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Adrien Lagarde

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Économie Écologique pour une Soutenabilité et Résilience Écosystémique des Pêches face aux Changements Globaux

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Ecole Doctorale Entreprise, Économie, Société (N°42)

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Dirigée par Luc DOYEN et Olivier THÉBAUD

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Résumé en français: Économie Écologique pour une Soutenabilité et Résilience Écosystémique des Pêches face aux Changements Globaux

Mots-clefs: Économie, Écologie, Soutenabilité, Résilience, Écosystémique, Pêches, Changement Climatique

Atteindre les différents objectifs du développement soutenable, incluant notamment la protection des océans, la conservation de la biodiversité marine et une utilisation soutenable des ressources halieutiques constitue un défi pour les pêcheries industrielles et artisanales du monde entier. Les changements globaux, qui affectent les dynamiques écologico-économiques des écosystèmes marins et côtiers, compliquent significativement l'atteinte de ces objectifs. A travers la prise en compte de critères écologiques, économiques et sociaux liés respectivement au maintien de la biodiversité halieutique, de la viabilité économique des pêcheries et de la sécurité alimentaire, cette thèse a pour objectif d'identifier des stratégies de gestion de pêche permettant d'assurer la soutenabilité et la résilience des pêcheries maritimes.

La recherche s'appuie sur deux études de cas: les pêcheries industrielles du Golfe de Gascogne et les petites pêcheries côtières coralliennes de Polynésie Française. Plusieurs modèles écologico-économiques sont développés et calibrés. Dynamiques, spatialement explicites, multi-flottes, multi-espèces et stochastiques, ces modèles prennent en compte la complexité des interactions propres aux socio-écosystèmes étudiés, tout en s'inscrivant dans la famille des modèles de complexité intermédiaire pour une gestion écosystémique des pêches. Les paramètres des modèles sont calibrés en utilisant à la fois des données locales propres aux deux études de cas, et des données climatiques. Le choix de l'effort de pêche comme variable de contrôle du système permet de prendre en compte la relative flexibilité et la diversité d'outils de régulation propres aux systèmes de gestion étudiés. Une attention particulière est portée aux stratégies de pêche multi-espèces que sont le Multi-Species Maximum Economic Yield (MMEY), le Multi-Species Maximum Sustainable Yield (MMSY) ou l'éco-viabilité. Les stratégies, scénarios et projections dérivés des modèles sont comparés en utilisant des critères écologico-économiques de soutenabilité, de résilience et de viabilité tels que le maintien dans le temps des profits de l'activité de pêche, le maintien de la production alimentaire dont dépendent les populations insulaires et côtières et de la conservation des fonctions écologiques et de la biodiversité.

Au delà des modèles, des stratégies de pêche et des scénarios construits pour les 2 études de cas, cette thèse présente une série de résultats transversaux importants: en premier lieu, l'analyse met en exergue la nécessité d'une redistribution temporelle, spatiale et par espèce des efforts de pêche pour la soutenabilité et la résilience des pêcheries maritimes. Plus concrètement et précisément, la recherche montre qu'une stratégie de diversification temporelle, spatiale, de métiers de pêche et d'espèces ciblées peut favoriser la soutenabilité et la résilience de tout le système pêche face aux incertitudes et à la complexité. Les stratégies stochastiques MMEY, MMSY et d'éco-viabilité apparaissent particulièrement pertinentes dans cette perspective. Ainsi une meilleure flexibilité des flottes de pêches (dépendant de la flexibilité des instruments de régulation), qui serait fonction des incertitudes globales et du contexte écologico-économique, soutiendrait la viabilité et la résilience des pêcheries, artisanales comme industrielles. D'une manière générale, ces

différents résultats mettent en évidence l'importance d'inscrire l'approche écosystémique dans une perspective écologico-économique et la nécessité de planifier le développement des pêcheries par une gestion multi-critère, adaptative, résiliente afin de minimiser les risques futurs.

Résumé en anglais: Ecological Economics for an Ecosystem Sustainability and Resilience of Fishing facing Global Changes

Keywords: *Economics, Ecology, Sustainability, Resilience, Ecosystem, Fishing, Climate Change*

Achieving the various objectives of sustainable development, including notably the conservation of the oceans and a sustainable use of fishery resources constitutes a challenge for industrial and artisanal fisheries around the world. Global changes, which affect the ecological and economic dynamics of marine and coastal ecosystems, significantly complicate the achievement of these objectives. By taking into account ecological, economic and social criteria linked respectively to the maintenance of fishery biodiversity, the economic viability of fisheries and food security, this thesis aims at identifying fishing management strategies allowing to ensure the sustainability and resilience of marine fisheries.

The research is based on two case studies: the industrial fisheries of the Bay of Biscay and the small coastal coral fisheries of French Polynesia. Several ecological-economic models are developed and calibrated. Dynamic, spatially explicit, multi-fleet, multi-species and stochastic, these models take into account the complexity of the interactions specific to the socio-ecosystems studied, while being part of the family of models of intermediate complexity for ecosystem-based fisheries management. The model parameters are calibrated using both local data relating to the two case studies, and climatic data. The choice of fishing effort as a system control variable takes into account the relative flexibility and diversity of regulatory tools specific to the management systems studied. Particular attention is paid to the multi-species fishing strategies that are the Multi-Species Maximum Economic Yield (MMEY), the Multi-Species Maximum Sustainable Yield (MMSY) or Eco-viability. The strategies, scenarios and projections derived from the models are compared using ecological and economic criteria of sustainability, resilience and viability such as maintaining the profits of the fishing activity over time, maintaining the food production on which island and coastal populations depend and the conservation of ecological functions and biodiversity.

Beyond the models, fishing strategies and scenarios constructed for the 2 case studies, this thesis presents a series of important transversal results: first, the analysis highlights the need for temporal, spatial, by species and by type of fishing reallocation of fishing efforts for the sustainability and resilience of marine fisheries. More concretely and precisely, this research shows that a strategy of temporal, spatial, fleet (type of fishing) and targeted species diversification, can promote the sustainability and resilience of the whole fishing system in the face of uncertainties and complexity. The MMEY, MMSY and Eco-viability stochastic strategies appear particularly relevant in this perspective. Thus a better flexibility of fishing fleets, which would depend on global uncertainties and would be function of the ecological and economic context, would support the viability and resilience of fisheries, both artisanal and industrial. In general, these different results highlight the importance of embedding the ecosystem approach within an ecological-economic perspective and the need to plan the development of fisheries through a multi-criteria, adaptive and resilient management in order to minimize future risks.

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Production Scientifique

a. Articles de recherche

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Article en préparation

Lagarde, A., Doyen, L., Claudet, J., Thébaud, O., 2020. "To what extent fishing viability strategies promote ecological-economic resilience of coral reef facing climate change".

b. Participation à des Conférences

World Conference on Natural Resource Modelling (Barcelone, Espagne, 6-9 Juin 2017)

- How Does MMEY Mitigate the Bioeconomic Effects of Climate Change for Mixed Fisheries (Présentation orale)

The 10th Indo-Pacific Fish Conference (Tahiti, Polynésie Française, 2-6 Octobre 2017)

- Ecological-economic resilience of a fished coral reef through Stochastic MMSY (Présentation orale)

1st Green-Econ Spring School in Environmental Economics (Marseille, France, 22-25 Mai 2018)

- Ecological-economic resilience of a fished coral reef through Stochastic MMSY (Présentation orale)

BIOECON XX: The 20th Annual BioEcon Conference (Cambridge, Angleterre, 12-14 Septembre 2018)

- How Does MMEY Mitigate the Bioeconomic Effects of Climate Change for Mixed Fisheries (Présentation orale)

World Conference on Natural Resource Modelling (Montréal, Canada, 22-24 Mai 2019)

- Ecological-economic resilience of a fished coral reef through Stochastic MMSY (Présentation orale)

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Pour obtenir les algorithmes/codes et fichiers de données:

Trouvable sur le *Google Drive* de l'adresse mail : these.algorithm.al@gmail.com. Mot de passe : **eco-viability**.

Étymologie et Philosophie

Il est communément admis par la grande majorité de la population que l'économie et l'écologie sont deux termes antinomiques. Cependant, si nous observons les racines grecques, nous nous apercevons qu'il n'en est rien. En effet, le terme *économie* provient de "*oikos*" qui signifie la maison et "*nomos*", la gestion ; l'économie n'est alors qu'un mot représentant initialement "*la gestion de la maison*", donc la gestion d'une ville, d'une région, d'un pays ou de la planète plus globalement. Le mot *écologie* contient également le préfixe éco mais fait également référence, avec son suffixe *logie* ("*logos*" en grec), au "*discours*" à la "*raison*", à la "*parole*" ou à la "*logique*" et par extension, depuis Platon et Aristote à l'intelligence. On pourrait l'interpréter de plusieurs manières, mais Haeckel (1866) lui attribua une définition plus moderne qui dès lors statuera sur son sens actuel: "la science des relations des organismes avec le monde environnant, c'est-à-dire, dans un sens large, la science des conditions d'existence" en d'autres termes "*la science de l'habitat*" ou pour faire le parallèle avec l'économie "*la science de la maison*". Si nous prenons alors les deux ancestrales définitions de l'économie et de l'écologie, il apparaît évident que ces deux mots sont liés non pas pour être mis en opposition mais au contraire pour se compléter. Tandis que l'écologie, nous explique comment notre maison (la Terre) fonctionne, l'économie, quant à elle, nous apprend à la gérer. Par conséquent, les deux termes semblent indissociables. L'économie de l'environnement, l'économie écologique ou *éconologie*¹ bien qu'étant une discipline récente, peu connue et peu comprise du grand public et des libéraux néoclassiques, s'affiche dès lors comme un domaine scientifique essentiel pour l'avenir, conditionnant un probable et possible développement durable.

¹Terme ou mot-valise né vers la fin du XXème siècle afin de réconcilier ces deux entités que sont l'écologie et l'économie, bien que leur contraction serait plus précise étymologiquement en parlant d'*économologie*.

Table des Matières

Étymologie et Philosophie	11
I Introduction Générale	17
1 Cadre et Enjeux	18
1.1 La surexploitation de la biodiversité et des écosystèmes marins	18
1.2 La gestion classique des pêches	22
1.3 L'approche écosystémique: Une nouvelle génération de défis	26
1.4 Soutenabilité, viabilité et résilience	29
2 Méthodes et Modélisation écosystémiques	34
2.1 Modèles écosystémiques	34
2.2 Indicateurs écologico-économiques	34
2.3 Seuils de viabilité	36
2.4 Stratégies exploratoires de pêche	36
2.5 Stratégies normatives de pêche	38
2.6 Questions de recherche autour de l'EBFM	39
3 Synthèse des travaux	41
3.1 La stratégie multi-espèces MEY (MMEY) atténue les effets du changement climatique dans les pêcheries mixtes	41
3.2 La résilience écologico-économique d'un récif corallien exploité favorisé par une stratégie de pêche stochastique	44
3.3 Les stratégies de viabilité de pêche pour une résilience écologico-économique des récifs coralliens face au changement climatique	48
4 Discussion Générale	51
4.1 L'intérêt des stratégies de gestion multi-espèces pour l'EBFM	51
4.2 L'intérêt des MICE écologico-économiques pour l'EBFM	52
4.3 La gestion de l'incertitude pour l'EBFM	53
4.4 La diversification écosystémique pour l'EBFM	54
4.5 Perspectives	56
Références	56
II How does MMEY mitigate bioeconomic effects of climate change in mixed fisheries	73
1 Introduction	75
2 Bay of Biscay case study	77
2.1 Sole	77
2.2 Hake	77

2.3	Economical role of hake and sole fisheries	77
2.4	Data sources	79
3	The bio-economic model	80
3.1	Multi-species age-class dynamic model	80
3.2	Stock-recruitment dynamics	80
3.3	Economic scores	81
3.4	Management strategies	82
3.5	Climate scenarios	83
4	Results	85
4.1	Impact of warming on stock-recruitment model	85
4.2	Status-quo : not ecologically and economically viable	86
4.3	MMSY : not ecologically viable but economically viable	88
4.4	MMEY : ecologically viable and economically viable	89
4.5	MMSY, MMEY : reduction of efforts especially for sole gill-netters	90
4.6	Bioeconomic synthesis displays the MMEY as the best strategy	91
5	Discussion	92
5.1	MMEY as an ecological and economic win-win strategy	92
5.2	Diversification of fleets produces greater benefits in the face of climate change	92
5.3	Perspectives	93
	References	94

III Ecological-economic resilience of a fished coral reef through Stochastic MMSY 117

1	Introduction	119
2	The Moorea Island small-scale fishery	120
3	The ecological-economic model	122
3.1	Conceptual model	122
3.2	Mathematical model	122
3.3	Calibration	124
3.4	Fishing strategies	126
4	Results	127
4.1	Closure (CL): Limited ecological gains and not socially sustainable	127
4.2	Closure on lagoon and + 100% on reef (2R): Ecologically limited and not socially sustainable	128
4.3	Closure on fore reef and increase of 100% on lagoon (2La): Ecologically and socially sustainable	128
4.4	Status-Quo (SQ): Ecologically and socially sustainable	129
4.5	SMMSY: Ecologically and socially sustainable	129
5	Discussion	132
5.1	MICE modelling to operationalize ecosystem based fishery management	132
5.2	SMMSY to promote ecological-economic resilience	133
5.3	Fishing as a driver of resilience	134
5.4	Diffusion and spatial management	134

5.5	Policy recommendations for Moorea Island	135
References		136
IV	To what extent fishing viability strategies promote ecological-economic resilience of coral reef facing climate change	155
1	Introduction	157
2	The Moorea Island small-scale fishery	159
3	The ecological-economic model	161
3.1	Conceptual model	161
3.2	Mathematical model	161
3.3	Calibration	163
3.4	Viability assessment	164
3.5	Fishing strategies	167
4	Results of shock scenarios	169
4.1	$p = 10\%$: A viable and resilient ecosystem	169
4.2	$p = 20\%$: A viable and resilient ecosystem	171
4.3	$p = 30\%$: A viable and less resilient ecosystem	173
4.4	$p = 50\%$: A viable and non-resilient ecosystem	175
5	Discussion	179
5.1	PVA and EcoVA: the most viable ecological-economic strategies	179
5.2	Viability \neq Resilience	180
5.3	Is a viable coral reef, still resilient under a gradual increase of shocks ?	180
5.4	Eco-viability: a toolbox for resilience assessment and management	181
References		182
Liste des Figures		200
Liste des Tables		204

Chapitre I
Introduction Générale

1. Cadre et Enjeux

1.1 La surexploitation de la biodiversité et des écosystèmes marins

Dans la lignée des travaux de Malthus (1852), Meadows et al. (1972) se posent la question de l'existence d'une croissance infinie au sein d'un monde aux ressources épuisables. Brundtland et al. (1987) tentent alors d'y répondre en proposant un nouveau modèle de société, un développement durable ("*sustainable development*") conditionnant *Notre avenir à tous*. Ce concept vise une équité inter-générationnelle et se définit par un développement reposant sur un équilibre entre ses trois piliers: économique, social et écologique. Quelques décennies plus tôt, la nature même des écosystèmes halieutiques généra les premières réflexions et conclusions de soutenabilité. Précurseuses pour l'économie écologique, ces idées aboutirent à l'élaboration du concept de Rendement Maximum Durable¹ (*Maximum Sustainable Yield* - MSY) introduit par Russell (1931) puis popularisé, grace à son estimation, par Schaefer (1954) et appliqué en tant qu'objectif de gestion au début des années 1950 aux Etats-Unis.

1.1.1 Surexploitation des stocks marins

Avec l'industrialisation et le développement technologique des deux derniers siècles ainsi que l'accroissement de la demande de produits halieutiques lié en partie à la croissance démographique, les captures des pêches maritimes à l'échelle internationale n'ont cessé d'augmenter. Les premiers relevés de 1950 nous permettent de rétroactivement observer une constante augmentation jusqu'aux années 1990 où une stagnation se fait alors ressentir depuis (Pauly and Zeller, 2016 - Figure I.1). Un seuil critique commence à être atteint dans beaucoup de pêcheries (FAO, 2018). Néanmoins, "en 2017, les stocks exploités au niveau durable maximal représentaient 59,6 % des stocks évalués, contre 6,2 % pour les stocks sous-exploités, une augmentation depuis 1989, reflétant en partie une meilleure mise en œuvre des mesures de gestion" (FAO, 2020).

Plus précisément, une classification menée par le projet Sea Around Us², considère qu'une pêcherie est "en développement" si les captures sont inférieures à 50% du maximum historique des captures ; si les captures sont supérieures à 50%, la pêcherie est dite "exploitée" ; sinon si les captures sont comprises entre 10 et 50% du maximum elle est considérée comme "surexploitée"

¹Le Rendement Maximum Durable est "la plus grande quantité de captures que l'on peut extraire d'un stock halieutique à long terme et en moyenne, dans les conditions environnementales existantes (moyennes), sans affecter significativement le processus de reproduction" (Définition donnée par la FAO - <http://www.fao.org/home/en/>)

²<http://www.seaaroundus.org/>

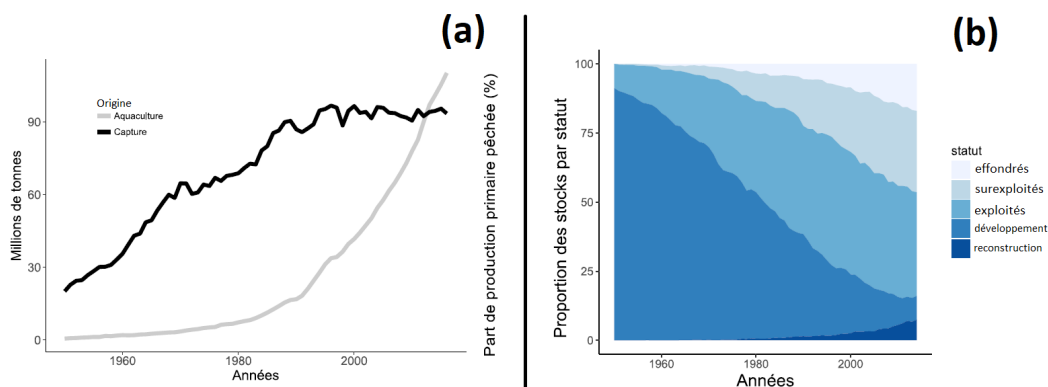


Figure I.1: Suivi des pêcheries mondiales: (a) Quantités capturées par les pêcheries marines (en noir) et produites par l'aquaculture (en gris) ; (b) Pourcentage des stocks selon leur statut d'exploitation. Sources: FAO (a) et Sea Around Us Project (b).

; et la pêche est dite “effondrée”, si elles sont inférieures à 10% du maximum. A la suite de l’effondrement, si les captures augmentent pour revenir vers des niveaux entre 10 et 50% du maximum, la pêche est en “reconstruction”. Depuis les années 1960, la part de stocks surexploités (ou effondrés) était en constante augmentation (Pauly et al., 1998). Deux décennies plus tard, grâce à la mise en place de programmes économiques durables et équitables dotés de considérations écologiques, le nombre de stocks “en reconstruction” commencent alors à augmenter (Kleisner et al., 2013; FAO, 2020). La proportion de ces stocks est montrée sur la Figure I.1 (b).

1.1.2 La biodiversité et les écosystèmes sous pression

Ainsi, depuis 1950, au delà de la surexploitation des différents stocks halieutiques pris séparément et face à la continuelle augmentation de l’effort de pêche partout à travers le monde, c’est la structure des écosystèmes et de la biodiversité qui a changé. En particulier, au sein des grandes pêcheries, l’effort de pêche est tellement fort que Pauly et al. (1998) observe un “fishing down effect” (Figure I.2). Cet effet se définit par un impact de l’industrie de la pêche capturant les grands prédateurs en premier puis descendant la chaîne alimentaire au fur et à mesure que l’abondance des plus gros groupes fonctionnels s’épuise jusqu’à ne capturer que des moyennes puis petites espèces. A terme, l’écosystème, en plus de perdre sa diversité fonctionnelle, se voit caractérisé par un niveau trophique moyen de plus en plus faible générant des réactions en chaîne au sein du réseau trophique, appelées *cascades trophiques*. Ce phénomène peut avoir des effets bénéfiques à court terme (Doyen et al., 2018, Chap. III) mais la plupart du temps néfastes à long terme (Myers et al., 2007; Casini et al., 2008).

Cependant, le niveau trophique moyen des captures semble en apparence observer quelques récentes augmentations (Figure I.3). Cela s’explique notamment par une plus forte expansion spatiale des pêcheurs vers des zones plus faiblement exploitées historiquement (Kleisner and Pauly, 2011)³. Il est également nécessaire de préciser que sur le long terme, les changements et impacts anthropiques sur la structure de l’écosystème ne sont pas encore totalement connus et compris. Par ailleurs, il est à noter que le niveau trophique moyen de l’écosystème n’est pas équivalent au niveau trophique moyen des captures⁴ (Branch et al., 2010). Par conséquent, le manque d’informations sur le véritable niveau trophique de l’écosystème nous incite à nuancer de tels résultats.

Plus globalement, la résilience et soutenabilité d’un écosystème dépend majoritairement de sa diversité fonctionnelle. En effet, certaines espèces dites spécialistes sont indispensables au bon fonctionnement de l’écosystème. Si certaines d’entre elles viennent à disparaître, et qu’aucune redondance fonctionnelle n’est observée (i.e aucune autre espèce ne peut reprendre son rôle dans la chaîne trophique), l’écosystème peut alors s’effondrer (Clavel et al., 2011). L’exemple de la surpêche des petits herbivores dans les Caraïbes empêchant la fonction de broutage de ce groupe fonctionnelle et transformant ainsi la zone à dominance corallienne en une zone colonisée par les algues illustre ce type de processus (Mumby, 2009).

1.1.3 Causes socio-économiques

Les raisons de cette surexploitation sont multiples: une croissance démographique trop importante et par extension une demande en ressources halieutiques qui ne cessent de croître ; des développements technologiques liés aux méthodes d’extractions et de captures qui favorisent une amplification des impacts anthropiques et une augmentation des rendements à court terme (Pauly et al., 1998; FAO, 2018). Gordon (1954) précise ces phénomènes comme étant liés aux

³Néanmoins, un nouvel indicateur nommé Index de Soutenabilité des Pêcheries (Fisheries Sustainability Index - FSI) nous montre que le processus de *cascades trophiques* continue malheureusement à se dérouler à travers les écosystèmes mondiaux (Figure I.3 - Kleisner and Pauly, 2011).

⁴Le niveau trophique moyen des captures compte parmi les huit indicateurs attestant de la santé d’un écosystème. Il est nommé Marine Trophic Index (MTI - Section 2.2).

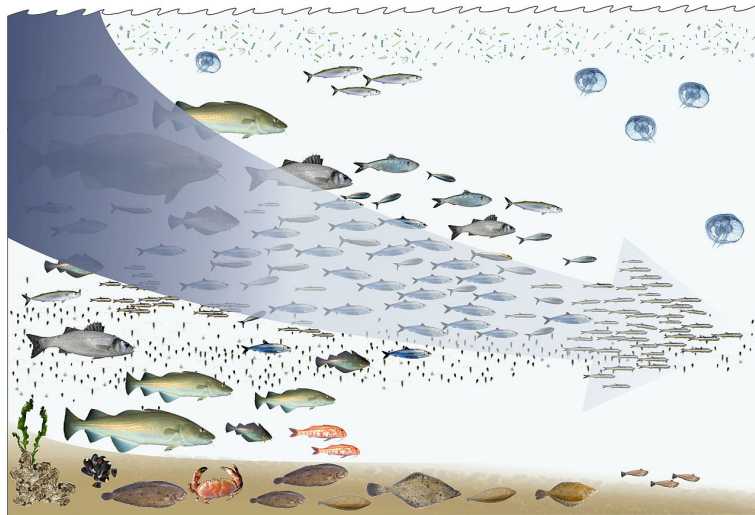


Figure I.2: Représentation schématique du “Fishing Down Effect” décrit par Pauly et al. (1998).

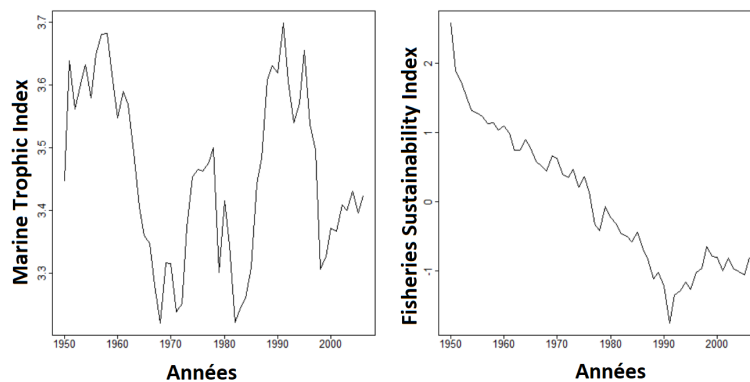


Figure I.3: Tendances de l’indice trophique marin (Marine Trophic Index - Gauche) et de l’indice de soutenabilité des pêcheries (Fisheries Sustainability Index - Droite) (Kleisner and Pauly, 2011).

conséquences de l’exploitation d’une ressource commune⁵ par des agents économiques poursuivant chacun un objectif de maximisation d’utilité (rationalité de l’homo-economicus). Supposons une pêche⁶ donnée, considérée “en développement” ou “exploitée”. La zone étant économiquement avantageuse comparativement aux autres pêcheries surexploitées ou effondrées, le nombre de pêcheurs tendra à augmenter. Ce nombre va continuer de croître jusqu’à ce que la rentabilité de la pêche soit réduite aux niveaux des autres ou qu’elle affiche un profit nul. Cela s’explique par des revenus marginaux (par unité d’effort) qui augmentent avec l’effort de pêche. Comme le souligne également Hardin (1968) dans sa “tragédie des communs”, la nature de bien en accès libre d’une pêche a pour principale conséquence: sa surexploitation économique et parfois écologique ; cela engendre bien souvent un effondrement des stocks, qui se veut “optimal” économiquement (pour chaque acteur de la pêche a priori) mais qui, à long terme, (et/ou pour le collectif) ne l’est plus.

Les politiques publiques notamment par le biais d’incitations économiques (subventions) ou de

⁵C’est la nature commune (non exclusive mais rivale d’utilisation) de la ressource halieutique qui est à l’origine du problème.

⁶“Une pêche est une entité de gestion d’une capacité de pêche circonscrite à une zone géographique donnée, où opèrent différents métiers qui capturent des espèces occupant des habitats de caractéristiques semblables.” (IFREMER - <https://wwz.ifremer.fr/peche/Glossaire/Glossaire/Peche>)

mesures restrictives (taxes, quotas...) peuvent également être responsables d'une aggravation de la situation (OECD, 2017). Le manque de moyen pour limiter la pêche IUU (*Illegal, Unreported and Unregulated* - illégale, non-reportée et non-régulée)⁷ et le pillage des ressources des pays du "Sud" via des accords commerciaux déloyaux (Grynberg, 2003), sont autant de causes créant ou exacerbant les problèmes actuels. Plus globalement, on trouve des politiques publiques intégrant la notion de durabilité mais dont les moyens mis en oeuvre sont insuffisants dans sa prise en compte, des politiques publiques qui n'en tiennent pas compte, et d'autres qui n'utilisent cette notion que ponctuellement.

Par ailleurs, les agents économiques qui ont une préférence pour le présent peuvent à terme entraîner l'extinction du stock (Clark, 1973). De plus, dans certains cas, même si la ressource est gérée par un propriétaire unique (échappant ainsi à la tragédie des communs), il peut être dans l'intérêt de ce propriétaire de l'extraire dans son intégralité, plutôt que de l'exploiter durablement, du fait de sa vitesse de renouvellement trop lente (et de la préférence pour le présent de l'agent - Clark, 1973). Tout ce raisonnement s'explique par une mauvaise gestion temporelle de l'effort de pêche. En effet, la surpêche génère des pertes de revenus **futurs** (à travers une diminution du stock fécond notamment) qui pèsent moins que les bénéfices présents liés à l'extraction immédiate de la ressource (long/moyen terme) (Clark, 2010). Le profil des espèces et la répartition des niveaux trophiques peuvent également s'en voir affectés.

La variabilité environnementale générée par le changement climatique est également un facteur qui modifie la structure des écosystèmes et qui impacte négativement les performances socio-économiques. De plus, les systèmes de gouvernance locaux n'ont que très peu de recours pour faire face à ce phénomène global.

1.1.4 Changement climatique et incertitude globale

Alors que l'humanité fait déjà face aux conséquences de l'impact anthropique local (pollution, étalement urbain, effondrement de la biodiversité, etc...), le changement climatique global et toutes ses implications et effets nous guident encore plus vers un futur incertain (Lam et al., 2016). Les conséquences locales sont nombreuses. Les petites pêcheries dont font partie les récifs coralliens subissent différentes perturbations localisées pour la plupart imputables au changement climatique (espèces invasives, cyclones, blanchissements, perte de nutriments...) qui semblent augmenter en intensité et en occurrence depuis plusieurs décennies (Cinner et al., 2013; Diop et al., 2015; Cheung and Oyinlola, 2018). Présentant des caractéristiques différentes, les grandes pêcheries partagent quelques similarités (espèces invasives, augmentation de la température de l'eau...) mais sont également affectées par des mouvements de population de poissons vers les pôles notamment (Poloczanska et al., 2016) ainsi que par une pollution (e.g micro- et macro-plastiques...) généralement plus présente due à une densité humaine plus importante dans les zones en question.

En réponse à ces enjeux, l'UNEP (*United Nations Environment Program* - Programme des Nations Unies pour l'Environnement) et sa nouvelle plateforme l'IPBES⁸ (*Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* - Plateforme intergouvernementale scientifique et politique sur la biodiversité et les services écosystémiques) à l'instar de la FAO⁹ tentent d'agir globalement pour atténuer les effets des changements globaux et s'affichent en tant qu'assistant dans le développement des politiques environnementales de nombreux pays. Localement, notamment pour la Nouvelle-Aquitaine, l'association Acclimaterra¹⁰ tente par exemple d'apporter des réponses scientifiques aux enjeux climatiques locaux et globaux.

⁷<http://www.fao.org/3/ca6916en/ca6916en.pdf>

⁸<https://ipbes.net/>

⁹<http://www.fao.org/home/en/>

¹⁰<http://www.acclimaterra.fr/qui-sommes-nous/>

1.1.5 Services écosystémiques

L'un des principes de l'économie de l'environnement est d'évaluer ce que produit la nature et ce qu'elle peut nous apporter i.e les services rendus par la nature nommés services écosystémiques. Ces services peuvent être *de provision* (e.g productivité des pêches, des forêts...), *de régulation* (e.g espèces détritatives, purification des eaux...), de *support* (e.g cycle de l'eau, formation des sols, production primaire) et *culturels* (e.g tourisme) (Millenium Ecosystem Assessment¹¹ - MAE). Ils peuvent également être néfastes (e.g requins vs baigneurs, ours vs berger...).

Mais l'un des défis de ce début de siècle est d'arriver à attribuer une valeur ou donner un poids économique à ces services afin de comparer les coûts et bénéfices et/ou avantages d'une protection/conservation de la nature face à son exploitation ; en d'autres termes, l'analyse économique pose la question des choix à prendre en matière d'utilisation des écosystèmes. Cette valeur économique peut être évidente pour certaines ressources matérielles dont il existe déjà un marché et des prix sur lesquelles s'appuyer (marché du poisson, de bois, du gibier...). Mais cela demeure complexe, d'une part car il est difficile de réellement comptabiliser la totalité des services d'un écosystème (e.g une forêt produit du bois, mais également de l'oxygène, abrite de la biodiversité, permet le tourisme, etc...) et d'autre part car certains biens non-matériels, qui sont souvent des biens communs (e.g oxygène), ne disposent pas de marché ; nous remarquons également que de nombreuses dimensions du bien-être ne peuvent être captées par des prix marchands ou dérivés de marchés simulés. Cette remarque est conditionnée par l'existence d'une véritable corrélation entre bien-être (ou bonheur) et développement économique. L'un des moyens les plus utilisés actuellement bien que controversé pour donner une valeur économique aux services non monétarisés, vise à "déclarer" ou identifier le consentement à payer (*willingness to pay*) des individus pour le service rendu par la nature (De Groot et al., 2002). Le prix des services écosystémiques mondiaux a été ainsi estimé à 33 billions de dollars, dont 63% est dû aux écosystèmes marins (De Groot et al., 2012), mais ces évaluations ont leurs limites (Pendleton et al., 2016).

Bien évidemment, il faut souligner que donner une valeur économique à un service naturel, ne signifie pas que ce dernier peut être remplacé par un service moins cher et plus efficient (Silvertown, 2015). Au contraire, certains services ont une valeur intrinsèque et sont bien souvent non-susbtituables (Zimmerman, 2001; McCauley, 2006).

