18 403
50	1 907
60	71
70	2 377
80	12 914
90	26 930
100	695
Total	201 019

The present situation in the Elbe RBD is characterised by a largely barrier-free mainstem for fish migration – with the exception of the weir in Geesthacht, the only barrier in German territory, and 19 cooling-water intake points. Significant cooling water intake occurs in the lower tidal Elbe by heavy industry. Estimates of the total quantity of injured fish exist only for the Brunsbüttel power station. Multi-annual investigations in the cooling-water intake there show that up to 6 tonnes of eel per year are destroyed (Rauck, 1980, Möller *et al.*, 1991). Since the 6 cooling-water intakes in the lower reaches of the mainstem must be passed by all eel moving upstream and downstream, in this river section alone a not insignificant injury potential of 3% of the escaping silver eels per year must be assumed.

Potential barriers to the upstream and downstream migration of eels in the Elbe RBD are concentrated in the tributaries. So far a total of 4897 transverse structures and cooling-water intakes have been recorded. Quantification of their effects on the eel stock of the Elbe RBD is currently possible only by means of a rough estimate. Since about the last 100 years, restocking with eels has been undertaken in many sub-basins of the Elbe in order to maintain natural habitats for distribution and growth for the eel population where these have been cut off by obstructions and thereby compensate for the effects of barriers to upstream migration at least in part. At the same time, the escapement of silver eel from sub-basins is considerably impeded or made completely impossible by several successive hydroelectric installations. In contrast to this, the only weir in the mainstem of the Elbe at Geesthacht impedes the upstream and downstream migration to a distinctly lesser extent.

An average overall mortality rate from hydroelectric installations and cooling-water intakes for silver eel during escapement currently comes to 24 % (Table 3).

Pollution

The effects of contaminants on the number of recruitments per escaping silver eel from the Elbe RBD (Recruit per Spawner, RSP) cannot be assessed at present.

Diseases and parasites

There is so far no indication that eels in the Elbe RBD are infected to a significant effect by herpes viruses (HVA) or rhabdoviruses (EVEX) or that restocking with eels from aquaculture production could contribute to an increase in the rate of infection with these viruses already existing in our waters.

The eel stock in the Elbe RBD is subject to a rather lesser rate and intensity of infestation by the parasite *A. crassus* compared to other German and European waters. The average infestation intensity lies between 3 and 6 nematodes per eel, the average rate of infestation between 40 and 70%.

Restocking

Restocking statistics of fishers, anglers and fisheries undertakings from the period 1985-2007 were evaluated. Original or extrapolated restocking data were available for most of the river basin; for 25% of the water areas, data missing in part had to be supplemented by estimates. On the basis of the restocking quantities, in most cases available in the form of biomass (kg), the number of restocked eels was calculated, broken down according to 3 sizes of restocking fish on the basis of known or estimated average unit biomass:

- glass eel (0 to 2 g individual weight, 0 to11.5 cm total length);
- advanced farm eel (2 to 17 g individual weight, 11.5 to 22.5 cm total length);
- bootlace (17 to 50 g individual weight, 22.5 to 32 cm total length).

A graphical account of the trend in the eel restocking quantities over the period 1985-2007 is given in Figures 5.33 and 5.34.

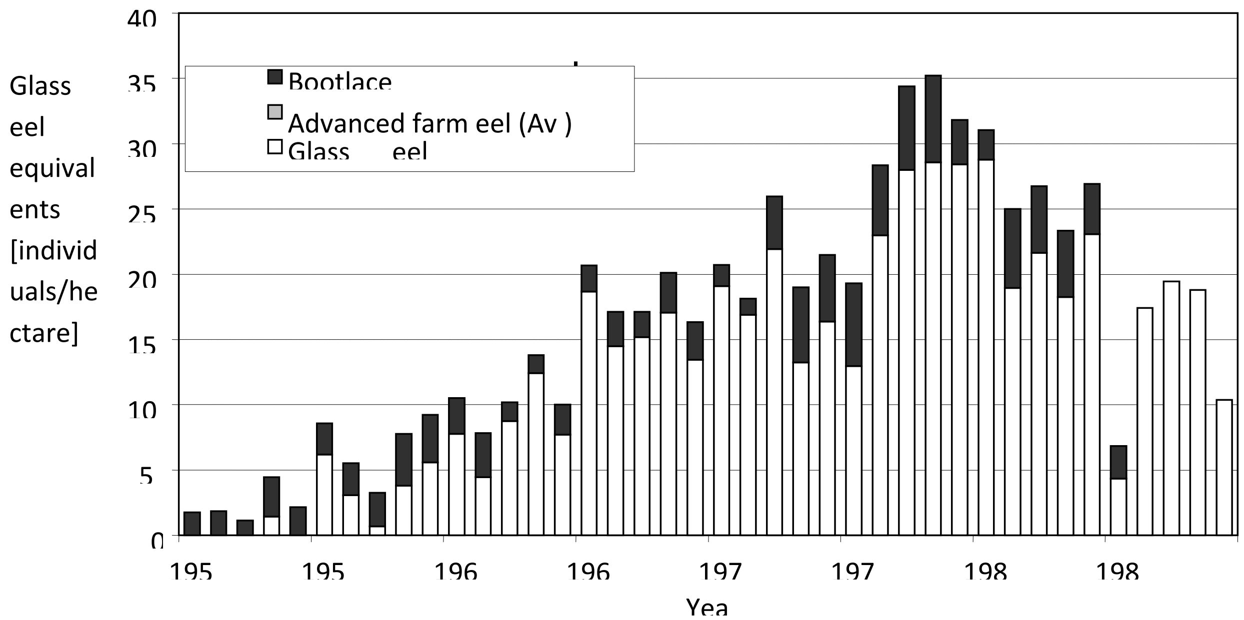


Figure 5.33. Eel restocking in the former GDR 1950-1989 (in glass eel equivalents per hectare, conversion 1 Av = 3.0 Ao equivalents, 1 As = 4.5 Ao equivalents)

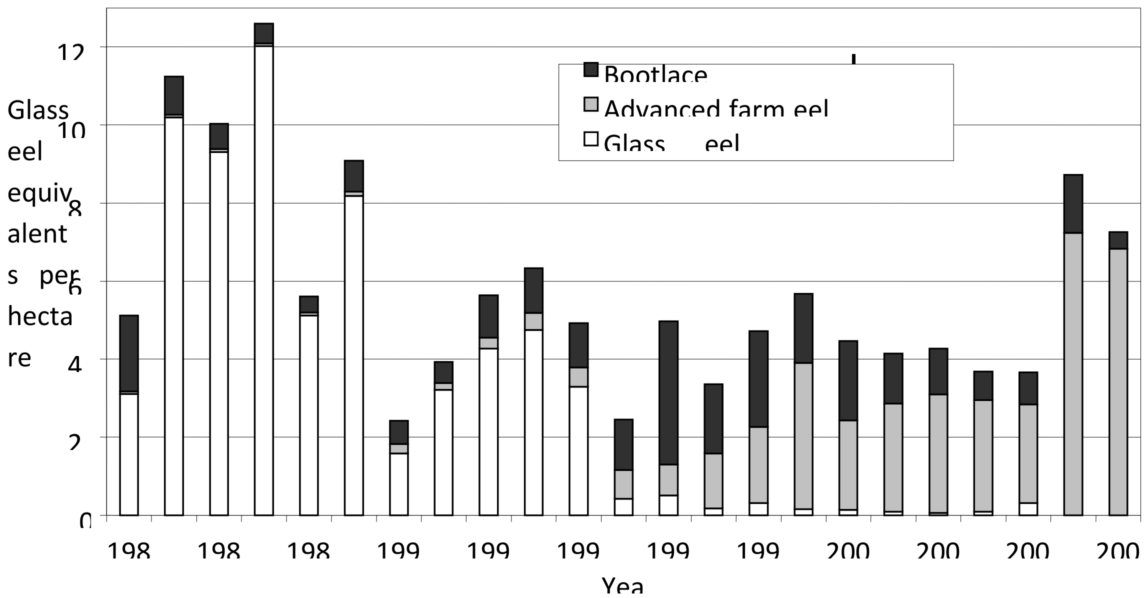


Figure 5.34. Eel restocking in the Elbe RBD 1985-2007 (in glass eel equivalents per hectare, conversion 1 Av = 3.0 Ao equivalents, 1 As = 4.5 Ao equivalents)

Data for POSE

The River Elbe RBD of is considered as data rich for POSE. For the entire eel catchment area data or qualified estimates are available for the period from 1985 until 2007 for the following parameters:

- Water surface area
- Growth, age, age-length-weight-relation
- Recruitment (numbers of stocked eel and recent numbers of immigrating eel)
- Commercial and recreational catch data
- Eel mortality by cormorants
- Eel mortality by power plants

Moreover, data on stocking and commercial catches exist for parts of the RBD back to 1950. Projects to estimate immigrating and emigrating eel numbers are ongoing.

As the German Eel Model (GEM) was developed with and for the River Elbe and its eel stock, the application of GEM to this data set is documented in the detailed description of the development of GEM (see Annex A3).

5.6. Application of DemCam to the Elbe river data set

Model simulations were run from 1900 to 2007.

Input data

Life history parameters

The a and b constants for the allometric length (cm) /weight (g) relationship were set as $a = 8.34 \times 10^{-4}$, $b = 3.17$.

The average age of male and female silver eels were set as 6.7 and 13.6 years, respectively. The average lengths of male and female silver eels were set as 401 and 641 mm, respectively.

Recruitment

In the absence of survey data, we assumed that natural elver recruitment prior to 1985 was constant at a level of 33.5 t per annum, which is equivalent to about 500 individuals per hectare. From 1985 to 2007, recruitment was considered to be primarily by stocking of eel at ages 1 to 4 years, and therefore a time series of data were available from stocking surveys.

The average weight of glass eel recruits was set as 0.33 g, as for other populations across Europe, and the average weight of an elver was set at 7 g.

The survival rate of glass eel to elvers was set at 25% for the first year.

Carrying capacity

The carrying capacity for elvers was set at 1000 elvers per hectare, after a series of trials revealed this level to produce sex ratios values close to those observed in the eel catches.

Environment

The simulation was run on a single compartment with a wetted area of 201,019 hectares, and assuming that the entire wetted area was a homogenous habitat. The annual average water temperature was set at 11.7 C.

Impacts

The area has an established history of fishing for eel. Fishing effort was assumed to be constant throughout the time series (1990 to 2007). Local data suggested catches were about 1 kg/ha. DemCam requires fishing to be input as an effort described as numbers of gear per day, and with associated catchability coefficients for yellow and silver eels. Therefore, we set an annual fishing effort equal to 150,000 gear*day, which corresponds approximately to the effort executed by 100 fishermen, each using 5 gears and operating 300 days per year.

In the absence of data for the Elbe, the catchability of silver eel was set at 0.1, and that of yellow eel was set at 0.01, based on estimates for other environments (Bevacqua, unpublished data).

Results

The results of the DemCam simulations are presented in Figures 5.35 to 5.38, which illustrate the predicted time series of glass eel recruitment, silver eel production, yellow eel standing stock and catches of yellow and silver eel.

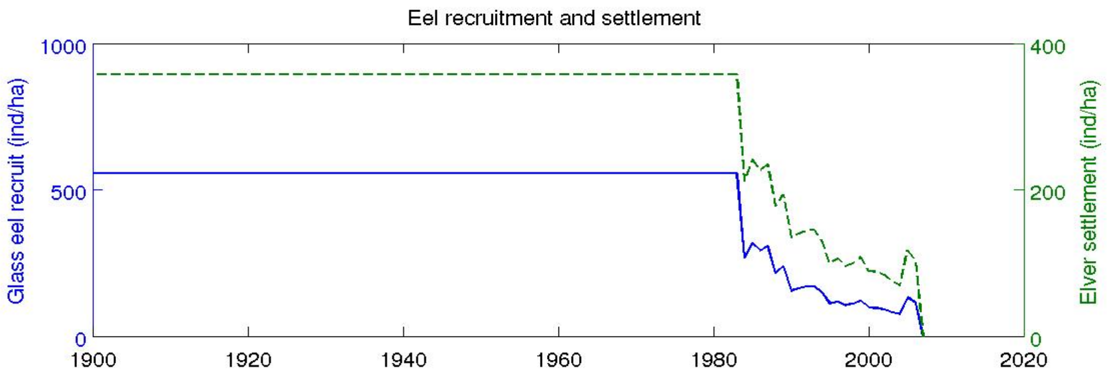


Figure 5.35. DemCam simulation of glass eel recruitment and elver settlement for the Elbe. Note the different scales on the two y-axes.

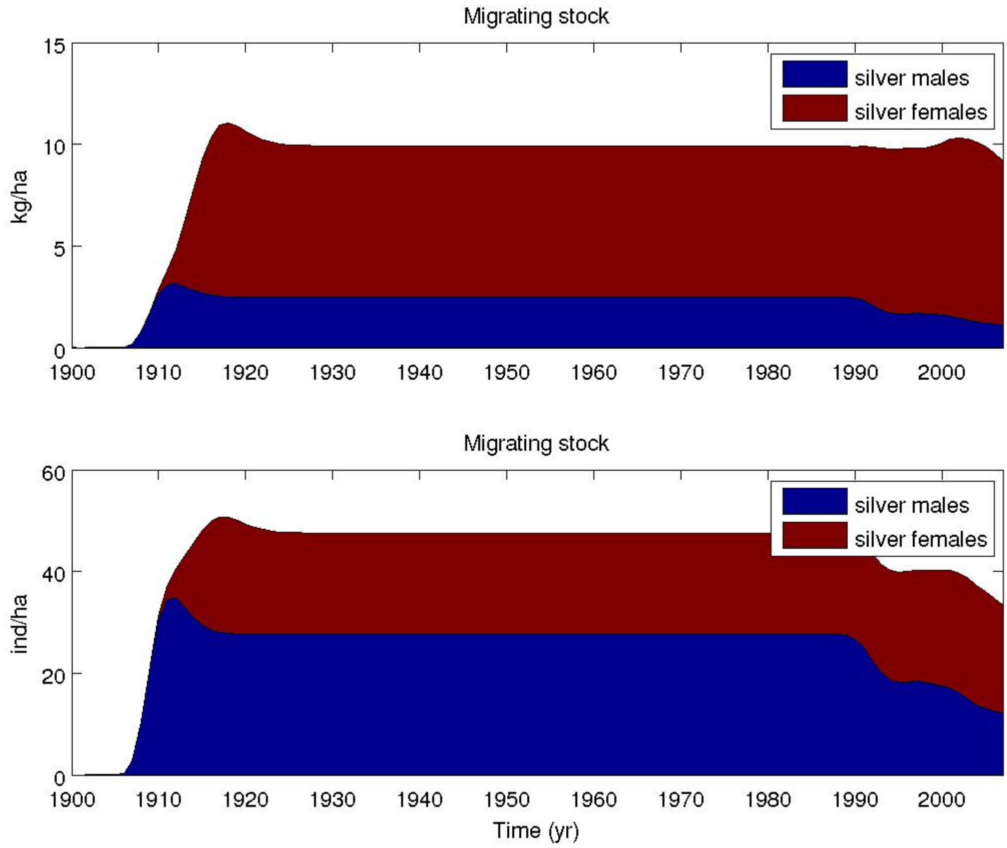


Figure 5.36. DemCam simulation of silver eel production in the Elbe. The top chart shows silver eel production for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

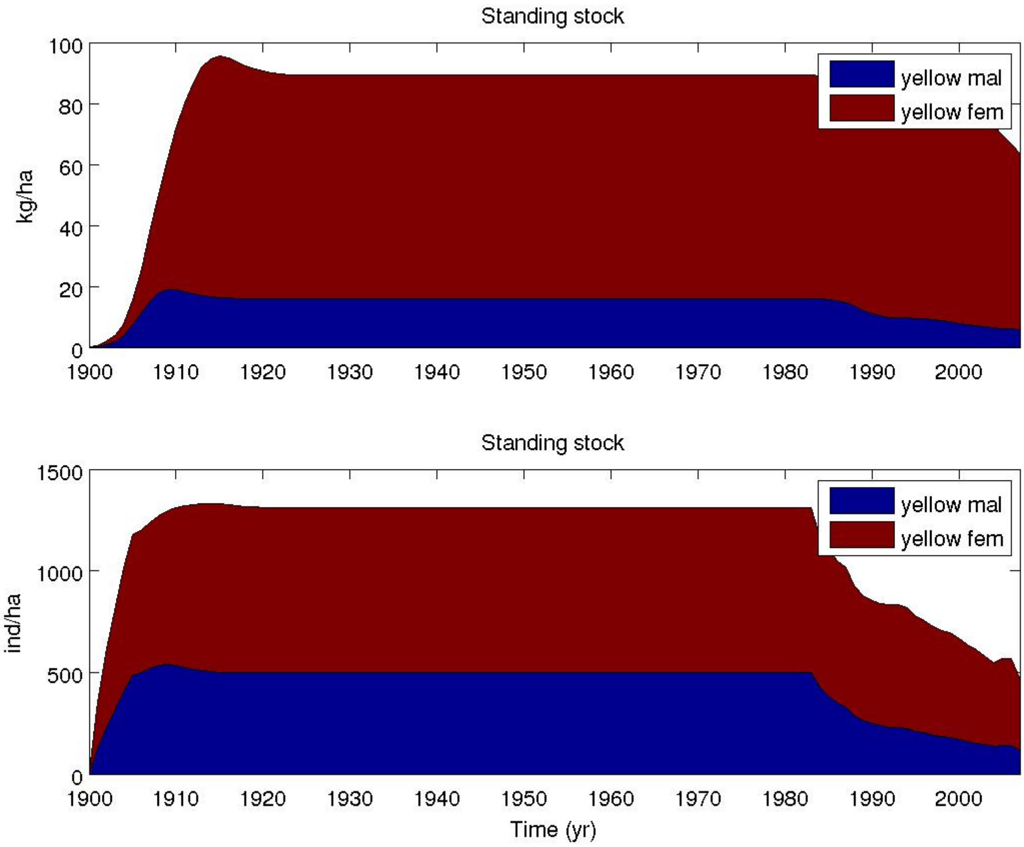


Figure 5.37. DemCam simulation of standing stock of yellow eel in the Elbe. The top chart shows yellow eel stock for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

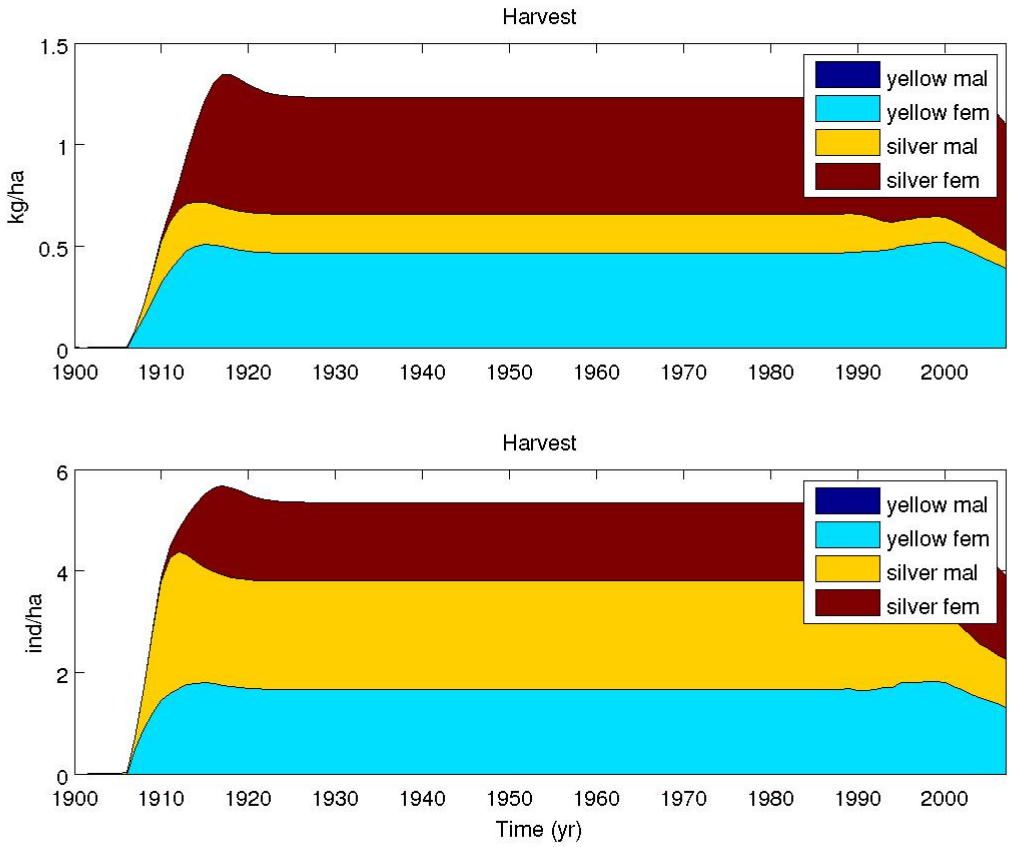


Figure 5.38. DemCam predictions of the catch of male and female yellow and silver eels during the time series, expressed as kg per hectare in the upper chart, and individuals per hectare in the lower chart.

Given the extensive wetted area, and the relatively low recruitment level supported by stocking in recent years, it is not surprising that DemCam predicted that stock dominated by female eels. The decrease in silver eel production as a consequence of the drop in natural recruitment after 1985 became evident only in recent years, probably because of the initial compensatory effects of reduced density dependent influences on mortality, and changes to the conditions determining gender 'choice' resulting in higher proportion of female silver eels.

Considerations for model application

The process of 'tuning' DemCam to the eel data from the Elbe revealed that the simulations were sensitive to the parameterisation of recruitment and the selection of an appropriate carrying capacity.

Compared with the results given by GEM the DemCam-model estimated production and standing stock for current years (2007) about 4-5 times higher than GEM, meanwhile estimated catches were lower than in GEM (see following table).

Model output data of DEMCAM and GEM concerning Elbe-data (base year 2007)

	DemCam	GEM	factor
Silver eel production (kg/ha)	~ 8 (25 % mortality subtracted)	1.6	5.0
Silver eel production (ind/ha)	~ 26 (25 % mortality subtracted)	5.4	4.8
Standing stock (kg/ha)	~ 62	14	4.4
Standing stock (ind/ha)	~ 480	174	2.8
Catch (kg/ha)	~ 1.1	1.5	0.7
Catch (ind/ha)	~ 3.8	7.3	0.5

Differences concerning catch are not substantial, the more so as estimated number of individuals caught depends on different sizes of selected individuals, length-weight relationships, etc.

Differences in standing stock and production are considerably large. First results of silver eel monitoring in the Elbe estimate 2 – 6 individuals/ha and are therefore within the range of the GEM modelling results (Simon & Fladung, 2009). Estimates from other European River systems for emigrating silver eel are 0.3 – 10.0 kg/ha; most are in the range of 0.5 – 5.0 kg/ha (ICES, 2010).

In the light of a low natural eel immigration in the Elbe and a stock mainly recruited by stocking, it seems DemCam is overestimating eel survival und thereby silver eel escapement.

5.7. The lagoons of Sardinia (Italy)

Sardinia (Figure 5.39) is the second largest island in the Mediterranean (after Sicily). It is an autonomous region of Italy and constituted a single eel management unit (EMU) in the Italian national plan for eel recovery.

Figure 5.39. Surface water and eel habitat of Sardinia

Eel habitat in Sardinia mainly consists of coastal lagoons. In fact, although eels are present in internal waters, they are very scarce when compared to that of coastal lagoons.

Sardinian rivers are scarce (ca. 3000 km) and characterized by droughts during summer and autumn that impairs the presence of fish species. Moreover, most of the rivers have been obstructed by the construction of artificial catchments. Spano (1956) estimated an annual fish production of 1.8 t from the entire river system, but eels represented an unknown fraction.

Sardinia has only one small natural lake (Baratz, 40 ha) and other artificial lakes (8,300) do not represent suitable habitat for eels given the presence of dykes that impair migration. Stocking of eel to support fisheries was never conducted in artificial lakes because of the possibility for eels to escape downstream, thus making stocking economically unprofitable.

Almost the all eel production of Sardinia comes from the 77 lagoons present in the island, corresponding to 15,000 ha. The biggest lagoon (Cabras) is 2,228 ha and 19 are larger than 100 ha.

Actual lagoon area is much less than that at the beginning of the 20th century given reclamation programs conducted during years 1920-1940.

Sardinian lagoons are shallow (ca. 1 m) and salinity is very high due to limited freshwater discharge and elevated exchange with marine water. Salinity level increased in the last few decades due to modification of river courses and widening of outlets to prevent anoxic conditions.

Eel fishery

The eel fishery occurs in 32 lagoons by means of 'lavorieri' targeting silver eels and fyke nets targeting both yellow and silver eels. Fish catches in 21 lagoons registered an annual production of around 1000 t during 1991-1993. Eels constituted 14.2% of the catch, suggesting an eel production of 142 t, equivalent to an annual production of 22 kg per hectare. Documents of the 1950s indicate that production was even higher at times where salinity level was lower and there were fewer piscivorous birds.

Fishing in the other lagoons is not profitable due to their reduced dimensions. An estimate of Cannas *et al.* (2008) indicates an annual production of around 20 tons of eels from those lagoons where fishing is not profitable.

Studies on eel in Sardinia

Most of studies on eel in Sardinia belong to grey literature with the exception of some works conducted in the 1980s (see Rossi and Cannas, 1984). Rossi and Cannas (1984) observed a mean age of 5 and 6 years for silver eel, respectively males and females. However, the study was conducted in an oligotrophic lagoon (Porto Pino) and it is likely that growth is faster in other lagoons (A. Cannas, pers. comm.).

Data for POSE

The Sardinia EMU offers POSE an opportunity to test the application of the models to eel habitats dominated by lagoons, and with limited (poor) data on eel. In terms of eel life history, some data are available on the size and age distributions of recruiting eels but there are no enumerations of recruitment. Growth rates, including a length-weight relationship, are available for recruits, yellow and silver eels, and some data are available on sex ratios and the length distribution of silver eels. Natural mortality rates are available, including rates at length and age, and information on possible effects of density on mortality and sex differentiation.

In terms of potential impacts on eel production, commercial fisheries data are available, as are some data for predation rates and amounts of potential habitat that has been lost to the eel.

Finally, GIS and mapping data are available describing the habitat available to the eels.

5.8. Application of DemCam to the data from Sardinia

Model simulations were run from 1900 to 2010.

Input data

Life history parameters

The a and b constants for the allometric length (cm) /weight (g) relationship were set as a = 8.34x10^{-4}, b = 3.17.

The average age of male and female silver eels were set as 3.2 and 6.4 years, respectively. The average lengths of male and female silver eels were set as 340 and 580 mm, respectively.

Recruitment

In the absence of survey data, we assumed a time series of glass eel recruitment equivalent to the DEA index of the European eel stock (see ICES, 2010). We assumed a constant recruitment prior to 1950 of 100 t per annum, which is equivalent to about 10,000 individuals per hectare. Thereafter, we assumed that the recruitment followed the trend of the GEA time series.

The average weight of glass eel recruits was set as 0.33 g, as for other populations across Europe, and the average weight of an elver was set at 7 g.

Carrying capacity

The carrying capacity for elvers was set at 2000 elvers per hectare, which is a level typical of a highly productive environment (Bevacqua, unpublished data).

Environment

The habitat available to eel in Sardinia is mainly the lagoons. Until 1975, these lagoons provided about 30,000 hectares of suitable habitat, but this has since dropped to 15,000 hectares due to land reclamation. The simulation was run assuming that this entire wetted area was a single compartment with a homogeneous habitat. The annual average water temperature was set at 15.0 °C.

Impacts

Although fishing was historically present in the system, we set a null fishing effort until 1950 in order to estimate pristine spawning escapement in absence of fisheries. From 1950 onwards, we assumed a fishing effort equal to 210,000 gear*day which roughly corresponds to an effort executed by 100 fishermen, each using 5 fishing gears and operating 300 days per year. We supposed that such effort halved after 1975 as a consequence of the habitat reduction.

The selectivity of fishing gears was set according to a stretched mesh of 18 mm.

In the absence of data for the Sardinian lagoons, the catchability of silver eel was set at 0.1, and that of yellow eel was set at 0.01, based on estimates for other environments (Bevacqua, unpublished data).

Results

The results of the DemCam simulations are presented in Figures 5.40 to 5.43, which illustrate the predicted time series of glass eel recruitment, silver eel production, yellow eel standing stock and catches of yellow and silver eel.

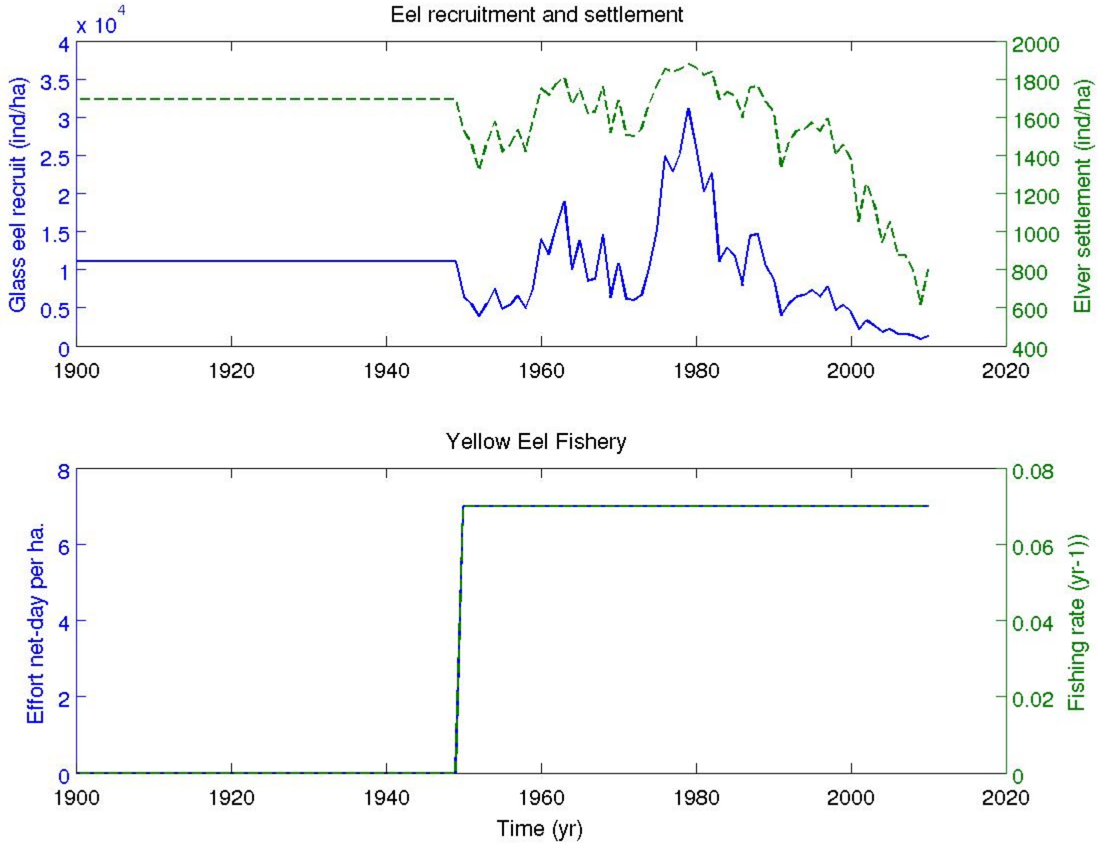


Figure 5.40. DemCam simulation of glass eel recruitment and elver settlement for the Sardinian lagoons (upper chart), and the evolution of fishing pressure (lower chart). Note the different scales on the two y-axes of both charts.

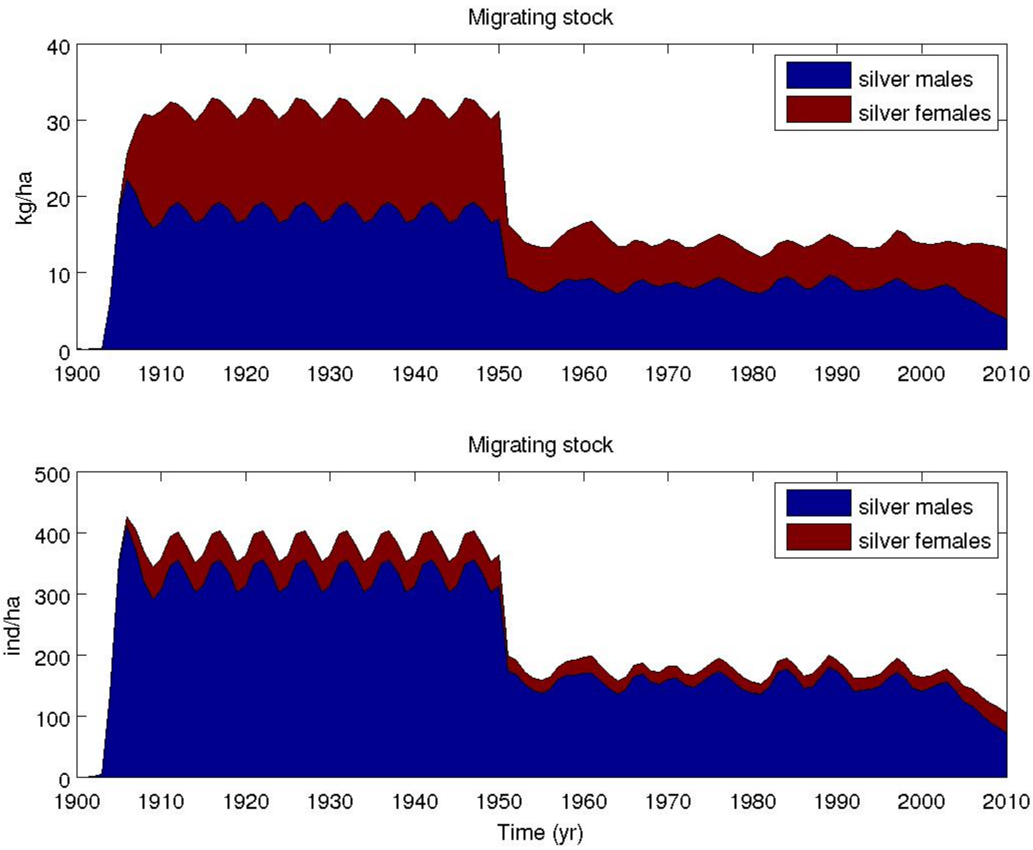


Figure 5.41. DemCam simulation of silver eel production in the Sardinian lagoons. The top chart shows silver eel production for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

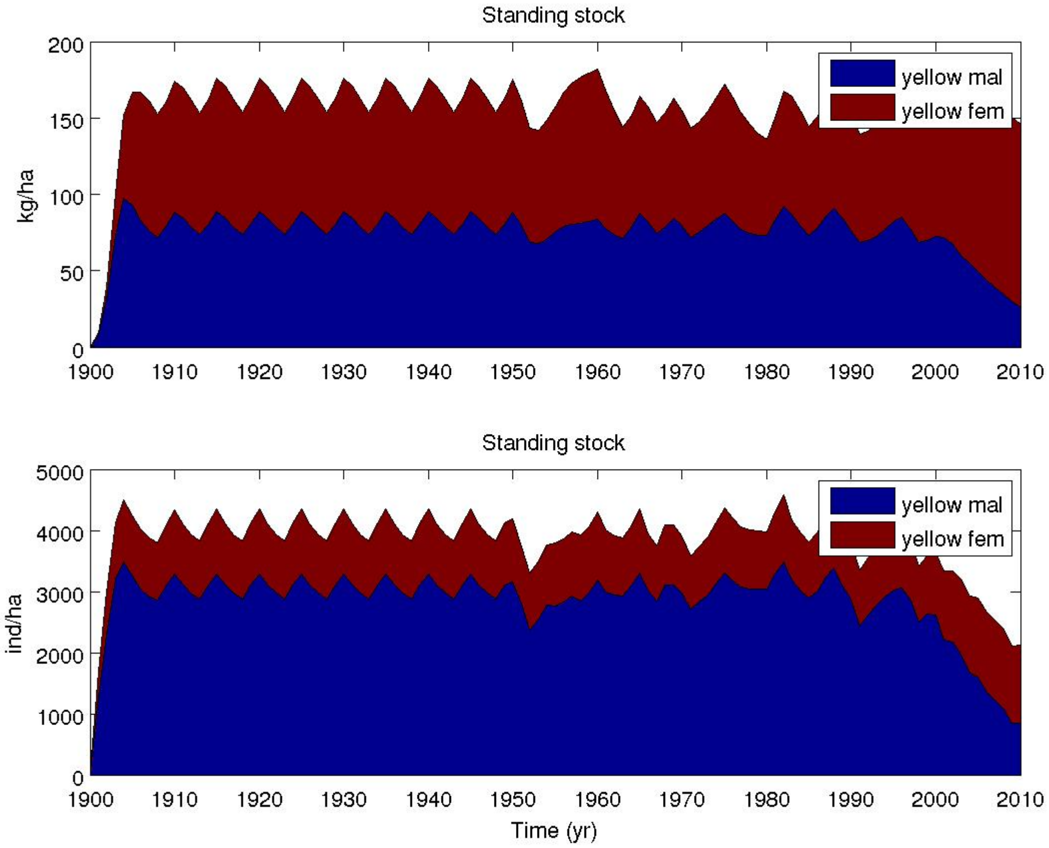


Figure 5.42. DemCam simulation of standing stock of yellow eel in the Sardinian lagoons. The top chart shows yellow eel stock for males (blue) and females (red) measured in kg per hectare, while the lower chart shows the same time series but expressed as individuals per hectare.

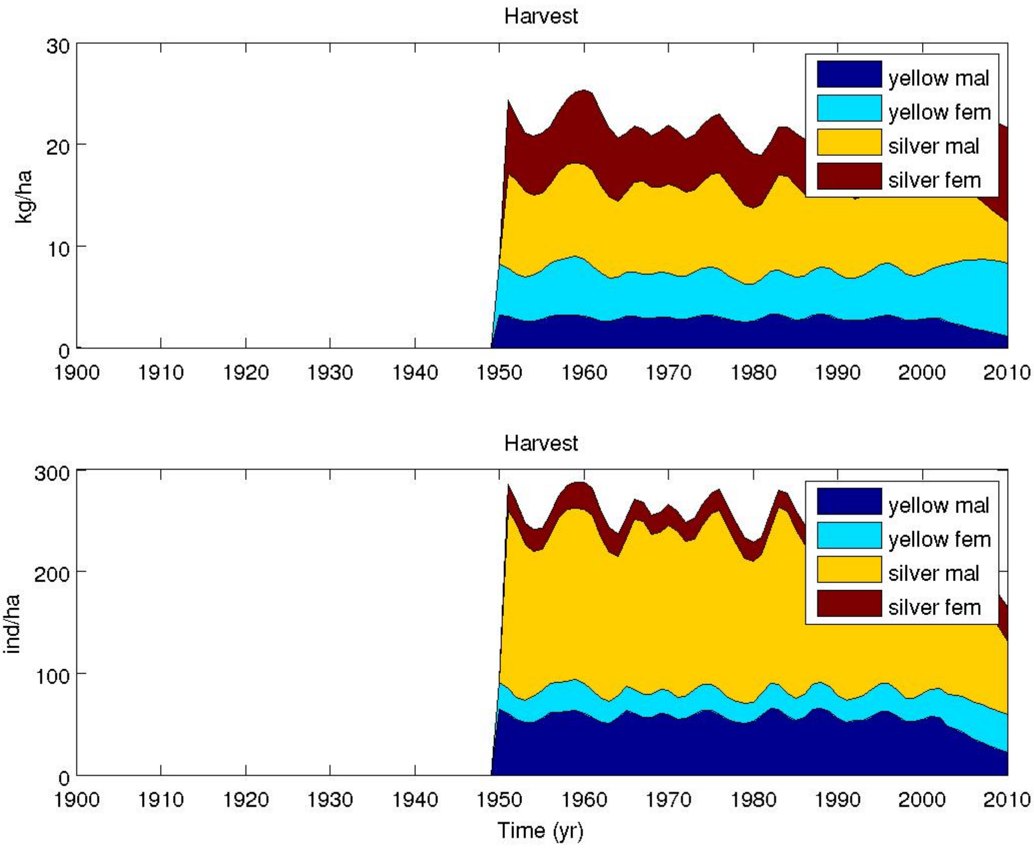


Figure 5.43. DemCam predictions of the catch of male and female yellow and silver eels during the time series, expressed as kg per hectare in the upper chart, and individuals per hectare in the lower chart.

The DemCam simulation provided estimates of pristine spawning escapement of 30 kg/ha (Figure 5.41) which are in accordance with existing literature on lagoon productions. However, since 1975, the overall production of Sardinia was halved due to habitat loss. Migrating stock was almost halved by the presence of fisheries which mainly impair production of silver eel females which need more years to grow until maturation size and are hence more susceptible to eel fishery. Catches were estimated around 20 kg/ha and did not significantly decrease when considered per hectare, given the fact that habitat decreased and density descendent effect might have initially masked the effect of recruitment drop.

Considerations of the model application

The process of 'tuning' DemCam to the eel data from the Sardinian lagoons revealed that the simulations were sensitive to the parameterisation of recruitment and the selection of an appropriate carrying capacity.

5.9. The Brittany EMU (France)

Contributors: Cedric Briand, Laurent Beaulaton

France overview

The French eel management plan is separated into 10 EMU (Figure 5.44), from which two have been included in the POSE project (Brittany and Rhone – Mediterranean coast). Since national surveys provide the basic data available in every French EMU, they are described here.

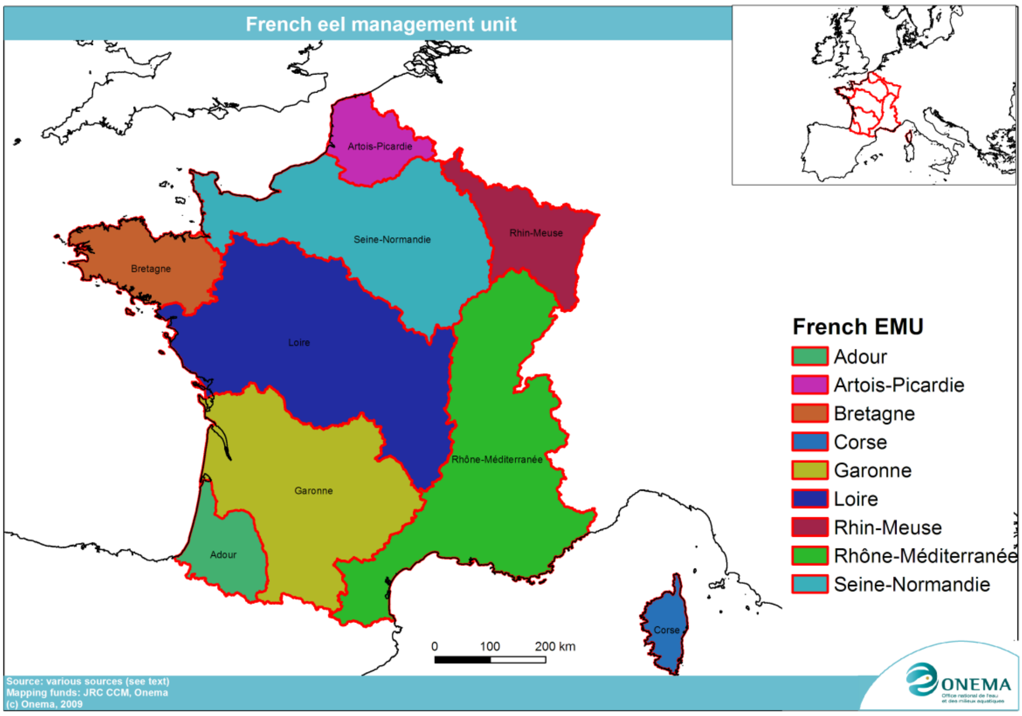


Figure 5.44. Eel Management Units in France

National surveys: Electrofishing

France has employed a structured, multi-species survey network since 1995 called RHP (Réseau Hydrobiologique et Piscicole), comprising about 700 fishing points sampled every year. Since the entry into force of the Water Framework Directive (WFD), this network has increased to cover 1500 points (RCS – Réseau de Contrôle de surveillance) fished alternatively every two years. The results of these electrofishing surveys, as well as those from various studies operated by Onema (formerly CSP) are gathered into a national database (BDMAP).

The BDMAP contains data from more than 22,000 electrofishing sampling occasions from the late 1970s to date.

National surveys: Fishery survey

Eel fisheries in France and their monitoring are extensively described in Beaulaton *et al.* (2009). Basically, marine and river (freshwater) fishers, fishing respectively downstream and upstream of the saline limit, and professional, amateur fishers with gears and anglers are separated because they have different regulation and monitoring systems.

For the POSE project, data on catches of glass eel, yellow and silver eels are available from the SNPE (*Suivi National de la Pêche aux Engins*) survey, operated by Onema, are available at a sufficient scale to preserve the anonymity of fishers. SNPE is the official declaration system for both professionals and amateurs with gears in public freshwater rivers.

Obstacles census

All obstacles to river flow (including obstructions to fish migration) in France are now gathered in a database called ROE (*Référentiel National des Obstacles à l'Ecoulement*). The first release (April 2010) is available at http://carmen.carmencarto.fr/66/ROE.map (Figure 5.45).

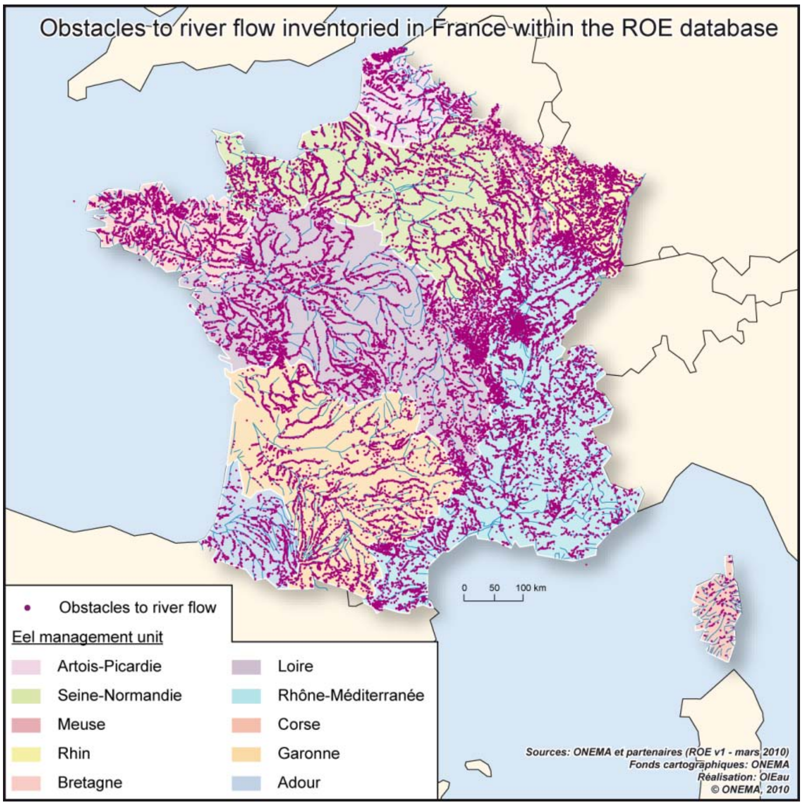


Figure 5.45. Obstacles to river flow inventoried in France (ROE database)

Glass eel status

Most of glass eel time series in France come from fisheries surveys. An analysis of these time series is given in Beaulaton *et al.* (2009). They clearly show a decrease in glass eel recruitment in France since 1980, with an annual rate of decrease of about 8%.

Yellow eel status

An analysis of BDMAP data was conducted in the French Eel Management Plan to explore trends in the yellow eel stock. 150 sampling points, with more than 3 sampling occasions between 1977 and 2007 each recording eels, were selected for trend analysis. The analysis (Figure 5.46) clearly shows a decreasing trend of about 3.4% per year since 1983. This is despite the fact that the selected sampling points would have been expected to be most resilient to stock declines because they are near the sea or in main rivers.

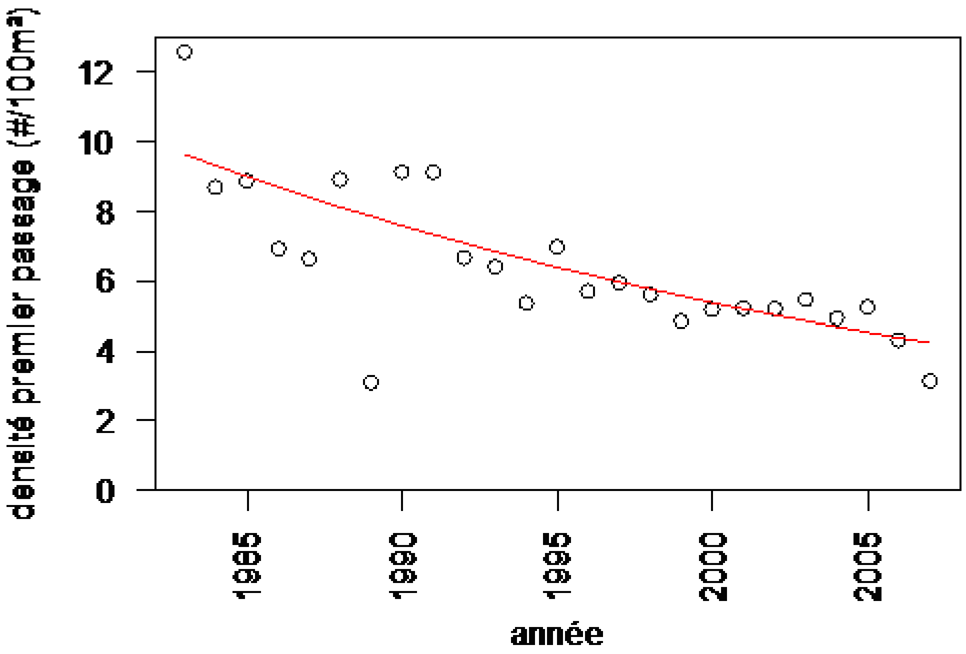


Figure 5.46. Least square mean density (point) and density trend (line) of yellow eels from 150 survey locations in France, sampled between 1977 and 2007. Densities are reported in eel/100m^2.

Brittany EMU

Overview

The Brittany Eel Management Unit is a part of the Loire-Brittany WFD River Basin District (Figure 5.44).

The Loire-Brittany district extends over 155 000 km² and covers 28% of the metropolitan France national territory. The Brittany EMU covers 27 209 km² and is characterized by a very long coastal range (2730 km) accounting for about a third of the French coasts. There are 560 watersheds in Brittany with over 30 000 km of river length. Most rivers are short, with only 15 basins having an area greater than 250 km^2. The largest basin, the Vilaine, lies in the east of Brittany, and is formed mostly of lowland, slow flowing rivers. In contrast, most rivers lying in the west of Brittany are fast flowing rivers flowing from the Armoricain mounts.

Stock status

Historically, the densities of yellow eels tended to be high in the Brittany region, with eel comprising 80% of the fish biomass caught in rivers in the 1980's (Legaut & Porcher, 1990). An analysis of eel distribution based on electrofishing data shows that the largest densities are now found in the area surrounding the Vilaine, and in the Vilaine itself, despite the large fishing pressure for glass eel in this basin. The analysis from a previous application of the EDA model (Hoffman, 2008) also shows that eel densities are quite high in the rivers flowing to the north, though glass eel fisheries are relatively underdeveloped in this area (Figure 5.47).

Due to a local bottleneck between the Brittany and Normandy coasts, the tide amplitude is very high, and favourable tide currents are thought to bring large amounts of glass eel to this region. A similar observation can be made for the Vilaine, which benefits from large recruitments but where high exploitation rates remove most of the glass eels.

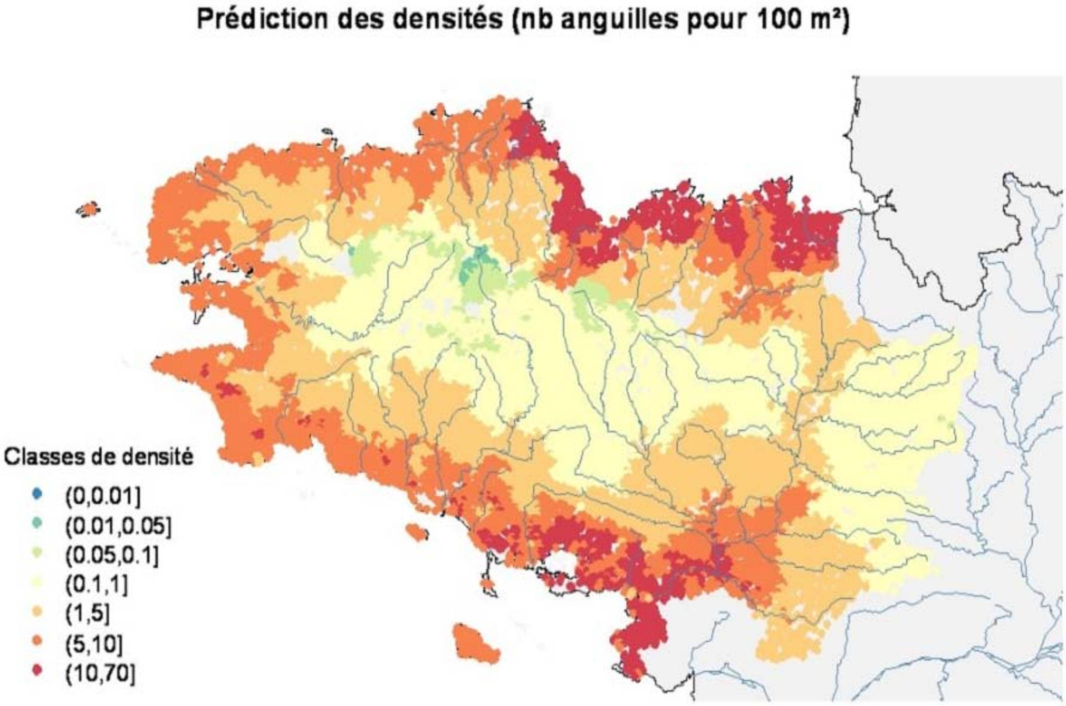


Figure 5.47. Densities (eel/100m^2) predicted by the EDA model in Brittany (Hoffman, 2008). Values reported in the legend indicate the lower and upper boundaries for each category of density.

Impacts on eel production

Dams

The Brittany streams are segmented with a large number of dams. A full GIS database of the dams and their impacts on eel is available. The cumulated impact of dams on eel migrations has been evaluated (Leprevost, 2007).

Fishery

The professional glass eel fishery is the main type of eel exploitation in Brittany. Most of the fishery is concentrated in the Vilaine: the 2008 catch from the Vilaine was 5300 kg, compared to only 550 kg for the remainder of Brittany. The largest estuaries in Brittany (Blavet, Aulne, Elorn) are those where most of the glass eel fishery occurs.

A professional yellow fishery of some importance is located in the Golfe du Morbihan and in the neighbouring estuaries in the southern part of Brittany near the mouth of the Vilaine. Recent landings were around 25 tons per year. However, the amateur fishery is larger, with historic landings reported at around 60 t declining to about 40 t in most recent years. The largest catches occur in the eastern part of Brittany (Figure 5.48).

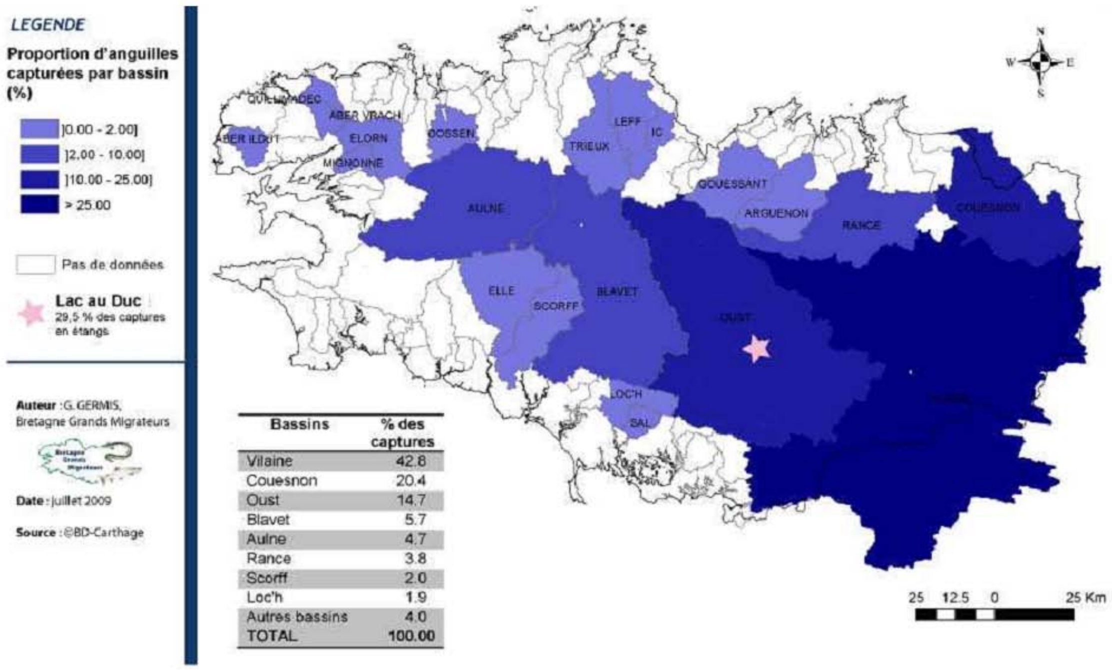


Figure 5.48. Distribution (%) of amateur river catches in Brittany (Gemis, 2009).

5.10. Application of the EDA to the Brittany data set

Input data

The model requires data about abundance of yellow eel. For France (Brittany and Rhone EMU) the data are obtained from the Aquatic environment and fish database (BDMAP - with 19309 fishing samples, collected on 8140 sampling stations from 1966 to 2008).The electrofishing surveys have been carried out by the French National Office of Water and the Aquatic Environments (ONEMA). Several electrofishing protocols have been used, and the data were restricted to electrofishing operations at two passes for sectors with a complete prospecting method by foot.

