Controlling the Biological Invasion of a Commercial Fishery by a Space Competitor: A Bioeconomic Model with Reference to the Bay of St-Brieuc Scallop Fishery

Marjolaine Frésard and Jean Boncoeur

This paper presents a bioeconomic model of a commercial fishery facing biological invasion by an alien species acting as a space competitor for the native species. The model is illustrated in a case study of the common scallop fishery of the Bay of St-Brieuc (France), where biological invasion by a slipper-limpet (*Crepidula fornicata*) is now addressed by a control program. First we present the model, which combines the dynamics of the two competing stocks. We then use the model to analyze the equilibrium of the fishery under various assumptions concerning invasive species control, and to assess the social cost of the invasion. Finally we propose a set of dynamic simulations concerning the ongoing program, emphasizing the influence of its starting date on its overall economic results.

Key Words: aquatic invasive species, biological invasion control, common scallop, ecosystemic fisheries management, plurispecies bioeconomic modeling, slipper-limpet

According to Article 8(h) of the Convention on Biological Diversity (1992), invasive alien species are exotic species that are introduced, establish, and spread in an ecosystem, causing environmental and economic harm by threatening habitats and native species. Invasive alien species may impose significant losses in terms of foregone output, biodiversity loss, and reduced ecological services.

In order to estimate the loss of welfare due to invasion, it is necessary to link a model of the underlying ecological relationships with an economic model (Perrings, Williamson, and Dalmazzone 2000). This implies representing the dynamics of the invasion and its various interactions with valuable species, ecosystem services, and biodiversity loss. Knowler and Barbier (2000) focus their analysis on the economic impact of an invader competing with a valuable resident species. They present a general model describing the dynamics of this species subject to interspecific competition and environmental changes caused by the invasive species, and they measure the economic impact of the invasion as the difference between the steady-state returns of the harvesting industry in the ex-ante and ex-post situations. Their theoretical analysis is followed by an empirical case study dedicated to the economic impact of the comb-jelly *Mnemiopsis leidyi* on the Black Sea anchovy fishery.

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This paper studies the economic impact of the invasion of a commercial fishery by an exotic shellfish acting as a space competitor for the targeted native species. The case study is located in the Bay of St-Brieuc, a 800 km^2 bay on the northern coast of Brittany (France). The targeted native species is the common scallop (*Pecten maximus*), and the invasive species is a slipper-limpet (*Crepidula fornicata*). Originating from North America, this species was accidentally imported some decades ago and has gradually settled in many French coastal waters (Blanchard 1995). Devoid of commercial value, it mainly acts as a space competitor for common scallop, a highly valued commercial species in France. By reducing the size of areas that are suitable for scallop beds (Chauvaud 1998), the spread of slipper-limpets threatens the long-run viability of several scallop fisheries. Among them is the scallop fishery of the Bay of St-Brieuc, the second largest common scallop fishery in France. This fishery is seasonally operated by some 250 artisanal boats subject to a limited entry license system (Fifas, Guyader, and Boucher 2003). During the 2001/2002 scalloping campaign, these boats landed 5,529 tons of common scallop, representing an ex-vessel value of €11.8 million (Anon. 2002). The presence of slipper-limpets in the bay was first noticed in 1974 (Dupouy and Latrouite 1979); twenty years later, the stock of slipper-limpets was estimated to be 250,000 tons (Hamon and Blanchard 1994). This invasive process led to the establishment of a control program, based on yearly dredging campaigns (Anon. 2005).

After an empirical economic study dedicated to the case of the Bay of Brest (Frésard and Boncoeur 2004), we present here a bioeconomic model based on the case of the Bay of St-Brieuc. Following Knowler and Barbier (2000), this model describes the impact of an invasive alien species on the profit generated by harvesting a valuable resident species in a bounded ecosystem. In our model, the negative impact of the invasive alien species relies on space competition with the native species. To represent the combined dynamics of the invasive and native stocks, we use a competing species bioeconomic model derived from Flaaten (1991).[1] This model allows comparisons between pre- and post-invasion steady-states (under various assumptions concerning the control policy), and is also suitable for dynamic simulations.

The paper is organized in three sections. The first section is devoted to a general presentation of the model. In the second and third sections, the model is used for different types of analysis. In a comparative static perspective, the second section analyzes the equilibrium of the fishery as a function of the level of invasive species control, and assesses the economic impact of the invasion according to Knowler and Barbier's (2000) definition. In the third section, dynamic scenarios of the evolution of the fishery are explored with the help of the model. The simulations are based on the invasive species control program which is ongoing in the Bay of St-Brieuc fishery.

The Model

Assumptions

- We consider a fishery that is affected by the proliferation of an invasive alien species. This proliferation is the only environmental perturbation affecting the fishery.
- Only the consequences of the proliferation of the invasive species on the stock of native species are considered. In the case of the Bay of St-Brieuc, other possible impacts of invasion by slipper-limpets, such as biodiversity reduction, have not yet been ascertained (Hamon et al. 2002).
- The invasive species acts as a space competitor for the native species.[2] This competition is asymmetric: while the occupation of space by slipper-limpets makes it increasingly difficult for scallop juveniles to settle down, the presence of scallops does not hinder the development of slipper-limpets, which can stick to scallop shells and develop colonies on these shells (Hamon et al. 2002, Chauvaud et al. 2003). It is assumed that uncontrolled growth of the invasive stock leads to eradication of the native stock. All areas in the bay are potentially

[1] The differences between our model and Flaaten's (1991) model are the following: in our model, (i) the competition between the two stocks is asymmetric, (ii) the competition affects the ecosystem's carrying capacity for the native stock, and (iii) the elasticity of CPUE (catch per unit of effort) to stock abundance may differ from one.

[2] Trophic competition between slipper-limpets and scallops, if any, is considered to be a minor phenomenon compared with competition for space (Chauvaud et al. 2003).

subject to colonization by slipper-limpets, and once the density of slipper-limpets is high enough in a given place, it becomes impossible to harvest any scallop in this place.

- A deterministic logistic model is used to represent the natural dynamics of native and invasive stocks. This model has already been applied to the scallop stock of the Bay of St-Brieuc (Mahé and Ropars 2001). Its capacity to predict short-run changes in the stock biomass is limited, since it does not capture the high variability of scallop recruitment under the influence of fluctuating hydroclimatic conditions (Boucher and Dao 1989). However, this drawback becomes less important when analyzing the long run.[3] As regards the invasive stock, the logistic assumption is merely a conjecture, since, up to now, only one assessment of the slipper-limpet stock has been carried out in the Bay of St-Brieuc (Hamon and Blanchard 1994).
- It is assumed that the invasive process is spatially homogenous, i.e., that the density of the invasive species grows at a uniform rate in all parts of the bay. This is clearly an oversimplification, since the spatial development of slipper-limpets is reported to be patchy (Hamon et al. 2002). Taking this feature into account would necessitate a spatially explicit model, distinguishing various areas in the bay according to their degree of invasion. However, in the case under survey, this invasion takes the form of new spots appearing in various places of the bay, rather than a continuous increase in the size of an initially invaded spot (Hamon et al. 2002). As a result, controlling the invasion cannot be thought of as defending a "frontline" between a pristine and a fully invaded area, which makes the simplifying assumption of space homogeneity more acceptable. In our model, the impact of invasion takes the form of a negative relationship between the invasive stock biomass and the carrying capacity of the ecosystem for the native stock.
- For each stock, a constant elasticity of catch per unit of effort (CPUE) to stock abundance is assumed. In the case of the native stock, available data (Table 1) do not provide any evidence that this elasticity is significantly different from one, which supports the standard assumption of proportionality between CPUE and biomass. In the case of the invasive stock, it is suspected that the elasticity of CPUE to stock abundance is greater than one. In other words, if one tries to fish down this stock, CPUE might well decrease at a faster rate than biomass. This conjecture, which corresponds to the case that Hilborn and Walters (1992) named "hyperdepletion," could explain the irreversibility of the invasive process. In the model, both "proportionality" and "hyperdepletion" hypotheses will be considered for the harvest of the invasive stock.
- Unit prices and costs are exogenous. In the case of common scallop, this simplifying assumption is open to criticism, since available data suggest a connection between landings and ex-vessel price (Table 1).[4]

Equations

For each stock, the time variation of biomass X is equal to the difference between its natural variation N and fishing mortality Y:

$$(1) \qquad dX_i / dt = N_i - Y_i \quad (i = 1, 2),$$

where subscript 1 refers to the native stock and 2 refers to the invasive stock. The natural growth of each stock biomass follows a logistic path:

$$(2) \qquad N_i = r_i X_i \left(1 - \frac{X_i}{K_i}\right) \quad (i = 1, 2),$$

[3] The stability of hydroclimatic conditions cannot be taken for granted in the long run. However, in the case of common scallop, "long run" does not necessarily mean many years, since recruitment normally occurs at age 2, and most scallops are harvested at age 2 or 3 (Boucher and Dao 1989, Fifas, Guyader, and Boucher 2003).