1.2 La gestion classique des pêches

L'approche mono-spécifique demeure le modèle de gestion le plus utilisé à travers le monde actuellement (MSY via une Politique Commune des Pêches - European Union (2013) ; MSY aux USA - NOAA (2007), MEY en Australie¹²).

Se basant sur les travaux des écologues afin de définir la croissance des populations, Schaefer (1954) définit pour la première fois un modèle de l'évolution des pêcheries tel que:

$$x(t + 1) = x(t) + G(x(t)) - H(t) \quad (\text{I.1})$$

où $x(t)$ représente la biomasse du stock à l'instant t , $G(x)$ est une fonction de croissance biologique de population et $H(t)$ correspond aux captures de pêche à l'instant t (Conrad et al., 1987; De Lara and Doyen, 2008). La dynamique de croissance biologique souvent présentée dans la littérature¹³ est la croissance logistique définie par:

$$G(x) = rx \left(1 - \frac{x}{K}\right) \quad (\text{I.2})$$

¹¹<http://www.millenniumassessment.org/>

¹²<https://www.afma.gov.au/about/economic-role>

¹³Aujourd'hui, en pratique, la recherche liée à la gestion des pêches, retient plutôt les modèles de populations structurées en âge ou en taille.

avec r est son taux de croissance intrinsèque, K la capacité de charge du milieu (i.e l'abondance maximum pouvant être observée dans le milieu étudié).

La partie exploitation (captures) peut évoluer linéairement par rapport à la biomasse et l'effort comme dans Schaefer (1954) tel que:

$$H(x(t), e(t)) = qx(t)e(t) \quad (\text{I.3})$$

avec q qui est égal à la capturabilité du stock (i.e la probabilité pour la flottille de rencontrer et capturer une unité de biomasse par unité d'effort) et $e(t)$ l'effort de pêche.

Les prélèvements équilibrés (i.e le taux de prélèvement est égal au taux de croissance naturel tel que $H(x) = G(x)$) sont définis par:

$$H(x) = rx \left(1 - \frac{x}{K}\right) \quad (\text{I.4})$$

Si l'on remplace l'expression de x donnée par l'éq. (I.3) dans l'équation précédente ($x = \frac{H}{qe}$), nous obtenons la formulation du prélèvement équilibré par rapport à l'effort:

$$H(e) = qKe \left(1 - \frac{qe}{r}\right) \quad (\text{I.5})$$

Par conséquent, lorsque $H(e) > G(x)$, le stock biologique décline, et à l'inverse quand $H(e) < G(x)$, la biomasse augmente.

Nous pouvons ensuite définir une fonction de profits π qui dépend de la biomasse x et de l'effort e telle que:

$$\pi(x, e) = pH(x, e) - C(e) = pqxe - C(e) \quad (\text{I.6})$$

avec p le prix d'une unité de biomasse, $C(e)$ une fonction de coût dépendant de l'effort de pêche e . Par souci de simplicité, nous considérons que les coûts évoluent linéairement avec l'effort tel que $C(e) = ce$ avec c un coût unitaire constant.

Si nous prenons la formule du prélèvement équilibré (I.5), et que nous la remplaçons dans l'équation précédente, on obtient les profits équilibrés:

$$\pi(e) = pH(e) - C(e) = pqKe \left(1 - \frac{qe}{r}\right) - ce \quad (\text{I.7})$$

Maximum Sustainable Yield (MSY)

Afin de conserver une croissance naturelle du stock maximal, la plupart des modèles de gestion vise une production maximale équilibrée (*Maximum Sustainable Yield* - MSY). Cet objectif est devenu le point de référence en matière de gestion des pêches (Finley, 2009). Mathématiquement, il résulte d'une maximisation des captures à l'équilibre tel que:

$$\max_e H(e) \quad (\text{I.8})$$

En utilisant les conditions d'optimalité du premier ordre et en égalisant à zéro la dérivée première des prélèvements à l'équilibre (éq. (I.5)) par rapport à e , on peut ensuite obtenir le niveau de prélèvement (i.e effort) assurant le rendement maximum durable (MSY). Ainsi:

$$e_{MSY} = \frac{r}{2q} \quad (\text{I.9})$$

On obtient le niveau de biomasse pour lequel la croissance naturelle du stock est au maximum simplement par la maximisation de $G(x)$ ce qui nous donne en utilisant les conditions d'optimalité du premier ordre ($\frac{dG(x)}{dx} = 0$):

$$x_{MSY} = \frac{K}{2}$$

Ainsi, en remplaçant l'expression de x_{MSY} dans l'éq. (I.2), on obtient la production maximale équilibrée (MSY):

$$H_{MSY} = G(x_{MSY}) = \frac{rK}{4} \quad (\text{I.10})$$

D'origine américaine¹⁴, le MSY se diffuse, et intègre le droit international en 1982 puis en 2002 avec le Sommet Mondial sur le Développement Durable (Mesnil, 2012). Il s'insère assez tard dans la politique commune des pêches (PCP) européenne, mais devient en 2013, un axe principal des réformes proposées et mises en place (European Union, 2013).

Maximum Economic Yield (MEY)

De nombreux scientifiques ou pays contestent l'utilisation du MSY comme seul objectif de rendement et posent la question de l'optimalité d'un tel mode de gestion (Larkin, 1977; Kompas et al., 2009; Christensen, 2010; Grafton et al., 2010; 2012). En effet, d'un point de vue économique, le MSY peut être critiquable notamment car la pêche entraîne des coûts fixes mais surtout variables (carburant, réparations...). Ainsi plutôt d'essayer de maximiser les captures, Gordon (1954) suggère de maximiser les rendements économiques (i.e profits). Cette objectif est aujourd'hui nommée le Rendement Maximum Economique (*Maximum Economic Yield* - MEY).

Afin d'obtenir le MEY, nous maximisons la fonction de profits à l'équilibre (eq. (I.7)) comme suit:

$$\max_e \pi(e) \quad (\text{I.11})$$

En utilisant les conditions d'optimalité du premier ordre et en égalisant à zéro la dérivée première de la fonction de profits par rapport à e (eq. I.6), on obtient le niveau d'effort correspondant au MEY:

$$e_{MEY} = \frac{r}{2q} \left(1 - \frac{c}{pqK} \right) \quad (\text{I.12})$$

Si l'on remplace $e = e_{MEY}$, dans l'éq. I.3 et que l'on pose $H(x, e_{MEY}) = G(x)$, on obtient ensuite le stock x_{MEY} à l'optimum économique:

$$x_{MEY} = \frac{K}{2} + \frac{c}{2pq}$$

Ainsi, en remplaçant l'expression de x_{MEY} dans l'éq. (I.2), on obtient le rendement maximum économique (MEY):

$$H_{MEY} = G(x_{MEY}) = \frac{rK}{4} \left(1 - \left(\frac{c}{pqK} \right)^2 \right)$$

MSY vs MEY

En comparant les résultats théoriques de ces deux points de références, on peut observer (s.s.i les coûts variables c sont strictement positifs) que l'effort au MEY est inférieur à l'effort au MSY et à l'inverse, la biomasse au MEY est supérieure à celle au MSY (Figure I.4). Par conséquent, le MEY étant plus profitable, et plus soutenable que le MSY, de nombreux économistes plaident pour l'utilisation du MEY en tant que point de référence pour la gestion des pêches (Larkin, 1977; Kompas et al., 2009; Christensen, 2010; Grafton et al., 2010; 2012).

¹⁴Initialement introduit par le concept d'*Optimum Yield* entré en vigueur en 1976 avec le *Magnuson-Stevens Fishery Conservation and Management Act* (NOAA, 2007).

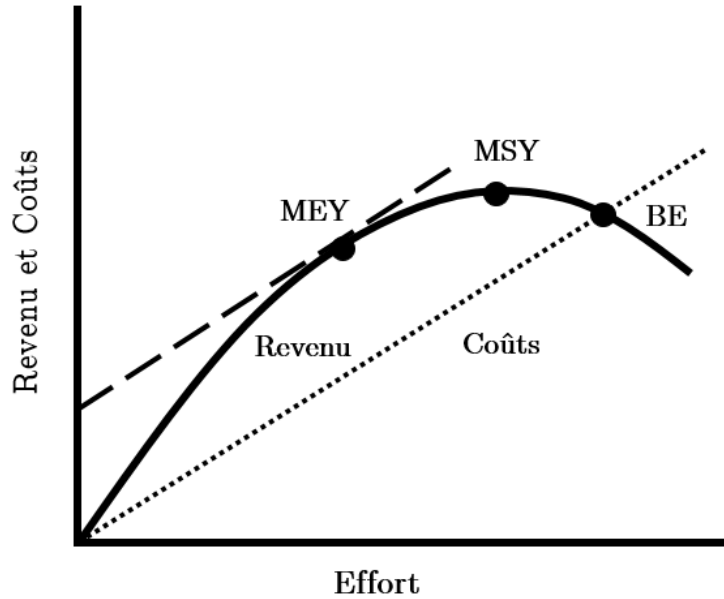


Figure I.4: Modèle de Gordon-Schaefer où le revenu et les coûts sont fonction de l'effort de pêche. Le revenu égalise les coûts à l'équilibre bioéconomique (BE). (Christensen, 2010)

Ces points de références (MSY, MEY) sont alors fondamentaux dans la définition des instruments de gestion de pêche actuels tels que les Totaux Autorisés de Capture (TAC), les Quotas Individuels Transférables (QIT ou *ITQ - Individual Transferable Quota*) ou les licences de pêche. Une fois ces TACs fixés en fonction du point de référence poursuivi, ils cherchent à répondre à un autre enjeu: éviter la course au poisson et les phénomènes de course au poisson et de développement de surcapacités qui en résultent. D'autres méthodes de gestion alternatives (en matière de protection du potentiel de production et de reproduction des stocks exploités), en revanche, n'utilisent pas nécessairement ces points de références, on retrouve ainsi les droits d'usage territoriaux (TURFs - *Territorial Use Rights for Fishing programs*), les AMP (Aires Marines Protégées), ainsi que tous les outils législatifs et réglementaires (taille minimale des mailles de filets, saisons de pêche, taille minimale des espèces ciblées, techniques de pêches légales ou non...). Ces instruments dépendent bien évidemment des caractéristiques des pêcheries (taille, nombre d'agents, abondances relatives, diversité fonctionnelle, vulnérabilité aux chocs, etc...) et des choix des décideurs.

MEY dynamique

Par ailleurs, il est également important de noter que comme n'importe quel actif, le flux de profits peut varier dans le temps. Ainsi, les profits globaux d'une pêcherie à long terme peuvent alors être définis comme la somme des profits escomptés dans le temps i.e la Valeur Actuelle Nette (Net Present Value - NPV) comme suit (nous prenons le cas d'un modèle discret):

$$NPV(e) = \sum_{t=t_0}^T \delta^t \pi(x(t), e(t)), \quad (\text{I.13})$$

où δ est le facteur d'escompte.

Nous pouvons ensuite maximiser cette fonction dans le but d'identifier l'effort optimal de pêche au MEY escompté tel que:

$$\max_e NPV(e) \quad (\text{I.14})$$

Pour trouver l'équilibre de long terme (MEY dynamique), on se sert des conditions d'optimalité via une formulation Hamiltonienne, combinant le critère de profits et la dynamique du système comme détaillées par Conrad et al. (1987); De Lara and Doyen (2008).

Des limites et une remise en cause du mode gestion classique

Plus globalement, une gestion des pêcheries fondée sur l'implémentation de points de référence mono-spécifiques (MSY ou MEY voir MEY dynamique) demeure l'approche la plus utilisée à travers le monde. Cependant, ces modèles ne prennent pas en compte toute la complexité d'un écosystème. C'est pourquoi de nombreux scientifiques (Hilborn et al., 2004; Guillen et al., 2013; Voss et al., 2014; Doyen et al., 2017) recommandent la mise en place d'une approche multi-spécifique notamment car de nombreuses flottilles (e.g chalutiers) ne capturent pas seulement une espèce.

En effet, Larkin (1977) critique l'hégémonie du MSY sur deux aspects: le premier étant que "les espèces de moindre productivité [seront] progressivement pêchées jusqu'à la limite (MSY) de leur extinction" ; et la deuxième étant que les modèles mono-spécifiques n'intègrent pas la complexité des interactions trophiques et écologiques s'opérant au sein d'un milieu naturel (e.g le MSY des proies dépend de la biomasse des prédateurs et réciproquement). On observe ainsi une corrélation négative proie-prédateur (May et al., 1979). A cela, il est possible d'ajouter toutes les relations indirectes liées à l'habitat, au changement climatique et aux espèces qui ne sont pas en lien direct (Yodzis, 2001) ainsi que les interactions techniques (e.g prises accessoires) (Tromeur and Doyen, 2019). Par conséquent, comme le souligne Walters et al. (2005), la gestion au MSY (et MEY) d'une espèce peut d'une part, réduire la biomasse des hauts niveaux trophiques, et d'autre part mettre en danger la résilience et la structure même de l'écosystème exploité. Cela s'explique notamment car aucun objectif écologique (maintien de la résilience, conservation de la biodiversité...) explicite n'est formulé.

1.3 L'approche écosystémique: Une nouvelle génération de défis

Origines et Définitions

L'approche écosystémique émerge en parallèle des premiers questionnements sur les limites écologiques vers la fin des années 60 et le début des années 70 suite à la conférence de Stockholm de 1972. L'objectif est alors de développer des approches intégrées de la gestion de l'environnement, basées sur les fondations de l'écologie scientifique (Meadows et al., 1972). A la suite des questions, viennent les réponses. Près de 3 décennies plus tard, en 2003, la FAO propose une définition de la gestion écosystémique des pêcheries ou *ecosystem-based fisheries management* (EBFM) qui demeurera la plus répandue et utilisée jusqu'à aujourd'hui:

"L'approche écosystémique des pêcheries vise à trouver un équilibre entre divers objectifs sociaux, prenant en compte les connaissances et les incertitudes des composants biotiques, abiotiques et humains des écosystèmes et leurs interactions, appliquant ainsi une approche intégrée des pêcheries dans le cadre de limites ayant un sens écologique." (FAO, 2003).

Pikitch et al. (2004) résume cette définition: *"L'EBFM est une nouvelle direction pour la gestion des pêches inversant essentiellement l'ordre des priorités de gestion pour commencer par l'écosystème plutôt que les espèces cibles [avec comme objectif global] de maintenir les écosystèmes marins en bonne santé et les pêcheries qu'ils soutiennent"*.

En d'autres termes, l'EBFM tente de prendre en compte les différentes échelles de complexité écologique (multi-espèces), socio-économique (multi-flottilles) et environnementale (changement climatique) afin d'atteindre un modèle un gestion équilibré visant un développement durable.

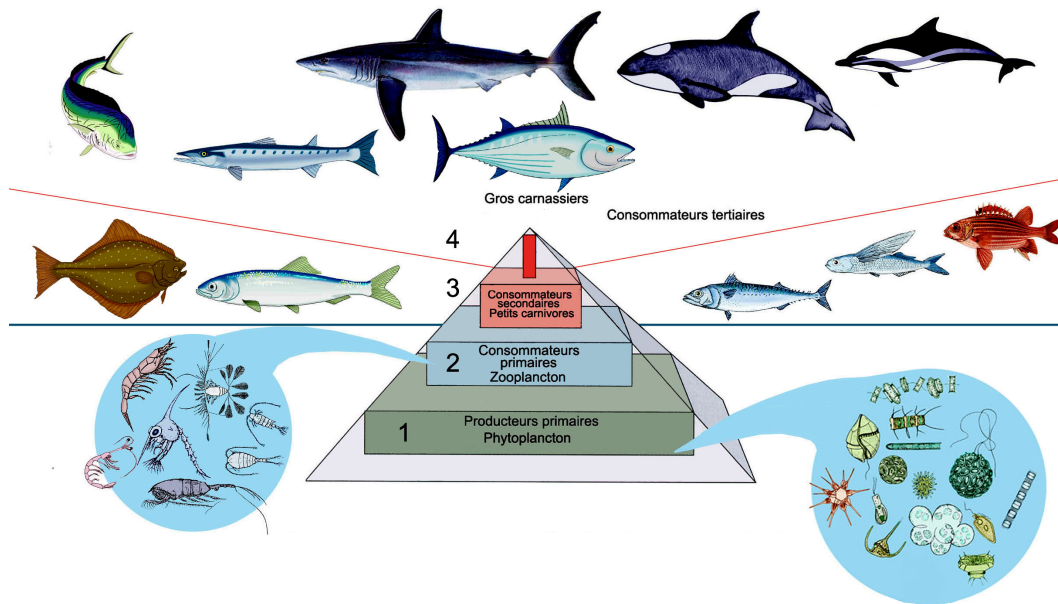


Figure I.5: Illustration schématique d'un réseau trophique.

Défi N°1: De l'étude de la complexité écologique, ...

En premier lieu, l'approche écosystémique consiste à comprendre, analyser et intégrer la complexité des pêcheries dans les modèles de gestion (Link et al., 2017). Il faut donc supposer que toutes, ou au moins une grande majorité des interactions inter- et intra-spécifiques, liées à l'habitat et impactées par le climat sont préalablement comprises par les biologistes, écologues et climatologues. A la suite de l'étude de la complexité, plusieurs approches de modélisation furent développées (Plagányi, 2007). Ces modèles diffèrent de la gestion classique par leur niveau de complexité et de réalisme.

Ecopath ou sa version dynamique plus récente, *Ecopath with Ecosim* (Christensen and Walters, 2004 - *EwE*) tentent par exemple de décrire les flux de biomasse à l'équilibre. Ces modèles font partie d'une première catégorie qui intègrent l'ensemble des niveaux trophiques d'un milieu, du super prédateur (e.g Requins) au producteur primaire (e.g Phyto-planctons) comme illustré par la Figure I.5. Le modèle *Atlantis* inclut quant à lui les rétroactions entre les dynamiques écologiques et les décisions de gestion (Fulton et al., 2011). D'autres modèles se concentrent sur des espèces en particulier et apportent des réponses (dans certains cas) plus précises (*krill-predator-fishery model* - Plagányi, 2007).

Défi N°2: ...socio-économique...

Les caractéristiques multi-spécifiques des pêcheries sont associées à une multiplicité de flottilles, de profils de pêche, et d'engins (fileyeurs, chalutiers spécialistes et mixtes, etc...). Cette complexité de production, qui se définit par une diversité de bateaux et techniques de pêches, permet potentiellement d'assurer une diversification des cibles (Burgess, 2015). Cependant, ces flottilles sont souvent en compétition pour les mêmes ressources, avec lesquelles elles interagissent à des degrés divers suivant leur stratégies et leurs techniques de pêche (Ulrich et al., 2001).

A travers une variation de l'impact anthropique via une croissance démographique perpétuelle et une volatilité relative des prix liée au carburant (et par extension des marchés des produits halieutiques), le contexte socio-économique (Demande/Offre) ajoute un nouveau degré de complexité économique à la pré-existante complexité écologique. Les pressions humaines étant multiples, on peut ainsi parler de **multi-pressions**.

En résumé, la gestion de la complexité des écosystèmes n'est pas seulement d'ordre écologique.

Elle requiert également une gestion écosystémique liée à la gouvernance des flottilles qui devrait être définie en s'appuyant sur une répartition soutenable de l'effort de pêche ou de quotas entre les flottilles.

Défi N°3: ...à l'analyse, l'évaluation et la gestion de l'incertitude globale...

Cette complexité écologico-économique pourrait être fortement impactée par le changement climatique et ses conséquences. L'évaluation et l'intégration de l'incertitude climatique et de ses conséquences s'ajoute alors à l'approche écosystémique en tant que préalable indispensable conditionnant l'aide à la décision.

L'incertitude climatique est représentée dans les modèles écosystémiques par une stochasticité qui est principalement écologique mais peut également être économique (Bockstael and Opaluch, 1983; Doyen and De Lara, 2010; Doyen et al., 2012; Gourguet et al., 2013; Lagarde et al., 2018). Son évaluation dépend alors de notre connaissance de l'évolution du milieu (amplitude de changement de température, de nutriments...). Une fois que nous connaissons les changements potentiels des caractéristiques de l'habitat, il est ensuite possible d'étudier l'impact que cette évolution génère sur les différentes espèces et sur l'écosystème en général. La difficulté est ensuite d'évaluer le poids de cet impact (qui peut être différent ou non pour chaque espèce en fonction de leur sensibilité, vulnérabilité et résistance écologique) sur le recrutement, la mobilité et la prédation des espèces étudiées ainsi que sur les conséquences dans la chaîne trophique. Il y a plus de 100 ans, Hjort (1914) recommandait déjà l'intégration de paramètres environnementaux dans l'évaluation des stocks de poissons. De nombreux chercheurs, à l'instar de P. Cury, ont depuis pris le relais et recommandent une gestion écosystémique dépendant de l'environnement (Cury et al., 2005).

L'incertitude peut également être d'ordre socio-économique comme évoqué précédemment (volatilité des prix du carburant, matières premières).

Par ailleurs, il existe différents moyens d'évaluer et de modéliser l'incertitude: par régression¹⁵, par des scénarios stochastiques¹⁶ (Doyen et al., 2017) ou par des scénarios d'émissions de gaz à effet de serre¹⁷ déterminés par les climatologues (Groupe Intergouvernemental sur l'Etude du Climat - GIEC).

Défi N°4: ...pour une gestion multi-dimensionnelle.

L'approche écosystémique vise également à trouver "un équilibre entre divers objectifs sociaux" telle que l'expliquait la définition de la FAO (2003). Ils peuvent également être d'ordre socio-économique (e.g demande alimentaire - Cissé et al., 2015), si l'on prend en compte les services écosystémiques conditionnant (en partie) la rentabilité des pêcheries.

Les objectifs socio-économiques peuvent être préalablement définis par le modélisateur ou déterminés pendant le calcul (*multi-objective programming*). Ils peuvent par exemple être globaux (objectif de rentabilité pour toute la pêcherie) ou équitables (chaque flottille doit générer des profits). Une optimisation multi-critère nous indique ainsi l'ensemble des solutions possibles dans l'aide à la gestion. Le gestionnaire peut ensuite favoriser celles qu'il préfère. On peut trouver des méthodes spécifiques à la définition de politiques de gestion soutenable partiellement décrites dans une étude réalisée par Mardle and Pascoe (1999).

Plus concrètement, l'implémentation de tels modèles de gestion est conditionnée par la préférence des acteurs pour les objectifs recherchés (Kempf et al., 2016). Une méthode récente a émergé et consiste en un contrôle des composants écologiques et économiques des pêcheries liées

¹⁵ Analyse d'une quelconque corrélation significative entre l'environnement et le stock à laquelle on rajoute une amplitude plus ou moins forte.

¹⁶ Probabilité qu'un événement climatique ou environnemental déjà observé se reproduise.

¹⁷ Analyse des conséquences anthropiques sur l'évolution du climat et ses effets sur le milieu étudié du pire au meilleur scénario.

à des seuils et des contraintes qui ont le mérite d’assurer une performance minimale au niveau de chaque critère. Cette méthodologie se nomme l’éco-viabilité et montre, en théorie, la possibilité d’améliorer les performances écologico-économiques (Béné et al., 2001; Baumgärtner and Quaas, 2009; Gourguet et al., 2013; Schuhbauer and Sumaila, 2016; Doyen et al., 2017; Oubraham and Zaccour, 2018).

Pour résumer, même si la plupart des pêcheries mettent en place des modèles de gestion explicitement pourvus de divers objectifs (écologiques, économiques et sociaux), les modes de gestion véritablement écosystémiques restent rares mais semblent être le préalable théorique nécessaire à un développement durable et au maintien de l’équilibre entre ses 3 piliers.

De la théorie à la pratique

En pratique, de nombreuses pêcheries intègrent plusieurs objectifs (écologiques et économiques), le MSY n’étant plus un objectif de gestion mais un seuil, un plafond au dessous duquel les stratégies de gestion doivent se situer afin d’être optimales (Pitcher et al., 2009). En effet, un effort de pêche (i.e une pression de pêche) plus faible permet bien souvent des rendements proches du MSY, et une biomasse plus élevée (Hilborn, 2010). On peut ainsi parler *d’optimum yield* qui se définit comme tel: c’est la “quantité de poisson apportant les plus grands bénéfices à la Nation [qui est estimée] sur la base du MSY, réduit par tout facteur écologique, social, ou économique” (Larkin, 1977; Patrick and Link, 2015a). La politique de pêche américaine est principalement basée sur ce concept depuis les années 50 (NOAA, 2007) tandis que l’Union Européenne (2016) ne fait pas mention du concept *d’optimum yield* mais assure que ses objectifs sont d’ailleurs: soutenabilité écologique et maintien de bénéfices économiques et sociaux (i.e développement durable).

Dans une méta-analyse de 1200 stocks, Skern-Mauritzen et al. (2016) révèlent que seulement 24 pêcheries affichaient une gestion intégrant des paramètres biophysiques ou interactions écologiques. Néanmoins, il semble nécessaire de noter qu’à l’instar de l’intérêt croissant des scientifiques pour cette approche, il est plus que probable que ce taux de 2 % augmente. Deux questions émergent alors: cela sera-t-il suffisant à terme ? Et sommes-nous capables de réaliser de bonnes performances, en utilisant l’EBFM? Selon un bilan de Pitcher et al. (2009), aucun pays utilisant partiellement l’approche écosystémique ne réalise de bonnes performances¹⁸.

Opérationnaliser l’EBFM et relever ces 4 défis semble compliqué. Cette thèse tente d’intégrer ces dimensions pour progresser vers des modèles de gestion durable et écosystémique.

1.4 Soutenabilité, viabilité et résilience

La soutenabilité (Brundtland et al., 1987), la résilience (Holling, 1973) et la viabilité (Aubin, 1991) sont trois concepts qui jouent un rôle important pour aborder les enjeux environnementaux et les questions des limites écologiques posées vers la fin du XXème siècle. L’implémentation de ces trois concepts conditionne la pérennité de l’humanité dans les siècles à venir (OECD, 2020).

Soutenabilité

Un développement soutenable/durable est:

“Un développement qui répond aux besoins du présent sans compromettre la capacité des générations futures à répondre aux leurs”. (Brundtland et al., 1987)

Synthétiquement, un état est soutenable lorsqu’il peut perdurer indéfiniment. Se basant sur la définition du rapport Brundtland, les Nations Unis définissent en 2015, 17 objectifs économiques, sociaux et environnementaux inter-dépendants de développement soutenable à atteindre pour 2030 (Figure I.6). L’équité inter-générationnelle souhaitée par l’ONU est alors soutenue par

¹⁸De plus, l’IDH (Indice de Développement Humain) des pays semblent également être corrélé avec les résultats



Figure I.6: Les “17 objectifs de développement durable pour transformer le monde à l’horizon 2030” (UN, 2015).

de nombreux indicateurs. Ceux utilisés dans cette thèse sont définis dans la section 2.2 et d’autres sont listés non-exhaustivement par Singh et al. (2009). Selon Reid and Rout (2020), ces indicateurs de soutenabilité, devenant de plus en plus intégrés, complexes et composites (multi-dimensionnels) apparaissent parfois arbitraires, opaques et seulement qualitatifs car souvent inquantifiables au regard de certains concepts. Un équilibrage de valeur, un facteur d’équilibre entre les considérations écologiques et les objectifs économiques (e.g en y incorporant un facteur et un poids de non-susstituabilité) pourrait permettre d’obtenir de meilleurs indicateurs de soutenabilité effectuant le lien entre environnement et exploitation (Pearson, 2016).

Déjà mis en cause par Malthus (1852), la volonté de croissance perpétuelle, liée à l’idéologie libérale (et la nature humaine ? Marx, 1844), semble en contradiction avec les objectifs de soutenabilité, le mode de production capitaliste et la mondialisation exacerbant l’ensemble des conséquences (Meadows et al., 1972). Ainsi, on voit apparaître des propositions de modèles de sociétés, sans croissance, redéfinissant de nombreux concepts tels que la richesse, le bien-être ou la prospérité (Jackson, 2016) ainsi que des créations de partis politiques (Parti pour la Décroissance¹⁹) pronant cette *Halte à la croissance* souhaité par le Club de Rome de 1968.

Plus particulièrement, le concept de soutenabilité forte²⁰ semble désormais prévaloir sur l’idée de soutenabilité faible et de substituabilité du capital naturel parmi les économistes écologiques (Vivien, 2009). Cependant, l’aquaculture (capital objectivement créé par l’Homme) a dépassé depuis quelques années la production halieutique mondiale (Figure I.1 - FAO, 2018) revenant ainsi au concept de soutenabilité faible²¹ conditionnée par l’innovation salvatrice oubliée par Malthus (1852), expliquée par Schumpeter (1912) et modernisée par Drucker (1985). Dès lors, les économistes recommandent une innovation recentrée sur des objectifs écologiques et sociaux (Nakamori, 2020). Depuis quelques années le progrès technique a, en effet, été le moteur de nombreuses actions écologiques²² et a priori soutenables. Néanmoins, il y a 2 siècles, c’est ce même progrès technique qui fut le déclencheur des problèmes actuels. Auparavant source des problèmes écologiques, l’innovation n’est donc pas la solution mais désormais l’une des clefs à la disposition de l’Humanité pour assurer sa soutenabilité.

Globalement, le concept de soutenabilité (plus connu en France sous le synonyme de durabilité) occupe une place croissante dans les objectifs de politique publique ; reste à déterminer et opérationnaliser les outils de gestion, les instruments de politique publique et les incitations.

¹⁹ <https://www.partipourladecroissance.net/>

²⁰ Concept définissant le capital naturel comme non-substituable au capital créé par l’Homme.

²¹ Concept considérant l’existence d’une substituabilité entre le capital naturel et celui créé par l’Homme.

²² Énergie éolienne, géothermique, hydroélectrique, solaire, de biomasse, le biocarburant, voiture électrique, procédé de désalinisation, agriculture biologique, aquaculture plus productive, etc...

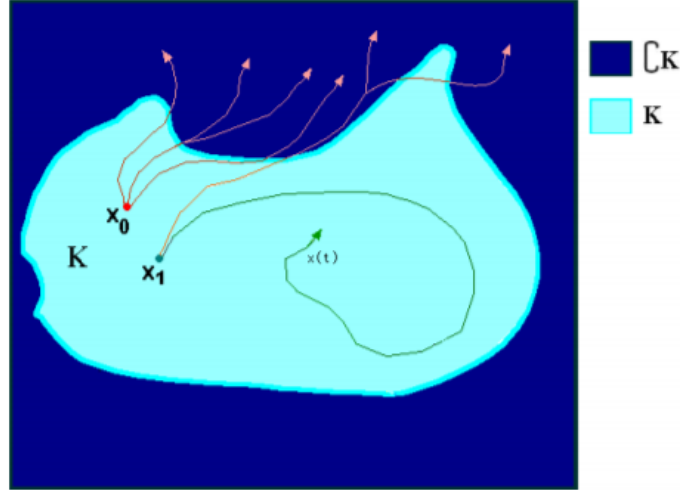


Figure I.7: Représentation mathématique du concept de viabilité où 2 états initiaux du système x_0 et x_1 sont présentés. Les évolutions temporelles du premier état x_0 violent toutes les contraintes dans un temps fini, tandis que l'autre x_1 affiche une évolution viable (qui demeure dans K) et une autre non viable qui n'appartient plus à K (Aubin et al., 2011).

Viabilité

Pour aborder les enjeux de soutenabilité, l'approche de viabilité est pertinente car elle consiste en une interprétation des contraintes minimales socio-économico-écologiques à respecter pour toutes les générations (Martinet and Doyen, 2007; Baumgärtner and Quaas, 2009; Martinet, 2011; Doyen and Martinet, 2012).

En 1991, Aubin fournit un cadre mathématique²³ dynamique au concept de viabilité créant une rupture radicale avec les approches « statiques » et d'optimisation des mathématiques de la décision. Il la définit comme tel:

[Cette théorie] conçoit et développe des méthodes mathématiques et algorithmiques pour étudier l'adaptation aux contraintes de viabilité des évolutions régies par des systèmes complexes en situation d'incertitude que l'on retrouve dans de nombreux domaines (celui des êtres vivants, de l'évolution biologique à l'économie, des sciences de l'environnement aux marchés financiers, de la théorie du contrôle et de la robotique aux sciences cognitives). (Aubin et al., 2011)

Les différents seuils sont expliqués dans la section 2.3 tandis que la représentation schématique du concept de viabilité est affichée dans la Figure I.7.