For Brittany, these data were extracted from data for 259 sites sampled during 1984 to 2008 and resulting in 745 discrete data points (Figures 5.49 to 5.52). The density (d) was calculated for all electrofishing operations and expressed in number/100m².

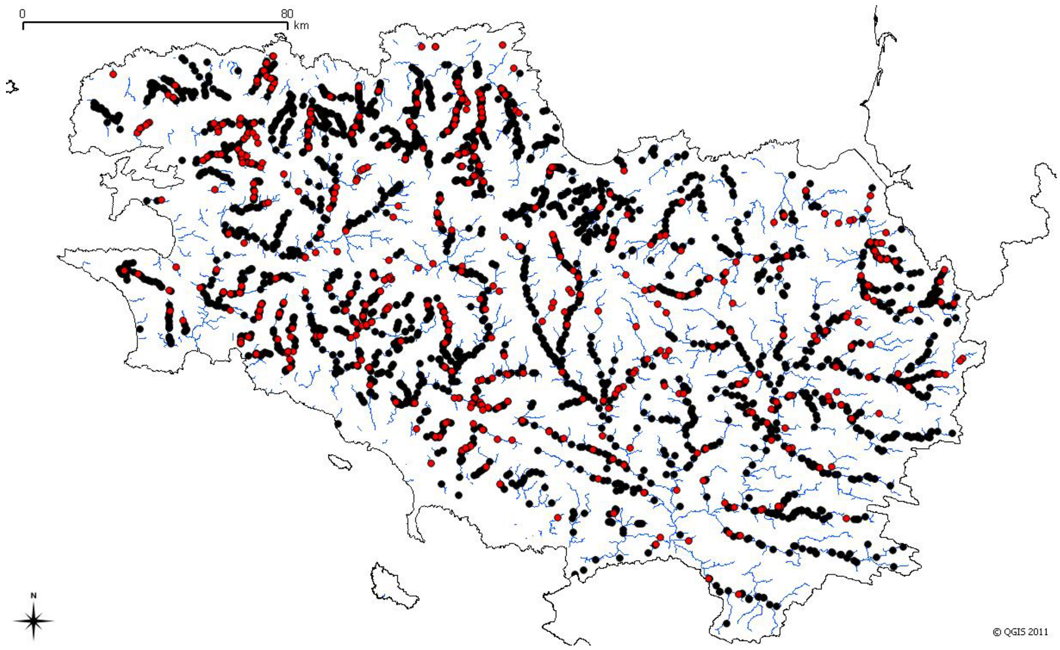


Figure 5.49. Sampling sites (in red) in the Brittany EMU catchment on the CCM river network (in blue) with the dams location (in black).

In the analysis, 745 electrofishing operations on 259 electrofishing stations from June and November from 1984 to 2008 were selected.

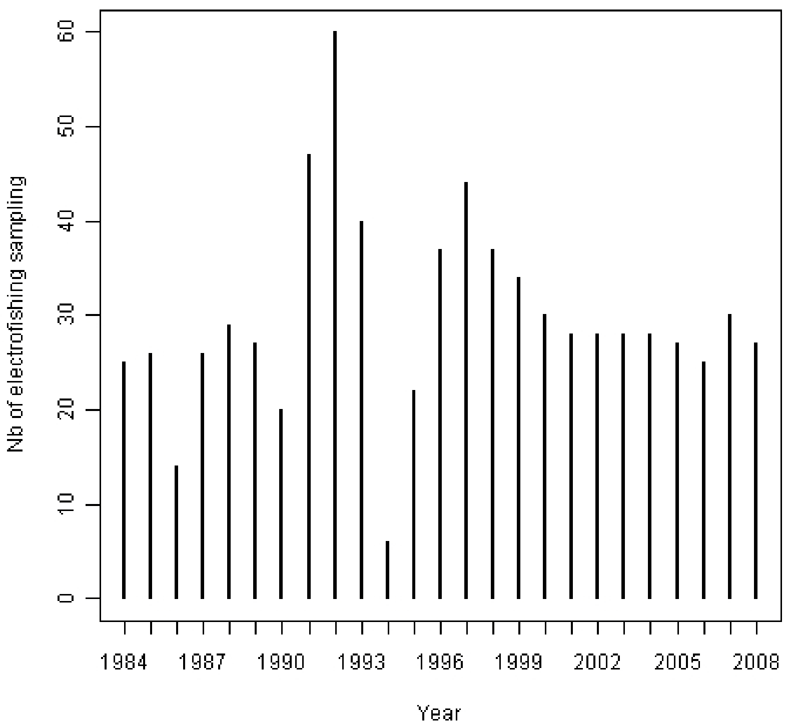


Figure 5.50. Number of electrofishing sampling per year in the Brittany Emu.

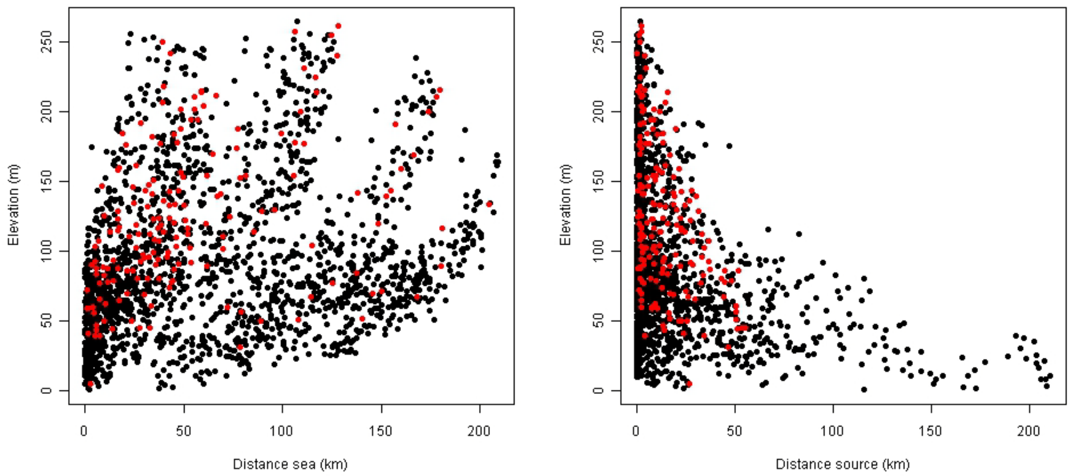


Figure 5.51. Sampling sites (in red) in the Brittany EMU catchment on the CCM river network (in black) depending on the elevation and the distance from the sea or from the source.

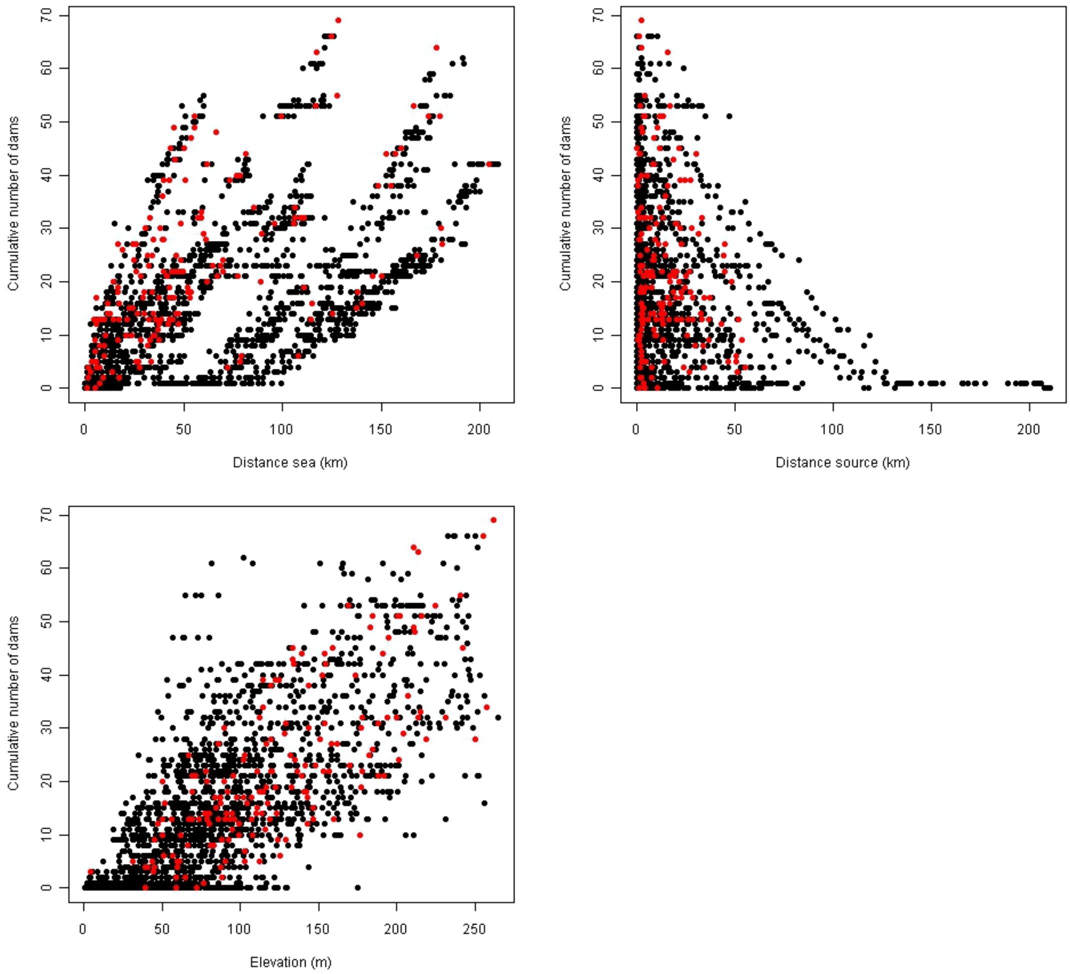
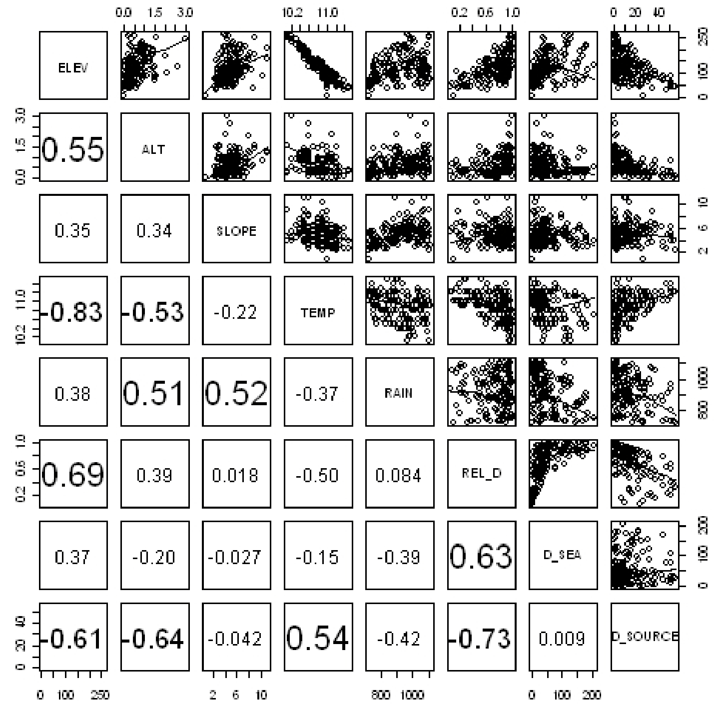


Figure 5.52. Sampling sites (in red) in the Brittany EMU catchment on the CCM river network (in blue) depending on the cumulative number of dams and the distance from the sea, the distance from the source or the elevation.

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CCM data set for the Brittany EMU provided information for each survey site on the distances to sea and source, relative distance, mean slope and elevation, altitudinal gradient, the area of land drainage upstream, mean annual temperature and rainfall, land use cover (urban, agricultural and no impact) , and the number of dams downstream of the site. The cumulative number of dams from sea to site was used to characterise the obstacle pressure to upstream migrating eels. In total there were 3801 dams distributed throughout the Brittany EMU.

A combination of 17 variables (year, month and a set of 15 variables) was tested. The Figures 5.53 and 5.54 show that there are several variables with tightly correlated predictors.



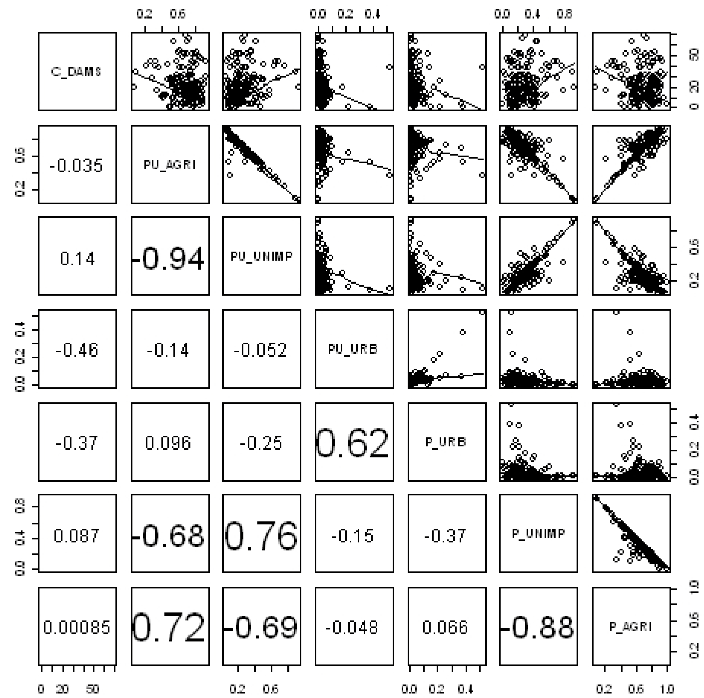


Figure 5.53. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With ELEV: elevation mean, ALT: altitudinal gradient, SLOPE: slope mean, TEMP: temperature mean, RAIN: rain mean, REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, C_DAMS: cumulative number of dams, PU_AGRI : upstream percent in agricultural use, PU_UNIMP: upstream percent in unimpact land use (p_up_no_impact), PU_URB: upstream percent in urban use, P_URB: local percent in urban use, P_UNIMP: local percent in unimpact land use (p_no_impact), P_AGRI: local percent in agricultural use. The numbers below the diagonal are correlations coefficients. The font size of the cross-

correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

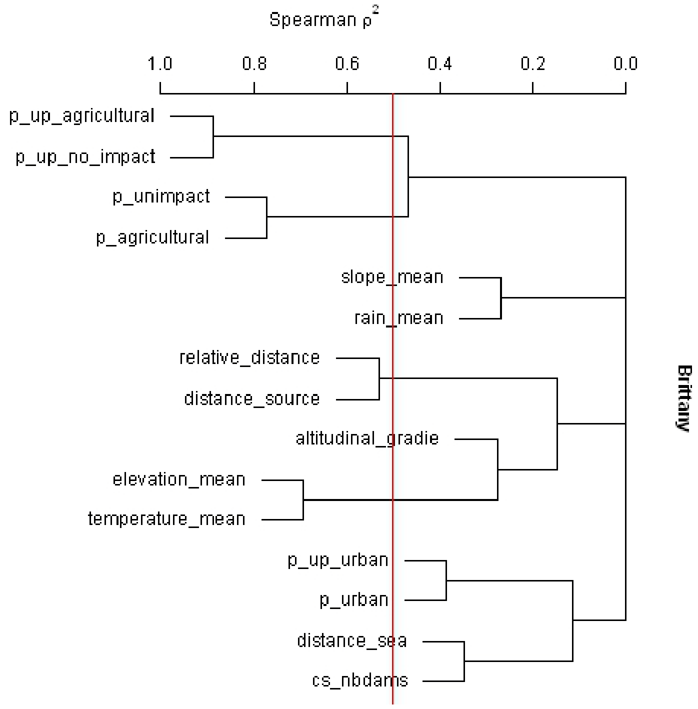


Figure 5.54. Dendrogram obtained by hierarchical cluster analysis of 17 candidate predictors for Brittany dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ²=0.5.

According to the threshold of 0.5 for ρ², the following variables should be grouped:

- p_up_agricultural, p_up_no_impact
- p_no_impact, p_agricultural
- relative distance, distance source
- elevation, temperature

A separate entry was used for the pairs of variables with significant relationship (Figure 5.54) and for the 4 groups of variables described in Figure 5.53 to avoid spurious correlation.

The combination of the different groups resulted in 3267 models to be tested.

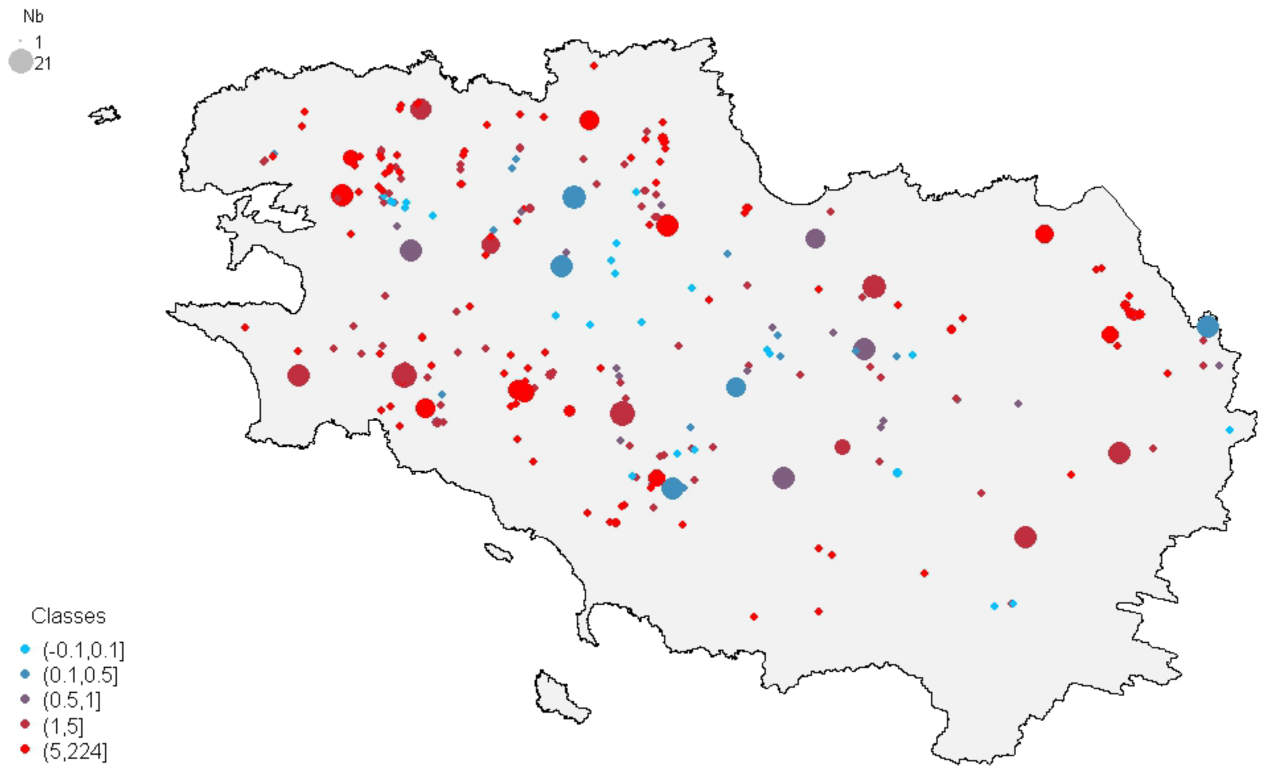


Figure 5.55. Map of density observed densities. Eel.100 m^{-2} All years.

Model Testing Procedure

We calibrated statistical models with a Generalized Additive Model (Hastie and Tibshirani, 1990) to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/). EDA model is based on a delta-gamma model (Stefánsson, 1996) which combines two generalized additive model (GAM). There are three steps of modelling:

- a presence/absence model (delta model) based on a GAM with a binomial distribution and a logit link to determine the probability of a positive catch;
- a density model with the positive data (gamma model) using a GAM with a gamma distribution and logarithm link to determine the level of positive catch; and,
- the multiplication of the two previous models (delta-gamma model).

The GAMs were computed with the library 'gam' (Hastie, 2010) with a cubic spline smoother (3 degree of freedom) for each environmental variable. The delta-gamma (ΔΓ) generalized additive models explain a large portion of the variability in eel abundance data, as there are many occasions where densities are null.

Presence-absence model (Delta model)

The results for tests of 3267 models are summarised in Table 5.11. The best density model is selected by the Akaike's Information Criterion (AIC) with a lower AIC indicating a better fit (Akaïke, 1974, Sakamoto *et al.,* 1986). The AIC function indicates a better fit of the response variable at the year of electrofishing sample with the rain fall, the temperature, the distance from the sea, the distance from the source, the upstream percent of urban use and the upstream percent of no impact land use (model 1 in Table 5.11). For this model the kappa = 0.557 ± 0.048 with a threshold = 0.68. GAM explained only 33 % of the deviance of the abundance of the yellow eel. Four explanatory variables are significant (Table 5.12): temperature, distance source, upstream percent of urban use and upstream percent of unimpacted land use.

Table 5.11. Model selection results using Akaike's information criterion (AIC) for presence-absence model analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	rain mean	temperature mean	slope mean	distance sea	distance source	cs_nbdams	p_up_urban	p-up_agricultural	p_up_no_impact	AIC s=3
1	x		x	x		x	x		x		x	396.2
2	x		x	x		x	x		x	x		396.7
3	x			x		x	x		x		x	397.7
4	x		x	x		x			x		x	397.7
5	x			x		x	x		x	x		398.2
6	x			x	x	x	x		x		x	398.5
7	x		x	x		x			x	x		398.6
8	x			x	x	x	x		x	x		399
9	x		x	x		x		x	x		x	400.3
10	x	x	x	x		x	x		x		x	400.6

Model Goodness of fit (presence-absence model)

The best presence-absence model selected (a binomial model with a logit link and a cubic spline smoother – s =3) is:

d != 0 ~s(year,3)+ s(rain_mean,3)+ s(temp_mean,3)+ s(distance_source,3)+ s(distance_sea,3)+ s(p_up_urban,3)+ s(p_up_no_impact,3)

matrice confusion

observed

predicted 1 0

1 626 33

0 34 52

correctly predicted 0.91

present correctly predicted 0.95

absents correctly predicted 0.61

Table 5.12. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar Chisq	P(Chi)	
(Intercept)	1				
s(annee, 3)	1	2	1.9514	0.3769180	
s(rain_mean, 3)	1	2	5.7872	0.0553859	.
s(temperature_mean, 3)	1	2	25.4104	3.035e-06	***
s(distance_source, 3)	1	2	9.3835	0.0091688	**
s(distance_sea, 3)	1	2	2.2644	0.3223256	
s(p_up_urban, 3)	1	2	13.9506	0.0009347	***
s(p_up_no_impact, 3)	1	2	7.5127	0.0233708	*

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

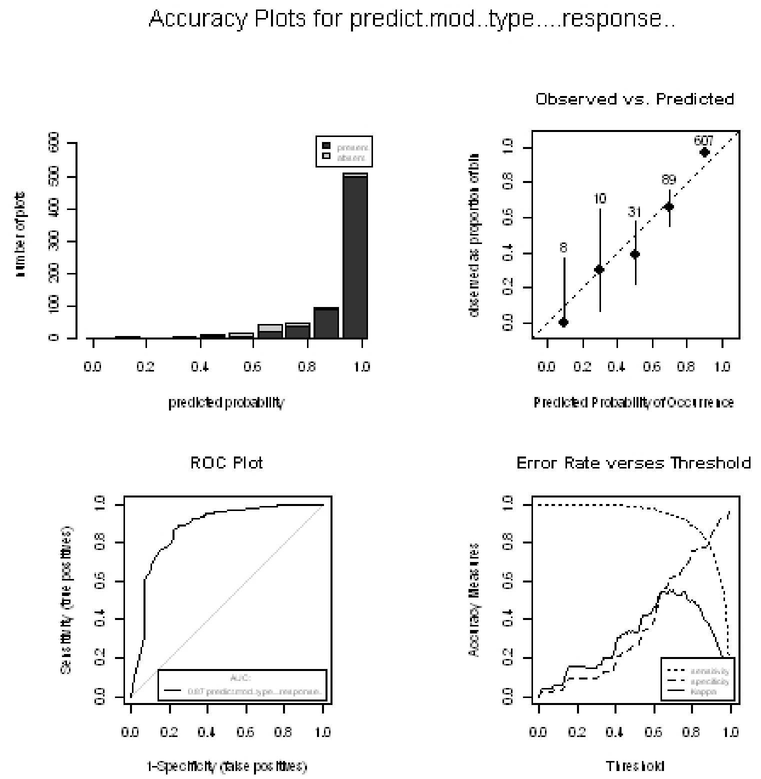


Figure 5.56. Model quality and threshold selection graphs for presence-absence model selected with a histogram plot (upper left), a calibration plot (upper right), a ROC plot with the associated Area Under the Curve (AUC) (lower left), and an error rate verses threshold plot (lower right).

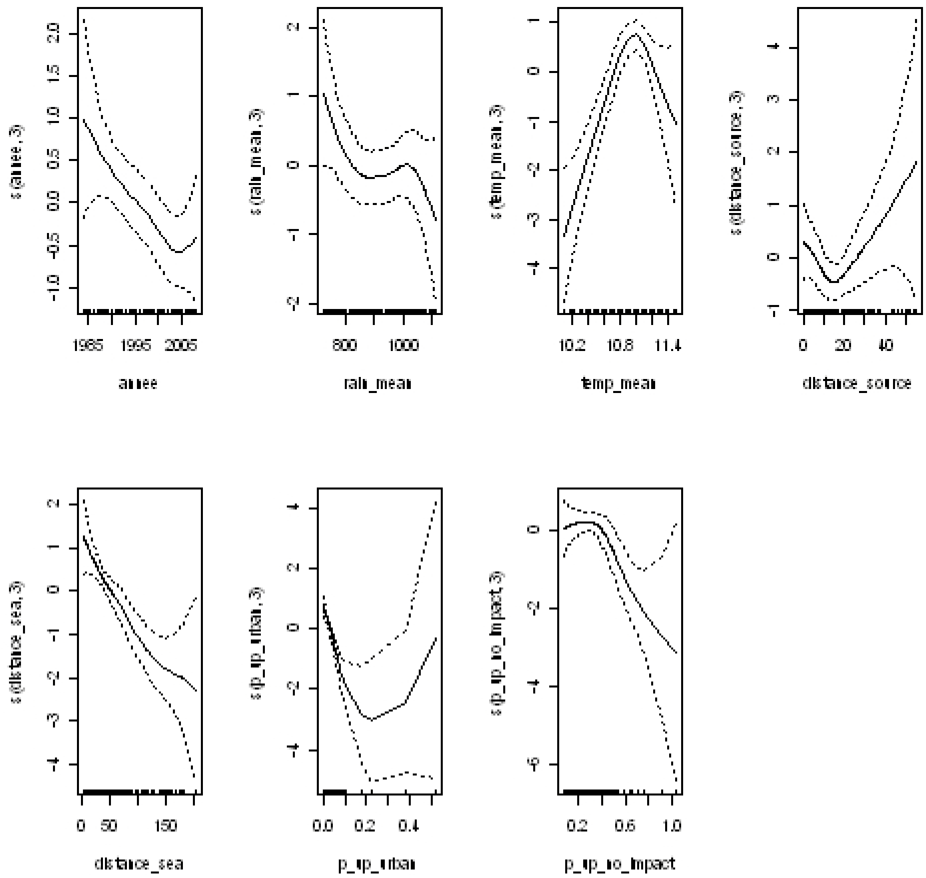


Figure 5.57. Response curves of each variable included in the generalized additive model (GAM) for the presence/absence model for Brittany EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The residual of the presence absence model are plotted for each year (Figure 5.58). If the results of the fishing operation are in overplot (several year point on top of the other), they are slightly moved around their origin to show on the map. Several operations for the same year will remain in overplot. The residuals correspond to observed -predicted values, hence a blue spot (negative value) will indicate a predicted value larger than the observed one, and a red point (positive value) a predicted value lower than the observed one.

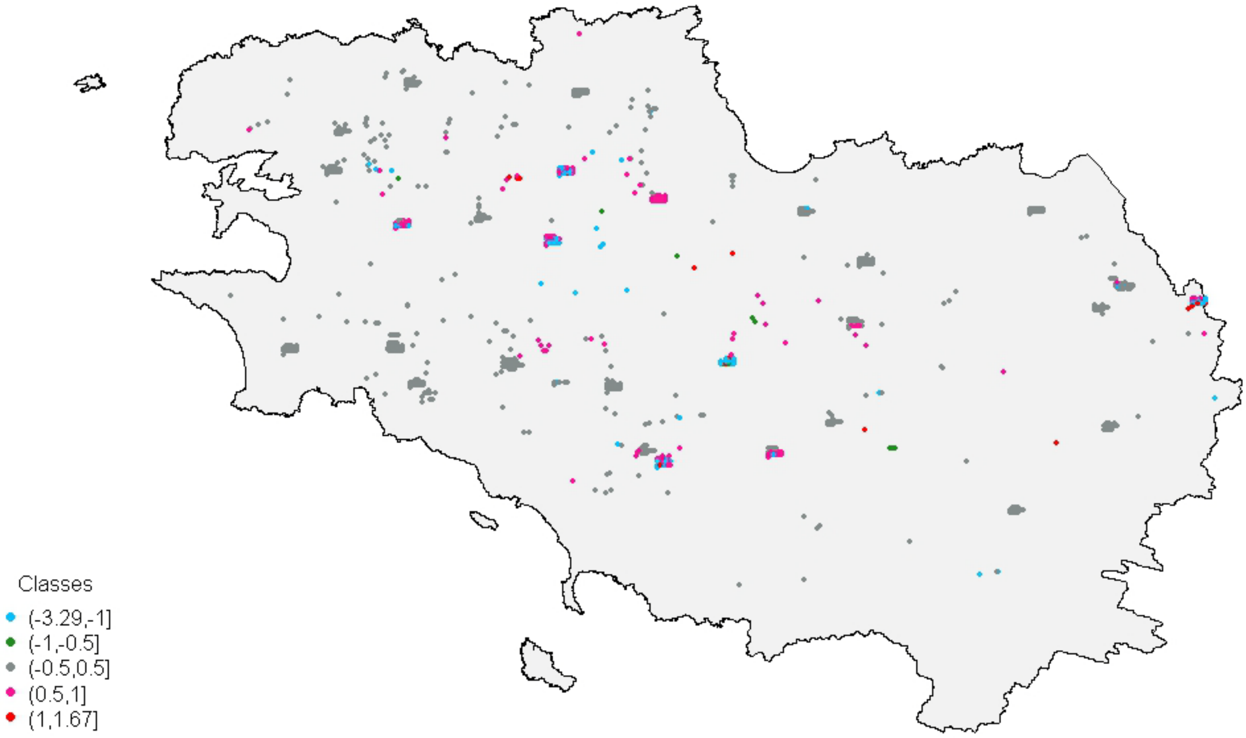


Figure 5.58. Residuals per year: presence-absence model.

In the next graph (Figure 5.59), the residuals are averaged for each station. A blue spot is a station with on average over the years predicted values far larger than the observation. The size of the point is related to the number of electrofishing operation in the station.

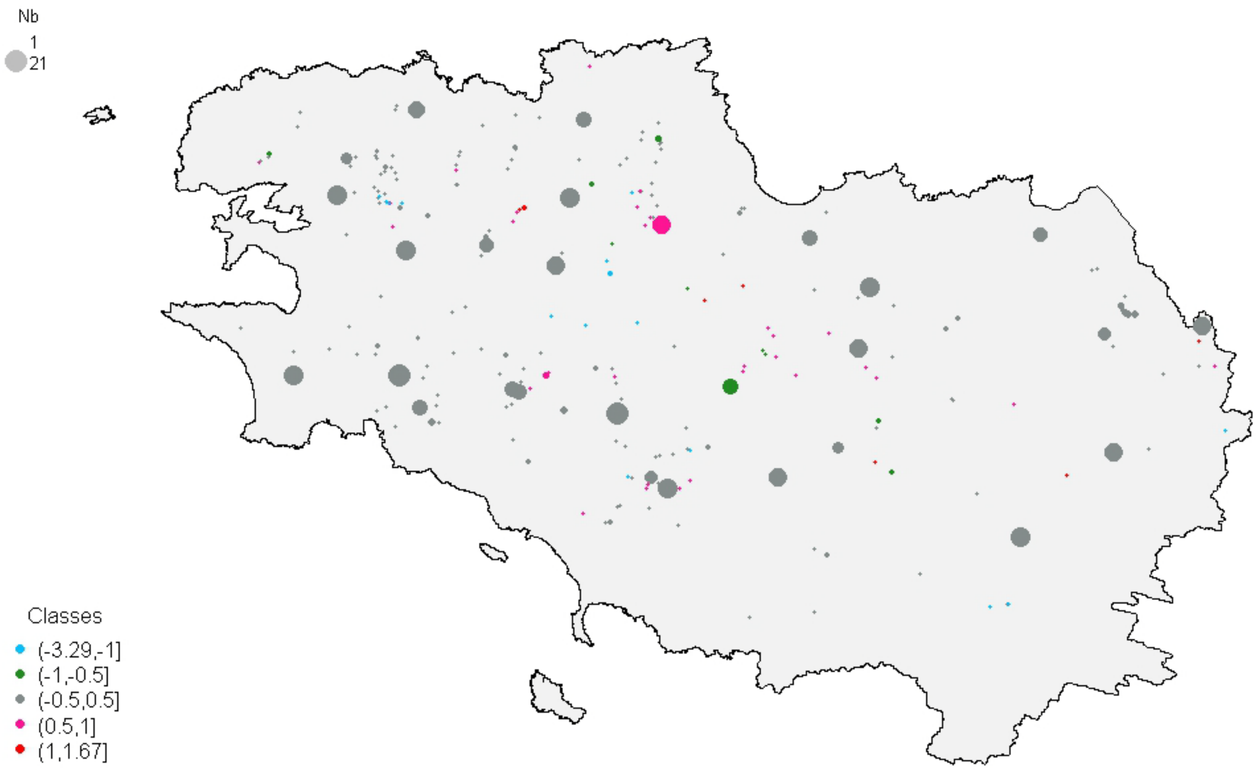
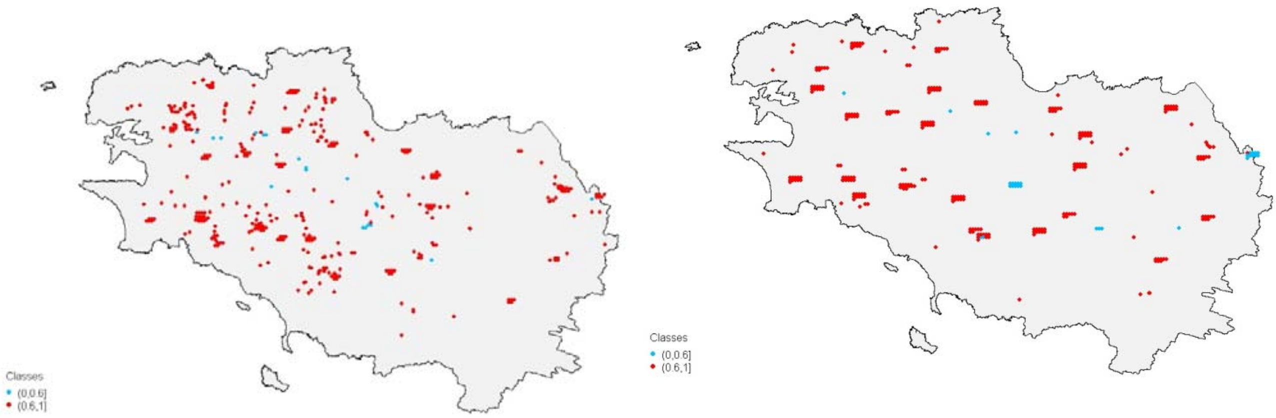


Figure 5.59. Mean Residuals per stations: presence-absence model.

The map of predicted value (Figure 5.60) from the model is computed before and after the break point in the year variable response trend. The threshold chosen for the response is the value giving the best Kappa, i.e. trade-off between the lowest number of wrongly predicted presence (false positive), and wrongly predicted absence (false negative).



(a) (b)

Figure 5.60. Predicted value: presence-absence model year<1998 (a), Predicted value: presence-absence model year≥1998.

Density model

The density model is applied only to positive value. It is a gam model with a gamma distribution and a logarithm link. 3267 models have been tested.

Model results for density model are summarized in Table 5.12. The AIC function indicates a better fit of the response variable at the year of electrofishing sample with altitudinal gradient, slope, relative distance, cumulative number of dams, upstream percent of urban use and the upstream percent of agricultural land use). GAM explained 54% of the deviance of the abundance of the yellow eel. The effects of six explanatory variables in the model (year, altitudinal gradient, slope, relative distance, cumulative number of dams p_up_agricultural) are significant (Table 5.13). The Spearman rank correlation between the observed values and the fitted values is statistically significant (ρ=0.645, p-value< 2.2 10^{-16}, Figure 5.61).

Table 5.12. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	altitudinal gradient	slope mean	distance sea	distance source	relative distance	cs_nbdams	p_urban	p_agricultural	p_no_impact	p_up_urban	p_up_agricultural	p_up_unmpact	AIC s=3
1	x		x	x			x	x				x	x		3500.6
2	x		x	x			x	x				x		x	3501.7
3	x	x	x	x			x	x				x	x		3502.9
4	x	x	x	x			x	x				x		x	3503.2
5	x	x	x	x	x	x			x		x				3503.5
6	x		x	x			x	x				x			3507.5
7	x	x	x	x			x	x	x		x				3508.5
8	x	x	x	x			x	x	x						3508.8
9	x	x	x	x			x	x	x						3509.6
10	x	x	x	x			x	x	x	x					3510.4

Model Goodness of fit (density model)

The best density model selected is: d ~ s(year,3) +s(slope,3)+ s(altitudinal_gradient,3)+ s(relative distance,3)+ s(cs_nbdams,3)+ s(p_up_urban,3)+ s(p_up_agricultural,3)

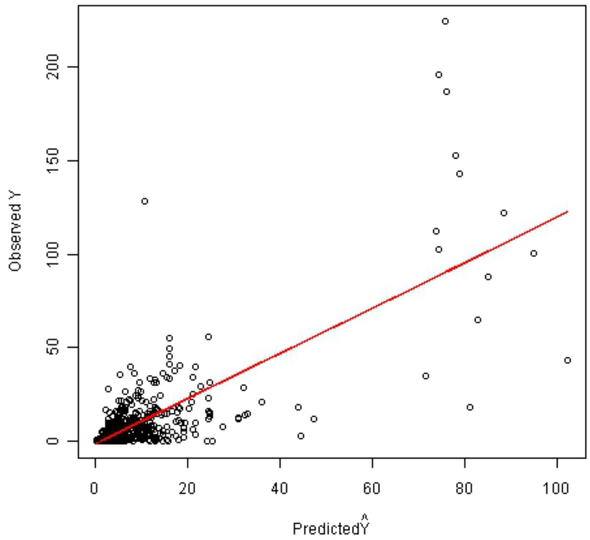


Figure 5.61. Observed vs. predicted regression scatter plot.

Table 5.13. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar F	Pr(F)	
(Intercept)	1				
s(annee, 3)	1	2	5.8755	0.002962	**
s(slope_mean, 3)	1	2	9.9142	5.752e-05	***
s(alt_gradie, 3)	1	2	31.1032	1.290e-13	***
s(relative_distance, 3)	1	2	10.8388	2.349e-05	***
s(cs_nbdams, 3)	1	2	17.8470	2.872e-08	***
s(p_up_urban, 3)	1	2	2.5523	0.078700	.
s(p_up_agricultural, 3)	1	2	5.8411	0.003064	**

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

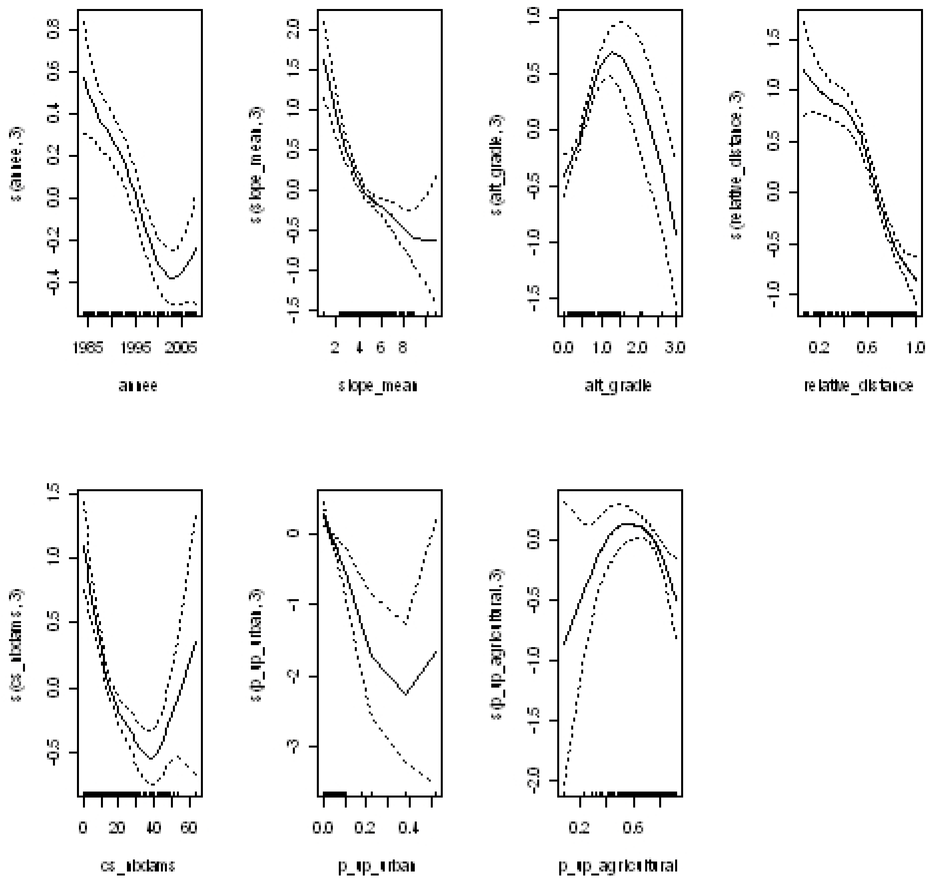


Figure 5.62. Response curves of each variable included in the generalized additive model (GAM) for the density model for Crepe EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The map below (Figure 5.63) shows the mean residuals for operations where positive densities were observed. The sign is calculated according to predicted – observed. The size of the points gives an indication of how many electrofishing operations occurred at that point and thus of the 'weight' of that station in the model.

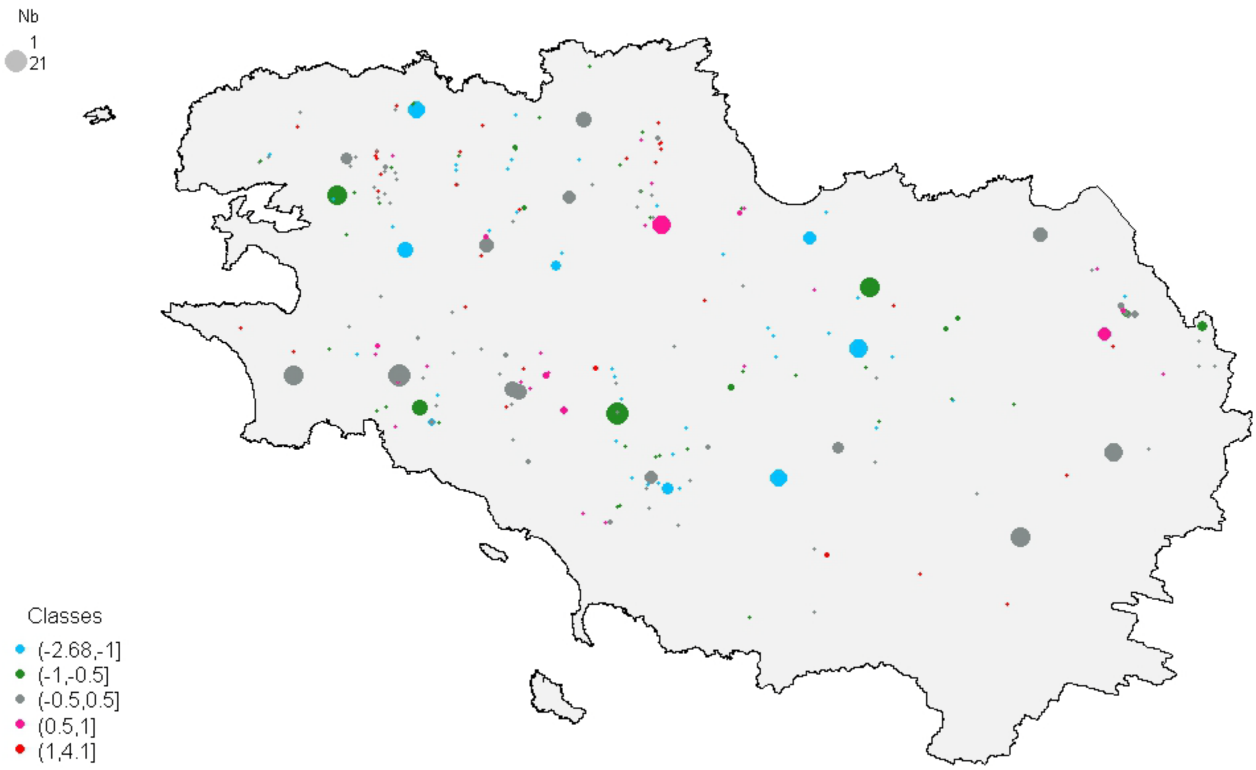


Figure 5.63. Mean residuals per stations: density model.

The next map (Figure 5.64) gives the same information but residuals are 'jittered' around the point. There is no clear spatial trend in the presence of large positive or negative residuals.

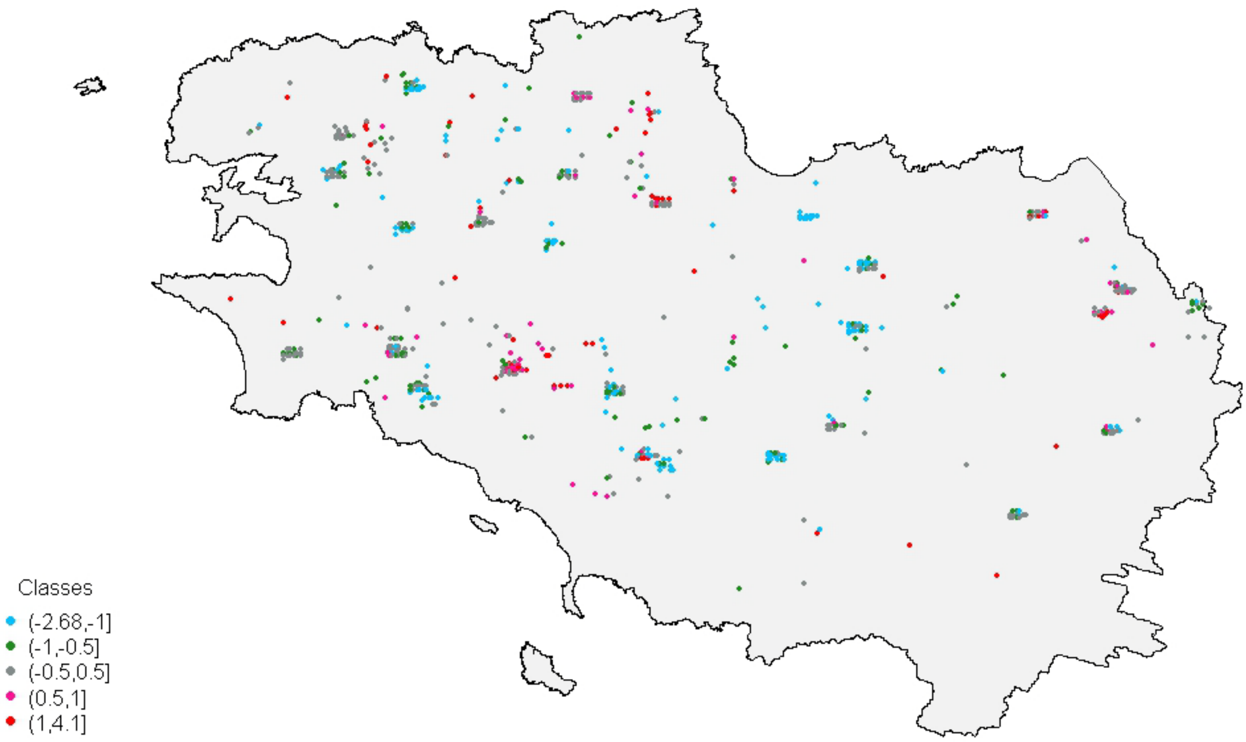
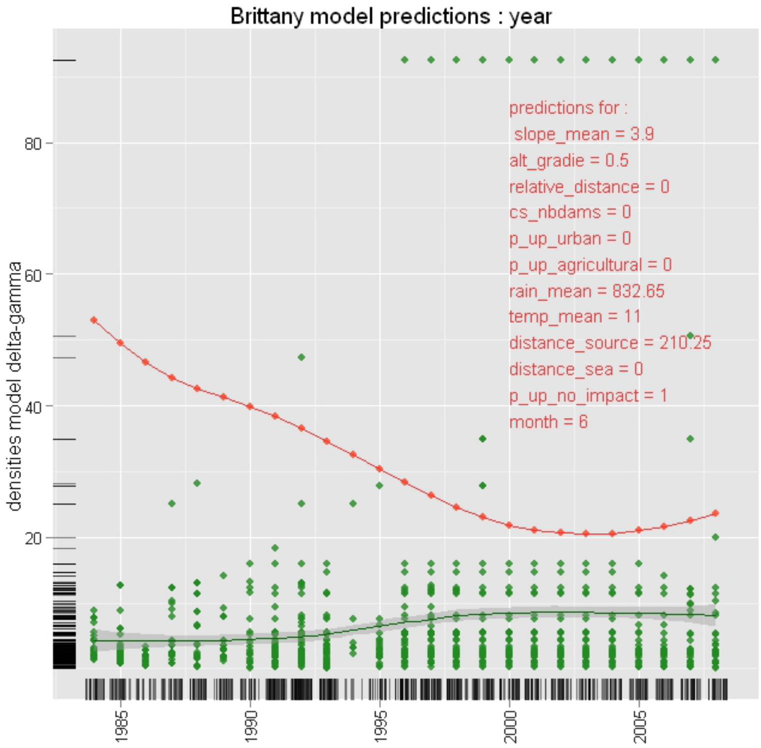
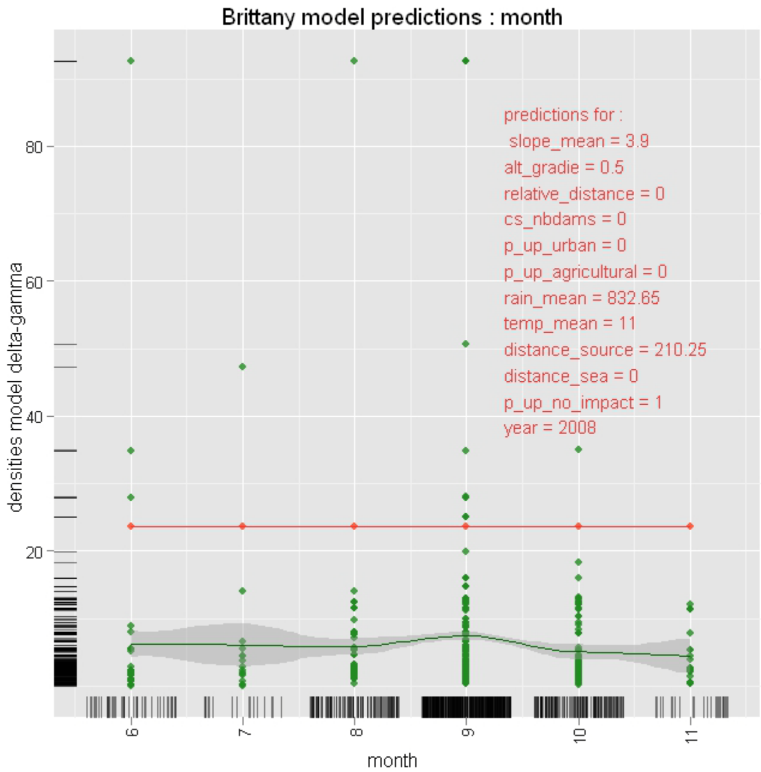


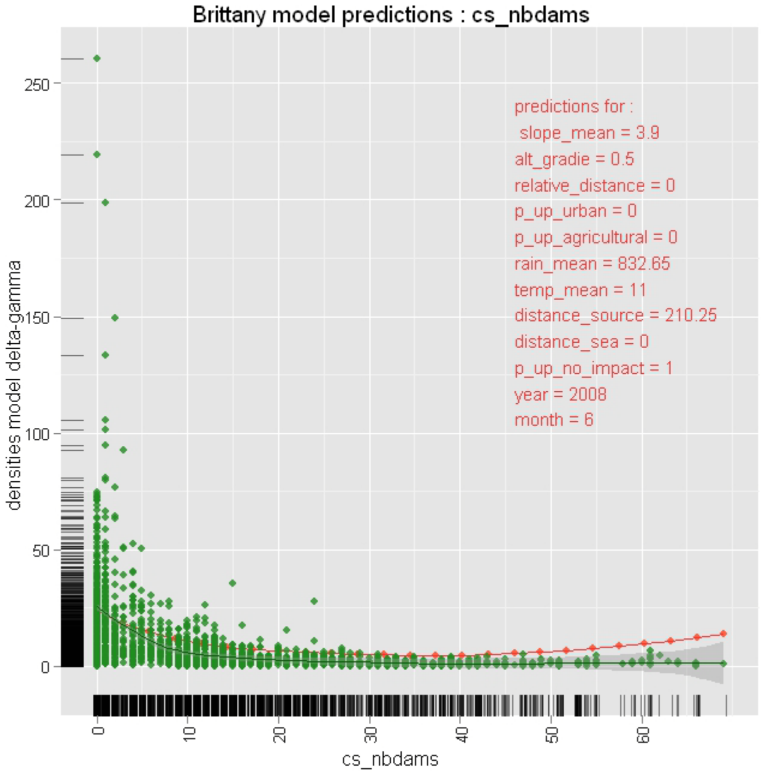
Figure 5.64. Residuals per year: density model.