[4] However, the interpretation of the decrease in the ex-vessel price of scallops in the first part of the 1990s is not straightforward, since the growing abundance of scallops during this period was observed in all major French scallop fisheries, and not only in the Bay of St-Brieuc. Moreover, industrial processing of scallops harvested in the Bay of St-Brieuc area has developed in the second part of the last decade (Anon. 2002), which might increase the dependence of the Bay of St-Brieuc ex-vessel price on the world market price of frozen pectinids.

Table 1. The Bay of St-Brieuc Common Scallop Fishery: Resource and Activity

Harvesting season	Harvestable biomass (tons)	Landings (tons)	Number of licensed boats	Fishing effort (hours)	Average ex-vessel price (€/kg)	
					Nominal	Real[a]
1990–1991	3,220	1,670	293	7,102	2.89	3.62
1991–1992	4,650	3,320	279	5,690	1.88	2.28
1992–1993	6,770	4,700	266	8,321	1.59	1.89
1993–1994	10,080	5,300	261	5,686	1.61	1.88
1994–1995	10,850	5,530	260	6,921	1.69	1.94
1995–1996	9,430	4,550	260	6,038	1.82	2.05
1996–1997	7,390	3,730	260	8,345	1.87	2.07
1997–1998	8,560	3,710	254	8,949	1.87	2.05
1998–1999	6,710	2,830	251	7,636	2.04	2.23
1999–2000	5,780	2,710	244	7,609	2.12	2.28
2000–2001	9,900	3,570	241	9,357	2.35	2.49
2001–2002	7,990	5,600	243	8,358	2.13	2.22
2002–2003	16,260	6,200				
2003–2004	15,720	6,510				

[a] In constant 2003 euros (using as a deflator the general consumption price index).

Sources: Fifas, Guyader, and Boucher (2003) and Fifas (2004) for harvestable biomass and landings, INSEE for price deflator, and Anon. (2002) for other data.

where r_i is the intrinsic growth rate of stock i, and K_i is the carrying capacity of the ecosystem for stock i.[5] For each stock, the relationship between CPUE and biomass is written as

$$\frac{Y_i}{E_i} = q_i\, X_i^{\alpha_i} \qquad (i = 1, 2), \tag{3}$$

where q_i and α_i are positive parameters, and E_i is the fishing effort devoted to the harvest of stock i.[6] The following relationship describes the asymmetric competition between invasive and native stocks:

$$K_1 = K_{1\max}\left(1 - \frac{X_2}{K_2}\right) \qquad (X_2 \le K_2), \tag{4}$$

where $K_{1\max}$ is the virgin (pre-invasion) carrying capacity of the ecosystem for the native stock. Finally, for each harvesting activity, we write profit (π_i) as the difference between revenue and cost:

$$\pi_i = P_i\, Y_i - C_i\, E_i \qquad (i = 1, 2), \tag{5}$$

where P_i is the ex-vessel unit price of catch of species i, and C_i is the unit cost of effort devoted to harvesting stock i.

Numerical Illustration

Available data are too scarce for an econometric validation of the model. What is proposed here is only a numerical illustration, where the model parameters are given "reasonable" values, whenever information is available.

Common scallop (native species). The common scallop stock of the Bay of St-Brieuc is assessed

[5] A more general version of this natural growth model, where

$$N = rX(1 - (X/K)^{\theta}),$$

has been recently tested on 1,780 biological populations (Sibly et al. 2005). It appeared that a majority of these populations were characterized by $\theta < 1$. This result suggests that density-dependent effects tend to manifest themselves in early life stages, and that populations above their carrying capacity are slow to decline, while populations that are below their carrying capacity are quick to increase.

[6] Parameter α_i represents the elasticity of CPUE to biomass of stock i. If its value is set to 1, CPUE is proportional to stock abundance, and the submodel describing the dynamics of the corresponding harvested stock (given its carrying capacity) is the classical Schaefer model (Schaefer 1954, 1957). The case where $\alpha_i > 1$ corresponds to hyperdepletion.

at the beginning of each fishing campaign by the French marine research institute, Ifremer. Concerning biomass and catch, data covering fishing campaigns from 1990–1991 to 2001–2002 have been published by Fifas, Guyader, and Boucher (2003) [updated by Fifas (2004) for 2002–2003 and 2003–2004]. These data are reproduced in Table 1. Using Ordinary Least Squares (OLS), we estimate the following equation:

(6) $$Y_{1t} = a\,X_{1t} + b\,X_{1t}^2 + u_t\,.$$

The results are

$$\begin{array}{lll} a = 0.649 & b = -1.596 \times 10^{-5} & R^2 = 0.74 \\ (9.36) & (-2.82) & \end{array}$$

(T-values are shown in parentheses). Variables X_{1t} and $X_{1t}{}^2$ are statistically significant with a 95 percent probability. Requirements of normal distribution of residuals (85 percent probability), non-autocorrelation, homoskedasticity, and time-stability are satisfied. Assuming equilibrium and making use of equations (1) and (2), we may then identify r_1 with a, and ($-r_1 / K_1$) with b. As a result, we get

$$r_1 = 0.649 \quad \text{and} \quad K_1 = 40{,}689 \text{ tons.}$$[7]

These estimations are open to criticism for several reasons. First, due to the multicolinearity between variables X_{1t} and $X_{1t}{}^2$ (Farrar-Glauber test), the estimated variance of coefficients is inefficient, and, consequently, the significance of the variables may be highly dependent on the sample. Furthermore, a bias is due to the fact that the estimation of K_1 involves a non-linear transformation of estimated coefficients. Another concern is the fact that the equilibrium assumption is at odds with important variations in biomass and landings during the observation period (see Table 1). It is worth recalling that the model neglects the strong influence exerted by hydroclimatic conditions on the recruitment of scallops. While catch and biomass have experienced parallel changes during the survey period, fishing effort has been much more stable, which suggests that the major part of the changes in biomass and catch during this period is due to hydroclimatic fluctuations that are not taken into account by the model. For these reasons, the values of parameters r_1 and K_1 used in the numeric simulations of this paper should be regarded as illustrative, rather than as the results of a genuine calibration.[8]

The carrying capacity for scallops in the Bay of St-Brieuc during the 1990s was adversely affected by the level of slipper-limpet biomass. Assuming that this biomass represented 25 percent of the bay's carrying capacity for slipper-limpet at that time (see below) and combining this assumption with the estimated value of K_1, we may derive from (4) the estimated pristine (pre-invasion) carrying capacity for scallops:

$$K_{1\max} = 54{,}252 \text{ tons.}$$

The remark concerning the illustrative character of the growth function parameters also applies to the parameters of the catch function (3). Making use of data presented in Table 1, we first estimate the following double log transformation of equation (3):

(7) $$\ln Z_{1t} = c + \alpha_1 \ln X_{1t} + v_t\,,$$

where Z_{1t} is the CPUE at year t, and c is the neperian logarithm of q_1. The results are

$$\begin{array}{lll} c = -7.37 & \alpha_1 = 0.75 & R^2 = 0.44 \\ (-3.1) & (2.8) & \end{array}$$

(T-values are shown in parentheses). Requirements of normal distribution of residuals, non-autocorrelation, homoskedasticity, and time-stability are satisfied. According to these results, stock biomass plays a significant role in explaining CPUE. The elasticity of CPUE to biomass is

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[7] Applying the same model to the same stock, Mahé and Ropars (2001) propose the following estimations: 0.535 for r_1 and 30,398 for K_1. The t-statistics are respectively 3.01 and 2.6, and R^2 is equal to 0.4. But the covered period and the number of observations are not specified.

[8] Alternatively, available data could be used to calculate the change in stock at each period, from which natural growth could be derived by adding observed landings. This reconstructed surplus production could then be regressed on X_1 and $X_1{}^2$ to estimate the parameters in equation (2). The advantage of this method is that it does not assume stock equilibrium. However, in the case we deal with, regressing N_1 on X_1 and $X_1{}^2$ does not provide statistically significant results. This is no surprise, considering the strong impact of fluctuating hydroclimatic conditions on yearly scallop recruitment.

not statistically different from one: the limits of the 95 percent confidence interval for α_1 are 0.15 and 1.35. Therefore we keep the hypothesis of proportionality between CPUE and biomass ($\alpha_1 = 1$), which is fairly standard in fisheries literature, and makes analytic resolution of the model easier. To obtain a numerical value for the catchability coefficient, we estimate the following equation:

$$Z_{1t} = q_1 X_{1t} + w_t . \tag{8}$$

The results are

$$q_1 = 7.05 \qquad R^2 = 0.39$$
$$(11.74)$$

(T-value is shown in parentheses).

