Measuring agricultural and congestion externalities in recreational fisheries: The case of salmon in France

Julien Salanié, Yves Surry, Philippe Le Goffe, . Inra, . Région Bretagne, . Agrocampus-Ouest, . Métropole de Rennes

Measuring agricultural and congestion externalities in recreational fisheries: The case of salmon in France[1]

Julien Salanié [a], Yves Surry [b], Philippe Le Goffe [a],

[a] ENSAR – Département Économie Rurale et Gestion
65, rue de Saint-Brieuc – CS 84215 – 35042 Rennes cedex – France
phone: 02 23 48 54 16 – fax: 02 23 48 54 17

[b] Swedish University of Agricultural Sciences
Department of Economics – Box 7013 – S-750-07 Uppsala -Sweden
phone: 018 67 10 00 – fax: 018 67 35 02
yves.surry@ekon.slu.se

Résumé

En France, la valeur sociale tirée de la pêche de loisir est probablement limitée. De nombreuses externalités environnementales, de stock et de congestion sont présentes. Elles sont dues, pour partie, à des mesures encourageant l'abandon des droits de pêche et la généralisation du libre accès. Dans ce contexte, on étudie les facteurs de variation de la demande de pêche récréative au saumon dans l'Ouest de la France par la méthode des coûts de déplacements. Des indicateurs des pollutions agricoles et de la congestion des parcours de pêche sont introduits dans des modèles à utilité stochastique (RUM). En particulier, des modèles logit multinomiaux, emboîtés et mixtes sont estimés. Les résultats indiquent que la congestion et les captures influencent le choix des sites par les pêcheurs . Dans la perspective de l'application du principe bénéficiaire-payeur et de la restauration des droits de pêche, on évoque, en conclusion, l'utilisation des résultats pour simuler une tarification en fonction de l'effort de pêche.

Mots-clés : droits de pêche, pollution, demande récréative, méthode des coûts de déplacement.

Abstract

In France, social welfare from recreation fishing is probably burdened. Environmental, stock and congestion externalities affect this welfare. They are partly due to institutional measures favoring property rights renunciation and open-access generalization. In this context, we study demand shifting factors in the case of salmon angling in western France, using the travel cost method. Indicators for ecosystem degradation from agriculture and for site congestion are introduced in random utility models (RUM). Multinomial, nested and mixed logit models are estimated. In the perspective of the application of the beneficiary-pays principle and property rights enforcement, we evoke in conclusion the use of these results to simulate resource pricing as a management option to increase angling social welfare.

Keywords: fishing rights, pollution, recreation demand, travel cost method.

[1] The authors gratefully acknolwedge the financial support of the Conseil régional de Bretagne (Programme Breizh, Analyses et méthodes pour un développement agricole et rural durable en Bretagne).

Introduction

In France, recreational fishing in rivers and streams is managed by angling associations. The number of anglers in rivers is consta decreasing for two decades by 3% per year (40 000 anglers). In the same time, rivers ecosystems are put under high pressure by hun activities (e.g. agriculture, waste disposals, urbanization, hydroelectric power generation, etc.).

Despite private fishing rights belonging to landowners, recreational fishing is managed as a public good. Fishing rights are associated v many constraints concerning river quality maintenance. In practice, fishing right owners (mostly farmers) renounce to these rights and ç them for free to anglers associations. This situation results from the actual legislation that favors this renunciation. Fishing rights own have the obligation to belong to an angling association to use their right (so they have to pay the fishing fee anyway). Financial aids to fu river quality maintenance constraints are subject to membership in angling or environmental associations. Next, there is a legal and histor framework for property rights renunciation for free. Therefore most of the fishing rights are given-up for free to angling associations. Tl manage these rights on an egalitarian basis by keeping annual fishing fees at a very low level. They also develop reciprocitarian agreeme to allow anglers to fish on larger territories (at present 60% of France) by just paying a small additional fee (16 € p.a.).

This generalization of open-access is the source of externalities (stock and congestion externalities) because fishing effort is not limi (Anderson, 1983 and 1993). It also results in environmental externalities (e.g. neglected riverbanks, fallow lands development, wa pollution, fish habitat destruction) because riverside landowners, notably farmers, have no incentives to maintain and protect ri environment.

We expect these features to burden the social rent derived from recreational fishing. Anglers' welfare might be lowered and farmers show interest in recreational functions of the environment. The main stakes of resolving these externality problems are:

- to increase anglers' welfare,
- to give incentives to farmers to protect rivers' environment and, in the same time, to raise revenues in a period of declining pul support to agriculture,
- to insure sustainable tourism and regional development,
- and to identify and calibrate the management tools to improve social welfare.

Our work aims at assessing and measuring inefficiencies affecting recreational fishing in France. We study the variation factors recreational fishing demand using the travel cost method. We look at how recreational trip decisions are affected by site attributes ir random utility framework applied to salmon angling in western France. From that, we derive damage functions for environmental a congestion externalities.

There are a few studies on the linkages between agriculture and recreation demand. Examples of applications are found on the values agricultural landscape to hikers (Fleisher and Tsur, 2000), croplands distribution to pheasant hunters (Hansen *et al.*, 1999) or water flo (Fadali and Shaw, 1998) and water pollution (Patrick *et al.*, 1991) to recreational fishermen. These studies suggest that recreational valu are not negligible in comparison to costs of pollution reduction or benefits from amenity production. They confirm the idea that the externalities could be provided on the beneficiary-pays principle basis through market implementation.

Moreover, few studies focus on congestion problems. Among them, recent papers from Boxall *et al.* (2003) and Schuhmann and Schwa (2004) gave evidence for congestion negative effects on recreationists welfare.

1. The random utility framework

The travel cost method can be used to measure externalities affecting salmon anglers. We built a random utility model that deals both w agricultural and congestion externalities.

The use of the random utility model (RUM) is very popular among recreational economists. It is a model of an individual's choice of a site a recreation trip. Each site is assumed to give the person some utility that is assumed to be a function of trip costs and site characterist (aesthetics, congestion, pollution, etc.).

The utility for site *i* is assumed to be linear as shown in equation (1).