Final model

The final model is the product of the delta model by the gamma model. The delta model is the probability of having a positive density. The gamma model is the level of a density for positive values. The model is used to predict densities on all the river stretches of the EMU. The following figures (not numbered) give the trend (red curve) of an effect given that all other parameters are fixed (see figure for the value of fixed parameters) as well as the predicted density of each river stretch of the CCM (green points) and its average along the examined effect (green curve). The rugs on the scale have been moved slightly to indicate the density of observation. The densities are provided in eel.100 m^{-2}


Brittany model predictions : year
predictions for :
slope_mean = 3.9
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
month = 6
densities model delta-gamma
20
40
60
80
1985
1990
1995
2000
2005


Brittany model predictions : month
predictions for :
slope_mean = 3.9
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
densities model delta-gamma
20
40
60
80
6
7
8
9
10
11
month


Brittany model predictions : cs_nbdams
predictions for :
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alt_gradie = 0.5
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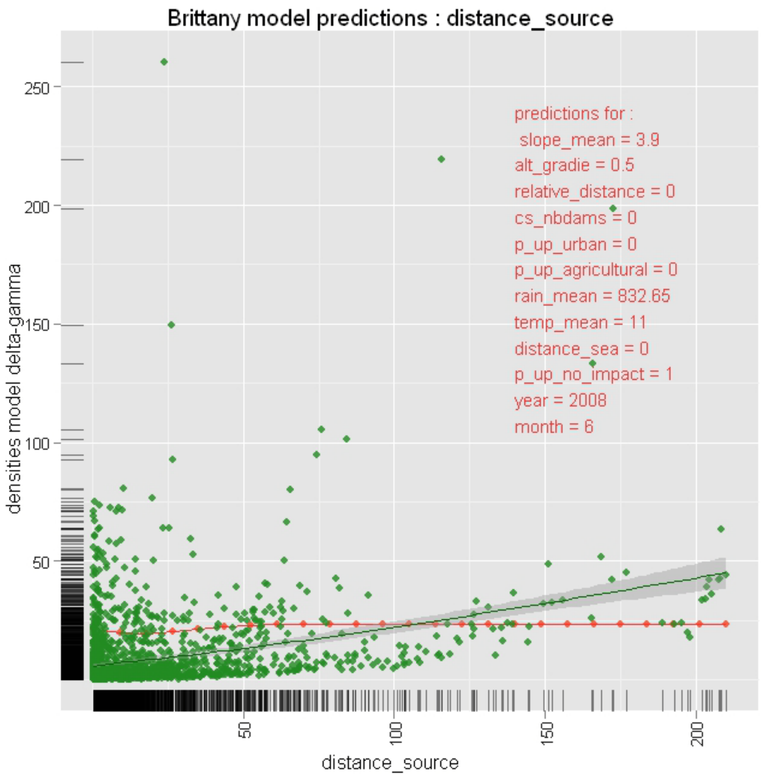
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
cs_nbdams
0
50
100
150
200
250
0
10
20
30
40
50
60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predictions for :
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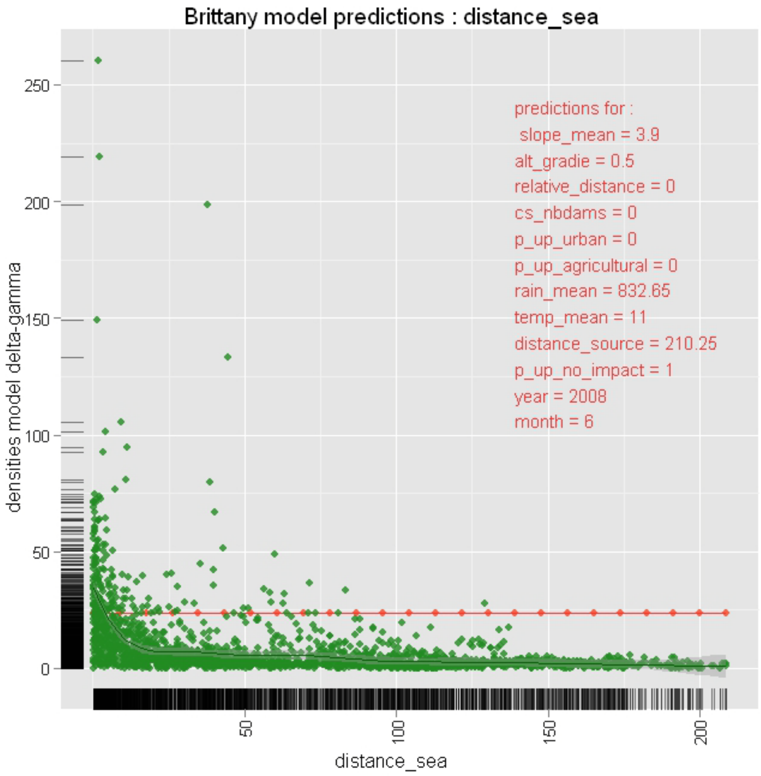
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
distance_source
50
100
150
200
250
50
100
150
200


Brittany model predictions : distance_sea
predictions for :
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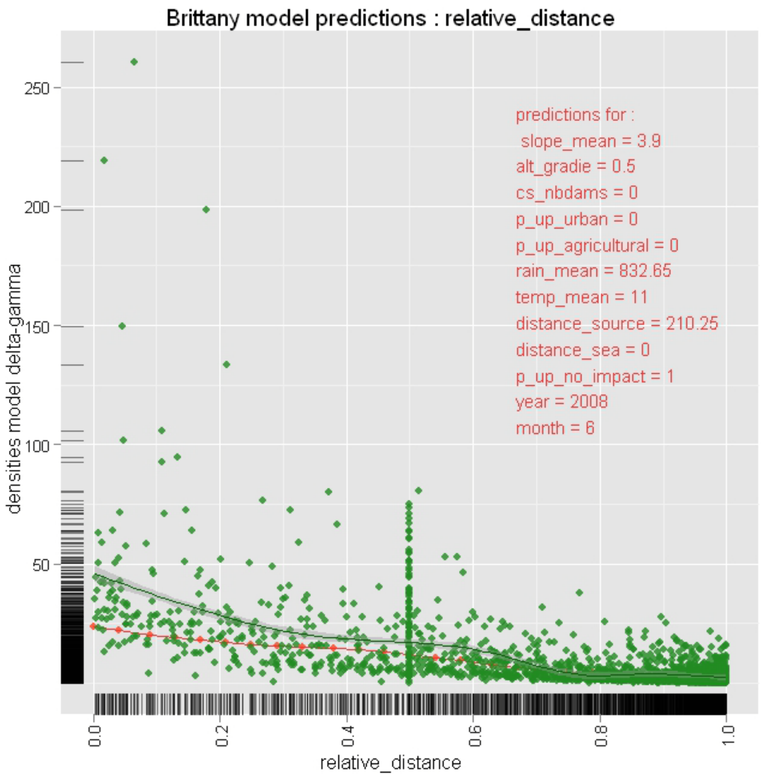
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
distance_sea
0
50
100
150
200
250
50
100
150
200


Brittany model predictions : relative_distance
predictions for :
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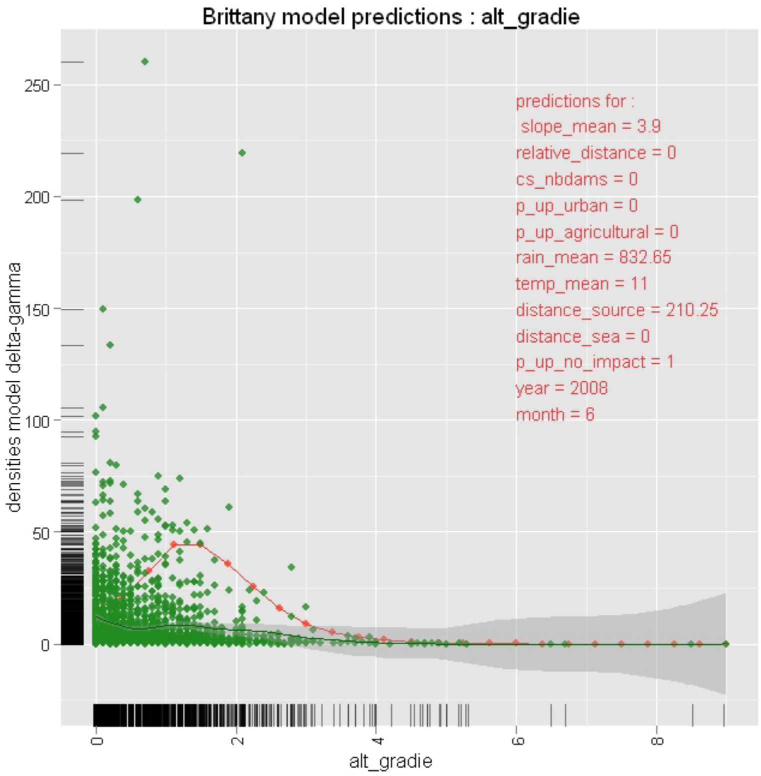
alt_gradie = 0.5
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
relative_distance
50
100
150
200
250
0.0
0.2
0.4
0.6
0.8
1.0


Brittany model predictions : alt_gradie
predictions for :
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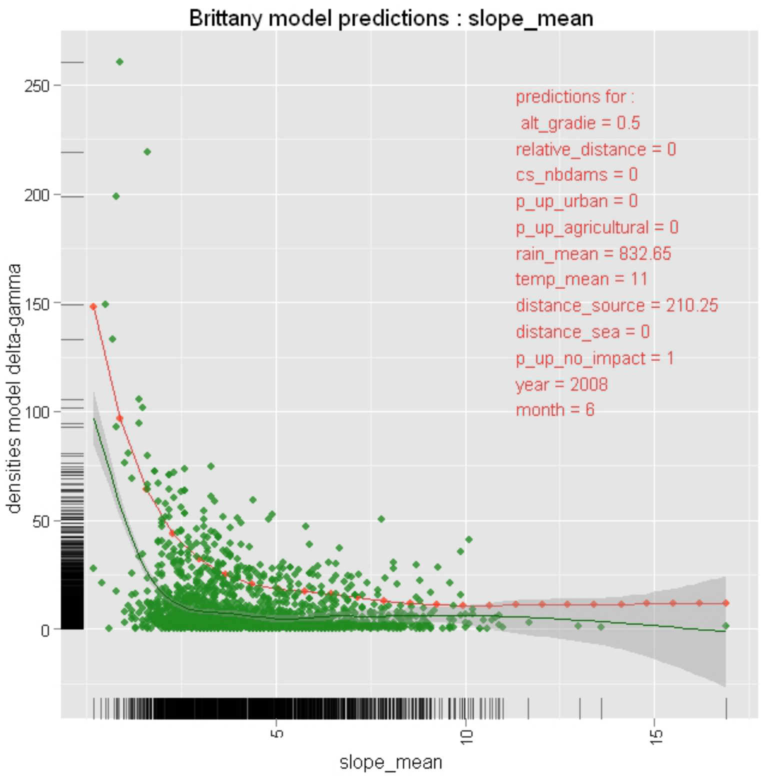
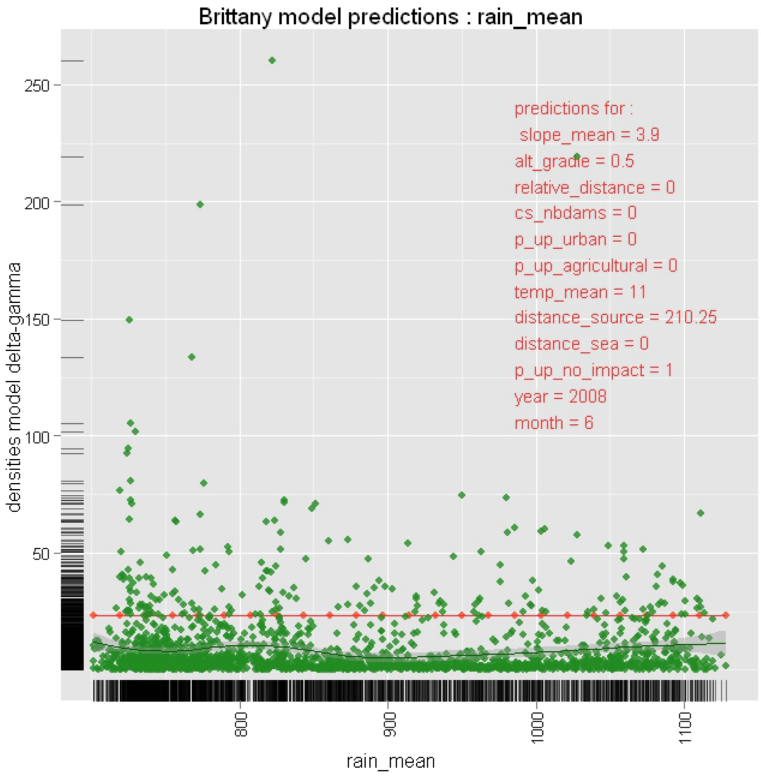
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
alt_gradie
0
50
100
150
200
250
2
4
6
8


Brittany model predictions : slope_mean
predictions for :
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
slope_mean
0
50
100
150
200
250
5
10
15


Brittany model predictions : rain_mean
predictions for :
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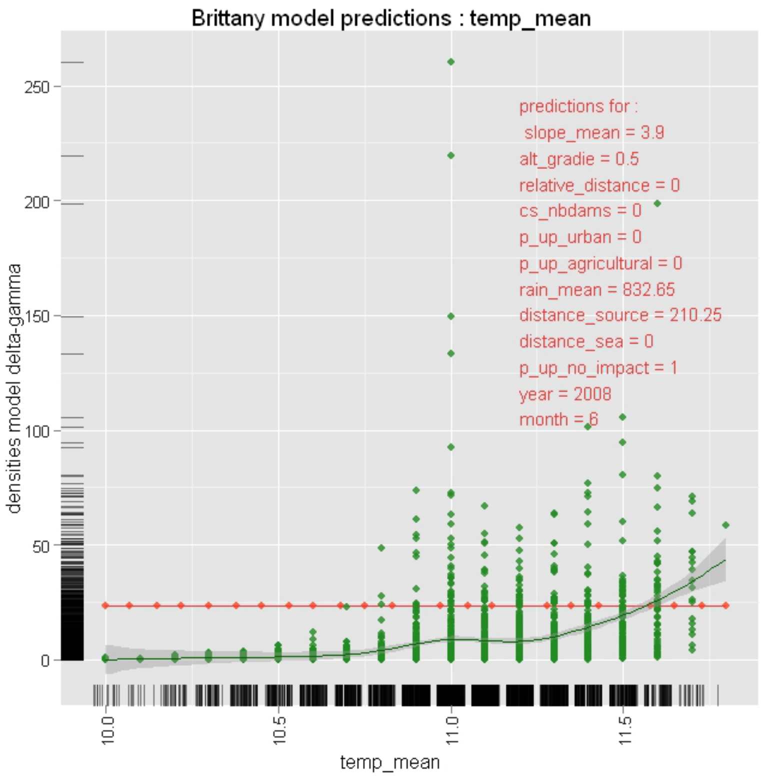
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
250
200
150
100
50
800
900
1000
1100
rain_mean


Brittany model predictions : temp_mean
predictions for :
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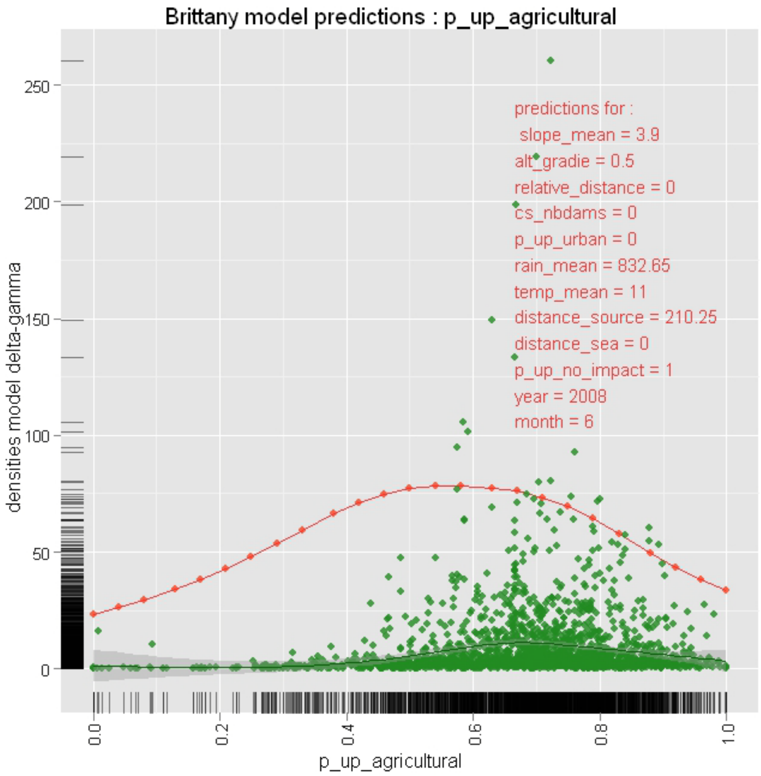
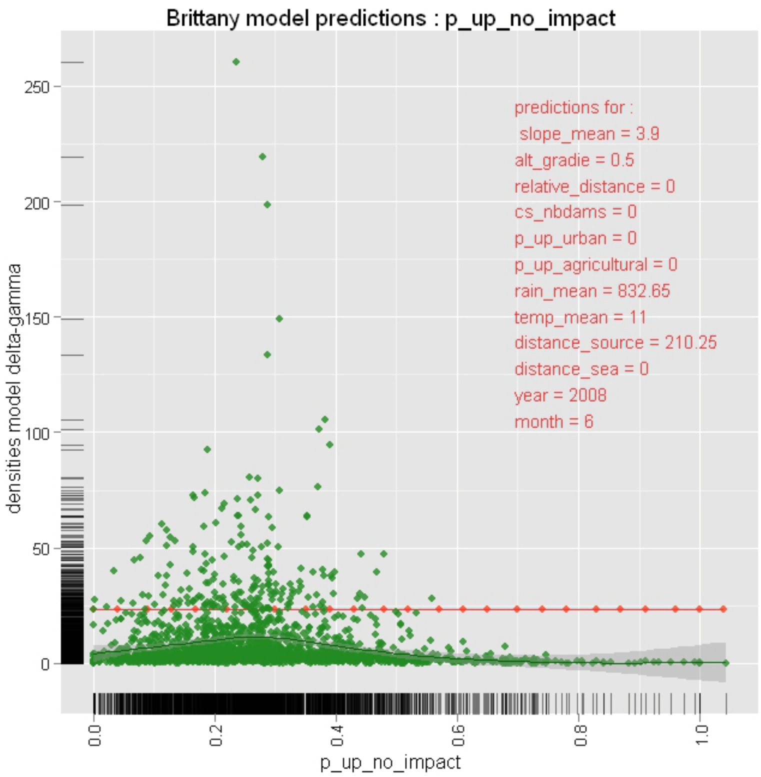
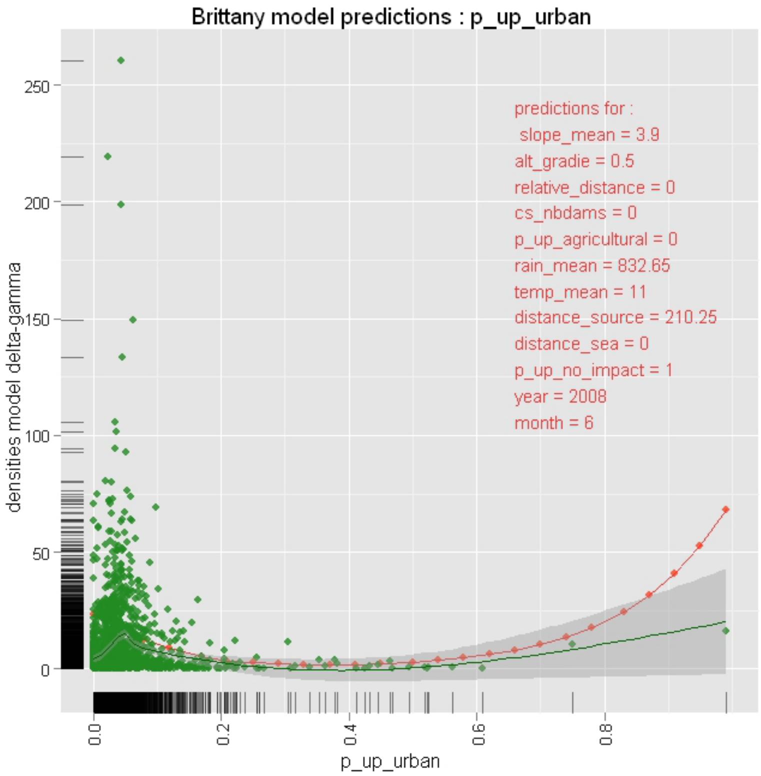
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
250
200
150
100
50
0
10.0
10.5
11.0
11.5
temp_mean


Brittany model predictions : p_up_agricultural
predictions for :
slope_mean = 3.9
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
p_up_agricultural
0
50
100
150
200
250
0.0
0.2
0.4
0.6
0.8
1.0


Brittany model predictions : p_up_no_impact
predictions for :
slope_mean = 3.9
alt_gradie = 0.5
relative_distance = 0
cs_nbdams = 0
p_up_urban = 0
p_up_agricultural = 0
rain_mean = 832.65
temp_mean = 11
distance_source = 210.25
distance_sea = 0
year = 2008
month = 6
densities model delta-gamma
p_up_no_impact
0
50
100
150
200
250
0.0
0.2
0.4
0.6
0.8
1.0



The following figures (5.65 and 5.66) give the yellow and silver eel production along the examined variable. On the upper corner they also give the cumulated wetted area along the same variable that supports that production.

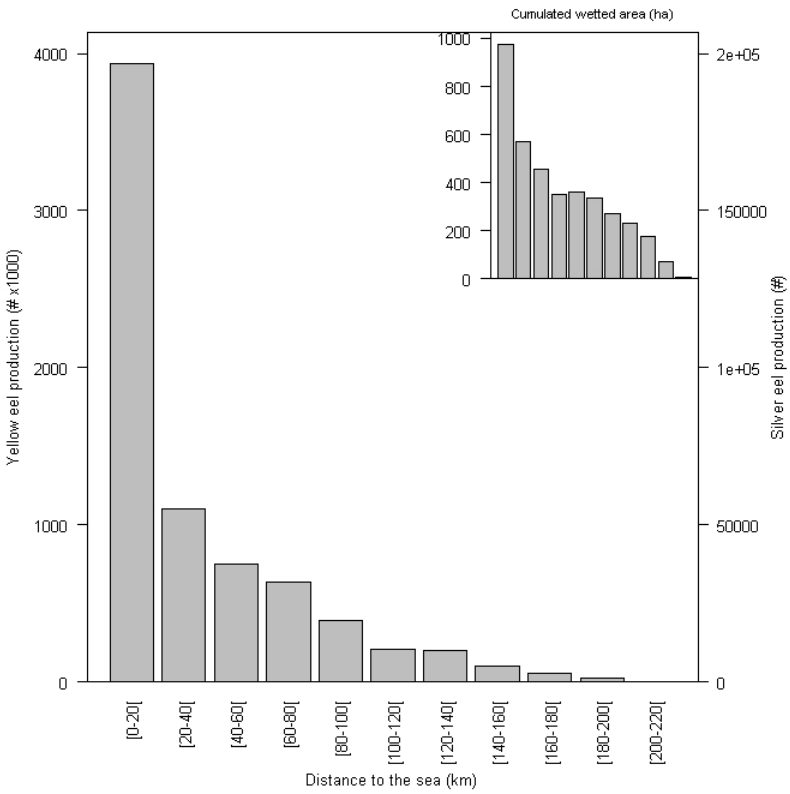


Figure 5.65. Yellow and silver eel production along the distance from the sea.

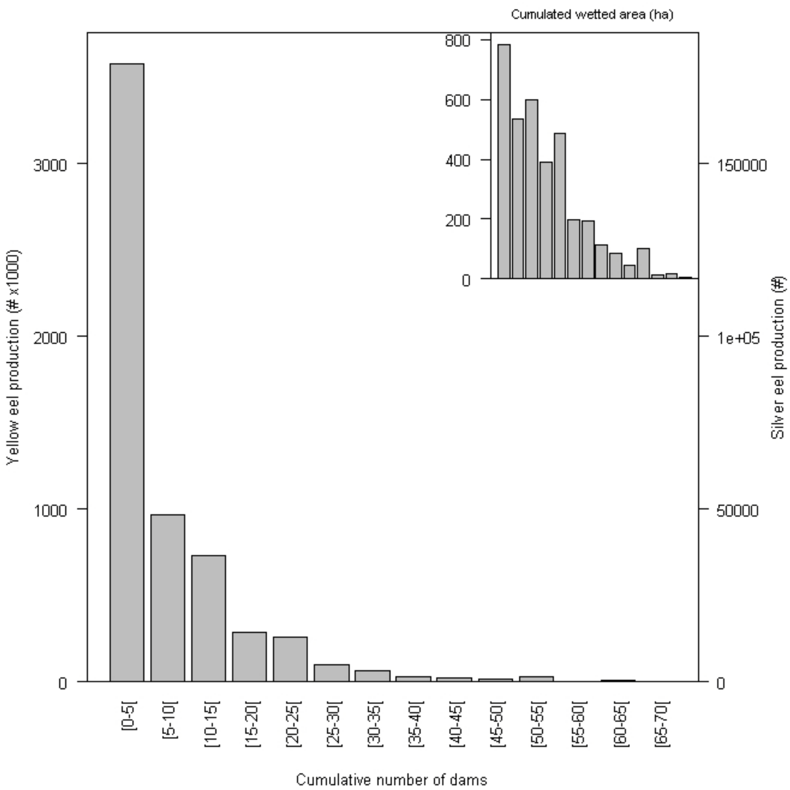


Figure 5.66. Yellow and silver eel production along the cumulative number of dams.

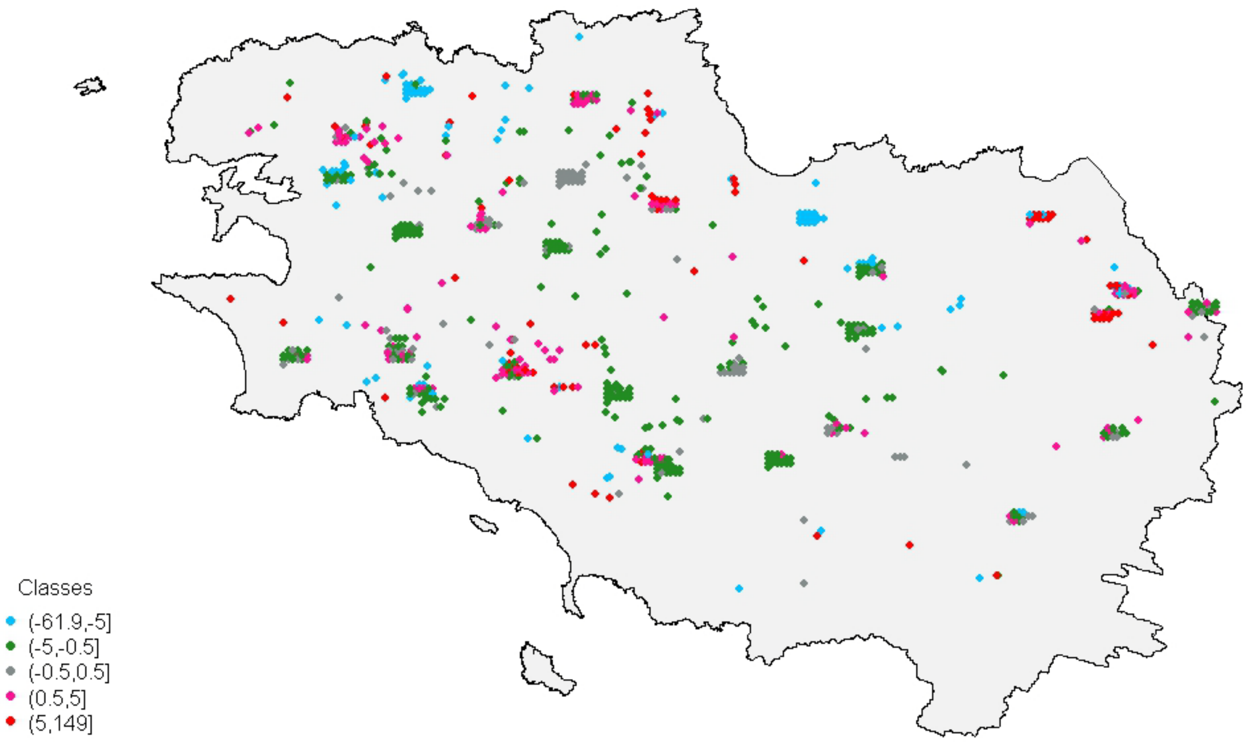


Figure 5.67. Map of residuals (predicted - observed), for the full model. Each year of data a point is shown with a small distance around the point.

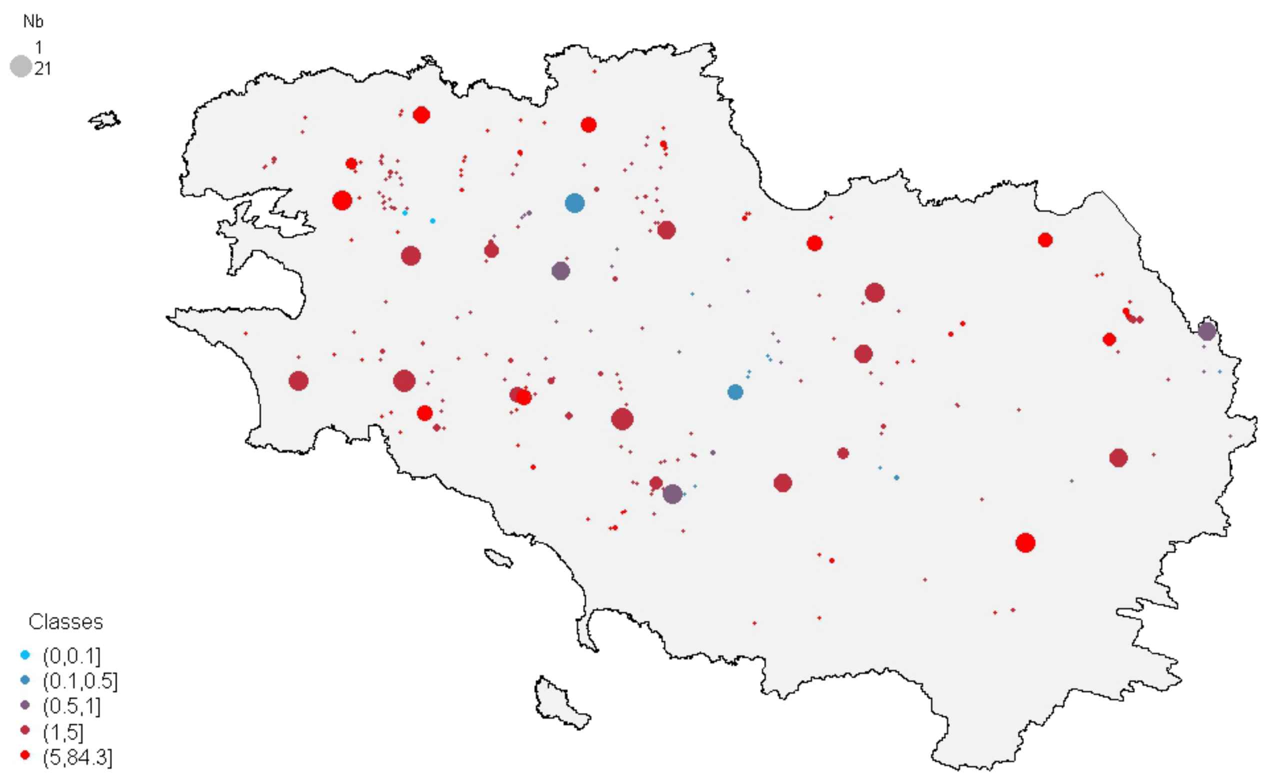


Figure 5.68. EDA predictions for the full model, the size model is given according to the number of electrofishing at that point. Classes in eel.100 m^{-2}

EDA predictions for the Brittany EMU

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch with a conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al.* (2000) have estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA a constant conversion rate of 5% was chosen as a default value. This constant rate conversion of eel in equivalent silver eel is based on the assumption of density-independent biological processes.

This potential escapement in number in then converted into biomass with the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for Brittany EMU. The current biomass ($B_{current}$) is calculated by subtracting the anthropogenic mortalities on silver eel to the potential escapement in biomass.

Best achievable escapements (B_{best}) can be calculated by the current biomass artificially forced to null anthropogenic impact (no dam, land use mortality to “no impact” and silver eel catch to 0) added with silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historical level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment.

Table 5.14 summarises the EDA estimates of the total numbers of yellow and silver eels in the Brittany EMU.

Table 5.14. Model outputs for EDA simulations of the Brittany EMU

Model output	Value
Total water surface (km²)	38.08
Average number of yellow eel per 100 m²	9.487
Average number of silver eel per 100 m²	0.474
Total number of yellow eel	3612147
$N_{current}$: Total number of silver eel	180607
$B_{current}$(t=2010) in kg	144485
B_{best} (t=2010) in kg	8093355

Changes made during the tuning process for the application of EDA to the Brittany EMU

In the second application phase, to evaluate Bcurrent, Bbest and Bpristine, the mean weight of each life stage (glass eel $\overline{w}_{glass}$, yellow eel $\overline{w}_{yellow}$ and silver eel $\overline{w}_{silver}$) and the anthropogenic mortalities on glass eel Y_{glass}, yellow eel Y_{yellow} and silver eel Y_{silver} have been used.

The model parameters used to evaluate Bcurrent, Bbest and Bpristine are given in Table 5.15. Table 5.16 summarises the EDA estimates of the total numbers of yellow and silver eels in the Brittany EMU.

Table 5.15. Data input for silver eel estimation for the Brittany EMU

With Y_{glass}(t= $2010-\tau$), Y_{yellow}(t= $2010-\tau+\lambda_{yellow}$) and Y_{silver}(t=**2010**) in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	12 (Briand et al. 2008) or 9 (Lambert, 2008)	4	0.33 (Briand et al. 2008)	100 (Briand et al. 2008)	800 (Briand et al. 2008)	15300 (Vilaine) + 2 000 (elsewhere) (in 1998)	1297 (in 2002)	0 (in 2010)

Table 5.15. Model outputs for EDA simulations of the Brittany EMU

Model output	Value
Total water surface (km²)	38.08
Average number of yellow eel per 100 m²	9.487
Average number of silver eel per 100 m²	0.474
Total number of yellow eel	3612147
$N_{current}$: Total number of silver eel	180607
$B_{current}$(t=2010) in kg	144486
B_{best} (t=2010) in kg	377439
B0 ($B_0 = \overline{B_{best}\left(t<1980\right)}$ in kg	678624

5.11. Rhone-Méditerrannée EMU (France)

Overview

The Rhône – Méditerrannée EMU can be separated into three compartments:

- the Rhone basin (97 800 km²)
- many river basins flowing directly to the Mediterranean Sea (25 000 km²)
- Mediterranean lagoons (550 km²)

The Mediterranean lagoons are particularly suitable habitats for eel.

Stock status

Eel density quickly decreases with the distance from the sea both in the Rhone basin (Figure 5.69) and in other Mediterranean basins.

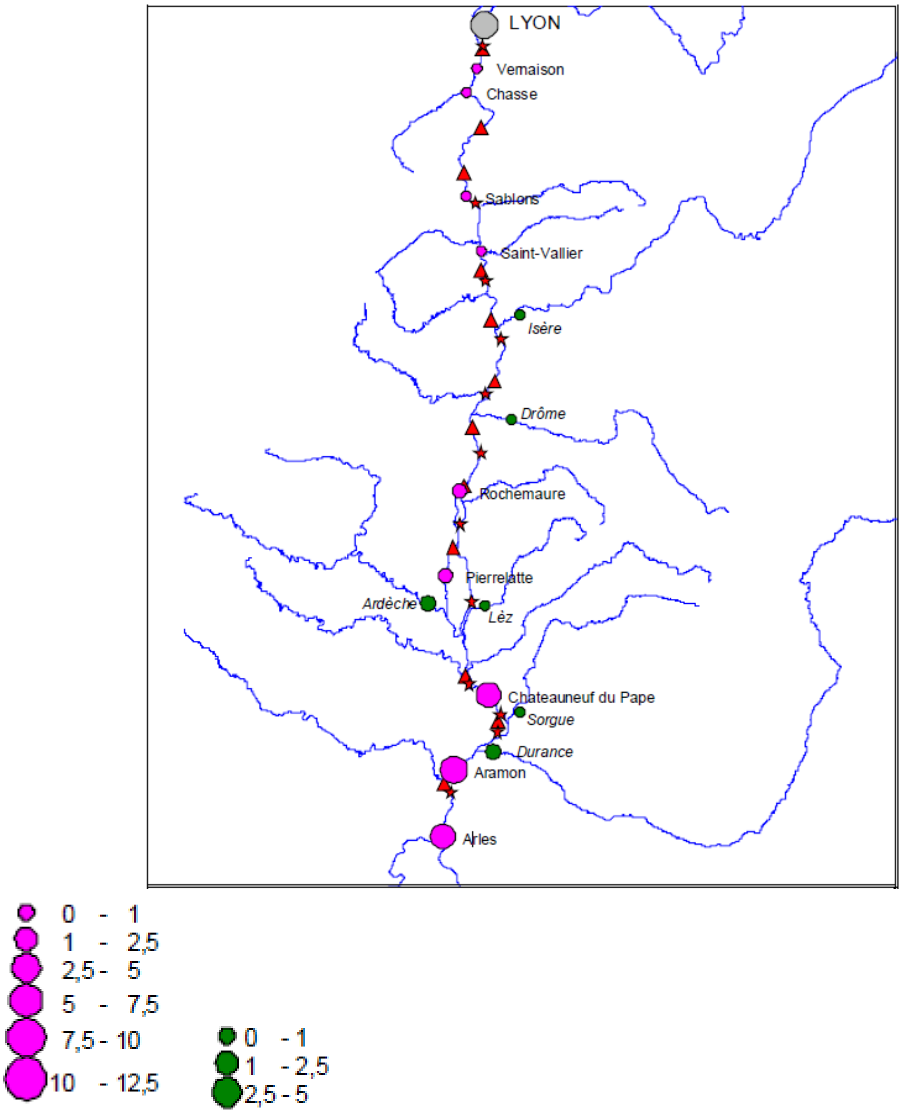


Figure 5.68. Eel densities (eel/100m²) in the Rhone basin. Pink: RCS data in the Rhone, river autumn 2007, EPA sampling ; green: other sampling in tributaries, mean densities between 2000 and 2004 ; red: main dams - source EMP Rhône – Méditerrannée EMU

Impact on eel production

Fishery

Glass eel fishing is forbidden in the Rhône – Méditerrannée EMU.

The main yellow eel fishery takes place in Mediterranean lagoons, though catches have declined in recent years. Recent fishery data are available for three areas but as they relate to three different time periods they are not immediately comparable. The mean average annual landing for Languedoc-Rousillon lagoons from 2003 to 2005 were estimated at 512 t (Cepralmar 2003, 2004, 2005). In 2007, catches in PACA lagoons were estimated at 111 t (Pôle relais lagunes méditerranéennes, 2009). For 2008, Demenache *et al.*, (2009) estimated the production of yellow eels in continental French Mediterranean coast has dropped further to about 294 t (precision between 211/395t).

In the Rhone river itself, fishing in the estuary caught 17.5 t of yellow eels in 2007 (MRM, 2008). In comparison, the catch from anglers fishing the freshwater parts of the river in 2007 was about 0.5 t (Onema data). No estimate is available of the number of anglers associated with this 2007 catch, but in 1999, 500,000 anglers fished in the EMU, of which about 5 to 10% were thought to target eels (Barral, 2000). This number is likely to have declined somewhat in recent years due to restrictions imposed because of the levels of PCBs found in eels in the River Rhone.

The main silver eel fishery takes place in Mediterranean lagoons, with an average annual catch over 2003 to 2005 of around 241 t (Cepralmar 2003, 2004, 2005). In comparison, marine fishers in the downstream part of the Rhone river caught 7 t of silver eels in 2007 (MRM, 2008).

Dams

A high proportion of obstructions in the mainstream of the River Rhone are equipped to produce hydropower. It is estimated that the mean survival rate of silver eel after passing all the dams with turbines is 21% for those coming from the Lyon and the Saone rivers, 51% for those coming from the Drôme tributary, and 72% for silver eels coming from the Ardèche tributary.

Restocking

There is a tradition of eel restocking in the upper part of the Rhone basin (Barral, 2000), with about 500 kg of eel have been restocked each year between 1978 and 2000. Restocking material is sourced from a variety of origins and the eels can be as long as 40 cm.

The Rhone dataset is considered as poor for the purposes of POSE.

5.12. Application of the EDA to the Rhone data set

The general method of applying the EDA model to an EMU is described in Chapter 4. Here, we describe the work achieved to apply the model to the Rhone dataset, and the resultant predictions of yellow eel stock and silver eel production.

In the analysis, 2890 electrofishing operations on 1361 electrofishing stations from April to November from 1984 to 2008 were selected (Figure 5.69). These sites were considered in relation to their distribution over the time period (Figure 5.70), their elevation and distances from the sea and source (Figure 5.71), and the cumulative number of dams against distances from the sea and source, and site elevation (Figure 5.72).

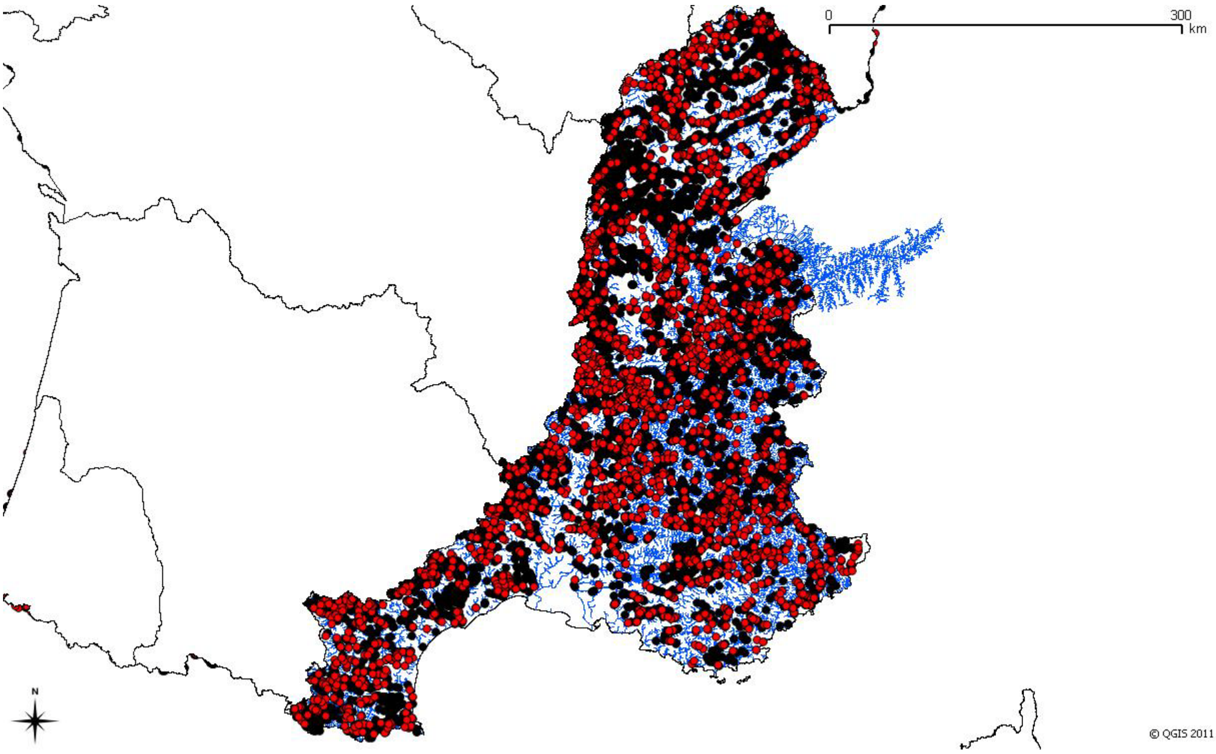


Figure 5.69. Sampling sites (in red) in the Rhone Emu catchment on the CCM river network (in blue) with the dams location (in black).

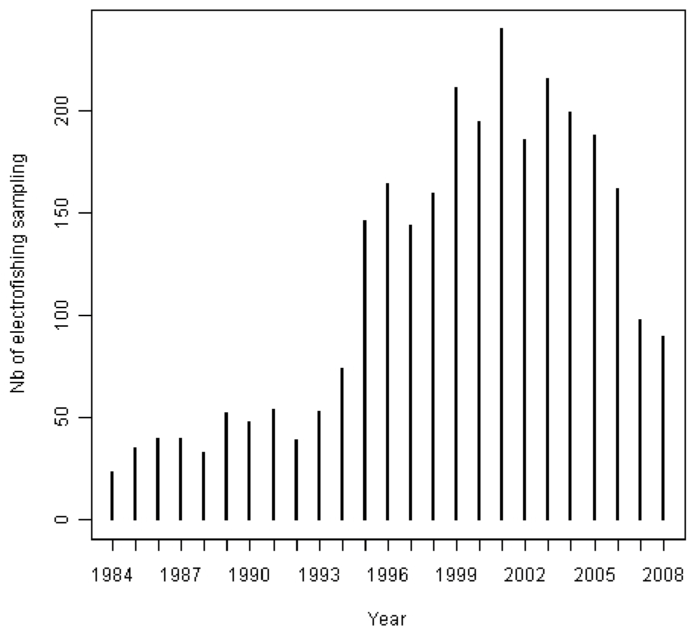


Figure 5.70. Number of electrofishing sampling per year in the Rhone Emu.

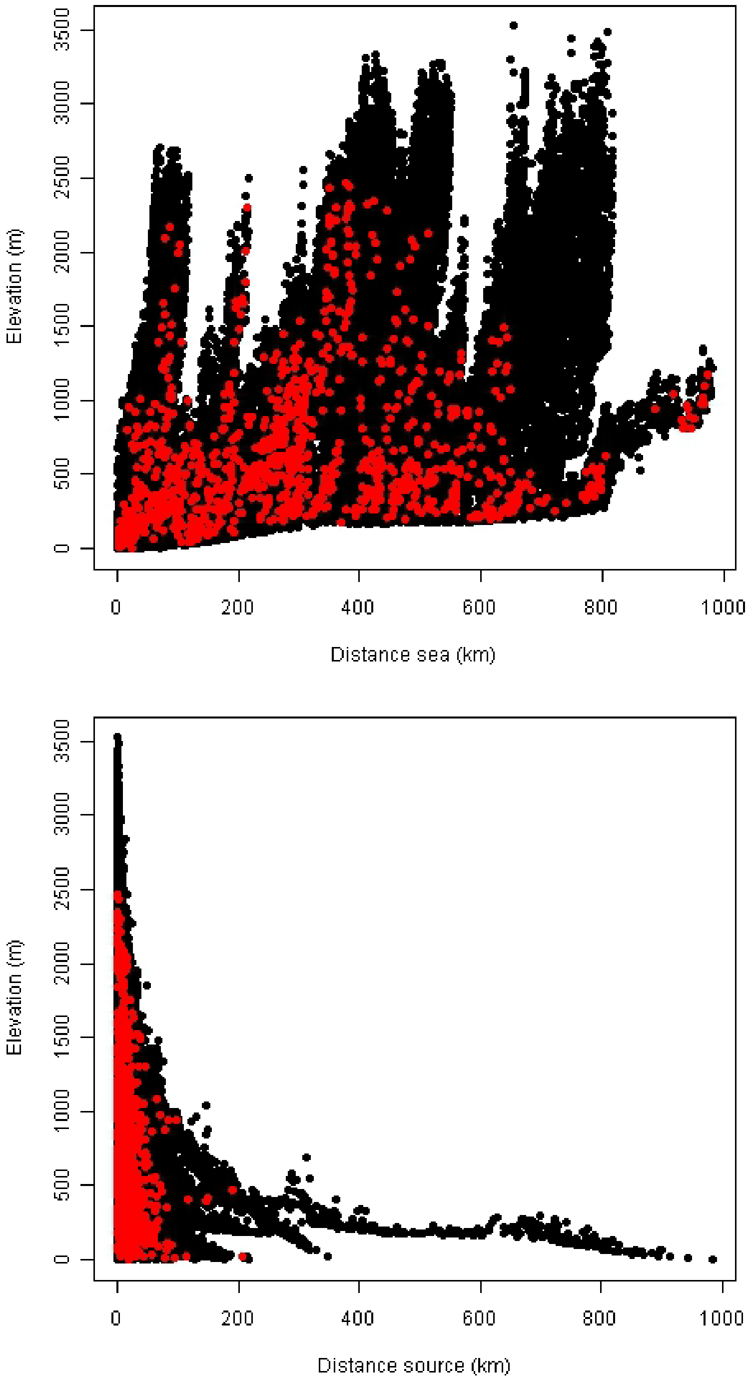
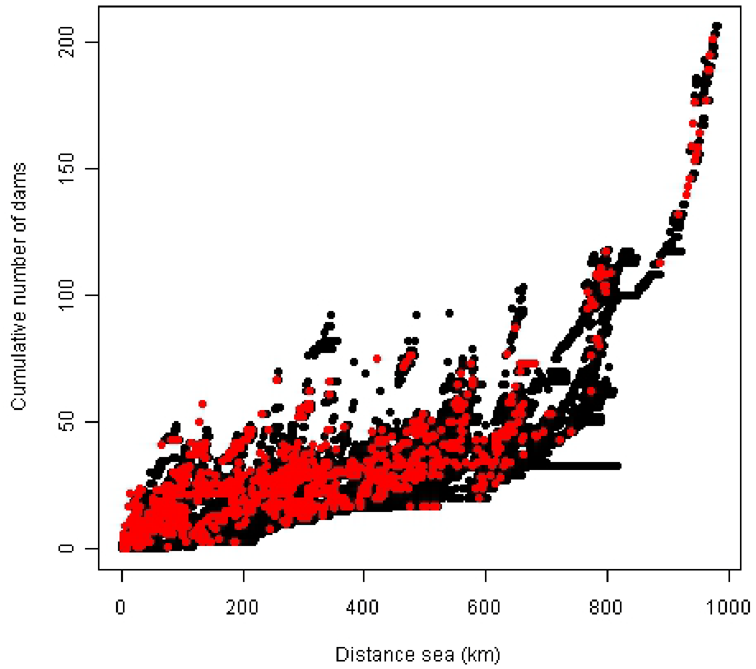
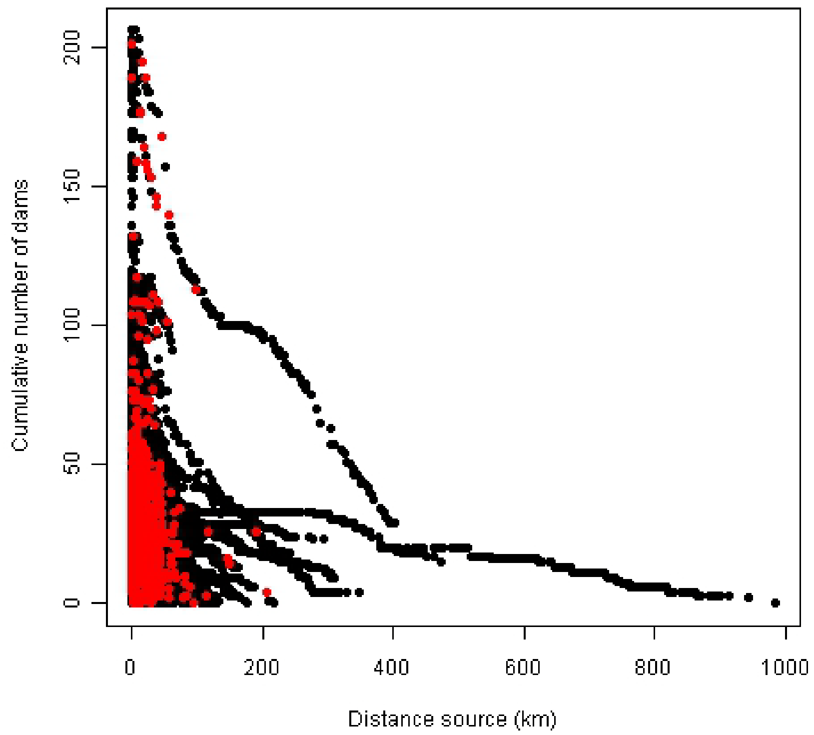


Figure 5.71. Sampling sites (in red) in the Rhone Emu catchment on the CCM river network (in black) depending on the elevation and the distance from the sea or from the source.



Cumulative number of dams
0
50
100
150
200
0
200
400
600
800
1000
Distance sea (km)



Cumulative number of dams
0
50
100
150
200
0
200
400
600
800
1000
Distance source (km)

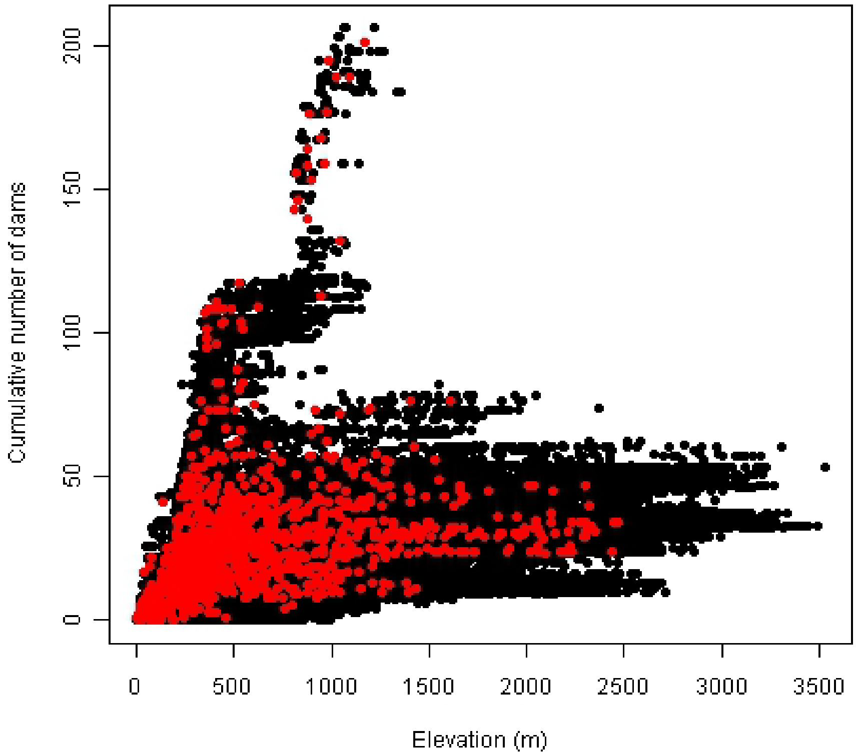


Figure 5.72. Sampling sites (in red) in the Rhone Emu catchment on the CCM river network (in black) depending on the cumulative number of dams and the distance from the sea or from the source and the elevation.

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CCM data set for the Rhone EMU provided information for each survey site on the distances to sea and source, relative distance, mean slope and elevation, altitudinal gradient, the area of land drainage upstream, mean annual temperature and rainfall, land use cover (urban, agricultural and no impact) , and the number of dams downstream of the site. The cumulative number of dams from sea to site was used to characterise the obstacle pressure to upstream migrating eels. In total there were 5469 dams distributed throughout the Rhone EMU.

A combination of 17 variables (year, month and a set of 15 variables) was tested. Figures 5.73 and 5.74 show that there are several variables with tightly correlated predictors.

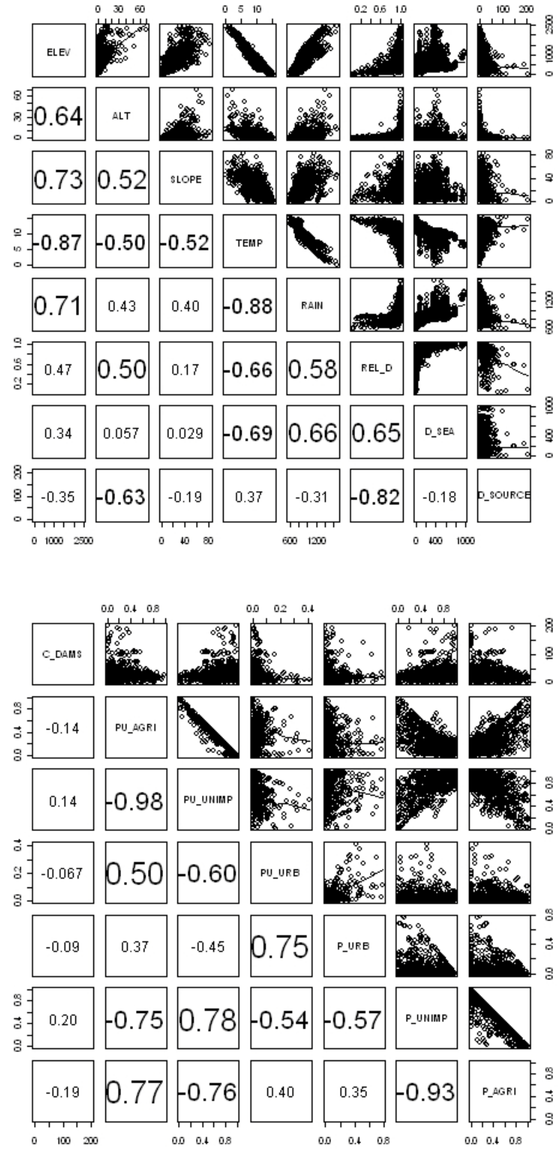


Figure 5.73. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With ELEV: elevation mean, ALT: altitudinal gradient, SLOPE: slope mean, TEMP: temperature mean, RAIN: rain mean, REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, C_DAMS: cumulative number of dams, PU_AGRI : upstream percent in agricultural use, PU_UNIMP: upstream percent in unimpact land use (p_up_no_impact), PU_URB: upstream percent in urban use, P_URB: local percent in urban use, P_UNIMP: local percent in unimpact land use (p_no_impact), P_AGRI: local percent in agricultural use. The font size of the cross-correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

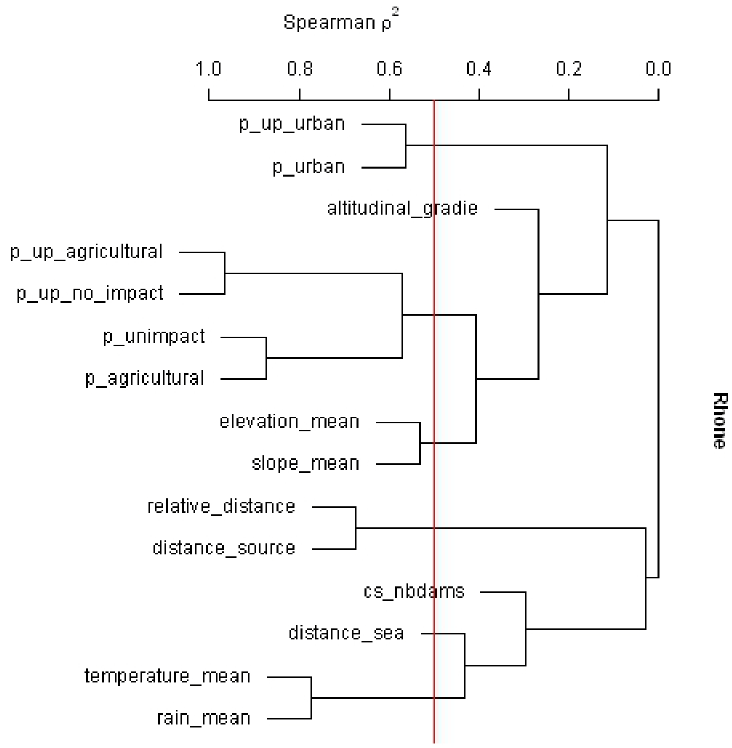


Figure 5.74. Dendrogram obtained by hierarchical cluster analysis of 15 candidate predictors for Rhone dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ^2=0.5.

According to the threshold of 0.5 for ρ^2, the following variables should be grouped:

- elevation and slope
- temperature and rain
- relative distance and distance from the source
- p_agricultural, p_unimpact
- p_up_agricultural and p_up_no_impact

A separate entry was used for the pairs of variables with significant relationships (Figure 5.73) and for the 5 groups of variables described above (Figure 5.74), to avoid spurious correlations.

The combination of the different groups resulted in 2160 models to be tested.