Unit prices and costs, expressed in real terms, are assumed to be constant over time. Their estimation is based on available data for the 1990s. Nominal data are deflated with the help of the general index of consumption prices (INSEE). All "real" prices and costs are expressed in terms of constant 2003 euros. Data concerning ex-vessel prices are provided by auction markets (Table 1). According to this source, the average ex-vessel real price of scallops harvested in the Bay of St-Brieuc was €2.15 per kg during the 1990–2002 period. Deducting landing taxes (6 percent of ex-vessel prices on the average) leads to the following real net price of landed scallops:

$$P_1 = €2 \text{ per kg.}$$

Time series concerning scalloping costs in the Bay of St-Brieuc are not available. As a proxy, we use the results of a field survey (Pascoe 2000) providing estimations for 1997 costs in the French scallop fisheries of the western part of the English Channel (a group of fisheries where the Bay of St-Brieuc is by far the major component). Table 2 presents the estimations concerning non-wage variable costs, by boat-length class. According to these estimations, the average non-wage variable cost of dredging (expressed in nominal terms) was €69.44 per day and per boat in 1997. With a dredging time usually limited to 50 minutes per day, this daily cost corresponds to €83.33 per hour and per boat on the average (in nominal terms). We use the legal minimum wage to assess the opportunity cost of labor.[9] This wage rate[10] (expressed in nominal terms) was €8.35 per hour in 1997. With an average 2.4 fishermen per boat, the resulting estimation for the opportunity cost of labor is €20.04 per fishing hour and per boat (in nominal terms). Adding this cost to the variable non-wage cost leads to an average €103.37 (nominal terms), or €113.10 in real terms. On this basis, we assume that the real overall variable cost of effort in the fishery is[11]

$$C_1 = €113 \text{ per hour and per boat.}$$

Slipper-limpet (invasive species). Quantitative knowledge is much more limited about slipper-limpet than about common scallop. Presence of the species in the Bay of St-Brieuc was first mentioned in 1974 (Dupouy and Latrouite 1979). In 1994, it covered approximately 200 km^2, i.e., 25 percent of the whole bay area, with a biomass estimated at 250,000 metric tons (Hamon and Blanchard 1994). No stock assessment has been carried out since 1994, but there is evidence that the invasion process is ongoing (Hamon et al. 2002). At present, slipper-limpet biomass grows at a rate that probably does not exceed 5 percent per year (Blanchard 2005). A quantitatively realistic population dynamics model cannot be derived from this limited information. For the sake of numeric illustration, we assume that the bay's carrying capacity for slipper-limpet is four times the 1994 estimated stock, i.e.,

$$K_2 = 1 \text{ million tons,}$$

and that the intrinsic growth rate is

$$r_2 = 0.04,$$

which, according to the logistic model, corresponds to a growth rate of 3 percent in 1994.

Some harvest data are available from the invasion control program that was started in 1998

[9] As usual in artisanal fisheries, wages of crew members (including the skipper) are fixed on the basis of the so-called "share system," which implies a participation in profit.

[10] Including national insurance costs.

[11] Fixed costs are not taken into account, because they are not specific to the scalloping activity, which is only a part-time activity for fishermen operating in the Bay of St-Brieuc.

Table 2. French Scalloping Fleet of the Western English Channel: Variable Non-Wage Daily Dredging Costs (1997)

Boat length class	7–10 meters	10–12 meters	12–13 meters
Total number of boats	148	103	69
Number of surveyed boats	9	21	14
Daily costs (€)[a]			
Mean	42.69	78.82	111.75
Standard deviation	37.65	37.65	49.24

[a] Current prices.

Source: Pascoe (2000).

(Anon. 2005). After an experimental phase during the years 1998–2001, the program has operated routinely since 2002. During the period 2002–2004, a specialized boat harvested 35,796 tons of slipper-limpet with a hydraulic dredge. The total fishing time was 375 hours, and the corresponding cost was €533,715.[12]

There are not enough available data to estimate statistically significant parameters of the harvest function [equation (4)]. Concerning the elasticity of CPUE to stock (parameter α_2), two scenarios will be run in parallel: a "proportionality" scenario ($\alpha_2 = 1$) and a "hyperdepletion" scenario ($\alpha_2 > 1$). In the latter case, we will assume, for the sake of numeric illustration, that ($\alpha_2 = 2$). Taking these values for parameter α_2, we derive the corresponding values of parameter q_2 from equation (4), in which catch and effort are set at their average value for years 2002–2004 ($Y_2 = 11{,}932$ tons; $E_2 = 125$ hours) and biomass at its level derived from the logistic model for the same period ($X_2 = 321{,}475$ tons):

$$q_2 = 2.963 \times 10^{-4}$$
(proportionality scenario: $\alpha_2 = 1$)

$$q_2 = 9.237 \times 10^{-10}$$
(hyperdepletion scenario: $\alpha_2 = 2$).

Harvested slipper-limpets are delivered for free to a processing factory, where they are rendered into fertilizers. In the present situation, the revenue derived from this operation scarcely covers its cost. Therefore we will suppose

$$P_2 = 0.$$

[12] Public subsidies covered more than 90 percent of this cost.

Under this assumption, profit generated by harvesting the invasive species (π_2) is always negative, and its absolute value is equal to the harvesting cost. Our estimation of the unit cost of effort is based on the control program data for years 2002–2004:

C_2 = €1,423 per hour of hydraulic dredging.

Equilibrium Analysis

In this section, we assume biological equilibrium for both stocks, i.e., that the time-variation of each of them is zero. Should there be no control program, such an equilibrium would be reached only when the invasive species had completely superseded the native stock. However, with a control program, the invasive stock may be stabilized at a level that leaves some room for the native stock, and, as a consequence, that allows sustainable harvesting of this stock.

According to (1), the equilibrium condition implies that catch are equal to natural variation of biomass. Such catch are called "sustainable" because, under steady environmental conditions, they may be sustained indefinitely with a constant fishing effort. By introducing the equilibrium condition into (1) and combining it with (2), we can express, for each stock, sustainable catch as a function of the equilibrium biomass:

$$(9) \qquad Y_i = r_i X_i \left(1 - \frac{X_i}{K_i}\right) \qquad (X_i \le K_i) \;\; (i = 1, 2).$$

After presenting the consequences of the equilibrium assumption on each stock and corresponding fishing activity, we investigate the question of the optimal level of control of the invasive stock.

Native Stock

For the native stock ($i = 1$), we consider equations (9) and (3), with ($\alpha_1 = 1$). Combining these two equations makes it possible to express the equilibrium biomass and the corresponding level of catch as functions of fishing effort:

$$(10) \qquad X_1 = K_1\left(1-\frac{q_1 E_1}{r_1}\right) \quad (E_1 \leq q_1/r_1)$$

$$(11) \qquad Y_1 = q_1 E_1 K_1\left(1-\frac{q_1 E_1}{r_1}\right) \quad (E_1 \leq q_1/r_1).$$

In the standard Schaefer model used here, the equilibrium biomass is a decreasing linear function of the steady-state level of fishing effort and, consequently, the corresponding level of catch is a quadratic concave function of fishing effort. The maximum level of catch in the long run, or *maximum sustainable yield* (MSY), is equal to ($r_1.K_1 / 4$), and the corresponding levels of equilibrium biomass and effort are respectively ($K_1 / 2$) and ($r_1 / 2.q_1$). If, in the long run, fishing effort exceeds the level corresponding to MSY, sustainable catch becomes a decreasing function of effort (overfishing). Equilibrium biomass and catch go to zero if effort remains above (r_1 / q_1).

Equations (10) and (11) depend on the level of the carrying capacity of the ecosystem for the native stock, which, according to equation (4), is negatively influenced by the presence of the invasive species. The impact of the invasion on (10) and (11) is illustrated in Figure 1.

Steady-state profit may be expressed as a function of fishing effort by combining (5) and (10):

$$(12) \qquad \pi_1 = P_1 q_1 E_1 K_1\left(1-\frac{q_1 E_1}{r_1}\right) - C_1 E_1 \quad (E_1 \leq q_1/r_1).$$

According to this relationship, profit cannot be positive in the long run if the carrying capacity for the native stock is less than

$$(13) \qquad \tilde{K}_1 = \frac{C_1}{P_1 q_1}.$$

If K_1 exceeds this break-even capacity, the long-run maximum profit is

$$(14) \qquad \pi_1^* = \frac{P_1 r_1 K_1}{4}\left(1-\frac{\tilde{K}_1}{K_1}\right)^2 \qquad (K_1 \geq \tilde{K}_1).$$

Because (14) is an increasing function of K_1, long-run maximum profit is negatively influenced by the invasive process.