En pratique, une des difficultés est alors de définir l'ensemble d'états viables. La définition de seuils de viabilité socio-écologico-économiques est ainsi le préalable nécessaire à l'implémentation

²³Mathématiquement, si nous reprenons les travaux de Aubin (1991); Doyen and De Lara (2010), la viabilité peut se définir comme suit. Considérons d'abord un système de contrôle dynamique (exemple en temps discret):

$$x(t+1) = F(x(t), u(t), \omega(t))$$

où $x(t)$ est l'état du système à l'instant t , $u(t)$ est le contrôle (la décision, l'action) du système à l'instant t et $\omega(\cdot)$ représente des scénarios exogènes (non contrôlables) compris dans un domaine de scénarios possibles (vecteur) noté Ω . F est alors une fonction d'évolution du système dépendant d'un état du système $x(t)$, du contrôle $u(t)$, et de scénarios stochastiques $\omega(t)$. On définit ensuite un sous-ensemble K considéré comme *viable*. Une trajectoire $x(\cdot)$ est dite une solution viable pour le système si:

$$\forall t \geq 0, x(t) \in K$$

Concept	Définition
Résistance	La capacité d'un système à activement changer en conservant son identité ou à passivement maintenir ses performances à la suite d'un ou plusieurs événements indésirables.
Robustesse	La probabilité pour un système de maintenir son identité et de ne pas descendre en dessous d'un seuil indésirable (ou irréversible) à la suite d'un ou plusieurs événements indésirables.
Récupération	Une mesure du temps de récupération d'un système pour revenir à ses fonctionnalités désirées à la suite d'un ou plusieurs événements indésirables.

Table I.1: Définitions des 3 R d'une résilience socio-écologique selon Grafton et al. (2019)

de cette méthode. De ce point de vue, des liens importants ont été récemment mis en évidence entre les approches de viabilité et du 'maximin' (Doyen and Martinet, 2012; Doyen and Gajardo, 2019). Plus largement, des liens majeurs ont aussi été mis en évidence entre l'approche de viabilité et d'autres approches importantes pour les sciences de la durabilité (Doyen et al., 2019); en particulier, on trouve l'approche 'safe operating space' (SOS, Rockström et al., 2009).

Résilience

Le concept de résilience joue un rôle croissant dans les prises de décision, y compris dans la gestion des risques du secteur privé (Sheffi, 2015), les investissements pour le développement (OECD²⁴, 2018) et les objectifs d'agences multilatérales (e.g FAO²⁵, 2020 et World Bank, 2013). Cependant, les scientifiques et les décideurs manquent d'une compréhension commune de la résilience et les applications pratiques pour la gestion de l'environnement sont rares (Béné and Doyen, 2018).

La définition exacte de la résilience socio-écologique fait toujours débat au sein de la communauté scientifique, néanmoins ce concept est communément compris comme étant:

“les caractéristiques d'un système qui lui permettent de se maintenir en termes de performances ou fonctionnalités malgré un ou plusieurs événements indésirables.” (Grafton et al., 2019)

Grafton et al. (2019) tentent de définir plus précisément ce concept en lui attribuant trois dimensions nommées les 3 R: Résistance, Récupération, Robustesse (*Resistance, Recovery, Robustness* - Table I.1). La Figure I.8 illustre le concept de Résilience socio-écologique et des 3 R en identifiant 3 scénarios possibles suivant la situation initiale (T_0).

Dans le scénario 1, représenté par la courbe verte, aucun événement indésirable ne se produit et par conséquent aucun impact sur les performances du système n'est observé. La résilience socio-écologique du système n'est donc pas mise à l'épreuve avec une résistance maintenant les performances, un temps de récupération fixé à zéro, et une robustesse déterminée par une probabilité inchangée (p_1) de ne pas franchir le seuil.

Dans le scénario 2, représenté par la courbe jaune, un événement indésirable d'impact modéré se produit. La résilience socio-écologique du système est caractérisée par: une résistance diminuée générant une perte des performances du système K par rapport au niveau précédent de performances M ; une récupération diminuée car disposant d'un temps de récupération plus grand que dans le premier scénario ($T_3 - T_0 > 0$); et une robustesse diminuée déterminée par une probabilité de ne pas franchir le seuil plus faible ($p_2 < p_1$) que dans le premier scénario.

²⁴<http://www.oecd.org/dac/conflict-fragility-resilience/>

²⁵Cadre institutionnel pour soutenir une paix durable dans le contexte de l'Agenda 2030.

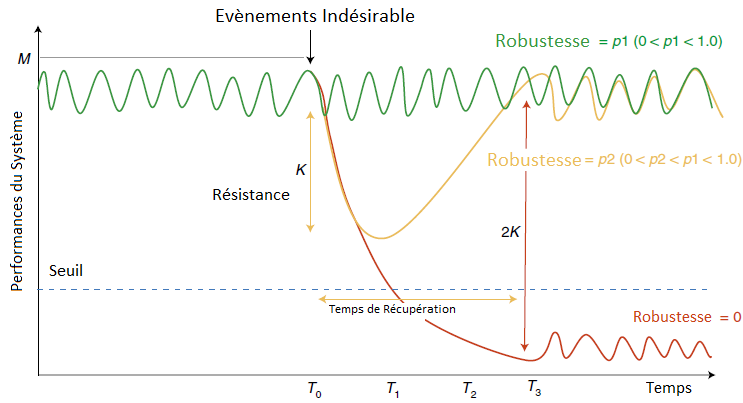


Figure I.8: Exemple conceptuel d'effets possibles d'un événement indésirable sur la Résistance, la Récupération et la Robustesse (Grafton et al., 2019). Les courbes colorées correspondent à trois scénarios différents ou comment un écosystème peut réagir face à un événement indésirable.

Dans le scénario 3, représenté par la courbe rouge, un événement indésirable avec une faible probabilité mais doté d'un large impact se produit. La résilience socio-écologique du système est caractérisée par: une résistance diminuée générant une perte des performances du système $2K$ par rapport au niveau précédent de performances M ; une récupération impossible car disposant d'un temps de récupération infini ; et une robustesse déterminée par une probabilité de ne pas franchir le seuil égale à 0.

De plus, un système socio-écologique affiche une hystérésis et une dépendance vis à vis de son histoire (Tilman, 1999; David, 2007). Par conséquent, l'adaptation et la transformation des performances du système, grâce à la gestion de la résilience, peuvent probablement (et doivent?) se produire avant, pendant et après un événement indésirable (Grafton et al., 2019).

La mise en place d'approches multi-critères impliquant des normes/valeurs monétaires (profits socio-économique) et non monétaires (biodiversité, services écosystémiques) favorisent cette mise en pratique.

2. Méthodes et Modélisation écosystémiques

2.1 Modèles écosystémiques

Parmi les modèles écosystémiques, il peut exister différents degrés de complexité. Les scientifiques recommandent de passer d'une gestion mono-spécifique à une gestion multi-spécifique (Plagányi, 2007; Tromeur and Doyen, 2019). Certains modèles n'intègrent que les interactions trophiques à un instant t (EcoSim - Christensen et al., 2005), d'autres intègrent des dynamiques temporelles (EwE - Christensen et al., 2005, Lotka Volterra). À partir de la théorie du contrôle et des systèmes dynamiques comme dans Thébaud et al. (2014); Doyen (2018); Doyen et al. (2019), on peut considérer le modèle **stochastique, dynamique, multi-spécifique, multi-flottille** suivant (dans le cas d'un modèle discret):

$$x_i(t+1) = F_i(x(t), e(t), \omega(t)) \quad (\text{I.15})$$

où F_i est la fonction d'évolution de la population d'une espèce i . Elle dépend d'un vecteur $x(t) = (x_1(t), x_2(t), \dots, x_n(t))$ qui inclut toutes les biomasses $x_i(t)$ d'une espèce i en t et d'un vecteur $e(t) = (e_1(t), e_2(t), \dots, e_m(t))$ rassemblant tous les efforts de pêche d'une flottille f , noté $e_f(t)$ ainsi que de scénarios stochastiques $\omega(t) = (\omega_1(t), \omega_2(t), \dots, \omega_p(t))$ qui intègrent quant à eux l'incertitude liée au changement climatique, aux chocs environnementaux et à la variabilité économique (carburant, matières premières) à chaque période t .

Un tel modèle est suffisamment générique pour être décomposé en **classe d'âge** (Quinn and Deriso, 1999; Gourguet et al., 2013; 2014; Lagarde et al., 2018) ou être spatialement explicite (Doyen et al., 2007; Mouysset et al., 2013; Lagarde et al., 2020).

2.2 Indicateurs écologo-économiques

L'analyse des résultats des modèles écosystémiques présentés précédemment est ainsi conditionnée par l'intégration d'indicateurs socio-économiques et de biodiversité.

2.2.1 Indicateurs de biodiversité

En toute généralité, les indicateurs de biodiversité sont de type indicateur d'état pouvant être noté en tant que vecteur I et dépendent d'un état du système $x(t)$ tel que:

$$I(x(t)) = I(x_1(t), x_2(t), \dots, x_n(t)) \quad (\text{I.16})$$

Parmi la multitude d'indicateurs de diversité pouvant exister dans la littérature (Magurran, 2007), nous avons choisi de présenter l'indice de Simpson (SI) ainsi que l'indice trophique marin (Marine Trophic Index - MTI) car utilisés dans les prochains chapitres et nécessaires pour l'analyse de la soutenabilité, la résilience et de la viabilité, dont la définition a été donnée précédemment. Ces indicateurs sont souvent utilisés dans l'évaluation des performances écologo-économiques des pêcheries et apparaissent ainsi relativement pertinents pour la suite de cette thèse.

Indicateurs	Définition	Equation
Marine Trophic Index	Le niveau trophic moyen des captures d'une pêcherie (Pauly and Watson, 2005).	(I.18)
Simpson	La mesure de l'uniformité/homogénéité (ou d'équitabilité) d'un écosystème (Somerfield et al., 2008).	(I.17)

Table I.2: Tableau d'indicateurs de biodiversité.

Indice de diversité de Simpson (SI)

L'indice de Simpson estime l'homogénéité d'un système i.e la répartition de chaque espèce considérée dans un milieu donné. Il peut être calculé à partir des abondances, biomasse ou stock fécond (Chap II) tel que:

$$SI(t) = \left(\sum_{i=1}^n \left(\frac{x_i(t)}{\bar{x}(t)} \right)^2 \right)^{-1} \quad \text{avec} \quad \bar{x}(t) = \sum_{i=1}^n x_i(t) \quad (\text{I.17})$$

où $x_i(t)$ est la biomasse de l'espèce i en t . L'indice varie entre 0 et n . Il est à son maximum si on observe une équi-répartition des espèces et vaut alors n le nombre d'espèces totales. Néanmoins, cet indicateur est très sensible aux espèces abondantes.

Indice Trophique Marin (MTI)

L'indice trophique marin (des captures par définition) estime le niveau trophique moyen de l'écosystème exploité. Bien qu'en réalité, il ne soit qu'un proxy du niveau trophique moyen de l'écosystème, cet indicateur nous montre l'évolution des captures dans la chaîne trophique. Il varie de 2 (petites proies/herbivores) à 5 (grands prédateurs/piscivores) et est fonction du niveau trophique de chaque espèce i débarquée noté T_i et du nombre d'espèces capturées $H_i(t)$ (décrits à l'éq. (I.19)) à l'année t (Pauly and Watson, 2005).

$$MTI(t) = \sum_i H_i(t)T_i \quad (\text{I.18})$$

Plus globalement, il existe une myriade d'indicateurs de biodiversité tels que la richesse spécifique (Mesure la diversité d'espèces au sein d'un écosystème - Gotelli and Colwell, 2011), l'indice de Shannon (Mesure la diversité d'espèces et leur répartition au sein d'un écosystème - Spellerberg and Fedor, 2003), le nombre de Hill (Mesure la diversité d'espèces et leur homogénéité au sein d'un écosystème - Chao et al., 2016) ou la diversité fonctionnelle¹.

2.2.2 Indicateurs socio-économiques

Parmi les indicateurs socio-économiques, nous trouvons le niveau de captures (éq. (I.19)), les profits (éq. (I.20)) ainsi que les salaires qui en résultent et l'effort de pêche (pouvant être un indice de l'intensité de la pression anthropique ou de l'emploi) ainsi que l'évolution des prix des espèces ciblées. Les captures par espèces $H_i(t)$ peuvent être définies de manière générique comme suit:

$$H_i(t) = H(q_{i,f}, e_f(t), x_i(t), \omega(t)) \quad (\text{I.19})$$

où H est une fonction de captures pouvant se formaliser de différentes manières telle que le modèle de Gordon-Schaefer par exemple. $q_{i,f}$ est la capturabilité² de la flottille f pour une

¹L'indice de Diversité Fonctionnelle considère et évalue le nombre d'espèces ou de groupes d'espèces dotés de différentes fonctions au sein d'un écosystème (e.g herbivores, piscivores, corallivores... - Chap III). Cela permet notamment d'intégrer le concept de redondance écologique (i.e deux espèces ayant la même fonction au sein d'un écosystème). Cet indicateur semble mieux expliquer et refléter la bonne santé d'un écosystème (Lande, 1996) car il s'appuie non pas sur la diversité des individus mais sur la diversité des fonctions écosystémiques des espèces. Il existe différentes manières de calculer et définir cet indicateur (Petchey and Gaston, 2006) mais la principale méthode est de regrouper les espèces en groupes fonctionnels (Chap III).

Dans le cas d'habitats (animal et végétal) tels que le corail ou les algues, le regroupement s'effectue en fonction de leur famille, ordre, classe ou règne selon la principale fonction du rang taxonomique que l'on souhaite étudier.

Cet indicateur est notamment souvent associé à l'évaluation de la résilience d'un écosystème et par conséquent apparait particulièrement pertinent dans l'évaluation de systèmes complexes tels que les récifs coralliens (Lande, 1996).

²La capturabilité peut également être un indicateur de la capacité de pêche des flottilles. Plus la probabilité de capturer une espèce est forte, plus l'impact anthropique de la pêche est fort.

espèce i (i.e la probabilité pour la flottille de rencontrer et capturer une unité de biomasse par unité d'effort), $e_f(t)$ l'effort de la flottille f en t , $x_i(t)$ le stock de l'espèce i en t et $\omega(t)$ des scénarios stochastiques. Les profits par flottille $\pi_f(t)$ peuvent également être définis comme la différence entre les coûts et les revenus génériquement comme suit:

$$\pi_f(t) = \sum_i p_i(\omega(t))H_i(t) - C_f(\omega(t))e_f(t) \quad (\text{I.20})$$

où $p_i(\omega(t))$ représente le prix de l'espèce i évoluant selon des scénarios stochastiques écologico-économiques $\omega(t)$, $C_f(\omega(t))$ les coûts (variables et constant) de la flottille f variant quant à eux selon des scénarios stochastiques socio-économiques $\omega(t)$ (carburant, salaires, équipements, etc...).

Parmi les indicateurs les plus difficiles à estimer, nous retrouvons la valeur "monétaire" des services écosystémiques rendus par la nature comme expliquée dans la section 1.1.5 et par Heal (2000). Par exemple, les revenus produits par l'activité touristique liée aux aménités environnementales (diversité d'espèces, de couleur, etc...) peuvent être un bon indicateur de la santé économique (et partiellement écologique) du système (Boncoeur et al., 2002; Weijerman et al., 2018). L'intensité de la pêche récréative (via l'effort ou le nombre de captures - Chap III), montre également l'attrait de la population locale ou non-résidente pour ce socio-écosystème.

2.3 Seuils de viabilité

Afin de définir différents états du socio-écosystème, des seuils de **viabilité** peuvent être définis (Aubin, 1991; Doyen et al., 2017). Certains planchers écologiques ont déjà été fixés parmi les grandes pêcheries grâce notamment à l'évaluation des stocks. Entre autres, le B_{lim} indique par exemple "la biomasse limite en dessous de laquelle on est pas sûr que le stock puisse se reconstituer" et le B_{pa} représente "la biomasse de précaution"³, intégrant une marge de sécurité par rapport au B_{lim} .

Cependant, dans le cas de petites pêcheries où l'évaluation des stocks est bien souvent approximative, il est difficile de déterminer des seuils. On part alors d'une année de base t_0 à laquelle l'écosystème est "considéré" comme en "bonne santé". Il est ensuite possible de déterminer de manière plus ou moins arbitraire ou hypothétique les différents seuils de viabilité biologiques et économiques notés avec l'indice *lim* (e.g avec les captures $H_{lim} = H(t_0)$). Comme théoriquement et génériquement présenté par Doyen and De Lara (2010) et Doyen et al. (2017), les modèles vont alors rechercher les chemins de gestion viable pour lesquels les indicateurs écologico-économiques (Biomasse, Captures, Profits ou ceux présentés ci-dessus, etc...) ne dépassent les contraintes mises en places i.e les seuils de viabilité (liste non-exhaustive) tels que $\forall t$:

$$I(x(t)) > I_{lim} \quad (\text{I.21})$$

$$H(t) \geq H_{lim} \quad (\text{I.22})$$

$$\pi_f(t) \geq 0 \quad (\text{I.23})$$

où $H(t)$ sont les captures en t , $I(x(t))$ un indicateur écologique en t et $\pi_f(t)$ les profits de chaque flottille f en t .

2.4 Stratégies exploratoires de pêche

Afin de simuler et de prévoir l'évolution écologico-économique d'un écosystème et d'évaluer les résultats des modes de gestion mis en place grace notamment aux indicateurs écologico-économiques et de viabilité présentés ci-dessus, il est nécessaire de considérer différents scénarios

³<https://wwz.ifremer.fr/peche/Archives/Precisions-sur-les-diagnostics>

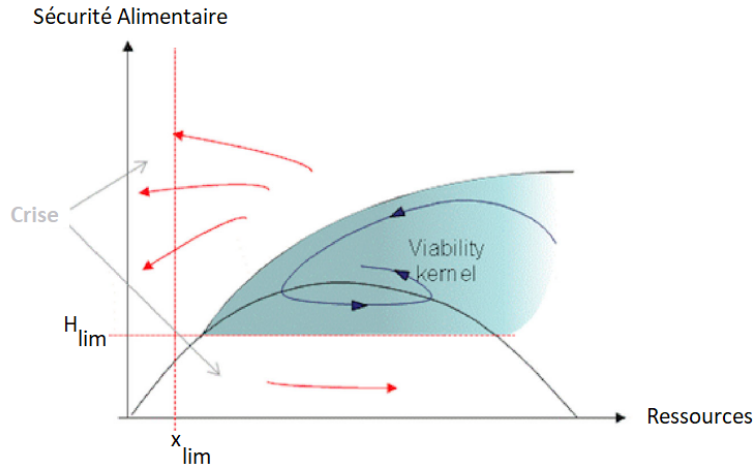


Figure I.9: Nouvelle représentation schématique du concept de viabilité écologico-économique où le viabilité de Kernel (*Viability kernel*) est l'objectif illustré pour le cas de seulement 2 contraintes faisant référence à la conservation écologique versus la sécurité alimentaire. La partie en vert/bleu représente les états viables du système dynamique selon le pourcentage de confiance β introduit dans l'éq. (I.25) tandis que les parties en blanc correspondent aux états non viables (Doyen et al., 2019).

de pêche (Börjeson et al., 2006; Pereira et al., 2010; Doyen et al., 2017; IPBES⁴). Ces scénarios correspondent à des stratégies de gestion visant différents objectifs. Deux grands groupes sont alors étudiés: des scénarios exploratoires et normatifs. Les premiers se contentent d'appliquer un niveau d'effort donné dans le temps et d'évaluer comment se comporte l'écosystème tandis que les deuxièmes sont des scénarios prospectifs qui tentent d'atteindre un objectif ou qui répondent à des contraintes de viabilité.

Status-Quo (SQ)

Ce scénario (appliqué ici sur l'effort de pêche) consiste simplement à maintenir l'effort historique observé⁵ pour chaque flottille f et au cours du temps et d'attester de l'évolution des résultats écologico-économiques. Cela nous donne ainsi un diagnostic de soutenabilité en fonction des conditions actuelles d'exploitation et du contexte écologique tel que:

$$e_f(t) = e_f(t_0) \quad \forall f, \forall t$$

Spatialement explicites (SpE)

Si l'on considère plusieurs zones d'exploitation z , il peut être également intéressant de modifier l'effort d'une zone et de représenter explicitement les possibilités de reports d'efforts de pêche entre différentes zones (grâce à un multiplicateur d'effort $u_z(t)$ dépendant de la zone z et du temps t) sur une autre zone adjacente (ou proche) et réciproquement tel que:

$$e_{zf}(t) = u_z(t)e_{zf}(t_0) \quad \text{avec} \quad \sum_z u_z(t) \leq \alpha(t) \quad \forall f, \forall t$$

où $\alpha(t)$ peut être l'effort de pêche maximum, soutenable ou viable défini par le décideur pour l'ensemble des zones z en t .

⁴<https://ipbes.net/>

⁵Il peut résulter de la moyenne de la période historique étudiée, ou peut juste être l'effort de la dernière année observée qui évolue selon la croissance démographique des pêcheurs.

Cette stratégie⁶ de gestion peut ainsi nous indiquer quelle combinaison favorise un meilleur effet de refuge (lorsque les espèces sont mobiles) et laquelle produit de meilleurs résultats socio-économiques i.e quelles stratégies sont soutenables. Une spatialisation plus précise des modèles de gestion, ainsi qu'une répartition de l'effort plus soutenable peut alors s'opérer.

Plus simplement, cette stratégie de pêche fait référence aux stratégies d'Aires Marines Protégées (AMP), où l'exploitation des ressources halieutiques est bien souvent interdite (effort de pêche nul) ainsi en reprenant l'équation précédente, cela nous donne, pour une zone donnée ($z = 1$): $e_{1f}(t) = 0$ ainsi $u_1(t) = 0$ pour un intervalle de temps donné. Si l'activité de pêche, dans la zone étudiée, est de subsistance, l'effort sera alors reporté sur une zone adjacente ou proche pour favoriser une soutenabilité socio-économique des populations humaines ainsi $u_z(t) > 0$ pour $z \neq 1$ et un intervalle de temps donné. Dans certains cas, ce type de report peut parfois aussi se traduire par des impacts écologiques et socio-économiques défavorables (Soulié and Thébaud, 2006).

Par ailleurs, au sein de ces AMPs, la pêche n'est pas nécessairement totalement interdite mais peut également être juste règlementée (diminution de l'effort, protection de certaines espèces, fermeture temporaire à court, moyen ou long terme, interdiction de certaines techniques de pêche, etc...).

2.5 Stratégies normatives de pêche

Tandis que les scénarios exploratoires nous indiquent certaines tendances d'évolution de l'écosystème, les scénarios normatifs, en revanche, sont plus précis pour obtenir des niveaux d'effort satisfaisant les contraintes écologico-économiques ou atteignant les objectifs recherchés.

Dans ce cadre normatif, on fixe un horizon temporel et on tente d'identifier les combinaisons optimales d'effort atteignant les objectifs à chaque pas de temps, à chaque période ou à l'issue de la période de scénarisation. Pour identifier de tels scénarios normatifs, on peut s'appuyer sur la théorie du contrôle optimal ou viable. Dans le cadre d'incertitude stochastique, on parle généralement de contrôle stochastique (Doyen et al., 2017). Dans l'esprit du MEY escompté (équation (I.13)), le problème de contrôle stochastique peut généralement se formaliser de la manière suivante (Doyen and De Lara, 2010; Cissé et al., 2013; Gourguet et al., 2013; Lagarde et al., 2018, 2020)

$$\max_{e(t_h+1), \dots, e(t_n)} \mathbb{E} \left[\sum_{t=t_h+1}^{t_n} W(t, x(t), e(t), \omega(t)) \right] \quad (\text{I.24})$$

où $W(t, x(t), e(t), \omega(t))$ peut être une fonction d'utilité, de production (MMSY) ou de profits (MMEY) qui dépend du temps t , d'un état de l'écosystème $x(t)$, d'un effort de pêche $e(t)$ et de scénarios stochastiques $\omega(t)$. Les efforts $e(t_h + 1), \dots, e(t_n)$ sont ainsi les niveaux optimaux maximisant l'objectif/la fonction à chaque pas de temps à partir de l'année suivant la dernière année historique notée t_h jusqu'à la fin de l'horizon temporel i.e la dernière année de simulation notée t_n .

Éco-viabilité

Par ailleurs, des contraintes biologiques et économiques peuvent être fixées discutant ainsi du concept de viabilité économique-écologique ou d'éco-viabilité (Doyen and De Lara, 2010; Doyen et al., 2012; Gourguet et al., 2013; Cissé et al., 2015). Les indicateurs de viabilité préalablement introduits dans la section 2.2 sont déterminants.

⁶A condition que la connaissance de la distribution spatiale des biomasses des espèces impactées par la pêche permette de calibrer de tels modèles, et sous réserve de pouvoir efficacement réguler la distribution spatiale des activités de pêche.

Dans un contexte où les incertitudes sont de nature probabiliste, les scénarios et stratégies d'éco-viabilité ou de viabilité écologico-économique peuvent être définis comme la satisfaction de plusieurs contraintes avec une probabilité suffisamment élevée (Doyen and De Lara, 2010) telle que:

$$\begin{aligned} \mathbb{P}(\text{Contraintes (I.21), (I.22), (I.23) sont satisfaites pour } t = t_h + 1, \dots, t_n) \\ = \mathbb{E} \left[\prod_{t=t_h+1}^{t_n} \mathbb{1}_C(x(t), e(t), \omega(t)) \right] > \beta \end{aligned} \quad (\text{I.25})$$

où β correspond à des taux de confiance préalablement déterminés (99%, 90%, ...) et avec la fonction indicatrice étant:

$$\mathbb{1}_C(x, e, \omega) = \begin{cases} 1 & \text{si Contraintes (I.21), (I.22), (I.23) sont satisfaites} \\ 0 & \text{sinon} \end{cases}$$

La probabilité \mathbb{P} est alors calculée par rapport à l'incertitude ω qui résume la stochasticité sur la dynamique de l'écosystème (croissance, interactions entre espèces), la capturabilité (ou les interactions techniques), les coûts ou les prix. Plus globalement, la probabilité d'éco-viabilité peut inclure plusieurs contraintes, il en revient au modélisateur de choisir le nombre de contraintes bioéconomiques qu'il souhaite intégrer. La Figure I.9 montre schématiquement comment sont représentés les résultats de viabilité.

La stratégie de gestion d'éco-viabilité peut se formaliser via la maximisation de sa probabilité décrite à l'équation précédente (éq. (I.25)) telle que:

$$\max_{e(t_h+1), \dots, e(t_n)} \mathbb{P}(\text{Contraintes (I.21), (I.22), (I.23) sont satisfaites pour } t = t_h + 1, \dots, t_n) \quad (\text{I.26})$$

Les efforts $e(t_h + 1), \dots, e(t_n)$ sont ainsi les niveaux d'effort viables maximisant la probabilité de respecter les contraintes écologico-économiques par rapport aux scénarios stochastiques $\omega(t)$. Il est ici nécessaire de considérer que les stratégies d'éco-viabilité stochastiques reposent sur le "feedback control" c'est à dire un contrôle rétroactif (Doyen and De Lara, 2010).

2.6 Questions de recherche autour de l'EBFM

En prenant en compte des critères écologiques, économiques et sociaux liés respectivement au maintien de la biodiversité, de la viabilité économique de la pêche, et à la sécurité alimentaire, cette thèse vise à identifier des stratégies de gestion de pêche permettant d'assurer la soutenabilité et la résilience des pêcheries marines. En prenant en compte la complexité écologique et économique liée à l'exploitation des écosystèmes marins, cette thèse vise également à identifier des stratégies de gestion de pêche en lien avec l'EBFM.

Dans cette perspective générale, cette thèse pluridisciplinaire apporte des éclairages aux questions de recherche suivantes:

- Quel niveau de complexité écologique est nécessaire pour opérationnaliser l'EBFM?
- Quel niveau de complexité socio-économique est nécessaire pour opérationnaliser l'EBFM?
- Comment modéliser et gérer l'incertitude liée aux changements globaux ?
- Quelles stratégies de pêche favorisent la viabilité, la soutenabilité et la résilience des socio-écosystèmes pour l'EBFM ?

Cette thèse s'appuie sur 2 cas d'études contrastés: les pêcheries mixtes démersales du Golfe de Gascogne et la pêche artisanale côtière et de subsistance de l'île de Moorea. Malgré la différence entre les caractéristiques socio-économiques et écologiques de ces deux études de cas, la discussion des résultats identifie des parallèles entre les zones étudiées. Cela permet d'en extraire des idées sur la gestion d'autres socio-écosystèmes non-traités ici et ouvre la porte à de futures recherches qui permettront (grâce au développement technologique et à un plus grand nombre de données fiables et disponibles) une amélioration des résultats, des prévisions et des modèles de gestion.

3. Synthèse des travaux

3.1 La stratégie multi-espèces MEY (MMEY) atténue les effets du changement climatique dans les pêcheries mixtes

Les travaux de cette sous-section sont inspirés de l'article, publié dans *Ecological Economics, How Does MMEY Mitigate the Bioeconomic Effects of Climate Change for Mixed Fisheries*¹. Les détails de cet article sont donnés dans le Chapitre II. Ils concernent les cas d'études des pêcheries mixtes du Golfe de Gascogne (zones VIIIa et VIIIb - Figure I.10) dans un cadre de changement climatique. Dans un contexte de surexploitation du stock de sole, d'une excellente santé du stock de merlu (ICES, 2019) et d'une forte augmentation des températures de la surface de la mer (*Sea Surface Temperature* - SST - Opec²), cet article explore la capacité des stratégies de gestion de pêche multi-espèces à faire face à ces changements climatiques. Pour ce faire, un modèle dynamique, stochastique, multi-espèce, multi-classe d'âge et multi-flottille est développé et calibré à partir des données biologiques, économiques et environnementales pour les flottilles françaises³. Pour chaque espèce, la dynamique des populations, décrite sur une base annuelle par classe d'âge, est d'abord caractérisée par des mécanismes de mortalité naturelle et de pêche (Quinn and Deriso, 1999; Gourguet et al., 2013) comme suit:

$$x_{i,a}(t+1) = x_{i,a-1}(t) \exp(-M_{i,a-1} - F_{i,a-1}(t)) \quad (\text{I.27})$$

où $x_{i,a}(t)$ représente l'abondance des espèces exploitées $i = 1, 2$ (sole, merlu respectivement) à l'âge $a = 2, \dots, A_i$ en t . Par ailleurs, nous supposons qu'il n'existe pas d'interactions biologiques entre la sole et le merlu. Ainsi, l'abondance des espèces $x_{i,a}(t)$ évolue en fonction de la mortalité naturelle $M_{i,a}$ et de la mortalité totale $F_{i,a}(t)$ de l'espèce i à l'âge a en t .

L'incertitude climatique est ici représentée par les SST et impacte le recrutement des espèces. Basée sur le modèle de Ricker (1954) auquel un paramètre environnemental θ est ajouté, la dynamique de recrutement se définit comme suit:

$$x_{i,1}(t+1) = f(SSB_i(t - \tau_i), \theta(t - \tau_i), \varepsilon_i(t - \tau_i)) \quad (\text{I.28})$$

où $SSB_i(t)$ représente le stock fécond/mature sexuellement (**Spawning Stock Biomass** - SSB) de l'espèce i en t , $\theta(t)$ les SST en t et $\varepsilon_i(t)$ le bruit i.i.d en t . Ces 3 variables dépendent d'un facteur de décalage temporel τ_i indexé sur le temps nécessaire à un oeuf de l'espèce i pour devenir une recrue capturable.

Concernant la partie socio-économique, les captures sont fonctions de l'effort⁴, de l'abondance et de la mortalité (Equation des captures de Baranov, 1925) comme suit:

$$H_{i,a,f}(u_f(t)) = x_{i,a}(t)u_f(t)F_{i,a,f}(t) \frac{1 - \exp(-M_{i,a} - \sum_{f=1}^m u_f(t)F_{i,a,f}(t))}{M_{i,a} + \sum_{f=1}^m u_f(t)F_{i,a,f}(t)} \quad (\text{I.29})$$

Les profits $\pi_f(t)$ (Dérivation de l'équation (I.6) et (I.20)- détaillés au complet dans le Chap III) dépendent quant à eux de facteurs socio-économiques fixes et stochastiques (carburant).

¹ <https://doi.org/10.1016/j.ecolecon.2018.07.001>

² Operational Ecology (End date: 31/12/2014) - <http://marine-opec.eu/>

³ Données collectées via le Système d'Information Halieutique (SIH) de l'IFREMER et de la direction française de la pêche et de l'aquaculture (DPMA) via le cadre européen de collecte des données (DCF).

⁴ On applique un multiplicateur d'effort noté $u_f(t)$ à l'effort $e_f(t)$. Les explications sont détaillées dans le chapitre II.