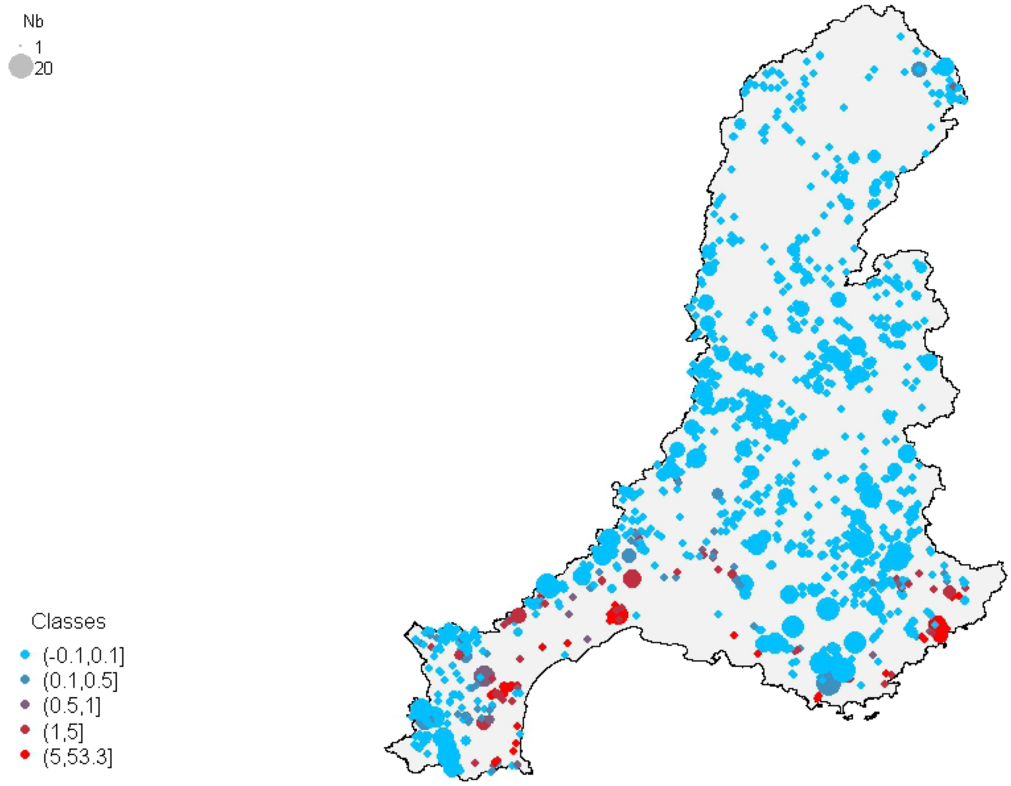


Figure 5.74. Map of density observed densities. All years.

Model Testing Procedure

We calibrated statistical models with a Generalized Additive Model (Hastie and Tibshirani, 1990) to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/). EDA model is based on a delta-gamma model (Stefánsson, 1996) which combines two generalized additive model (GAM). There are three steps of modelling:

- a presence/absence model (delta model) based on a GAM with a binomial distribution and a logit link to determine the probability of a positive catch;
- a density model with the positive data (gamma model) using a GAM with a gamma distribution and logarithm link to determine the level of positive catch; and,
- the multiplication of the two previous models (delta-gamma model).

The GAMs were computed with the library ‘gam’ (Hastie, 2010) with a cubic spline smoother (3 degree of freedom) for each environmental variable. The delta-gamma (ΔΓ) generalized additive models explain a large portion of the variability in eel abundance data, as there are many occasions where densities are null.

Presence-absence model (Delta model)

The results for tests of 2160 models are summarised in Table 5.15. The best density model is selected by the Akaike's Information Criterion (AIC) with a lower AIC indicating a better fit (Akaïke 1974, Sakamoto et al. 1986). The AIC function indicates a better fit of the response variable at the year and the month of electrofishing sample with elevation, distance from the sea, cumulative number of dams, upstream percent of urban use, upstream percent of unimpacted land use (model 1 in Table 5.15). For this model the kappa = 0.637 ± 0.02. GAM explained 52 % of the deviance of the abundance of the yellow eel. Six explanatory variables are significant (Table 5.16): year, month, elevation, distance from sea, upstream percent of urban use and upstream percent of unimpacted land use.

Table 5.15. Model selection results using Akaike's information criterion (AIC) for presence-absence model analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	elevation mean	distance sea	distance source	relative distance	cs_nbdams	cs_score	p_up_urban	p_up_agricultural	p_up_no_impact	AIC s=3
1	x	x	x	x			x		x		x	1147.6
2	x	x	x	x	x				x		x	1148.3
3	x		x	x	x				x		x	1148.6
4	x		x		x		x		x		x	1149.4
5	x	x	x	x			x		x	x		1149.6
6	x	x	x	x	x				x	x		1151.2
7	x		x	x	x				x	x		1151.5
8	x		x	x			x		x	x		1151.6
9	x	x	x	x			x				x	1152.4
10	x	x	x	x					x		x	1154.0

Model Goodness of fit (presence-absence model)

The best presence-absence model selected (a binomial model with a logit link and a cubic spline smoother – s =3) is: d != 0 ~ `s(year,3)+ s(month,3)+ s(elevation,3)+ s(distance sea,3)+ s(cs_nbdams,3)+ s(p_up_urban,3) + s(p_up_no_impact,3)`

matrice confusion

observed

predicted	1	0
1	253	108
0	126	2398

correctly predicted 0.92

present correctly predicted 0.67

absents correctly predicted 0.96

Table 5.16. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar Chisq	P(Chi)	
(Intercept)	1				
s(annee, 3)	1	2	16.289	0.0002904	***
s(month, 3)	1	2	6.744	0.0343262	*
s(elev_mean, 3)	1	2	53.627	2.264e-12	***
s(distance_sea, 3)	1	2	93.688	< 2.2e-16	***
s(cs_nbdams, 3)	1	2	5.592	0.0610425	.
s(p_up_urban, 3)	1	2	10.378	0.0055770	**
s(p_up_no_impact, 3)	1	2	12.108	0.0023492	**

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

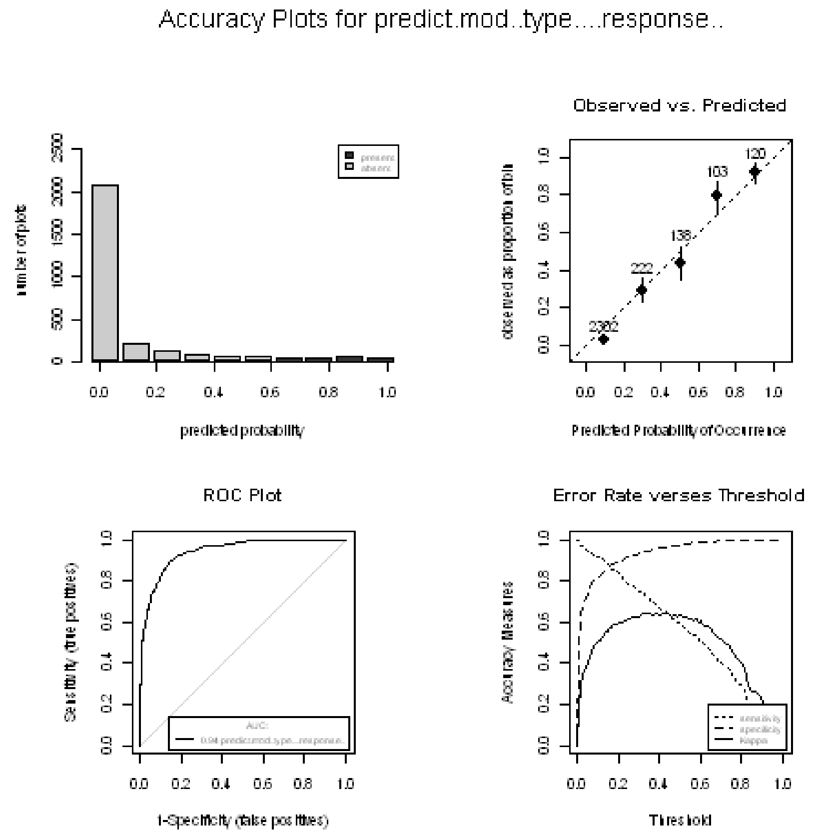


Figure 5.75. Model quality and threshold selection graphs for presence-absence model selected with a histogram plot (upper left), a calibration plot (upper right), a ROC plot with the associated Area Under the Curve (AUC) (lower left), and an error rate verses threshold plot (lower right).

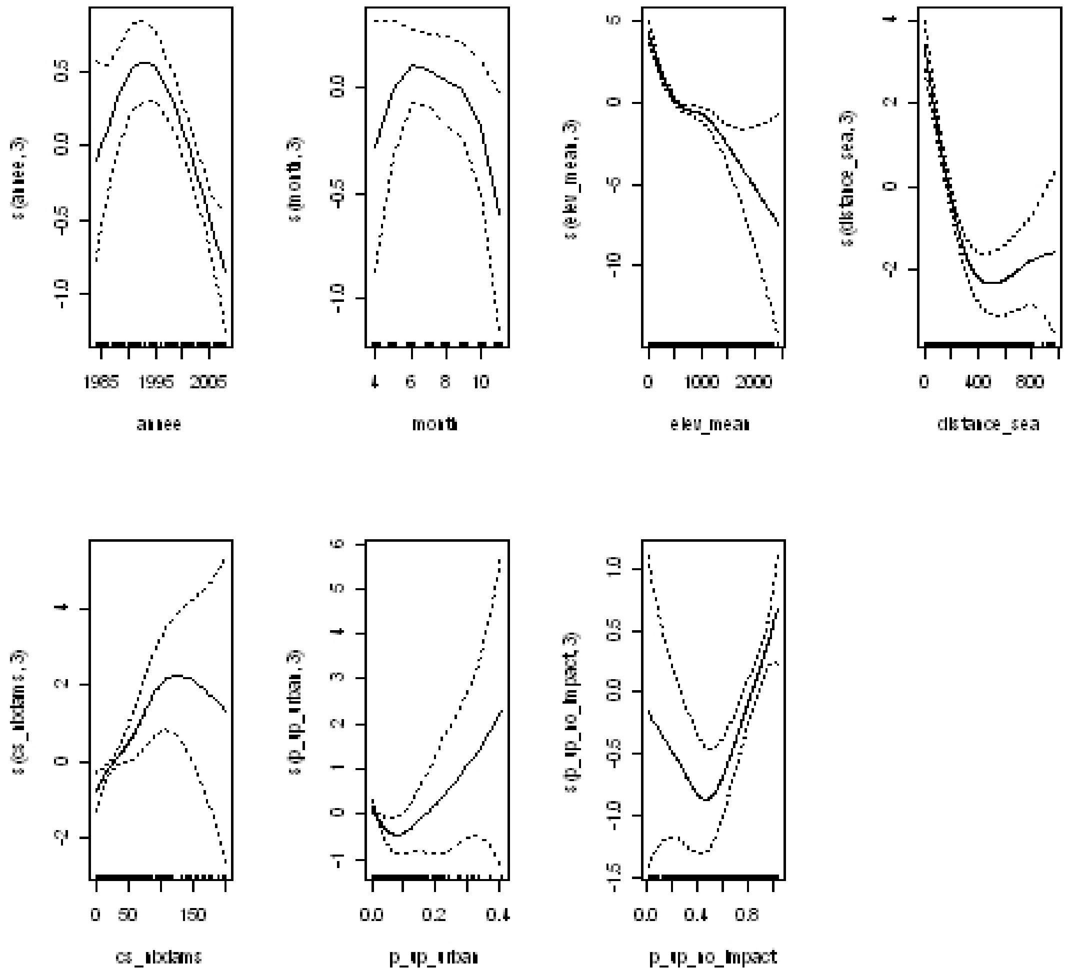


Figure 5.76. Response curves of each variable included in the generalized additive model (GAM) for the presence/absence model for Rhone EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The residual of the presence absence model are plotted for each year (Figure 5.77). If the results of the fishing operation are in overplot (several year point on top of the other), they are slightly moved around their origin to show on the map. Several operations for the same year will remain in overplot. The residuals correspond to observed -predicted values, hence a blue spot (negative value) will indicate a predicted value larger than the observed one, and a red point (positive value) a predicted value lower than the observed one.

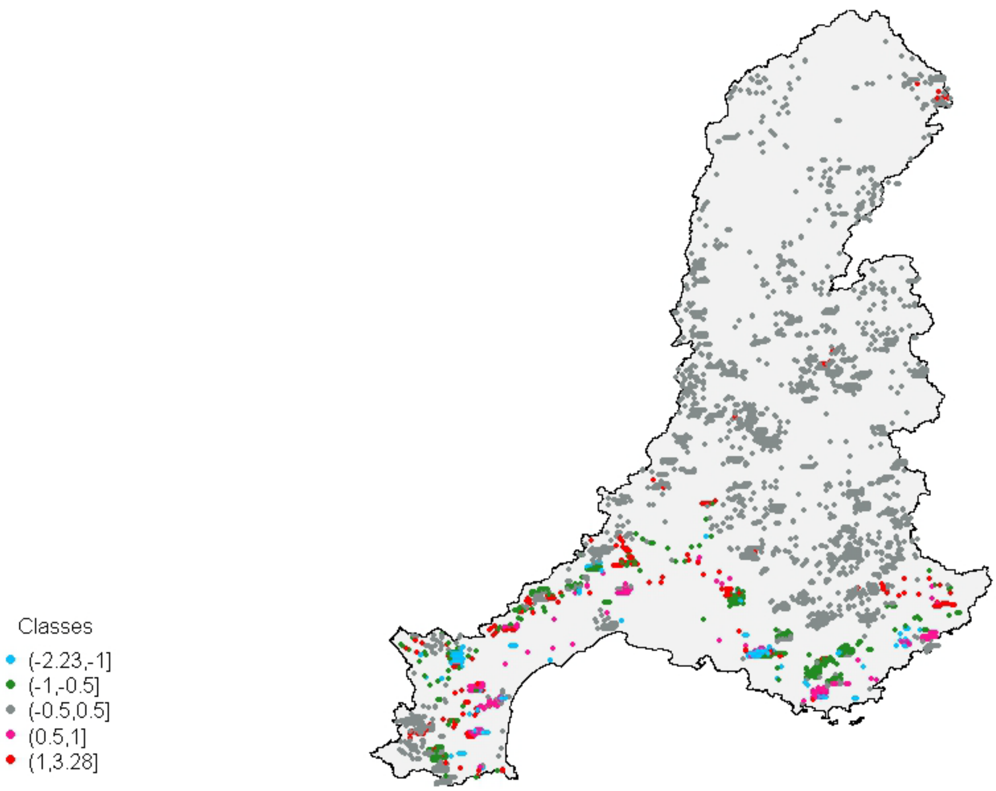


Figure 5.77. Residuals per year: presence-absence model.

In the next graph (Figure 5.78), the residuals are averaged for each station. A blue spot is a station with on average over the years predicted values far larger than the observation. The size of the point is related to the number of electrofishing operation in the station.

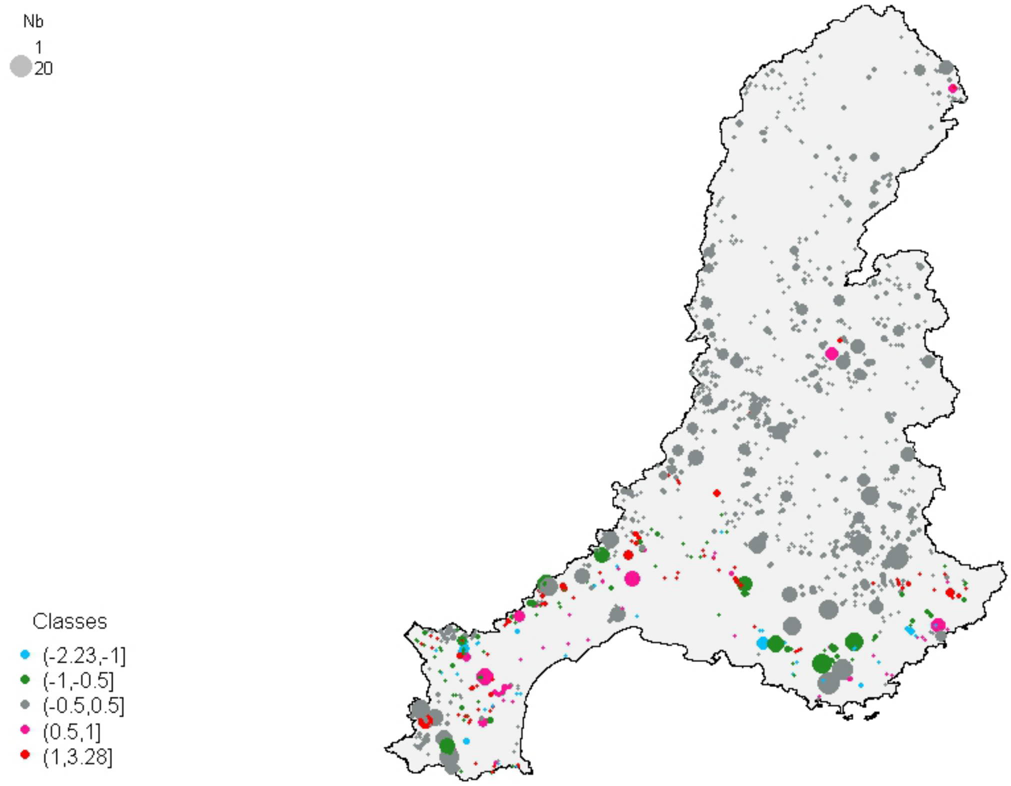
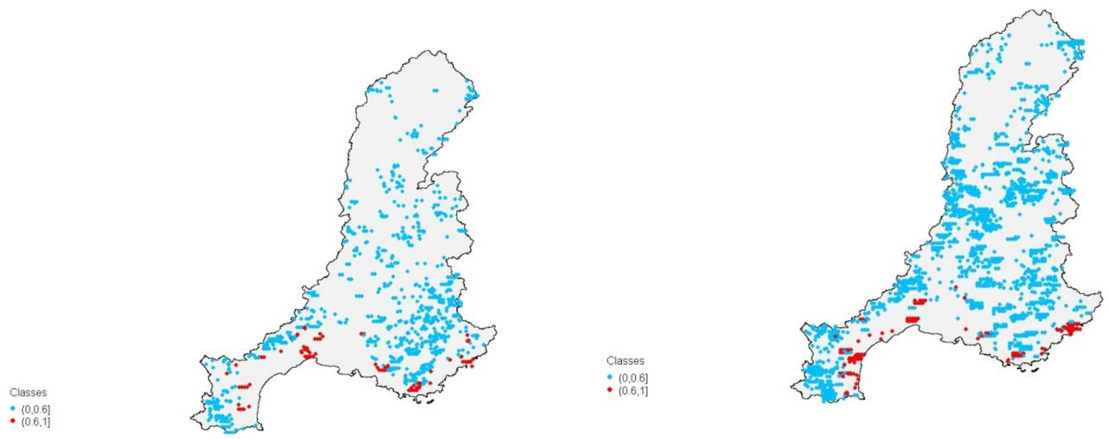


Figure 5.78. Mean Residuals per stations: presence-absence model.

The map of predicted value (Figure 5.79) from the model is computed before and after the break point in the year variable response trend. The threshold chosen for the response is the value giving the best Kappa, i.e. trade off between the lowest number of wrongly predicted presence (false positive), and wrongly predicted absence (false negative).



(a) (b)

Figure 5.79. Predicted value: presence-absence model year < 1998 (a), Predicted value: presence-absence model year ≥ 1998 (b).

Density model

The density model is applied only to positive value. It is a gam model with a gamma distribution and a logarithm link. 2160 models have been tested.

Model results for density model are summarized in Table 5.17. The AIC function indicates a better fit of the response variable at the year of electrofishing sample with elevation, distance to the sea, distance to the source, local percent of urban use and local percent of agricultural land use). GAM explained 48% of the deviance of the abundance of the yellow eel. The effects of five explanatory variables in the model (year, elevation, distance to the sea, p_urban, p_agricultural) are significant (Table 5.18). The Spearman rank correlation between the observed values and the fitted values is statistically significant (ρ=0.68, p-value < 2.2 10^{-16}, Figure 5.80).

Table 5.17. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	elevation mean	temperature mean	distance sea	distance source	cs_nbdams	p_urban	p_agricultural	p_unimpact	AIC s=3
1	x		x		x	x		x	x		1459.9
2	x		x		x	x				x	1460.0
3	x		x		x		x	x	x		1460.1
4	x	x	x		x		x	x	x		1460.9
5	x	x	x		x	x				x	1461.9
6	x	x	x		x			x	x		1461.9
7	x			x	x	x		x	x		1462.8
8	x		x		x			x	x		1463.0
9	x	x	x		x			x	x		1463.9
10	x		x		x					x	1464.5

Model Goodness of fit (density model)

The best density model selected is: d ~ s(annee,3)+ s(elevation_mean,3)+ s(distance_source,3)+ s(distance_sea,3)+s(p_urban,3)+s(p_agricultural,3)

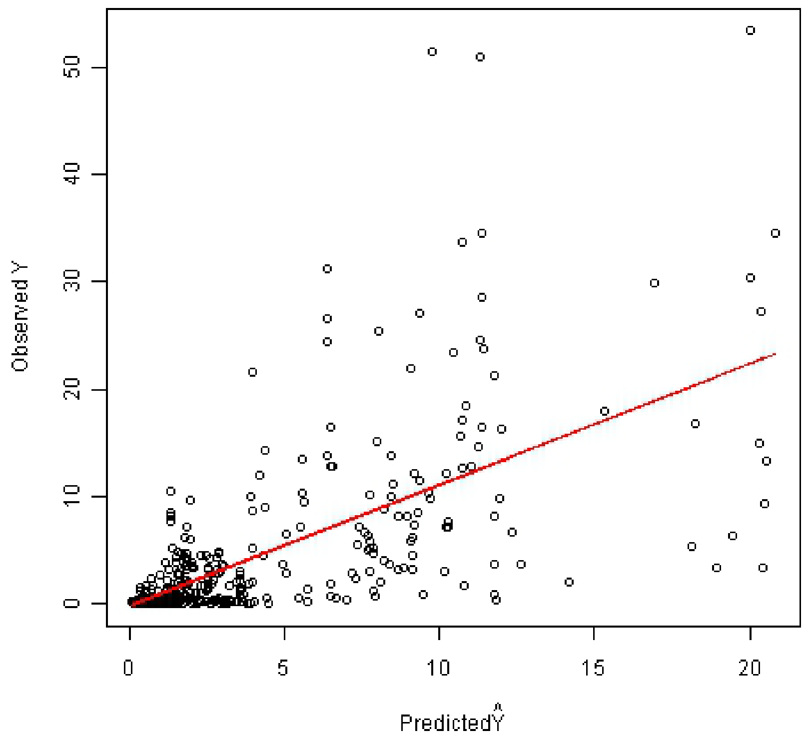


Figure 5.80. Observed vs. predicted regression scatter plot.

Table 5.18. Table of effects, DF for terms and Chi-squares for non-parametric effects.

Df Npar Df Npar F Pr(F)
(Intercept) 1
s(annee, 3) 1 2 7.832 0.0004686 ***
s(elev_mean, 3) 1 2 8.835 0.0001796 ***
s(distance_source, 3) 1 2 1.846 0.1593303
s(distance_sea, 3) 1 2 53.900 < 2.2e-16 ***
s(p_urban, 3) 1 2 3.360 0.0358078 *
s(p_agricultural, 3) 1 2 5.944 0.0028864 **

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

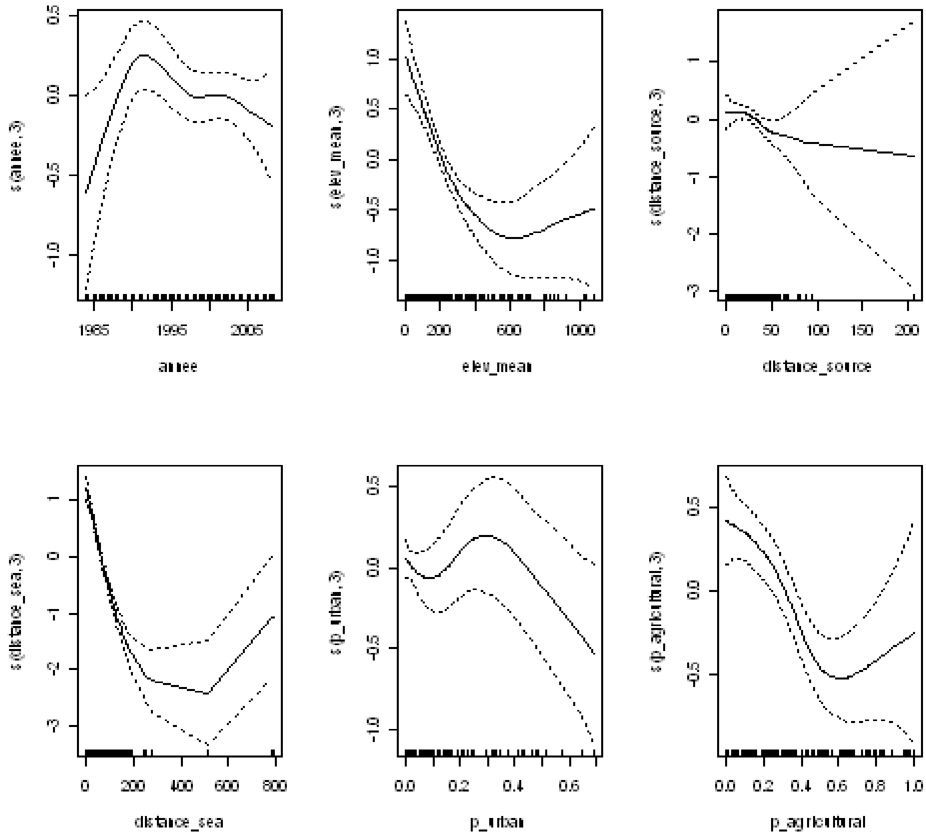


Figure 5.81. Response curves of each variable included in the generalized additive model (GAM) for the density model for Crepe EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The map below (Figure 5.82) shows the mean residuals for operations where positive densities were observed. The sign is calculated according to predicted – observed. The size of the points gives an indication of how many electrofishing operations occurred at that point and thus of the 'weight' of that station in the model.

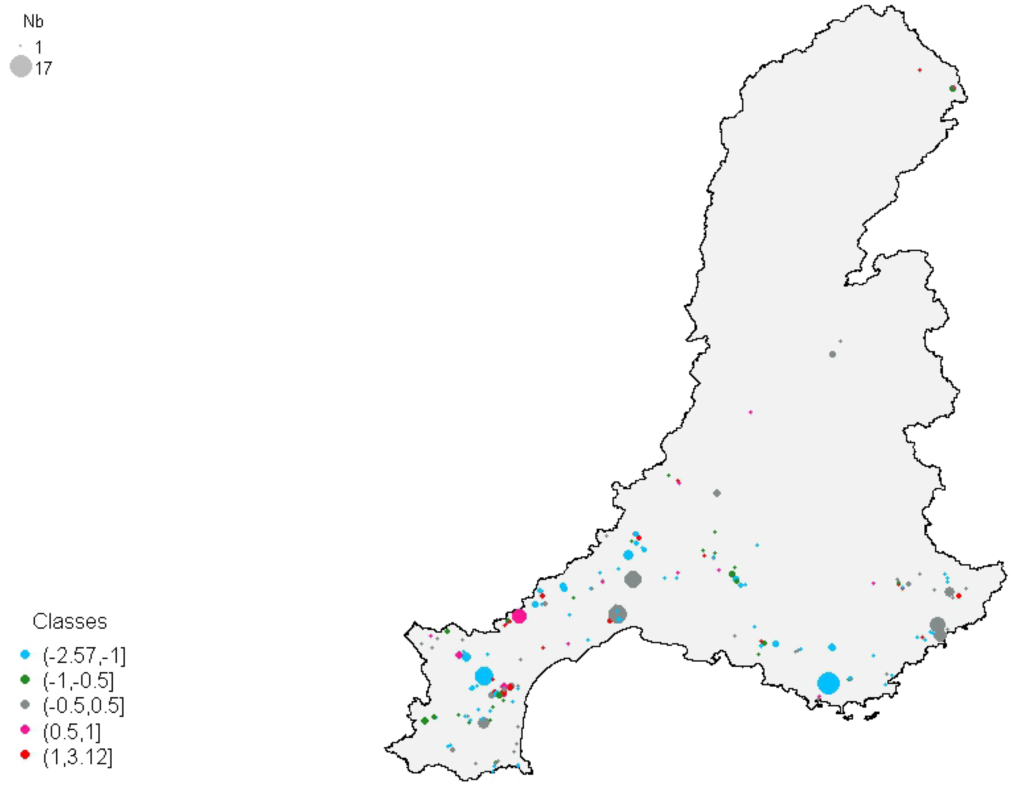


Figure 5.82. Mean residuals per stations: density model.

The next map (Figure 5.83) gives the same information but residuals are ‘jittered’ around the point. There is no clear spatial trend in the presence of large positive or negative residuals.

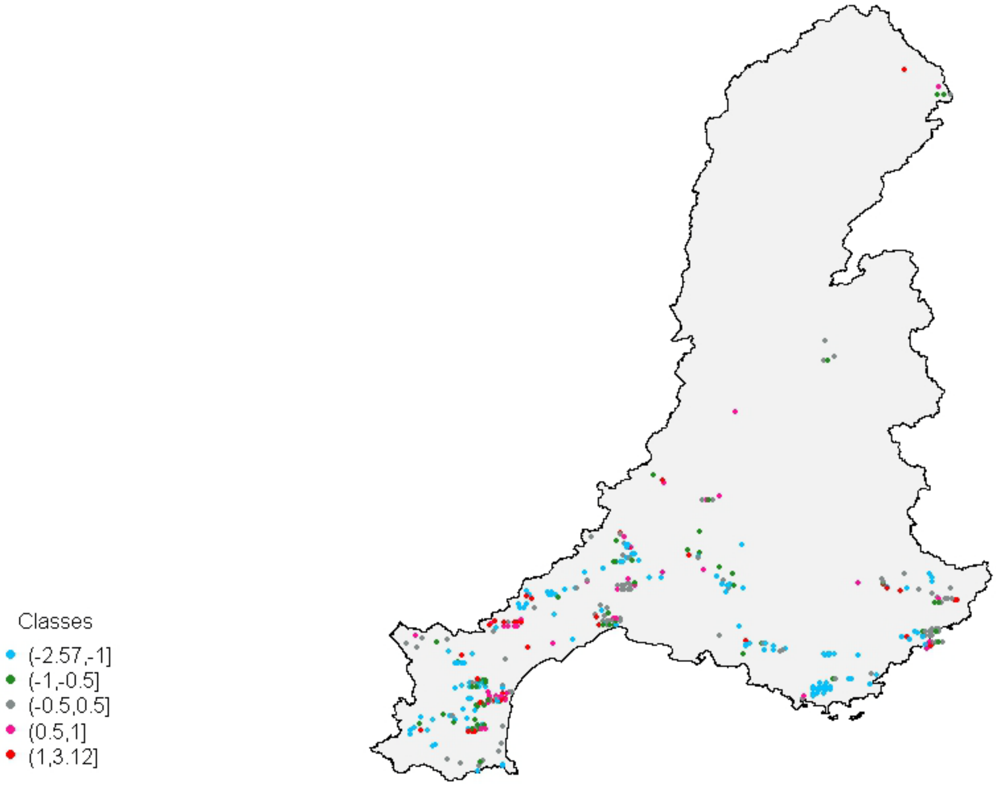
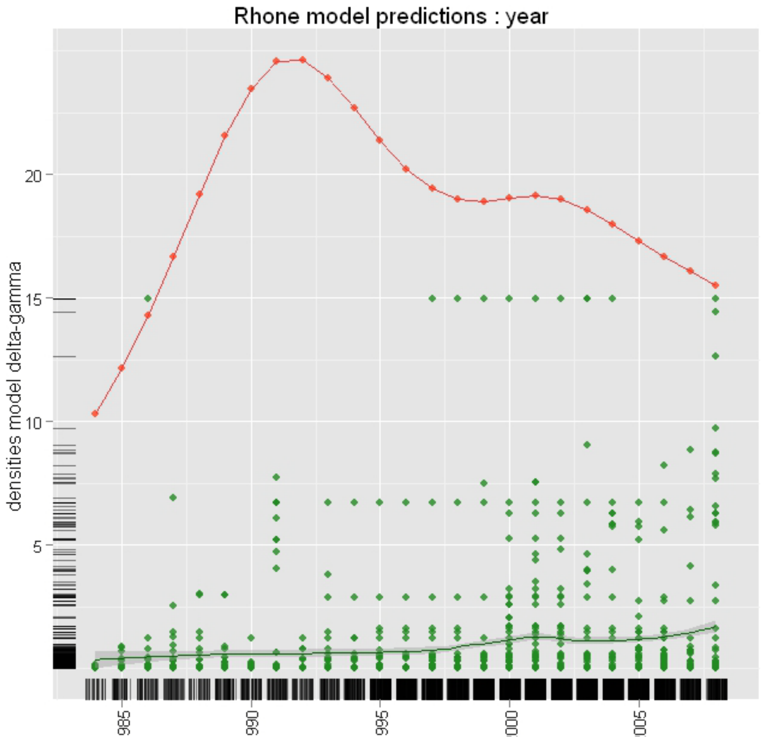


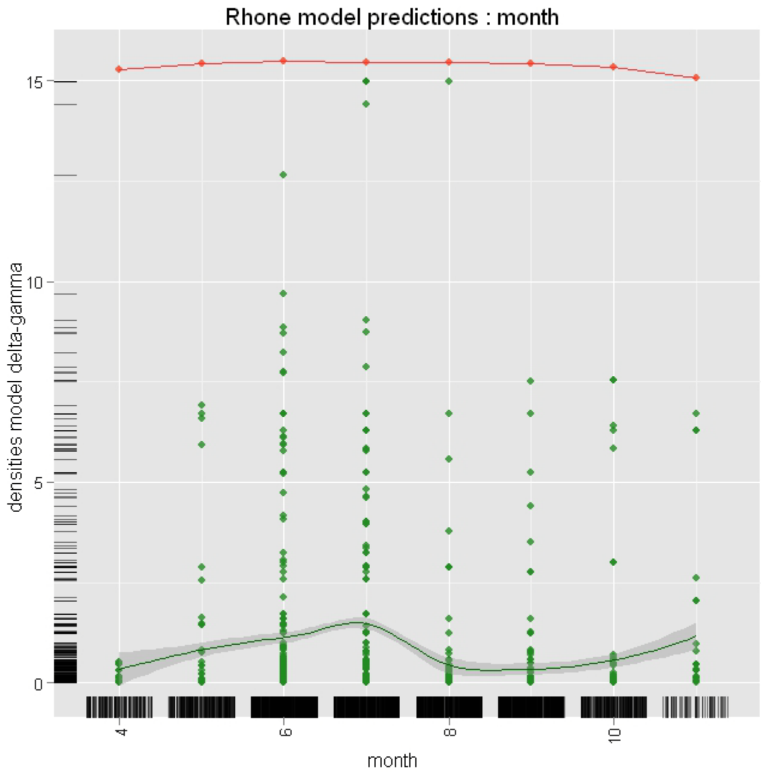
Figure 5.83. Residuals per year: density model.

Final model

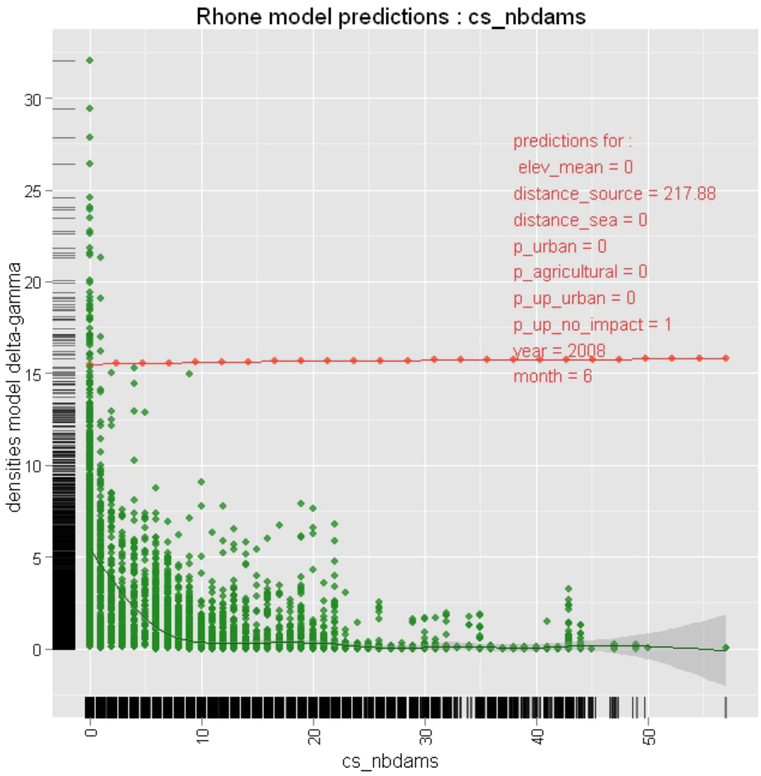
The final model is the product of the delta model by the gamma model. The delta model is the probability of having a positive density. The gamma model is the level of a density for positive values. The model is used to predict densities on all the river stretches of the EMU. The following figures (not numbered) give the trend (red curve) of an effect given that all other parameters are fixed (see figure for the value of fixed parameters) as well as the predicted density of each river stretch of the CCM (green points) and its average along the examined effect (green curve). The rugs on the scale have been moved slightly to indicate the density of observation.

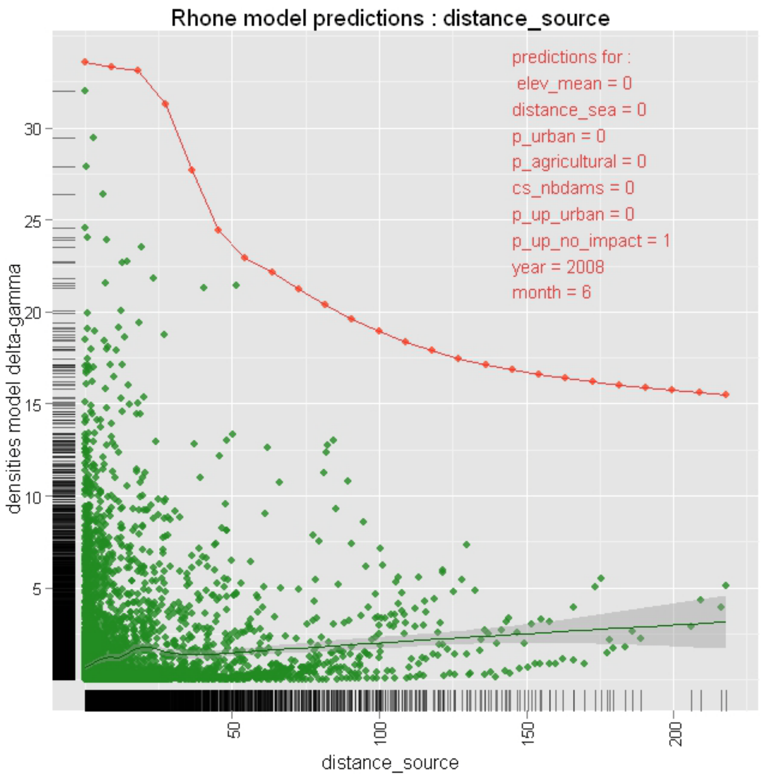


Rhone model predictions : year
densities model delta-gamma
20
15
10
5
985
990
995
000
005



Rhone model predictions : month
densities model delta-gamma
15
10
5
0
4
6
8
10
month


Rhone model predictions : cs_nbdams
predictions for :
elev_mean = 0
distance_source = 217.88
distance_sea = 0
p_urban = 0
p_agricultural = 0
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
cs_nbdams


Rhone model predictions : distance_source
predictions for :
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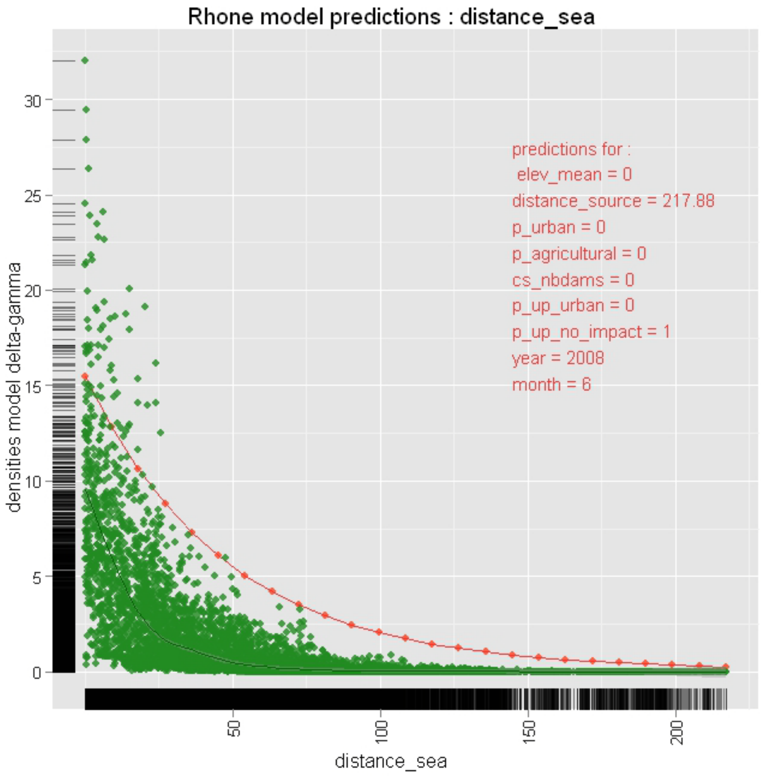
distance_sea = 0
p_urban = 0
p_agricultural = 0
cs_nbdams = 0
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
distance_source


Rhone model predictions : distance_sea
predictions for :
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distance_source = 217.88
p_urban = 0
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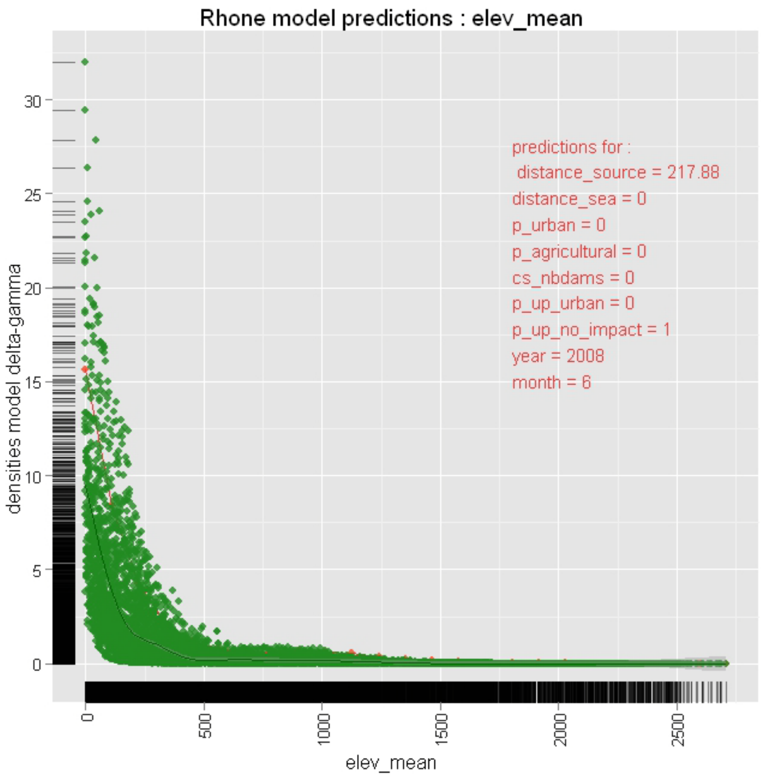
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
distance_sea


Rhone model predictions : elev_mean
predictions for :
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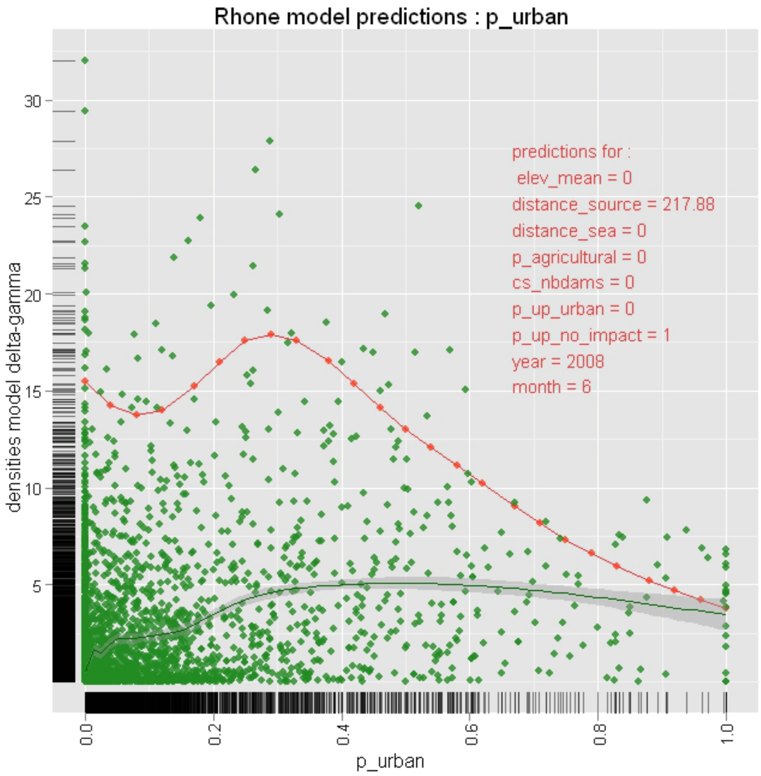
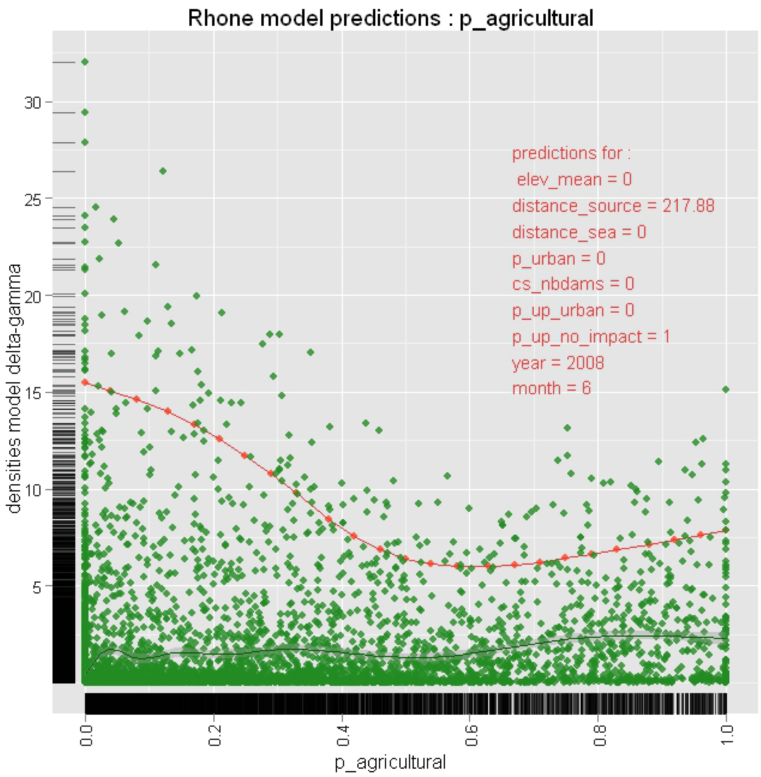
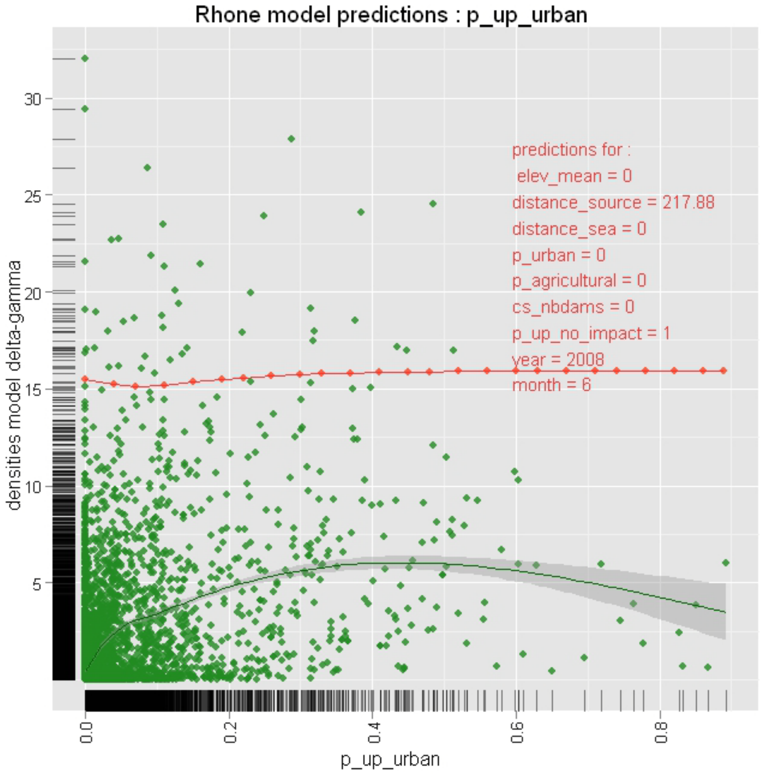
distance_sea = 0
p_urban = 0
p_agricultural = 0
cs_nbdams = 0
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6
densities model delta-gamma
elev_mean


Rhone model predictions : p_urban
densities model delta-gamma
p_urban
predictions for :
elev_mean = 0
distance_source = 217.88
distance_sea = 0
p_agricultural = 0
cs_nbdams = 0
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6


Rhone model predictions : p_agricultural
densities model delta-gamma
p_agricultural
predictions for :
elev_mean = 0
distance_source = 217.88
distance_sea = 0
p_urban = 0
cs_nbdams = 0
p_up_urban = 0
p_up_no_impact = 1
year = 2008
month = 6



The following figures (Figure 5.84 and 5.85) give the yellow and silver eel production along the examined variable. On the upper corner they also give the cumulated wetted area along the same variable that supports that production.

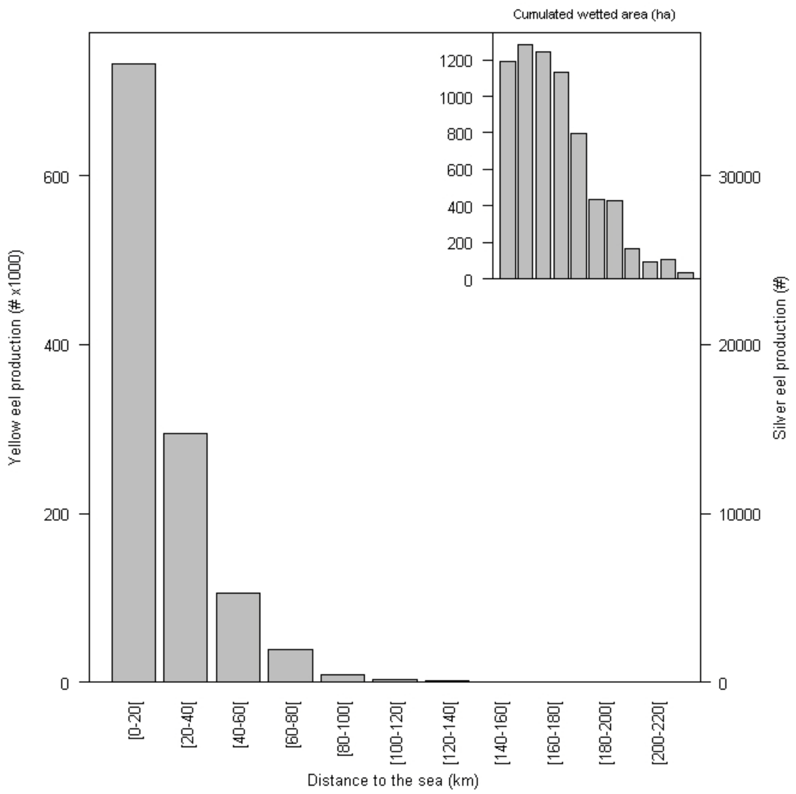


Figure 5.84. Yellow and silver eel production along the distance from the sea.

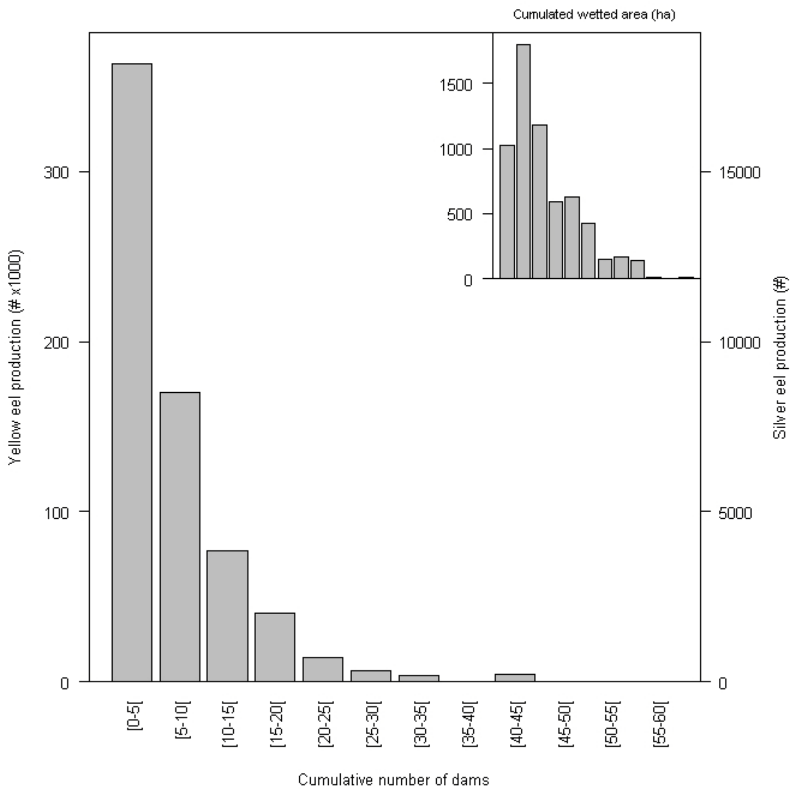


Figure 5.85. Yellow and silver eel production along the cumulative number of dams.

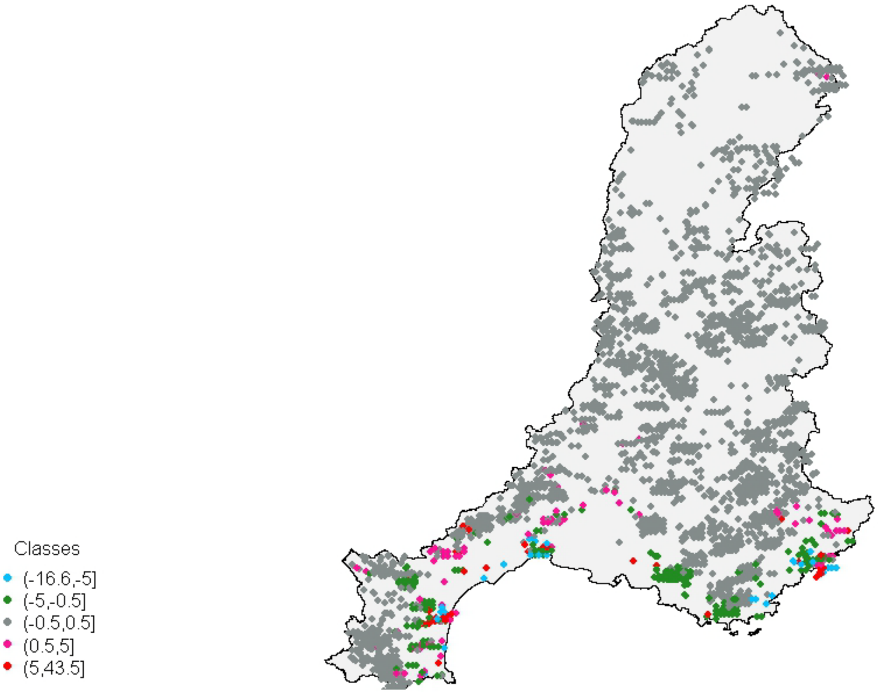


Figure 5.86. Map of mean residuals (predicted - observed), for the full model. Each year of data a point is shown with a small distance around the point.

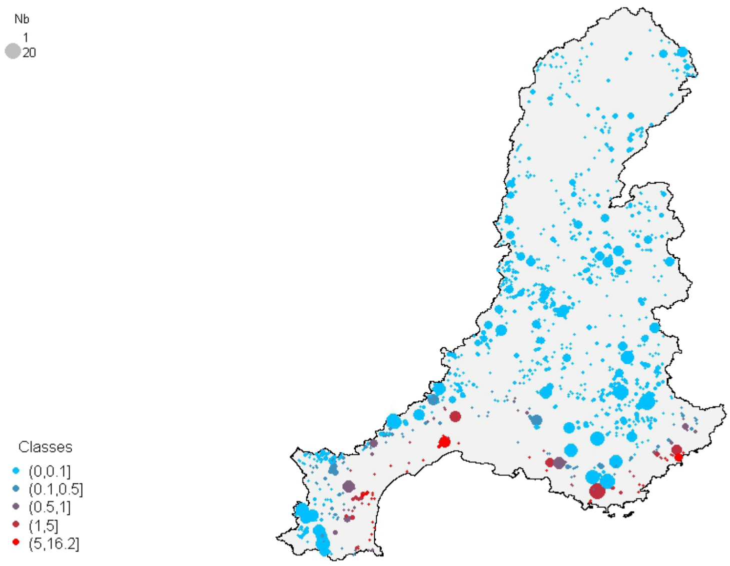


Figure 5.87. EDA predictions for the full model, the size model is given according to the number of electrofishing at that point.

EDA predictions for the Rhone EMU

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch with a conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou, 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al.* (2000) have estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA a constant conversion rate of 5% was chosen as a default value. This constant rate conversion of eel in equivalent silver eel is based on the assumption of density-independent biological processes.

This potential escapement in number in then converted into biomass with the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for the Rhone EMU. The current biomass ($B_{current}$) is calculated by subtracting the anthropogenic mortalities on silver eel to the potential escapement in biomass.

Best achievable escapements (B_{best}) can be calculated by the current biomass artificially forced to null anthropogenic impact (no dam, land use mortality to "no impact" and silver eel catch to 0) added with silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historical level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment. Table 5.19 summarises the EDA estimates of the total numbers of yellow and silver eels in the Rhone EMU.