Invasive Stock

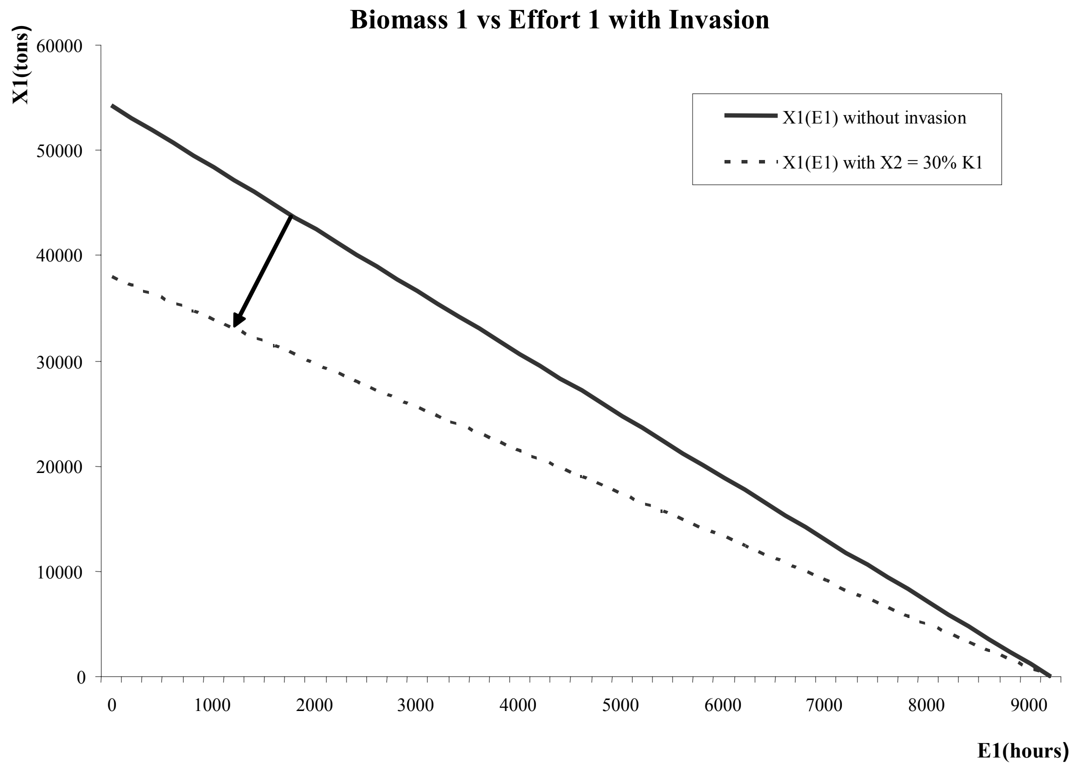
In the case of stock 2, we have equilibrium relationships between fishing effort, biomass, and catch by combining equations (9) and (3). These relationships depend on the value of parameter α_2, which is the elasticity of CPUE to stock biomass.

If this parameter is equal to one (the "proportionality" scenario), the equilibrium relationships between E_2, X_2, and Y_2 are similar to the equilibrium relationships between E_1, X_1, and Y_1 (Figure 2, proportionality scenario). In this case, complete eradication of the invasive stock is technically feasible, by setting fishing effort at (r_2/q_2). However, this is not a steady-state effort level: once complete eradication has been achieved, the invasive species harvesting effort falls to zero.

In the more general case where ($\alpha_2 \in [0,\infty[$), the equilibrium relationship between E_2 and X_2 is written as

$$(15) \qquad E_2 = \frac{r_2}{q_2} X_2^{1-\alpha_2}\left(1-\frac{X_2}{K_2}\right) \quad (X_2 \leq K_2).$$

If α_2 is greater than one (the "hyperdepletion" scenario), it is not always possible to express analytically X_2, and hence Y_2, in terms of E_2. However, it appears from (15) that, for any X_2 between zero and K_2, E_2 is a positive, differentiable, strictly decreasing, and convex function of X_2.



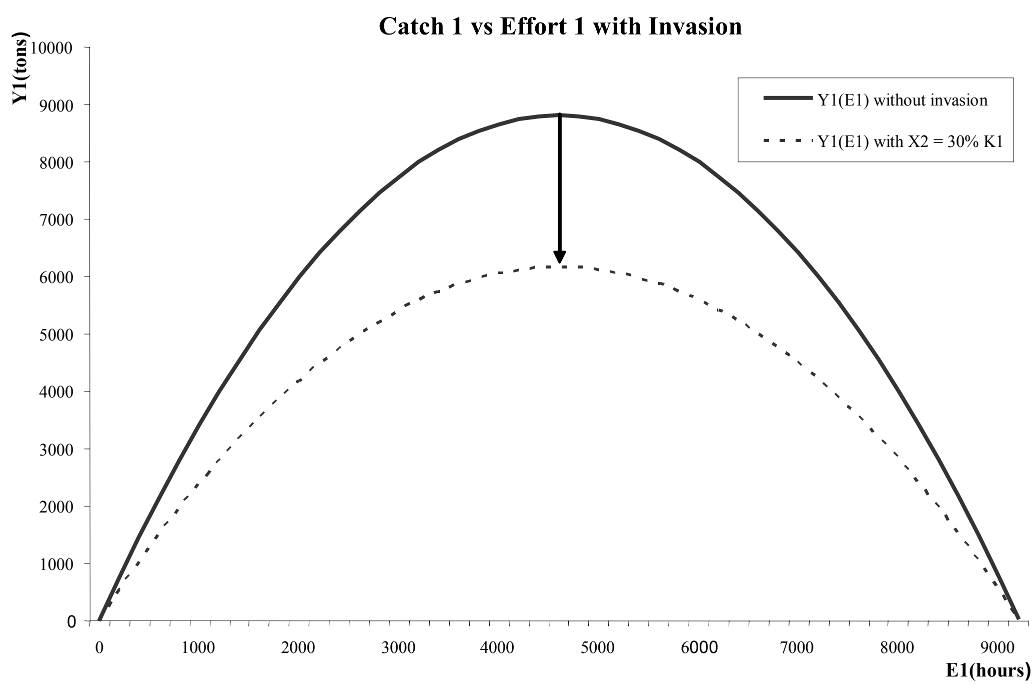


Figure 1. Impact of Invasion on Native Species Equilibrium Biomass and Catch, as Functions of Fishing Effort

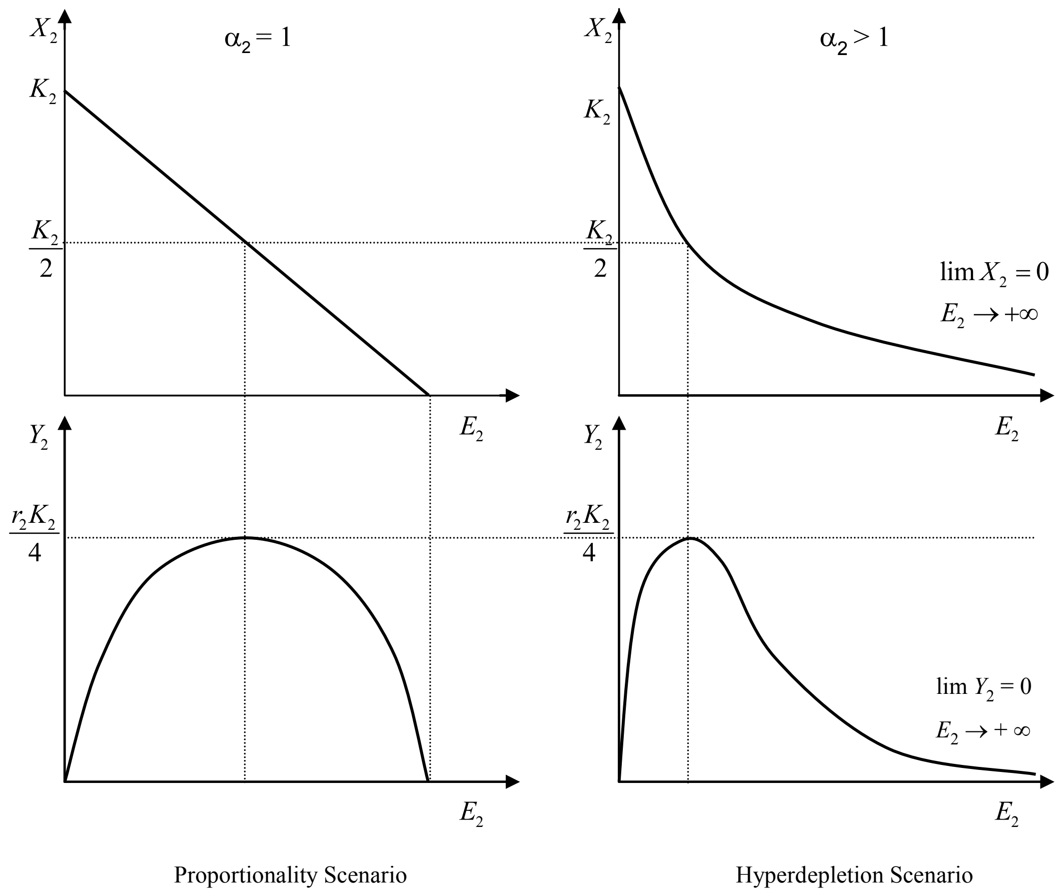


Figure 2. Relation Between Invasive Species Fishing Effort, Biomass, and Catch

Therefore, the reciprocal function $X_2(E_2)$ is defined as differentiable, positive, strictly decreasing, and convex for any non-negative level of effort.[13] It is equal to K_2 when ($E_2 = 0$), and has a zero asymptotic limit as E_2 increases indefinitely (Figure 2, hyperdepletion scenario). This result indicates that total eradication of the invasive stock is impossible in the hyperdepletion case.