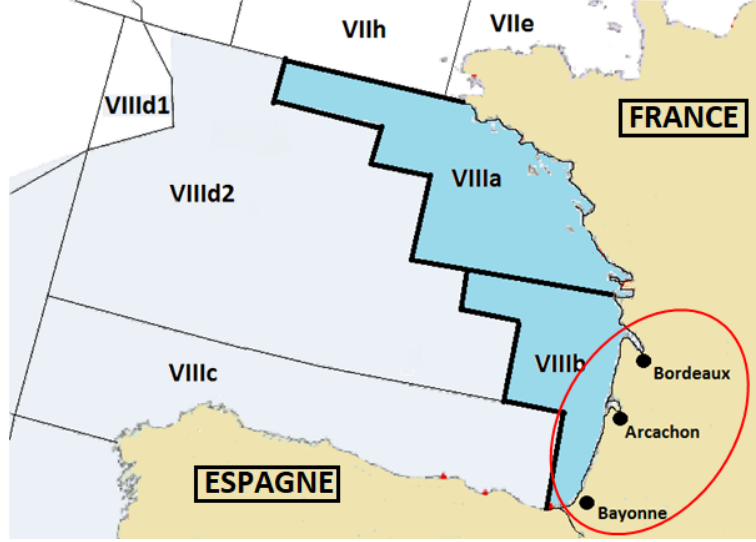


Figure I.10: Carte du Golfe de Gascogne et des divisions réalisées par le CIEM. La zone d'étude est en bleue.

Deux scénarios d'émission de gaz à effet de serre (RCP⁵ 2.6 et RCP 8.5) basés sur le 5ème rapport mené par le Groupe Intergouvernemental sur l'Etude du Climat (GIEC - *Intergovernmental Panel on Climate Change* - IPCC) en 2013, nous indiquent deux évolutions contrastées et probables des SST à l'horizon 2100.

Grace à ces modèles écologico-économiques et selon les scénarios climatiques, trois stratégies de gestion sont ensuite comparées en termes de performances écologico-économiques: le Status-Quo (SQ), le Multi-species Maximum Sustainable Yield (MMSY) and le Multi-species Maximum Economic Yield (MMEY). La stratégie MMEY se définit à partir de la valeur actuelle nette (*Net Present Value* - NPV) telle que:

$$NPV(e) = \mathbb{E} \left[\sum_{t=t_0}^{t_n} \delta^t \sum_{f=1}^m \pi_f(t) \right] \quad (\text{I.30})$$

où $\delta = \frac{1}{1.04}$ est le facteur d'escompte, t_0 l'année de base, t_n la dernière année de simulation, m le nombre de flottilles et $\pi_f(t)$ la fonction de profits détaillée au Chapitre III. Si l'on maximise la valeur actuelle nette comme dans l'équation (I.14), nous obtenons l'effort optimal e^{MMEY} au MMEY maximisant les profits escomptés et agrégés dans le temps tels que:

$$NPV(e^{MMEY}) = \max_e NPV(e) \quad (\text{I.31})$$

La Figure I.11 montre les trajectoires biologiques (SSB de la sole et du merlu) et économiques (Profits) de la stratégie optimale MMEY tandis que la Figure (II.9) synthétise le classement de ces stratégies en intégrant la dimension écologique (Index de Simpson) et économique (NPV).

Les résultats montrent que la stratégie SQ n'est pas soutenable et est caractérisée par un déclin majeur de l'espèce commerciale clef (sole). En revanche, la stratégie MMSY améliore l'état écologique de l'écosystème et les performances économiques de la pêche. La stratégie MMEY produit même de meilleurs résultats écologico-économiques. Ces bénéfices bioéconomiques sont cependant altérés et atténués par les effets du changement climatique. En effet, le stock de sole diminue avec l'augmentation des SST et affiche même, sous le pire scénario climatique, un SSB en dessous de son seuil de précaution à la fin de l'horizon temporel. Dans le cadre de la meilleure stratégie MMEY, les flottilles dotées de structures de capture plus diversifiées affichent

⁵Representative Concentration Pathways - Chemins de Concentration Représentatives.

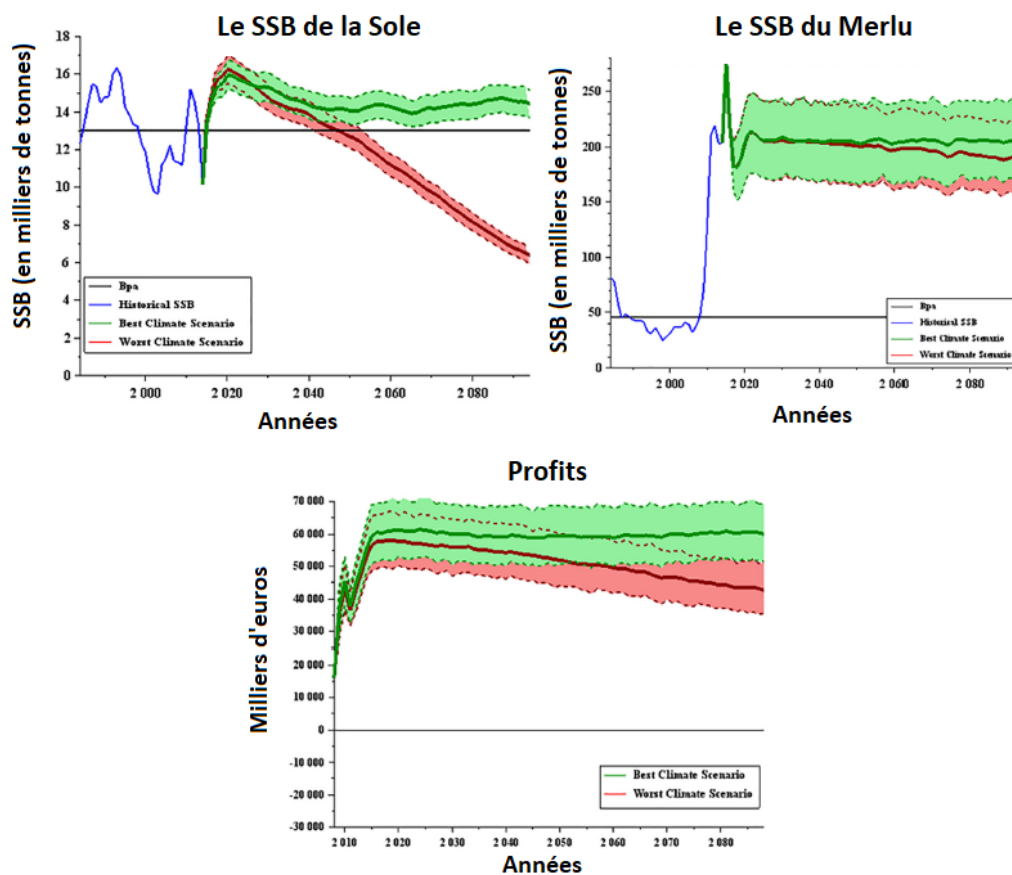


Figure I.11: Stratégie MMEY - Les trajectoires de SSB de la sole (en haut à gauche) et du merlu (en haut à droite) sont représentées pour les deux scénarios climatiques en milliers de tonnes. La ligne noire continue représente le seuil de précaution (du CIEM - Bpa) du stock de l'espèce. La troisième figure (en bas) représente les profits totaux de l'ensemble des treize sous-flottes. Les données historiques sont affichés en bleu. Les 500 trajectoires simulées sont représentées par les zones colorées en fonction du scénario climatique (vert: meilleur / RCP 2.6 et rouge: pire / RCP 8.5). Les courbes en gras dans les zones vertes et rouges affichent la moyenne de ces 500 trajectoires.

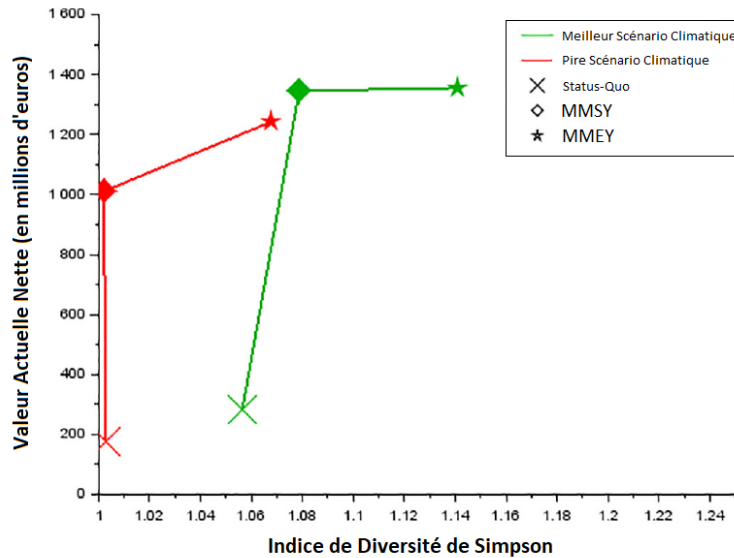


Figure I.12: L'Index de Diversité de Simpson face à la Valeur Actuelle Nette (Pour les deux espèces, sole et merlu).

de meilleurs résultats face au changement climatique. Cela s'explique par leur plus grande flexibilité de métiers de pêche. Lorsque l'activité d'une flottille, spécialisée dans la capture d'une unique espèce, voit le stock de sa cible diminué en dessous de son seuil de précaution, il est alors très difficile pour les bateaux de cette flottille de s'adapter aux nouvelles réglementations/quotas. À l'inverse, les petits bateaux utilisant déjà 2 à 3 outils de pêche différents par an, sont plus à même de répondre à l'impact de la variabilité climatique sur les stocks halieutiques en changeant régulièrement de cible, cela est décrit plus en détails dans le chapitre II et ses Appendices.

L'approche écosystémique de cet article montre la compréhension accrue acquise via une approche plus intégrée, qui devrait déboucher sur la mise en œuvre de stratégies de gestion plus intégrées. Concernant l'aide à la gestion, cet article montre que l'approche multi-spécifique produit de meilleurs résultats que les approches mono-spécifiques classiques et nous permet de faire face à l'incertitude du changement climatique dans le cas des grandes pêcheries industrielles.

3.2 La résilience écologico-économique d'un récif corallien exploité favorisé par une stratégie de pêche stochastique

Soumis à la revue *Environmental Modelling and Assessment*, l'article nommé *Ecological-economic resilience of a fished coral reef through stochastic multi-species MSY*⁶ explore, à travers différentes stratégies de pêche, la résilience écologico-économique d'un écosystème corallien soumis à des pressions environnementales (changement climatique, chocs externes i.e cyclones, espèces invasives) et anthropiques (pêche, tourisme...). Les détails de ce papier sont précisés dans le Chapitre III.

Dans le cas de petites pêcheries (récifs coralliens notamment) telles que celle de Moorea (Polynésie Française), une approche stochastique multi-espèce (poissons, habitats) et multi-flottille demeure largement pertinente pour opérationnaliser l'EBFM. Les récifs coralliens et en particulier celui de Moorea comptent parmi les écosystèmes les plus complexes au monde. Ils font face à de multiples pressions (anthropiques et environnementales) qui mettent en danger leur structure dont dépendent fortement les populations insulaires. Heureusement caractérisé par une densité et une croissance démographique relativement faible (comparativement à d'autres

⁶ <http://bordeauxeconomicswp.u-bordeaux.fr/2020/2020-11.pdf>

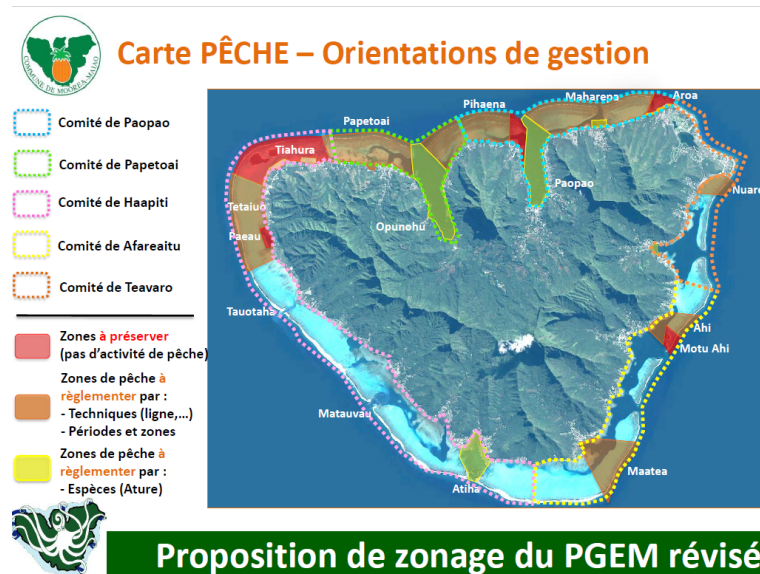


Figure I.13: Carte de Moorea divisée en comités de gestion de pêche selon le nouveau Plan de Gestion de l’Espace Maritime (PGEM) débutant en 2019-2020.

petites pêcheries coralliennes à travers le monde), le récif de Moorea est en meilleure santé que d’autres écosystèmes coralliens à travers le monde (Mumby et al., 2007; Galzin et al., 2016). Sa forte résilience (montrée à travers la récupération très rapide de la couverture de corail après l’invasion de COTS⁷ et d’un cyclone) lui permet pour l’instant de demeurer très diversifié en termes d’espèces et fonctions écosystémiques (Lamy et al., 2016; Dubois et al., 2017, données non-publiées). La mise en place d’un nouveau plan de gestion plus juste socialement et moins coercitif (réouverture de nombreuses AMP) atteste d’un état écologique satisfaisant à l’inverse de plein d’autres petites pêcheries coralliennes à travers le monde (e.g Caraïbes) où une dangereuse homogénéisation du milieu dominé par les algues se fait ressentir (Mumby, 2006; 2007; 2009).

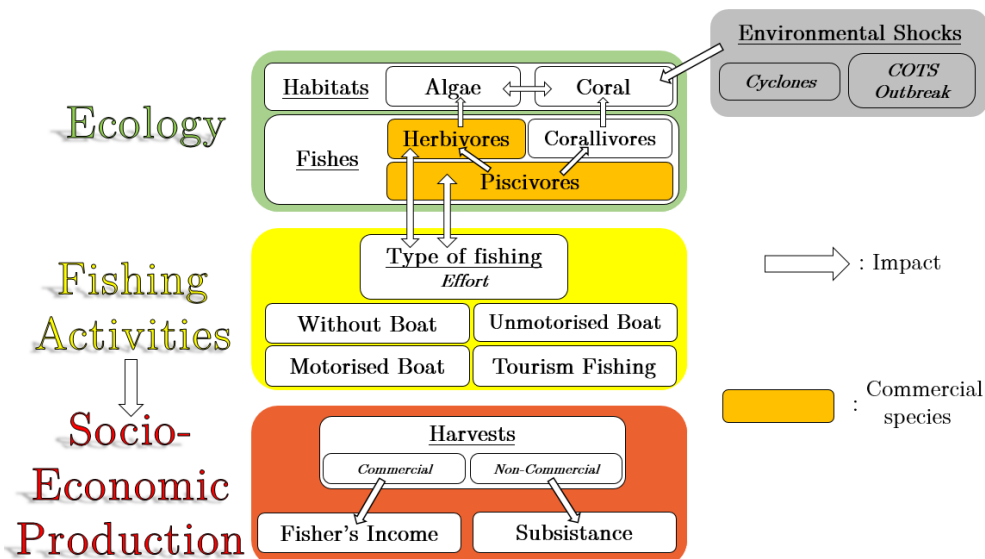


Figure I.14: Conceptual Model

Plus de 200 espèces de substrats (corail, algues) et de poissons vivent parmi le récif (Galzin et al., 2016; Moritz et al., 2018). Elles ont été regroupées en 5 groupes fonctionnels (Corail,

⁷ *Crown-of-Thorns Starfish* - Etoiles de Mer dévoreuses de Corail.

Algues, Corallivores, Herbivores et Piscivores). Ainsi, nous avons développé et implémenté un modèle (inspiré du modèle Lotka-Volterra - Lotka, 1927; Volterra, 1927; Mumby, 2006) dynamique, spatialement explicite (Deux patches: Lagon et Pente Externe⁸), multi-spécifique, multi-flottille. Le modèle conceptuel est explicité par la Figure I.14, tandis que le modèle mathématique se définit comme suit:

$$\widetilde{x}_{ip}(t+1) = x_{ip}(t) \left(1 + r_i + \sum_j s_{ij} \gamma_p(x_{1,p}(t)) x_{jp}(t) \right) (1 - \Delta r_p(\omega(t))) - H_{ip}(t) \quad (\text{I.32})$$

où $x_{ip}(t)$ est la densité du groupe i dans le patch p , r_i le taux de croissance intrinsèque du groupe i , s_{ij} le coefficient d'interaction trophique entre les groupes i et j ($s_{ij} > 0$ si i est prédateur de j , $s_{ij} < 0$ est négatif si i est une proie de j , $s_{ij} = 0$ est nul lorsqu'il n'existe aucune interaction directe entre les espèces i et j)⁹ et $\widetilde{x}_{ip}(t)$ représente la densité du groupe i dans le patch p sans la diffusion inter-patches des espèces mobiles. L'effet refuge $\gamma_p(x_{1,p}(t))$ correspond au poids illustrant l'intensité de prédation. Il est linéaire à la proportion de l'habitat corallien $x_{1,p}(t)$ du patch p permettant aux proies de se cacher et où il est plus difficile pour les prédateurs de se nourrir ($\gamma_p(x_{1,p}(t)) \geq 0$). $\Delta r_p(\omega(t))$ représente les perturbations stochastiques environnementales (cyclones, invasion d'étoiles de mer dévoreuse de corail) qui impactent la couverture de corail sur la pente externe et dépendent de scénarios stochastiques environnementaux $\omega(t)$. $H_{ip}(t)$ représente la somme des captures de toutes les flottilles f pour une espèce i dans un patch p en t détaillée par l'équation (I.3) avec $H_{ip}(t) = \sum_f H_{ifp}(t)$.

Un taux de diffusion $d_{ip'p}$ (Proportion du groupe fonctionnel i allant d'un patch p' au patch p avec $d_{ip'p} \in [0, 1]$) est intégré (uniquement pour les espèces mobiles i.e Herbivores et Piscivores), afin de rendre le modèle spatialement explicite tel que:

$$x_{ip}(t+1) = \sum_{p'=lagon}^{fore\ reef} d_{ip'p} \widetilde{x}_{ip'}(t+1) \quad (\text{I.33})$$

Les captures sont déterminées par une équation dérivée de celle présentée plus tôt (éq. (I.3)) telle que:

$$H_{ifp}(t) = q_{ifp} e_{fp}(t) L_f(t) x_{ip}(t) \quad i = 3 \text{ (herbivores)}, 4 \text{ (piscivores)} \quad (\text{I.34})$$

où le paramètre q_{ifp} mesure la capturabilité du groupe i pour chaque flottille ou type de pêche f dans un patch p . L'effort de pêche $e_{fp}(t)$ est le nombre d'heures par an passées par un pêcheur de la flottille f dans le patch p tandis que $L_f(t)$ correspond à la main d'oeuvre engagée dans la flottille f (exprimée en nombre de pêcheurs par km²).

Quatre scénarios exploratoires (SQ, Closure¹⁰, Deux SpE¹¹) et un scénario normatif SMMSY sont comparés sur la base d'indicateurs écologico-économiques attestant de l'état et de la résilience écologico-économique de l'écosystème.

Optant pour une approche stochastique "non-anticipative" en boucle fermée (i.e contrôle de rétroaction) de Shapiro et al. (2009); Doyen and De Lara (2010), la stratégie SMMSY consiste en la maximisation de la valeur attendue de la somme des débarquements au fil du temps et cela afin d'obtenir un effort optimal e_0 et e_1 (fonction des deux périodes de contrôle et de l'état du système) satisfaisant l'objectif comme suit:

⁸ Fore Reef.

⁹Par ailleurs, si $s_{ij} < 0$ et $s_{ji} < 0$ pour un i et j donné, on ne parle plus de prédation mais de compétition (e.g Corail et Algues sont en compétition pour la colonisation des substrats), à l'inverse si $s_{ij} > 0$ et $s_{ji} > 0$ pour un i et j donné, on parle de mutualisation (e.g cleaner fish).

¹⁰Fermeture pure et simple et des zones exploitables i.e l'effort de pêche est fixé à zéro.

¹¹Le premier scénario spatialement explicite ferme la pente externe et double l'effort du lagon, et le deuxième fait l'inverse.

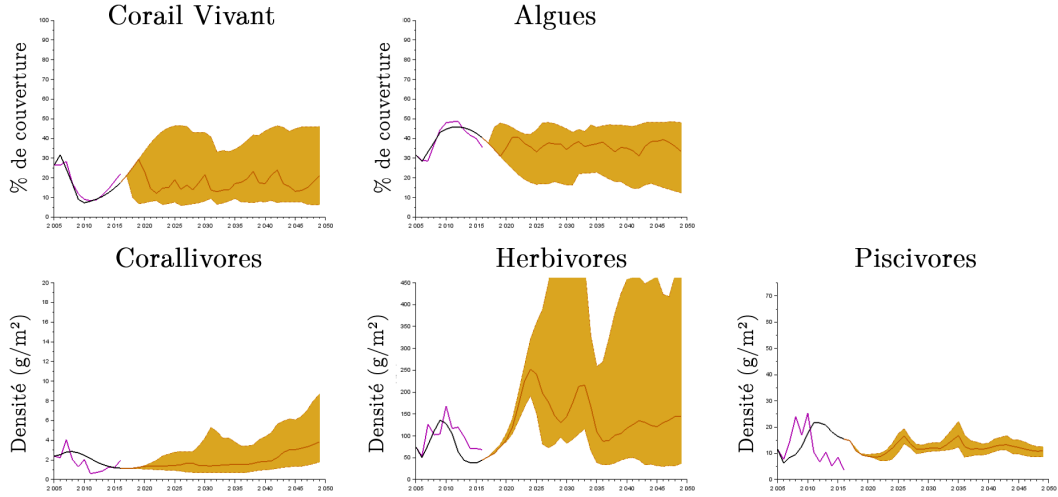


Figure I.15: Stratégie SMMSY - Trajectoires écologiques - Les cinq figures représentent les pourcentages de couverture des deux habitats et les densités agrégées (Lagoon - Fore Reef) des trois groupes fonctionnels de poissons. Les 100 trajectoires simulées sont représentées par la zone orangée. Les courbes au sein des champs colorés indiquent la moyenne de ces 100 trajectoires tandis que les courbes violettes attestent des trajectoires historiques et les courbes noires de la calibration.

$$\max_{e(\dots)} \mathbb{E} \left[\sum_{t=t_h+1}^{t_n} \left(\sum_i \sum_f \sum_p H_{i,f,p}(t) \right) \right] \quad (\text{I.35})$$

où t_h est la dernière année historique observée dont nous avons des données et t_n l'horizon temporel.

La stratégie SMMSY favorise la résilience écologico-économique. Cela est mis en lumière par la Figure I.16 où elle affiche la plus grande surface selon les différents indicateurs écologico-économiques calculés. En effet, grâce à la stratégie SMMSY, le système **résiste** et conserve son identité fonctionnelle et sa viabilité ; il est également **robuste** car il ne descend pas en dessous d'un seuil critique où un groupe fonctionnel pourrait s'éteindre ; et son **temps de récupération** est très faible concernant le corail et nul pour les herbivores et piscivores. Sa dynamique de densité de poisson est montrée par la Figure I.15.

Elle se caractérise par une augmentation globale de l'effort de pêche sur toutes les zones soulignant ainsi l'actuelle sous-exploitation de la pêcherie. Il faut néanmoins préciser que l'incertitude climatique croissante ainsi que l'éventuel biais existant parmi les données pourraient nuancer de tels résultats. De plus, la stratégie SMMSY rend soutenable le niveau trophique de l'écosystème en maintenant la fonction essentielle de broutage des petits herbivores malgré les chocs environnementaux. La stratégie SMMSY est également plus diversifiée en termes de temporalités, d'équipements, de zones et de cibles.

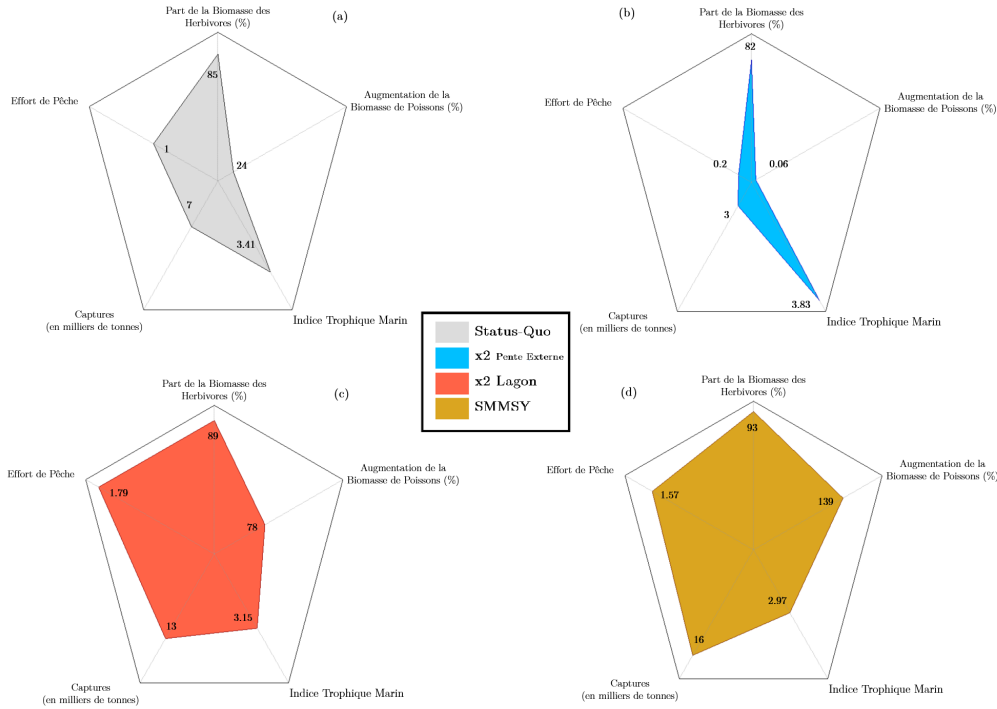


Figure I.16: Diagramme en radar - Les quatre radars sont colorés selon les stratégies de gestion et les cinq échelles des composants (SHB: 0-100%, Fishing Effort: 0-2, Catches: 0-20, Marine Trophic Index: 0-4, Fish Biomass Increase: 0-200%). Les nombres dans les radars représentent la moyenne des 100 simulations liées à chaque composant correspondant. Les 5 composants sont décrits en appendices au Chap. III.

3.3 Les stratégies de viabilité de pêche pour une résilience écolo-économique des récifs coralliens face au changement climatique

Dans la synthèse de ce troisième et dernier chapitre (Chap IV), nous reprenons le modèle stochastique, multi-espèce et multi-flottille présenté dans la section précédente. Nous nous concentrons encore une fois sur l'île de Moorea (Polynésie Française) dont les caractéristiques sont également présentées plus haut, mais nous utilisons dans cet article l'approche de viabilité (Aubin, 1991).

L'article nommé *To what extent fishing Viability strategies promote ecological-economic Resilience of coral reef facing climate change*, examine, à travers différentes stratégies de gestion de pêche, le lien existant entre la viabilité d'un récif corallien et sa résilience. Plus précisément, il s'attache à comparer une stratégie exploratoire (Status-Quo) avec une stratégie préalablement déterminée mais normative (SMMSY) et également avec trois autres stratégies de viabilité (la première est écologique: PVA, la deuxième est d'ordre socio-économique: EVA, et la troisième englobe ces deux dimensions et est donc écolo-socio-économique: Eco-Viabilité ou EcoVA). Ces stratégies de gestion de pêche sont comparées en termes de résultats écologiques, socio-économiques, de viabilité et de résilience. Afin d'analyser la réponse du socio-écosystème et des modèles de gestion face au changement climatique, quatre scénarios d'occurrence de choc environnementaux sont implémentés. A l'instar des observations historiques (COTS - 2006-2009, Cyclone OLI - 2010), ces chocs sont supposés affecter uniquement la couverture de corail sur la pente externe. Nous implémentons donc quatre scénarios disposant d'une probabilité qu'un choc survienne chaque année (10%, 20%, 30% et 50%).

Ces scénarios de choc sont modélisés par la variable $\Delta r_p(\omega(t))$ dans l'équation (I.32) et

dépendent des quatre scénarios stochastiques environnementaux représentés avec la variable $\omega(t)$. Cette dernière variable fluctue donc en occurrence mais demeure la même en terme d'intensité (perte de 70% de couverture de corail sur la pente externe lorsqu'un choc survient).

Afin d'analyser les stratégies en terme de viabilité, nous nous inspirons des travaux de Aubin (1991); Doyen et al. (2017) et nous implémentons 3 contraintes écologiques et une contrainte socio-économique tel qu'il existe un état viable du système dans lequel ces seuils sont respectés à chaque pas de temps. Ainsi les contraintes écologiques sont:

$$x_{coral}(t) \geq x_{coral}^{lim} \quad \forall t \quad (I.36)$$

$$x_{herb}(t) \geq x_{herb}^{lim} \quad \forall t \quad (I.37)$$

$$x_{pisc}(t) \geq x_{pisc}^{lim} \quad \forall t \quad (I.38)$$

et les seuils socio-économiques pour les différentes flottilles sont définis tels que:

$$H_f(t) \geq H_f^{lim} \quad for f \geq 6 \quad \forall t \quad (I.39)$$

Grace à ces contraintes x_{coral}^{lim} , x_{herb}^{lim} , x_{pisc}^{lim} , H_f^{lim} , nous analysons si pour chacune des simulations les densités $x_{coral}(t)$, $x_{herb}(t)$, $x_{pisc}(t)$ et les captures par pêcheur $H_f(t)$ ¹² demeurent au dessus des seuils. Cela nous indique ainsi la probabilité que chaque stratégie respecte ces contraintes à chaque pas de temps (année) t . Mathématiquement, la probabilité (nous prenons l'exemple de l'éco-viabilité i.e la viabilité écologico-socio-économique¹³) cela se traduit comme suit:

$$\begin{aligned} & \mathbb{P}(Ctrs (I.36) (I.37) (I.38) (I.39) \text{ sont satisfaites pour } t = t_h + 1, \dots, t_n) \\ &= \mathbb{E} \left[\prod_{t=t_h+1}^{t_n} \mathbb{1}_{EcoVA}(x(t), e(t), \omega(t)) \right] \end{aligned} \quad (I.40)$$

où t_h est la dernière année historique observée dont nous avons des données et t_n l'horizon temporel et avec la fonction indicatrice étant:

$$\mathbb{1}_{EcoVA}(x, e, \omega) = \begin{cases} 1 & \text{si } Ctrs (I.36) (I.37) (I.38) (I.39) \text{ sont satisfaites} \\ 0 & \text{sinon} \end{cases}$$

Les stratégies SQ et SMMSY sont les mêmes que celles présentées dans la section 3.2. Les stratégies de viabilité, quant à elle, sont déterminées par la maximisation des probabilités de viabilité (nous présentons ici seulement la viabilité écologico-socio-économique i.e l'éco-viabilité - EcoVA) comme suit:

$$\mathbb{P}_{EcoVA}(e^{EcoVA}) = \max_e \mathbb{P}_{EcoVA}(e)$$

En termes de viabilité et de résilience et pour tous les scénarios de choc (10%, 20%, 30%, 50%), les deux meilleures stratégies sont celles de la viabilité écologique (PVA) et de l'éco-viabilité (EcoVA) ; analyse détaillée au Chap IV, section 4. De plus, les gains de l'éco-viabilité par rapport au SQ sont de plus en plus significatifs par rapport à l'amplitude des chocs. Cette analyse est évidente lorsque l'on observe les probabilités d'éco-viabilité atteignant 100% pour chaque scénario de choc (Table I.3) à l'inverse de celles correspondant au SQ qui diminuent lorsque la fréquence des chocs augmentent.

¹²Les détails du calcul des captures par pêcheur sont expliqués dans le Chapitre IV, section 3.4.2.

¹³Le calcul des probabilités écologiques et socio-économiques seules est détaillé dans le chapitre IV section 3.4.

Stratégies de pêche	Probabilité d'éco-viabilité (en %)			
	p=10%	p=20%	p=30%	p=50%
SQ	50	45	22	0
SMMSY	92	79	90	100
PVA	100	100	100	100
EVA	86	96	92	75
EcoVA	100	100	100	100

Table I.3: Probabilités d'éco-viabilité selon la stratégie de pêche et pour tous les scénarios de choc: p=10%, p=20%, p=30%, p=50%.