Table 5.19. Model outputs for EDA simulations of the Rhone EMU

Model output	Value
Total water surface (km²)	69.06
Average number of yellow eel per 100 m²	1.718
Average number of silver eel per 100 m²	0.086
Total number of yellow eel	1185883
$N_{current}$: Total number of silver eel	59294
$B_{current}$(t=2010) in kg	47435
B_{best} (t=2010) in kg	52329

Changes made during the tuning process for the application of EDA to the Rhone EMU

In the second application phase, to evaluate Bcurrent, Bbest and Bpristine, the mean weight of each life stage (glass eel $\overline{w}_{glass}$, yellow eel $\overline{w}_{yellow}$ and silver eel $\overline{w}_{silver}$) and the anthropogenic mortalities on glass eel Y_{glass}, yellow eel Y_{yellow} and silver eel Y_{silver} have been used.

The model parameters used to evaluate $B_{current}$, B_{best} and $B_{pristine}$ are given in Table 5.20. Table 5.21 summarises the EDA estimates of the total numbers of yellow and silver eels in the Rhone EMU.

Table 5.20. Data input for silver eel estimation for the Rhone EMU

With $Y_{glass}(t=2010-\tau)$, $Y_{yellow}(t=2010-\tau+\lambda_{yellow})$ and $Y_{silver}(t=\mathbf{2010})$ in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	4	2	0.33 (Briand et al. 2008)	100	800	0 (in 2006)	128 (in 2008)	0 (in 2010)

Table 5.21. Model outputs for EDA simulations of the Rhone EMU

Model output	Value
Total water surface (km²)	69.06
Average number of yellow eel per 100 m²	1.718
Average number of silver eel per 100 m²	0.086
Total number of yellow eel	1185883
$N_{current}$: Total number of silver eel	59294
$B_{current}(t=2010)$ in kg	47435
$B_{best}(t=2010)$ in kg	103953
B0 ($B_0 = \overline{B_{best}(t<1980)}$) in kg	117557

5.13. Basque Country Eel Management Unit

Contributors: Estibaliz Díaz, Eider Andonegi, Aizkorri Aranburu, María Korta. AZTI-TECNALIA

AZTI-Tecnalia would like to thank to Diputación de Gipuzkoa; Departamento de Medio Ambiente, Planificación Territorial, Agricultura y Pesca and Iker Azpiroz (EKOLUR SL).

Overview

The Basque Country Eel Management Unit (BCEMU) is located on the Northeast coast of Spain (Figure 5.88) and includes 11 hydrological units, six in Bizkaia (Barbadun, Ibaizabal, Butroe, Oka, Lea and Artibai) and five in Gipuzkoa (Deba, Urola, Oria, Urumea, Oiartzun). They constitute a total area of 4,850 km^2 and 1800 km of main rivers and tributaries. All the rivers flow within the BCEMU, except the upper areas of the Oria and the Ibaizabal rivers that belong to the Cantabrian river basin district.

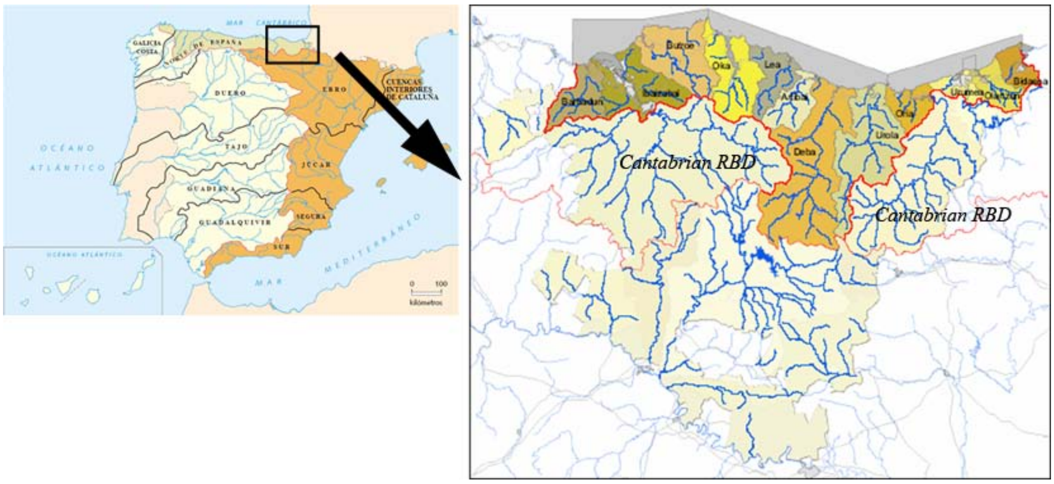


Figure 5.88. Map of Spain showing Basque Country Eel Management Unit.

These rivers are very short and flow over a calcareous and steep geology. Eel is currently present in all the rivers and areas except from the higher part of Gipuzkoan rivers, mainly due to the presence of dams.

These rivers have historically supported a very high human population density and a high morphological pressure, with highly modified water masses and destroyed ecosystems and many dams in the watercourses. Nowadays, even if their ecological status has improved there are some areas where the habitat and water status is still poor and where pollution is a considerable problem.

All this information has been collected within the framework of the Basque Country Eel Management Plan (BCEMP) which can be consulted at (http://www.nasdap.ejgv.euskadi.net/r507393/es/contenidos/nota_prensa/plan_anguila/es_dapa/plan_anguila.html).

Stock status

Eel distribution area has increased in Bizkaia and Gipuzkoa in recent years, due to improvements in water quality and the demolition of some dams. In general terms, the downstream areas of all rivers in the BCEMU are mainly occupied by individuals below 30 cm size while upper areas are dominated by those above 45 cm, which are females.

Eel abundance has generally increased during the last 10-15 years in Gipuzkoa, and maintained or decreased in Bizkaia despite the general recruitment decrease across Europe.

Eel reproductive potential for the Oria river basin was estimated for the first time in 2007, based on the determination of silvering stages by Durif *et al.* (2005) (see BCEMP for more details). It was estimated that there is an annual escapement of about 4200 (400 kg) male and 1750 (1075 kg) female silver eels. The reproductive potential has increased in 2008 and 2009, but has been similar in both years, with 10,200 (1111 kg) and 11,966 (1069 kg) males, and 7,807 (3,645 Kg) and 8,826 (3,081 kg) females for 2008 and 2009 respectively. However, these values are liable to be modified due to new data; since the BCEMP includes a research plan that aims to obtain more information regarding escapement.

Impacts to eel production

Commercial and Recreational Fisheries

The glass eel fishery is a recreational and traditional activity practiced for many centuries in the BCEMU. The historic centre of the glass eel market in Spain was located there, specifically along the Oria River. In 1960, the estimates of total catches of glass eel came to 275 tons (Navaz y Sanz, 1964) but these have decreased to around 200 kg in recent years. The glass eel fishery is regulated by the Basque Government Decree 41/2003 (amended by Decree 107/2005), which prohibits the sale of the glass eel catches.

The greatest number of boat licenses is granted for the Oria River, and this area has the highest catches. Fishing from the boats is the most effective type of fishing. The largest catches are obtained in general between November and January, when the highest amplitudes of tide occur and the rainfalls are more abundant.

The analysis of the catch data provided by the fishermen indicates that the catches have decreased significantly. Updated data on CPUEs confirms the abundance of the glass eels is decreasing.

There is neither yellow nor silver eel professional fishing in the BCEMU and recreational fishing is almost insignificant. Recreational fishing was regulated by means of country council laws. Until recently it was possible to fish eel above 20 cm size within the BDEMU using fishing rods, but this was prohibited from 2008 and 2009 in Gipuzkoa and Bizkaia, respectively.

Obstacles to migration

There are a large number of artificial barriers to fish migration; at least there is an obstacle every 3.3 km^2 of basin and every 1.2 km of river length. However, several obstacles have been demolished in recent years. The overall number of obstacles for upstream migration in the BCEMU is 1467, 408 and 206 of which are working in Bizkaia and Gipuzkoa, respectively. Particularly, 81 of total 252 dams are working in the Oria River nowadays.

There is also a significant number of obstacles to downstream migration. Their impact is unknown, however, and will be studied in the BCEMU research plan. In addition, there are 2 fish farms along the Oria River, one of which is situated in the middle-low area of the river and possibly affects eel migration. There is one specific fish pass for eel in the lower area of Oria since 2005. There are 32 fish passes within the BCEMU but their efficiency for eel is unknown.

Water quality

Until early 1980s the main watercourses and estuaries of the BCEMU received the waste products from the iron and steel industry, shipyard, paper mill, urban and agriculture sewage, etc. The situation has improved due to sanitary work, construction of water-treatment plants, decline of heavy industry, adoption of control measures for wastes, construction of reservoir, etc. Thus, the ecological status has improved in general, although the status of half the rivers in the BCEMU is still below poor condition. However, the situation in estuaries is a little bit better (Figure 5.89). There are other less known sources with a negative effect on water quality like livestock, forestry, mines and quarries, communication routes, rubbish tips, etc.

An extensive monitoring network established in late 1990s provides comprehensive information on the water quality of the BCEMU.

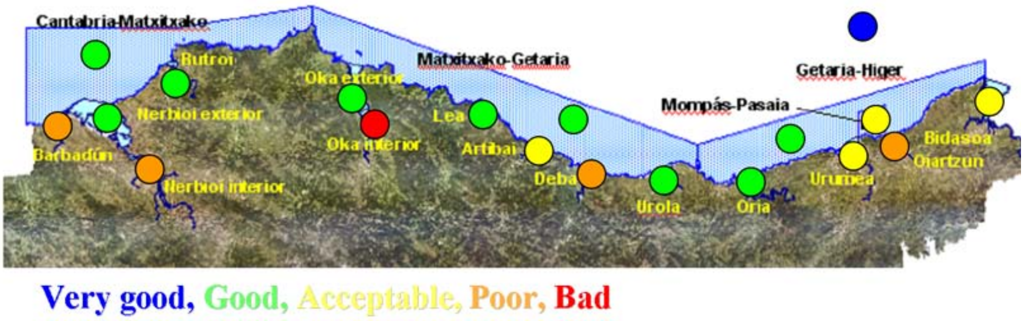


Figure 5.89. Ecological status of transitional water in the BCEMU.

Pollution

The water quality monitoring network indicates that although the ecological status of the river has improved in the last years, there are some areas where the habitats and water status in the BCEMU are still poor and pollution is a considerable problem due to PCBs, pesticides and heavy metals.

Diseases and parasites

Anguillicoloides crassus is present in all rivers of the BCEMU (Díaz *et al.,* 2007). The prevalence of the parasite presents high with inter-, and intra-variability of 15-80% and 15-54% respectively, and the mean intensity is 2 nematodes per eel. Updated data on the prevalence and intensity of *A. crassus* in the BCEMU shows mean values comparable to those obtained in other European rivers. In the last six years the prevalence of individuals that have been parasitized by the nematode has increased but the intensity has decreased (Gallastegi *et al.*, 2002; Diaz *et al.*, 2007; Rallo *et al.*, 2007). This suggests that there are some rivers where the infestation levels of the parasite are still increasing.

Eel data available to POSE for the catchments in the BCEMU

The morphology and characteristics of these basins are explained in the previous sections. The eel data available for these rivers of the BCEMU includes glass eel fishing catches reports since 2003. The abundance and biometric data of yellow and silver eel are obtained in multi-specific electrofishing surveys since 1988 (Bizkaia) and 1992 (Gipuzkoa) which are GIS referenced. Data on the habitat affinity of the different phases of the biological cycle is also available as well as habitat and geographic data according to the Water Frame Directive (WFD, 2000/60/EC). There is a dam and hydroelectric power stations inventory and *Anguillicoloides spp.* presence study (2007).

In addition, the Oria River has more data as it was used as a pilot basin in the INDICANG project where several indicators were estimated. These data include glass eel migration and abundance data (from 2005), providing fishery independent recruitment data; yellow eel migration has been sampled in an ascending trap located in the tidal limit (from 2005); and, eel specific electro-fishing surveys with silvering data (from 2004). Reproductive potential has been estimated since 2007.

5.14. Application of the EDA to the Basque EMU data set

The general method of applying the EDA model to an EMU is described in Chapter 4. Here, we describe the work achieved to apply the model to the Basque dataset, and the resultant predictions of yellow eel stock and silver eel production.

In the analysis, 897 electrofishing operations on 277 electrofishing stations between June and November from 1981 to 2009 were selected (Figure 5.90). Five electrofishing stations (with st_id = KAR130, END10200, AGUI26, BID04100, BID555) had been deleted because of a lack of information about the dams in these watersheds; thus 15 electrofishing operations were deleted (Figure 5.91).

These sites were considered in relation to their distribution over the time period (Figure 5.92), their elevation and distances from the sea and source (Figure 5.93), and the cumulative number of dams against distances from the sea and source, and site elevation (Figure 5.94).

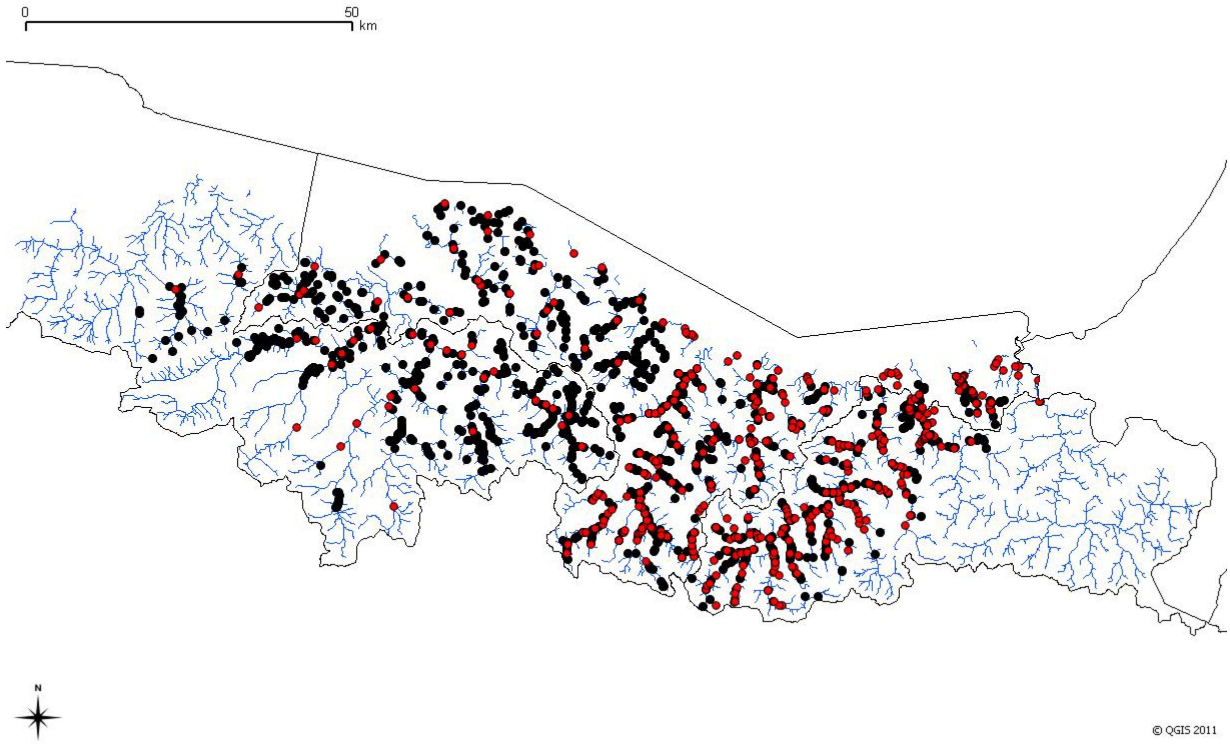


Figure 5.90. Sampling sites (in red) in the Basque EMU catchment on the CCM river network (in blue) with the dam locations (in black).

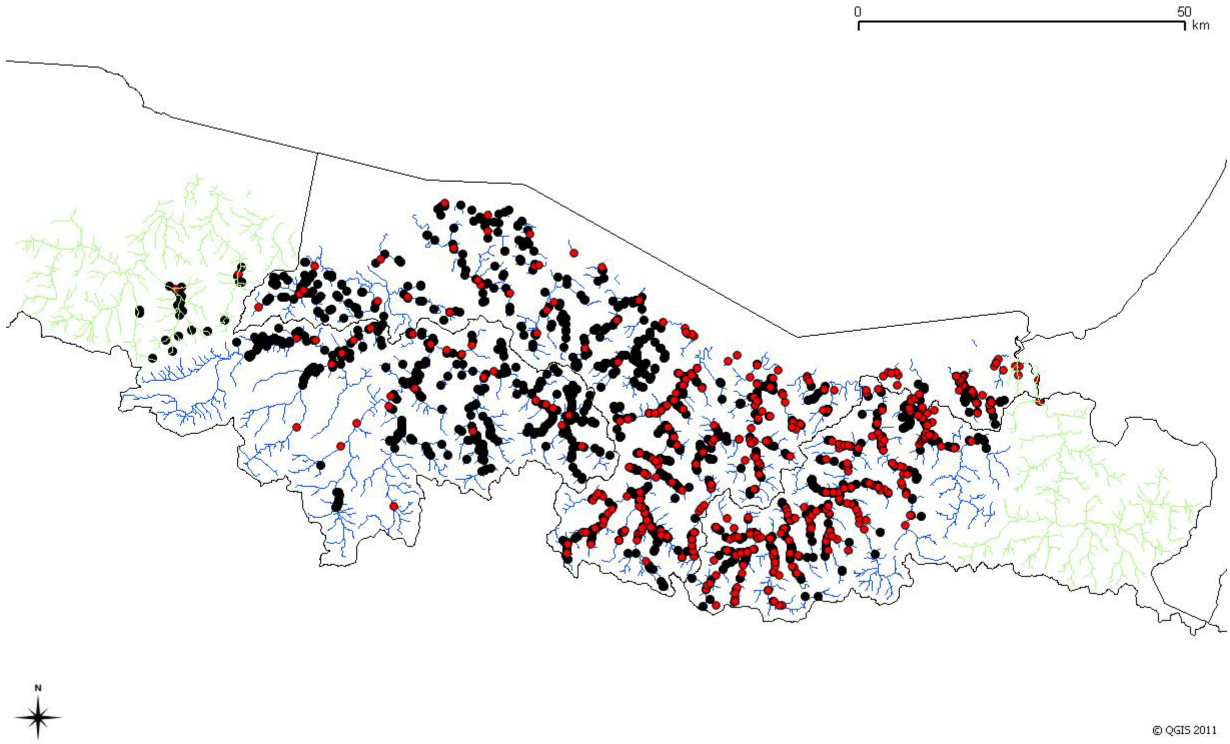


Figure 5.91. Sampling sites (in red) in the Basque EMU catchment on the CCM river network (in blue) with the dam locations (in black). In green the watershed deleted, thus the sampling sites located on these watersheds are deleted in the analysis.

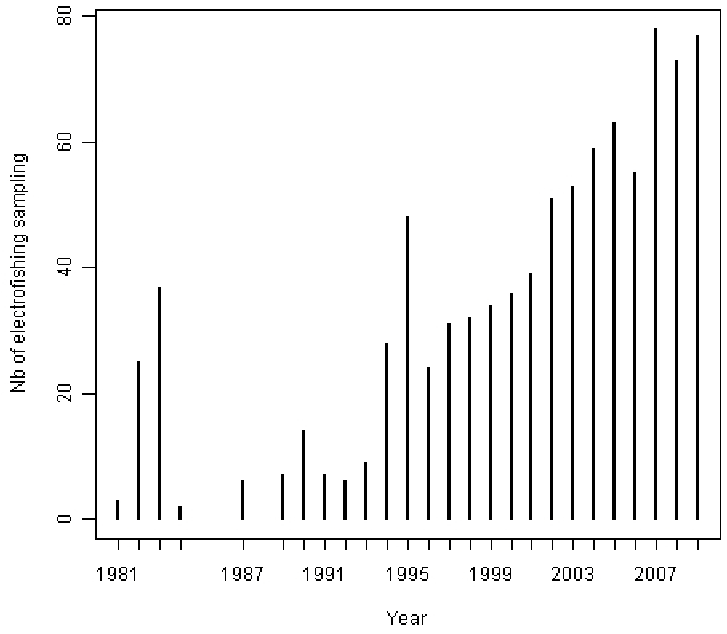


Figure 5.92. Number of electrofishing sampling per year in the Basque EMU.

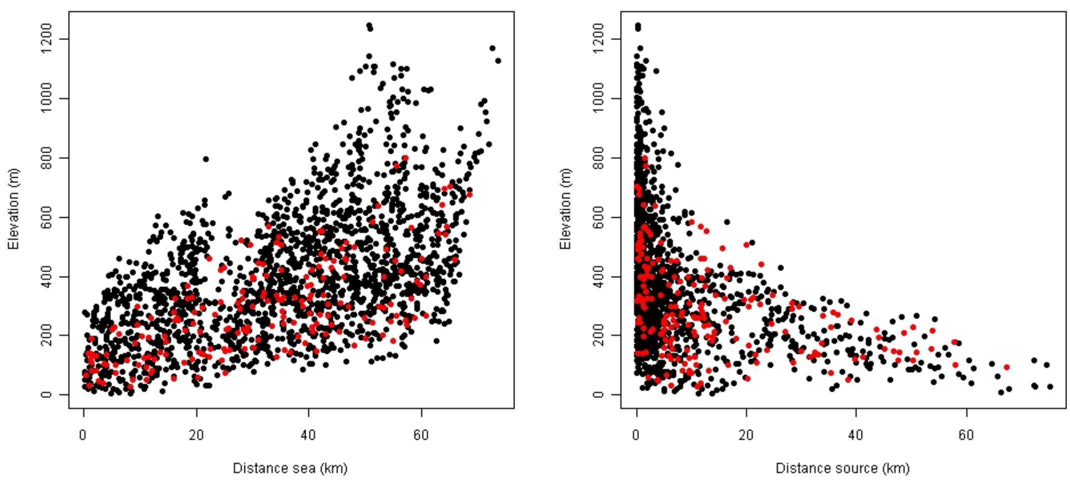


Figure 5.93. Sampling sites (in red) in the Basque EMU catchment on the CCM river network (in black) depending on the elevation and the distance from the sea or from the source.

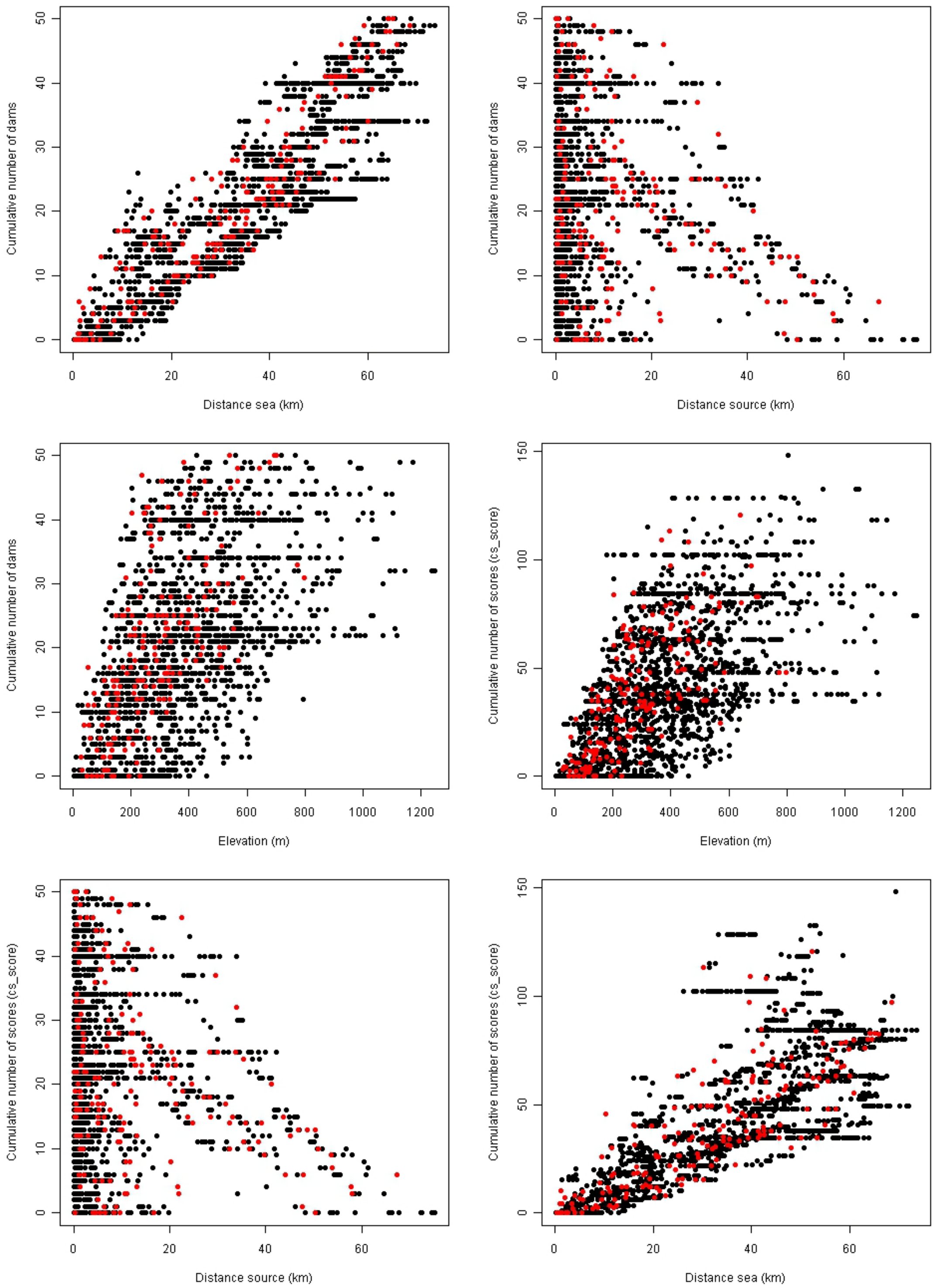


Figure 5.94. Sampling sites (in red) in the Basque EMU catchment on the CCM river network (in black) depending on the cumulative number of dams or scores and the distance from the sea or from the source.

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CCM data set for the Basque EMU provided information for each survey site on the distances to sea and source, relative distance, mean slope and elevation, altitudinal gradient, the area of land drainage upstream, mean annual temperature and rainfall, land use cover (urban, agricultural and no impact) , and the number of dams downstream of the site. The cumulative number of dams from sea to site was used to characterise the obstacle pressure to upstream migrating eels. In total there were 5469 dams distributed throughout the Basque EMU.

A combination of 16 variables (year, month and a set of 14 variables) was tested. The Figures 5.95 and 5.96 show that there are several variables with tightly correlated predictors.

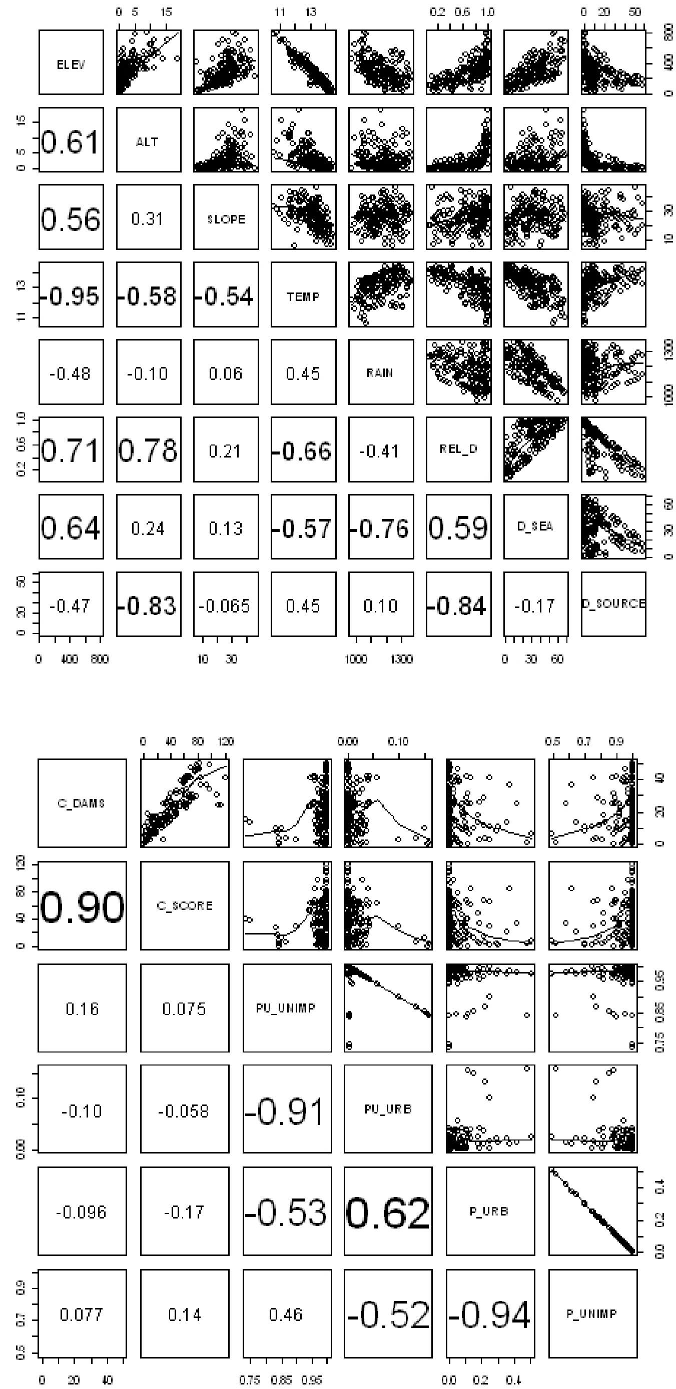


Figure 5.95. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With ELEV: elevation mean, ALT: altitudinal gradient, SLOPE: slope mean, TEMP: temperature mean, RAIN: rain mean, REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, C_DAMS: cumulative number of dams, C_SCORE: cumulative score, PU_UNIMP: upstream percent in unimpact land use (p_up_no_impact), PU_URB: upstream percent in urban use, P_URB: local percent in urban use, P_UNIMP: local percent in unimpact land use (p_no_impact). The font size of the cross-correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

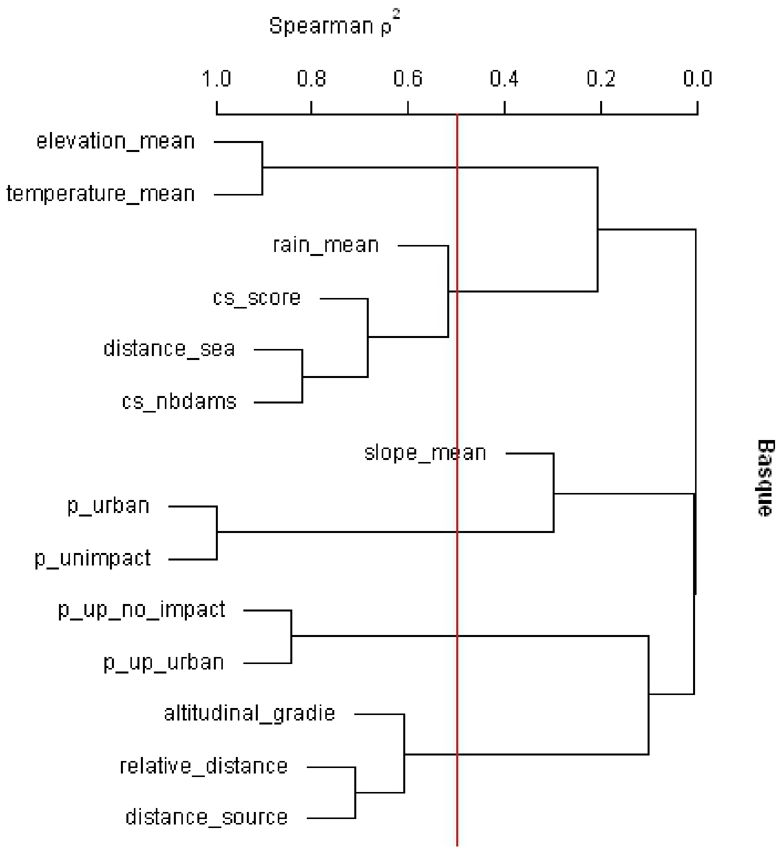


Figure 5.96. Dendrogram obtained by hierarchical cluster analysis of 19 candidate predictors for Basque dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ²=0.5.

According to the threshold of 0.5 for ρ^2, the following variables should be grouped:

- elevation and temperature
- rain, cumulative number of scores, distance from the sea, cumulative number of dams
- p_urban, p_no_impact
- p_up_no_impact and p_up_urban
- altitudinal gradient, relative distance and distance from the source

A separate entry was used for the pairs of variables with significant relationship (Figure 5.95) and for the 5 groups of variables described in Figure 5.96 to avoid spurious correlation.

The combination of the different groups resulted in 765 models to be tested.

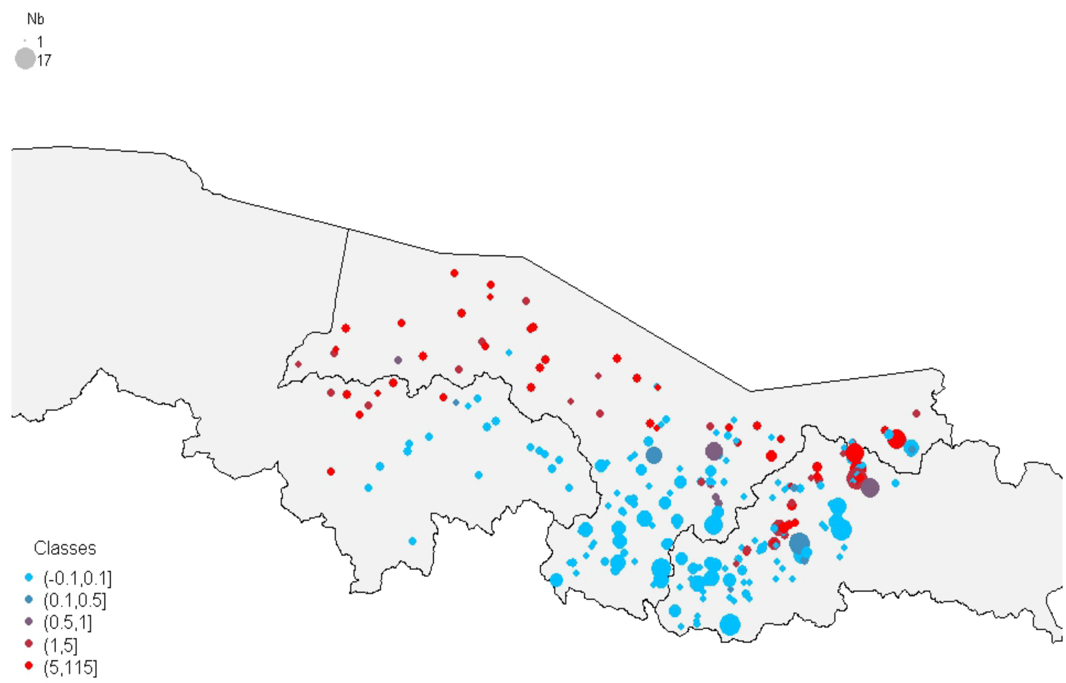


Figure 5.97. Map of density observed densities per electrofishing. All years.

Model Testing Procedure

We calibrated statistical models with a Generalized Additive Model (Hastie & Tibshirani, 1990) to assess how the densities of yellow eel varied between years and according to characteristics of river network, land use and obstacles pressures. All computations were carried out with the R2.12.1 statistical software (R Development Core Team, 2011, cran.r-project.org/). EDA model is based on a delta-gamma model (Stefánsson, 1996) which combines two generalized additive model (GAM). There are three steps of modelling:

- a presence/absence model (delta model) based on a GAM with a binomial distribution and a logit link to determine the probability of a positive catch;
- a density model with the positive data (gamma model) using a GAM with a gamma distribution and logarithm link to determine the level of positive catch; and,
- the multiplication of the two previous models (delta-gamma model).

The GAMs were computed with the library 'gam' (Hastie, 2010) with a cubic spline smoother (3 degree of freedom) for each environmental variable. The delta-gamma (ΔΓ) generalized additive models explain a large portion of the variability in eel abundance data, as there are many occasions where densities are null.

Presence-absence model (Delta model)

The results for tests of 765 models are summarised in Table 5.20. The best density model is selected by the Akaike's Information Criterion (AIC) with a lower AIC indicating a better fit (Akaïke, 1974, Sakamoto *et al.,* 1986). The AIC function indicates a better fit of the response variable at the year and the month of electrofishing sample with the elevation, the distance from the sea, the cumulative number of dams and the upstream percent of urban use (model 1 in Table 5.20). For this model the kappa = 0.668 ± 0.028 with a threshold = 0.5. GAM explained 48 % of the deviance of the abundance of the yellow eel. The six explanatory variables in the model are significant (Table 5.21).

Table 5.20. Model selection results using Akaike's information criterion (AIC) for presence-absence model analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	elevation mean	rain mean	temperature mean	distance sea	distance source	relative distance	cs_nbdams	cs_score	p_up_urban	p_up_no_impact	AIC s=3
1	x	x	x			x			x		x		599.6
2	x	x	x				x		x		x		601.0
3	x	x		x				x		x		x	601.6
4	x	x	x			x			x			x	602.6
5	x	x		x				x	x			x	605.1
6	x	x	x					x	x			x	605.6
7	x	x	x					x	x		x		606.1
8	x	x	x				x		x			x	606.3
9	x	x	x						x			x	606.9
10	x	x	x						x		x		607.3

Model Goodness of fit (presence-absence model)

The best presence-absence model selected (a binomial model with a logit link and a cubic spline smoother – s =3) is:

d != 0 ~ s(year,3)+ s(month,3)+ s(elevation,3)+ s(distance sea,3)+ s(cs_nbdams,3)+ s(p_up_urban,3)

matrice confusion

observed

predicted 1 0

1 197 63

0 59 578

correctly predicted 0.86

present correctly predicted 0.77

absents correctly predicted 0.9

Table 5.21. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar Chisq	P(Chi)	
(Intercept)	1				
s(annee, 3)	1	2	44.208	2.514e-10	***
s(month, 3)	1	2	26.185	2.061e-06	***
s(elev_mean, 3)	1	2	16.221	0.0003003	***
s(distance_sea, 3)	1	2	12.243	0.0021941	**
s(cs_nbdams, 3)	1	2	15.311	0.0004736	***
s(p_up_urban, 3)	1	2	17.249	0.0001796	***

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

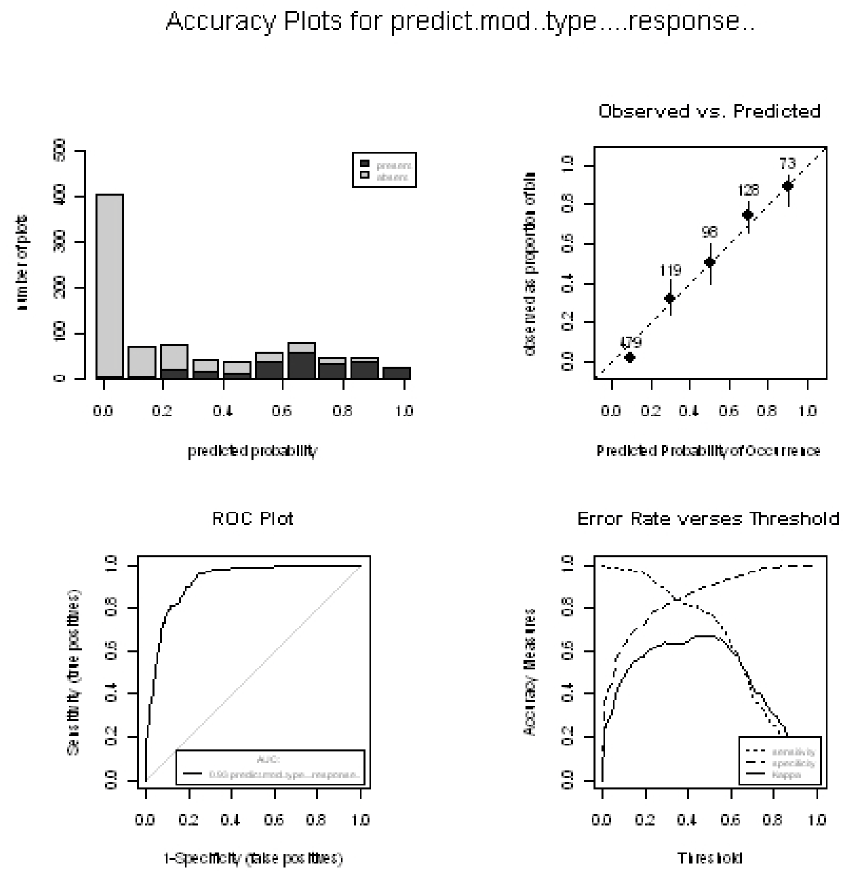


Figure 5.98. Model quality and threshold selection graphs for presence-absence model selected with a histogram plot (upper left), a calibration plot (upper right), a ROC plot with the associated Area Under the Curve (AUC) (lower left), and an error rate verses threshold plot (lower right).

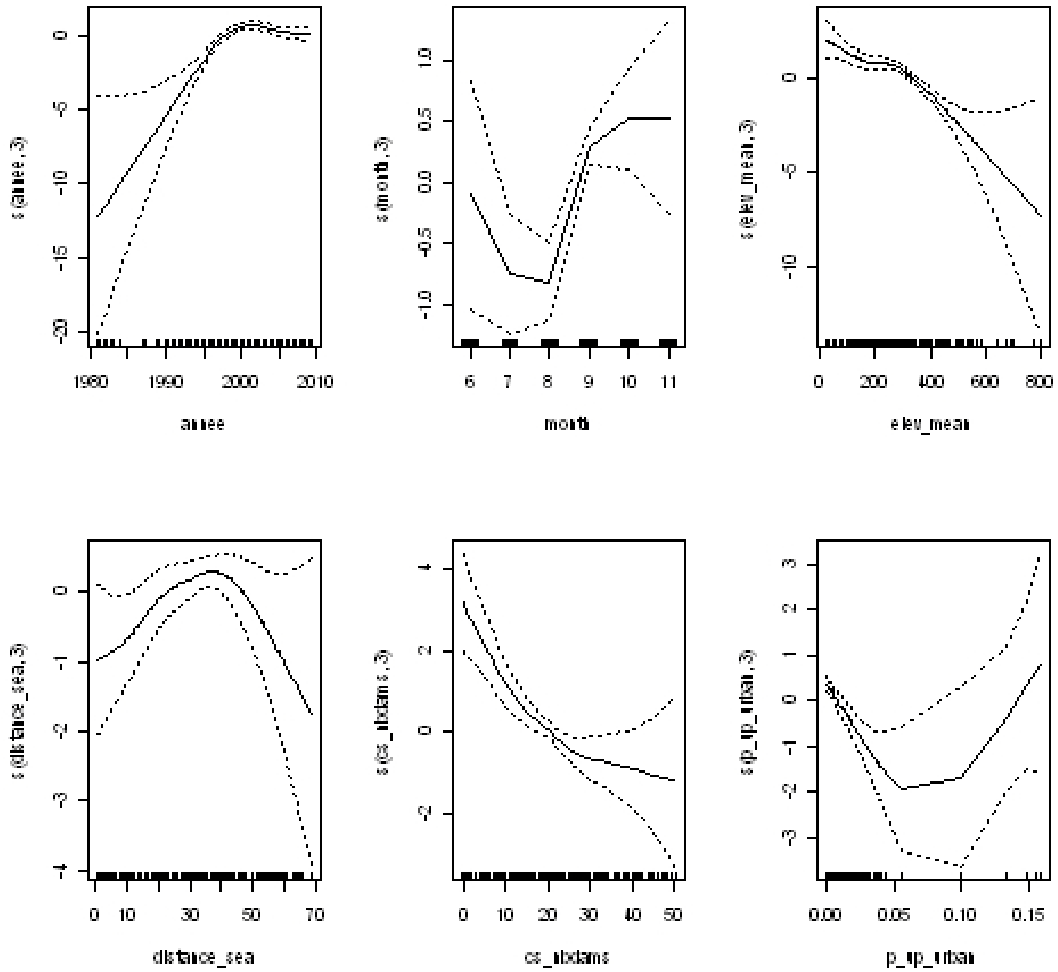


Figure 5.99. Response curves of each variable included in the generalized additive model (GAM) for the presence/absence model for Basque EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The residual of the presence absence model are plotted for each year (Figure 5.100). If the results of the fishing operation are in overplot (several year point on top of the other), they are slightly moved around their origin to show on the map. Several operations for the same year will remain in overplot. The residuals correspond to predicted-observed values, hence a blue spot will indicate a predicted value larger than the observed one, and a red point a predicted value lower than the observed one.

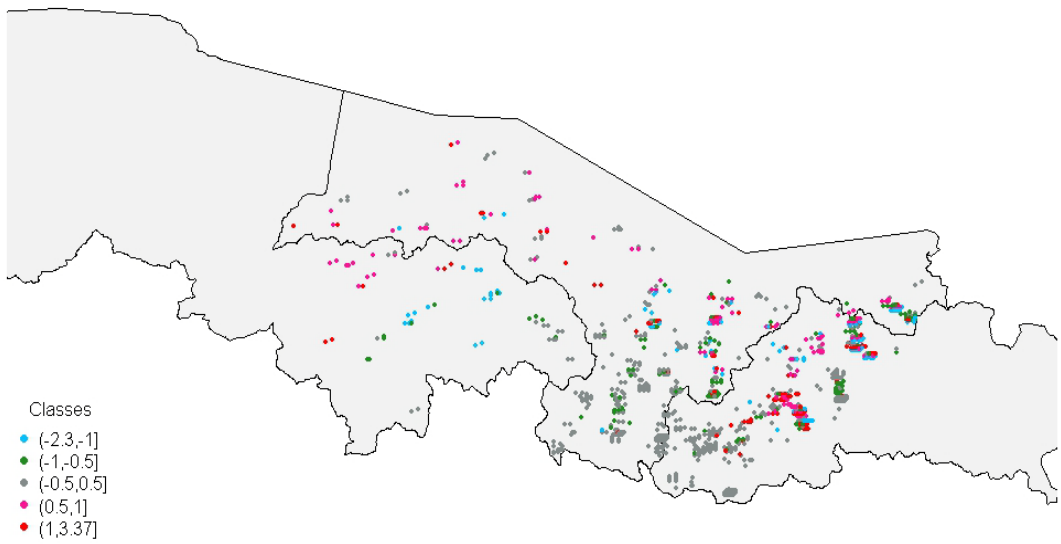


Figure 5.100. Residuals per year: presence-absence model.

In the next graph (Figure 5.101), the residuals are averaged for each station. A blue spot is a station with on average over the years predicted values far larger than the observation. The size of the point is related to the number of electrofishing operation in the station.

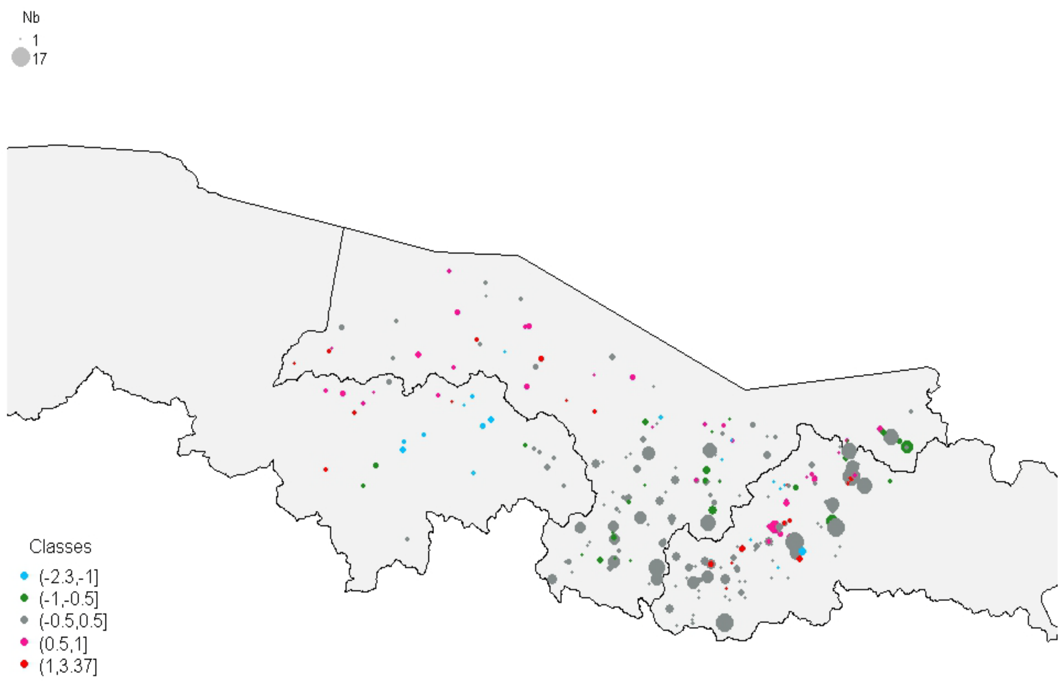
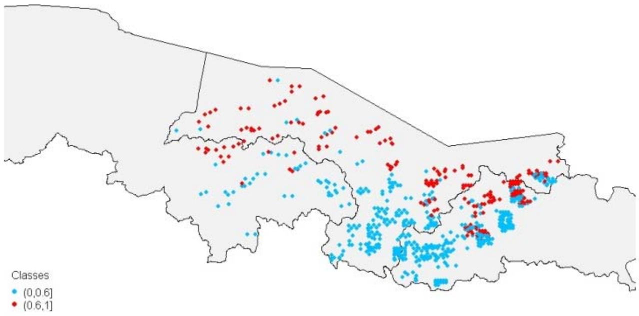


Figure 5.101. Mean Residuals per stations: presence-absence model.

The map of predicted value (Figure 5.102) from the model is computed before and after the break point in the year variable response trend. The threshold chosen for the response is the value giving the best Kappa, i.e. trade-off between the lowest number of wrongly predicted presence (false positive), and wrongly predicted absence (false negative).

(a)



(b)

Figure 5.102. Predicted value: presence-absence model year < 1998 (a), Predicted value: presence-absence model year ≥ 1998 (b).

Density model

The density model is applied only to positive value. It is a gam model with a gamma distribution and a logarithm link. 765 models have been tested.

Model results for density model are summarized in Table 5.22. The AIC function indicates a better fit of the response variable at the year and month of electrofishing sample with the elevation, the distance to the sea, the cumulative number of score, and the upstream percent of no impact land use). GAM explained 51% of the deviance of the abundance of the yellow eel. The effects of four explanatory variables in the model (month, elevation, cs_score, p_up_no_impact) are significant (Table 5.23). The Spearman rank correlation between the observed values and the fitted values is statistically significant (ρ=0.696, p-value < 2.2 10^{-16}, Figure 5.103).

Table 5.22. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	elevation mean	rain mean	temperature mean	distance sea	distance source	relative distance	cs_nbdams	cs_score	p_up_urban	p_up_no_impact	AIC s=3
1	x	x	x			x				x		x	1552.6
2	x	x	x			x				x			1555.2
3	x		x			x				x		x	1555.4
4	x	x	x			x				x	x		1555.6
5	x	x	x				x					x	1555.7
6	x		x			x				x	x		1557.6
7	x		x			x				x			1557.8
8	x	x	x			x							1559.4
9	x	x	x			x	x						1560.4
10	x	x	x			x						x	1560.8

Model Goodness of fit (density model)

The best density model selected is: d ~ s(annee, 3) + s(month, 3) + s(elevation mean, 3) + s(distance sea, 3) + s(cs_score, 3) + s(p_up_no_impact, 3)

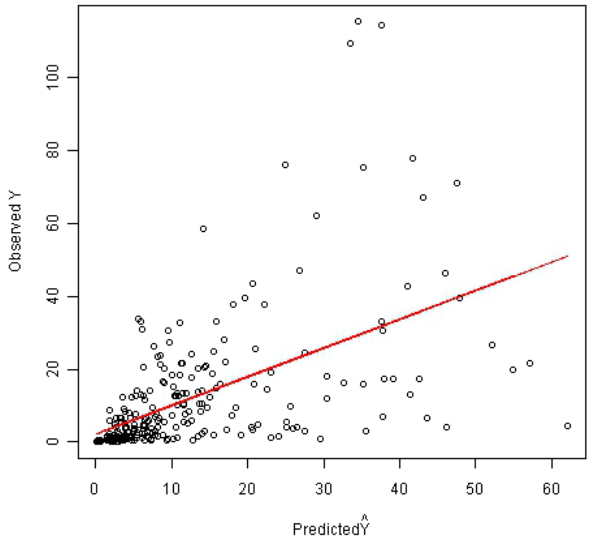


Figure 5.103. Observed vs. predicted regression scatter plot.

Table 5.23. Table of effects, DF for terms and Chi-squares for non-parametric effects.

Df Npar Df Npar F Pr(F)
(Intercept) 1
s(annee, 3) 1 2.0 0.9403 0.39195
s(month, 3) 1 2.0 2.6477 0.07291 .
s(elev_mean, 3) 1 2.0 21.5014 2.629e-09 ***
s(distance_sea, 3) 1 2.0 1.7430 0.17723
s(cs_score, 3) 1 2.0 3.6828 0.02660 *
s(p_up_no_impact, 3) 1 2.1 3.4107 0.03220 *

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

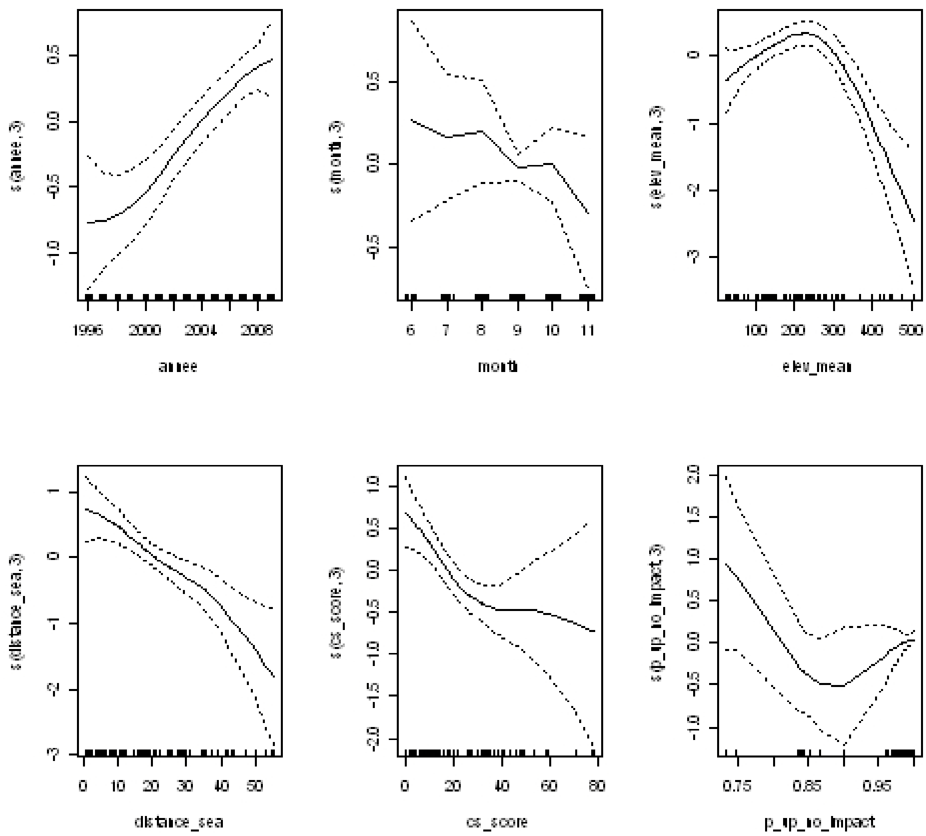


Figure 5.104. Response curves of each variable included in the generalized additive model (GAM) for the density model for Basque EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The map below (Figure 5.105) shows the mean residuals for operations where positive densities were observed. The sign is calculated according to predicted – observed. The size of the points gives an indication of how many electrofishing operations occurred at that point and thus of the « weight » of that station in the model.

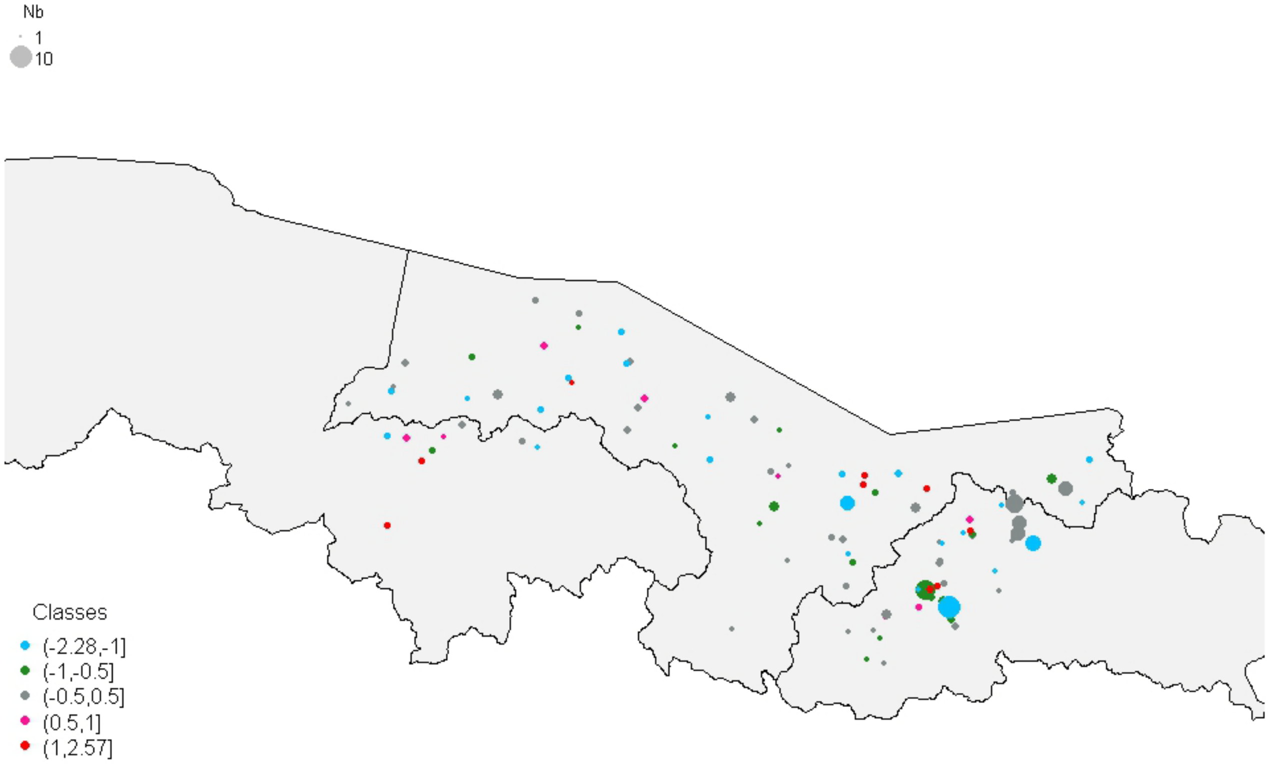


Figure 5.105. Mean residuals per stations: density model.