The level of sustainable catch Y_2 is a function of the equilibrium biomass X_2 [equation (11)], itself a function of effort E_2 for any ($\alpha_2 \geq 1$). Consequently, Y_2 may also be considered as a function of E_2:

$$(16) \qquad Y_2(E_2) = Y_2[X_2(E_2)] .$$

The properties of this composed function may be derived from the properties of the component functions $Y_2(X_2)$ and $X_2(E_2)$. The function $Y_2(E_2)$ is increasing and strictly concave as long as E_2 is less than

$$\frac{r_2}{2q_2}\left(\frac{K_2}{2}\right)^{1-\alpha_2} .$$

[13] A similar result would be obtained with a variant of the simple logistic model, where

$$N_2 = r_2 X_2 (1 - (X_2 / K_2)^{\theta}),$$

provided θ is less than 1. Assuming $\theta < 1$ would imply greater ongoing control costs to contain the invasive stock than would assuming $\theta = 1$. While the implication of $\theta < 1$ on the relationship between steady-state biomass and harvesting effort is similar to the implication of hyperdepletion ($\alpha_2 > 1$), the mechanism of action is a consequence of changes in the population growth rate rather than changes in catchability. In the case under survey, it is not possible to decide, on the basis of currently available information, which mechanism squares better with the facts.

When E_2 is set at this level, $Y_2(E_2)$ is maximum (MSY) and equal to $(r_2.K_2 / 4)$. It becomes a decreasing function when E_2 goes beyond this point. In the case where α_2 is greater than one (hyperdepletion), $Y_2(E_2)$ tends asymptotically towards zero as E_2 increases indefinitely (Figure 2, hyperdepletion scenario).

Control Program and Social Cost of Invasion

We now consider the optimal program for controlling the population of the invasive species, assuming biological equilibrium conditions. The native species fishery is supposed to be managed optimally, with fishing effort set at its long-run profit-maximizing level (E_1*). This level depends on the carrying capacity of the ecosystem for the native stock (K_1), and, consequently, on the equilibrium level of the invasive stock biomass (X_2). If the invasive stock is not harvested, its biomass will reach equilibrium when the ecosystem is fully invaded ($X_2 = K_2$), which implies that ($K_1 = 0$) and ($E_1* = 0$). Therefore, the problem is to determine the optimum level of invasive species biomass control, i.e., the one that maximizes the overall long-run surplus of the control program. This surplus is defined as

$$\pi = \pi_1^* + \pi_2 . \tag{17}$$

With a zero-price for invasive species, π_2 is necessarily non-positive, and its absolute value is simply the cost of harvesting this species. Both this cost and the optimal profit of the native species fishery (π_1*) will be expressed hereafter in terms of invasive species equilibrium biomass (X_2).

As regards π_1*, this operation is achieved by combining (4) and (14). By doing so, we get

$$\pi_1^*(X_2) = \frac{P_1 r_1}{4}\left(1 - \frac{X_2}{K_2}\right)\left(1 - \frac{K_2 - \tilde{X}_2}{K_2 - X_2}\right)^2 \quad (X_2 \le \tilde{X}_2) \tag{18}$$

with

$$\tilde{X}_2 = K_2\left(1 - \frac{\tilde{K}_1}{K_{1\max}}\right), \tag{19}$$

the level of invasive species biomass bringing the carrying capacity of the ecosystem for the native species down to the point where optimal profit in the native fishery is zero, i.e., such that

$$X_2 = \tilde{X}_2 \Rightarrow K_2 = \tilde{K}_2 .$$

As long as X_2 is below this level, function $\pi_1*(X_2)$ is strictly positive, decreasing, and convex. Its extreme values are

$$\pi_{1\max}^* = \frac{P_1 r_1 K_{1\max}}{4}\left(1 - \frac{\tilde{K}_1}{K_{1\max}}\right)^2 \text{ when } (X_2 = 0) \tag{20}$$

$$\pi_1^* = 0 \text{ when } (X_2 = \tilde{X}_2) . \tag{21}$$

Expressing the cost of harvesting the invasive species in terms of X_2 requires multiplying both sides of (15) by C_2, which is the unit cost of effort devoted to this activity:

$$|\pi_2(X_2)| = C_2\, E_2 = \frac{C_2\, r_2}{q_2} X_2^{1-\alpha_2}\left(1 - \frac{X_2}{K_2}\right) \tag{22}$$

$$(\alpha \ge 1) \quad (X_2 \le K_2) .$$

Due to the negative relation between equilibrium biomass and harvesting effort, $|\pi_2|$ is a decreasing function of X_2. It is equal to zero when ($X_2 = K_2$), i.e., when the invasive species is not harvested. In the proportionality case ($\alpha_2 = 1$), the relation between $|\pi_2|$ and X_2 is linear, and the cost of eradicating stock 2 is equal to $(C_2.r_2 / q_2)$ (however, this is not a permanent cost: once full eradication has been achieved, the invasive species harvesting cost becomes zero). In the hyperdepletion case ($\alpha_2 > 1$), the relation between $|\pi_2|$ and X_2 is strictly convex, and the cost of harvesting stock 2 tends to be infinite as the equilibrium value of this stock gets close to zero (eradication is impossible in this case).

By combining (17), (18), and (22), we may express the global surplus of the control program, assuming biological equilibrium, as a function of the stabilized invasive species biomass:

$$\pi(X_2) = \pi_1^*(X_2) - |\pi_2(X_2)| . \tag{23}$$

In Figure 3, this surplus is visualized as the vertical distance between the two curves representing respectively $\pi_1{}^*(X_2)$ and $|\pi_2(X_2)|$.

Making the control program profitable requires the existence of some level of invasive species biomass, $\hat{X}_2$, such that

$$(24)\qquad 0 \le \hat{X}_2 < \tilde{X}_2 \ \text{ and } \ \pi_1^*(\hat{X}_2) = \left|\pi_2(\hat{X}_2)\right| .$$

In the proportionality case (Figure 3, proportionality scenario), this requirement corresponds to the condition

$$(25)\qquad \frac{C_2\, r_2}{q_2} \le \pi_{1\max} .$$

If this condition holds, the intersection between the two curves is unique, and represents the break-even point of the program (not to be confused with the break-even point of the native species fishery, which does not take into account the invasive species harvesting cost):

$$(26)\qquad X_2 < \hat{X}_2 \Leftrightarrow \pi > 0 .$$

When X_2 is kept below that point, the distance between the convex curve representing $\pi_1{}^*(X_2)$ and the straight line representing $|\pi_2(X_2)|$ increases as X_2 gets closer to zero. As a result, the optimal program follows an "all or none" rule, corresponding either to complete *laissez-faire* (27a) or to complete invasive species eradication (27b):

$$(27a)\qquad \frac{C_2\, r_2}{q_2} > \pi_{1\max} \Rightarrow (X_2^* = K_2 \ ; \ E_2^* = 0)$$

$$(27b)\qquad \frac{C_2\, r_2}{q_2} \le \pi_{1\max} \Rightarrow \left(X_2^* = 0 \ ; \ E_2^* = \frac{r_2}{q_2} \right).$$

In the first case, the corollary of the *laissez-faire* attitude towards invasion is the complete extinction of the native species fishery.

This dilemma between complete invasive stock eradication and extinction of the native species fishery is a consequence of the proportionality assumption. Assuming hyperdepletion makes complete invasive species eradication impossible, and makes room for an intermediate optimal level of control scenario.

With an elasticity of CPUE to stock larger than one, the harvesting cost curve for the invasive species is strictly convex (Figure 3, hyperdepletion scenario). In this case, the intersection between this curve and the one representing $\pi_1{}^*(X_2)$, if it exists, usually is not unique. In Figure 3's depiction of this scenario, there are two positive levels of invasive species stock biomass:

$$\hat{X}_{2a} \ \text{ and } \ \hat{X}_{2b} > \hat{X}_{2a} ,$$

satisfying condition (24). The global surplus of the program is positive between these levels, and negative outside them:

$$(28)\qquad \hat{X}_{2a} < X_2 < \hat{X}_{2b} \Leftrightarrow \pi > 0 .$$

The optimal program for controlling the invasive stock corresponds to a stabilized biomass level $X_2{}^*$ such that

$$(29)\qquad \hat{X}_{2a} < X_2^* < \hat{X}_{2b} \ \text{ and } \ \frac{d\pi_1^*}{dX_2^*} = \frac{d\left|\pi_2\right|}{dX_2^*},$$

i.e., the marginal benefit of decreasing invasive stock for the native species fishery is balanced by the marginal harvesting cost of this decrease.