Shock Occurrence	Strategies	Specialization	Diversification	Resilience
10%	PVA	\	Spatial, Metier, Target	Maintained
	EcoVA	\	Spatial, Metier, Target	Maintained
20%	PVA	\	Spatial, Metier, Target	Maintained
	EcoVA	Metier (Motorised Boats) Target (Piscivores)	Spatial	Maintained
30%	PVA	Metier (Unmotorised Boats)	Spatial, Target	Not Maintained (High Recovery Time)
	EcoVA	\	Spatial, Metier, Target	Not Maintained (High Recovery Time)
50%	PVA	Target (Piscivores)	Spatial, Metier	Not Maintained (Infinite Recovery Time)
	EcoVA	Metier (Motorised Boats) Target (Piscivores)	Spatial	Not Maintained (Infinite Recovery Time)

Table I.4: Type de strategies utilisées par les scénarios de pêche PVA et EcoVA et les conséquences sur la résilience selon l'occurrence des chocs.

La composante écologique apparait ici la plus importante. En effet, l'éco-viabilité est d'abord induite par la viabilité écologique comme l'atteste la stratégie PVA qui demeure viable alors que la stratégie EVA ne l'est que pendant le scénario de choc le plus extrême.

Concernant la résilience, celle-ci est affectée notamment sur le temps de récupération de la couverture de corail qui tend vers l'infini pour les deux scénarios de choc les plus extrêmes (30%, 50%). La Table I.4 met en exergue les différentes stratégies d'effort de pêche de la PVA et de l'EcoVA et montre le maintien majoritaire d'une stratégie de diversification pour les trois scénarios les modérés de choc (10%, 20%). Une stratégie de spécialisation de cibles (piscivores) s'opère pour les stratégies PVA et EcoVA lors des deux scénarios de choc les plus extrêmes (30%, 50%) afin de palier à la perte de l'effet refuge engendrée par une couverture de corail très faible.

4. Discussion Générale

4.1 L'intérêt des stratégies de gestion multi-espèces pour l'EBFM

Alors que Clark (1973) montrait déjà les limites des modèles de gestion mono-spécifique, une meilleure connaissance des interactions trophiques et de la dimension écosystémique des pêcheries permet désormais de montrer que de tels modèles monospécifiques de gestion peuvent induire une diminution des stocks et de la biodiversité (Walters et al., 2005). Des points de référence et objectifs multi-espèces sont alors proposés afin d'intégrer les caractéristiques multi-dimensionnelles et écosystémiques des pêcheries (Moffitt et al., 2016). Cependant bien que ces nouveaux modèles génèrent de meilleurs résultats bioéconomiques, ils laissent également entrevoir quelques limites.

Avantages

Les stratégies de gestion multi-espèces que sont le MMSY ou le MMEY ou l'éco-viabilité permettent de prendre en compte les interactions techniques i.e les prises simultanées de différentes espèces par les flottilles (Punt et al., 2010). En effet, la capture jointe de multiples espèces est une caractéristique de nombreuses pêcheries mixtes impliquant des équipements de pêche non selectifs (e.g chalutiers mixtes - Gourguet et al., 2013). Par conséquent, cela induit des problèmes dans la définition de politiques de pêche optimales à caractère mono-spécifique (Anderson, 1975).

Ainsi, comme présenté dans le chapitre III, l'un des avantages de l'approche multi-espèce se caractérise par des efforts se redirigeant vers des espèces aux stocks plus abondants (MMSY) ou aux valeurs économiques plus élevées (MMEY). Cela permet une meilleure adaptation des modèles de gestion. En effet, dans le cas de deux espèces, l'effort de pêche au MMSY pourra par exemple diminuer l'effort de la moins abondante et augmenter celle dont la biomasse est plus élevée afin de compenser les captures perdues (par la diminution de l'effort sur la moins abondante). Dès lors, une stratégie multi-espèce détient implicitement des considérations de conservation écologique. En effet, il sera moins productif pour la stratégie MMSY de se focaliser sur une espèce peu abondante (donc souvent en-dessous de son seuil de précaution). Et il sera moins rentable pour la stratégie MMEY d'épuiser totalement le stock d'une espèce à forte valeur économique.

Plus globalement, l'approche multi-espèce semble plus en lien avec l'approche écosystémique car elle intègre déjà plusieurs composants et ne se focalise plus sur une espèce particulièrement mais sur toutes celles qui sont exploitées au sein de la pêcherie. De plus, dans le cas où il existe des interactions trophiques entre les espèces ciblées, les stratégies de gestion multi-espèces prendront implicitement en compte cela dans le calcul de l'effort ce qu'une stratégie mono-spécifique ne peut faire par définition.

Limites

Dans le cas de flottilles visant plusieurs espèces, la maximisation de la production totale pour ces dernières a montré qu'il était possible que certaines espèces soient surexploitées au profit d'autres stocks délaissés (Ricker, 1958; Legović and Geček, 2010). Dans le Golfe de Gascogne, par exemple, Guillen et al. (2013) a montré que la maximisation des captures totales mène à une surexploitation du merlu. Fort-heureusement, la recrudescence de cette espèce dans cette zone a semblé compenser ces conclusions (ICES, 2019). De plus, si nous prenons le cas d'une pêcherie où aucun stock exploité ne dispose d'un stock fécond 3 ou 4 fois supérieur à son seuil de précaution (tel que peut actuellement l'être le merlu dans le Golfe de Gascogne), les populations de poissons dotées d'un potentiel biotique relativement faible pourraient s'éteindre car les stratégies multi-espèces n'auraient d'autres choix que se focaliser sur toutes les espèces de la pêcherie. Ainsi dans le cas où les stratégies multi-espèces ne peuvent rediriger leurs efforts vers des stocks plus abondants ou plus rentables au sein de la pêcherie (e.g modèle mono-flottille), les populations

de poissons avec une faible productivité biotechnique et une faible valeur économique peuvent alors être surexploités ou même s'éteindre au MMSY ou MMEY (Tromeur and Doyen, 2019).

Economiquement, il est fondamental de préciser que le MMEY ou le MMSY ne prennent pas en compte la forte disparité pouvant exister à travers les flottilles dont certaines peuvent être moins flexibles en termes de cibles, plus âgées ou moins rentables. Par conséquent, une pêcherie productive (au maximum de sa capacité de production au MMSY) ou profitable (au dessus de son seuil de rentabilité au MMEY) ne veut pas nécessairement dire une pêcherie socialement équitable. Pour cette dimension, l'approche de viabilité expliquée plus tôt pourrait contraindre les efforts afin de satisfaire l'objectif d'équité implicite au développement durable.

L'avenir des modèles de gestion

Les modèles de gestion mono-spécifique (MSY, MEY) semblent tendre vers leur limite et la mise en place d'une gestion multi-espèce, multi-flottille et plus globalement écosystémique apparaît comme l'unique solution actuellement soutenable pour faire face aux pressions dites multi-drivers.

De grandes institutions telles que la FAO (Food and Agriculture Organization of the United Nations - Organisation des Nations unies pour l'alimentation et l'agriculture), le GIEC (Groupe Intergouvernemental sur l'Etude du Climat) ou encore l'IPBES (*Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* - Plateforme intergouvernementale scientifique et politique sur la biodiversité et les services écosystémiques) peuvent fournir aux états des tendances d'évolutions, des scénarios ou juste des idées sur les probables conséquences locales grâce à leur bases de données globales (bien que les degrés de précision soient encore extrêmement variables).

Comme montrée par cette thèse, la mise au point de scénarios prospectifs (i.e normatifs), permettant de comparer différentes alternatives de gestion aux objectifs écologico-économiques, peut permettre d'évaluer d'éventuels synergies et compromis entre rationalité économique et considération écologique. Cependant, la compréhension du fonctionnement écologique complexe des pêcheries est un des défis relevé par la communauté scientifique tout comme son évaluation¹. A la suite de quoi, la mise en place de modèles de gestion écosystémiques adaptés à chaque écosystème s'affiche comme le préalable inhérent à l'utilisation de ce premier défi.

4.2 L'intérêt des MICE écologico-économiques pour l'EBFM

Les modèles présentés dans cette thèse sont conformes aux modèles de complexité intermédiaire (MICE) préconisés dans Plagányi et al. (2014); Doyen et al. (2017) afin d'implémenter l'EBFM. En effet, les MICEs sont axés sur des questions spécifiques et visent à limiter la complexité en se focalisant uniquement sur les composants et les interactions minimales nécessaires pour traiter les principaux effets de la question de gestion considérée. En effet, il est actuellement difficile de véritablement intégrer tous les processus et interactions s'opérant au sein d'un écosystème (de plus au coeur d'un milieu aussi complexe que les récifs coralliens) ou d'un socio-écosystème ; d'une part car la connaissance de ces milieux n'est pas complète et d'autre part car le temps de calcul est dépendant de la puissance moyenne des ordinateurs².

¹Parmi les écologues et biologistes (affiliés au centre de recherche du CRILOBE dont la mission principale est l'étude du lagon et du récif) auxquels j'ai pu poser la question suivante: "L'écosystème de Moorea est-il en bonne santé ?", leurs réponses furent toutes différentes (Pers. Observation). Par ailleurs, la profusion d'indicateurs écosystémiques et la non-standardisation de ces derniers dans les recherches mondiales prouvent partiellement un manque de connaissance de l'état d'équilibre des milieux. La complexité des écosystèmes et leur évolution est telle, que les chercheurs peinent à comprendre certains mécanismes et sont surpris par de nombreux processus (e.g récupération incroyable de la couverture de corail de Moorea post-COTS-perturbation i.e après 2016).

²Bien que l'innovation technologique nous fournit des processeurs et des machines de plus en plus puissants, la généralisation et la moyennisation d'ordinateurs capables de réaliser des billions d'opérations à la seconde n'est pas encore d'actualité. La création d'ordinateur quantique (l'un des derniers nommé *Sycamore* a été fabriqué, en 2019, par un partenariat entre Google et la NASA qui assurent détenir la suprématie quantique - Arute et al.)

Les MICE sont donc plus ciblés que les modèles globaux de l'écosystème (*Global Ecosystem Model*), comme Atlantis (Fulton et al., 2011) ou Ecopath with Ecosim (EwE - Christensen and Walters, 2004) bien qu'ils intègrent la dynamique complexe de l'écosystème marin dans le cadre des changements mondiaux, les processus sous-jacents aux pêches et leurs interactions avec les services écosystémiques marins. Ainsi, ils demeurent suffisamment simples pour permettre une adaptation pratique et pour faciliter la communication entre les disciplines et les porteurs d'enjeux y compris les non-scientifiques. À cet égard, les modèles développés dans cette thèse intègrent les grandes complexités écologiques des écosystèmes (notamment coralliens) en s'articulant autour de la dynamique de l'habitat, des interactions trophiques et des perturbations environnementales. En mettant l'accent sur les résultats de la pêche, ils tiennent également compte des facteurs économiques, sociaux et des problèmes de gestion. Plus particulièrement, la comparaison entre les différentes stratégies de pêche et le Status-Quo donne un aperçu fondamental des améliorations potentielles de la situation actuelle, des états des socio-écosystèmes et de leur gestion.

Par rapport à d'autres secteurs, l'approche économique-écologique est particulièrement pertinente lorsque l'on s'intéresse aux ressources halieutiques et à leur exploitation. En effet, à l'inverse de certains secteurs (énergie, production, échanges commerciaux...) où il est (pour l'instant) difficile d'avoir une politique écologique rentable et socialement juste, les pêcheries, qu'elles soient grandes ou petites, nécessitent une gestion dotée de considération écologique afin d'obtenir des résultats économiques positifs sur le long terme (Hardin, 1968). Comme expliqué précédemment, la surexploitation des ressources halieutiques conduit inévitablement vers une dégradation des captures (en nombre et en taille - Pauly et al., 1998) et par extension des profits affaiblissant le secteur et précarisant les pêcheurs.

4.3 La gestion de l'incertitude pour l'EBFM

Qu'ils soient économiques, environnementaux ou climatiques, les facteurs d'incertitudes sont nombreux. Une grande partie de ces incertitudes est globale: la volatilité du marché (des carburants, des matières premières, etc...), les espèces invasives ou les catastrophes naturelles (cyclones, raz-de-marée, tremblement de terre, etc...) ainsi que les conséquences des émissions de gaz à effet de serre. À cette échelle globale, la gestion de ces incertitudes est difficile car un très petit nombre d'états ou d'institutions peuvent réellement avoir un impact global sur les conséquences locales. À cet égard, les solutions soutenables pour les décideurs locaux et leurs conseillers scientifiques sont très restreintes.

Une solution est alors de tenter de modéliser l'incertitude et ses conséquences localisées en se basant sur des facteurs historiques déjà observés au sein de leur socio-écosystème. Implémenter un intervalle de variations probables (ou *range*) des composants potentiellement stochastiques de l'écosystème afin d'en déterminer des modèles de gestion intégrés et adaptatifs est l'un des principaux apports de cette thèse. En des termes plus simples, évaluer une fourchette des possibles afin d'en déduire des solutions qui sont, dans le meilleur des cas, soutenables et dans le pire des cas, atténuants.

Par ailleurs, la gestion de l'incertitude est particulièrement importante lorsque l'on s'intéresse aux processus de résilience (Miller et al., 2010).

Il faut cependant préciser une chose fondamentalement importante. De par sa nature, l'intégration de l'incertitude au sein de modèles de gestion sous-entend des risques dans la mise en place concrète de ces derniers.

En effet, comme le souligne parfaitement Hill et al. (2007):

“les résultats des modèles doivent être présentés d'une manière qui présente les risques et contreparties associés aux actions alternatives et au degré d'incertitude des prédictions”.

uniquement utilisable pour certaines tâches spécifiques, donne l'espoir de voir un jour une modélisation ultra réaliste d'un socio-écosystème.

Ils continuent en statuant:

“[La] *présentation* [des résultats] *ne devrait pas déguiser le fait que, dans de nombreux cas, les estimations d’un modèle d’incertitude, peuvent être basées sur des critères subjectifs*”.

De l’honnêteté des résultats présentés et de la coopération entre les différentes disciplines (économie, sociologie, écologie, biologie, climatologie, mathématiques, psychologie...) dépendra donc l’amélioration des prévisions scientifiques et la réduction de l’incertitude.

4.4 La diversification écosystémique pour l’EBFM

Dans cette thèse, nous avons montré qu’il est possible d’améliorer les résultats socio-économiques d’une pêcherie tout en promouvant la biodiversité et ses fonctions.

L’une des réponses proposées est en ligne avec l’intuition globale et de nombreuses recherches menées précédemment qui est: Appliquer une stratégie de diversification afin de minimiser les risques futurs. Cela fait le parallèle avec la théorie du portefeuille (Markowitz, 2010) et l’hypothèse d’assurance en écologie (Yachi and Loreau, 1999).

L’outil de la modélisation (notamment à travers la modélisation de toutes les interactions écologico-économiques d’un écosystème et des processus d’optimisation) semble, en effet, nous guider vers ces stratégies de diversification plus adaptatives. Il semble très intuitif d’en arriver à cette conclusion. Des stratégies de gestion émergentes telles que le “balanced harvesting”³ (i.e capture équilibrée) se basant sur la productivité biologique des espèces remettent en cause les modèles de gestion uniquement sélectifs (Zhou et al., 2010; Garcia et al., 2012).

Alors que ces stratégies dites “balanced” font face à des critiques par leur manque de preuves pratiques et de difficultés dans leur implémentation (Burgess, 2015; Froese et al., 2016), cette thèse présente des résultats concrets de structures d’efforts de pêche qui nous guident effectivement vers des stratégies moins sélectives et dotées de méthodes plus diversifiées⁴ i.e des stratégies de diversification multi-dimensionnelle: **Une diversification spatiale, temporelle, d’équipements et de cibles.**

4.4.1 Une diversification spatiale

Concernant la **spatialité**, l’instauration de “rahui” (petites AMP dans les archipels polynésiens et indo-pacifiques - Royal, 2003; Maxwell and Penetito, 2007), nous montre un souhait croissant d’une gestion spatialement explicite notamment pour générer un effet refuge. Cette thèse va dans ce sens en intégrant la dimension de diffusion pour les espèces mobiles au sein des modèles de gestion. En demeurant à des niveaux de complexité intermédiaire, les modèles spatialement explicites sont également en ligne avec les MICE car sans en complexifier la structure, ils ajoutent une autre dimension et de nouvelles informations sur les tendances d’évolution des écosystèmes. Écologiquement, cette dimension spatiale favorise la conservation des nourriceries, des états larvaires et juvéniles des espèces de poissons (Fogarty and Botsford, 2007). Socio-économiquement, une gestion spatiale diversifiée des stratégies de pêches est déjà partiellement intégrée dans les modèles de gestion de nombreuses pêcheries à travers le monde (Fogarty and Botsford, 2007) et continue d’être un sujet largement discuté parmi les scientifiques (Wilén, 2004). Par ailleurs, cette thèse est notamment en lien avec un nouveau modèle de gestion spatiale et écosystémique nommé *Ecosystem-based spatial management (EBSM)* testé au sein de la réserve marine des Galapagos et dont l’efficacité des applications demeure nuancée (Castrejón and Charles, 2013).

Cette analyse spatiale permet de déterminer quelles zones sont les plus rentables à un instant t et lesquelles ont vocation à devenir des AMPs. Comme expliqué pour l’étude de cas sur Moorea

³Capter toutes les espèces et tailles dans un écosystème proportionnellement à leur productivité.

⁴Répartir les impacts est l’un des outils les plus utilisés en économie et dans le développement durable (Verschuur and Reysoo, 2002; OCDE, 2012).

(Chap III et IV), les études des données locales d'exploitation halieutique et d'abondance des stocks nous informe sur les applications propres à chaque socio-écosystème. Ainsi, cela améliore la pertinence des résultats notamment en affichant des efforts de pêche par zone ou par patches.

4.4.2 Une diversification des équipements et des cibles

Nous discutons dans cette thèse de la nécessité de favoriser une plus grande diversification des profils de pêche et donc des espèces capturées (i.e diversification des **équipements** et des **cibles**). Cela permettrait une gestion adaptative de l'effort en fonction de l'abondance des stocks, de leur productivité (Zhou et al., 2010) et du contexte climato-écologique. La diversification des métiers de pêche augmente la production et réduit les menaces sur les stocks en danger (Burgess, 2015). A l'instar du secteur agricole ou financier, un agriculteur ou un investisseur a tout intérêt à diversifier ses outils de cultures ou son portefeuille d'actifs afin de minimiser ses risques. Si l'un de ses produits (naturel ou financier) voit son prix diminué ou son rendement mis à mal (causé par le changement climatique, une crise financière, etc...), il vaut mieux qu'il dispose d'un autre produit lui permettant de garder une soutenabilité socio-économique. Ce raisonnement peut s'appliquer aux cas des pêcheries.

Plus concrètement, cette thèse recommande une **augmentation de la flexibilité des flottilles**, comme cela a déjà été montré par Gourguet et al. (2014) pour la pêcherie australienne de crevettes nordiques (*Australian Northern Prawn Fishery - NPF*) faisant ainsi le lien avec les stratégies d'adaptation face aux changements globaux. Des subventions de pêche (contrôlées et soumises à certaines conditions ou objectifs - Lindebo, 2005), non pas pour augmenter la productivité des flottilles comme cela sera sous interdit et généralisé à travers le monde (14.6 objectifs de l'ONU - UN, 2015 - Figure I.6), mais pour améliorer l'adaptabilité et la flexibilité ou plus globalement la capacité **de réponse**⁵ des pêcheurs face aux changements globaux, pourraient avoir des effets bénéfiques.

4.4.3 Une diversification temporelle

La **temporalité** est également fondamentale dans les modèles de gestion. Si nous reprenons l'exemple des agriculteurs, les fruits et légumes (produits dans des conditions naturelles) sont soumis aux saisons et au temps. Le développement technologique devrait donc tendre vers la construction (ou la modification) de bateaux équipés d'engins différents (qui sont fonctions de la saisonnalité et des périodes de reproduction) capables de s'adapter. Nous rappelons ici que la considération écologique ne doit pas primer sur les résultats socio-économiques (les trois piliers doivent demeurer en équilibre). Ce genre de raisonnement s'applique déjà pour de nombreuses pêcheries où les équipements des flottilles adaptent leurs techniques de pêche aux saisons et espèces ciblées (Kruse et al., 2001). Cela est également appliqué pour les espèces en voie d'extinction. Cependant, comme expliqué précédemment et dans le Chapitre II, la capacité d'adaptation et de réponse des flottilles peut être faible au regard de l'âge et de la taille des bateaux.

Par ailleurs, une limite souvent rencontrée par les flottilles est lorsque les possibilités de pêche restent incertaines au niveau individuel (même en présence de TACs globaux), induisant des phénomènes de course au poisson qui peuvent réduire la durée des saisons de pêche, et les concentrer sur des périodes qui ne sont pas forcément les plus favorables en termes de rendements ou de commercialisation. En s'appuyant sur une saisonnalité de la pêche plus stricte et dotée d'une vision à long terme (pour éviter les conséquences d'une surexploitation "optimale" de la pêcherie à court ou moyen terme - Hardin, 1968; Clark, 2010), nous verrions ainsi sur les étales des poissonniers, des espèces uniquement disponibles à certains moments de l'année tout comme les

⁵La diversification peut généralement permettre à des acteurs de mieux gérer les risques au niveau individuel ; les conséquences socio-économiques et écologiques associées aux phénomènes de reports peuvent en revanche être fortement dépendantes de la capacité d'encadrement des activités de pêche (Soulié and Thébaud, 2006).

fruits et légumes. Pour de nombreuses pêcheries, l'implémentation d'une telle gestion temporelle suggère une refonte du marché, par une offre moins diversifiée, et une demande (notamment professionnelle) qui doit être plus souple ou compensée en majorité par l'aquaculture (ce qui commence doucement à être le cas - FAO, 2018 - Figure I.1).

Même si les institutions (inter-) gouvernementales assurent intégrer des considérations écologiques dans leurs objectifs de gestion, les moyens mis en oeuvre semblent parfois (et pour l'instant) insuffisants lorsque nous observons les conséquences écologiques. En effet, d'une part, malgré la diminution des quotas et en dépit des recommandations des scientifiques, certaines espèces demeurent sous leurs seuils de précaution et d'autre part comme évoquée précédemment, l'incertitude globale actuelle et à venir nous place face à des enjeux globaux dont nous peinons à évaluer les conséquences locales.

4.4.4 Une diversification pour une soutenabilité, une viabilité et une résilience des pêcheries

En résumé, de telles stratégies de diversification favorise la soutenabilité, la viabilité et la résilience des pêcheries car elle permettent de demeurer au coeur d'un espace d'états viables et durables sur le long terme. En effet, les espèces exploitées se maintiendraient au dessus de leur seuil de précaution (grâce à une diversification des cibles/équipements et temporelle) tandis que les pêcheurs profiteraient, à chaque changement de cible ou de zone, de la régénération du stock protégé ou de la zone réouverte (diversification spatiale). Cette gestion adaptative atténuerait le temps de récupération d'une pêcherie (car les pressions multi-drivers seraient durablement réparties), augmenterait sa robustesse (par définition, sa probabilité de rester au dessus des seuils socio-écologiques), et permettrait au système de respirer par moment ce qui favoriserait sa diversité spécifique et fonctionnelle (ainsi que ses services écosystémiques) et donc sa résistance aux chocs i.e sa résilience socio-écologique.

4.5 Perspectives

Cette thèse a montré l'importance d'intégrer l'approche écosystémique (EBFM) dans la gestion durable des pêches, en particulier, en s'appuyant méthodologiquement sur les MICEs. Dans ce cadre, les stratégies écologico-économiques de diversification favorisent la soutenabilité, la viabilité et la résilience des socio-écosystèmes marins par une meilleure conservation des ressources naturelles ainsi qu'une amélioration des performances socio-économiques. Néanmoins de nombreux défis demeurent. Tout d'abord la question de la gouvernance.

Actuellement, la gouvernance et les instruments pour gérer les socio-écosystèmes sont divers et discordants. Les quotas (TAC - *Total Allowable Catch*), les marchés de quotas (ITQ) sont pour l'instant des solutions qui paraissent correctes sans être parfaites tant sur le plan écologique que social (Thébaud et al., 2012). Plus de démocratie participative, de politiques publiques de long terme scientifiquement fondées permettraient une amélioration de la situation.

Une chose fondamentale qui ne devrait pas être négligée concerne notre capacité d'innovation technologique. L'innovation salvatrice discutée par Schumpeter sera bénéfique pour l'avenir si elle continue de s'orienter vers la soutenabilité, et l'économie écologique et si elle est accompagnée d'une définition claire des règles d'accès aux ressources communes⁶. L'aquaculture et les énergies renouvelables en sont des exemples intéressants. Le progrès technique est l'un des facteurs qui devrait soutenir la pérennité de nos sociétés en liaison avec les ressources renouvelables incluant énergies renouvelables et biodiversité.

La future gouvernance mondiale et étatique sera déterminante dans la mise en place de stratégies de gestion écosystémique, durable, viable et résiliente.

⁶Par ailleurs, l'innovation pose nécessairement la question des arbitrages entre rendements économiques et emploi.

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Chapitre II

How does MMEY mitigate bioeconomic effects of climate change for mixed fisheries

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Abstract

This paper examines the impact of climate warming on the bio-economic performances of Bay of Biscay mixed fisheries and provides insights into sustainable management strategies for coping with global warming. To achieve this, a dynamic multi-species, multi-class, multi-fleets model is developed and calibrated using biological, economic and environmental data from ICES, European Data Collection Framework and IPCC. Fishing and economic data have been collected within the European Data Collection Framework. The climate represented by the sea surface temperature is assumed to affect species recruitment. Three management strategies are then compared in terms of bio-economic outcomes: Status-Quo (SQ), Multi-species Maximum Sustainable Yield (MMSY) and Multi-species Maximum Economic Yield (MMEY). These strategies are ranked with respect to two constrained RCP climate scenarios. Results exhibit that the SQ strategy is not sustainable and is characterized by a major decline of Sole. By contrast, the MMSY and the MMEY strategies improve the ecological state and economic performance of fisheries. Furthermore, the MMEY strategy yields higher bio-economic performances than MMSY. These bio-economic benefits are however altered by climate change effects. Under the MMEY, fleets with more diversified catch structures perform better facing climate change.

Key words : bio-economics, scenarios, global warming, fisheries, sustainability, Bay of Biscay

1. Introduction

Marine biodiversity and ecosystem are under extreme pressure worldwide because of the intensification of fishing methods and an overall increase of seafood demand. Thus, according to FAO (2014), around 80% of worldwide commercial fish species are overexploited or fully exploited. Climate change complicates and exacerbates the issues by inducing new, or intensifying existing, risks, uncertainties and vulnerabilities.

In that context, the European Union explicitly accounts for the objectives of mitigating and adapting to the effects of climate change in the area of maritime spatial planning and integrated coastal zone management¹. The Common Fisheries Policy (CFP - Reg. UE 1380/2013 11/12/2013) also reaffirms the obligations associated with the international commitments of a sustainably management of fisheries and to strengthen the existing arrangements. It also puts forward a more regional approach for optimizing the various devices in order to reach, in particular, the maximum sustainable yield by 2020. In the meantime, the Marine Strategy Framework Directive² aims at protecting and conserving the marine biodiversity. Positive economic and social benefits as well as food security are also targeted. In that multi-facet perspective, our paper investigates the ecological and economic performances of the Bay of Biscay mixed fisheries facing climate warming.

In the Bay of Biscay, a warming of $\cong 0.2$ ° C / decade has been observed for the period 1965-2004 between the surface and 200 m depth (Decastro et al., 2009). Such climate changes already impact some fish species. In particular for flat fish species, including Sole, with a wide area within the Bay of Biscay, recent studies have shown spatial correlations between their abundances and the increase in temperature (Hermant et al., 2010). For boreal species, abundance decreases in the Bay while for southern species it increases. Recruitment appears to be the main process affected by climate change (Koutsikopoulos et al., 1998). As the definition of boundaries and access rights is a crucial issue for fisheries management, climate change raises a new challenge to institutions and regulating agencies (Badjeck et al., 2010). Variations in the spatial distribution of species indeed question the update of fishing rights and the geographical redeployment of fleets (Rajudeen, 2013).

As a consequence, designing management tools and public policies that ensure the long-term bioeconomic sustainability of marine fisheries constitutes a major challenge (FAO, 2014). In response, there has been a growing need of integrated assessment tools to support management advices (Thébaud et al., 2014) such as ecosystem-based fishery management (EBFM - Pikitch et al., 2004 ; Link et al., 2017) that aims at integrating the ecological and economic complexities of fisheries, instead of focusing on isolated target species. However, the way to operationalize the EBFM approach remains challenging, especially from the bioeconomic viewpoint (Sanchirico et al., 2008 ; ?). New models are needed, notably to integrate the multiple complexities at play (Plagányi, 2007). In particular, these models are expected to account for the multispecies and multi-fleets nature of fisheries, for the multiple ecosystem services or values they provide as well as for climate impacts. They should also help evaluating the bioeconomic effectiveness and sustainability of current regulatory instruments such as fishing quotas or financial incentives, and designing relevant ecosystem-based management quantitative tools (Patrick and Link, 2015b).

Many fish stocks are currently managed to reach their maximum sustainable yield (MSY), through limitations on fishing quotas or efforts (Mace, 2001). At MSY, catches are maximized at levels where the stock can regenerate. This strategy has been set as the main reference point of many world fisheries and is one of the key objective of the new CFP. However the sustainability of this monospecific strategy in multispecies contexts is disputed (Larkin, 1977). In particular, applying MSY policies from single-species assessments in multispecies communities with trophic interactions has been shown to induce biodiversity losses (Walters et al., 2005). Instead of MSY,

¹https://ec.europa.eu/clima/publications_en#Mainstreaming

²Directive 2008/56/EC – EU action in the field of marine environmental policy (Marine Strategy Framework Directive) - <http://eur-lex.europa.eu/legal-content/FR/TXT/?uri=LEGISSUM:l28164>

many resource economists advocate the use of maximum economic yield (MEY) targets, at which profits are maximized (Dichmont et al., 2010). Harvesting at MEY is notably known to favor higher biomasses than harvesting at MSY (Clark, 2010 ; Grafton et al., 2012). In a single-species context, harvesting at MEY is thus a more profitable and viable strategy than maximizing sustainable yield. In that perspective, the MEY management strategy has been chosen as a reference point for Australian fisheries (Dichmont et al., 2010). However, maximizing profits from a single stock can also induce overexploitation and extinction, provided its price is higher than the cost of depleting the stock (Clark, 1973).

To account for the multispecies nature of fisheries, multispecies reference points and targets are now proposed (Moffitt et al., 2015). However, the potential bioeconomic consequences of such multispecies harvesting policies remain largely unknown. There have been attempts at designing multispecies MSY (MMSY) policies, at which total catches are maximized (Mueter and Megrey, 2006). But in mixed fisheries where technical interactions occur, that is when one fishing fleet harvests different species, maximizing total yields has been suggested to endanger some species (Ricker, 1958 ; Legović and Geček, 2010 ; Guillen et al., 2013). Potential consequences of multispecies MEY (MMEY), at which total profits are maximized, have also been investigated (Anderson, 1975). As in the single-species case, MMEY is found to be more profitable than MMSY (Guillen et al., 2013), however, MMEY can induce the overexploitation of stocks with low value (Chaudhuri, 1986 ; Guillen et al., 2013 ; Tromeur and Doyen, 2019). In other words, if a multispecies fishery is seen as a portfolio of natural assets, maximizing total profits could neglect the conservation of inferior assets, thus inducing biodiversity losses.

The aim of this paper is to evaluate and compare the bioeconomic merits of MMSY and MMEY policies respectively as well as to question their relevance for operationalizing ecosystem-based management for mixed fisheries facing climate change in the Bay of Biscay. More specifically, this paper examines the impact of climate warming on the bio-economic performances of Bay of Biscay mixed fisheries and gives insights into the sustainable management strategies to cope with global warming. To achieve this, a multi-class, multi-fleets and dynamic model for common sole (*solea solea*) and European hake (*merluccius merluccius*) is developed and calibrated using ICES, European Data Collection Framework and IPCC data.

2. Bay of Biscay case study

Our study deals with the mixed fisheries of the Bay of Biscay operating in divisions VIIIa and VIIIb according to the International Council for the Exploration of the Sea (ICES) sections (Figure II.1). Regarding fish species, we focus on common sole and hake.

2.1 Sole

Common sole (*solea solea*) is a benthic species whose distribution extends from the West African coasts to the Baltic. In the Bay of Biscay, common sole is in the center of its range area (average latitude 44.5 ° N). So far, the work carried out by Hermant et al. (2010) did not reveal a clear trend in the evolution of its spatial distribution.