(1) $$v_i = \beta_{tc} TC_i + \beta_q Q_i + \varepsilon_i$$

where TC_i is the trip cost to site *i*, Q_i a vector of site characteristics, ε_i a random error term (unknown to the researcher) and β_{tc}, β_q are t parameters to be estimate.

The individual is assumed to choose the site that maximizes his utility. Site k is then chosen if $\beta_{tc}TC_k + \beta_q Q_k + \varepsilon_k > \beta_{tc}TC_i + \beta_q Q_i + \varepsilon_i$ for all i belonging to the individual's choice set (S).

Assuming that the error terms ε_i are i.i.d. Weibull leads to the commonly used multinomial logit model (MNL), with the probability of choosing site k given by expression (2).

(2)
$$P(k) = \frac{exp(\beta_{tc}TC_k + \beta_q Q_k)}{\sum_{i=1}^{S} exp(\beta_{tc}TC_i + \beta_q Q_i)}$$

(3)

The MNL model is subject to the independence of irrelevant alternatives (IIA). This property of the MNL model implies that the probability ratio between two alternatives is independent from any variation of the probabilities in another choice alternative. This restriction is usually not verified which results in a mispecified model. The IIA restriction can be released using variants of the MNL such as the nested multinomial logit (NMNL) or the mixed multinomial model (MMNL).

Nesting is the most widely used solution to release the IIA restriction. Nested models are estimated in several papers on recreational fishing demand (Kaoru, 1995 ; Kling and Thomson, 1996 ; Parson and Hauber, 1998). In this case, alternatives are grouped in categories (i.e. nests) of similar alternatives. Many nesting structures are found in the literature, however, the most common are grouping by geographical proximity, by type of water bodies (e.g. lake, river, ocean), by fishing mode (e.g. from boat or from the shore) or by targeted species.

In the nested model, the probability of choosing an alternative k in the choice set S is conditioned by the probability of choosing its corresponding nest j. It is equal to $P(k) = P(k|j) \quad P(j)$

These probabilities can be written as follows :

$$P(k|j) = \frac{exp((\beta_{tc}TC_i + \beta_q Q_i)/\rho_j)}{\sum_{i=1}^{J} exp((\beta_{tc}TC_i + \beta_q Q_i)/\rho_j)}$$

$$P(j) = \frac{exp(\alpha_j + \rho_j I_j)}{\sum_{i=1}^{J} exp(\alpha_j + \rho_j I_j)}$$

Where I_j is the inclusive value associated to nest j and can be written : $I_j = \sum_{i=1}^{J} exp((\beta_{tc}TC_i + \beta_q Q_i)/\rho_j)$

ρ_j parameters (dissimilarity coefficients) measure the degree of dissimilarity between each alternatives within a nest. They are theoretically comprised between 0 and 1 (Herriges and Kling, 1997). ρ_j parameters greater than 1 indicate a misspecificied nesting structure with alternatives belonging to different nests being better substitutes than alternatives within the nests.

Mixed models constitute another flexible alternative characterized by random parameters. Recent papers like Train (1998), Chen and Cosslett (1998) and McConnell and Tsen (1999) provide applications of these models to recreational fishing. While releasing the IIA restriction, mixed models allow to capture heterogeneous angling behaviors.

Mixed models are a generalization of the tradition multinomial logit model. In this model, parameters β_i are supposed random and distributed following a density function $f(\beta|\theta)$, where θ is a vector of parameters describing the density function like its mean and variance. Train (2003) shows that the probability of choosing an alternative k is equal to :

$$P(k) = \int L_k(\beta_{tc}, \beta_q) f(\beta_{tc}, \beta_q|\theta) d\beta$$

This integral is a weighted average of the L_k probabilities from the logit model. Weights are the probabilities of randomly sampling each combination of β_j. To estimate such a model we proceed using random draws $\beta_i(t)$ for each parameter for each individual making assumptions over the density function of each parameter. Usually, there are assumed to follow a normal distribution. However, ar distribution can be used. Train (1998) used a log-normal distribution for the β_{tc} parameter to insure that it is negative for all individual in i sample.

In this paper, we apply these three types of RUM models[2], namely MNL, nested-MNL and mixed-MNL to a sample of salmon anglers in tl Western part of France.

2. Agriculture and salmon stocks

Atlantic salmon is found in many French rivers. Because of threatened stocks, some big rivers (Loire and Garonne) are closed professional and recreational fishing. While some rivers are found near the Pyreneans (southern France), most salmon rivers are located western France (Brittany and Lower Normandy). Over the last decade, 85%[3] of French salmon recreational catches were made in this are

Many salmon stocks in France are threatened[4]. In its overview of salmon repartition in French rivers over the last century, Thibault (199 shown that many salmon stocks collapsed (e.g. Loire river, Allier river, Garonne river). Dams are mostly responsible for this situati preventing salmons migration to reproduction areas. Other species (e.g. sea lamprey and sea trout) face the same situation. However, dar cannot be considered as the only reason for the decline in salmon recreational catches. This decline is observed on rivers not cut hydroelectric power generation dams.

Salmonids (salmon, trout and sea trout) are very sensitive to ecosystems degradation. In particular, they require pristine running waters w high dissolved oxygen levels and low silt contents. Degradations from anthropogenic influence are mostly due to urbanization a agricultural and industrial activities. Although urbanization and industrial activities have significant impacts on salmonids stocks, agriculture recognized to have a high responsibility in river ecosystems degradations. Hendry *et al.* (2003) provide an extensive review of the literatu on salmonids habitat requirements and the solutions to improve stocks. They identify three criteria influencing salmonids stocks :

- (i) water quality
- (ii) water quantity
- (iii) physical habitat.