Following Knowler and Barbier (2000), we may assess the social cost of the invasion as the difference between the optimal equilibrium profit of the uninvaded native species fishery and the equilibrium optimal net benefit of the invasive species control program:

$$(30)\qquad \left|SC_{inv}\right| = \pi_{1\max}^* - \pi^* .$$

If condition (24) does not hold, it is not possible to make the control program profitable under equilibrium conditions. The social cost of the invasion is then equal, in absolute value, to the optimal equilibrium profit of the uninvaded native species fishery. It is lower if condition (24) holds, because then sustainability of the native species fishery may be reconciled with the presence of the invasive species by means of a socially profitable control program.

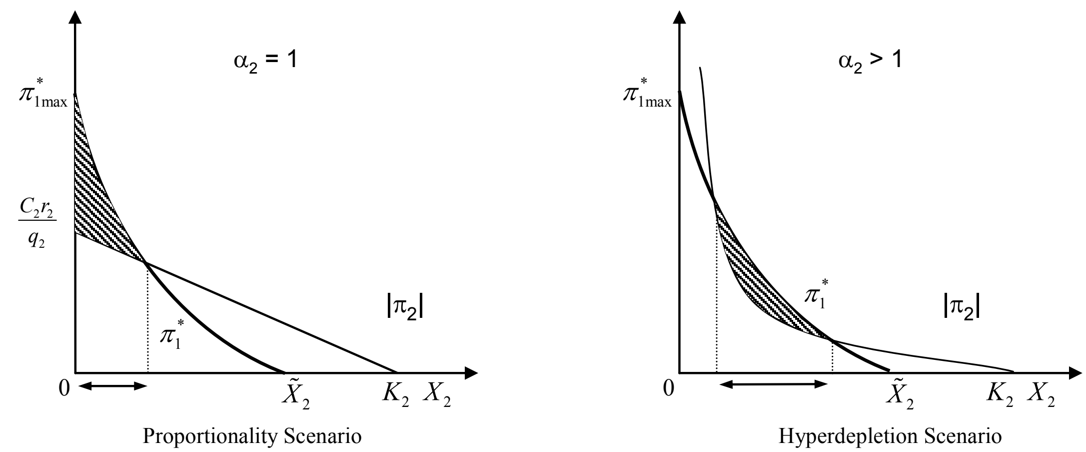


Figure 3. Net Benefit of the Invasive Species Control Program, as a Function of the Stabilized Invasive Biomass

In order to give a quantitative illustration of the equilibrium analysis, we present in Table 3 the equilibrium values based on the numeric illustration of the model presented in the first section of the paper. Results show that, under the proportionality scenario, the optimal eradication solution generates a greater net benefit (€17 million) than the optimal level of stabilized invasive stock biomass of the hyperdepletion scenario (€15 million). But the comparability of these figures is limited by the fact that the equilibrium surplus of the proportionality scenario does not include the eradication cost, which is not permanent. In the case of hyperdepletion, the social cost of invasion, computed according to equation (28), would be €2 million, instead of €17 million if the control program was not carried out.

Sensitivity tests were carried out on two highly uncertain parameters of the invasive species dynamics: present growth rate of the stock and carrying capacity of the ecosystem. Table 4 displays the impact of these tests on the social cost of invasion, assuming hyperdepletion. Concerning the present growth rate of the slipper-limpet stock, two scenarios have been run: 1 percent and 5 percent. According to the results of the sensitivity tests, the social cost of invasion increases with the value of the present estimated growth rate of the invasive stock, and is more than two times greater in the 5 percent scenario than in the 1 percent scenario. Two scenarios have also been run concerning the carrying capacity for slipper-limpets: 625,000 tons (i.e., 2.5 times the 1994 estimated stock biomass), and 2.5 million tons (10 times the 1994 estimated stock biomass). According to tests, the social cost of invasion is a decreasing function of the carrying capacity of the ecosystem for the invasive stock, and its value is divided by approximately 2.4 when the carrying capacity goes from 625,000 tons to 2.5 million tons.

Dynamic Simulation

In this section, we analyze the invasive species control program which is now being implemented in the Bay of St-Brieuc common scallop fishery (Anon. 2005). For this purpose, we use the bioeconomic model that was presented in the first part of the paper, but we drop the assumptions of steady-state and optimal management of the fishery.

Steady-state analysis neglects transition costs. Even though a certain type of long-run equilibrium may be regarded as desirable, reaching this position is usually expensive and time-consuming, and this type of consideration has to be taken into account in a cost-benefit analysis.

As regards fisheries management, despite the fact that nominal effort is controlled and a total allowed catch (TAC) is fixed each year, no empirical evidence suggests that the management of the Bay of St-Brieuc common scallop fishery is ruled by economic optimization (Fifas, Guyader,

Table 3. Equilibrium Analysis: Numerical Simulation Under Two Scenarios Concerning the Invasive Species Harvest Function: Proportionality (α_2 = 1) and Hyperdepletion (α_2 = 2)

Type of equilibrium	IS Biomass[a]	NS carrying capacity[a]	NS fishery profit[b]	IS harvesting cost[b]		Net benefit[b]	
				$\alpha_2 = 1$	$\alpha_2 = 2$	$\alpha_2 = 1$	$\alpha_2 = 2$
No invasion[c]	**0**	54,252	17,099	0	0	**17,099**	17,099
Program lower break-even ($\alpha_2 = 2$)	3,604	54,056	17,036	191	17,036	16,845	0
Optimum ($\alpha_2 = 2$)	**59,200**	51,040	16,057	180	979	15,876	**15,078**
Program break-even ($\alpha_2 = 1$)	983,520	894	3	3	1	0	2
Program upper break-even ($\alpha_2 = 2$)	984,305	851	1	3	1	-2	0
NS fishery break-even	985,220	802	0	3	1	-3	-1
Full invasion	**10^6**	0	0	0	0	**0**	0

[a] In tons.

[b] In 1,000 euros.

[c] And optimum when (α_2 = 1). In this case the invasive species harvesting cost is not permanent, and does not appear in the equilibrium data.

Notes: IS = invasive species, NS = native species.

Table 4. Equilibrium Analysis: Social Cost of Invasion Under Hyperdepletion Scenario[a] (in 1,000 euros)

Invasive species carrying capacity[b]	625	1,000	2,500
Present invasive species growth rate			
1 %	1,676	1,167	684
3 %	2,823	2,021	1,181
5 %	3,592	2,593	1,524

[a] Difference between optimal profit of the uninvaded native species fishery and optimal net benefit of the invasive species control program (equilibrium values).

[b] In 1,000 tons.

and Boucher 2003). Annual authorized fishing time is not correlated to the variations of harvestable biomass, and, during years 1990 to 2003, actual (declared) catch have exceeded scientific recommendations by 22 percent on the average (Fifas 2004). The results of the model presented in the first part of this paper suggest that overfishing is the rule, and the apparently high profitability of the fishery is not at odds with this diagnosis: as scalloping in the bay is only a part-time activity for fishermen and fishing boats, only direct (specific) costs are taken into account when computing the gross margin generated by this activity. These considerations suggest an approach of the invasive species control program in terms of dynamic simulation, rather than static optimization.

The strategy of this program relies on a distinction between two stages, which might be called "rollback" and "containment": the first one is characterized by a high level of invasive species harvesting, in order to decrease significantly the level of invasion; the second step is intended to consolidate the outcome of the "rollback" phase, by harvesting each year the natural surplus produced by the existing biomass.

In the case of the St-Brieuc fishery, the first stage of the program consists of harvesting 100,000 tons of slipper-limpet over five years (according to our model, this represents two times the MSY, estimated at 10,000 tons a year). This stage has started in 2002. For external reasons (limited availability of the boat used for the harvest), the level of harvest during the two first years (9,034 tons in 2002, 8,681 tons in 2003) has lagged far behind the targeted level, but a sharp increase was realized in 2004 (18,081 tons), and the program is now operated in the vicinity of what was initially planned. In our analysis, we will neglect the gap between initial plans and implementation, and assume that 20,000 tons of slipper-limpet are actually harvested each year during five years.

The second stage of the program ("containment") is not yet precisely scheduled, which may be explained by the uncertainty concerning the dynamics of the invasive species: the annual level of harvest in this stage depends on the speed at which cleaned areas are recolonized, a parameter which is not yet well known. Our simulation of this stage will rely on the simple logistic model presented in the first part of the paper, associated with the two variants of the harvest function (proportional CPUE and hyperdepletion).

To simulate the impact of the program on the scallop fishery, we will assume that fishing effort is exogenous: as long as the gross margin generated by the fishery is positive, the annual fishing time is kept equal to the average level observed during years 1990–2001 (approximately 7,500 hours per year). This is probably an oversimplification, but it is intended to reflect in a stylized way the evidence that effort management is hardly influenced by resource abundance. Another simplification concerning fishing effort is the fact that no provision is made for the increase in the catchability coefficient due to technical progress.