The sole appears as a vulnerable stock and is subject since 2002 to a management pattern aiming at restoring the spawning biomass at its level of precaution (Bpa). This goal was reached in 2009 (Figure II.2). However, due to surprisingly low recruitment in 2010, the stock is again declining. Consequently, since 2016, a 10% reduction in total allowable catches (TAC) as compared to 2015 and 2014 has been imposed (ICES, 2017) by the European commission inducing a quota of 3420 tons for French fleets (European Union, 2016). Although the spawning biomass of sole recovered for three years, it still remains below the sustainable reference point (Bpa = 13000 tons) since 2013 (ICES, 2017).

2.2 Hake

Distributed in the North-East Atlantic, European hake (*merluccius merluccius*) is present along the coasts from Norway to Mauritania. Temperature is a driver that affects the early stages of hake life (Hermant et al., 2010). Experiments in a controlled environment for the development of hake eggs at different temperatures indeed showed high mortalities outside the range 10-13° (Guevara-Fletcher et al., 2016). Similarly, studies in the Mediterranean using habitat models show that nurseries require stable background temperatures (11.8-15 ° C), low background velocities (<3.4 cms-1) and productive plankton fronts (Druon et al., 2015). Moreover, as growth or survival of hake juveniles is increased with the availability of adequate feeding, changes in ocean conditions affect prey availability and thus affect migration behavior and hake growth (Benson et al., 2002). Thus, Goikoetxea and Irigoien's work (2013) in the Northeast Atlantic on hake highlighted the role of the North Atlantic Oscillation (NAO) in the success of recruiting hake for several years (Figure II.3). More specific informations about species ecology can be found from the bibliographical synthesis made by Caill-Milly et al. (2018).

2.3 Economical role of hake and sole fisheries

Hake and common sole are among the first four species in terms of economic values on the Atlantic coast. In 2015, on the Atlantic coast, Hake represents 20% of the overall production in values while Sole reaches 7% (IFREMER, 2017). For the French landings, hake represents 7% of the total value while the sole rise to 11% (Gourguet et al., 2013). The sole is less abundant than hake which is the dominant species for fisheries in the European Union (EUMOFA, 2015). Indeed, although hake has suffered from severe overexploitation with a fall in its recruitment in the 1990s (Figure II.3), a recovery of its spawning stock has been observed (ICES, 2016) in the following years strongly induced by better recruitments and by the European mono-specific management plan set up at that time (MSY). The price per kilo of the sole is much more important than hake. In 2015, it is worth about 12 € per kilo with a 60 million € market to be compared with 3€ per kg for hake within a market which represents 45 million €. These values derive from

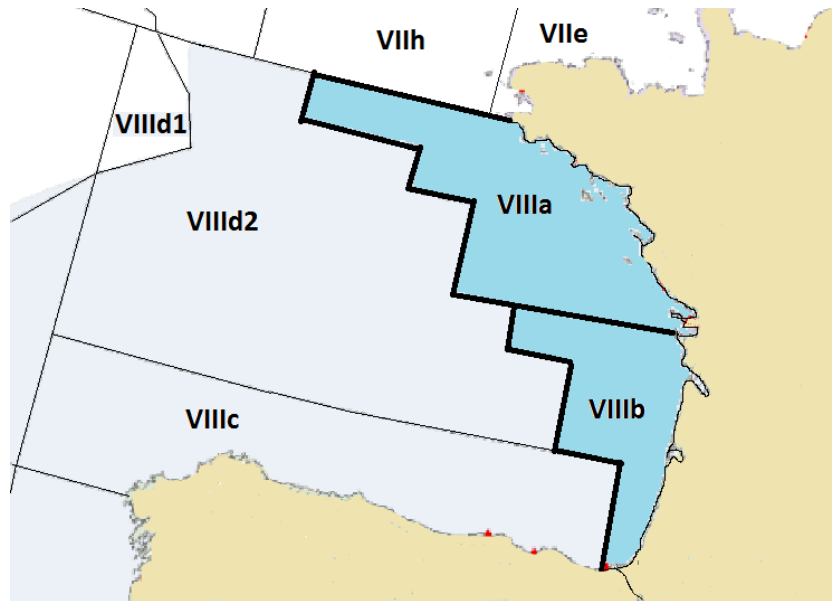


Figure II.1: Map of the Bay of Biscay and ICES divisions. The studying area is in blue.

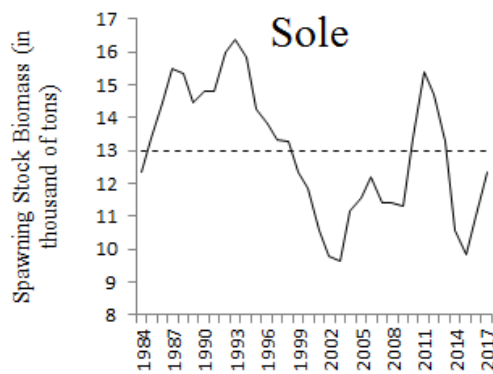


Figure II.2: Historical evolution of the spawning biomass for the common sole. The dashed line refers to the precautionary threshold B_{pa} . (ICES, 2017)

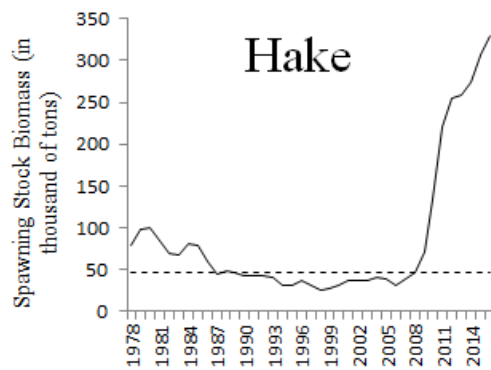


Figure II.3: Historical evolution of the spawning biomass for the European hake. The dashed line refers to the precautionary threshold (B_{pa}) estimated at 46200 tons (ICES, 2016)

the marked preference of consumers for sole. Consequently, the high abundance of hake and the high price of sole explains why both species are of major economical interest.

The main French fleets targeting those two species, include 400 vessels across the Bay of Biscay and can be divided into three groups of vessels based on their main gear: various fish trawlers, sole gill-netters and various fish gill-netters. These three fleets can then be separated into 13 sub-fleets ranked by size (Gourguet et al., 2013).

2.4 Data sources

Recruitment and spawning biomass estimation of the sole and hake have been extracted from population models by the ICES on an annual basis for sole and quarterly for hake¹ from 1991 to 2013. Sole data are derived from a population dynamics model named XSA (Extended Survivors Analysis - Shepherd, 1999) while hake data have been estimated through the SS3 (Stock Synthesis 3) model based on commercial catches and on abundance data (ICES). Biological parameters are displayed in appendix (Tables II.5 and II.6) while fishing mortality on Hake and Sole are detailed in Tables II.7 and II.8. Economic data and transversal data of effort and production by fleet and gear have been collected from the Fisheries Information System of IFREMER and the French Directorate for Fisheries and Aquaculture (DPMA) through the European Data Collection Framework (DCF). Sea Surface Temperatures (SST) arise from a project led by the European Union called OpEc² which aimed at rebuilding the history of all marine ecosystems, biological and historical data such as water temperature, oxygen, salinity. The geographical coordinates used in this study are: latitude (43.75, 47.39) and longitude (-6.90, -2.77). They do not refer to the entire Bay of Biscay but only to ICES divisions VIIIa and VIIIb. For the SST projections until 2100, we rely on the more recent IPCC (Intergovernmental Panel on Climate Change) report which provides, according to four emission scenarios (RCP)³, many environmental forecasted data. In this paper, we choose to focus on the worst and best climate scenario, respectively, RCP 8.5 and RCP 2.6.

¹In the report made by the ICES, the sea council hypothesizes that no recruits is observed in the fourth quarter, the sum of the three previous quarters represents here the annual and actual spawning stock (ICES)

²Operational Ecology (End date : 31/12/2014) - <http://marine-opec.eu/>

³Representative Concentration Pathways

3. The bio-economic model

We rely on a multi-species, multi-class, multi-fleets and dynamic model in discrete time inspired by Quinn and Deriso (1999), Doyen et al. (2012) and Gourguet et al. (2013). Environmental, biological, economic components and links of the model are described in Figure II.4. These links highlight how the different interactions occur: SST impact recruitment through the specific responses of the Spawning Stock Biomass (SSB) with respect to the environmental context. Then stock levels along with the fishing effort determine the catches, profits and biological outcomes from 2014 until 2094.

3.1 Multi-species age-class dynamic model

For each species, population dynamics described on a yearly basis by age group is first characterized by natural and fishing mortality mechanisms as follows :

$$\begin{cases} x_{i,a}(t+1) = x_{i,a-1}(t) \exp(-M_{i,a-1} - F_{i,a-1}(t)) \\ x_{i,A_i}(t+1) = x_{i,A_i-1}(t) \exp(-M_{i,A_i-1} - F_{i,A_i-1}(t)) + x_{i,A_i}(t) \exp(-M_{i,A_i} - F_{i,A_i}(t)) \end{cases} \quad (\text{II.1})$$

where $x_{i,a}(t)$ stands for the abundance of the exploited species, $i = 1, 2$ (sole, hake respectively) at age $a = 2, \dots, A_i$ at time t . The age class starts at two because the first one stands for recruitment. We assume that there is not interactions between sole and hake. Thus, abundances of species $x_{i,a}(t)$ evolves according to both natural $M_{i,a}$ and total fishing $F_{i,a}(t)$ mortalities of the species s at age a and time t . Furthermore, the total fishing mortality $F_{i,a}(t)$ is derived from the sum of the fishing mortality of the $m = 13$ sub-fleets f at year $t_0 = 2014$ described by:

$$F_{i,a}(t) = \sum_{f=1}^m u_f(t) F_{i,a,f}(t_0) \quad (\text{II.2})$$

where $u_f(t)$ stands for the fishing effort multiplier of the sub-fleet f at time t . The initial fishing mortality, $F_{s,a,f}(t_0)$, depends on catchability, effort and number of boats as follows :

$$F_{i,a,f}(t_0) = q_{i,a,f} e_f(t_0) K_f(t_0) \quad (\text{II.3})$$

with $e_f(t_0)$ is the mean value of fishing effort by vessels of sub-fleet f expressed in number of days at sea, $K_f(t_0)$ is the number of vessels by sub-fleet f , both for the baseline year 2014 and $q_{i,a,f}$ the catchability of the sub-fleet f on species i at age a . Thus, the fishing mortality is assumed to be proportional to effort as in the seminal Graham-Schaefer model. Such an assumption arises from the real situation in the Bay of Biscay where congestion effects (Chu and Kompas, 2014) are very limited. Indeed, for instance, the number of vessels which operates in the sole fishery has been decreasing of 21% between 2000 and 2011 while French vessels targeting the sole account for about 90% of the total number of boats (Guyader et al., 2017).

3.2 Stock-recruitment dynamics

The spawning biomass $SSB_i(t)$ for the two species is described by:

$$SSB_i(t) = \sum_{a=1}^{A_s} \gamma_{i,a} v_{i,a} x_{i,a}(t) \quad (\text{II.4})$$

where $\gamma_{i,a}$ stands for the share of fertile individuals of species i at age a and $v_{i,a}$ represents the weights (in tons) of individuals of species i at age a and $a = 1, \dots, A_i$.

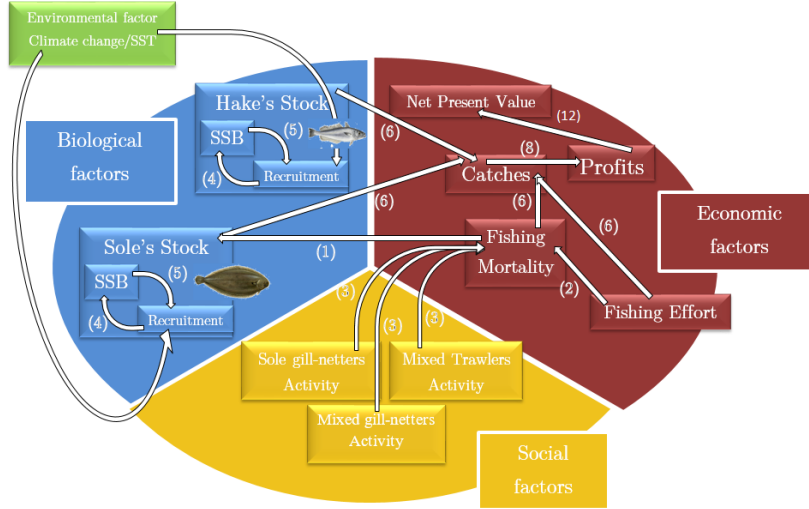


Figure II.4: Relations existing between environmental, biological and economic factors within the bio-economic model. Arrows stand for the interactions between variables while figures between brackets refer to the equations/models that link the various factors within the bio-economic model.

We assume that the recruitment dynamics depends on both SSB and sea surface temperature θ in a stochastic way as follows:

$$x_{i,1}(t+1) = f(SSB_i(t - \tau_i), \theta(t - \tau_i), \varepsilon_i(t - \tau_i)) \quad (\text{II.5})$$

Here $x_{i,1}(t)$ represents the recruits, $\theta(t)$ stands for the sea surface temperature at time t while $\varepsilon_i(t)$ captures the environmental stochasticity affecting the recruitment. τ_i is a lag with respect to the time necessary for the egg to become a catchable recruit (about two years for the sole $\tau_1 = 1$; about one year for hake $\tau_2 = 0$). The integration of environmental factors in recruitment is in line with Cushing (1982), Glantz (1992) and Laevastu (1993). Recruitment may be affected by sea temperature through many behavioral and physiological processes during spawning and larval phase such as metabolic cost of spawners, natural mortality of eggs and larvae, food availability etc. (Hermant et al., 2010). Different recruitment functions f have been considered here including the Ricker (1958), Beverton-Holt (1957) and Cushing models as displayed in Table II.4. Most of these stock-recruitment models are derived from a generalisation of the Ricker and Beverton-Holt model (Hilborn and Walters, 1992). With these different formulations, we performed regressions¹ of recruits over SSB and SST time series² in order to find the recruitment model that best fits the data. As the biological interactions between temperature and recruitment are complex, another possible approach would have been to use a neural network approach as in Kompas and Chu (2018) which does not require a specific form but is more time-consuming. We detail and discuss the time lags τ_i obtained in the results section.

3.3 Economic scores

Assuming that discards are neglectable, landings of the m different sub-fleets equals catches (Harvests) and are defined by the Baranov catch equation:

¹Ordinary Least Squared for the log-linearised model of the sole with 22 observations and autoregressive process of order 1 for the log-linearised model of the hake to correct the autocorrelation of its errors with 66 observations

²By using the Scilab software and one of its econometric modules named GRO CER - <http://dubois.ensae.net/grocer.html>

$$H_{i,a,f}(t) = x_{i,a}(t)u_f(t)F_{i,a,f}(t_0) \frac{1 - \exp(-M_{i,a} - \sum_{f=1}^m u_f(t)F_{i,a,f}(t_0))}{M_{i,a} + \sum_{f=1}^m u_f(t)F_{i,a,f}(t_0)} \quad (\text{II.6})$$

Incomes derived from catches reads as follows:

$$Inc_f(t) = \sum_i \sum_{a=1}^{A_i} p_{i,a}(t)v_{i,a}H_{i,a,f}(t) \quad (\text{II.7})$$

where $v_{i,a}$, as in equation (II.4), is the mean weight of individuals of species i at age a and price $p_{i,a}(t)$ corresponds to the market value (euros by kg) of species i at age a for year t assumed to fluctuate randomly according to a Gaussian law.

Profits $\pi(t)$ as the difference between incomes and costs are defined by:

$$\pi_f(t) = (Inc_f(t) + \alpha_f u_f(t)K_f(t_0)e_f(t_0))(1 - \tau_f) - \left(V_f p(t)e_f(t_0) + c_f^{var} e_f(t_0) + c_f^{fix} \right) u_f(t)K_f(t_0) \quad (\text{II.8})$$

α_f corresponds to the income per unit of effort of sub-fleet f of other species involuntarily caught. The dynamic of these non-targeted species is not detailed but we assume constant values for these species per unit of effort, thus $Inc_f(t)$ is only a part of the global income. τ_f is the landing cost by sub-fleet as a proportion of the gross income, V_f represents the volume of fuel used by fishing effort unit and c_f^{var} and c_f^{fix} corresponds respectively to the variable³ and annual⁴ (fixed) costs by a vessel of sub-fleet f . These parameters are based on economic data available for 2008 (IFREMER, SIH, DPMA⁵, Table II.9 and II.10). The price of fuel is considered constant over time, set at a price of 0.5 € per liter.

3.4 Management strategies

We here consider three management strategies in order to compare them in terms of bio-economic outcomes: Status-Quo (SQ), Multi-species Maximum Sustainable Yield (MMSY), Multispecies Maximum Economic Yield (MMEY).

Status-quo strategy: The first management strategy entitled Status-Quo (SQ) maintains fishing efforts constant throughout the period of interest $t_0 = 2013$ to $t_n = 2094$ such as:

$$u^{SQ}(t) = 1 \quad \forall f = 1, \dots, m \quad \text{and} \quad t = t_0, \dots, t_n$$

Multi-species Maximum Sustainable Yield (MMSY) strategy: The second fishing strategy aims at reaching a maximum sustainable yield over all species considered, that is to say, to maximize the aggregated long-term landings of the different fleets. The objective is to find the best constant effort multiplier vector noted u_f^{MMSY} that maximizes total catches. In fact, we maximize the mean total catches over time defined as the average of the total catches over the entire temporal period. Moreover, to account for the stochasticities affecting both the species

³The variable cost includes oil, supplies, ice, bait, gear, and equipment costs

⁴The annual cost includes maintenance, repair, management and crew costs, fishing firms, licenses, insurances and producer organisation. Those costs date from 2008

⁵DPMA stands for Direction des Peches Maritimes et de l'Aquaculture which corresponds to the Directorate for Sea Fisheries and Aquaculture at the French Ministry of Agriculture and Fisheries. SIH means Systeme d'Informations Halieutiques, the fisheries information system monitored by Ifremer, the French Research Institute for the Exploitation of the Sea (http://www.ifremer.fr/institut_eng)

prices $p_s(t)$ in equation (II.7) and recruitment dynamics (II.5) through $\varepsilon_s(t)$, we consider the expected value of the mean catches::

$$H^{MMSY}(u) = \mathbb{E} \left[\frac{1}{t_n - t_0} \sum_{t=t_0}^{t_n} \sum_{i=1}^2 \sum_{a=1}^{A_i} \sum_{f=1}^m H_{i,a,f}(t) \right] \quad (\text{II.9})$$

Once we have the expected catches, we indentify the vector of best fishing effort multipliers denoted by u_f^{MMSY} that maximize the previous metrics:

$$H^{MMSY}(u^{MMSY}) = \max_u H^{MMSY}(u) \quad (\text{II.10})$$

As explained in the introduction, by adopting a multi-species point of view, the MMSY management takes into account the fact that most fleets do not target (voluntarily or not) only one species. Thus, this management model potentially offers a more relevant management from an ecosystem and multi-species perspective as compared to a single-species point of view (Voss et al., 2014).

Multi-species Maximum Economic Yield strategy: The third strategy we consider consists in maximizing the Net Present Value (NPV) over the m fleets defined by:

$$NPV(u) = \mathbb{E} \left[\sum_{t=t_0}^{t_n} \delta^t \sum_{f=1}^m \pi_f(t) \right] \quad (\text{II.11})$$

with profits $\pi(t)$ defined in equation (II.8) and $\delta = \frac{1}{1.04}$ the discount factor. Again, \mathbb{E} corresponds to the expectations with respect to the stochastic parameter ε_i and prices p_i . Maximizing the NPV relates to the maximum economic yield for both species which explains why we introduce the notation u_f^{MMEY} .

$$NPV(u^{MMEY}) = \max_u NPV(u) \quad (\text{II.12})$$

To compute numerically the optimal solutions, we have also used the SCILAB software.

3.5 Climate scenarios

In our study, we consider two extrem climate scenarios (IPCC, 2013 - RCP 2.6, RCP 8.5) illustrated by the Figure II.5 for the Sea Surface Temperature in the Bay of Biscay. We notice an upward trend for historical temperatures and a recent and sharp increase for the last few years. Indeed, from 2007, after a fall of more than 0.5°C, the Bay of Biscay is getting warmer with a rise of nearly 1.5°C in just 6 years. This outcome is the result of an increase in warming of 0.06/0.07°C per year over the last 30 years (Le Treut, 2013). Inter-annual variations induced by atmospheric flux and ocean currents (Michel et al., 2009) are the main sources of uncertainty and are very difficult to predict even with complex climate models. Yet, the accuracy of climate models is steadily increasing since the 1990s because of the advancement of research, more available data and also due to some major technological discoveries (IPCC, 2013). Therefore, even if these models cannot predict what the temperature will be to the tenth of a degree in 80 years, they are getting closer to reality by relying on verifiable physical principles and on emission scenarios more than likely due to our human activities (IPCC, 2013).

These projected temperature in the Bay of Biscay are integrated each year in the recruitment formula of equation (II.5) which affects the species dynamics as a whole and by extension the economy.



Figure II.5: Historical trajectories of SST in the Bay of Biscay from 1991 to 2013 and projections of SST trajectories according to the two climate scenarios (RCP 2.6 and 8.5) from 2014 to 2100

4. Results

This section presents the merits of integrating a temperature-dependent stock-recruitment model into our bioeconomic model in order to determine the management strategy, that best mitigates warming effects among SQ, MMSY and MMEY.

4.1 Impact of warming on stock-recruitment model

Sole	Standard Error (σ)		0,2037519		
	Sum of squared residuals		0,8302967		
	R ² = 0,51		a	b	c
	Estimation	58,106969	-0,0000743	-0,012258	
t(17)	5,0245006	-3,3391635	-3,6149265		
p-value	0,000065	0,0032687	0,0017274		

Table II.1: Parameters and standard errors of the estimated Ricker model (equation II.13) accounting for temperature for sole. t(17) stands for the Student test with 17 observations.

Hake	Standard Error (σ)		0,633434		
	Sum of squared residuals		25,278034		
	R ² = 0,27		a	b	c
	Estimation	4,4805325	-0,0000067	-0,0020034	
t(62)	6,5098007	-4,2417837	-2,163427		
p-value	1,418E-08	0,000074	0,034309		

Table II.2: Parameters and standard errors of the estimated Ricker model (equation II.14) accounting for SST for hake. t(62) stands for the Student test with 62 observations.

In Tables II.1 and II.2, we present the main results of the statistical analysis of recruitment models. The equations (II.13) and (II.14) notably highlight the importance of lags between recruitment and SSB and SST. They are lagged proportionally to the time necessary for the species to become an egg, a larvae, a juvenile then a catchable recruit : two years for the sole et one year for hake in average. We know that a recruitment model only driven by SSB is likely to appear less explanatory than a model which includes an environmental factor (Cury et al., 2014). This is highlighted in table II.4. Moreover, all estimated coefficients (a, b, c) are statistically significant at the 5% level (Tables II.1 and II.2). For both species, the Ricker model turns out to be more relevant than the Beverton-Holt, Cushing or Cobb-Douglas model and all coefficients are statistically significant. These conclusions are consistent with the study carried out by Anneville and Cury (1997) which explains that the Ricker model is “the best pattern [...] because it ensures a much stronger regulation”. Beverton and Iles (1998) also confirms that the Ricker model is the best pattern to explain the stock-recruitment relationship especially if the effect of temperature is integrated. The influence of temperature on recruitment, explained by the coefficient c in Tables II.1 and II.2, is negative for both species.

Sole’s SR model

$$x_{1,1}(t+1) = aSSB_1(t-1)e^{-bSSB_1(t-1)-c\theta(t-1)^2} + \varepsilon_1(t-1) \quad (\text{II.13})$$

Hake's SR model

Hake's model is first built with quarterly data. Given that no recruits is observed during the winter quarter so the first three quarters are equals to an entire year. Thus, to harmonize it, we sum the recruits of the three quarters of the previous year in order to transform the quarterly model into a yearly model such as:

$$x_{2,1}(t+1) = \tilde{x}_{2,1}(t_1(t)) + \tilde{x}_{2,1}(t_2(t)) + \tilde{x}_{2,1}(t_3(t))$$

with $\tilde{x}_{2,1}(t_j(t))$ the number of hake recruits of quarter i of year t such as:

$$\tilde{x}_{2,1}(t_j(t+1)) = aSSB_2(t_j(t))e^{-bSSB_2(t_j(t))-c\theta(t_j(t))^2} + \varepsilon_2(t_j(t)) \quad \text{with } j = [1, 2, 3]$$

so the yearly basis model is described as:

$$x_{2,1}(t+1) = \sum_{j=1}^3 \left(aSSB_2(t_j(t))e^{-bSSB_2(t_j(t))-c\theta(t_j(t))^2} + \varepsilon_2(t_j(t)) \right) \quad (\text{II.14})$$

4.2 Status-quo : not ecologically and economically viable

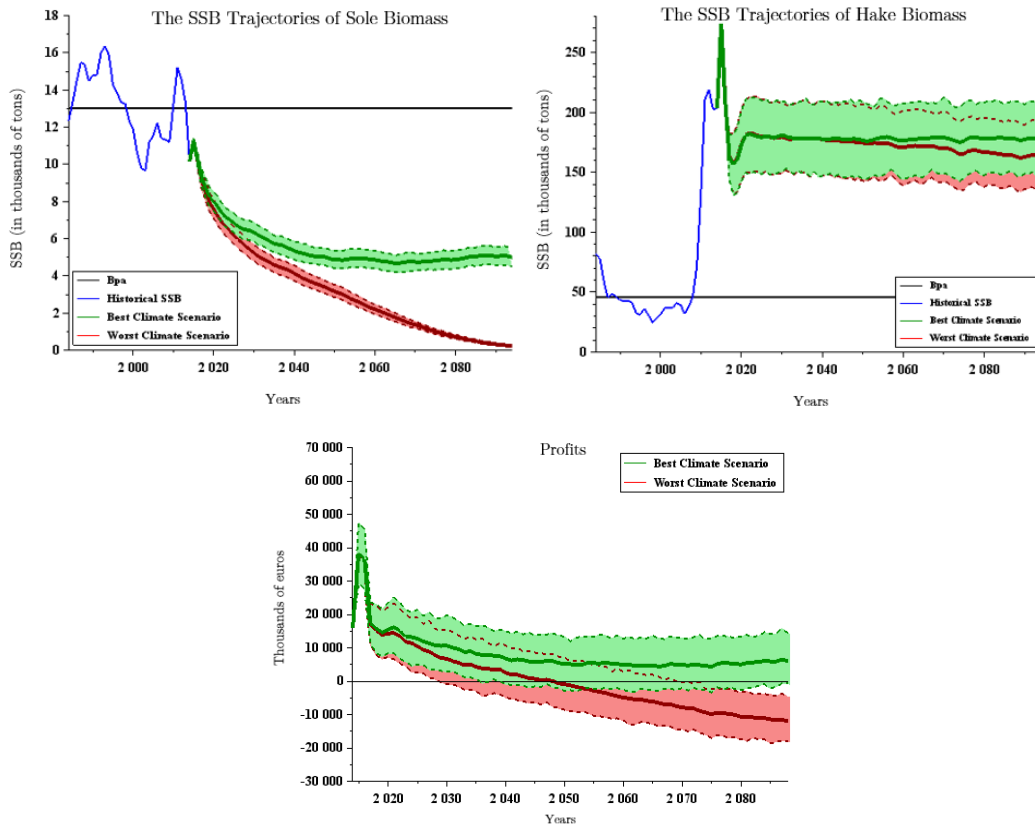


Figure II.6: Strategy Status-Quo - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.

Figure II.6 describes the estimated¹ bio-economic performances of the SQ strategy under the two climate scenarios : best scenario/RCP 2.6 and worst scenario/RCP 8.5 over the period 2014-2088. On the top, are plotted the SSB of Sole and Hake while profits are displayed on the bottom.

Figure 6 shows that the SQ strategy is not ecologically or economically viable. The ecological vulnerability relates to sole biomass which indeed declines and violates the ICES precautionary limit, even with a favorable climate change scenario, and collapses under the pessimistic climate scenario. The economic vulnerability arises from the worst climate scenario which leads to negative profits in the fishery.

More globally, the SQ strategy highlights the fact that if fishing efforts are not adjusted, global warming will amplify the current fall of the sole SSB and could lead to an economic collapse. Therefore, management strategies are required to adapt fishing effort in order to moderate the impact of global warming on bio-economic outcomes.

¹The 500 simulated trajectories are induced by Monte-Carlo replicates of uncertainties. $\varepsilon_i(t)$ assumed to be an i.i.d. centered Gaussian distribution with standard deviations of the species i denoted by σ_i and displayed in Tables II.1 and II.2 i.e $\varepsilon_i(t) \sim N(0, \sigma_i)$.

4.3 MMSY : not ecologically viable but economically viable

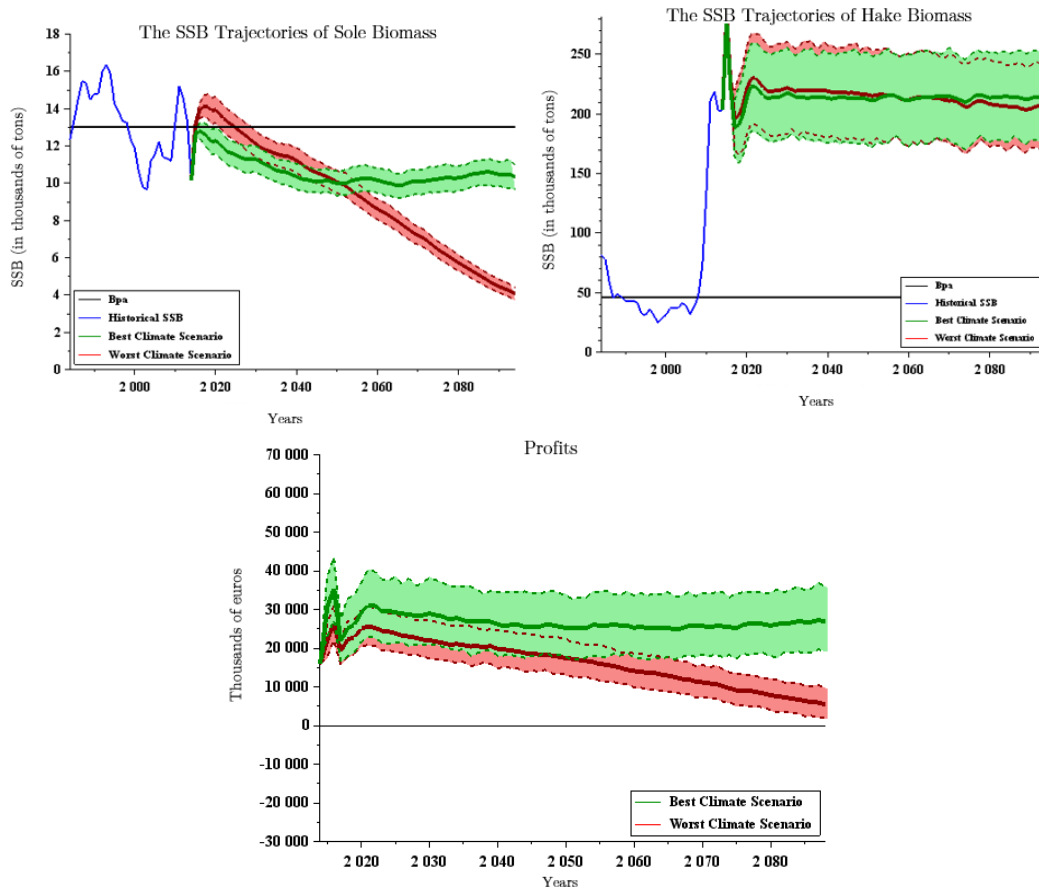


Figure II.7: Strategy MMSY - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.

As illustrated by Figure 7, the MMSY strategy performs better ecologically and economically than the SQ strategy. As expected, the more extreme the climate scenario, the more negative the impacts.

The decline of sole SSB below its Bpa appears unavoidable but is clearly mitigated by this strategy. Under the worst climate scenario, the weakness of the underlying MMSY fishing effort multipliers (Table II.3) first generates a recovery of this stock which however collapses in the long run because of the too high rise of temperatures. Under the best climate scenario, the sole stock first decreases to stabilize after 15 years at around 10 000 tons. Hake stock displays similar trends as compared to SQ scenario, with lower values regardless of climate scenarios.

Interestingly, although the purpose of this strategy is not to maintain the SSB above Bpa's, it significantly improves biological outcomes. Indeed, maximizing landings in the future can not be dissociated from sustaining high level of stock. Therefore, the MMSY strategy implicitly accounts for ecological objectives through the fishing effort mitigation, thus performing better than the SQ strategy.