Water quality can be affected by both point and non-point source pollutions. Point source pollution from farming arise from local spillov (e.g. livestock slurry and silage liquor). Flows of pollutants, among which pyrethroid, can equally be a problem. Occasionally, they c eradicate invertebrate populations on which juvenile salmonids feed. Non-point source pollutants effects are considered as a more import threat to salmonid stocks. Run-offs from agricultural lands, leading to nutrient enrichment and/or eutrophication dramatically alter ecosyst quality for salmonids requirements. In particular, they can lead to excessive macroalgae growth and subsequent oxygen depletion a salmon mortality at all biological stages. Silt arising from bank erosion, due to agricultural practices and drainage is acknowledged to be c of the main source of salmon stocks depletion (MAFF and NAW, 2000 + see infra). Hendry *et al.* (2003) argue that the European Comm Agricultural Policy (CAP) is responsible for increased siltation of rivers and streams. This is a result from subsidies maintaining high livest levels on unsuited lands resulting in overgrazing. Subsidies to highly erosive crops (e.g. corn) are also at stake.

Aside from quality issues, **water quantity** issues are also of importance for salmon ecology. Modified water flows affect salmonids migrati availability and distribution of the habitats used by juveniles salmonids and streams morphology, directly related to habitats. Flow regin can be modified by numerous elements such as hydroelectric power generation, flow derivations for aquaculture and agricultural and fore production (drainage or irrigation).

Finally, **physical habitats** issues are critical. Hendry *et al.* (ibid.) consider that habitat destruction or modification is responsible for mos the decline in salmonids stocks. The authors expressively link fish habitat modifications to farming practices on riparian lands. Overgrazi high livestock densities, lack of shepherding (or fencing), cropping near rivers, high erosive crops, lack of soils covering during winter drainage are equally damageable to fish habitat. They result in bank collapse, river siltation and sediments loads affecting spawning rearing habitats for salmonids. On that feature, Hendry *et al.* (ibid) state :

> *" ... The CAP combined with the current lack of any effective land-use control legislation to specifically police and control these detrimental effects of agriculture and forestry, is arguably the biggest single threat to the survival of many salmonid populations throughout the British Isles..."*

---

[2] Classical continuous and count-data pooled travel cost models were also estimated.

[3] 2000 salmons per year on average.

[4] The WWF (2001) find out in a survey that the Atlantic salmon is threatened on most rivers of the world.

3. Linking agriculture to anglers welfare

In our model, anglers may be affected by agriculture in two ways : through recreational catches and landscape aesthetics.

Landscape aesthetics are a composite notion that each angler may appreciate differently. It is hard to define due to the complexity of what is a "scenic" view. Tempesta and Thiene (2004)[5] and Rambonilaza (2004) provide the most up to date empirical literature on valuing landscapes. In the French case, Colson and Stenger-Letheux (1996) is probably the main effort conducted in agricultural landscape valuation. It is not the purpose of this work to contribute to this area of environmental and agricultural policy evaluation. We invite the interested reader to refer to the abundant literature or at least to the two cited references. However, it is noted that grove (or bocage) constitutes a main indicator of landscape attractiveness in western France (Colson and Stenger-Letheux, 1996). It is also commonly admitted that permanent grasslands are a good indicator for grove (Le Goffe, 2000). We retain this indicator as a general indicator fro agricultural landscape. It is calculated as the average percentage of permanent grasslands on adjacent districts to the rivers taken from the 2000 Agricultural Census. At the level of our study (28 salmon rivers) finding a satisfying indicator is burdensome. Data on riverbanks management are lacking on most rivers. Some rivers have been subject to riverbanks vegetation and agricultural practices accounting in the context of SDAGE[6] (Schéma départmental d'aménagement de gestion des eaux) or river contracts[7]. In the near future, the availability of these data for rivers[8] will allow to improve our work in this area.

Catches are always included as a quality or success variable in recreational fishing demand valuation studies. Average catch rates at a fishing site have been extensively used. McConnell *et al.* (1995) introduced catch rates[9] individual estimates from a Poisson model. This approach has the advantage to introduce heterogeneity across anglers behaviours relatively to their ability to catch fish. Catches rates are usually estimated from creel survey data collected apart from trips data. Explicative variables are usually the age or experience of the angler, fishing modes (gear, bait, etc.), fishing effort (hours spend fishing at the site) and some proxy for fish stocks or environmental quality (Kaoru, 1995).

However, these models do not explicitly link fish stocks to ecosystem degradations. Fish stocks estimates per river are rarely available at every considered fishing site. Anyway, they would already be deflated from ecosystem degradations if evaluated from fish sampling. In France, salmon stocks are subject to total allowable catch (TACs) regulations per river. TACs are calculated from the potential of a river to produce salmon in relation to the available surface of spawning areas expressed in riffles/rapids surface equivalents. This potential has been calculated relatively to the maximum carrying capacity of these habitats (Prévost and Porcher, 1996). Surface areas in riffles/rapids have been calculated for each salmon river of our study area (Porcher and Prévost, 1996) and converted into smolts[10] and adults production equivalent to a river without degradation.

Moreover, the National Fishing Council attempted to inventory the causes of ecosystems degradations at the national level. Data are gathered in the ROM[11] (Ecosystems Observatory Network). The ROM compiles data (Appendix 2), on a 5 points scale ranging from "pristine" to "very damageable", to quantify the effects of several anthropogenic activities on a reference specie (wild trout). Because salmonids are usually treated as a whole in the biological literature we will consider that Atlantic salmon is affected in the same way as trout by anthropogenic pressures measured in the ROM.

Therefore, we built a simple model to explain variations in salmon annual average catch[12] at a site accounting for potential stock, degradations, enhancements and total fishing effort. The model was specified as a linear function of dependent and explanatory variables (except stocks introduced in logarithm) estimated by ordinary least squares. Impacts of aquaculture, industries, canalization and urbanization were also introduced but were not significant. Estimation results are shown in Table 1. Some predicted values could be negative (but very close to zero) for some rivers. We then decided to bind them to non-negativity using a Tobit model. All the estimated parameters are of the expected sign. Most of the parameters are not significant at the 10% level but very close.

[5] This conference.
[6] River catchments management schemes.
[7] Contracts built on a voluntary basis.
[8] The application of the European Water Directive Framework forces to realize such an accounting.
[9] Number of fish caught per trips.
[10] i.e. juvenile.
[11] Réseau d'Observation des Milieux.
[12] Calculated over the last 3 years.