For the sake of dynamic simulations, we use a discrete-time version of the continuous-time model presented in the first part of the paper. As regards initial conditions (with year 1 being the first year of program implementation), we suppose that the slipper-limpet biomass is 250,000 tons (corresponding to the 1994 stock assessment), and that the scallop biomass is 7,611 tons (corresponding to the average of the years 1990–2001). The time discount rate is set to 5 percent (a sensitivity test will be presented below concerning this assumption). As above, unit prices and costs are exogenous, the ex-vessel price of slipper-limpets is assumed to be zero, and the cost of processing equals the revenue generated by this activity.

We first investigate the impact of the program on biomass and catch of each species. Then we analyze the economic results of the program, taking into account both the cost of invasive species harvesting and the resulting benefits for the scallop fishery. Finally, we investigate variants concerning the starting date of the program, and we present results of sensitivity tests. At each step, in a cost-benefit perspective, we compare the simulated results of the program with a "no control," or *laissez-faire* scenario.

Impact of the Control Program on Invasive and Native Stock Dynamics

Figure 4 illustrates the consequences of the control program on the dynamics of the invasive species. The two steps of the program appear on this figure: the harvest is set exogenously at 20,000 tons per year during the "rollback" stage, and during the following "containment" stage, it is set at a level that balances the natural growth of the invasive stock biomass, in order to stabilize this biomass. According to the model, this permanent level of harvest is slightly over 6,000 tons per year, i.e., 30 percent of the annual harvest during the "rollback" stage. As a result, the invasive species biomass, which was assumed to be 250,000 tons at the beginning of the program, falls to 184,750 tons at the end of the "rollback" stage (74 percent of the pre-program level), and is stabilized at this level during the second stage of the program. In the *laissez-faire* scenario, the invasive stock biomass increases steadily, and, according to the specification of the logistic model used in the simulation, it more than doubles within 3 decades, reaching 500,000 tons during year 29 (2.7 times the stabilized biomass of the invasive species control scenario).

The consequences of the program on the scallop fishery are displayed in Figure 5. With no control, the bay's carrying capacity for scallops would decline steadily due to increasing competition by the invasive species (approximately by one-third within three decades) as would annual catch, with a time lag due to the permanent disequilibrium generated in the fishery by the declining carrying capacity.

Contrasting with this negative trend, the control program results in stabilizing both carrying capacity and catch of scallops. After three decades,

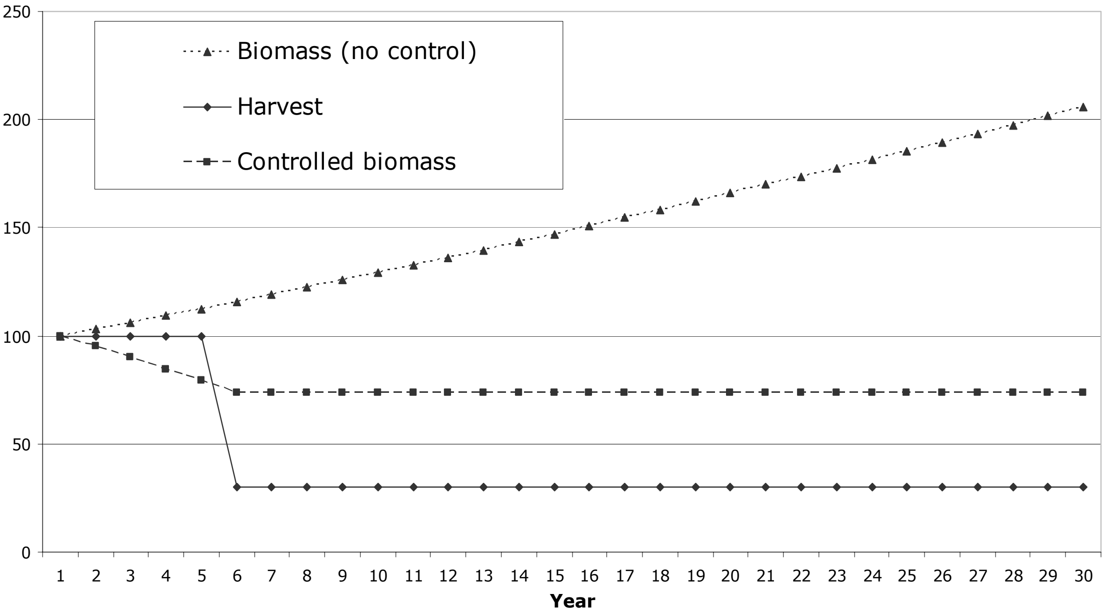
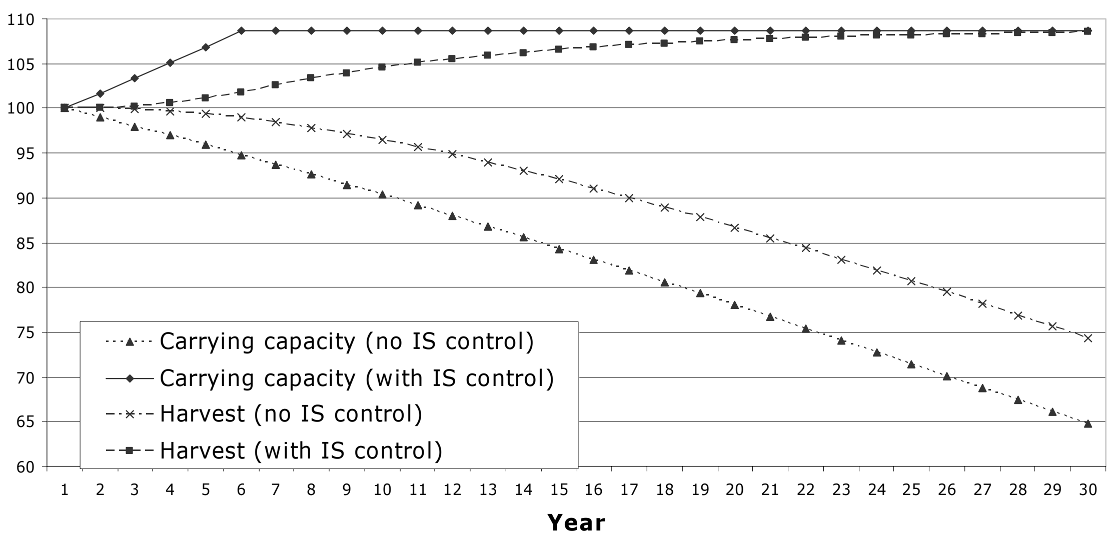


Figure 4. Impact of the Invasive Species Control Program on the Dynamics of the Invasive Species (*t*1 = 100)



Note: IS = invasive species.

Figure 5. Impact of the Invasive Species Control Program on the Native Species Fishery (*t*1 = 100)

these variables are respectively 66 percent and 44 percent higher with invasive species control than without control, and the gap continuously widens with time. Due to the "rollback" first step of the program, the stabilized values of scallop carrying capacity and catch are also higher than the immediate pre-program corresponding values. But, according to the simulation, the difference is less than 10 percent.

Economic Results of the Control Program

Figure 6 summarizes the economic results of the program, in annual (undiscounted) terms. The fishery annual gross margin, which decreases to zero in the *laissez-faire* scenario, would naturally benefit from the stabilization of environmental conditions generated by the program. The overall benefit of the program takes into account both its impact on the profitability of the scallop fishery and the slipper-limpet harvesting cost. This cost depends on the quantity harvested, but also on stock abundance and harvesting technical conditions (nature of the relation between CPUE and stock biomass). As in the first part of the paper, we consider two variants of the harvest function: proportionality between CPUE and biomass, and hyperdepletion.

Invasive species harvesting costs are higher in the "hyperdepletion" than in the "proportional" variant because the decrease in CPUE resulting from the lower abundance of slipper-limpet is more significant in the first case. Therefore, assuming hyperdepletion makes the annual overall benefit of the program less favorable than assuming proportionality between CPUE and stock biomass. But this difference is far from critical. In both variants, the overall annual benefit of the program is always positive, even during the "rollback" stage where slipper-limpet harvesting is high. This is due to the fact that slipper-limpet harvesting costs represent but a small proportion of the gross margin generated by the scallop fishery (always under 15 percent).

Compared to the *laissez-faire* scenario, both variants of the control scenario lead to a lower economic performance during the 4 first years, i.e., during the major part of the "rollback" stage of the program. However, the comparison favors the control scenario at the end of this stage. The superiority of control over *laissez-faire* increases steadily with time, due to the improving performance of the control scenario, but due even more to the continuously deteriorating economic performance of the fishery under the *laissez-faire* scenario.

The discounted present values of both scenarios are displayed in Table 5. According to the simulation, the control program generates a cumulated global present value which is greater by approximately 25 percent than the *laissez-faire* scenario, whatever the variant considered for the invasive species harvest function. Moreover, invasive species control allows sustainable fishing, while the *laissez-faire* scenario implies non-sustainability: because of declining carrying capacity of the invaded ecosystem for the native resource, native species fishing would stop after 96 years under this scenario. The length of this period is explained by two considerations: the intrinsic growth rate of the invasive stock which is assumed in the model to be relatively low, and the initial gross margin generated by scalloping, which is high. However, neither consideration should be regarded as providing solid ground for a "benign neglect" attitude towards the invasive process.