4.4 MMEY : ecologically viable and economically viable

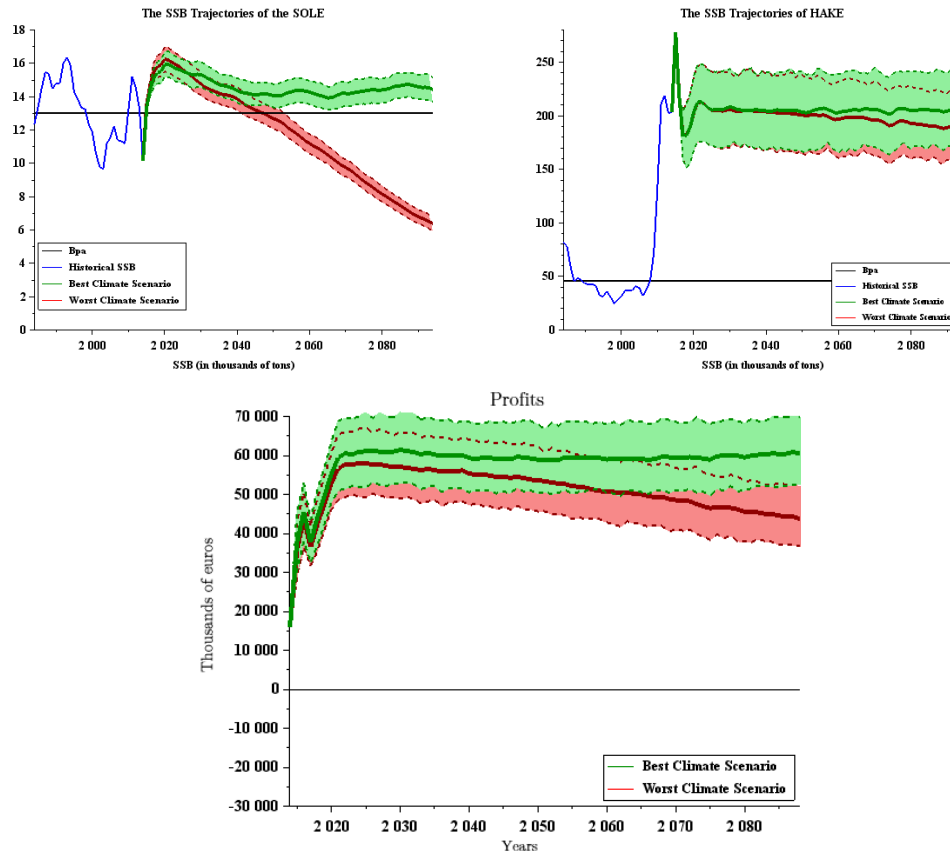


Figure II.8: Strategy MMEY - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.

As displayed by Figure 8, the MMEY strategy displays better bio-economic performances than the SQ and MMSY strategy.

Regardless of the climate scenario, the sole and hake SSB display the same trend than in the MMSY strategy (Figure II.8) but at higher levels for Sole. Nevertheless, for hake, the SSB remains at lower values whatever the climate scenario. Compared to the MMSY strategy, under the best climate scenario, profits are multiplied by more than two, reaching 60 million of euros per year. However, under the worst climate scenario, profits are lower (Figure II.8). This could be explained by the weak price of hake which leads the MMEY strategy to focus much more on Sole's profits and conservation. On the contrary, the MMSY strategy which aims at maximizing catches has a wider interest to protect hake, taking advantage of its high abundance.

The MMEY strategy emerges as the best way to mitigate climate change effects. In the next section, we elaborate on the explanation for such outcomes in terms of fishing efforts.

4.5 MMSY, MMEY : reduction of efforts especially for sole gill-netters

Type of fleet (number of vessels - $K_f(2008)$)	RCP 8.5		RCP 2.6	
	u_f^{MMSY}	u_f^{MMEY}	u_f^{MMSY}	u_f^{MMEY}
Mixed trawlers 0-12 m (110)	0	0.72	0.44	0.29
Mixed trawlers 12-16 m (45)	0.08	0.84	0	1.32
Mixed trawlers 16-20 m (49)	0.02	1.03	0.21	0.53
Mixed trawlers >20 m (37)	0.02	0.7	0.04	1.06
Sole gill-netters 0-10 m (28)	0.07	0.51	0.36	0.14
Sole gill-netters 10-12 m (42)	0.01	0.21	0.72	0.13
Sole gill-netters 12-18 m (40)	0.85	0.23	0.79	0.25
Sole gill-netters 18-24 m (23)	0.71	0.19	0.78	0.53
Mixed gill-netters 0-10 m (32)	0.22	0.74	0.66	0.44
Mixed gill-netters 10-12 m (30)	0.23	1.03	0.81	0.2
Mixed gill-netters 12-18 m (6)	0.74	0.64	0.66	0.36
Mixed gill-netters 18-24 m (9)	0.34	0.44	0.45	1.12
Mixed gill-netters >24 m (10)	0.49	0.96	1.12	0.83
Mean Landings (in thousands of tons)	142	137	148	144
Mean NPV (in millions of euros)	478	1242	658	1355

Table II.3: Fishing effort multipliers for MMEY and MMSY strategies with respect to the two climate scenarios. Numbers between brackets refer to the number of vessels in 2008 (Gourguet et al., 2013).

Table II.3 displays the different MMSY and MMEY fishing effort multipliers, mean landings and the NPV over the entire temporal horizon. Firstly, the MMSY and MMEY strategies imply an important mitigation of the number of boats as almost all the optimal multipliers (u_f^{MMSY} , u_f^{MMEY}) are smaller than 1. This is in line with the actual mitigation of the number of vessels on the Atlantic coast due to the CFP.

We also notice that, regardless of the climate scenarios, MMEY multipliers are globally higher than MMSY multipliers for mixed trawlers and mixed gill-netters. By contrast, MMEY multipliers for sole gill-netters are lower than MMSY multipliers for all climate scenarios². The choice of strategy has thus a major impact on the sole stock. In particular, the fishing effort multipliers of sole gill-netters plays a pivotal role for the sole stock.

Moreover, Table 3 indicates that climate change significantly affects the performances of these strategies as well as the computation of optimal management. In particular, in Table II.3, we ascertain that NPV and landings are lower under the worst climate scenario/RCP 8.5 with respect to each strategy.

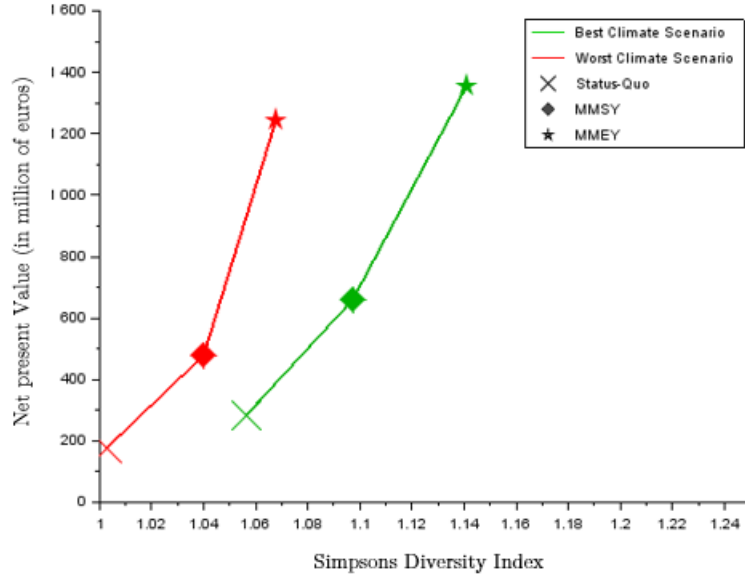


Figure II.9: Simpson's Index of Diversity versus Average Net Present Value.

4.6 Bioeconomic synthesis displays the MMEY as the best strategy

Figure II.9 synthesizes the bio-economic scores of the three management strategies through the average NPV of the entire temporal horizon on the Y-axis versus the Simpson's index of diversity³ on the X-axis. A Simpson's index close to 2 (because we have two species) means a more diversified ecosystem. By contrast, if the Simpson's index tends to one it means we have a lower level of diversity. The Figure shows an heterogeneous⁴ ecosystem which may be explained by a domination of one species on another (hake on sole) or a simple extinction of one species (in this case, Sole). Here, the values of Simpson index are weak < 1.2 indicating that diversity is at stake. This is due to low abundances of sole, which even collapses under the worst climate scenario for the SQ strategy.

²Except for the two smallest sub-fleets of the sole-gill netters fleet under the worst climate scenario. That is explained by their weaker contribution of the sole mortality (Figure II.12) than the two biggest sub-fleets

$$D = \left[\sum_{i=1}^2 \left(\frac{S\bar{S}B_i}{\sum_{s=1}^2 S\bar{S}B_i} \right)^2 \right]^{-1}$$

with $S\bar{S}B_i = \frac{1}{t_n - t_0} \sum_{t=t_0}^{t_n} SSB_i(t)$

⁴Some species are more abundant than others. In our case, it is Hake.

5. Discussion

In this last section, we discuss the benefits of managing a mixed fishery with the MMEY strategy and address specifically the question raised by the title of the paper : “How does MMEY mitigate bioeconomic effects of climate change for mixed fisheries”.

5.1 MMEY as an ecological and economic win-win strategy

Figure II.9 shows that the MMSY and the MMEY strategies improve both the ecological state and economic performance of the fishery as compared to the SQ strategy. Furthermore, the MMEY strategy yields bio-economic gains as compared to MMSY. This ranking $SQ < MMSY < MMEY$ (in the pareto sense) hold true for the two climate scenarios and as such is a win-win strategy. Such finding is aligned with general results obtained in Grafton et al. (2007) showing that under reasonable prices, costs, and discount rates, fishing at (dynamic) MEY promotes larger fish stocks and higher profits than fishing at MSY. Although Clark (1973) explains that maximizing NPV can lead to extinction if the discount rate r exceeds the intrinsic growth rate of the fishery, more recent studies (Grafton et al., 2010, 2012) have shown that bio-economic gains can occur for dynamic MEY even when the discount rate exceeds the intrinsic growth rate. This result applies in our case for several reasons including the recovery of hake in the past few years and the low discount rate used $r = 4$. Therefore, our economic strategy MMEY generates a positive effect on both fish stocks.

Nevertheless, there is no guarantee that the profit of each sub-fleet remains positive because we maximize the aggregated profits of all sub-fleets. Indeed fishing effort multipliers will be higher for the more profitable sub-fleets while less profitable sub-fleets will see their effort reduced in the MMEY strategy.

Moreover, the ecological gains of MMEY as compared to MMSY and SQ depend on climate change intensity. Under the highest climate change scenario, the simpson index gain is indeed very limited. This is due to the fact that the sole stock is strongly altered under this scenario as illustrated by Figure II.8 (top left).

5.2 Diversification of fleets produces greater benefits in the face of climate change

MMEY efforts draws on a strategy of diversification. First, we notice that fishing effort of the mixed trawlers and mixed gill-netters for the MMEY strategy are globally higher than those in MMSY strategy. That could explain why profits are higher for this latter strategy as compared to the MMSY strategy. On the contrary, fishing efforts of the sole gill-netters are globally lower for MMEY than for MMSY. This can be explained by the strong dependency¹ (Figures II.12 and II.10) of the sole gill-netters on the sole which MMEY efforts aims at avoiding because of the vulnerability of the sole due to global warming.

Therefore, many fleets relying on the sole may have to diversify their activities and change their targets especially if sole TAC and therefore landings continue to decrease as for almost 20 years (In appendix : Figure II.14). The price of sole has risen by 80% between 1994 and 2015 (In appendix : Figure II.13). This explains why sales in value remain high despite their limited share in volume² which is steadily decreasing (Aglia, 2014). Moreover, because of negative warming effects on targeted species, new commercial strategies and a reorganisation of the sector might be observed in the next years (Lagiere, 2012). This sectoral change may be restricted by a number

¹A high contribution to fishing mortality and a wide share in the overall income of the fleet.

²Sales in Volume = landings - unsold.

of factors : on the one hand, the French fleet is aging³ and the cost of renewing is high⁴ for new operators whose number has significantly decreased in the last years (In appendix : Figure II.11). On the other hand it is much more difficult for large vessels operators especially for sole gill-netters to adapt their fishing gears (Lagiere, 2012). Conversely, small vessels are already using 2 to 3 different gears per year. With the introduction of European regulations on discards, using case-by-case solutions for fleets to deal with discarding and gearing patterns appears to be one key of success (Morandea et al., 2014).

5.3 Perspectives

With this study, we underline the importance to integrate both multi-species, multi-fleet nature of fisheries as well as upstream temperature and more specifically SST in recruitment models and in management models of fisheries. This in line with Hughes et al. (2005) who claim : “restoring marine [...] ecosystems after they have degraded is much more difficult than maintaining them in good condition”. Such an ecosystem policy has already been tested by the Pacific Fishery management Council in 1998 in the management of sardine stocks (*Sardinops Sagax*). The council adopted a control of fishing depending on temperature increase (Hill et al., 2011).

More specifically, our results illustrate the potential of strategies aimed at MMEY to entail greater adaptation capacity in the face of climate change. This result is to place in the context of policy objectives which at the moment largely focus on mono-MSY or MMSY. So as to avoid harmful effects, global changes impacts have to be taken into account by regulating agencies which could be more efficient as almost half of mondial stocks are currently managed with the MSY method and as we have proved with the MMEY approach. This is in line with Chu and Kompas (2014) who claim that reaching the maximum economic yield, when combined with marines protected areas provides a better profitability and also ensures a higher conservation level regarding fish stocks.

In the medium to long-term horizon, it may be a large part of fleets which should adapt to changes induced by global warming. The large level of investment needed to rejuvenate the fleet and the decrease of quotas for some species will likely create major challenges in the future. Even if the increase in prices sustains profits, for the Bay of Biscay, governments or European institutions will have a crucial role to play.

The relevance of the adaptive, ecosystem and ecological-economic strategy advocated in our study should also draw on better knowledge on environmental changes. In that regard, the findings of Poloczanska et al. (2016) already observing a movement of marine species sensitive to warming towards the poles should be refined. In particular, the processes underlying the recruitment dynamics of the species are not fully understood yet such as the processes in survival success during the juvenile phase (Le Pape and Bonhommeau, 2015).

More globally, from a policy perspective, our results suggest to have an adaptative control of the fishery based on the economic, biological and social context and especially on the local and global environmental changes.

³The average age of French vessels is over 25 years old in 2012. Still in 2012, only 20% of the fleet was under 15 years. In contrast, almost 57% of the ships had more than 25 years (Aglia, 2014).

⁴Furthermore for big vessels.

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⁵ScENarios of bioeconomic VIability and RESilience for ecosystem-based fisheries management in Aquitaine

⁶Scenario, fishEry, ecologicAl-economic modelling and VIability nEtWork

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Appendix



Figure II.10: Fishing effort multipliers (left axis) and contribution to the overall fishing mortality of Sole in percentage (right axis). Sub-fleets (X-axis) are ranked by contribution to sole mortality with SGN=Sole gill-netters, MGN=Mixed gill-netters and MT=Mixed Trawlers. The first Figure accounts for the best climate scenario (RCP 2.6) and the second (bottom) for the worst climate scenario.

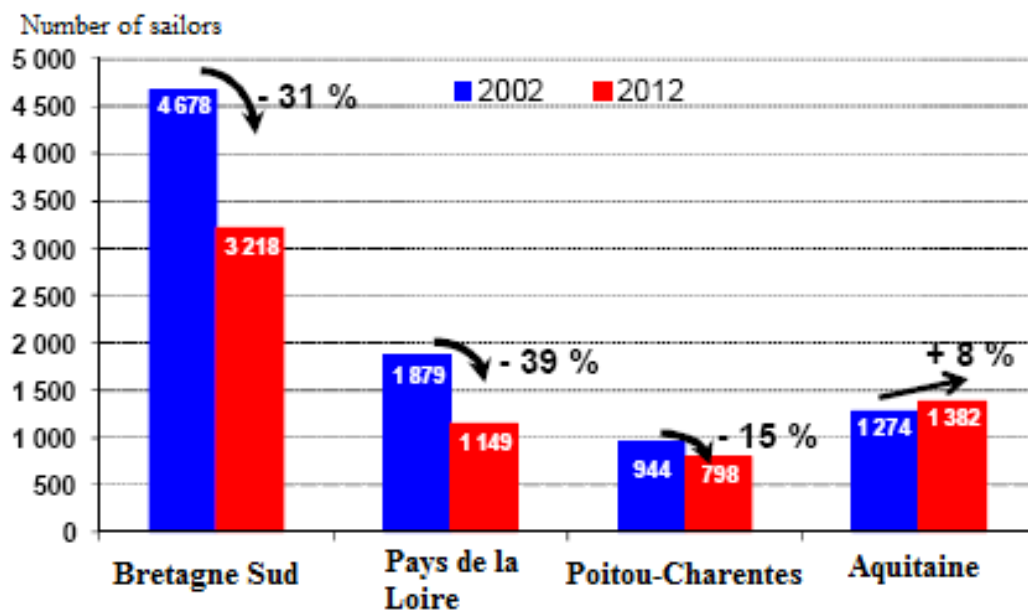


Figure II.11: Numbers of sailors in the Atlantic facade (Aglia, 2014)

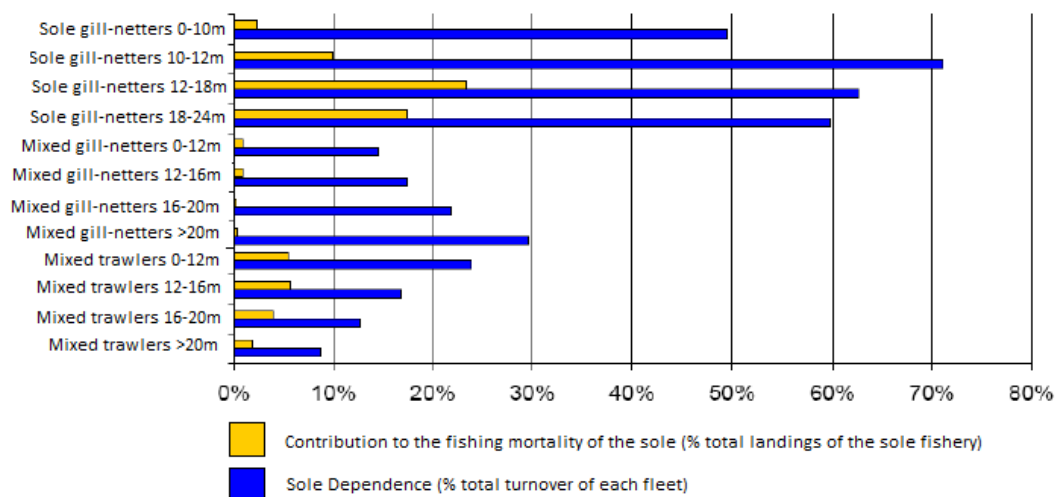


Figure II.12: Contribution to the fishing mortality and dependence on sole of fleets of Bay of Biscay fishery in 2010 (Aglia, 2014)

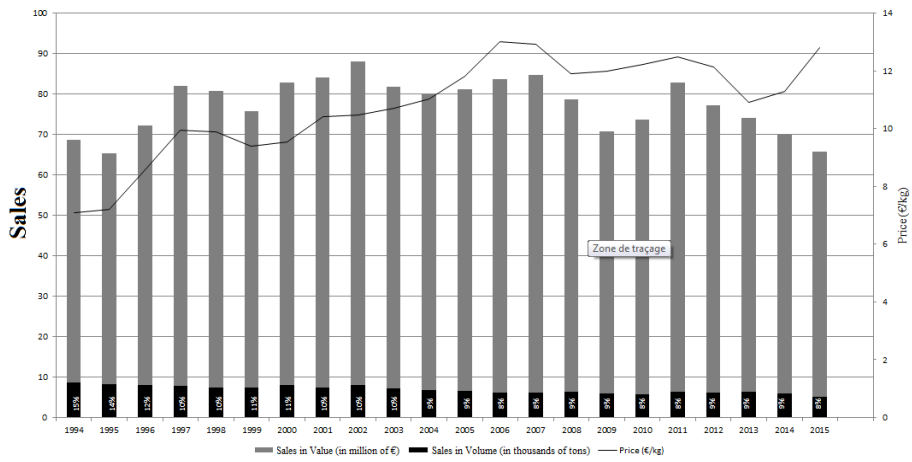


Figure II.13: Sales in volume and in value of the sole in all auction centres (histogram, left vertical axis) and price curve (right vertical axis). The percentages represent the part of sales in volume compared to sales in value. (data source : visionet.franceagrimer.fr)

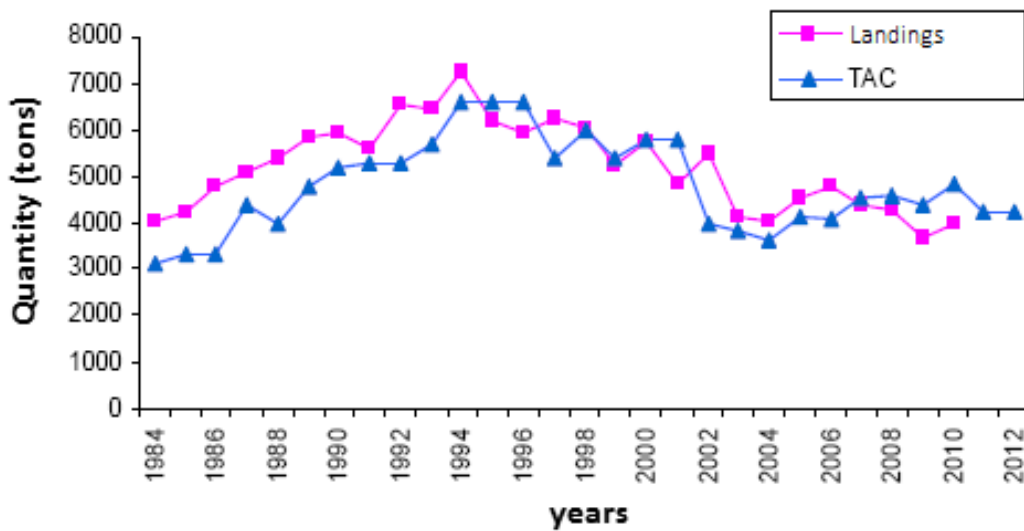


Figure II.14: Comparative evolution of the TAC and landings of the sole in the Bay of Biscay since 1984 (Lagiere, 2012)

<u>Cushing</u> (-3.31 0.56)	$x_{i,1}(t+1) = aSSB_i(t-\tau_i)^b \theta(t-\tau_i)^c + \varepsilon_i(t)$ (II.15)
<u>Ricker</u> (-2.5 0.25)	$x_{i,1}(t+1) = aSSB_i(t-\tau_i)e^{+bSSB_i(t-\tau_i)} + \varepsilon_i(t)$ (II.16)
Ricker 2(-3.37 0.66)	$x_{i,1}(t+1) = aSSB_i(t-\tau_i)e^{-bSSB_i(t-\tau_i)-c\theta(t-\tau_i)^2} + \varepsilon_1(t-\tau_i)$ (II.17)
<u>B-H</u> (-3.08 0.10)	$x_{i,1}(t+1) = \frac{SSB_i(t-\tau_i)}{b+aSSB_i(t-\tau_i)} + \varepsilon_i(t)$ (II.18)
<u>B-H 2</u> (-3.79 0.57)	$x_{i,1}(t+1) = \frac{SSB_i(t-\tau_i)}{b+aSSB_i(t-\tau_i)+c\theta(t-\tau_i)+d\theta(t)^2} + \varepsilon_i(t)$ (II.19)

Table II.4: Type of Stock-Recruitment models with and without environmental factor (θ) affecting recruitment according to the specie ($i = 1, 2$ respectively Sole, Hake). Numbers between brackets correspond respectively to the AIC criterion and R^2 associated to the model. The underlined models do not satisfy one or several associated statistical tests (test de White, Chow, Jarque and Bera and Durbin and Watson).

Age a	2	3	4	5	6	7	8+
Initial abund. $N_{1,a}(t_h)$ ($*10^3$ indv)	23191	17416	10707	4864	3425	2627	2590
Maturity $\gamma_{1,a}$	0.32	0.83	0.97	1	1	1	1
Mean weight (kg/indv) $v_{1,a}$	0.189	0.241	0.297	0.352	0.423	0.449	0.599
Natural mortality $M_{1,a}$	0.1	0.1	0.1	0.1	0.1	0.1	0.1

Table II.5: Sole parameters, ($i = 1$), $t_h = 2008$. Source: ICES; Ifremer, SIH, DPMA.

Age a	0	1	2	3	4	5	6	7	8+
Initial abund. $x_{2,a}(t_h)$ ($*10^3$ indv)	236062	132608	61571	25195	5219	1606	497	162	45
Maturity $\gamma_{2,a}$	0	0.11	0.73	0.93	0.99	1	1	1	1
Mean weight (kg/indv) $v_{2,a}$	0.029	0.25	0.716	1.572	2.503	3.452	4.393	5.773	6.747
Natural mortality $M_{2,a}$	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4

Table II.6: Hake parameters, ($i = 2$), $t_h = 2008$. Source: ICES; Ifremer, SIH, DPMA.

Sub-fleets	2	3	4	5	6	7	8+
Mixed trawlers 0-12 m	0.014	0.017	0.013	0.01	0.007	0.007	0.007
Mixed trawlers 12-16 m	0.014	0.018	0.014	0.012	0.013	0.013	0.013
Mixed trawlers 16-20 m	0.017	0.021	0.016	0.014	0.015	0.015	0.015
Mixed trawlers >20 m	0.007	0.009	0.007	0.006	0.007	0.006	0.006
Sole gill-netters 0-10 m	0.002	0.005	0.008	0.008	0.01	0.009	0.011
Sole gill-netters 10-12 m	0.011	0.028	0.042	0.045	0.053	0.052	0.059
Sole gill-netters 12-18 m	0.018	0.065	0.087	0.094	0.148	0.145	0.138
Sole gill-netters 18-24 m	0.015	0.054	0.072	0.078	0.123	0.121	0.115
Mixed gill-netters 0-10 m	0	0.001	0.002	0.002	0.002	0.002	0.002
Mixed gill-netters 10-12 m	0.001	0.003	0.005	0.005	0.006	0.006	0.007
Mixed gill-netters 12-18 m	0.001	0.003	0.004	0.004	0.006	0.006	0.006
Mixed gill-netters 18-24 m	0	0	0	0	0	0	0
Mixed gill-netters >24 m	0	0	0	0	0	0	0
Other Fleets	0.062	0.113	0.072	0.072	0.09	0.079	0.083

Table II.7: Values of fishing mortality on Sole ($i = 1$): $F_{1,a,f}(t_0)$. Source: ICES; Ifremer, SIH, 2008.

Sub-fleets	0	1	2	3	4	5	6	7	8+
Mixed trawlers 0-12 m	0.016	0.013	0.006	0.002	0.002	0.001	0	0	0
Mixed trawlers 12-16 m	0.018	0.015	0.007	0.002	0.003	0.001	0	0	0
Mixed trawlers 16-20 m	0.016	0.013	0.006	0.002	0.002	0.001	0	0	0
Mixed trawlers >20 m	0.011	0.009	0.004	0.001	0.002	0	0	0	0
Sole gill-netters 0-10 m	0	0	0	0	0.001	0	0	0	0
Sole gill-netters 10-12 m	0	0	0	0.001	0.002	0.001	0	0	0
Sole gill-netters 12-18 m	0	0	0	0.002	0.004	0.002	0.001	0	0
Sole gill-netters 18-24 m	0	0	0.001	0.005	0.008	0.004	0.001	0.001	0
Mixed gill-netters 0-10 m	0	0	0	0.001	0.002	0.001	0	0	0
Mixed gill-netters 10-12 m	0	0	0	0.001	0.002	0.001	0	0	0
Mixed gill-netters 12-18 m	0	0	0	0.002	0.004	0.002	0.001	0	0
Mixed gill-netters 18-24 m	0	0	0.005	0.025	0.044	0.023	0.008	0.003	0.002
Mixed gill-netters >24 m	0	0.001	0.013	0.067	0.119	0.062	0.022	0.009	0.005
Other Fleets	0.022	0.253	0.444	0.734	0.764	0.843	0.728	0.875	0.88

Table II.8: Values of fishing mortality on Hake ($i = 2$): $F_{2,a,f}(t_0)$. Source: ICES; Ifremer, SIH, 2008.

Fleets	Nb vessel	Fishing effort/vessel	Income from other species	Catches from other species
	$K_f(t_0)$	(nb day at sea) $e_f(t_0)$	(in €/effort unit) α_f	(in kg/effort unit) β_f
Mixed trawlers 0-12 m ($f = 1$)	110	157.7	622	201
Mixed trawlers 12-16 m ($f = 2$)	45	192.7	1375	429
Mixed trawlers 16-20 m ($f = 3$)	49	180.3	1751	490
Mixed trawlers >20 m ($f = 4$)	37	197.1	3597	1003
Sole gill-netters 0-10 m ($f = 5$)	28	139	311	71
Sole gill-netters 10-12 m ($f = 6$)	42	145.5	503	115
Sole gill-netters 12-18 m ($f = 7$)	40	202.9	765	162
Sole gill-netters 18-24 m ($f = 8$)	23	201.7	1150	251
Mixed gill-netters 0-10 m ($f = 9$)	32	153.8	303	59
Mixed gill-netters 10-12 m ($f = 10$)	30	178.8	847	173
Mixed gill-netters 12-18 m ($f = 11$)	6	145	1466	339
Mixed gill-netters 18-24 m ($f = 12$)	9	210.3	1500	348
Mixed gill-netters >24 m ($f = 13$)	10	260.6	1141	346

Table II.9: Initial number of vessels $K_f(t_0)$, effort by vessel $e_f(t_0)$, rate of extra fishing income α_f and rate of extra fishing catches β_f of the thirteen sub-fleets . Source: Ifremer, SIH, DPMA, 2008

Fleets	Landing cost	Volume of fuel	Variable cost by vessel	Annual cost by vessel
	τ_f	(in L/effort unit) V_f^{fuel}	(in €/effort unit) c_f^{var}	(in €) c_f^{fix}
Mixed trawlers 0-12 m ($f = 1$)	0.05	257	44	77779
Mixed trawlers 12-16 m ($f = 2$)	0.05	863	108	218506
Mixed trawlers 16-20 m ($f = 3$)	0.07	1076	188	245285
Mixed trawlers >20 m ($f = 4$)	0.07	1999	308	388951
Sole gill-netters 0-10 m ($f = 5$)	0.06	78	70	56601
Sole gill-netters 10-12 m ($f = 6$)	0.05	290	140	132326
Sole gill-netters 12-18 m ($f = 7$)	0.08	348	213	256373
Sole gill-netters 18-24 m ($f = 8$)	0.07	622	453	378872
Mixed gill-netters 0-10 m ($f = 9$)	0.05	59	28	42874
Mixed gill-netters 10-12 m ($f = 10$)	0.05	248	69	111911
Mixed gill-netters 12-18 m ($f = 11$)	0.06	396	230	223622
Mixed gill-netters 18-24 m ($f = 12$)	0.07	811	595	513353
Mixed gill-netters >24 m ($f = 13$)	0.03	1099	556	913096

Table II.10: Mean reference costs of the thirteen sub-fleets. Source: Ifremer, SIH, DPMA, 2008

Chapitre III

Ecological-economic resilience of a fished coral reef through stochastic multi-species MSY

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Abstract

This paper investigates the ecological-economic resilience of coral reef ecosystems under fishing and environmental pressures. To achieve this, a dynamic, spatially explicit, multi-species, multi-fleet model is developed. Stochastic environmental shocks are also assumed to alter coral cover and consequently the entire coral reef socio-ecosystem. The model is calibrated using ecological, economic and environmental data for Moorea (French Polynesia). Four exploratory fishing strategies and a goal-seeking strategy entitled SMMSY (Stochastic Multi-Species Maximum Sustainable Yield) are compared in terms of ecological-economic outcomes and resilience. The SMMSY turns out to promote ecological-economic resilience. It is first characterized by a global increase in fishing effort pointing out the relative current under-exploitation of the fishery. Secondly, SMMSY balances the trophic level of catches after natural shocks occurrences and sustains the fundamental herbivore grazing process. Fishing SMMSY strategies are also more diversified in terms of temporality, gears, location and targets. In a context of high food dependency, multi-criteria strategies pursuing socio-economic objectives along with ecosystem approach seem relevant for complex insular socio-ecosystems.

Key words: ecological-economics, biodiversity, ecosystems, scenarios, small-scale fisheries, sustainability, resilience, French Polynesia

1. Introduction

Indo-pacific coral reefs are among the most complex and diversified ecosystems worldwide. Most of these coral reefs are currently undergoing rapid changes (or degradations) due to an increase in both anthropogenic and natural disturbances (Nyström et al., 2000; Bellwood et al., 2004).