Table 1. Stock function estimation results

Variable	OLS	Tobit
Constant	-102.913	-102.913
	(-1.259)	(-1.448)
Log Stock (salmons)	10.653	10.653
	(1.331)	(1.571)
Agricultural degradation index	-6.146	-6.146
	(-1.438)	(-1.153)
Total effort (visits)	0.0135*	0.0135*
	(3.666)	(6.901)
Sée river (dummy)	241.776*	241.776*
	(3.245)	(5.713)
Stocking (dummy)	18.053	18.053
	(1.581)	(1.486)
σ		17.786*
		(7.483)
Log-likelihood at convergence	-120.326	-120.326
Pseudo-R^2	96.67	96.94

** : Significative at the 10% level or higher.*

Catch is increasing with fishing effort and stock. Stocking also has a positive impact on catch. A dummy was introduced to cont Sée river that is, by far, the river on which most salmons are caught in France (20% of national catches). Finally, the agriculture d index has the expected negative impact on catches. Predicted values from these models are introduced into the random utility moc

4. Data

Empirical data were obtained through a telephone survey from 828 salmon anglers who fished during the year 2002 on 28 salm Brittany and Lower Normandy (Appendix 1). Complete trip data were collected about angling destinations over these 28 rivers. information about anglers' socio-demographic characteristics and angling motivations (catching fish, resting, etc.), preferences (scenery and catch at rivers) and perception were also collected. Summary statistics on the sample are presented in Table attributes statistics in Table 3.

Table 2. Anglers summary statistics (n=828)

Variables	Mean	Std-dev.	Min.	Max.
Number of trips	42.4	42.9	1	220
Number of rivers visited	1.6	0.9	1	6
Trip distance to site (km)	55.7	112.7	0.1	1137.3
Wage rate (€/hour)	10.7	8.7	0	100.7
Age (years)	51.2	15.3	12	91
Fishing experience (years)	17.2	15.3	1	70

Table 3. Fishing sites attributes summary statistics (n=28)

Variables	Mean	Std-dev.	Min.	Max.
Total catch	52.5	114.8	0	600
Catch rate (catch/trip)	0.021	0.015	0.000	0.08(
Fishing trips	3345	4687	96	2298:
Average perceived congestion note (1-5 scale[a])	2.73	0.66	1.00	3.97
Actual congestion (trips/[km.m^3])	0.92	0.60	0.13	2.48
Permanent grasslands (%TAL)[b]	13.97	12.01	2.25	64.4!
Ecosystem degradation from agriculture (1-3 index)	1.86	0.71	1	3

[a] *Obtained from the survey (individual notation).* [b] *%TAL = percentage in total arable land.*

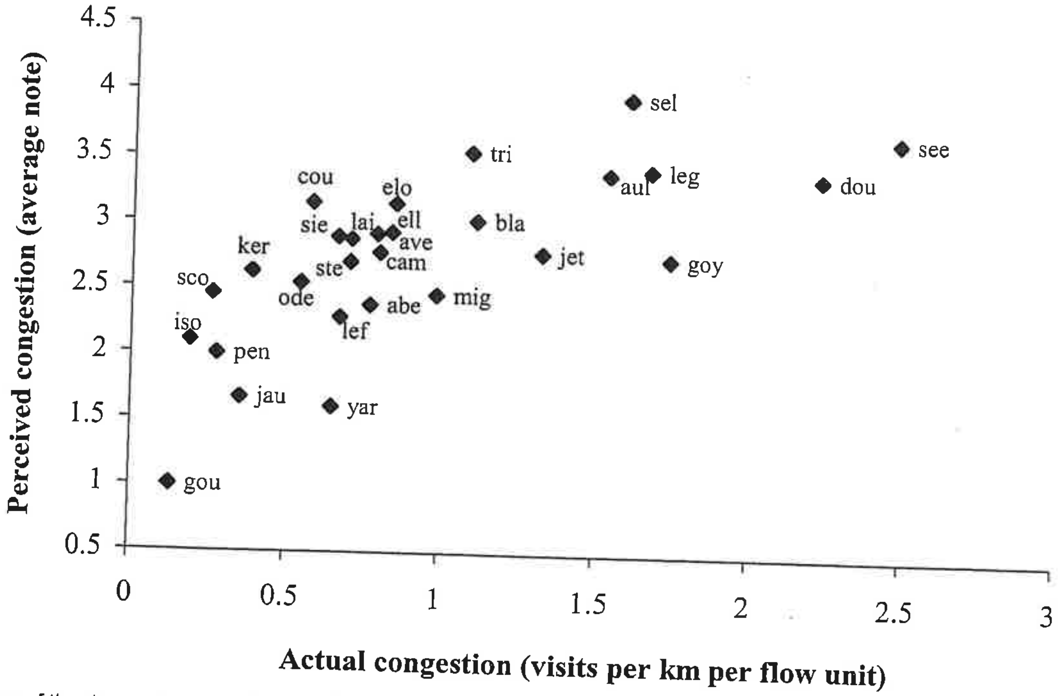
Including congestion indicators into recreation demand models leads to some difficulties. Most past studies used actual conges density) rather than anticipated or perceived congestion. Shelby (1991) and Jakus and Shaw (1997) emphasize that recreation likely to be sensitive to the perception they have, or even more the anticipation they make, of actual congestion. To our knowledge, the o study looking at the inclusion of different types of congestion measures in a RUM travel cost framework is Schuhmann and Schwabe (200 In this contribution, we are investigating two types of (objective or subjective) congestion measures:

- actual congestion = number of anglers per kilometer of reach and per unit of flow (Table 3),
- average perceived congestion = average congestion note (on a five point scale) given per individuals who visited a site in the surv (Table 3)

These two different measures are positively correlated following a logarithmic relationship (Figure 1.). This implies that perceived congesti increases more rapidly than actual congestion for low level of users density. After a point, it increases more slowly, indicating that havi more users influences less the level of congestion has perceived by anglers[13].

Trip cost calculations were designed to account for flexibility in work hours. We used measures similar to those used by Feather and Sha (1999). Our trip cost specifications include round trip cost from home to site (computed from a GIS) and opportunity cost of time as a fractic of the wage rate or the full wage rate times a dummy for individuals with flexible work hours (i.e. executive professions). Round trip cos were adjusted for car horsepower differences following the reference used for tax deduction in revenues declarations[14]

Figure 1. Relationship between the two congestion measures



Note : The names of the rivers are abbreviated to the 3 first letters.