The next map (Figure 5.106) gives the same information but residuals are 'jittered' around the point. There is no clear spatial trend in the presence of large positive or negative residuals.

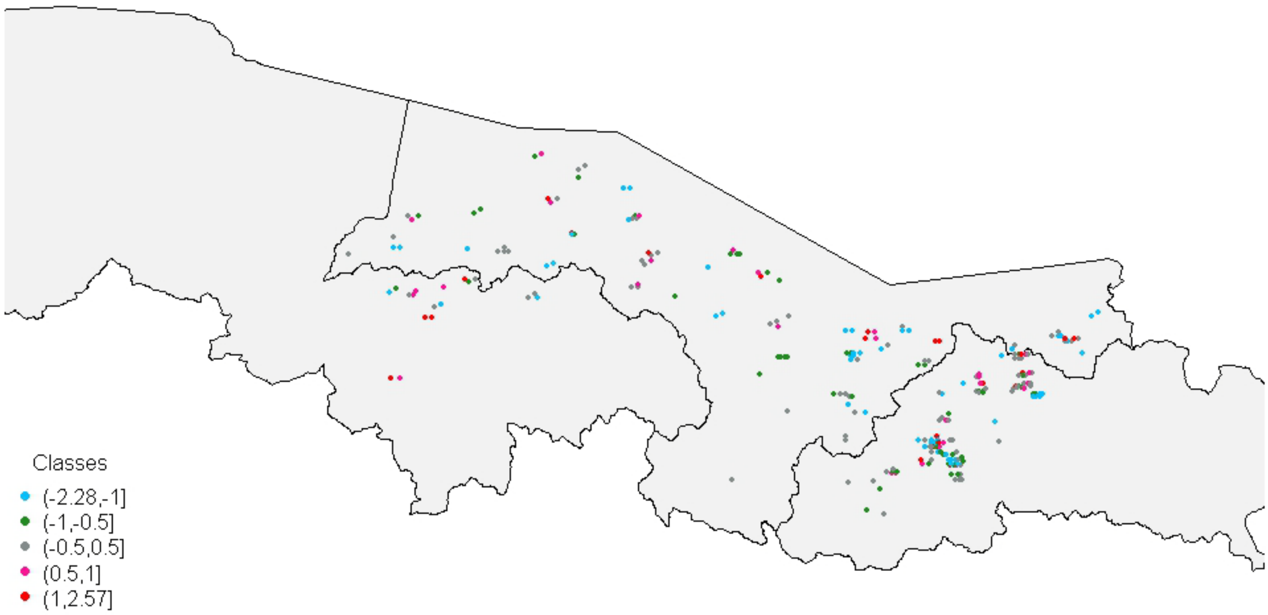
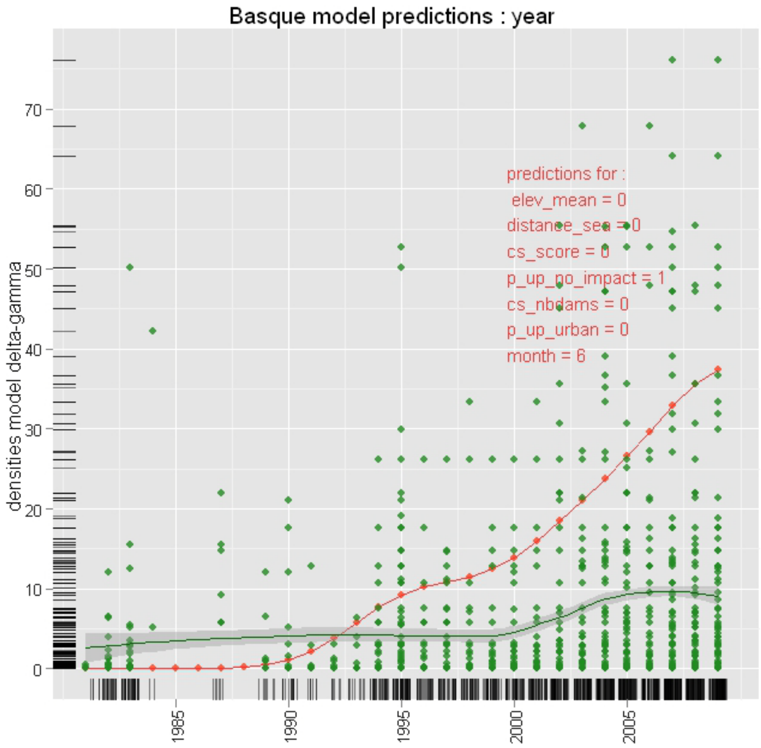
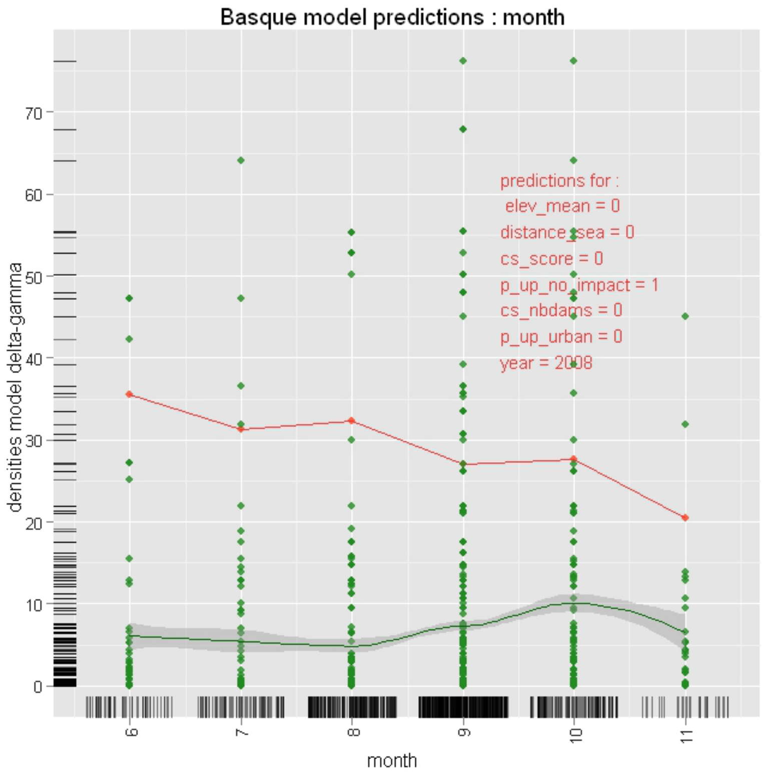


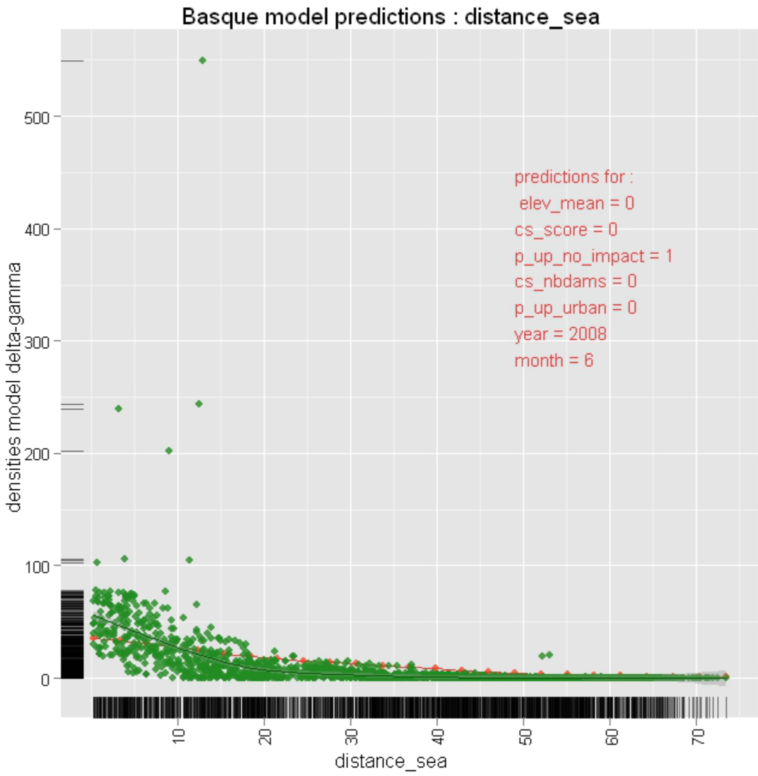
Figure 5.106. Residuals per year: density model.

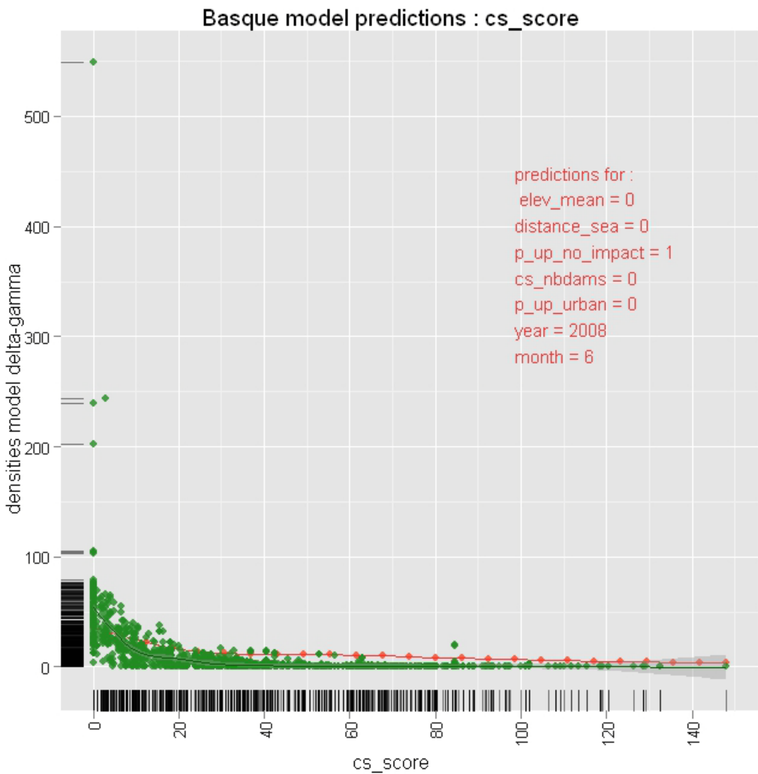
Final model

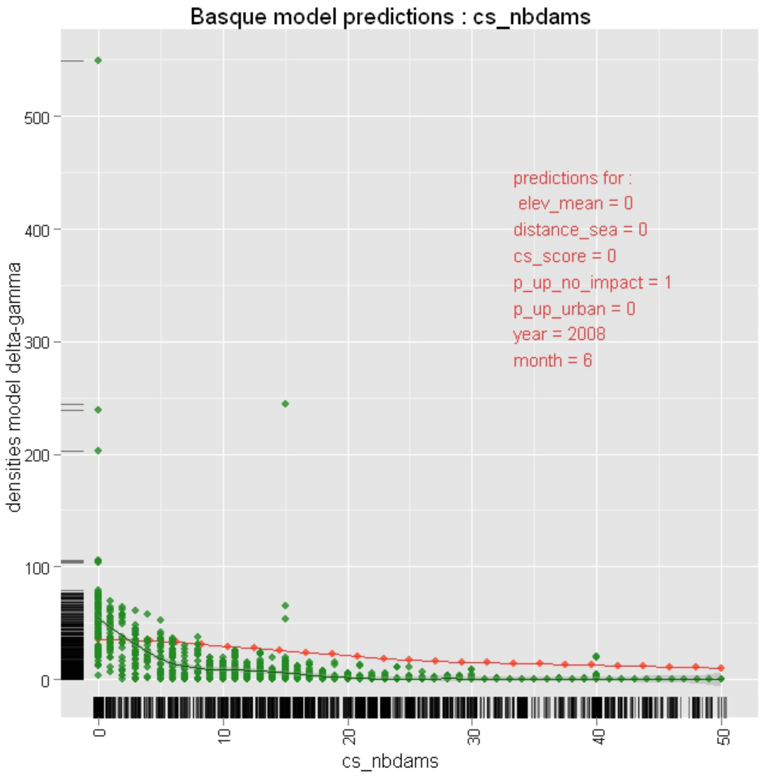
The final model is the product of the delta model by the gamma model. The delta model is the probability of having a positive density. The gamma model is the level of a density for positive values. The model is used to predict densities on all the river stretches of the EMU. The following figures (not numbered) give the trend (red curve) of an effect given that all other parameters are fixed (see figure for the value of fixed parameters) as well as the predicted density of each river stretch of the CCM (green points) and its average along the examined effect (green curve). The rugs on the scale have been moved slightly to indicate the density of observation.

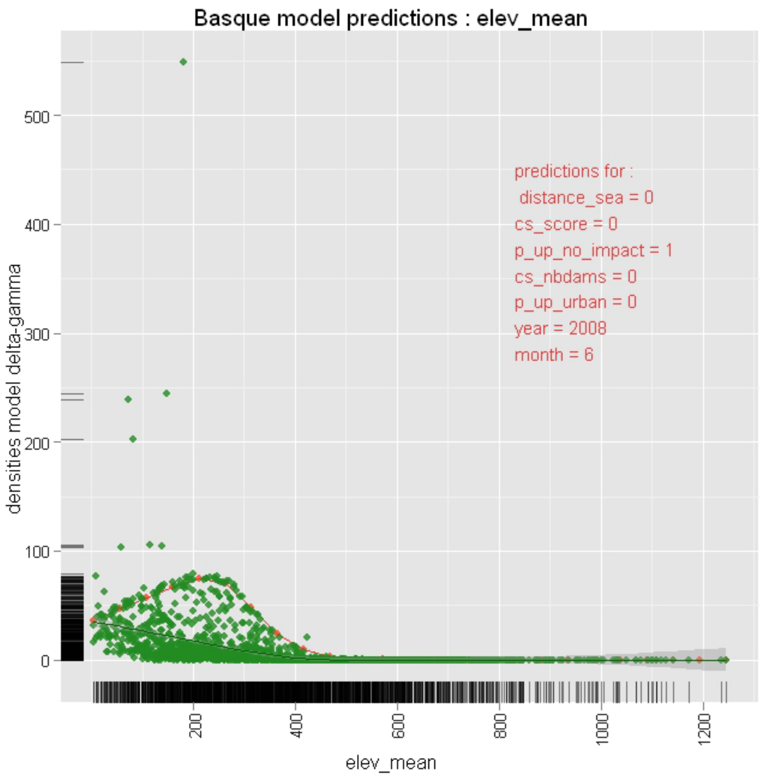

Basque model predictions : year
densities model delta-gamma
predictions for :
elev_mean = 0
distance_sea = 0
cs_score = 0
p_up_no_impact = 1
cs_nbdams = 0
p_up_urban = 0
month = 6
0
10
20
30
40
50
60
70
1985
1990
1995
2000
2005

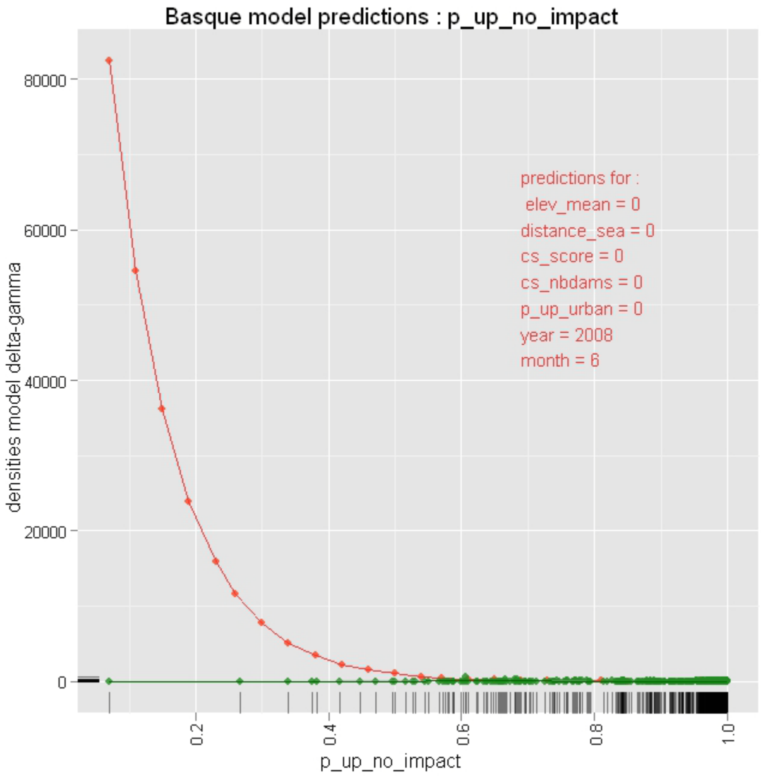

Basque model predictions : month
densities model delta-gamma
predictions for :
elev_mean = 0
distance_sea = 0
cs_score = 0
p_up_no_impact = 1
cs_nbdams = 0
p_up_urban = 0
year = 2008
0
10
20
30
40
50
60
70
6
7
8
9
10
11
month

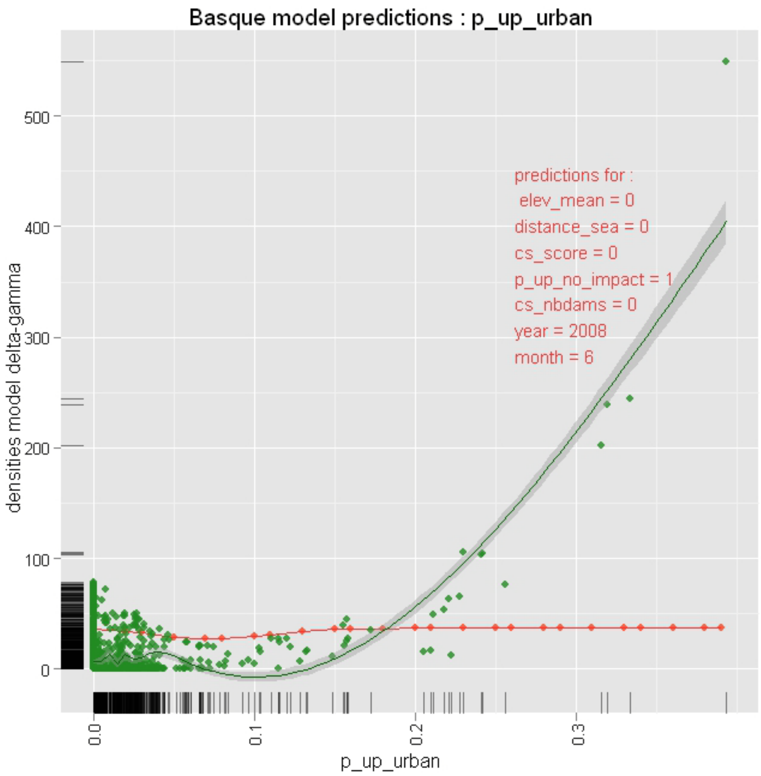

Basque model predictions : distance_sea
predictions for :
elev_mean = 0
cs_score = 0
p_up_no_impact = 1
cs_nbdams = 0
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
distance_sea
0
100
200
300
400
500
10
20
30
40
50
60
70


Basque model predictions : cs_score
predictions for :
elev_mean = 0
distance_sea = 0
p_up_no_impact = 1
cs_nbdams = 0
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
cs_score
0
100
200
300
400
500
0
20
40
60
80
100
120
140


Basque model predictions : cs_nbdams
densities model delta-gamma
cs_nbdams
predictions for :
elev_mean = 0
distance_sea = 0
cs_score = 0
p_up_no_impact = 1
p_up_urban = 0
year = 2008
month = 6


Basque model predictions : elev_mean
densities model delta-gamma
elev_mean
predictions for :
distance_sea = 0
cs_score = 0
p_up_no_impact = 1
cs_nbdams = 0
p_up_urban = 0
year = 2008
month = 6





The following figures (5.107 and 5.108) give the yellow and silver eel production along the examined variable. On the upper corner they also give the cumulated wetted area along the same variable that supports that production.

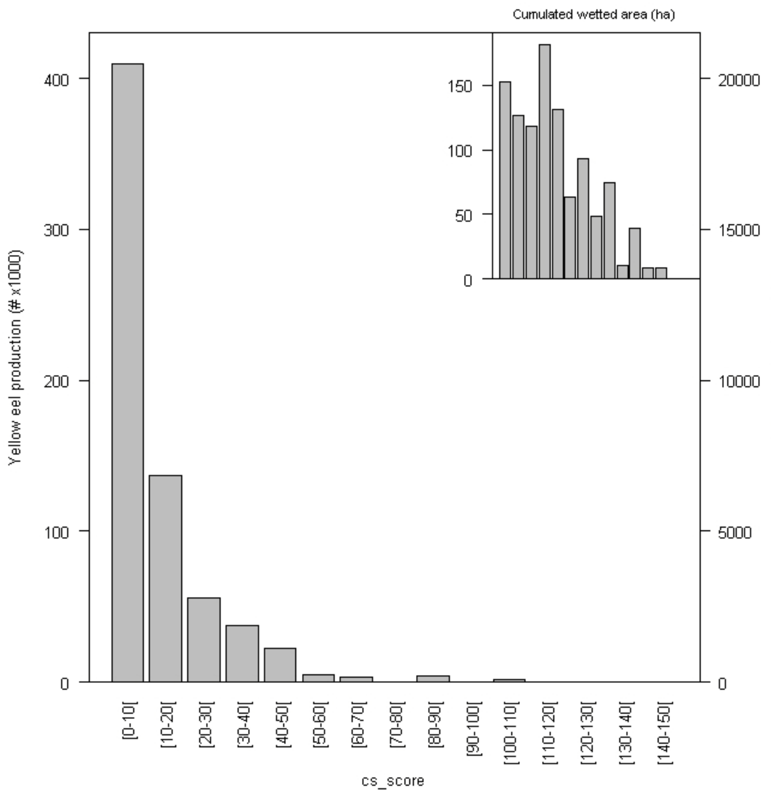


Figure 5.107. Yellow and silver eel production along the cumulative number of scores (cs_score).

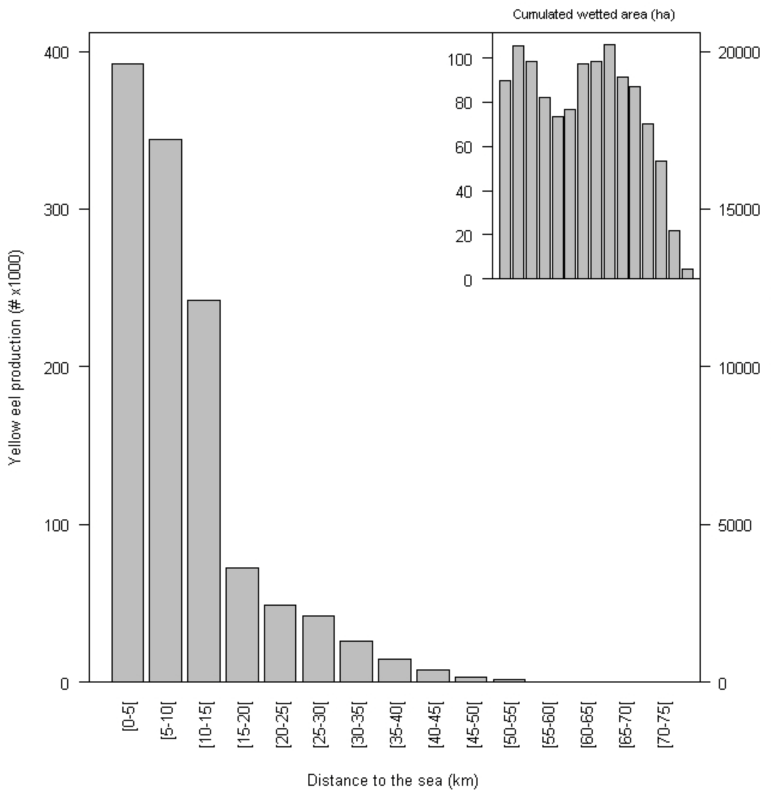


Figure 5.108. Yellow and silver eel production along the distance to the sea.

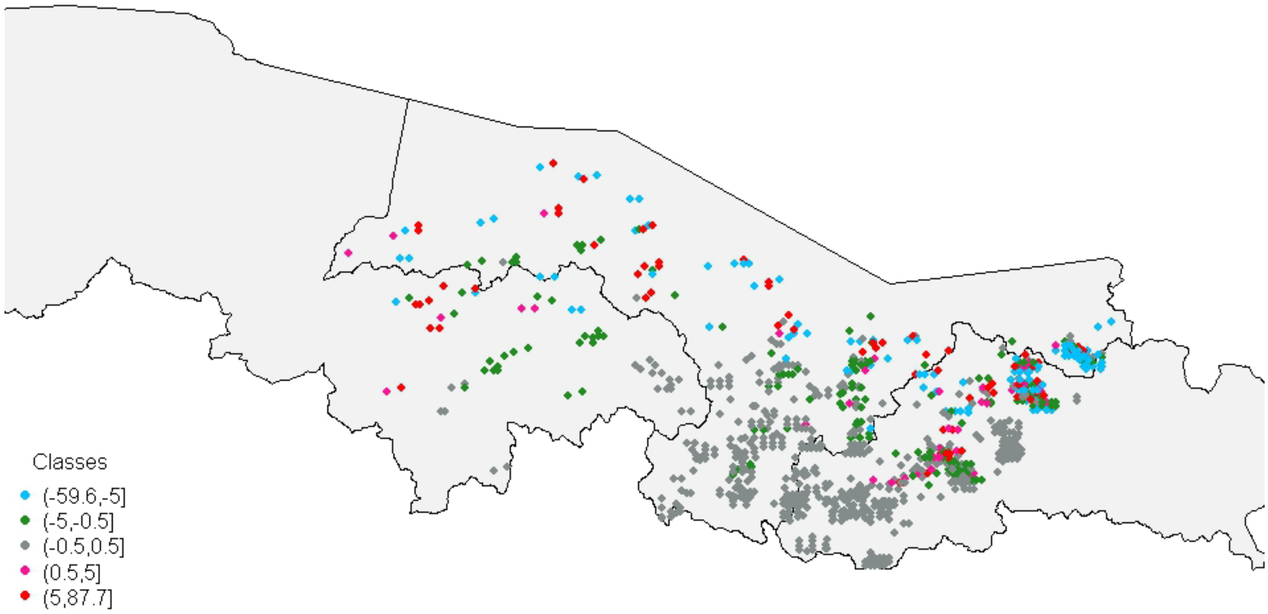


Figure 5.109. Map of mean residuals (predicted - observed), for the full model. Each year of data a point is shown with a small distance around the point.

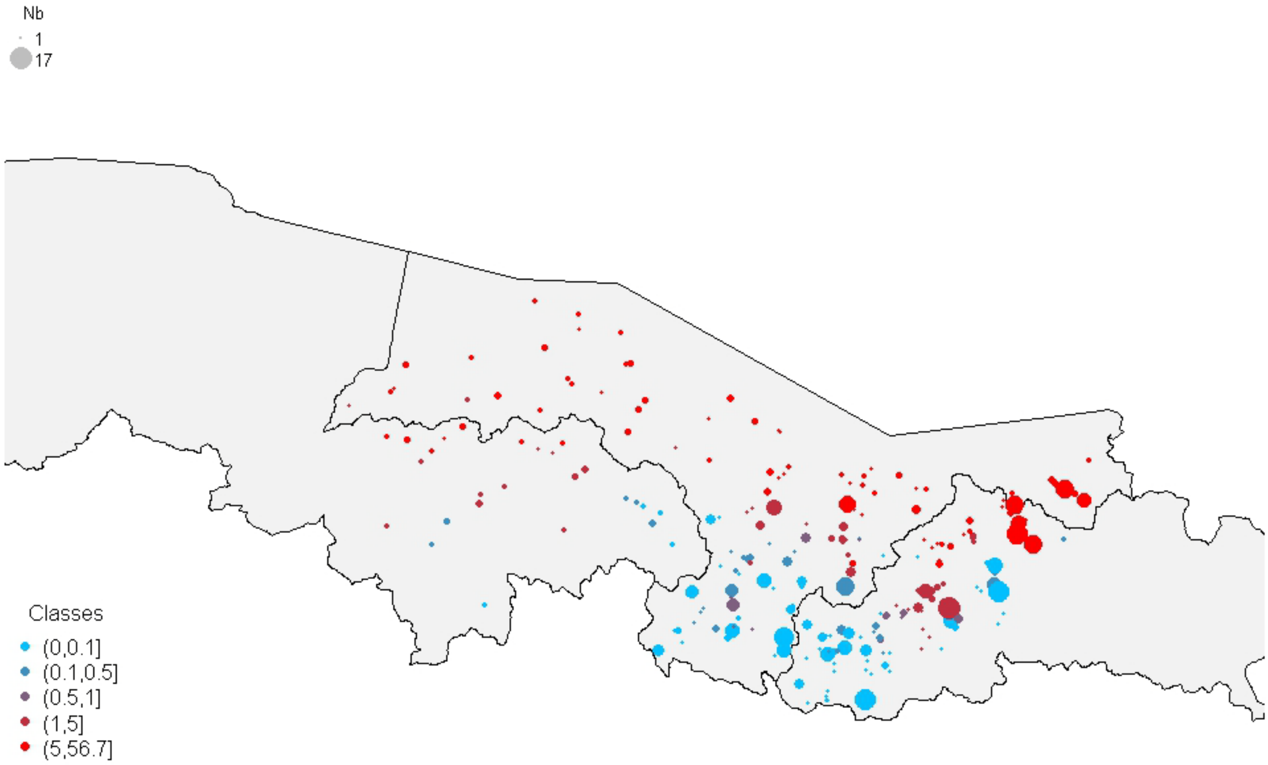


Figure 5.110. EDA predictions for the full model, the size model is given according to the number of electrofishing at that point.

EDA predictions for the Basque EMU

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch with a conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou, 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al.* (2000) have estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA a constant conversion rate of 5% was chosen as a default value. This constant rate conversion of eel in equivalent silver eel is based on the assumption of density-independent biological processes.

This potential escapement in number in then converted into biomass with the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for the Basque EMU. The current biomass ($B_{current}$) is calculated by subtracting the anthropogenic mortalities on silver eel to the potential escapement in biomass.

Best achievable escapements (B_{best}) can be calculated by the current biomass artificially forced to null anthropogenic impact (no dam, land use mortality to “no impact” and silver eel catch to 0) added with silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historical level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment.

The model parameters used to evaluate Bcurrent, Bbest and Bpristine are given in Table 4.24. Table 5.25 summarises the EDA estimates of the total numbers of yellow and silver eels in the Basque EMU.

Table 5.24. Data input for silver eel estimation for the Basque EMU

With Y_{glass}(t= $2010-\tau$), Y_{yellow}(t= $2010-\tau+\lambda_{yellow}$) and Y_{silver}(t=**2010**) in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	4	2	0.33	17.77	130.24 (254.6 from Diputación de Gipuzkoa)	614 (2010, from the data reported by the recreational fishermen)	No data	No data

Table 5.25. Model outputs for EDA simulations of the Basque EMU

Model output	Value
Total water surface (km²)	11.56
Average number of yellow eel per 100 m²	10.34
Average number of silver eel per 100 m²	0.517
Total number of yellow eel	1195163
$N_{current}$: Total number of silver eel	59758.14

Considerations of the model application

The EDA estimation of silver eel production for the water surface of the Basque river basin (11.56 km²) is similar to the rough estimation from the Eel Management Plan of the Basque Country (EMPBC) with 13.7 km² (Gipuzkoa 6.7km², Bizkaia 7.0 km²). In the EMPBC, the calculations were made from the samplings of three River Basins (Oria, Deba and Urola) and then extrapolated to all the Basque EMU. The estimation of the abundance of eels was made through electro-fishing sampling by the method of Seber *et al.* (1967) and based in the consecutive captures of De Lury (1947). The development stage was determined according to Durif *et al.* (2005). Regarding to, The data from Diputación Foral de Gipuzkoa (Gipuzkoa province) suggests a silvering rate with a wide range depending on the year and the basin (2.2-14.6%), but a mean value of 4 or 5% seems to fit quite well (see attached table below) and is comparable to the silvering rate we have used in the model (a 5% silvering rate). Depending on the year, the average yellow eel density varies from 7 to 10 yellow eel per 100 m². However, the average yellow eel density data may represent a wide variety of values because they are based on electro-fishing surveys conducted mainly in low reaches where eel populations are distributed. This result is similar to the prediction of EDA model (10.34 yellow per 100m²). The silver eel density has also a wide range, a mean and conservative value based on observations would be around 0.23-0.28 silver eel per 100m² in Gipuzkoa, which are low compared to those predicted by EDA (0.517 silver eel/10m² in EDA).

To summarize, the calibration of EDA on the Basque river basin provides fairly good estimates of the eel population.

Table 5.26. Silvering rates according to data from Diputación Foral de Gipuzkoa.

BASIN	YEAR	% SILVERING	NOTES
ORIA	2008	9.6	
ORIA	2009	12.2	
ORIA	2010	14.6	Population fall
DEBA	2009	2.2	Very low accessibility
UROLA	2010	4.0	Low accessibility

5.15. Anglian RBD (UK)

Overview

The Anglian RBD drains to the North Sea along the east coast of England, between the Thames estuary to the south, and the Humber estuary to the north.

There are over 750 rivers within the RBD, with a combined length of about 6,400 km (Figure 5.111). There are 36 lakes with a combined wetted area of about 9500 ha, only a little less than the wetted area of rivers (total combined 20,000 ha or 200 km^2). In comparison, however, transitional waters (estuaries) and coastal water bodies constitute much greater wetted areas, at about 330 and 2,300 km^2, respectively.

The RBD comprises several large catchments in the north and south west, e.g. the Nene, Welland, Witham, Great Ouse and Ely Ouse, as well as multiple smaller catchments throughout, including the Yare, Bure, Waveney, Gipping, Blackwater, Chelmer, Stour and Colne, which are in the south eastern part of the RBD. Much of the northern area is fertile low-lying agricultural land, and substantial impact has been made on the rivers by land drainage and intensive farming practices. The land use varies across the multitude of smaller catchments within the eastern half of the RBD, and includes forestry, industry, and localised pockets of heavy urbanisation, although overall, agriculture dominates.

Figure 5.111. Rivers of the Anglian RBD.

Prior to the late 1930s, large parts of the RBD were subjected to several hundred years of fenland drainage to create habitable and agricultural land. After the 1940s large areas were drained to optimise the amount of land available for intensive agriculture. This period led to the wide-scale loss of aquatic habitats. However the river systems were still relatively open to migrating eels, except those rivers with mills and locks. In response to the east coast floods in 1953, the emphasis of management changed from drainage to flood defence. This continues to be the leading factor to the present day. This emphasis on flood defence has lead to the closing off of river systems to migratory species with the construction of large-scale tidal defence schemes, tidal flap valves and locks. This has meant that the tidal limit, which used to influence rivers far inland, has been restricted to estuary level.

Due to the combination of the semi-arid nature of the Anglian RBD and the high water demands for human use and agriculture, there is a high level of water management and monitoring (requiring in-stream structures, weirs etc). This, combined with historic milling and land drainage means that the majority of rivers within the RBD are highly modified and regulated systems. River engineering structures such as dams, weirs, sluices and locks, barrages and flap valves may act as migration barriers to fish, thus having the potential to seriously affect eel populations in upstream freshwater habitats (Knights and White, 1998).

Stock Status

No data-based estimates of truly pristine escapement exist for English eel basins. The national method to estimate current silver eel escapement is based on yellow eel density and size frequency data and uses a probability mode (A.Knights & M. Aprahamian, in prep.). Silver eel estimation was made using the data collected at 17 sites on the River Colne. The probability model estimated silver eel output for the Colne to be 0.6 kg/ha and for the total RBD an estimate of 12.6 t / yr.

Recruitment data are available from two sites; on the Chelmer and Blackwater Canal at Beeleigh in Essex (from 2000) and on the River Stour at Flatford in Suffolk (from 2002). These traps are operated every year between April and July and provide an indication of the strength of the glass eel run for that year. They also catch pigmented elvers migrating upstream.

Catches have varied considerably between years at both sites, ranging from about 4,000 to <100 glass eel on the Chelmer and 11,000 to <1,000 glass eel on the Stour. These traps only provide semi-quantitative data, but indicate that relatively low numbers of glass eels are returning to the East Coast. This is probably due, in part, to the distance of the Anglian coast from the Continental Shelf, tending to produce low densities of eels within rivers on the North Sea coast of the UK.

Multi-species surveys are undertaken at 562 sites throughout the RBD, yielding information on the distribution of yellow eels throughout the RBD (Figure 5.112). Although the upper reaches of many catchments appear to support few eel populations, the only systems that appear to have obvious cut-off points for eels in their upper most reaches are the Great Ouse, Nene and possibly the Welland.

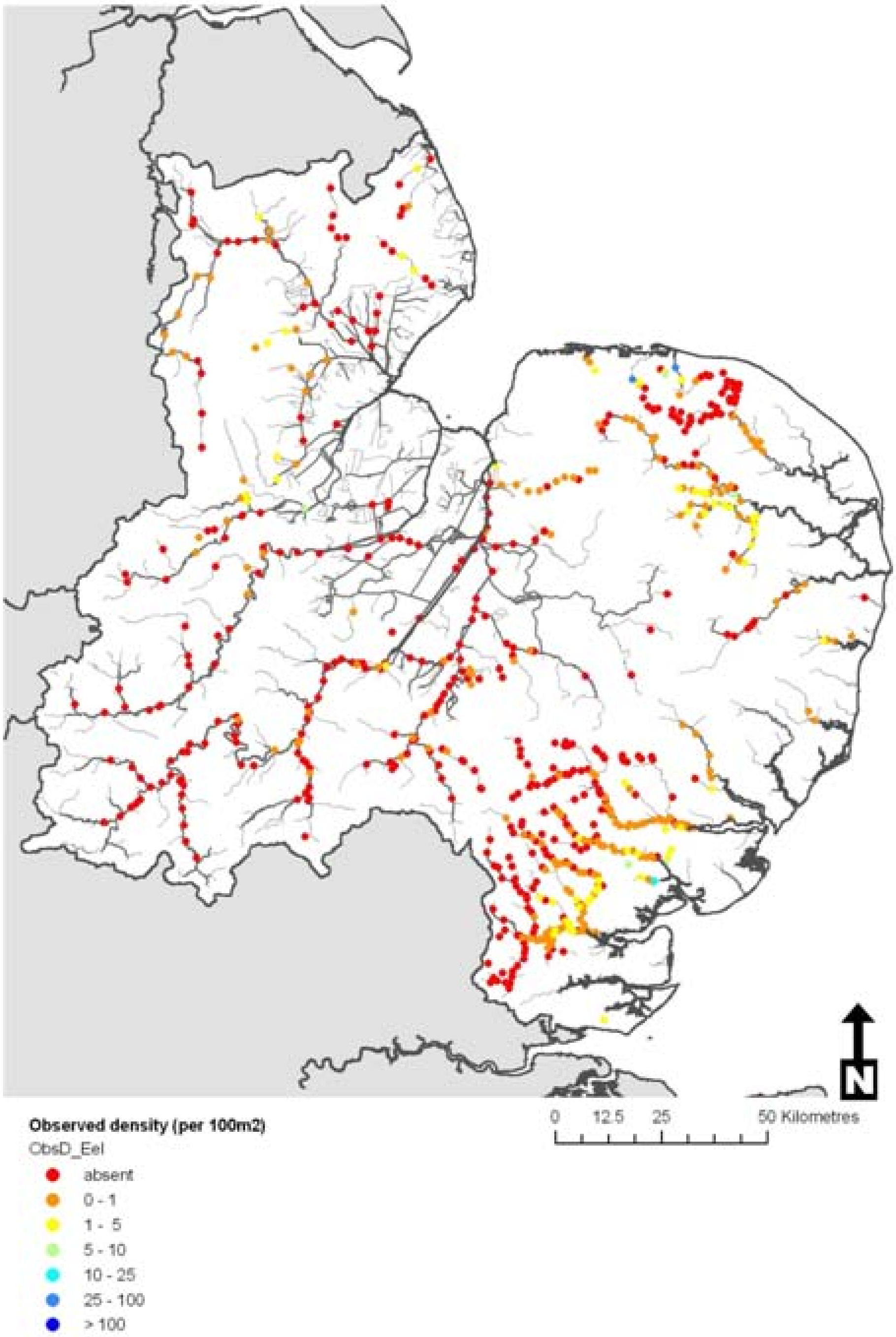


Figure 5.112. Distribution of eel in the Anglian RBD (2001-05 survey data combined).

Quantitative data on yellow eel populations are available from consistent long-term data sets for the majority of Essex and Suffolk catchments, in the south eastern part of the RBD. The overall trends for the Blackwater, Stour and Colne suggest a decline in eel densities from the earliest data in the mid-1980s to date (Figure 5.113).

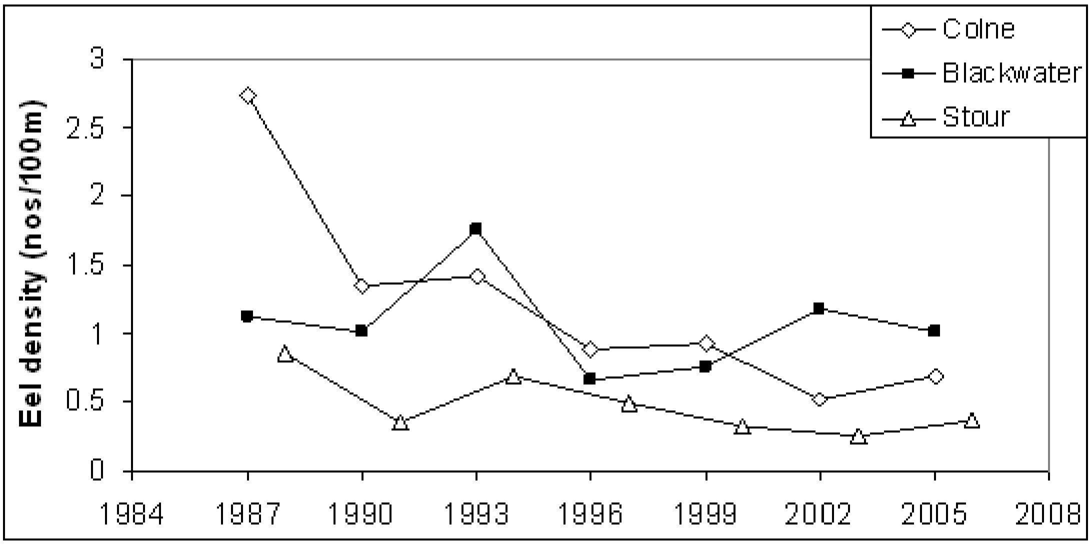


Figure 5.113. Mean density of eels (>99mm) on the rivers Colne, Blackwater and Stour. Data taken from 3-yearly multi-species electric fishing surveys.

The current status of yellow eel stocks in the four of the catchments in the RBD where accurate data have been recorded; the Blackwater, Chelmer, Suffolk Stour and Colne, demonstrates a clear dominance of larger (female) eels in the Blackwater, Chelmer and Stour catchments.

There is insufficient scientific information available to comment on stocks within transitional waters in the region where fisheries occur.

Impacts to eel production

Commercial and recreational fisheries

Licences to fish for eels and glass eels commercially are issued by the Environment Agency on a Regional basis. The Anglian RBD, due to the size of the coastline, has always had a strong coastal and estuarine fishery for eels. Recently, licences have been issued annually for about 200-270 small traps and 480-600 fykes: note that licences are issued per gear and not per person, so the number of fishers will be far less than gears.

There have been no elver catches reported in the Anglian RBD in the period from 2001, for which data were available.

The commercial yellow eel fisheries in the Anglian RBD operate throughout most of the year. Silver eel fisheries predominantly catch from September to December. The fishery operates in coastal waters, rivers and stillwaters across the RBD. The total declared catch data in the Anglian RBD and un-named sites likely to be in the RBD from 2005 to 2007, ranged from 13,065 to 3,709 kg of yellow eels, and 6,659 to 194 kg of silver eels.

Recreational angling for eels within Anglian RBD is generally carried out by specialist anglers, although historically eels often comprised a substantial component of fishing match weights. The vast majority of eels are captured whilst anglers are fishing for other coarse and game species and, in these circumstances, eel are usually returned alive to the water.

Habitat loss and Obstacles to migration

The rivers and waterways throughout the Anglian RBD have been highly modified over the last several centuries, and particularly in the latter half of the last century for the purpose of flood defence. This has lead to a restriction or loss of migratory routes and thus access to habitat, coupled with further complete loss of habitat resulting from land drainage practices to optimise the land available for agriculture.

The Anglian RBD is the driest region in the country and suffers with low flow problems across the region. These are greatly exacerbated by the high pressure posed by human consumption and agricultural irrigation, with many of the systems being artificially managed with cross-river pumping regimes in order to transport water to supply drinking water reservoirs. The net result of this is that many structures such as weirs and sluices pose an even greater obstacle to eel migration than their sheer physicality, due to the lack of water running over their surfaces at certain times of the year. This is coupled with the lack of water being allowed to pass out of these systems into the estuary.

A preliminary assessment has been carried out on the major obstructions to fish migration within the Anglian RBD, with obstacles graded individually in terms of the likelihood of eel passage.

There is currently little information on the level of eel entrainment within the Anglian RBD. However, it is likely that the numerous pumping stations across the RBD are responsible for mortalities of adult yellow eels and migrating silver eels. In the Anglian RBD there is only one hydropower installation recorded by the British Hydropower Association (**www.british-hydro.org**). The mortality of eel at this installation has not been estimated.

Predation

The Anglian RBD comprised 17% of the freshwater and lake habitat in England and Wales, and may expect to constitute 17% of eel consumption by cormorants: 5 to 7.4 tonnes. With the average length of eel taken at 40-55 cm or 150-200 g this suggests 25,000 to 50,000 eels consumed by cormorants within the Anglian RBD each year.

Water quality and pollution

Although a rural and predominantly farming region, there are also densely populated areas around many urban centres, with localised heavy industry. This largely reflects the close proximity of much of the RBD to London, combined with a long coastline that lends to import and export via shipping. Intensive agriculture is thought to apply the greatest pressure on water quality within the RBD.

Due to improvements in the regulation and practices of industry and waste-water treatment etc, the water quality within Anglian RBD is good. The General Quality Assessment results show that chemical quality (dissolved oxygen, biochemical oxygen demand and ammonia) for 90% of rivers was good or fair, with 47% classified as good to very good.

Biological quality (macro-invertebrates) showed 98% of the rivers were classed as good or fair quality with 79% classified as good or very good.

Nutrient (phosphate and nitrate) GQA status in the region for 2005 was high. These nutrient levels are the most likely parameter to be directly influenced by human activities. The results showed that 70% of the rivers in the Anglian RBD have high concentrations of nitrate, with 31% of these classified as very high. Also 77% of Anglian RBD rivers are classified as having "high" concentrations of phosphate, 9% of these are considered as "excessively" high. Many of the rivers and standing water bodies within the Anglian RBD are considered eutrophic (Environment Agency, 2004).

The seasonal low flows typically experienced across much of the RBD exacerbate the effects of any water quality problems, or pollution events, both in relation to reduced dilution capacity and general oxygen levels within the system.

Although pollutants such as PCBs and DDT were more common historically, they may still be present today in the waterbodies of the Anglian RBD due to the high levels of industry and agriculture in parts the region.

Diseases and parasites

Anguillicoloides crassus was first recorded in Britain in 1987, and in 1995 the National Rivers Authority (NRA) carried out investigations aimed at determining the status of *A. crassus* in eel stocks throughout the Anglian Region. The investigation took samples from a total of 44 still and flowing waters thought to best represent all the major catchments within the region, and from sites of particular local interest. The results showed all catchments within the region to be affected with the exception of the River Colne in Essex, which has subsequently been shown to have *A. crassus* in eel stocks (R.Wright, Environment Agency, pers. comm.). The results from the 1995 investigation were compared to those of a similar investigation carried out in 1989 and a strong increase in both the percentages of eels infected and the number of catchments affected by *A. crassus* was demonstrated (NRA, 1996).

Stocking

There are no records of applications to stock any life stage of eels within the Anglian RBD. However a regulated system of applying to stock is a relatively recent requirement and pre-regulation or illegal post-regulation stocking may well have been conducted. The only historic stocking activity known to have been conducted within the Anglian RBD was that of glass eels into the River Wensum between the 1920s and 1940s (G. Gamble, Environment Agency, pers. comm.). There is very little information available on numbers or the frequency of stocking, but it is thought that the owner of Lenwade Mill conducted this annually on a small scale. He stocked glass eels, believed to have originated from the Severn catchment, into the river above the mill and operated a fixed eel rack from the mill.

Data Catchments for POSE

There is a large amount of reliable and consistent long-term eel density and biomass survey data available for parts of the RBD, although individual length and weight data have only been collected for the last five years. However, more extensive monitoring of eel populations across the whole RBD is needed in order to enable greater confidence in estimating silver eel escapement. This includes glass eel recruitment into the RBD, targeted monitoring of yellow eel populations in freshwater, and silver eel escapement. By contrast, little is known about the potential or present production of eel from saline waters in the RBD.

In the absence of comprehensive information on recruitment, yellow eel production and silver eel escapement from the same river, the basins of the Anglian RBD are generally considered as data-poor for the purposes of this project.

5.16. Application of the EDA to the Anglian data set

The general method of applying the EDA model to an EMU is described in Chapter 4. Here, we describe the work achieved to apply the model to the Basque dataset, and the resultant predictions of yellow eel stock and silver eel production.

In the analysis, 4160 electrofishing operations on 911 electrofishing stations between June and November from 1981 to 2010 were selected (Figure 5.112).

These sites were considered in relation to their distribution over the time period (Figure 5.113), their elevation and distances from the sea and source and site elevation (Figure 5.114).

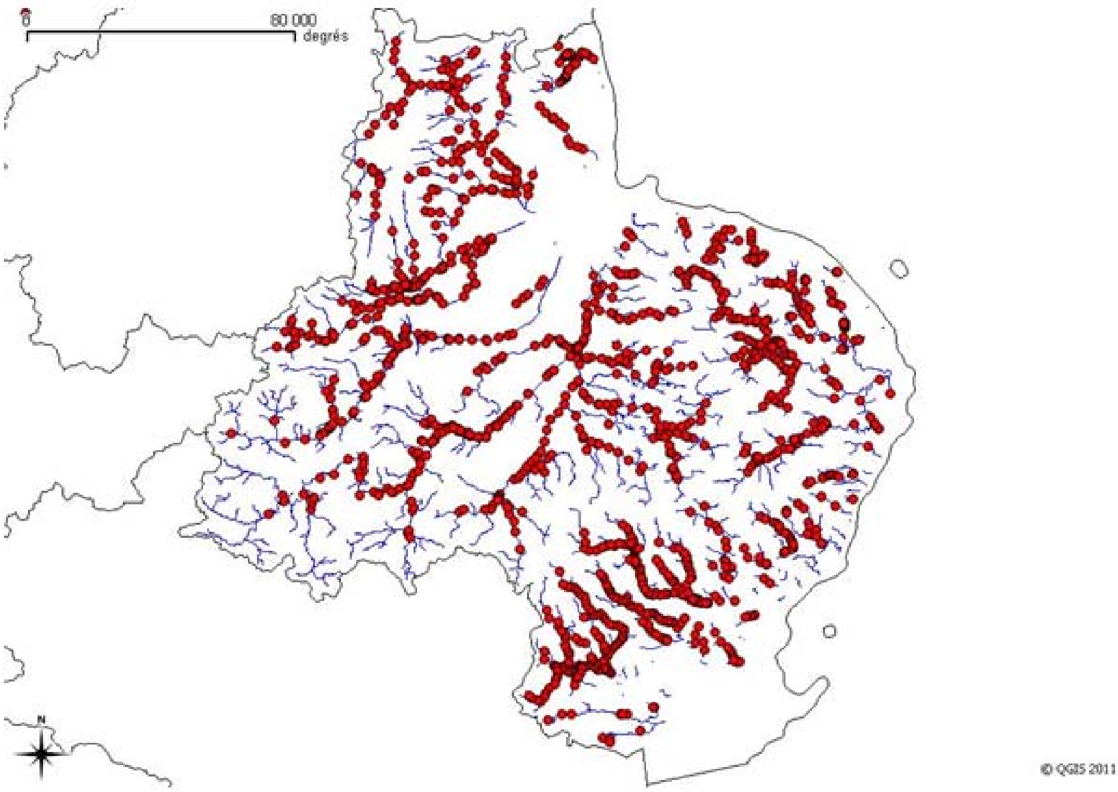


Figure 5.112. Sampling sites (in red) in the Anglian Emu catchment on the CCM river network (in blue).

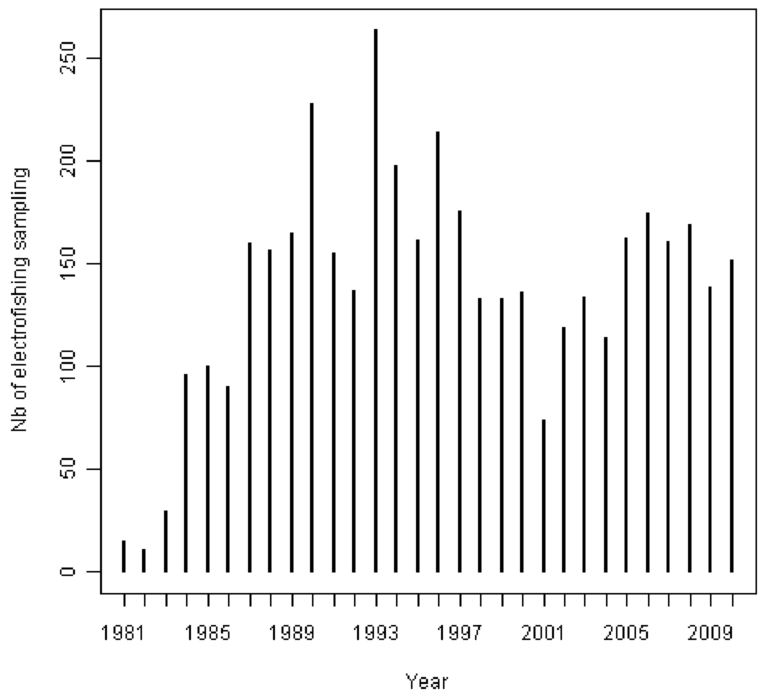


Figure 5.113. Number of electrofishing sampling per year in the Anglian Emu.

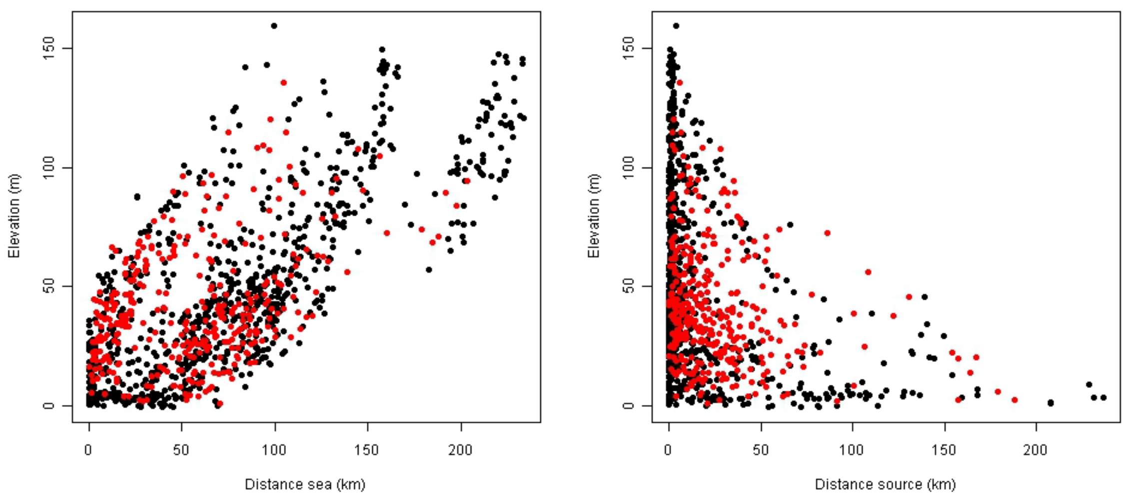
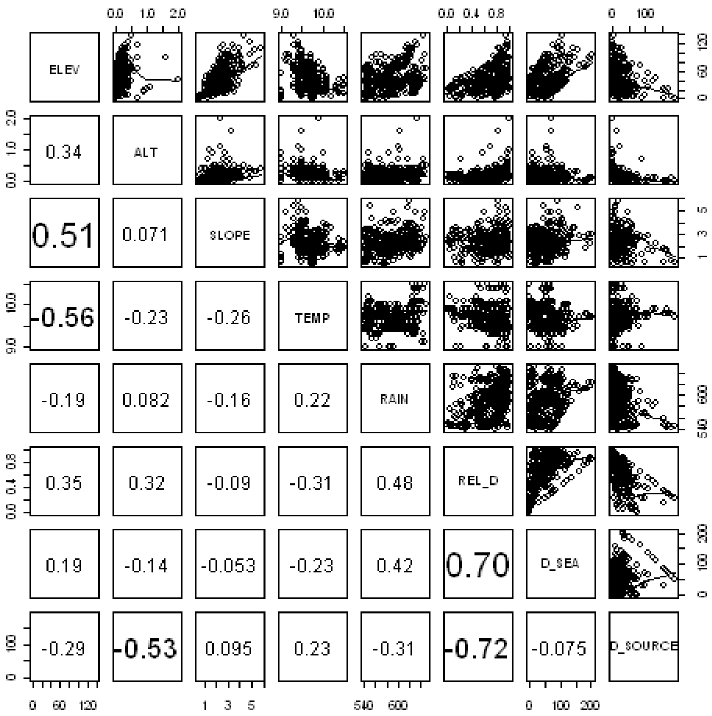


Figure 5.114. Sampling sites (in red) in the Anglian Emu catchment on the CCM river network (in black) depending on the elevation and the distance from the sea or from the source.