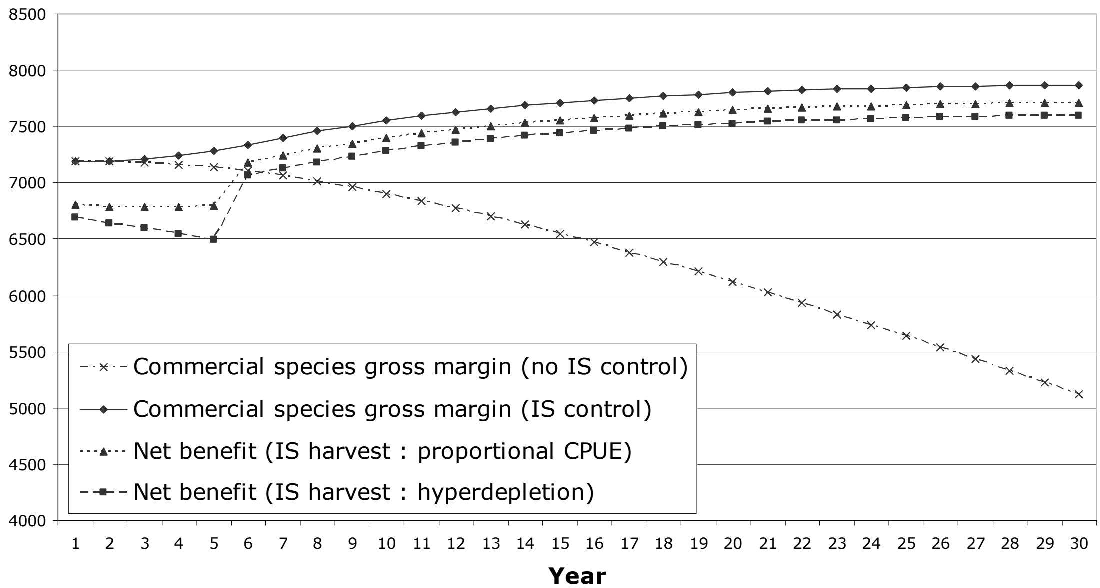
The Question of the Starting Date of the Program

In the described control scenario, the starting date of the program was set exogenously, and the only alternative considered was whether the program should be implemented or not. But slightly more sophisticated choices may be considered, including the question of the starting date of the program. Postponing this date is not neutral, for two distinct but interacting reasons: (i) the level of invasion which is to be addressed by the program is time-dependent, and (ii) the discounted present value of a set of cash flows depends on their time-pattern.

Basically, delaying the starting date of the program has opposite consequences on the profitability of the fishery and on the cost of the program.

It is clearly undesirable from the fishermen's point of view, both in the short run and in the long run: in the short run, postponing the control of the invasive process immediately deteriorates the environmental conditions of their activity, and in the long run, the level of the stabilized ecosystem carrying capacity for their target species is negatively influenced by the level of invasion reached at the starting date of the program. Moreover, delaying the benefits generated by the program decreases their discounted present value.

As regards the invasive species harvesting cost, it is also useful to distinguish short-run and long-run effects. In the short run, letting the invasion progress before the program starts results in higher CPUEs (especially in the case of hyperdepletion), and, as a consequence, reduces the cost



Notes: IS = invasive species, CPUE = catch per unit of effort.

Figure 6. Annual Economic Results of the Invasive Species Control Program (in 1,000 euros)

Table 5. Cumulated Time-Discounted Economic Results of Invasive Species Control and *Laissez-Faire* Scenarios (in million euros, infinite time horizon, time-discount rate 5 percent)

		Control scenario	
	Laissez-faire scenario	"Proportional" variant	"Hyperdepletion" variant
Native species fishery gross margin	116.9	152.4	152.4
Invasive species harvesting cost	0	4.3	6.9
Net benefit	**116.9**	**148.1**	**145.5**

of the "rollback" stage of the program. In the long run, the higher the stabilized biomass, the lower the corresponding permanent level of harvesting effort will be. Therefore, postponing the starting date of the program also reduces harvesting costs during the "containment" stage of the program, because it results in a higher biomass at the end of the former stage. Moreover, delaying the costs of the program decreases their discounted present value, *ceteris paribus*.

To illustrate these various effects, four scenarios have been simulated. The first one corresponds to the immediate starting of the program. In scenarios 2, 3, and 4, the starting date of the program is delayed by 5, 10, and 15 years, respectively.

According to simulation results, as the starting date of the program is delayed, the period of increasing invasive species biomass gets longer, and the stabilized level of controlled biomass gets higher. A higher level of invasive species biomass implies a lower level of carrying capacity for the native species, and, hence, lower catch and lower profits for a given level of fishing effort.

As delaying the starting date of the control program implies a higher level of invasive species biomass at the beginning of the "rollback" stage, it helps to decrease the cost of this stage because

CPUE is higher with stock abundance (particularly in the case of hyperdepletion). It also implies a higher level of invasive species biomass at the end of the "rollback" stage, and hence, as long as the MSY point is not reached, a higher permanent level of harvest to be realized during the "containment" stage. However, due to the increase in CPUE, this harvest is realized at a lower cost, particularly in the case of hyperdepletion. This permanent cost effect during the "containment" stage of the program adds up to the temporary one during the "rollback" stage. As a consequence, delaying the starting date of the program helps in reducing its cost.

Due to the difference of scale between the fishery gross margin and the operating cost of the control program, delaying the starting date of this program has negative long-term consequences on its annual net surplus: the positive impact on the invasive species harvesting cost is overbalanced by the negative impact on the fishery profitability. However, the consequences are not so simple during the transition period, where implementing the program involves extra costs. As a consequence, the ranking of the different scenarios on the basis of their economic performance may vary according to the time-discount rate, as illustrated by sensitivity tests.

Sensitivity Tests

Sensitivity tests were carried out on three parameters: the time discount rate, the invasive species growth rate, and the carrying capacity of the ecosystem for the invasive species.

As regards the first parameter, the results of the tests indicate that immediate implementation of the control program (scenario 1) is the best alternative as long as the time discount rate is less than 15 percent in the hyperdepletion case, and 16 percent in the proportionality case. When this level is reached, postponing the program by 5 years (scenario 2) is a better solution.[14]

Changing the value of the estimated present growth rate of the invasive stock from 1 percent to 5 percent (*ceteris paribus*) lowers the discounted benefit of the control program. But it lowers the discounted value of the *laissez-faire* scenario even more, and as a result, immediate implementation of the control program is still the best alternative.

Changing the carrying capacity of the ecosystem for slipper-limpets from 625,000 tons to 2.5 million tons (*ceteris paribus*) increases the discounted values of both control and *laissez-faire* scenarios, though at a higher rate for the second one. However, immediate implementation of the control program is still the best alternative with a 5 percent time-discount rate.

Conclusions

Making use of a simple two-species bioeconomic model, this paper has surveyed the economic consequences of the biological invasion of a commercial fishery by an exotic species acting as a space competitor for the native targeted species. For each stock, the model relies on the classical Schaefer assumptions, with a variant for the invasive stock, allowing for so-called "hyperdepletion" in the harvest function. The dynamics of the native stock is linked to that of the invasive stock by an asymmetric competition relation, where the carrying capacity of the ecosystem for the native species is a decreasing function of the level of invasion of the fishery by the invasive species. The equilibrium study performed with this model has underscored the importance of assuming hyperdepletion in the invasive species harvest function. Contrasted with the standard assumption of proportionality between CPUE and stock abundance, the hyperdepletion assumption makes it possible to model the irreversible character of the invasion. In the hyperdepletion case, an "optimal rate of invasion" may be found, differing from the "all or none" control that characterizes the proportionality assumption. The dynamic simulations realized with the help of the model have emphasized the importance of time when assessing the results of a control program.

The model was qualitatively based on a case study of the Bay of St-Brieuc common scallop fishery, facing biological invasion by slipper-lim-

[14] In the case (corresponding to the actual situation) where the "rollback" stage of the program has been implemented, the question of the starting date of the program is replaced by the question of the starting date of its containment stage. In this case, sensitivity tests indicate that it is worthwhile postponing the containment stage of the program during 5 years if the time discount rate reaches 15 percent (hyperdepletion) or 16 percent (proportionality).

pet. Its numerical illustration relies on the available biological and economic knowledge of this case. However, this model faces several important limits. Some are due to the deficiency of biological knowledge, especially in the field of slipper-limpet dynamics. Other limits are due to the modeling assumptions. In particular, describing the common scallop dynamics by means of a global deterministic model is open to criticism, due to the high variability of recruitment of this species. Considering prices and native species fishing effort (in the dynamic simulations) as exogenous variables is certainly another significant limitation of the model.

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