5. Estimations results

Choice set specification is an issue in estimating random utility models. Several authors, (Parsons and Hauber,1998 ; Haab and Hicks, 1997) shown that welfare measures are very sensitive to choice sets specifications. The issue is to answer the question : "Are all the sites I'm considering for my study relevant to users when taking trip decisions ?". Several approach to deal with choice sets are found in the literature. However, the three most used approach in choice set designing are :

-1. Endogeneous choice sets (Haab and Hicks, 1997) : choice sets definitions vary across individuals on an endogeneous basis

-2. Familiarity based choice sets (Hicks and Strand, 2000 ; Parsons *et al.*, 2000) : Only sites being familiar to individuals are considered

-3. Distance based choice sets (Parsons and Hauber, 1998) : Only sites being at a certain distance, chosen by the modeler, from the individual's home are considered.

[13] We estimated an ordered logit on the answers. It performed correctly and can be used to introduce individual specific measures of perceived congestion in the models.
[14] Bulletin Officiel des Impôts (5-F-1-03), n°10 du 17 janvier 2003.

Endogeneous and familiarity based choice sets definitions require data about anglers preferences and habits. Because we don't have information, we opted for distance based limitations on choice sets. We observed on our data that more that 95% of the total fishing ef a site is made by anglers leaving closer than a 2 hours trip (Appendix 3). We used this choice set definition to estimate our models method has the advantage to exclude most of the multiple days trips. Failing to account specifically for such trips leads to biased w estimations (Parsons, 2003).

We specified two different utility functions : one including our perceived congestion measure, the other using the actual congestion mea

Hausman tests were performed to assess the validity of the MNL models. These tests indicate that our MNL specifications violate t property and cannot be used for welfare calculations. We also tested for several nesting structures. Based on the dissimilarity coeffi (Herriges and Kling, 1997) we chose to retain a two level geographical nesting structure (Figure 2). Other nesting structures failed to p dissimilarity coefficients in the unit interval for both measures of congestion. Finally, the mixed models were estimated using 100 based on Halton replications (Train, 2003). Results for models including the perceived congestion variable are shown in table 4, thc actual congestion variable in Table 5.

Figure 2. Nesting structure of the models

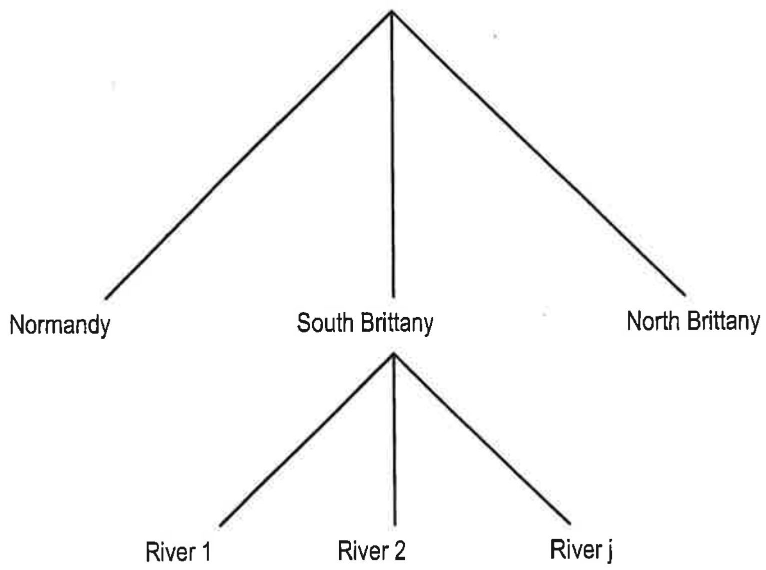


Table 4. Estimation results for "average perceived congestion" models

Variable	MNL	Nested MNL	Mixed MNL	
			Mean	Standard error
Trip cost	-0.090***	-0.121***	-0.158***	0.059***
	(-152.85)	(-81.55)	(-30.92)	(20.11)
Total Catch	0.004***	0.009***	0.027***	0.018***
	(66.19)	(19.27)	(19.87)	(16.95)
Congestion	2.643***	7.976***	-1.537**	0.079***
	(20.01)	(30.15)	(-2.12)	(2.62)
Congestion squared	-0.187***	-1.406***	0.420***	0.079***
	(-7.91)	(-26.57)	(2.84)	(2.62)
Grasslands	0.002**	0.070***	-0.478***	0.250***
	(1.97)	(13.09)	(-24.39)	(20.55)
Fishing days	0.003***	0.024***	0.054***	0.382
	(11.07)	(30.26)	(28.95)	(0.97)
Dissimilarity coefficients				
Normandy	-	0.104*	-	
		(1.70)		
South Brittany	-	0.489***	-	
		(28.76)		
North Brittany	-	0.654***	-	
		(32.79)		
McFadden's ρ2	0.475	0.906	0.967	

***, **, * : parameters significance at the 1%, 5% and 10% respectively. T-statistics are in parenthesis.

Table 5. Estimation results for "objective congestion" models

Variable	MNL	NMNL	Mixed MNL	
			Mean	Standard error
Trip cost	-0.089***	-0.107***	-0.156***	0.057***
	(-154.79)	(-86.59)	(-30.11)	(16.65)
Total Catch	0.008***	0.003***	0.026***	0.018***
	(58.24)	(6.52)	(18.83)	(10.58)
Congestion	4.570***	4.150***	6.968***	0.973***
	(79.78)	(31.84)	(18.07)	(6.07)
Congestion squared	-1.672***	-1.250***	-2.592***	1.070***
	(-63.44)	(-25.16)	(-14.19)	(15.05)
Grasslands	0.006***	0.054***	-0.031***	0.348***
	(5.20)	(10.27)	(-17.87)	(17.66)
Fishing days	0.006***	0.030***	0.056***	0.004
	(19.55)	(34.72)	(25.38)	(1.58)
Dissimilarity coefficients				
Normandy	-	0.518***	-	
		(3.86)		
South Brittany	-	0.498***	-	
		(25.13)		
North Brittany	-	0.819***	-	
		(36.74)		
McFadden's ρ2	0.478	0.902	0.969	

***, **, * : parameters significance at the 1%, 5% and 10% respectively. T-statistics are in parenthesis.

All variables are significant and have the expected signs. The trip cost coefficients are negative and have the highest t-ratios. As expected, anglers preferably choose sites with highest overall catches. We believe that anglers evaluate their own success chances in regard of this variable while it is not their personal "real" catch rate. The estimated coefficients on congestion and congestion squared indicate tl congestion affects welfare following an inverted U-shape function. We discuss this later. The probability to chose a site depends on number of open fishing days, indicating that site closures have effects on site choice.