Selection of potential explanatory variables

The descriptor parameters are related to the characteristics of the river basin and the anthropogenic conditions (obstacles and land use). The CCM data set for the Anglian EMU provided information for each survey site on the distances to sea and source, relative distance, mean slope and elevation, altitudinal gradient, the area of land drainage upstream, mean annual temperature and rainfall, and land use cover (urban, agricultural and no impact).

A combination of 16 variables (year, month and a set of 14 variables) was tested. The Figures 5.115 and 5.116 show that there are several variables with tightly correlated predictors.



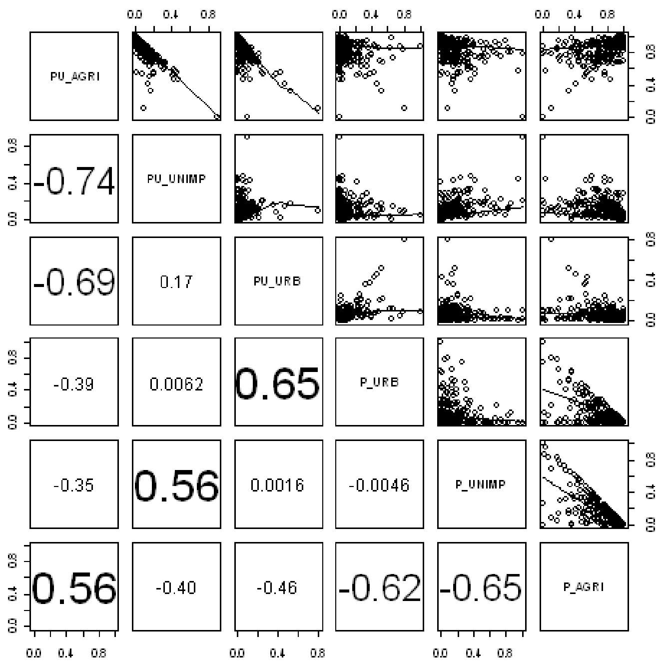


Figure 5.115. Pairwise correlation based on the Spearman rank correlation coefficient between pair of candidate predictors. With ELEV: elevation mean, ALT: altitudinal gradient, SLOPE: slope mean, TEMP: temperature mean, RAIN: rain mean, REL_D: relative distance, D_SEA: distance from the sea, D_SOURCE: distance from the source, PU_AGRI : upstream percent in agricultural use, PU_UNIMP: upstream percent in unimpact land use (p_up_no_impact), PU_URB: upstream percent in urban use, P_URB: local percent in urban use, P_UNIMP: local percent in unimpact land use (p_no_impact), P_AGRI: local percent in agricultural use. The numbers below the diagonal are correlations coefficients. The font size of the cross-correlation is proportional to its strength. The upper diagonal panels show the pair-wise scatterplots. The lines are Loess smoothers.

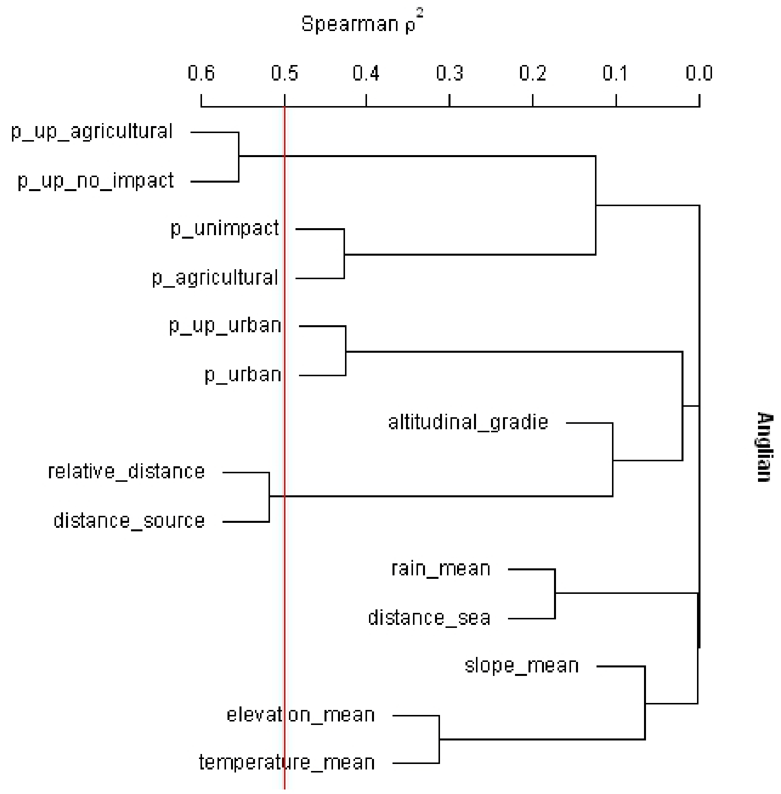


Figure 5.116. Dendrogram obtained by hierarchical cluster analysis of 14 candidate predictors for Brittany dataset, using the square of Spearman's rank correlation as similarity measure. The dendrogram is cut by a vertical line at Spearman ρ²=0.5.

According to the threshold of 0.5 for ρ^2, the following variables should be grouped:

- p_up_agricultural, p_up_no_impact
- relative distance, distance source

A separate entry was used for the pairs of variables with significant relationship (Figure 5.115) and for the 2 groups of variables described in Figure 5.116 to avoid spurious correlation.

The combination of the different groups results in 1890 models to be tested.

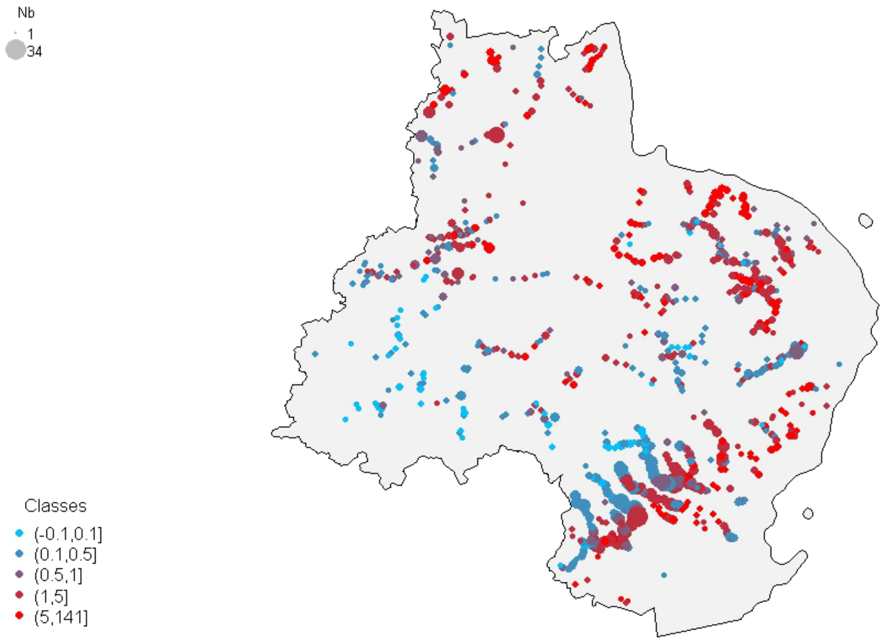


Figure 5.117. Map of density observed densities. All years. Eel.m-²

Model Testing Procedure

For Anglian, we have applied a gam model with a gamma distribution and a logarithm link (only a gamma model to the density because only positive densities were available). 1890 models have been tested.

Model results for density model are summarized in Table 5.25. The AIC function indicates a better fit of the response variable at the year and month of electrofishing sample with elevation, rain, distance to the source, distance to the sea, the upstream percent of urban use and the upstream percent of no impact land use). GAM explained 36% of the deviance of the abundance of the yellow eel. The effects of seven explanatory variables in the model (year, altitudinal gradient, elevation, distance source, distance sea, p_urban, p_no_impact) are significant (Table 5.26). The Spearman rank correlation between the observed values and the fitted values is statistically significant (ρ=0.4229597, p-value< 2.2 10^{-16}, Figure 5.118).

Table 5.25. Model selection results using Akaike's information criterion (AIC) for density model (gamma model) analysis of factors that affected eel abundance. Models within 2 AIC units of the minimum AIC had substantial support.

Model	year	month	elevation mean	slope mean	rain mean	distance sea	distance source	p_urban	p_agricultural	p_no_impact	p_up_urban	p_up_agricultural	p_up_unimpact	AIC s=3
1	x	x	x		x	x	x				x		x	12867.4
2	x	x	x		x	x	x						x	12889.2
3	x	x	x		x	x	x	x		x				12893.4
4	x	x	x		x	x	x		x					12900.2
5	x	x	x		x	x	x					x		12901.8
6	x	x	x		x	x	x			x				12939.1
7	x	x	x		x	x	x	x						12947.4
8	x	x	x		x	x	x				x			12971.8
9	x	x		x	x	x	x				x		x	12975.2
10	x	x		x	x	x	x						x	12983.1

Model Goodness of fit (density model)

The best density model selected is:

d ~ s(annee,3) +s(month,3)+ s(elevation_mean,3)+ s(rain_mean,3)+ s(distance_source,3)+ s(distance_sea,3)+ s(p_up_urban,3)+ s(p_up_no_impact,3)

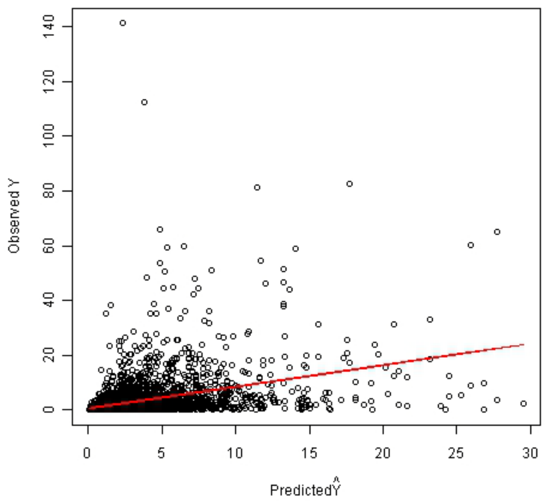


Figure 5.118. Observed vs. predicted regression scatter plot.

Table 5.26. Table of effects, DF for terms and Chi-squares for non-parametric effects.

	Df	Npar Df	Npar F	Pr(F)	
(Intercept)	1				
s(annee, 3)	1	2	4.143	0.015934	*
s(month, 3)	1	2	16.422	7.877e-08	***
s(elev_mean, 3)	1	2	3.616	0.026964	*
s(rain_mean, 3)	1	2	3.842	0.021516	*
s(distance_source, 3)	1	2	53.160	< 2.2e-16	***
s(distance_sea, 3)	1	2	6.440	0.001612	**
s(p_up_no_impact, 3)	1	2	15.037	3.114e-07	***
s(p_up_urban, 3)	1	2	5.546	0.003934	**

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

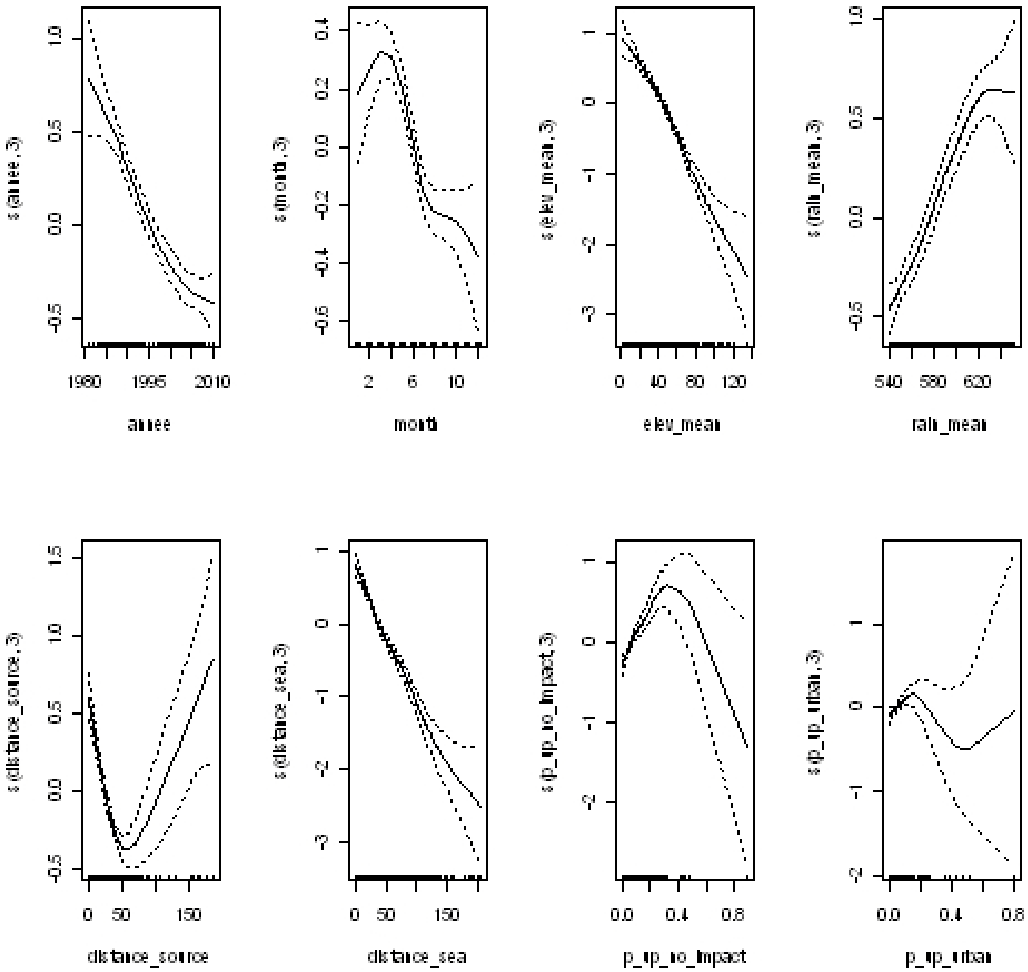


Figure 5.119. Response curves of each variable included in the generalized additive model (GAM) for the density model for Crepe EMU. The solid lines represent the estimated smooth function and the dashed lines the corresponding 95% confidence limits.

Map of model residuals and predictions

The map below (Figure 5.120) shows the mean residuals for operations where positive densities were observed. The residuals correspond to observed -predicted values, hence a blue spot (negative value) will indicate a predicted value larger than the observed one, and a red point (positive value) a predicted value lower than the observed one. The size of the points gives an indication of how many electrofishing operations occurred at that point and thus of the « weight » of that station in the model.

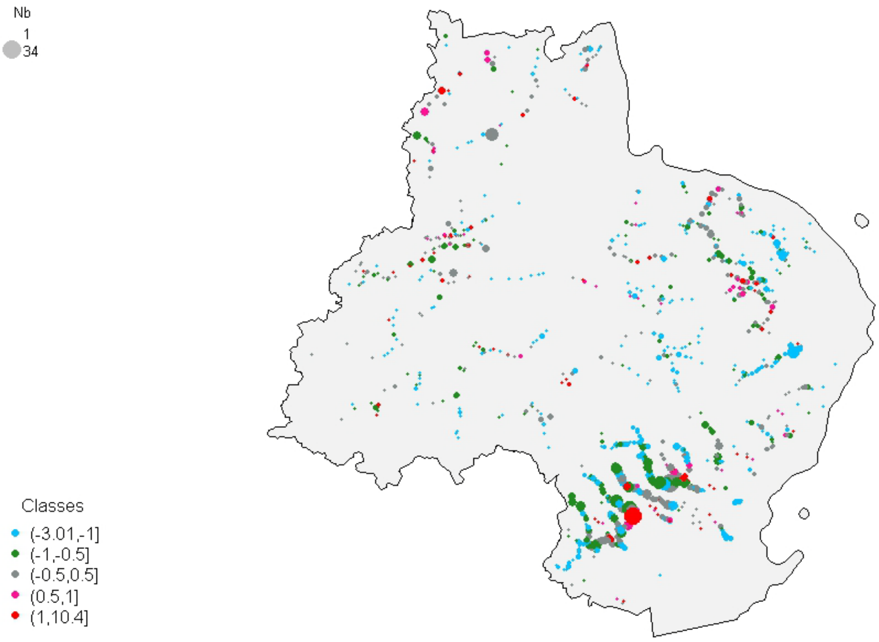


Figure 5.120. Mean residuals per stations: density model. Eel.m^{-2}

The next map (Figure 5.121) gives the same information but residuals are « jittered » around the point. There is no clear spatial trend in the presence of large positive or negative residuals.

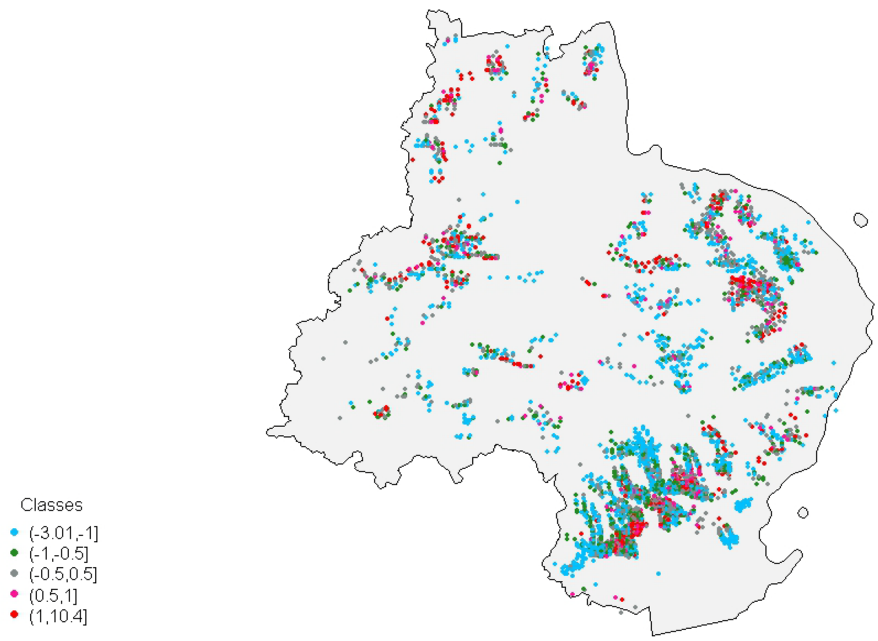
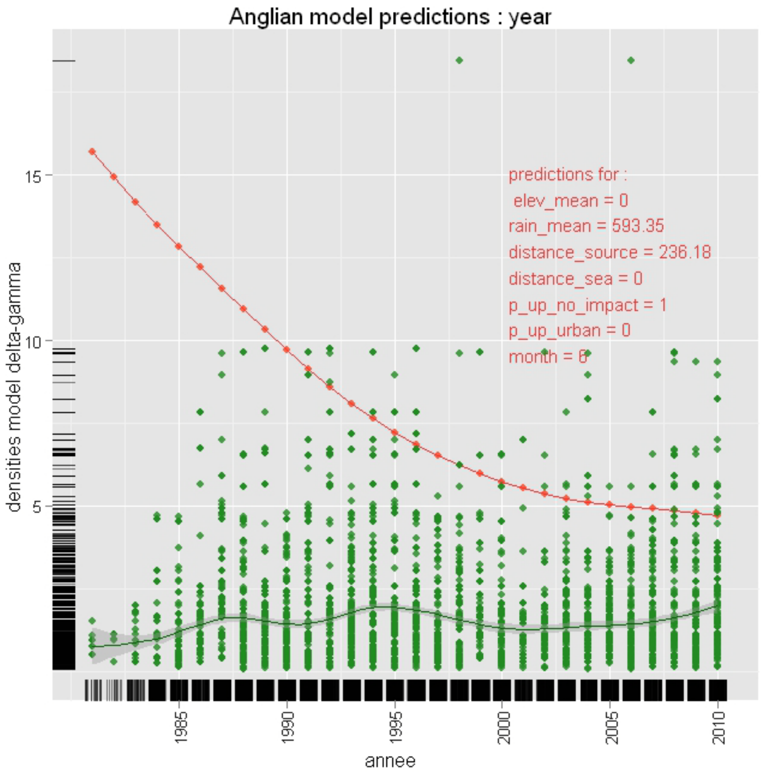


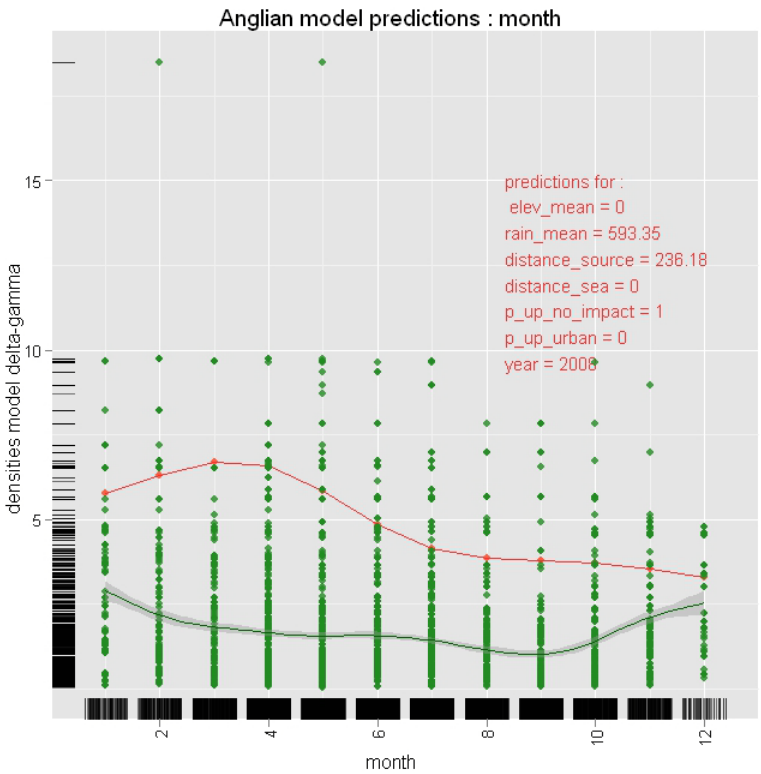
Figure 5.121. Residuals per year: density model. Eel.m-^{2}

Final model

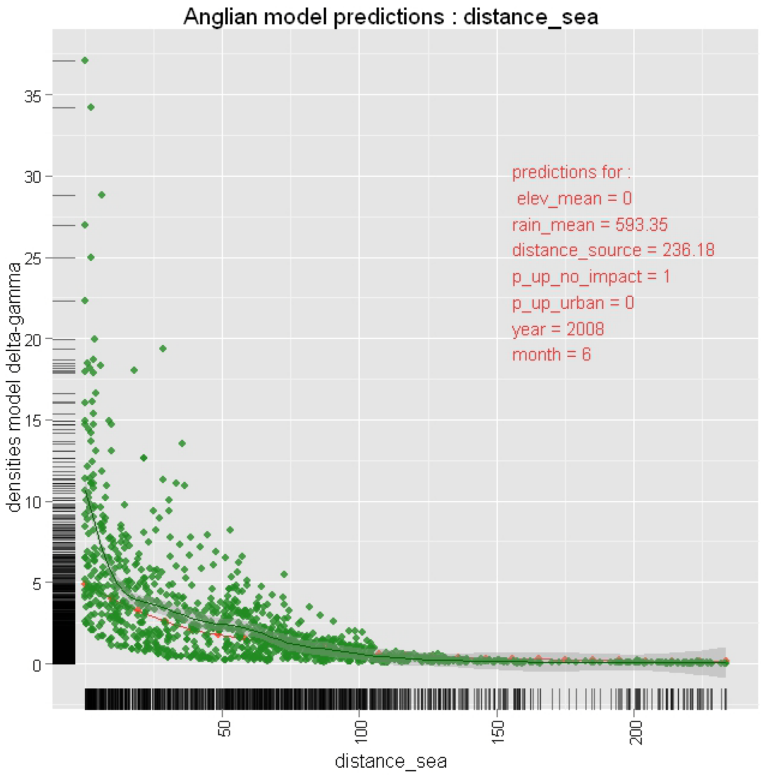
The final model is the gamma model. The model is used to predict densities on all the river stretches of the EMU. The following figures (not numbered) give the trend (red curve) of an effect given that all other parameters are fixed (see figure for the value of fixed parameters) as well as the predicted density of each river stretch of the CCM (green points) and its average along the examined effect (green curve). The rugs on the scale have been moved slightly to indicate the density of observation. All results presented are eel.m^{-2} but the model would need to access all null densities to be complete, densities presented there are overestimated.



Anglian model predictions : year
densities model delta-gamma
annee
predictions for :
elev_mean = 0
rain_mean = 593.35
distance_source = 236.18
distance_sea = 0
p_up_no_impact = 1
p_up_urban = 0
month = 6



Anglian model predictions : month
densities model delta-gamma
month
predictions for :
elev_mean = 0
rain_mean = 593.35
distance_source = 236.18
distance_sea = 0
p_up_no_impact = 1
p_up_urban = 0
year = 2008



Anglian model predictions : distance_sea
predictions for :
elev_mean = 0
rain_mean = 593.35
distance_source = 236.18
p_up_no_impact = 1
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
distance_sea
0
5
10
15
20
25
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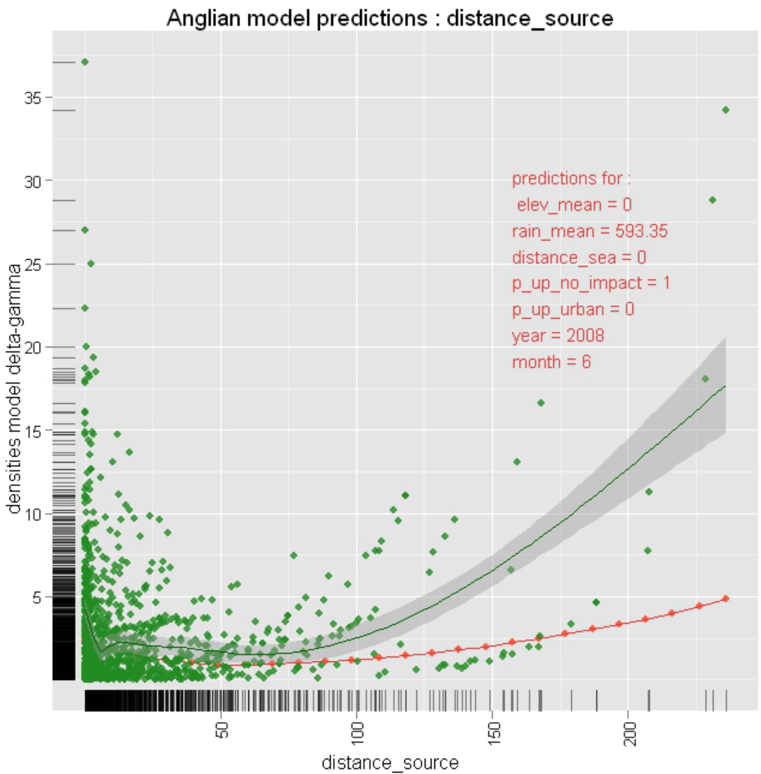
35
50
100
150
200



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rain_mean = 593.35
distance_sea = 0
p_up_no_impact = 1
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
distance_source
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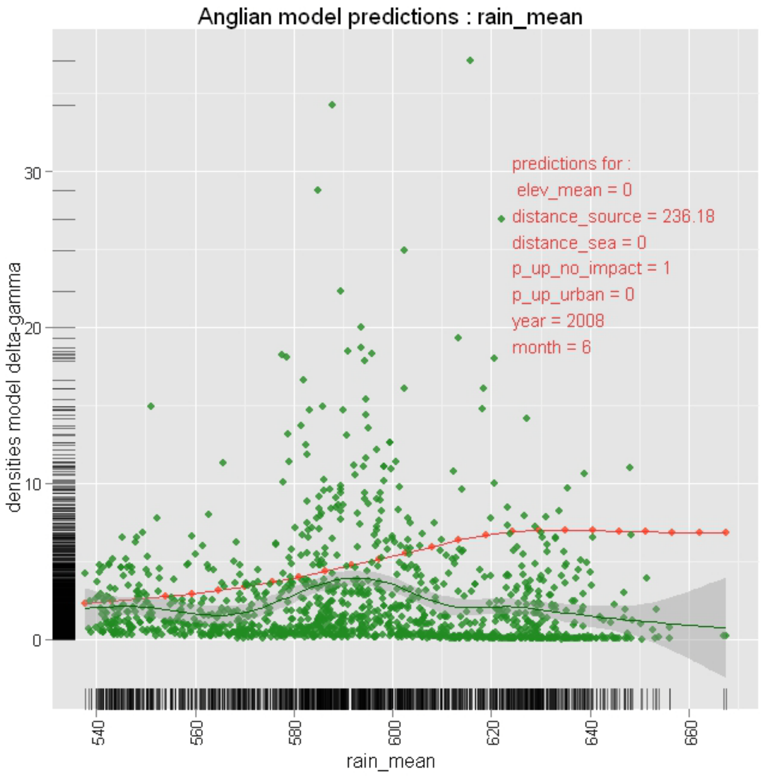
Anglian model predictions : rain_mean
predictions for :
elev_mean = 0
distance_source = 236.18
distance_sea = 0
p_up_no_impact = 1
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
rain_mean
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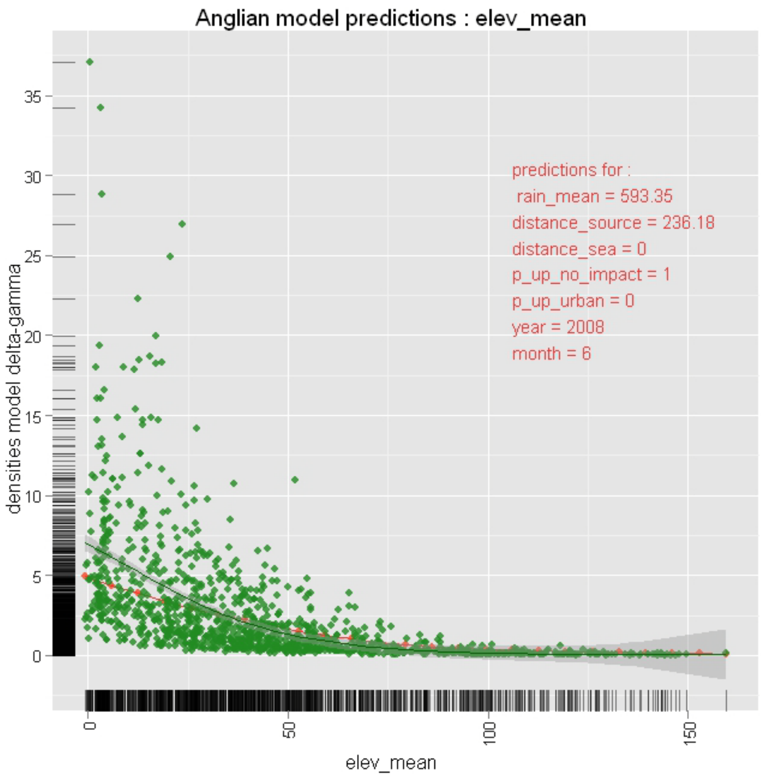
560
580
600
620
640
660


Anglian model predictions : elev_mean
predictions for :
rain_mean = 593.35
distance_source = 236.18
distance_sea = 0
p_up_no_impact = 1
p_up_urban = 0
year = 2008
month = 6
densities model delta-gamma
elev_mean
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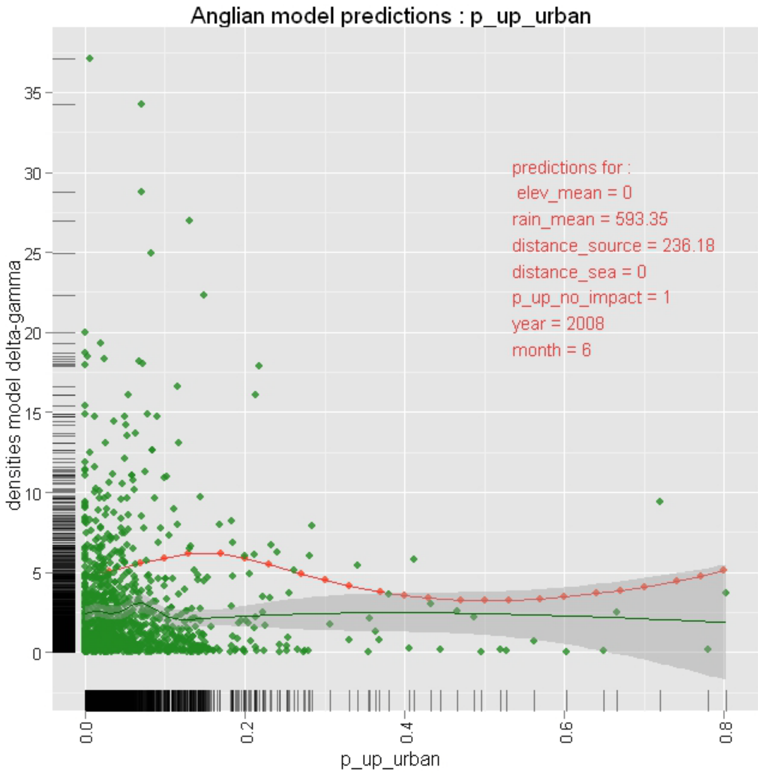


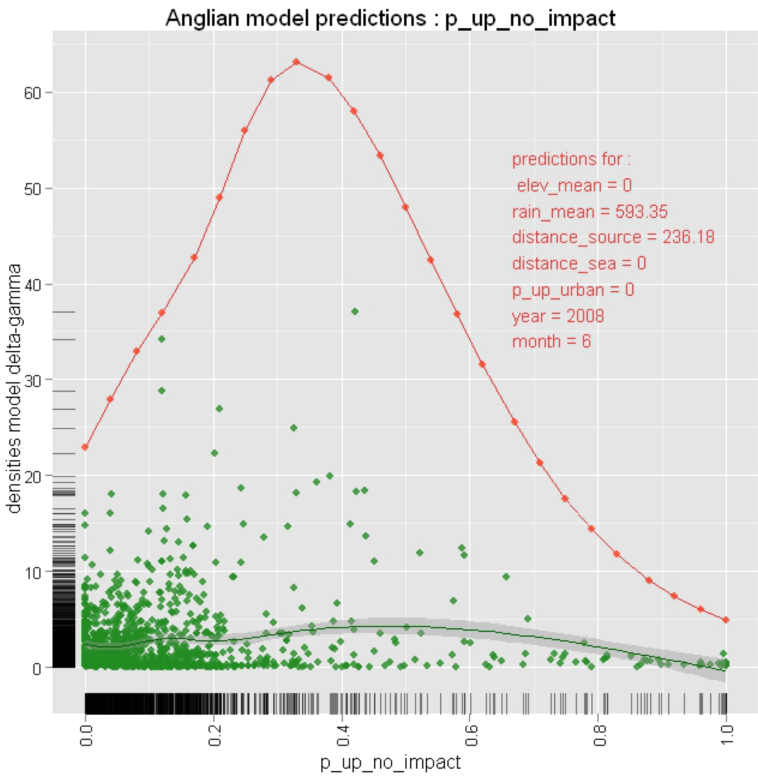


Figure 5.122 gives the yellow eel production along the examined variable. On the upper corner an inset chart also give the cumulated wetted area along the same variable that supports that production.

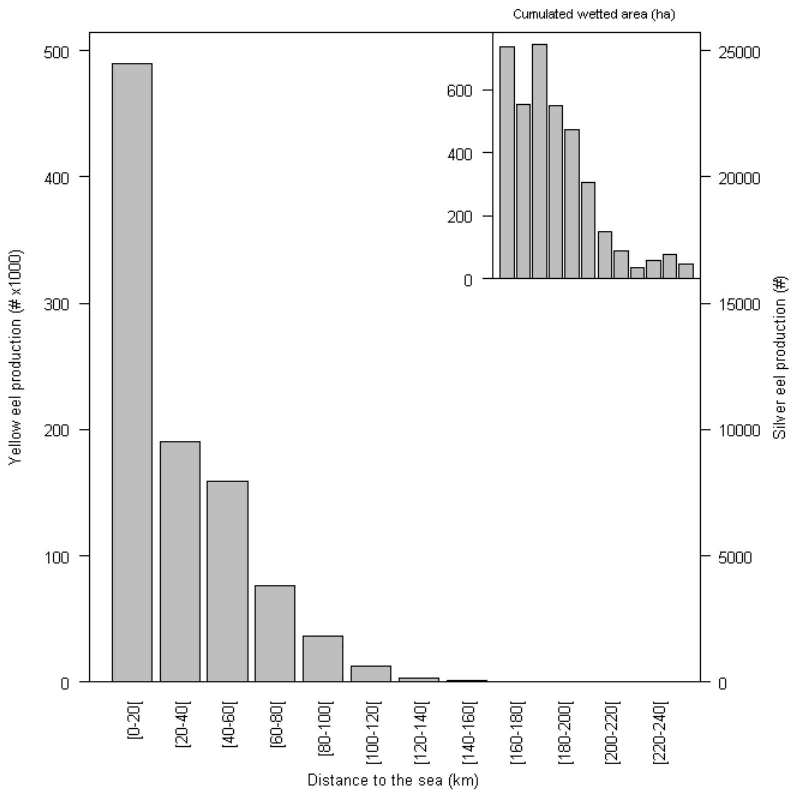


Figure 5.122. Yellow eel production along the distance from the sea.

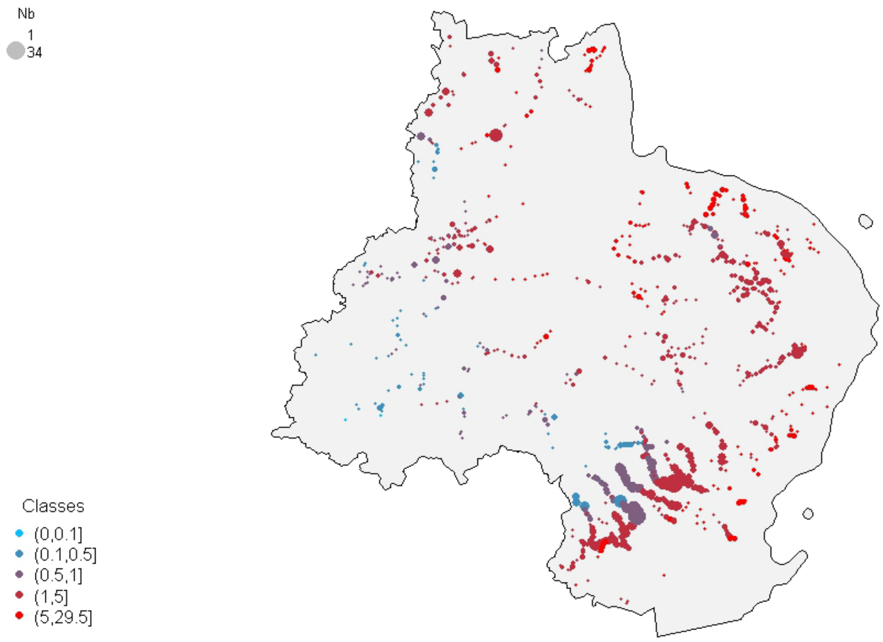


Figure 5.123. EDA predictions for the full model, the size model is given according to the number of electrofishing at that point.

EDA predictions for the Anglian EMU

$B_{current}$ is the silver eel escapement of a given year. The density of yellow eel is multiplied by the wetted surface of stretch (which is simply the product of the length of the stretch and the river width) to estimate the number of yellow eels in each stretch. The amount of yellow eels in the EMU is then estimated by summing the results for all stretches.

The potential escapement of silver eel is calculated by multiplying the yellow eel abundance in each stretch with a conversion rate. Little information is available about the relationship between yellow eel and silver eel stocks (Acou, 1999, Robinet *et al.,* 2007, Feunteun *et al.,* 2000). Feunteun *et al.* (2000) have estimated that between 5 and 12 % of the yellow eels start the silvering in the Frémur catchment. In the present version of EDA a constant conversion rate of 5% was chosen as a default value. This constant rate conversion of eel in equivalent silver eel is based on the assumption of density-independent biological processes.

This potential escapement in number in then converted into biomass with the mean weight of a silver eel specific to each EMU, in this case derived from the fishery data provided for the Anglian EMU. The current biomass ($B_{current}$) is calculated by subtracting the anthropogenic mortalities on silver eel to the potential escapement in biomass.

Best achievable escapements (B_{best}) can be calculated by the current biomass artificially forced to null anthropogenic impact (no dam, land use mortality to "no impact" and silver eel catch to 0) added with silver eel biomass corresponding to anthropogenic mortalities at glass eel and yellow eel stages (ICES, 2010).

The pristine biomass B_0 is the spawner escapement biomass produced when there were no anthropogenic impacts and recruitment was at its high historical level. In EDA, B_0 is simply the average of B_{best} for the period before the crash in recruitment.

Table 5.27 summarises the EDA estimates of the total numbers of yellow and silver eels in the Basque EMU. Figure 5.124 presents the predicted time series of silver eel escapement from 1980 to 2010.

Table 5.27. Model outputs for EDA simulations of the Anglian EMU

Model output	Value
Total water surface (km²)	38.23
Average number of yellow eel per 100 m²	2.537
Average number of silver eel per 100 m²	0.127
Total number of yellow eel	969 639
$N_{current}$: Total number of silver eel	48 481
$\mathbf{B_{current}}$(t=2010) in kg	18365
$\mathbf{B_{best}}$ in kg	27954
B0 (Bbest<1980) in kg	56246

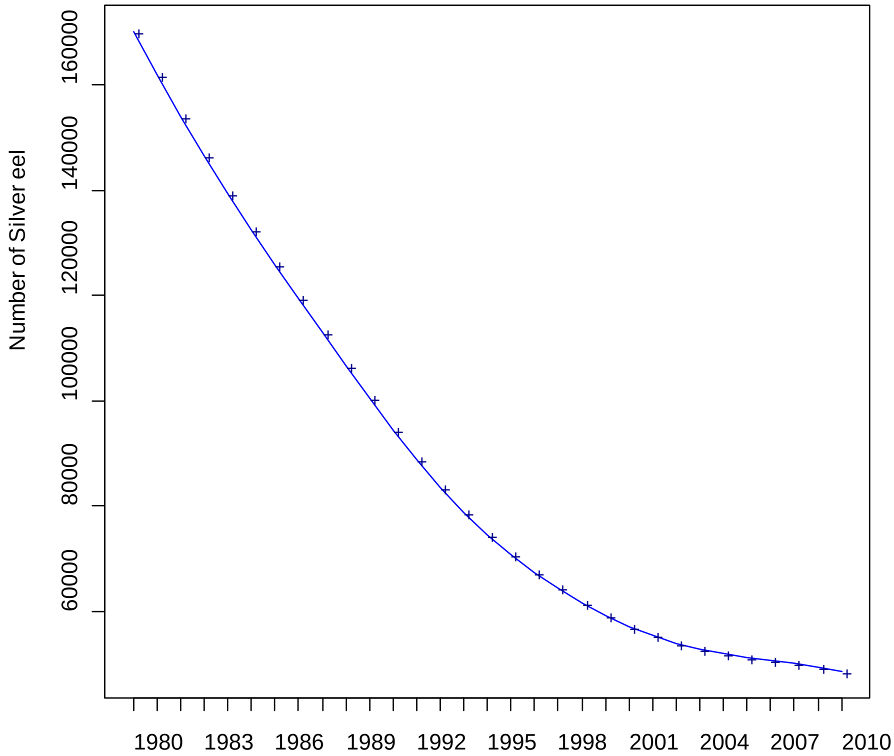


Figure 5.124. The EDA predicted silver eel production for Anglian EMU over the period from 1980 to 2010

Changes made during the tuning process for the application of EDA to the Anglian EMU

In the second application phase changes of units for yellow eel densities have been carried out. To evaluate Bcurrent, Bbest and Bpristine, the mean weight of each life stage (glass eel $\overline{w}_{glass}$, yellow eel $\overline{w}_{yellow}$ and silver eel $\overline{w}_{silver}$) and the anthropogenic mortalities on glass eel Y_{glass}, yellow eel Y_{yellow} and silver eel Y_{silver} have been used. The model parameters used to evaluate $B_{current}$, B_{best} and $B_{pristine}$ are given in Table 5.28. Table 5.29 summarises the EDA estimates of the total numbers of yellow and silver eels in the Basque EMU. Figure 5.124 presents the predicted time series of silver eel escapement from 1980 to 2010.

Table 5.28. Data input for silver eel estimation for the Anglian EMU

With Y_{glass}(t= $2010-\tau$), Y_{yellow}(t= $2010-\tau+\lambda_{yellow}$) and Y_{silver}(t=**2010**) in kg.

M (year^{-1})	τ (year)	λ_{yellow} (year)	$\overline{w}_{glass}$ (g)	$\overline{w}_{yellow}$ (g)	$\overline{w}_{silver}$ (g)	Y_{glass} (kg)	Y_{yellow} (kg)	Y_{silver} (kg)
0.1386 (Dekker 2000) or 4.81 for glass eel during ¼ year and 0.1386 after (Lambert, 2008)	18.8[1]	10	0.3	157.2[2]	382.8[3]	0	13065	194

[1] (SD = 0.27; range 13-31) as the eel will be mainly female. Data from the river Avon.
[2] (SD = 167.2)
[3] (SD = 12.7; range 132.4 - 1120.0) from the river Avon

Table 5.29. Model outputs for EDA simulations of the Anglian EMU

Model output	Value
Total water surface (km²)	38.23
Average number of yellow eel per 100 m²	2.537
Average number of silver eel per 100 m²	0.127
Total number of yellow eel	969 639
$N_{current}$[1]: Total number of silver eel	48 481
$B_{current}$(t=2010) **[1]** in kg	18365
B_{best}[1] in kg	27955[2]
B0[1] ($B_0 = \overline{B_{best}\left(t<1980\right)}$ in kg	65642[3]

[1] For Anglian, no data with null densities were available, so we should check those values again.
[2] should be t-tau+lambdayellow here I only have 2005 13065 kg
[3] Mean B best calculated on historical period 1981 :1985, earlier data not available

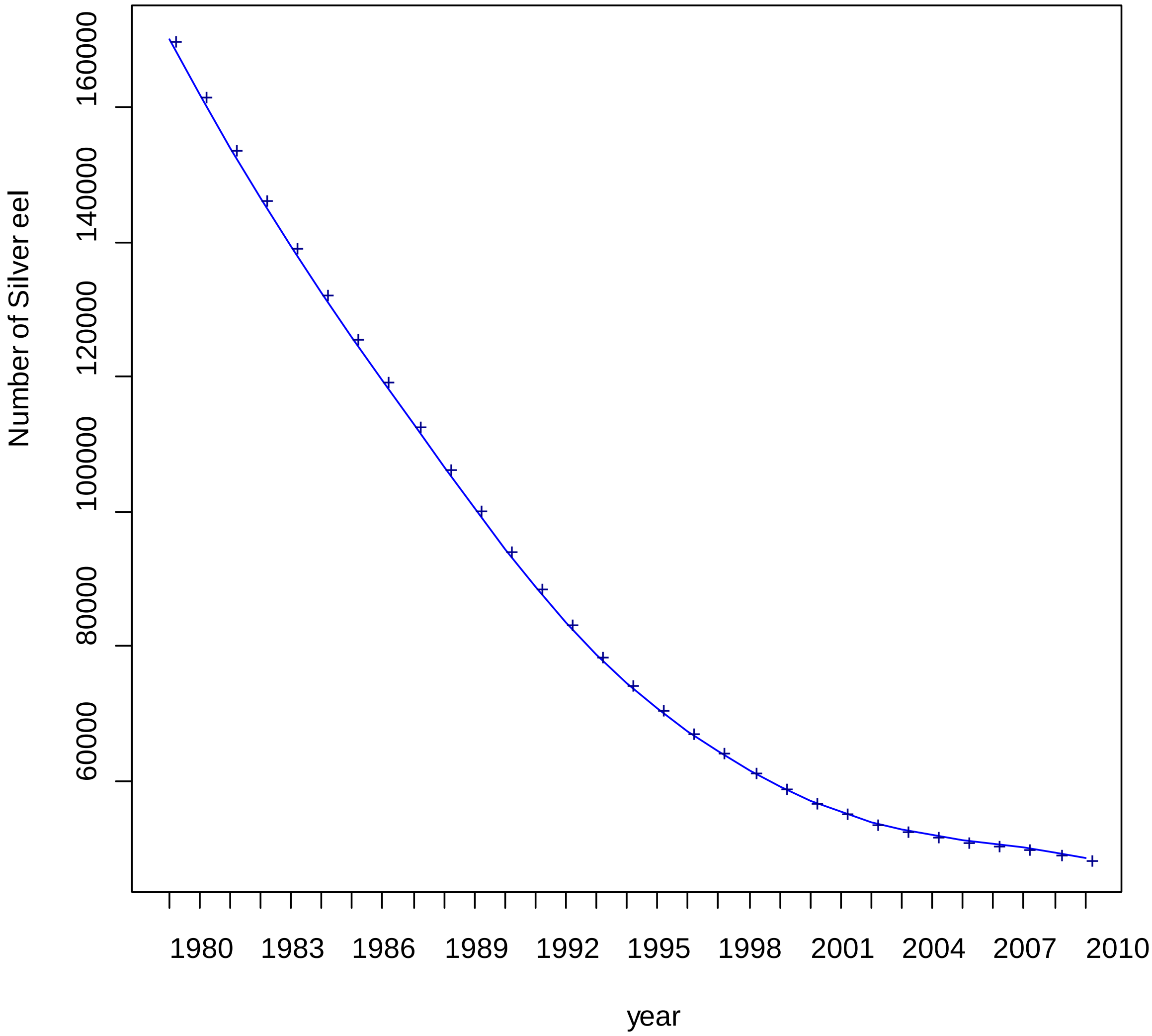


Figure 5.124. The EDA predicted silver eel production for Anglian EMU over the period from 1980 to 2010

5.17. Guide to Managers on the selection of suitable assessment models

This work identified that each of the models is suitable for application to a different character set of eel data and scenarios. Given the broad range of assessment data, impacts and management scenarios that may occur in eel management units across Europe, there are a vast number of possible combinations, making it impracticable to list these and assess each of the four models for their suitability. As an alternative, therefore, we have considered the broad types of scenarios to which the assessment models are probably best suited. This guide will assist managers in identifying the model or models that should be most appropriate for their specific circumstances.

DemCam and GEM are similar in that they are typically applied to a single spatial unit. As a consequence, they are best applied to those areas where the eel production processes are not expected to vary much throughout the management unit. DemCam was developed to model eel production in lagoons, so clearly it is best suited to being applied to such environments. GEM was developed for the River Elbe, but on the basis of a series of production process values being representative across the entire river network.

SMEP II is an eel life history model in the same manner as DemCam and GEM, albeit that they each only model the life history from recruitment to spawner escapement. However, SMEP II specifically incorporates the spatial complexities of a river basin, including any network of rivers, lakes, estuaries and lagoons. Although the descriptions of growth, natural mortality, sex differentiation and silvering are common throughout the river basin, the ability to model the dispersal of eel, the effects of density dependence, and to localize impacts allows a more complex and spatially explicit assessment.

DemCam, GEM and SMEP II all require information describing these biological processes. Although each model can use information taken from the literature, the results from our phase 1 tests of the Crepe data set highlight the risks this poses of introducing additional uncertainty in the model results and the potential pitfalls in assuming that information taken from other rivers, districts or regions is representative of the eel population in question. Local knowledge of growth and natural mortality rates, and of recruitment levels, appears particularly important and emphasizes the importance of collecting local field data.

The EDA model adopts a very different approach to the other three models, at its core relying on identifying relationships between yellow eel densities and habitat characteristics, extrapolating these across the area in question, accounting for any losses due to impacts, and applying conversion rates to produce results for silver eels. Two key consequences of this approach are that EDA is far less reliant on local knowledge of eel production processes, and that it is best applied at much larger spatial scales than the other models – typically at RBD / EMU scale.

This ability of EDA to produce results at the default spatial scale of the Eel Management Plans is no doubt appealing. The other three models have been applied at much smaller management units – even though the GEM was developed for the Elbe river basin district, this was in effect a single, albeit very large, river basin. As such, DemCam, GEM and SMEP II are best applied to "index" rivers and the results then extrapolated to other rivers, etc within the River Basin District. On the other hand, EDA requires a substantial distribution of eel density and habitat data from across the management unit, so is best applied to those areas with comprehensive national eel survey programmes.

Natural mortality data are rare, especially river-specific, and therefore most model applications will use the average values developed by Bevacqua *et al.* (2011). In the absence of better, more site-specific data, those values derived on the basis of Bevacqua *et al.* may be an improvement from using a single default value but the outputs should still be treated with caution. Applicants and assessors must recognise this limitation of the data and the models. To some extent, in the absence of site-specific data, it makes sense to standardise life history parameters across models and across regions so that we are at least all working to the same set of rules. A gap identified which will require attention in the future is the lack of assessment methodology(s) for quantifying eel production in large water bodies (e.g. lakes, large rivers, large estuaries, coastal waters). The ICES SGAESAW began the process of developing assessment methods for marine eels in 2007, with a synthesis of knowledge, but the next steps to actually develop methods and indicators have not been taken yet.

Chapter 6. The development and testing of a method to assess eel production in the complete absence of eel data

6.1. Introduction

Given the acknowledged paucity of Data Rich or monitored catchments with established biological reference points for eels (e.g. in Ireland there were 5 catchments with rich eel data, 17 with poor data and 242 with little or no eel data), extrapolation of knowledge gained from the data rich monitored catchments to catchments where little or no eel data exists is therefore required. Similar extrapolations may also be required within catchments between data rich areas (e.g. a lake) and data poor areas where no eel data has been gathered.

Setting and transporting biological reference points between data-rich and data-poor catchments has been well established for salmon (Crozier *et al.,* 2003; Prévost *et al.,* 2003; O'Maoileidigh *et al.,* 2004;) by using a Bayesian Hierarchical Analysis. The development of Bayesian hierarchical stock and recruitment analyses (BHSRA) for transporting stock and recruitment parameters from monitored rivers to rivers without stock and recruitment data has allowed Conservation Limits to be established for individual rivers independently of catch data. The BHSRA for salmon relies on a measure of the production units available in each river. In the absence of more refined and commonly available habitat variables, an intermediate, such as wetted area, can be used to quantify salmon production for widespread transport of stock and recruitment parameters from well-studied monitored rivers, provided the quantity of ''non-productive'' riverine habitat is in similar proportion from one river to the next. Wetted area and latitude were used as covariates to explain the variations in salmon stock-recruitment data series from 15 rivers in the NEAC area (Crozier *et al.,* 2003).

A simple methodology was developed for estimating silver eel production in Ireland based on linking the underlying geology with eel growth rates, calibrated by the few production estimates available, using associative regression analysis linking catchment habitat and eel production with eel growth and geology as proxy variables (see Irish Eel Management Plans National Report ( ) and the EIFAC/ICES Working Group on Eel 2008 Report, Chapter 4.3.2.2, (.

The limited understanding of eel biology and the relationships between habitat and production at present will inevitably include a significant degree of uncertainty in any estimates (eels are not alone in this) and to date this uncertainty has not been taken into account in many of the models.

This chapter addresses the need for a valid framework which uses silver eel production and escapement, where data exists, and attempts to transfer this information/knowledge to catchments where nothing is know about the local eel stock. It is critical that this framework accounts fully for uncertainty of its parameters and sub-models and the uncertainty implicit to extrapolation. It should be acknowledged that eels are not salmon, and the discrete stock-recruitment production relations described for salmon above do not exist for eel. The lack of recruitment data for individual catchments and the many anthropogenic influences on colonisation of catchments, such as obstructions to upstream migration, fisheries and pollution, may make it difficult to estimate actual silver eel production at the catchment level.

In establishing a possible framework for eel, a number of attempts have been made at describing the physical variables required at the catchment, and sub-catchment level (ICES, 2006; INDICANG). However, as the models become more complex, they require more parameters and they invariably become more sensitive to the parameter values. Therefore, before using the model in a predictive mode, a great challenge is to provide reliable parameter estimates and a fair assessment of their uncertainty (Rivot *et al.,* 2004).

In the case of salmon, wetted area and latitude were the co-variables which were available for all data-poor catchments and adequately captured the variation in stock recruitment across the range in the North Eastern Atlantic (Crozier *et al.,* 2003). The analysis for eel in Irish waters showed a relationship between geology and growth rate and between geology and silver eel production (ICES, 2008). A European wide analysis of eel growth rate data found that the yearly sum of temperatures above 13°C (TempSUP13), the relative distance within the catchment, sex, age class, salinity class and depth class were the best predictors of eel growth (Daverat et al., in press). Growth rate was greater in habitats close to the sea and in deep habitats. The temperature variable had one of the greatest predictive powers in the model.

Given that eel stocks vary widely even within a narrow geographical range, over-reliance on local monitored waters can lead to a major under-estimation of the uncertainty for rivers without data. Assuming that all rivers have the same fundamental eel characteristics is certainly an over-simplification, whereas assuming that each river is fully independent implies that there is nothing to learn from monitored rivers about eel productivity in general. We therefore propose an analysis framework that uses silver eel productions and proxies from data rich rivers to inform as to the variability of silver eel productions in general and their relation with covariates (e.g. climate, geology, water-chemistry). The analysis framework must also fully account for the stochasticity of each individual parameter (habitat quantification, silver eel weir efficiency and others) within the analysis framework and its sub-components (e.g. growth, mortality & production models) to fully quantify the uncertainty in estimates of silver eel production extrapolated from data-rich to data-poor situations.