The permanent grasslands variable is quite problematic. While we see in the MNL and NMNL results that it affects positively site choice, results from the mixed logit are indicated a much more complex reality. In both models, the distribution of the permanent grasslar parameters is spread (high standard-errors). This indicates that some anglers put value on permanent grasslands, by choosing site with highest level of this variable, while others not. Some even get disutility from grasslands. We believe that this situation is observed becau the grasslands variable is measured with error not at a pertinent level (i.e. not on riverbanks). We cannot make any conclusion on t variable while we believe that more accurate observations of permanent grasslands near the rivers would have shown the expected result

6. Welfare effects of agricultural scenarios

We then use our estimated models to analyze several agricultural scenarios. These scenarios are four-folds :

- **scenario 1 :** all sites improve to the best quality level (pristine environment). The agricultural degradation index (ADI) becomes 0 foi sites.
- **scenario 2 :** all sites improve to the good quality level (ADI=1 for all sites).
- **scenario 3 :** degradation incurs at all sites (ADI=3 for all sites).
- **scenario 4 :** all sites become highly polluted (ADI=5 for all sites).

Welfare computations were made for the two NMNL specifications. For the NMNL, the surplus variation ΔS associated to a change fron to Q^*_i in sites characteristics is then equivalent to :

$$\Delta S = \frac{ln\sum_{i=1}^{J} exp(\alpha_j + \rho_j l_j^*) \quad ln\sum_{i=1}^{J} exp(\alpha_j + \rho_j l_j)}{\beta_{tc}}$$

The welfare effects of these scenarios are calculated using total catches variations following our catch model (Table 1). Separate effects all rivers are shown in appendix 4. The results of these scenarios are calculated for the nested logit model using perceived conges measure as an independent variables. Per trip welfare calculations are shown in Table 6. Aggregate welfare values were calculatec multiplying the per trip estimates by the total number of salmon fishing trips estimated for the studied area[15] (Salanié, 2004).

Table 6. Surplus variations from the 4 scenarios.

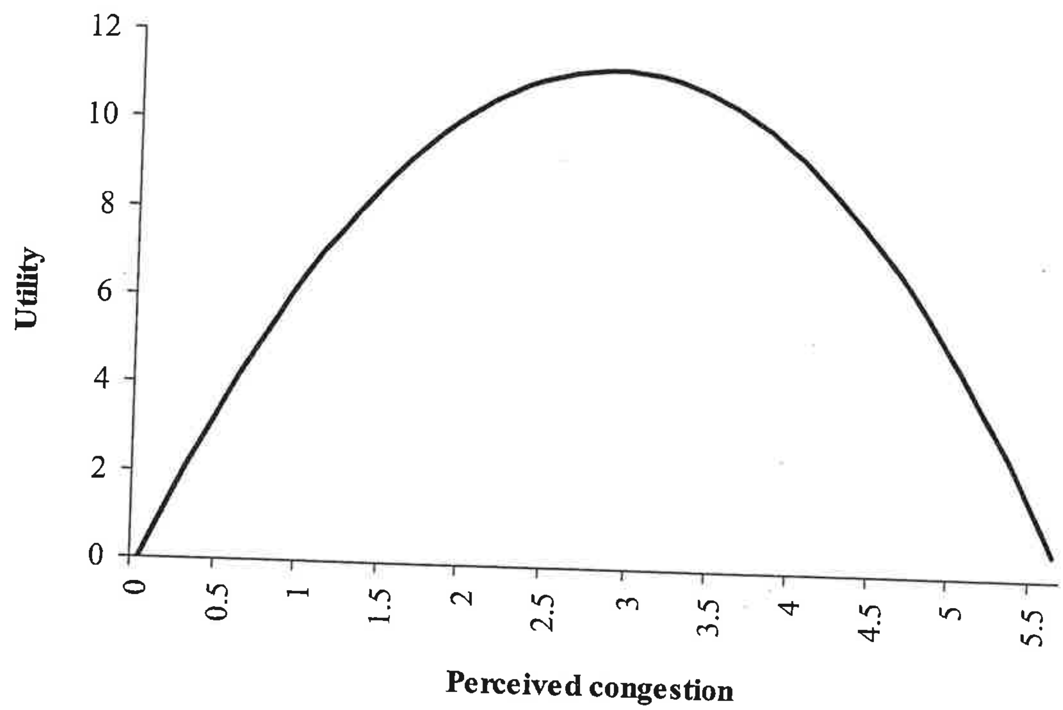
	Perceived congestion		Objective congestion	
	Per trip	Aggregate	Per trip	Aggregate
Scenario 1 (pristine)	1.56 €	146 049 €	0.32 €	30 082 €
Scenario 2 (improved)	0.41 €	38 272 €	0.16 €	15 275 €
Scenario 3 (decrease)	- 1.89 €	- 176 995 €	-0.13 €	-12 554 €
Scenario 4 (worst)	- 4.32 €	- 404 688 €	-0.43 €	-39 806 €

7. Optimal congestion

The estimated coefficients from the NMNL model using perceived congestion as an explanatory variable are used to plot utility ag congestion (Figure 3). Figure 3 clearly shows that there is an optimal level of congestion for which utility is maximum. This level is rea for an average perceived congestion of 2.84. 13 rivers are found above that value. This kind of inverted U-shape relationship has alr been demonstrated in recreational activities valuation studies (Schuhmann and Schwabe, 2004). In the first part of the curve, people utility from social interactions they have with other anglers. After the maximum, their welfare is lowered because the site becc overcrowded.

[15] 93 671 fishing trips in Brittany and Lower Normandy

Figure 3. Relationship between utility and congestion



8. Discussion and conclusion

This paper proposes an economic analysis of externalities affecting recreational anglers. Externalities arise from the public nature of goods considered and also from institutional failures. Recreational fishing becomes an open access resource while property rights exist but are attenuated. We propose a travel cost framework to evaluate these externalities, taking a close look at agricultural sources of environmental degradation and externalities of congestion.

Our work suggests that anglers exhibit positive willingness to pay for limiting fishing effort (at least on some rivers) and improving the environment. Measures of congestion include actual and perceived congestion. As suggested by Jakus and Shaw (1997), the "correct" congestion measure might be anticipated congestion. Agricultural pollution measured by an index for ecosystem degradation from agriculture is significant and affects negatively anglers welfare. The proxy for agricultural landscape patterns (measured by permanent grasslands in total arable land) tends to generate mixed results. We believe that this result is more due to the choice of the variable rather than a real situation. Agricultural pollution enters anglers utility through a catch function. It is used to model several scenarios. The aggregate values are not important at the region level. An small increase in agricultural ecosystem degradation would lead to an aggregated welfare loss of 180 000 €. However, these measures do not account for the trout anglers population which is more important.

The models shown in the paper do not account for variations in number of trips. It would be interesting to look at total trips variations following agricultural degradations increase or decrease using repeated NMNL (Morey *et al.*, 1991) or linked MNL (Hausman *et al.*, 1995).

Finally, Hendry *et al.* (2003) and Bilsby *et al.* (1998) argue that most of the degradations made to salmonids populations could be overcome using environmental buffers near riverbanks. At this level, transaction costs might be low. Regulating access to the resource *via* entrance fees should be a way to proceed applying the beneficiary-pays principle. Further work is needed to look at production costs of environmental improvements and the transaction cost associated with policies to provide them. Access pricing could result in limiting congestion and raising funds to remunerate fishing rights and improve the environment. This could help agri-environmental programs to be more self-supporting.

References

Anderson L.G. (1983). The demand curve for recreational fishing with an application to stock enhancement activities. *Land Econ* 59(3):279-286.

Anderson L.G. (1993). Toward a complete theory of the utilization and management of recreational fisheries. *Journal of Environ Economics and Management*, 24:272-295.

Bilsby M.A., Cragg-Hine D. and Hendry K. (1998). *Cost-benefit analysis : its role in recreational fisheries development and manageme* Recreational Fisheries – Social, Economic and Management Aspects (Hickley and Tompkins Eds), Fishing News Books, Lc UK, 310p.

Boxall P., Rollins K. and Englin J. (2003). Heterogeneous preferences for congestion during a wilderness experience. *Resource and E Economics*. 25:177-195.

Chen H. and Cosslett S.R. (1998). Environmental quality preference and benefit estimation in multinomial probit models: a sim approach. *American Journal of Agricultural Economics*, 80:512-520.

Colson F. and Stenger-Letheux A. (1996). Evaluation contingente et paysages agricoles : application au bocage de Loire-Atlantique. C *d'Economie et Sociologie Rurale*, 39-40:12-34.

Fadali E. and Shaw D.W. (1998). Can recreation values for a lake constitute a market for banked agricultural water ? *Contem Economic Policy*, 16:433-441.

Feather P. and Shaw D.W. (1999). Estimating the cost of leisure time for recreation demand. *Journal of Environmental Economic Management*, 38:49-65.

Fleisher A. and Tsur Y. (2000). Measuring the recreational value of agricultural landscape. *European Review of Agricultural Econ* 27(3):385-398.

Hansen L., Feather P. and Shank D. (1999). Valuation of agriculture's multi-site environmental impacts : an application to pheasant h *Agricultural and Resource Economics Review*, 28(2):199-207.

Hausman J.A., Leonard G.K. and McFadden D. (1995). A utlity-consistent, combined discrete choice and count-data model : ass recreational use losses due to natural resource damage. *Journal of Public Economics*, 56:1-30.

Hendry K., Cragg-Hine D., O'Grady M., Sambrook H. and Stephen A. (2004). Management of habitat for rehabilitation and enhancem salmonid stocks. *Fisheries Research*, 62:171-192.

Jakus P. and Shaw D.W. (1997). Congestion at recreation areas: empirical evidence on perceptions, mitigating behavior and manag preferences. *Journal of Environmental Management*, 50:389-401.

Kaoru Y. (1995). Using random utility models to estimate the recreational value of estuarine resources. *Resource and Energy Econ* 17:119-136.

Le Goffe P. (2000). Hedonic pricing of agriculture and forestry externalities. *Environmental and Resource Economics*, 15(4):397-401.

McConnell K.E. and Tseng W.C. (1999). Some preliminary evidence on sampling alternatives with the random parameters logit. *Resource Economics*, 14(4):317-332.

McConnell K.E., Strand I. and Black-Hedges L. (1995). Random utility models of recreational fishing: catching fish using a Poisson pr *Marine Resource Economics*, 10:247-261.

Ministry of Agriculture, Fisheries and Food [MAFF] and National Assembly for Wales [NAW]. (2000). *Salmon and freshwater fisheries r* MAFF and NAW, London, 204p.

Morey E.R., Shaw W.D. and Rowe R.D. (1991). A discrete-choice model of recreational participation, site choice and activity valuation complete trip data is not available. *Journal of Environmental Economics and Management*, 20:181-201.

Parsons G.R. (2003). The travel cost model. In: A primer on non-market valuation (Champ P., Boyle K.J. and Brown T.). Kluwer Academ Publishing, Amsterdam, Holland.

Patrick R., Fletcher J., Lovejoy S., Van Beek W., Holloway G. and Binkley J. (1991). Estimating regional benefits of reducing targete pollutants : an application to agricultural effects on water quality and the value of recreational fishing. *Journal of Environmenta Management*, 33:301-310.

Prévost E. and Porcher J.P. (1996). *Méthodologie d'élaboration des totaux autorisés de captures (TAC) pour le saumon atlantique (Salmo salar L.) dans le massif armoricain.* GRISAM, Document scientifique et technique n°1, 28p.