A full description of the modelling process and framework is given in Annex D, but here we provide a summary description as a background to the reporting and consideration of the results.

6.2. Data sources

The primary data input into the Bayesian model were as follows:

1. Growth rate data from a wide range of sites across Europe.

Daverat el al. (in press) have previously collated lengths and ages from eels from across Europe and these data were made available for use in this study (27861 eel, spanning 172 sites and 15 countries) (Figure 6.1). Some obvious outliers were excluded, and some data points were excluded owing to doubts about manner in which the eels were aged and therefore about the quality of the growth rate estimates. Data from water bodies where stocking was known to have taken place were also excluded.

2. Silver eel production estimates from a set of European sites were collated as part of the 2008 ICES/EIFAC working group on Eel in (ICES, 2008) and were made available for this study (See ICES, 2008 report for references).

Additional estimates of silver eel production data were extracted from published records or supplied by institutions and individuals (Moriarty, 1988; Bisgaard & Pedersen, 1990; Acou *et al.,* 2009; Bilotta *et al.,* 2011) (Diputación de Gipuzkoa 2009 & EKOLUR pers.comm; Environment Agency, Jon Hateley & Ros Wright pers. comm.; Norwegian Institute for Nature Research NINA, Hindar, Vollestad & Thorstad *pers comm*.) (Figure 6.1).

3. Candidate explanatory variables for growth rates and silver eel production were collated for all sites where eel growth rate and production data were available (Table 6.1).

The prerequisite for the explanatory variables was that the data should be openly available for any location in Europe, irrespective of whether there were any eel data (i.e. data-poor and no-eel-data areas). The CCM (Catchment characterisation and Modelling) dataset provided much of the catchment data required for modelling eel growth and production in freshwater (Vogt *et al.,* 2007). This dataset is publicly available, and contains information for all the catchments in Europe, including slope, elevation, area, river lengths, lake areas, temperature, rainfall, and Strahler index. The unit at which data were extracted from the CCM dataset was 'Seaoutlet' (WSO_ID). Each seaoutlet comprises a number of catchments in a common river basin, which combine to enter the sea at one point. Therefore, from the point of view of glass eel returning to the coast, the relevant data is for the seaoutlet as a whole, rather than individual catchments. Eel data (either growth rate of silver eel production or both) were available for 76 seaoutlets out of 80549 in the CCM dataset (Figure 6.1).

However, the CCM dataset does not contain any data on the possible productivity of a river basin or catchment, which was felt to be a crucial component of any model attempting to estimate eel production. This appears to be a very large data gap at a European level. Many countries have some national monitoring programs and geological surveys which can supply data on the trophic status of water bodies or the underlying geology of river basins. However, at the moment there does not appear to be any way to access this information, short of contacting local contacts and offices. This is what was undertaken for this project in order to include some data about trophic status and underlying geology of each seaoutlet.

A number of the data in the growth rate and silver eel production sets were taken from marine sites, where freshwater influences are presumed to be minimal. A smaller number of explanatory variables were available for these sites.

4. Recruitment indices for Europe ('Baltic' and 'rest'), were extracted from the 2010 ICES report with the permission of the Country Report leaders (ICES 2010) to allow the relationship between growth rate, mortality and production to be linked.

In order to match the relevant production year with the relevant recruitment index, the age of silvering was calculated for each site where production data were available. A linear regression between age at silvering and latitude was reported by ICES (2010), and this relationship was used to calculate the age at which male and female silver for each sampled latitude. This age was then subtracted from the production year to give the relevant recruitment index year. Where sex was not available, the age at silvering was taken to be midway between male and female age at silvering.

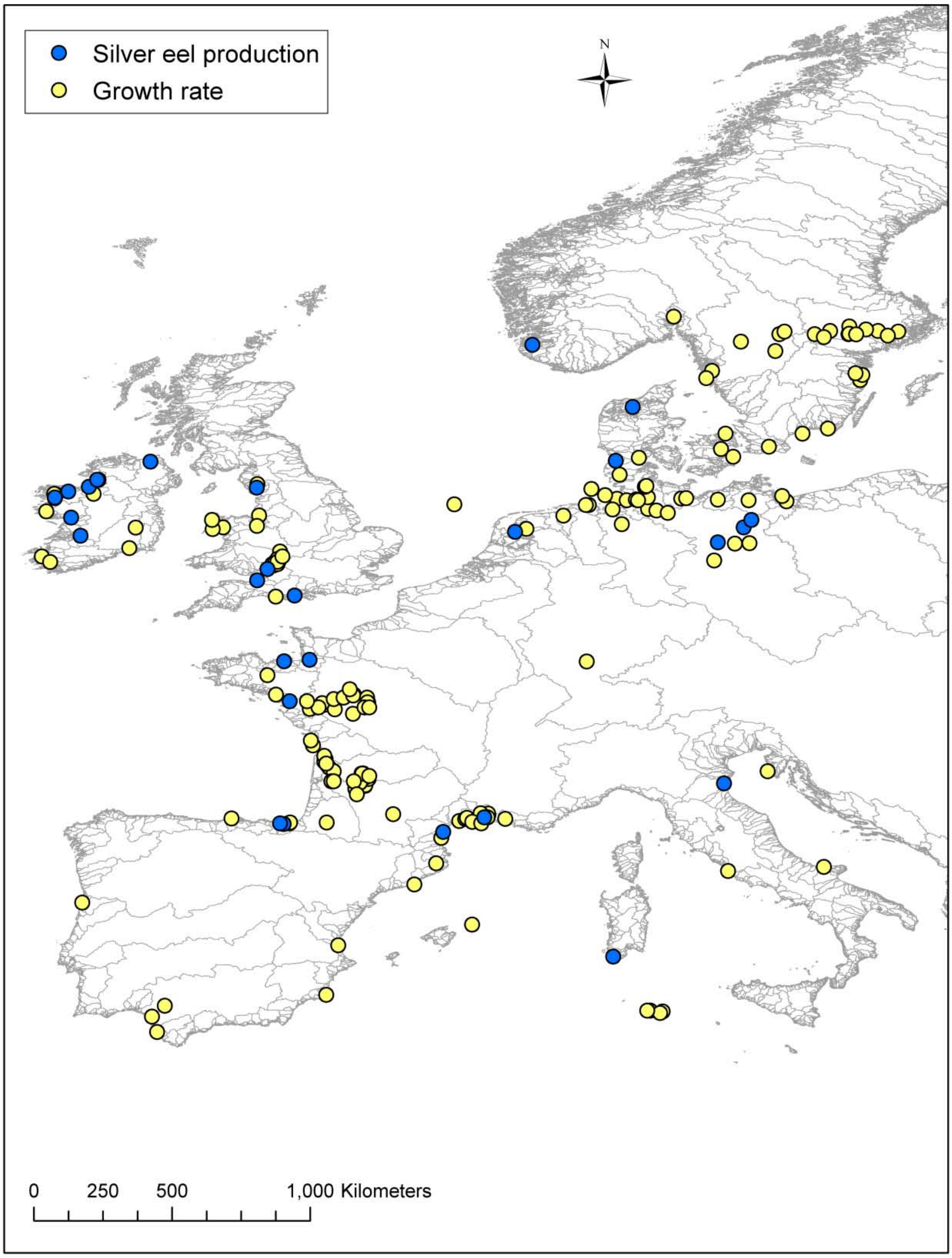


Figure 6.1. Sites where growth rate and silver eel production data were available. The grey outlines indicate sea outlets as defined in the CCM dataset.

Table 6.1. Candidate explanatory variables used to model growth rate of eel across Europe, and hence silver eel production.

Variable	Unit	Source
Latitude	Decimal degrees	
Longitude	Decimal degrees	
Trophic status	1. oligotrophic 2. mesotrophic 3. Meso-eutrophic 4. eutrophic 5. Hypertrophic na - not available	expert judgment / published data
Geology	1. 100% Siliceous 2. 75-99% siliceous 3. >25% Calcareous 4. clay na - not available	expert judgment / published data
Mean Elevation	m	CCM dataset
Mean Slope	%	CCM dataset
Mean Rainfall	mm	CCM dataset
MeanTemperature	Deg C	CCM extract or average 1950-2000 sea surface temp from ICES Oceanography data grid for 'at sea' sites
Max Strahler		CCM dataset
Area	km^2	CCM dataset
Lacustrine area	m^2	CCM dataset
Fluvial habitat	m	CCM dataset
tide	Tidal = '1', else '0'	Daverat et al (in press)
salinity	Saline =' 1,' else '0'	Daverat et al (in press)
Depth	'0' ≤ 1m, '1'>1m	Daverat et al (in press)

Introduction to modelling framework and notation

The aim of this modelling exercise was to develop a robust statistical model to estimate, with appropriate uncertainty, eel production at sites and locations where data are scarce. Foundational to this model are large datasets from a subset of regions around Europe where eels are present. From these data, and using existing knowledge of growth, mortality and recruitment, we developed a Bayesian model to allow production estimates to be derived using geographic and environmental data which are readily available for most locations in Europe where specific biological data might be absent.

Production (P) in Kg year^{-1} hectare^{-1} can be inferred in a variety of ways, but from a population biology perspective, it can be described mechanistically by the equation:

$$P(t) = \frac{N(0)S(t)M(t)}{A} \quad (1)$$

where,

P(t) is the yearly production in kg/ha at time t;
N(0) is the number of immature glass eels recruited to the population at time t = 0;
S(t) is the proportion surviving (survivorship) to time t;
M(t) is the mass in kg of eels at time t;, and
A is the area in ha of the catchment under review.

We drop the explicit reference to t and take logs so that

$$\log P = \log S + \log N + \log M - \log A \quad (2)$$

This formulation is the underlying model assumed for all our subsequently described statistical models which aim to take existing data and models of N, S and M as they relate to environmental variables in order to estimate production where such relationships are not known directly. This formulation comes with some inherent assumptions. Firstly, it is a static model of a population process and assumes that an entire cohort of new recruits mature and contribute to production at the same time *t*. Similarly, recruitment from years around the defined *t* = 0 are assumed to not contribute to production at time *t*. Given enough sequential years of data it would be possible to fit a dynamic population model to the data which would mechanistically describe the processes of recruitment survival and maturation. However, even a static model (as we are constrained to fit) will add considerable value to our understanding of this system.

In essence, the full model we construct involves modelling N, S and M separately using a combination of mechanistic algebraic models and data-driven Bayesian models where the mathematical relationships are unknown. These models are described in detail in Annex D, but here we provide a short summary of how these were estimated and included in the full model.

Recruitment is a major unknown variable in this model. Rather than having direct estimates of the number of recruited eels (hence N(0) is unknown), we have a dimensionless index of recruitment (R), a relationship we approximate via $N \approx aR^b$. The estimates of the coefficients a and b represent the unknown and estimated transformations linking recruitment index and recruitment number. Similarly, since Mass and survivorship are observed with uncertainty, we include parameters for multiplicative and additive biases in estimating their relationship with production.

Survival S is obtained as a deterministic function h of the growth rate of eels, as well as estimates of length at maturity, and various sex-related parameters which we denote as Ω. We thus write $S = h(G; L; \Omega)$ where G is the growth rate and L is the length at maturity. In our model, the values of L and Ω are fixed, being obtained from Bevacqua et al. (2010). The growth rate, however, is estimated (as discussed above) with predictions based on catchment covariates Z.

A directed acyclic graph (DAG) for the growth rate modelling (combined with that of the survival modelling outlined in Section 4 of Annex D) is shown in Figure 6.2. A DAG for the production modelling stage is shown in Figure 6.3.

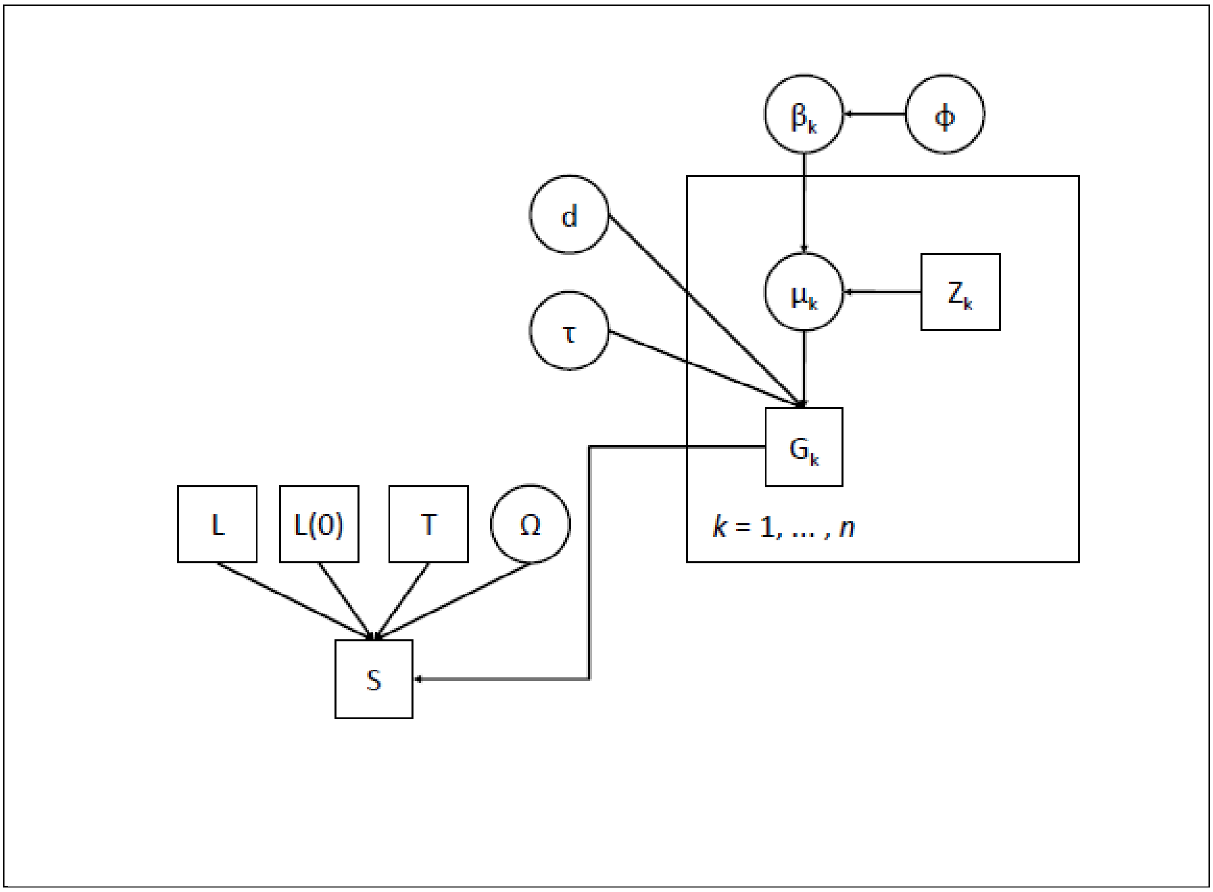


Figure 6.2. A directed acyclic graph (DAG) of the growth rate and mortality modelling process employed in developing the Bayesian eel model.

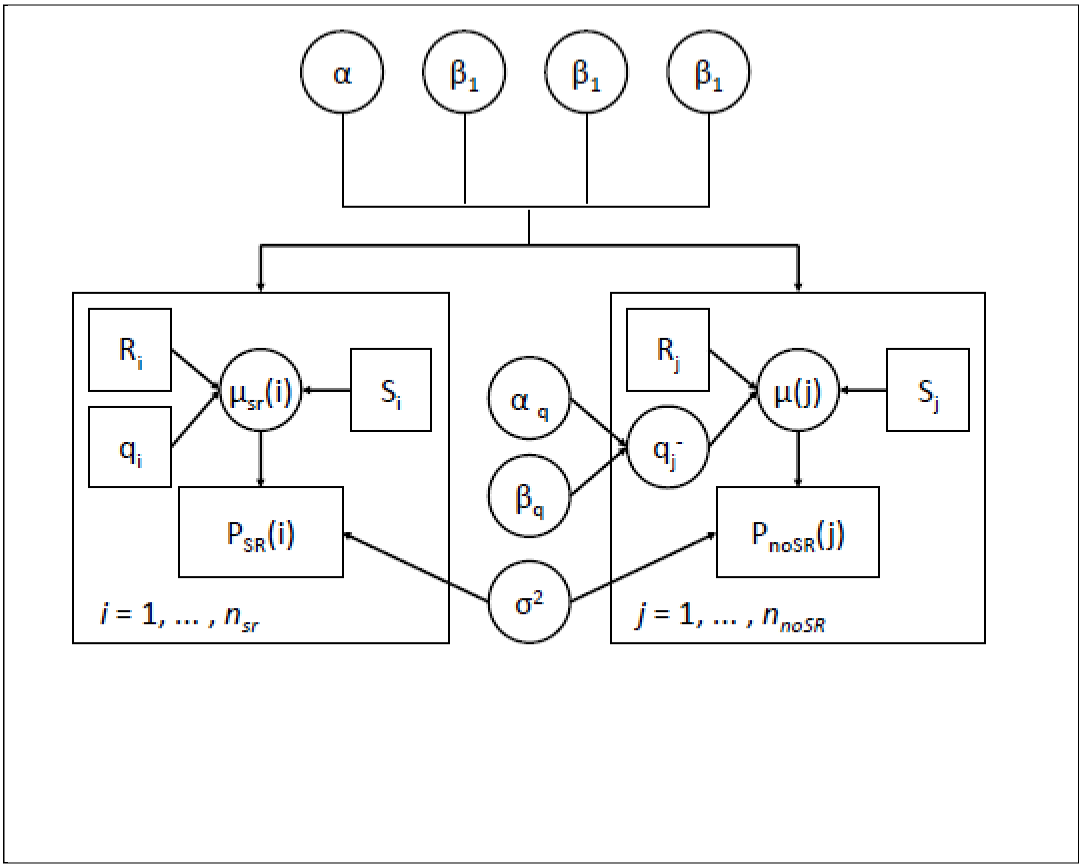


Figure 6.3. A directed acyclic graph (DAG) for the production modelling stage employed in developing the Bayesian eel model.

6.3. Results

The values of the coefficients for the production model are shown in Figure 6.4. None of the explanatory variables appear to exhibit a particularly strong relationship with production. However, the recruitment index seems to be broadly positive, and female survivorship appears broadly negative. The last finding may seem counter-intuitive (an increase in female survivorship decreases production), however recall that female survivorship is strongly correlated with mass at maturity and area of catchment. The value here is being driven by all three of these variables.

The sex ratio appears highly uncertain in its relationship with production. It has been left in the present model because it was found to increase the predictive variability in line with the data.

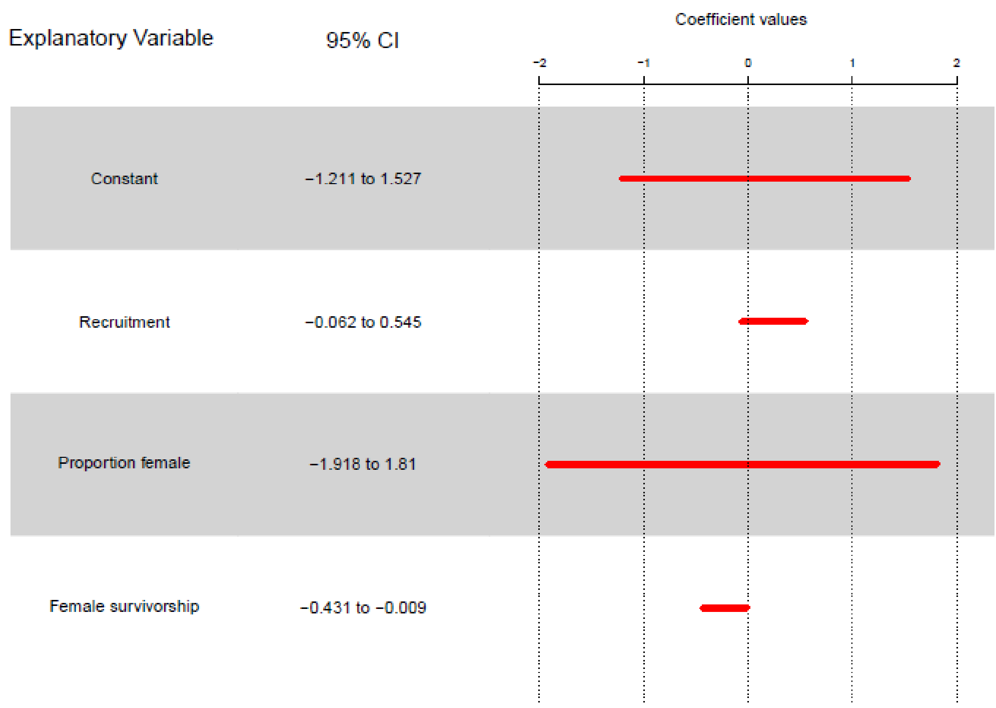


Figure 6.4. A graph of the final included explanatory variables for the production model, with 95% credible intervals given numerically and pictorially

Residual and fitted values plots are shown in Figure 6.5. With such a small data set, it is hard to comment on general performance of the model but it seems adequate. The fitted values plots show that the model has matched the mean relationships present in the data.

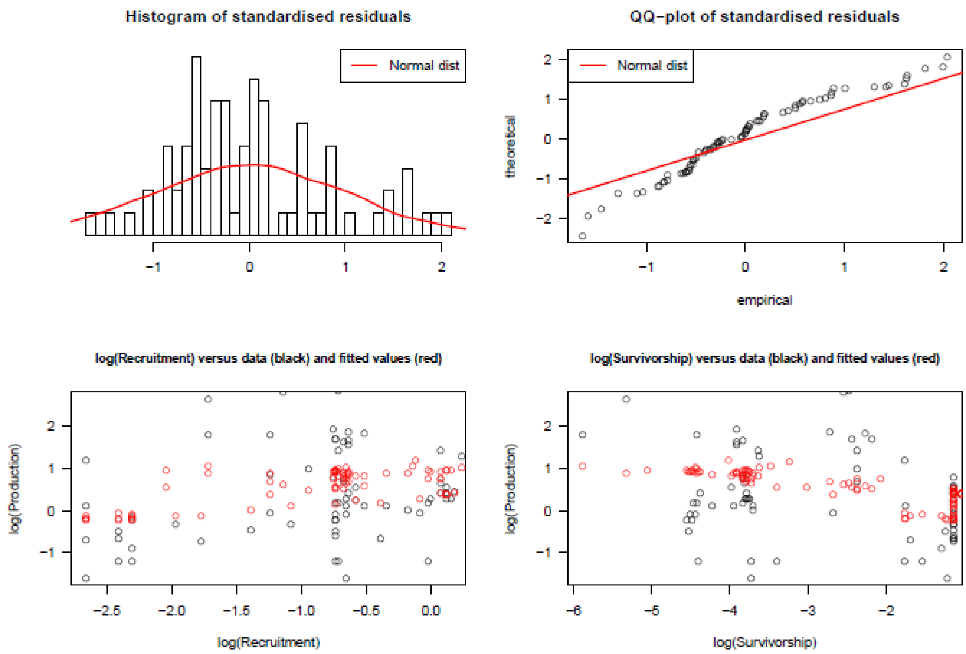


Figure 6.5. A four panel graph showing model performance for the production model. The top left panel shows a histogram of the standardised residuals together with a theoretical normal distribution. The top right panel shows the QQ-plot of the standardised residuals. The bottom left panel shows the fitted values (red) and data values (black) against log recruitment. The bottom right panel shows the fitted values (red) and the data values (black) against log survivorship.

6.4. Testing of predictive ability of eel production model

Eleven test sites were chosen at random from across Europe (Figure 6.6), and the relevant data inputs were collated for each site (Table 6.2). None of these sites were utilised for the model building, and as far as we know, have minimal or no eel data available. These sites therefore represent no-eel-data 'on land' catchments or 'at sea' sites. The data for each site were entered into the excel prediction sheet supplied by A. Jackson and A. Parnell (Figure 6.7), and the outputs recorded. The model was run using recruitment indices relevant to the 1980 escapement year and 2009 escapement year. The model generally predicted the growth rate well, and estimates from the test catchments and at sea sites showed a realistic range of growth rates, in line with empirical measurements for that geographical area (Daverat et al., in press) (Figure 6.8, top).

However, as discussed in the model description (Annex D), the estimates of silver eel production for each escapement year (2009 and 1980) were roughly similar across all geographical areas, although the high 95% prediction interval displayed some variation with site (Figure 6.8., bottom). The 1980 predictions were higher than those predicted for 2009, which shows that the model is sensitive to recruitment patterns to some extent. The recruitment index data were all higher for 1980 than for 2009, consistent with the decline in recruitment, although it should be noted that the recruitment data are relative indices and not quantitative values of absolute recruitment. In 1980, the mean estimated production of silver eel across the test catchments was 1.1 kg/ha, while the 2009 value was 0.75 kg/ha.

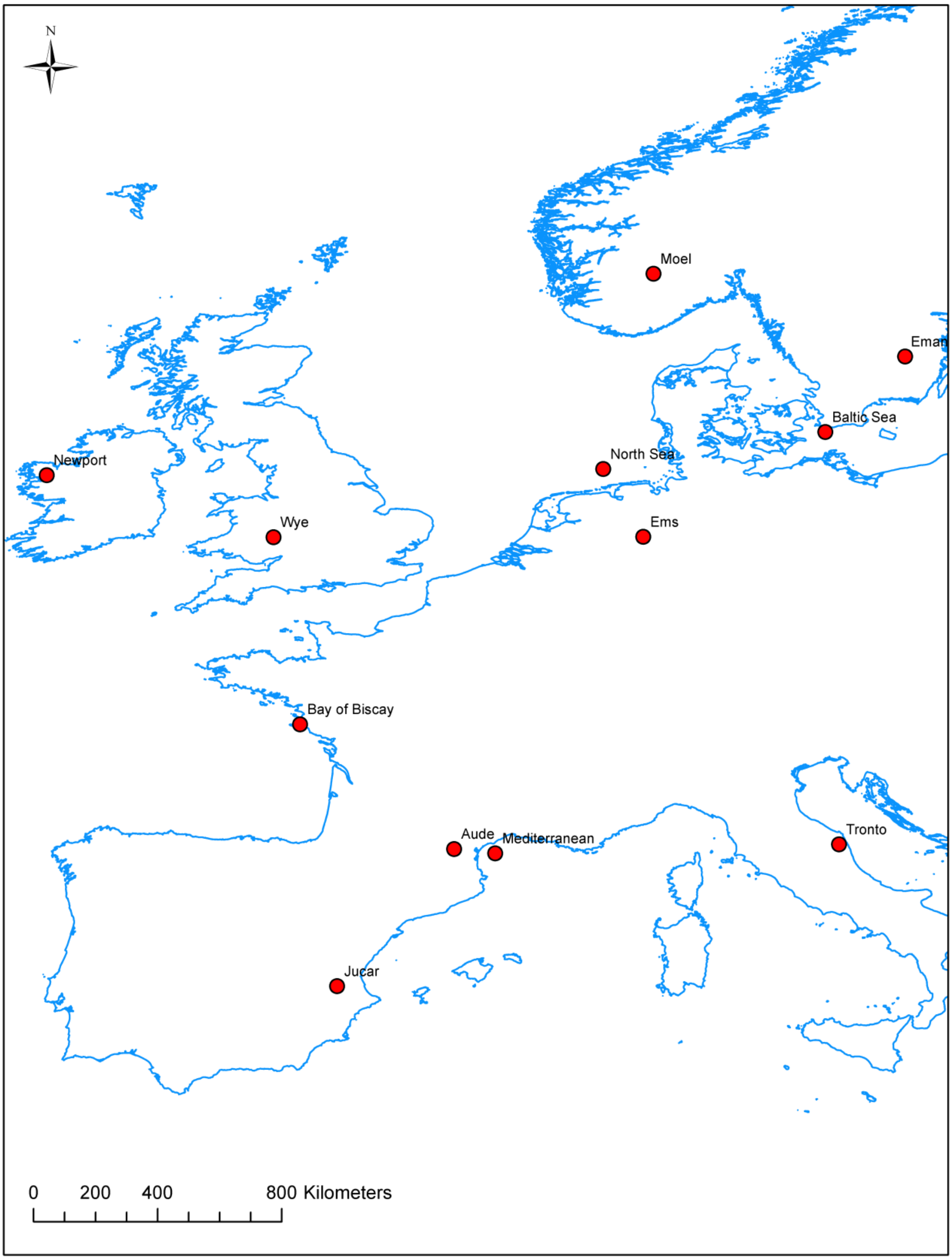
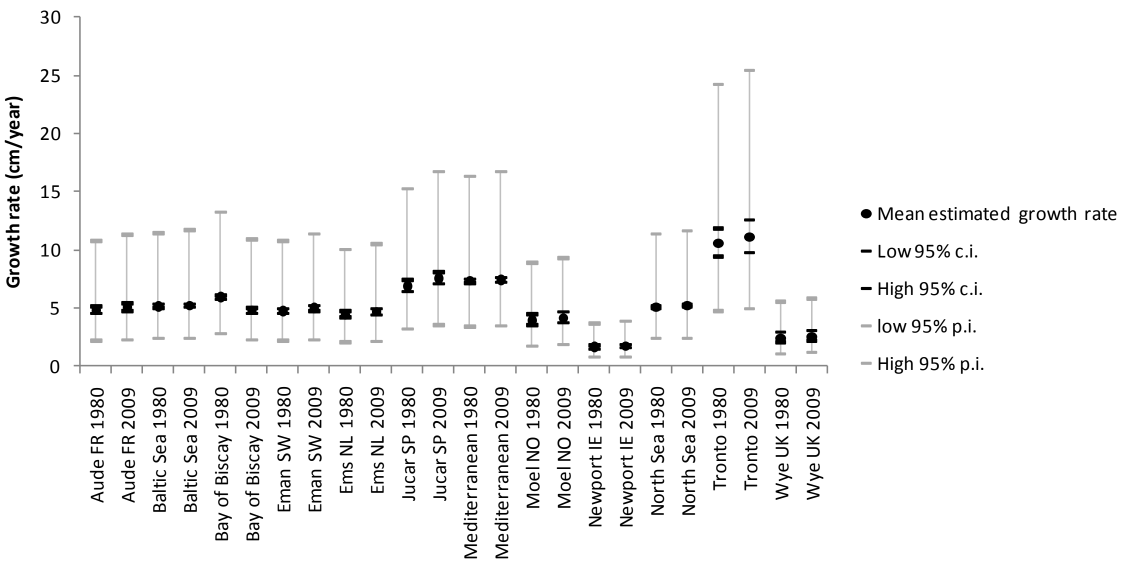


Figure 6.6. Map of Europe, showing location of data-poor sites where eel growth and silver eel production were estimated, using the Bayesian model.

C	D	E	F	G
ON LAND SITES				
Please input:			**Outputs:**	
Growth rate details			Mean estimated growth rate	2.53
Longitude	-2.87		Low 95% mean CI	2.20
Latitude	52.152		High 95% mean CI	3.06
Depth	0			
Tidal	0		Low 95% PI	1.16
Saline	0		High 95% PI	5.83
Rain	1081			
Slope	9.7154		Mean female survivorship	0.02
Eutrophic	0		Low 95% CI	0.02
Mesotrophic	1		High 95% CI	0.03
Catchment area	4144.9			
>25% Calcareous	1		Mean production (kg/ha)	1.17
Max Strahler order	5		Low 95% mean CI	0.53
Post 1980	1		High 95% mean CI	2.59
Survivorship details			Low 95% PI	0.10
Water temp (deg C)	8.7159		High 95% PI	12.91
Production				
Sex ratio	0.5			
Recruitment Index	1.21			

Figure 6.7. Screen shot of the prediction spreadsheet developed for WP5, allowing user input of explanatory variables.



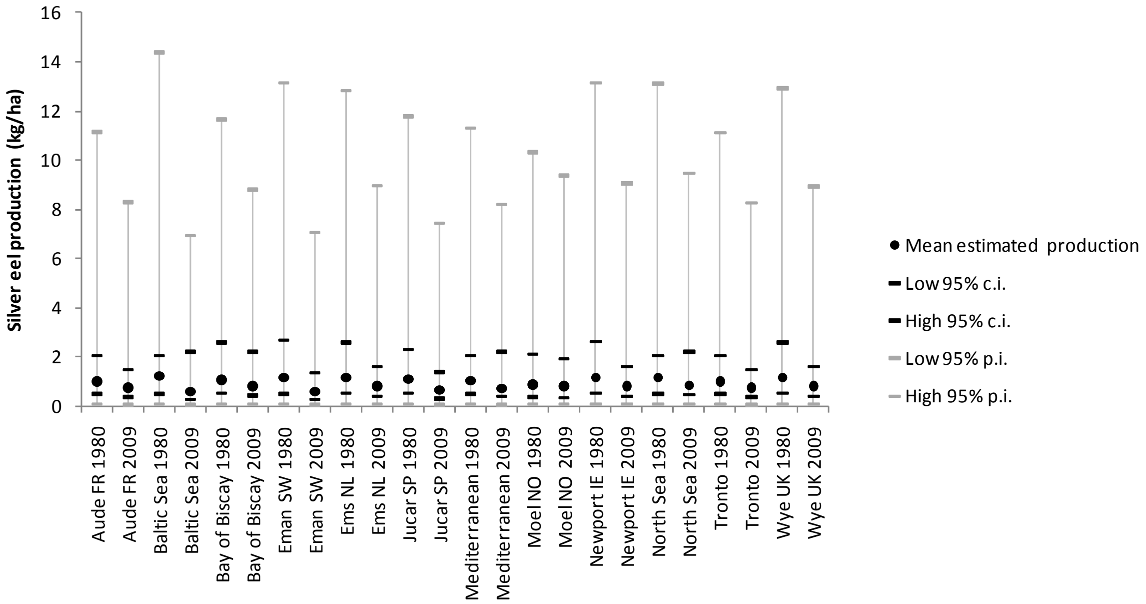


Figure 6.8. Estimated growth rate (top) and silver eel production (bottom) for 11 test sites across Europe. Estimates were calculated using the predictive Bayesian model developed for WP5, and were calculated for a silver eel escapement year of 1980 and 2009.

Table 6.2. Input values and results for 11 test sites across Europe.

Name	Newport	Wye	Moel	Ems	Bay of Biscay	Eman	Aude	North Sea	Baltic Sea	Mediterranean	Jucar	Tronto
Country	IE	UK	NO	DE/ NL	Sea	SW	FR	Sea	Sea	Sea	SP	IT
Latitude	53.95	52.15	59.79	52.16	46.73	57.39	43.12	54.13	55.20	42.99	39.14	43.25
Longitude	-9.44	-2.87	8.14	7.85	-2.10	15.43	2.36	6.69	13.12	3.55	-1.02	13.52
Depth	1	0	0	1	1	1	0	1	1	1	1	0
Tidal	0	0	0	0	1	0	0	1	1	1	0	0
water Temp	8.73	8.72	1.85	8.93	14.16	6.14	11.08	10.46	9.49	17.72	13.21	11.58
Sex ratio	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Recruitment index	0.28	0.27	0.53	0.27	0.26	0.09	0.18	0.32	0.07	0.16	0.10	0.18
Saline	0	0	0	0	1	0	0	1	1	1	0	0
Rain	1272	1081	995	813	na	602	852	na	na	na	466	856
Slope	13	10	17	3	na	1	20	na	na	na	10	25
Eutrophic	0	0	0	1	na	0	0	na	na	na	1	0
Mesotrophic	0	1	1	0	na	1	1	na	na	na	0	1
Catchment area	148	4145	11172	12185	na	4427	5226	na	na	na	21555	1342
>25% calcareous	1	1	0	1	na	0	1	na	na	na	1	1
max Strahler	2	5	6	5	na	5	5	na	na	na	6	5

Table 6.2 cont.. Results for 11 test sites across Europe. Results were calculated for an escapement year of 1980 and 2009, and appropriate recruitment indices for each geographical position applied (see text for details).

Name	Newport	Wye	Moel	Ems	Bay of Biscay	Eman	Aude	North Sea	Baltic Sea	Mediterranean	Jucar	Tronto
1980												
Mean estimated growth rate	1.66	2.41	3.93	4.46	5.94	4.76	4.80	5.12	5.16	7.35	6.91	10.54
Low 95% mean CI	1.51	2.08	3.55	4.21	5.80	4.55	4.51	5.00	5.02	7.16	6.38	9.41
High 95% mean CI	1.80	2.95	4.50	4.74	6.09	4.95	5.16	5.24	5.31	7.54	7.45	11.85
Low 95% PI	0.76	1.09	1.77	2.06	2.78	2.20	2.19	2.40	2.38	3.40	3.23	4.75
High 95% PI	3.68	5.56	8.89	10.00	13.26	10.75	10.77	11.35	11.45	16.30	15.29	24.24
Mean production (kg/ha)	1.19	1.17	0.88	1.17	1.08	1.16	1.01	1.19	1.25	1.04	1.09	1.01
Low 95% mean CI	0.54	0.54	0.38	0.53	0.52	0.50	0.50	0.51	0.51	0.50	0.51	0.50
High 95% mean CI	2.63	2.59	2.10	2.59	2.59	2.70	2.07	2.07	2.07	2.07	2.29	2.07
Low 95% PI	0.10	0.10	0.08	0.10	0.10	0.10	0.09	0.10	0.10	0.09	0.10	0.09
High 95% PI	13.13	12.92	10.32	12.83	11.65	13.16	11.14	13.12	14.37	11.29	11.78	11.13
2009												
Mean estimated growth rate	1.74	2.53	4.12	4.68	4.80	5.00	5.04	5.19	5.23	7.44	7.56	11.07

Low 95% mean CI	1.60	2.20	3.73	4.44	4.59	4.75	4.73	5.09	5.13	7.27	7.05	9.80
High 95% mean CI	1.88	3.06	4.67	4.96	5.04	5.21	5.42	5.30	5.34	7.63	8.11	12.60
Low 95% PI	0.80	1.16	1.88	2.17	2.23	2.31	2.31	2.42	2.41	3.46	3.55	4.97
High 95% PI	3.87	5.83	9.32	10.48	10.89	11.36	11.33	11.63	11.72	16.71	16.70	25.41
Mean production (kg/ha)	0.83	0.82	0.82	0.82	0.82	0.61	0.75	0.86	0.59	0.73	0.65	0.75
Low 95% mean CI	0.42	0.42	0.36	0.41	0.44	0.27	0.37	0.46	0.30	0.39	0.31	0.37
High 95% mean CI	1.63	1.62	1.95	1.62	2.22	1.35	1.49	2.22	2.22	2.22	1.38	1.49
Low 95% PI	0.08	0.07	0.07	0.07	0.07	0.05	0.07	0.08	0.05	0.07	0.06	0.07
High 95% PI	9.06	8.94	9.37	8.94	8.81	7.04	8.29	9.47	6.93	8.21	7.44	8.28

6.5. Discussion

The work above outlines a method for estimating eel production based on explanatory variables via the estimation of growth rate and survivorship. There is a large set of data available for estimating growth rate (Daverat et al., in press); this seems the most robust part of our modelling framework. Conversely, both the survivorship model and the production model suffer from a shortage of adequate data. The models we suggest for survivorship are strongly based on Bevacqua *et al.* (2010), but this work lacks estimates of uncertainty on many of the key parameters. Similarly the production models we have developed are based on very small amounts of data, some of which have key variables missing (e.g. sex ratio). The prediction intervals we thus provide for production are highly uncertain. Worse, because the uncertainty in the survivorship parameters is not included, the uncertainty is possibly an under-estimate. They are, however, our best guess at the production for the site given by users. In particular, 95% credible intervals for the mean production can be obtained with reasonable precision.

A major concern with the production model is that the mechanistic approach for estimating production (Equation 14: Annex D) does not produce any better (in terms of accuracy and precision) predictions than the phenomenological model (Annex D). Given that the mechanistic model is mathematically consistent with the units of production and therefore is unlikely to be incorrect in form, the mis-match must arise elsewhere. This means that at least one (and most likely more than one) of the variables used in the production model are incorrectly specified. The most obvious source is the estimated recruitment index; a more complicated bias adjustment than that used here may be necessary (though again we are hampered by the lack of calibration data at present). Other problems might include biases in the estimation of production for the calibration data set, or that of derived growth and hence survivorship. On a positive note, however, there is some evidence of relationships between recruitment index (I) and production (P), and this suggests there is at least some explanatory power in the model. Without considerably more observed production values, encompassing a wider range of explanatory factors, it is not possible to identify the actual source of the poor model performance.

Lastly, we outline possible recommendations for future modelling of eel production. The most obvious of these is an increase in the amount of production calibration data. Without more data here it remains extremely difficult to estimate the relationship between growth, survivorship and production. The development of a yellow eel density proxy for recruitment would be appropriate to fill gaps in recruitment knowledge. A re-examination of the existing data not available to the analysis should take place. We are clearly missing an important variable in survival/recruitment/area calculation which does not enable us to predict production with any great accuracy. Lack of catchment specific recruitment data and related dynamics such as natural mortality rates is a major hindrance in this exercise and has been alluded to elsewhere in the POSE report.

Additional explanatory information/data, such as the presence of obstacles to recruitment/upstream movement of eel and classifying tidal lagoons as a separate habitat grouping, might improve the predictive power of the model. It may be also possible to utilise fishery derived estimates of yield (rather than production) in order to better understand how recruitment index relates to actual recruitment number. There is considerably more data on yellow and silver eel yield, compared with silver eel production. Such an approach would require a better understanding of how yield is mathematically related to production, but enough comparisons are likely available to make this an achievable goal. This approach would require acquisition and consolidation of the yield data as well as more modelling work to incorporate the new statistical relationships into the existing analysis.

With further data, a number of advanced analysis options become available which may allow for far superior precision in the estimation of production. If data were available on an increased spatial scale, kriging methods (e.g. Banerjee *et al.,* 2004) may become more appropriate (they were attempted for the growth rate modelling stage here, but were found insufficiently superior to the t-regression modelling). With increased data on a temporal scale, state-space approaches (e.g. Pole et al., 1994) would allow for dynamic tracking of production over time, removing noise from individual catchments. Including more process-based approaches, particularly on a temporal scale, would allow some integration with the mathematical models such as CREPE, whereby we could statistically recover the population processes underlying the patterns in eel production within a Bayesian framework such as that underlying BENDM.

Chapter 7. SWOT analysis of models and recommendations for further developments

There are considerable differences in the types of information that each model requires in order to 'run' and the range of results that each of them provides. Although there are some differences in the input requirements, it is fair to say that all models are data intensive. This is either because they try to describe the complex population dynamics of the species or its wide spatial distribution in order to capture area-specific characteristics. Therefore, limitation of available data is expected to be a major obstacle in applying these models in many different situations.

One particular area of limited knowledge is the potential influence of density on life history processes, especially growth and natural mortality rates, sex determinism and dispersal between habitats. The relative importance/influence of density will undoubtedly have changed in some eel habitats over the past 30 years because of the substantial changes in recruitment, and we would anticipate more changes as and when recruitment recovers in the future. Although some of the models examined here include the facility to incorporate some changes in density dependent effects, these are limited by our very poor understanding of the underlying processes and the manner and variety in which density influences occur.

In SWOT analysis, strengths and weaknesses focus on internal causes whereas opportunities and threats stress external factors. The main points under each category are listed for each model below. In our case however, external factors are in common for all the models and are based on the actual context (EMP implementation and post-evaluation) where these models will be used. One opportunity is given by the availability of new data collected during implementation of the EMPs. Several threats are also identified. First, many long time series of data will stop as consequence of implementation of management measures (in particular closure of fisheries). Secondly, the Member States have autonomy to select one model or other approach rather than any other, but there is a risk of political pressure in this choice and application, and in the use of the results. Thirdly, there are so many 'unique' eel conservation and management scenarios in Europe that makes it difficult and probably inadvisable to pursue the development of a single methodology. Hence our approach to build a box of assessment tools, so the user can select the tool most appropriate to their assessment requirements.

Here we present an analysis of the potential strengths, weaknesses opportunities and threats of each of the models tested or developed in this project, and then a SWOT analysis of the overall POSE project followed by recommendations arising from the project.

DemCam

DemCam is a functional model that mimics the main characteristics of eel life cycle, for a single, homogeneous unit of productive area. In order to produce reliable results, the user must know reliable values of at least some of the main parameters for biological processes such as: juvenile recruitment, fishing harvest and effort, or spawner production.

The entire code was developed using Matlab 7.10. At present, therefore, the model can only be applied by someone who has expertise in Matlab programming. However, the development of a user-friendly interface is anticipated soon.

Strengths

- Provides a high level of detail in the simulation of eel dynamics
- All biological process functions are calibrated
- A stage-, age-, and length-structured model

Weaknesses

- Requires sufficient data to calibrate the biological processes
- Natural mortality is taken from the literature (not calibrated)
- The sex ratio at differentiation is fixed (when data are lacking)
- Only operates on single compartment so it is difficult to deal with variations in stock across a catchment.

Opportunities

- Supports bootstrapping and sensitivity analysis

Threats

- Limited data about recruitment
- Limited data to calibrate model
- Annual indicators of recruitment are not available
- Needs special programming skills to run

EDA

The EDA model offer approaches to estimate eel biomass at various spatial scales, and is the only model tested in this project that can be used to develop predictions for an entire EMU. The main strength of EDA is that the method is widely applicable to the whole European rivers since it uses European river network databases and open-source software. This approach is also based on actual data of yellow eel abundance; data which are classically collected during surveys like those conducted for the European Water Framework Directive (WFD). However, EDA requires a considerable amount of data on historical (pseudo-pristine of before management actions) and updated data (present or after management actions. The main weaknesses are the use of a poorly-known silvering rate and the absence of uncertainty evaluation, and that the framework does not take into account the wetted area lakes and lagoons, or the potential differences in eel stock characteristics between rivers and these other environments.

EDA users have to know R and SQL language but it is hoped that a user-friendly interface will be developed in the near future.

Strengths

- Spatial information readily available since it uses European river network databases and open-source software
- Can be used to make assessments for an entire EMU
- Model is calibrated using only data for yellow eel collected during surveys of river habitats

Weaknesses

- Requires a considerable amount of historical (pseudo-pristine) and recent data (present or after management actions)
- Users have to know R and SQL language
- Requires knowledge of yellow-to-silver conversion rates (which are, generally, poorly known)
- Absence of framework for uncertainty evaluation.
- The CCM dataset does not include the minor streams, so the wetted area producing eel within an EMU may be underestimated.
- The geospatial data does not at present include lakes or lagoons. This development is anticipated, but quantitative data on stock characteristics from lakes/lagoons/large rivers are not easy to obtain.

Opportunities

- Simulation of production on a European scale (contrary to river basin-specific calculations)
- Open-source software facilitates extension of the model

Threats

- Limited data about recruitment
- Limited data from past and present surveys

GEM

The model is realized in EXCEL to make it easy for users to adapt the model to the data, especially if deeper knowledge of EXCEL exists. The model is partly described within the EXCEL file, but, it is necessary to prepare a detailed description of the model before it can be submitted to other user without common work during the adaptation process. Until now three versions exist with partly different input data and model results. The version for the river Elbe system estimates only female eel and takes into account the catch of angler and the effect of increasing cormorant population. In addition the recruitment is divided into immigrating elvers and number of stocked individuals. Age ranged from 0 to 20 years. The version for the Corrib catchment did not incorporate catch of angler and the effect of cormorant because the factors could be neglected. However, variable proportions of male and female elvers were incorporated and age ranged from 0 to 35 years. The version of CREPE data used an age range from 0 to 20 years and incorporated the catch of fishermen by glass, yellow and silver eel. The proportion of female elvers was estimates based on survey data.

The descriptions illustrate that the model is flexible and that adaptation of the model to the situation in other catchments than the Elbe is possible. The quality of the output of the model increases if field data are available which can be used to optimize the model parameter in such a way that the difference between the field data and the model estimates get a minimum. Indeed, discussions between the modeller and the data provider will continue within the final months of POSE to improve the model optimizations.

Strengths

- The model takes into account the mortality caused by fishing, angling, cormorants, and hydropower plants, in addition to natural mortality
- Estimates the number of emigrating eel.

- Requires a low number of input variables
- Model assumptions can easily be substituted by estimates if they are available
- The parameterisation and the use of the model is easy due to the realization in EXCEL.

Weaknesses

- Requires a time series of input data of at least 15 years to avoid strong effects of the initial population which must be estimated.
- Only operates on single compartment so difficult to deal with variations in stock across a catchment.

Opportunities

- Facilitates stochastic calculations since it incorporates stochastic noise in the input data and use bootstrap methods to calculate confidence intervals of the model outputs.

Threats

- Limited data about recruitment
- Only short input time series is available
- Poor knowledge of natural mortality values

SMEP II

SMEP was originally developed as a generalist model for UK eel populations. The main strength of SMEP is that it is a general eel population model that works with limited data to simulate a variety of effects. It incorporates carrying capacity and density dependence, and its design is flexible to allow for different parameterisations of the model so, it could be applicable to any population. It also gives outputs for all life stages, thus providing for varying proxy reference points. SMEP II is a complex, detailed model which enables the user to increase the level of realism in simulating eel dynamics. However, that also increase the range of data it requires to run so, it could rely on assumptions about eel population dynamics for which there is limited scientific evidence.

SMEP II runs on any Windows-based pc, using MS-DOS and input/output files are formatted as universal .csv files. As such, no programming expertise is necessary to run SMEP II – however, it is not advisable to attempt to apply SMEP II without a good understanding of population modelling and of the biology of the eel and the characteristics of the eel production.

Strengths

- High level of detail in the description of eel dynamics
- All the main processes in eel dynamics (i.e. mortality, growth, sexual differentiation, maturation, etc) are explicitly modelled
- Time steps can be modified to better capture eel dynamics
- It does not require age-based information that might be difficult to obtain
- Incorporates different compartments within the river basin and copes with differences in stock characteristics between these compartments

Weaknesses

- The increase in realism means that this is a complex model
- Requires a considerable amount of input data

- User needs to have a good understanding of the theoretical framework behind the model to be able to parameterise it correctly.
- There is presently no statistical framework to provide estimates of uncertainty

Opportunities

- Appropriate for sensitivity analysis since the model parameter values can be easily changed and all key processes can be simulated.
- Reduces the need for age-specific information

Threats

- Limited data about recruitment
- Survey series cannot be standardised or are not available
- Poor knowledge of sources of mortality other than natural mortality
- Limited information about mortality at stage
- Can only be applied to a connected river basin, so is impractical for application to an EMU – solution is to apply to an Index River Basin and then extrapolate to other basins across the EMU, if appropriate.

BENDM (Bayesian Eel No Data Model)

The BENDM model attempts to establish a method for estimating eel production based on explanatory variables via the estimation of growth rate and survivorship. There is a large set of data available for estimating growth rate (Daverat et al., in press); this seems the most robust part of the framework. Conversely, both the survivorship model and the production model suffer from a shortage of adequate data. The survivorship models are strongly based on Bevacqua et al. (2010), but this work lacks estimates of uncertainty on many of the key parameters. Similarly the production models we have developed are based on very small amounts of data, some of which have key variables missing (e.g. sex ratio). The prediction intervals we thus provide for production are highly uncertain. Worse, because the uncertainty in the survivorship parameters is not included, the uncertainty is possibly an under-estimate. They are, however, our best guess at the production for the site given by users. In particular, 95% credible intervals for the mean production can be obtained with reasonable precision.

With further data, a number of advanced analysis options become available which may allow for far superior precision in the estimation of production.

Future development of the model will require new data relevant to the assessment year in question and expertise in spatial and Bayesian statistical modelling.

Strengths

- Provides a successful predictive tool for eel growth rate based on readily available catchment characteristics, with associated uncertainty intervals
- Provides mathematical equations that use published instantaneous mortality rates to model eel survivorship more accurately over time in place of simple percentage annual mortalities
- Provides a robust statistical framework for incorporating growth rate into a Bayesian model to estimate the production of silver eel in catchments where there are no such data

- Identification and collation of catchment characteristics datasets that are usable in areas of no eel data

Weaknesses

- The production part of the model suffers from a severe lack of data.
- The only recruitment data available for input are two dimensionless indices for Baltic and rest of Europe
- Owing to the low quality of production and recruitment data, the production part of the model was almost impossible to fit, and the results were unreliable and had high uncertainty
- Future development of the model requires specific statistical expertise

Opportunities

- Incorporate fisheries yield data into the production part of the model as a proxy for silver eel estimates
- Incorporate any additional production data that may become available in the short term
- Incorporate any recruitment quantification that may become available in the short term
- The growth rate models could be further improved with existing data by additional statistical analyses
- The mortality component could be extended to include uncertainties in parameter values

Threats

- No new data (production and recruitment) become available in the short term (2-3 years)
- The statistical resources (expertise, time and funding) which was used to develop this model may not be available in the future

SWOT Analysis of the POSE project

Strengths

- POSE has assessed, developed and tested a suite of models suitable to stock assessment under different data and local conditions
- POSE has developed CREPE for benchmark testing models under different scenarios (biological, management)
- Weaknesses and sensitivities in the various models have been identified
- POSE has established a possible framework for extrapolation from data rich to data poor scenarios, although current data weaknesses don't support the successful implementation of this. Growth
- POSE has developed a database structure for eel (DBEEL) which may be adopted at the national and/or international level for the management and exchange of eel data.

Weaknesses

- All models are complex, requiring high inputs of data or parameters
- Some models require time series data which are often lacking
- Some models operate at a single spatial unit making it difficult to apply to larger more complex catchments, for example where there are combinations of lakes and rivers

- Models require knowledge of life history process parameters (such as natural mortality rate, silvering rate, size at maturation). These are currently poorly understood and individual models may be quite sensitive to subtle changes in these parameters.
- Gaps in methodology exist, particularly for assessing stocking in large waterbodies such as lakes and coastal waters.
- Some models require absolute recruitment data which are lacking or absent at the catchment level.
- All models require historical data which are crucial to determining pristine biomass (B_0)
- CREPE has been built on a specific set of parameter assumptions for a southern European eel stock, which resulted in difficulties for some models developed on northern European stocks to adapt their parameters. Other CREPE parameter sets will support wider model testing.
- Uploading data to the DBEEL requires management and quality control

Opportunities

- POSE, subject to improving some of the identified weaknesses, may provide a standardised, benchmarked, suite of methodologies which can feed into the reporting requirements for the EU Regulation and facilitate assessments of the international stock.
- Expected improvements in the DCF for eel should fulfil many of the data requirements for the local modelling
- International co-ordination of data exchange and reporting, already in place for other species, will support both local assessments and standardised reporting
- Inclusion of non-fisheries related data collection under an agreed programme of surveys will avoid data gaps and improve the ability to undertake good quality stock assessments
- DBEEL provides a cost-effective and practical solution to eel data management and exchange

Threats

- International co-ordination is lacking and benchmarking and quality control of the local stock assessments becomes difficult or impossible
- Changes to the DCF for eel fail to close the gaps in data collection because data collection is set at inappropriate spatial and temporal scales and does not embrace both fisheries dependent and independent monitoring and surveys.
- Reduction/closures in fisheries effort and catch jeopardises data collection and the integrity of time series, so alternative data sources are required
- Recruitment data (local level) and time series (international stock assessment) remain absent or disappear
- No management of DBEEL is put in place for future data collection and exchanges.

Conclusions

In conclusion, POSE has provided a standardised, benchmarked suite of assessment methodologies that can feed into the reporting requirements for the EU Regulation and facilitate assessments of the international stock. Our work identified gaps and sensitivities in the models and their approach to stock assessments. Understanding these sensitivities means we are better informed about the modelling processes, and managers who employ the models as part of the EMP reporting process avoid unnecessary pitfalls.

A critical lesson learned during the project has been that it takes a lot of time to evolve the application of a model to a dataset to produce a confident result, and that is providing that the appropriate tuning data are available. Without these tuning data, the model application can be achieved quicker, but the results must be treated with considerable caution! Modelling fisheries data can be time consuming and this extensive time needs to be built into the whole process of stock assessment, particularly with respect to management and reporting of eel under the Regulation. Likewise, we caution against the blind use of model outputs without ground truthing.

We identified several crucial data requirements in each of the models, and these data are rare in the real world. The anticipated developments in the Data Collection Framework (DCF) sampling requirements for eel should fulfil many of the data requirements for local modelling that were identified during this project. The data collection must be coordinated and conducted at appropriate spatial and temporal scales, and embrace both fisheries and non-fisheries sources of data. The inclusion of non-fisheries related data collection under and agreed programme of surveys will close data gaps and improve our ability to undertake good quality assessments.

POSE developed CREPE which can be used as a framework to provide a series of baseline data sets benchmark test these and other models developed in the future. Countries adapting existing models or developing new models should benchmark test the model against CREPE. This will maximise the opportunity for successful reporting to the EU in future years and reduce the threat of uncoordinated assessment outputs. The original version of CREPE produced for POSE created a data set of virtual eel with characteristics closest to the southern parts of the eel's range, and this caused some difficulties for those models that had been developed for more northerly eel populations. Different scenarios (biological, management) will have to be included in different versions of CREPE in the future.

POSE also developed a database structure for eel (DBEEL) in order to facilitate the collation and dissemination of standard data. This structure could be adopted at the national or international level to support the coordinated assessment and management of *Anguilla anguilla*, and the intercalibrations requiring exchanges of eel data. The requirement for such a database has already been raised by the EIFAAC/ICES WGEEL (years, and add to ref list). However, management of the database is a substantial task also requiring quality control measures.

The international coordination of data exchange and reporting, already in place for other species, will support both local and international assessments and reporting. The DBEEL developed in the POSE project provides a cost-effective and practical solution to eel data management and exchange, but this database needs a home, management and a formal data exchange and quality assurance procedure.

However, international coordination of data collection is lacking and in its absence, benchmarking and quality assurance (planning stage) and control (ongoing) of the local stock assessments is difficult to say the least. It should also be acknowledged that some data sets are coming to an end because of reductions or closure of fisheries or other economic factors. Our model testing in POSE highlights the great importance of historic data and time series which are fundamental to deriving the historic eel production values required by the Eel Regulation. Clearly, time series data collections should be protected, and new time series commenced, especially of recruitment and silver eel escapement.

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