Porcher J.P. and Prévost E. (1996). *Pêche du saumon dans les cours d'eau du massif armoricain – Fixation du nombre total de captures autorisé (TAC) par bassin.* GRISAM, Document scientifique et technique n°2, 13p.

Salanié J. (2004). *La pêche au saumon dans le massif armoricain : enquête sur la population de pêcheurs et leur effort de pêche.* Document de travail ENSAR, Rennes, 29p.

Schuhmann P.W. and Schwabe K.A. (2004). An analysis of congestion measures and heterogeneous angler preferences in a random utility model of recreational fishing. *Environmental and Resource Economics*, 27:429-450

Shelby B. (1981). Crowding models for backcountry recreation. *Land Economics*, 56:43-55.

Tempesta T. and Thiene M. (2004). *The willingness to pay for the conservation of mountain landscape in Cortina d'Ampezzo (Italy).* 90th EAAE seminar "Multifunctional agriculture, policies and markets: understanding the critical linkage", 27-28 october 2004, Rennes, France, 15p.

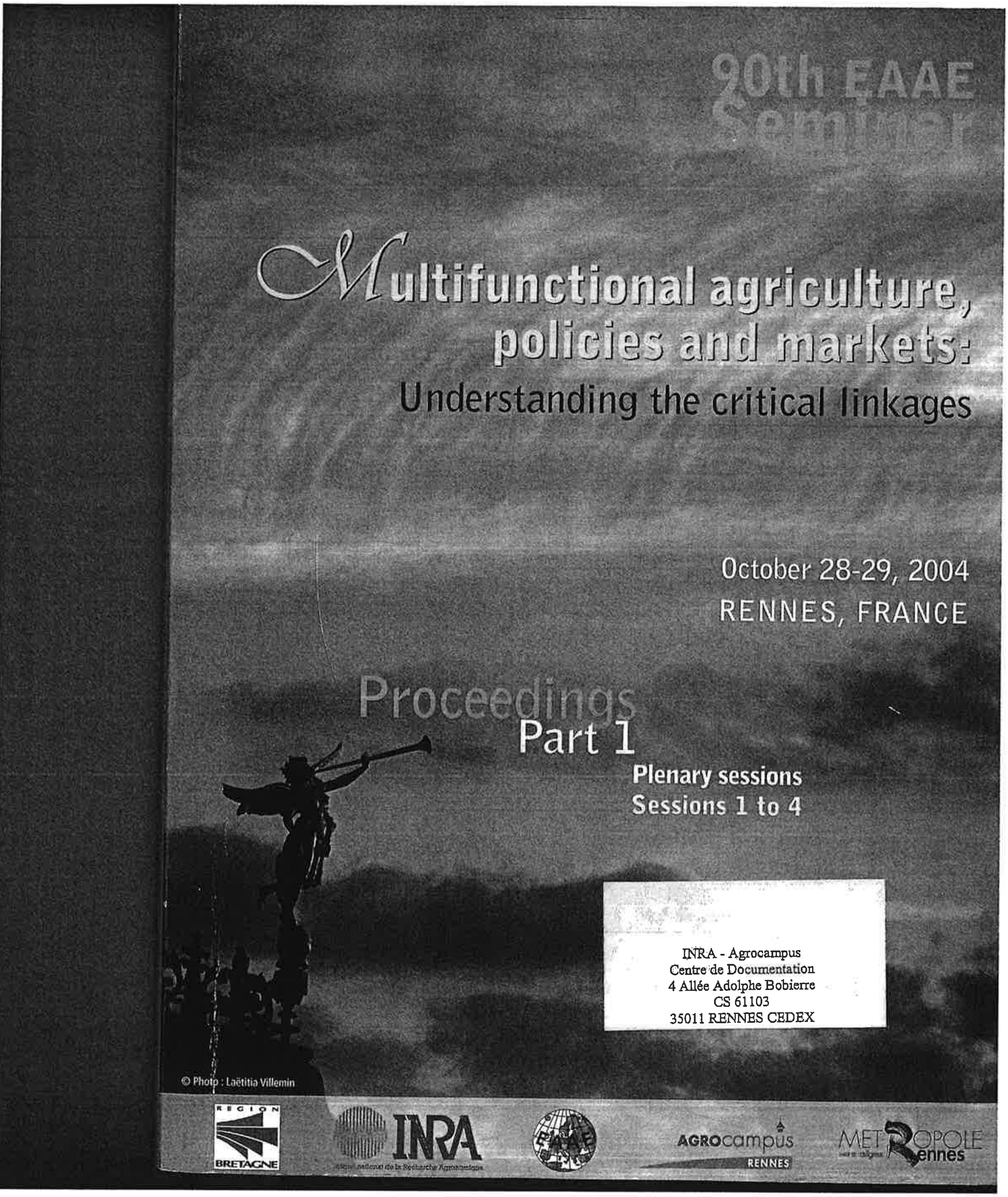
Thibault M. (1992). *Analyse historique de la mise en place des mesures de gestion des ressources halieutiques des eaux continentales.* In : Rencontres Halieutiques de Rennes – Les eaux continentales françaises : une richesse à gérér (Ombredane D. Ed.), Actes du colloques des 28-29 février 1992, ENSAR, Rennes.

Train K.E. (1998). Recreation demand models with taste differences over people. *Land Economics*, 74:230-239.

Train K.E. (2003). *Discrete choice methods with simulation.* Cambridge University Press, UK, 335p.

Willis K.G. and Garrod G.D. (1993). Valuing landscape: a contingent valuation approach. *Journal of Environmental Management*, 37:1-22.

World Wide Fund [WWF] (2001). *The status of wild Atlantic salmon: A river by river assessment.* WWF – Norway, Oslo, Norway, 179p.


90th EAAE Seminar
Multifunctional agriculture, policies and markets:
Understanding the critical linkages
October 28-29, 2004
RENNES, FRANCE
Proceedings
Part 1
Plenary sessions
Sessions 1 to 4
© Photo : Laëtitia Villemin
REGION BRETAGNE
INRA
EAAE
AGROcampus RENNES
METROPOLE Rennes