In particular, natural disturbances play an important role in the long-term dynamics of coral reefs, often causing periods of decline in coral cover and periods of recovery (Wilson et al., 2007; Adjeroud et al., 2010). Resulting of a giant human footprint, it is now assumed that their number might increase in intensity and occurrence within a near future (Cinner et al., 2013). Therefore, climate change and environmental shocks (e.g invasive species or cyclones) complicates and exacerbates the problems by creating, or intensifying existing risks, uncertainties and vulnerabilities (Cheung et al., 2010; Sumaila et al., 2011; FAO, 2018). As a consequence, ensuring the socio-economic sustainability of fisheries and preserving the marine biodiversity and ecosystems while facing uncertainty have become a major challenge for regional, national and international agencies (FAO, 2018). To implement these objectives, multidimensional and integrative management models are emerging, and that is why a wide number scientists advocate the use of ecosystem approach or ecosystem-based fishery management (EBFM) in order to take into account the various ecological and economic complexities at play (Pikitch et al., 2004; Doyen et al., 2017). These new ecosystem models are well described by Plagányi (2007) and are now used as a tool for building ecosystem scenarios¹. With the conception of ecosystem scenarios, we can then evaluate the ecological-economic outcomes that go along with them and identify the sustainable and resilient strategies.

We focus on a case study in French Polynesia, the Moorea’s coral reef. In this ecosystem, the interplay between coral and algae covers, the different trophic groups of fishes including corallivores, herbivores and piscivores along with migration of fish between the fore reef and lagoon are parts of the ecological complexity. The diversity of fishing techniques (diving, canoe, motorised boats) and tourism activities are parts of the anthropogenic complexity (Halpern et al., 2008; Fulton et al., 2011).

Our paper provides and compares in ecological-economic terms fishing scenarios for the management of coral fisheries in a context of environmental shocks. Hereafter, we denote these fishing scenarios as management strategies. In particular, a dynamic, spatially explicit, multi-species, multi-fleet model is developed and calibrated using ecological, economic and environmental data collected in Moorea. Stochastic environmental shocks are also assumed to alter coral cover and consequently the entire coral reef socio-ecosystem. From the calibrated model, four exploratory (forward-looking) and one normative fishing management strategies are compared in terms of ecological-economic outcomes and resilience management (Grafton et al., 2019). More specifically, we evaluate and compare biodiversity, catch and fishing effort performances of these management strategies. To do so, we adopt a multi-criteria viewpoint in line with the EBFM. The normative strategy entitled 'Stochastic Multi-Species Maximum Sustainable Yield (SMMSY) relies on the optimisation of expected harvests over time. This strategy is in line with some rare studies (Dalzell and Adams, 1997; Newton et al., 2007; McClanahan et al., 2016; McClanahan, 2018) which try to evaluate the sustainable food production per hectare for highly dependent regions such as insular populations. One major originality of our study is then to present stochastic results based on MSY approach for the entire area.

The rest of the paper is organized as follows. Section 2 presents the case study in French Polynesia. Section 3 describes the ecological-economic model as well as the five management strategies. Ecological-economic results are analysed in Section 4 and Section 5 across management strategies, discusses and concludes.

¹The creation of the IPBES (International Panel for Biodiversity and Ecosystem Services) at the interface between decision support and scientific knowledge aims at supporting these issues.

2. The Moorea Island small-scale fishery

Moorea Island (17°30'S, 149°50'W ; French Polynesia) is surrounded by a barrier reef characterized by a rich of its coral and fish diversity. Long of 61 km and wide of 750 m, the coral reef of Moorea has been subject to a long and steady historical monitoring and therefore displays one of the longest data set among the world coral reefs beginning in the early eighties (Galzin et al., 2016). Thus, it is a witness laboratory attesting the whole state of polynesian marine ecosystems.

All around the world, environmental shocks affect coral reef system. Depending on many physical and geographical features, the impacts differ from Pacific to Caribbean reefs (Nyström et al., 2000). However, small Pacific Islands globally share the same anthropogenic and environmental disturbances based on their isolated locations (Carlos et al., 2008). Thus, the following assumptions can then be further developed for other islands over the Pacific. Moorea island experienced a great number of disturbances through decades affecting the tridimensional structure of the reef (Lamy et al., 2016). Cyclones (1991, 2010) and crown-of-thorns seastar (COTS) outbreaks in particular (1979-1984, 2006-2009) and some bleaching event (1983, 1987, 1991) but not all caused recurrent declines and recoveries of coral cover and changes in the most abundant coral genera (Adjeroud et al., 2010). Despite these perpetual shift in coral assemblage along with changes in fish assemblages, fish functional diversity remains stable. Natural shocks plays a fundamental role in the dynamics of Moorea's reef as well as human population growth (Bellwood et al., 2004; Wilson et al., 2007). However, despite all the recurrent natural shocks experienced by Moorea's coral reef, it remains one of the most resilient ecosystem worldwide without any regime shift from a coral dominated-state to an algae dominated-state (Adjeroud et al., 2009; Traçon et al., 2011; Adam et al., 2011, 2014)

Regarding the demography, the growth in French Polynesia over the past decades, estimated to 2% by year, is important (Statistics Institute of French Polynesia , ISPF) although a slow-down in population growth is observed¹. Economically and socially, fishing activities are of key importance to local populations of Moorea as in most small islands of the Pacific (Bell et al., 2009; Leenhardt et al., 2012). However, the extent of the pressures entailed by fishing on coral reef ecosystems of Moorea, and the ensuing sustainability of observed fishing activities remains poorly known. Indeed, the strength of this anthropogenic pressure and in particular landing levels remains uncertain: for many reasons, the few analysis (Aubanel, 1993; Yonger, 2002; Kronen et al., 2006; Brenier, 2009) attempting to assess the global fishing catches of Moorea suggest very different results over the years whereas fishermen ascertain to harvest the same amount of fish than before (Leenhardt et al., 2012). A large proportion of the production is self-consumed² but the rest is sold by the so-called 'tui'³ to contribute to cash of households or communities. The weight of a tui is about 3 kilos and is sold around 1500 XPF⁴. At the outset of the 90s, Aubanel (1993) observe the same price than nowadays (pers. observation). Therefore, the socio-economic system relating to fishing appears to display a very stable pattern. It is only recently that more detailed analysis of tui composition and prices has been undertaken (Moussa, 2010), showing the potential for some heterogeneity in the prices of individual species, with implications for the changes in the composition of landings over time, following environmental perturbations and impacts of fishing. Furthermore, we must consider the Moorea Island as a closed system: importations and exportations of lagoon fish are negligible (Brenier, 2009).

In this study, we are focusing on the thirteen areas defined by the previous Maritime Space Management Plan⁵ (2005-2019) now gathered in five committees (Figure III.1). Biological data

¹Less than 1% between 2007 and 2017 (ISPF, 2017).

²More than 50% in average according to social surveys.

³A tui is "A wreath of fish consisting of one or more species, tied together with plant fibre drawn through their gills and then suspended on a metal holder, which forms the sales unit." (Leenhardt et al., 2012)

⁴Around 12€.

⁵PGEM (Plan de Gestion de l'Espace Maritime) in french - (Salvat et al., 2008)

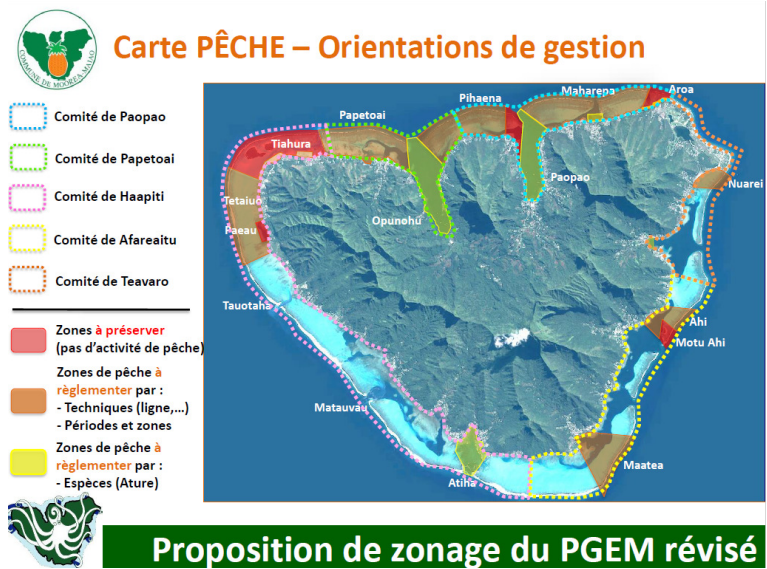


Figure III.1: Moorea Island and its new PGEM decomposed in five committees

such as fish density, % of cover of habitats, area in km² stem from the long-term monitoring carried out by the CRIOBE⁶ within the framework of the PGEM from 2005 to 2016. Economic data (capturability coefficients, fishing effort, fleets composition) are obtained from the PROC-Fish report (2010), through ANR ACROSS project and surveys which have been done in Maatea (One of the Moorea's commune).

⁶<http://www.criobe.pf/>

3. The ecological-economic model

To address the issues mentioned in the introduction, a dynamic, spatially explicit, multi-species (functional groups) and multi-fleet model is developed. Environmental adverse events are also assumed to alter coral cover and consequently the entire coral reef socio-ecosystem dynamics.

3.1 Conceptual model

The conceptual ecological-economic model of Figure III.2 illustrates the main interactions between the ecological, economic and social components of the fishery. The Table III.2 in the Appendix informs on the number of species underlying the different functional groups considered in the model. It also provides the notations used hereafter for these groups hereafter.

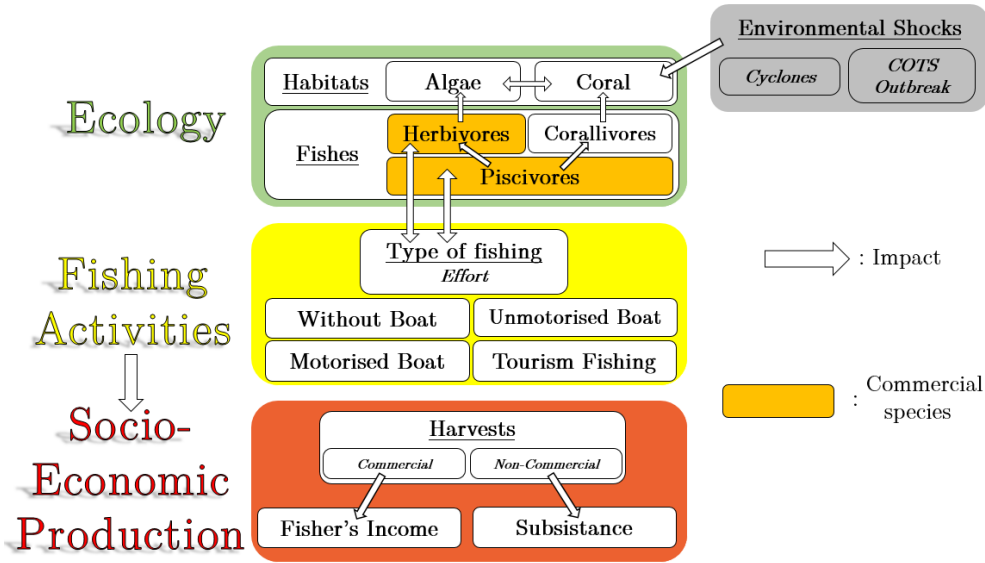


Figure III.2: Conceptual Model

3.2 Mathematical model

We now describe the ecological-economic model in mathematical terms. It is inspired by Mumby (2006); Doyen et al. (2007) and Dubois et al. (2017). The dynamics of the five functional groups are assumed to be governed by Lotka-Volterra type interactions including trophic (for fishes) or competition (for habitats) mechanisms. The model is spatially explicit with two patches to account for the movements of fishes between the Lagoon and the Fore Reef. In each patch p (Lagoon or Fore Reef), we now describe three distinct dynamics: the first one for the Coral (Eq. (III.1)), the second for Algae (Eq. (III.2)) and the third one for the three functional groups of fish (Eq. (III.6) (III.6) (III.7)) namely the piscivores, the herbivores and the corallivores. The stochasticity in the model arises from shocks (cyclones, cots outbreaks) damaging the coral cover on the fore reef.

Habitat dynamics:

The dynamics of coral cover $x_{1,p}(t)$ in patch p at time t reads as follows:

$$x_{1,p}(t+1) = x_{1,p}(t) \left(1 + r_1 - s_{1,1}x_{1,p}(t) - s_{1,2}x_{2,p}(t) - s_{1,5}x_{5,p}(t) \right) (1 - \Delta r_p(\omega(t))) \quad (\text{III.1})$$

where r_1 stands for the intrinsic growth rate of the coral cover, $s_{1,1} > 0$ relates to carrying capacity of coral while $s_{1,2} > 0$ represents the competition between coral and algae cover $x_{2,p}$ on the same patch p and $s_{1,5} > 0$ corresponds to the trophic interaction between corallivore biomass $x_{5,p}$ in patch p and coral. The variable rate $\Delta r_p(\omega(t))$ captures the environmental disturbances (cots outbreak, cyclones) on coral cover in patch p depending on stochastic scenarios $\omega(t)$. We assume that shocks only impact the coral group on the fore reef as it has been observed during these historical perturbations described in section 2.

The dynamics of algae cover $x_{2,p}(t)$ in patch p at time t reads as follows:

$$x_{2,p}(t+1) = x_{2,p}(t) (1 - s_{2,3}x_{3,p}(t)) \quad (\text{III.2})$$

where $s_{2,3} > 0$ relates to the trophic interaction between herbivore biomass $x_{3,p}$ in patch p and algae. $x_{2,p}(t)$ is also constrained by a surface inequality:

$$x_{2,p}(t) \leq g (K_p - x_{1,p}(t)) \quad (\text{III.3})$$

where the parameter g stands for the percentage of reef grazed (only algae) as in Mumby (2006) and K_p represents the carrying capacity for habitats in patch p i.e the entire available area which can be colonized by algae and coral (canal and sandy floor/depth excluded¹), in other words K_p is carrying capacity for habitats in patch p .

Fish dynamics:

The dynamics of corallivores $x_{5,p}(t)$ in patch p at time t reads as follows:

$$x_{5,p}(t+1) = x_{5,p}(t) \left(1 + r_5 - \gamma_p(x_{1,p}(t))s_{5,4}x_{4,p}(t) + s_{5,1}x_{1,p}(t) \right) \quad (\text{III.4})$$

where parameter $s_{5,1} > 0$ stands for the consumption of coral by corallivores. Parameter $s_{5,4} > 0$ stands for the predation of piscivores on corallivores and $\gamma_p(x_{1,p}(t))$ represents the refuge effect as explained in equation (III.5). In that regard, following Doyen et al. (2007), we capture the effect of habitat refuge for predation intensity by the difference between the carrying capacity of Coral (maximum of Coral cover depending on patches) and the percentage of Coral cover at time t such as:

$$\gamma_p(x_{1,p}(t)) = K_p - x_{1,p}(t) \geq 0 \quad (\text{III.5})$$

where $K_p - x_{1,p}(t)$, also presented above in eq. (III.2), stands for the non-coral area (where finding a refuge is harder for grazers/small herbivores).

Regarding the dynamics of mobile and harvested fish groups, namely herbivores $x_{3,p}(t)$ and piscivores $x_{4,p}(t)$ in patch p at time t , we first write it without accounting for dispersion mechanisms between the lagoon and the fore reef as follows:

$$\begin{cases} \widetilde{x_{3,p}}(t+1) = x_{3,p}(t) \left(1 + r_3 + s_{3,2}x_{2,p}(t) - \gamma_p(x_{1,p}(t))s_{3,4}x_{4,p}(t) \right) - H_{3,p}(t) \\ \widetilde{x_{4,p}}(t+1) = x_{4,p}(t) \left(1 + r_4 + \gamma_p(x_{1,p}(t))s_{4,3}x_{3,p}(t) - s_{4,4}x_{4,p}(t) + \gamma_p(t)s_{4,5}x_{5,p}(t) \right) - H_{4,p}(t) \end{cases} \quad (\text{III.6})$$

where $H_{3,p}(t)$ and $H_{4,p}(t)$ are the fishing harvests of herbivores and corallivores respectively. Again r_3 and r_4 stand for the intrinsic growth rates of the herbivores and piscivores. Parameter $s_{3,2} > 0$ represents the consumption by herbivores of algae. Symetric parameters $s_{3,4} > 0$ and $s_{4,3} > 0$ stand for the predation of herbivores by piscivores and $s_{4,4} > 0$ for the intra-specific competition i.e cannibalism within piscivores group. Again $\gamma_p(x_{1,p}(t))$ models the refuge effect generated by the coral $x_{1,p}(t)$.

¹ K_{lagoon} equals 50% for the lagoon and K_{reef} is set at 98% for the fore reef.

Spatially explicit dynamics

In order to capture the spatial movements of fishes between the lagoon and the fore reef, a diffusion factor depending on the mobile species and on the patch is implemented as in Costello and Kaffine (2016). The parameters $d_{i,p,p'}$ stand for the share of the biomass of group i which moves from patch p to patch p' . We assume that herbivores and piscivores are mobile species while the corallivores remain within their initial patch.

$$\begin{cases} x_{3,p}(t+1) = \sum_{p'=lagoon}^{reef} d_{3,p',p} \widetilde{x_{3,p'}}(t+1) \\ x_{4,p}(t+1) = \sum_{p'=lagoon}^{reef} d_{4,p',p} \widetilde{x_{4,p'}}(t+1) \end{cases} \quad (\text{III.7})$$

Fishing

The fishery production is represented by a standard Schaefer function where harvest is a linear function of both fishing efforts and resource biomass as follows:

$$H_{i,f,p}(t) = q_{i,f,p} e_{f,p}(t) L_f(t) x_{i,p}(t) \quad i = 3, 4 \quad (\text{III.8})$$

Here the parameter $q_{i,f,p}$ measures the catchability of group i by fishing pattern/fleet f in patch p . The fishing effort $e_{f,p}(t)$ is the number of hour per year spent by one fisherman of the fleet f in patch p while $L_f(t)$ stands for labor² engaged within fleet f (in number of fishers per km²).

Uncertainty

We assume that shocks only impact the coral group on the fore reef as it has been observed during these historical perturbations described in section 2. In other words,

$$\Delta r_{lagoon}(t) = 0.$$

We also assume that the environmental shocks $\Delta r_{reef}(t)$ follow a Bernoulli distribution of parameter μ and p . Therefore, the probability of the natural disturbance μ ($0 < \mu < 1$) on the fore reef for the coral is described by:

$$\mathbb{P}(\Delta r_{reef}(t) = \mu) = p, \quad \mathbb{P}(\Delta r_{reef}(t) = 0) = 1 - p \quad (\text{III.9})$$

where μ is the magnitude of shocks on coral reef. The Bernoulli distribution is supposed to be independant and identically distributed (i.i.d).

Uncertainty is here expressed by a probability of occurrence of a natural disturbance. We assume that COTS outbreak and cyclone (which are the two main shocks that affect coral cover) have the same impact based on historical observations and in order to simplify the model.

3.3 Calibration

We calibrate the model using ecological, economic and environmental data from French Polynesia and in particular Moorea island (Kronen et al., 2006; Lamy et al., 2016). The ecological parameters to estimate include the intrinsic growth rates r_i of groups, the interaction coefficients $s_{i,j}$ between groups and the diffusion rate $d_{i,p,p'}$ of mobile species between patches for each species i . The method to estimate the ecological parameters consists in minimizing the mean square error

²Historical Labor (2005-2016) is presented in Table III.4. Estimated labor is assumed to grow linearly following an estimated demographic rate noted l_f : $L_f(t+1) = l_f * L_f(t_h)$ with $t_h = 2016$.

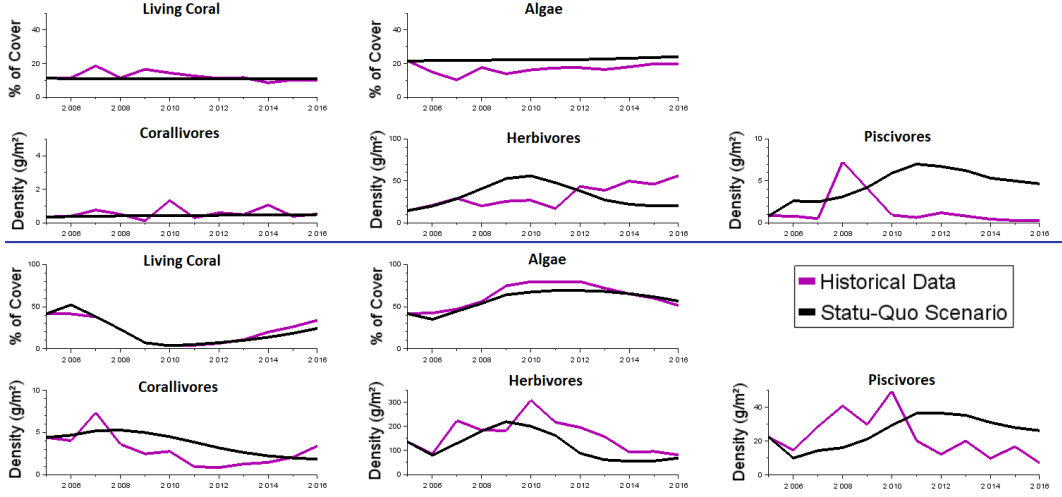


Figure III.3: Calibration - The five first figures (top) account for the lagoon and the five following figures (bottom) stand for the fore reef.

between the annually observed $x^{hist}(t)$ and simulated states $x(t)$ by the model. We rely on data from different zones z along the coast in Moorea as follows:

$$\min_{r,s,d} \left[\sum_{t=t_0}^{t_h} \sum_i \sum_p \sum_z \left(x_{i,p,z}^{hist}(t) - x_{i,p,z}(t) \right)^2 \right] \quad (\text{III.10})$$

with $t_0 = 2004$ and $t_h = 2016$ stands for first and the last year of observed data. The nonlinear optimization problem in equation (III.10) was solved numerically using the Scilab³ software and algorithms based on genetic optimization (optim_ga).

From Moorea's monitoring, we must consider that external shocks such as cyclones, COTS outbreak or bleaching event have a major impact only on coral cover on the fore reef⁴.

In Appendix, Table III.3 summarizes the different biological parameters of the model while Table III.4 displays the fishing parameters.

The stochasticity parameters relating to the coral dynamics and equation (III.9) are $\mu = 0.7$ and $p = 10\%$.

Regarding calibration dynamics, the deviation from the collected data is better on the fore reef than on the lagoon where an inertia is observed. This may be explained by the fact that other ecological drivers apply in the lagoon. In particular, other human activities (terrigenous inputs, physical development, navigation and tourism, ...) whose nature is not known negatively impact the piscivores notably in the second part of the historical period.

Furthermore, the smaller size of the fore reef implies a higher overall density (fishes) and cover (habitats), it is then difficult to compare them on the same scale. Moreover, collected data (notably for herbivores and piscivores) on the fore reef are likely to be more biased given the very high density and diversity in this particular patch.

³<https://www.scilab.org/>

⁴The biological dimension embeds these phenomena through a forcing of the coral cover in 2006, 2007, 2008 (COTS) and 2010 (cyclone OLI) to its historical and observed % of cover

$$x_{1,reef}(t_{shock}) = x_{1,reef}^{hist}(t_{shock}) \quad \text{with} \quad t_{shocks} = [2007, 2008, 2009, 2010]$$

3.4 Fishing strategies

We here consider different fishing strategies in order to compare them in terms of simulated trajectories over the period $t_h = 2016$ to $t_n = 2049$. The three first are more theoretical and no really applicable in management's models but allow us to explore how the system responds to shocks depending on the closing features implemented hereafter. Following Doyen (2018) and IPBES (2018), we distinguish between exploratory and goal-seeking strategies for fishing.

Closure (C) strategy:

The first exploratory strategy named Closure bans fishing for all fleets and patches p as follows:

$$e_{f,p}^C(t) = 0 \quad \forall t = t_h + 1, \dots, t_n \quad \forall f, p$$

This theoretical strategy makes it possible to focus on the ecological dynamics of the ecosystem in the long run.

Closure on lagoon and a 100% increase in effort on the fore reef (2R):

The second exploratory strategy bans fishing on the lagoon and double the fishing effort on the fore reef to compensate the closure within the lagoon:

$$e_{f,lagoon}^{2R}(t) = 0 \quad e_{f,reef}^{2R}(t) = 2 * e_{f,reef}(t_h) \quad \forall t = t_h + 1, \dots, t_n \quad \forall f, p$$

Closure on fore reef and a 100% increase in effort on the lagoon (2La):

Symmetrically, the third strategy explores a situation where fishing is stopped on the fore reef while it doubles within the lagoon:

$$e_{f,lagoon}^{2La}(t) = 2 * e_{f,lagoon}(t_h) \quad e_{f,reef}^{2La}(t) = 0 \quad \forall t = t_h + 1, \dots, t_n \quad \forall f, p$$

Status-Quo (SQ) strategy: The fourth exploratory fishing strategy entitled Status-Quo (SQ) maintains fishing efforts at the current level (of year $t_h = 2016$) throughout the period of interest for every fleet f and every patch p . In other words, it assesses the current way of managing the fishery such as:

$$e_{f,p}^{SQ}(t) = e_{f,p}(t_h) \quad \forall t = t_h + 1, \dots, t_n \quad \forall f, p$$

Stochastic Multi-species Maximum Sustainable Yield (SMMSY) strategy: This strategy differs from the exploratory fishing strategies (such as SQ) in the sense that it is a goal-seeking (or normative) strategy aiming at achieve specific goals in the future. The objective of this SMMSY strategy is to reach a maximum sustainable yield over all harvested species, that is to say, to maximize long-term landings induced by the different types of fishing f . To account for the uncertainty of shocks $\Delta r_{reef}(t)$ affecting coral reef dynamics in equation (III.1), we consider the expected value of the sum of the landings over time as follows:

$$\max_{e(\dots)} \mathbb{E} \left[\sum_{t=t_h+1}^{t_n} \left(\sum_i \sum_f \sum_p H_{i,f,p}(t) \right) \right] \quad (\text{III.11})$$

In this strategy, we assume that only two fishing effort choices e_0 et e_1 are identified at the start of two periods t_h and t_1 . Moreover, to deal with feedback controls and uncertainties, the choice of these two efforts relies on a 'non-anticipative' control strategy (Shapiro et al., 2009; Doyen et al., 2017).

4. Results

Figures III.4 to III.8 show the ecological trajectories of the fish functional groups aggregated over the two patches for the five different effort scenarios. In appendix, Figure III.11 depicts the dynamics of Coral and Algae cover. Figure III.9 (a) displays catch performances of the four fleets and (b) captures the global catches trajectories over time. Each strategy is then ecologically and economically analysed. The five management strategies are displayed using a color code (Green: Closure, Blue: 2R, Red: 2La, Grey: SQ, Brown: SMMSY). Prior to results' presentation, we need to notice that the greater variability observed for the herbivore group is explained by a higher spawning stock and density in comparison to other functional groups and by a likely higher bias in historical data due to its abundance.

4.1 Closure (CL): Limited ecological gains and not socially sustainable

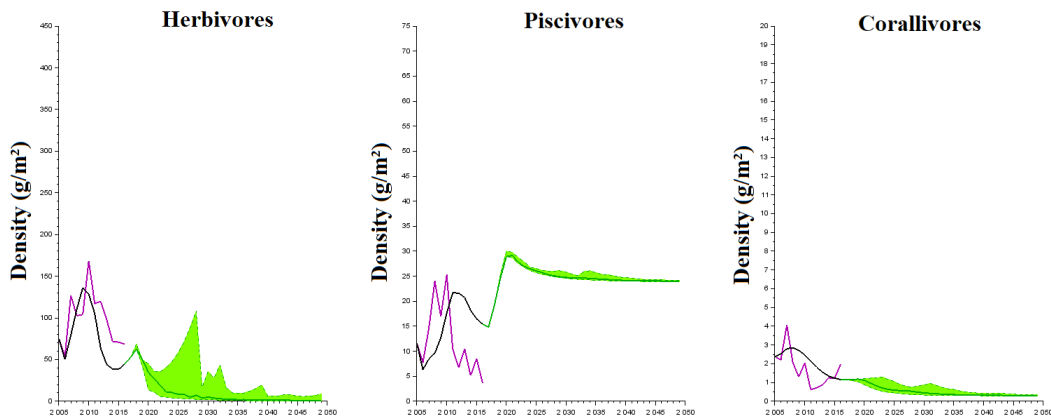


Figure III.4: Closure strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon - Fore Reef) fish densities. The 100 simulated trajectories are represented by the green area. The solid lines display the average of these 100 trajectories.

Drawn in green on Figure III.4, the Closure projections are characterized by the same habitat dynamics than the SQ, while it significantly differs for fish trajectories namely the herbivores and piscivores groups. It turns out that closing every zone to fishing does not clearly improve the ecological performances of the ecosystem and in particular fish states. This may sound counterintuitive; The explanation is that the fishing ban reinforces the top predators namely the piscivores which alters the herbivores because of trophic interactions. Moreover, this closure strategy is obviously not economically and socially viable given the high fishing dependency of the human population.

4.2 Closure on lagoon and + 100% on reef (2R): Ecologically limited and not socially sustainable

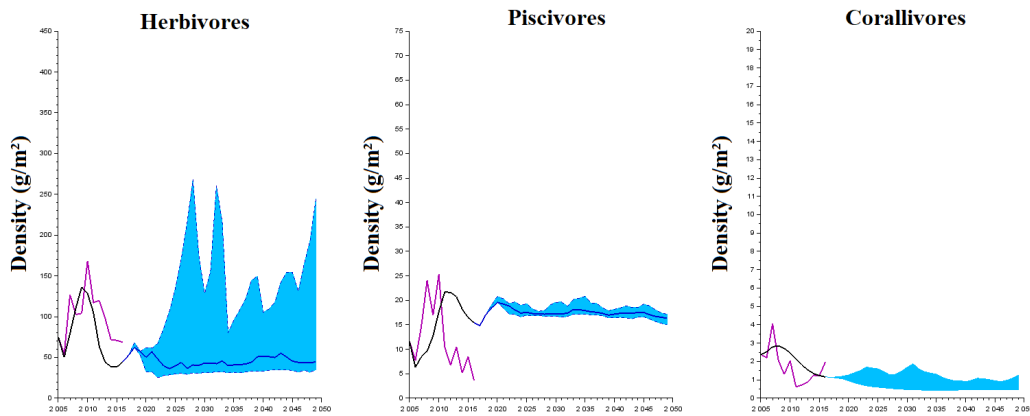


Figure III.5: 2R strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the blue area. The solid lines display the average of these 100 trajectories.

As shown by trajectories in blue on Figure III.5, closing only the lagoon induces similar trends as compared to the closure strategy CL. Indeed, given the large fishing area that represents the lagoon, preventing fishing in the lagoon increases the top predator density and as a consequence strongly and negatively impacts the herbivore biomass. Economically, as displayed by Figures III.9, we see that the level of catches is lower than under the SQ strategy mainly because the lagoon represents 90% of the Moorea coastal ecosystem's.

4.3 Closure on fore reef and increase of 100% on lagoon (2La): Ecologically and socially sustainable

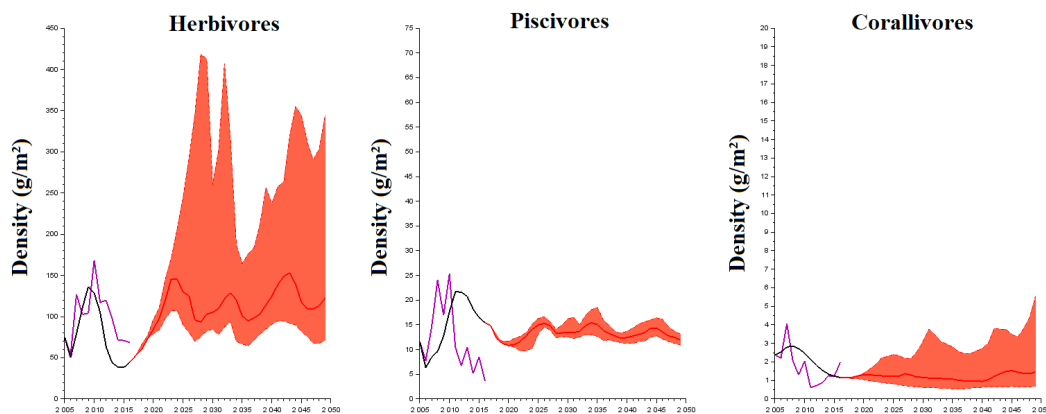


Figure III.6: 2La strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the red area. The solid lines display the average of these 100 trajectories.

As illustrated by trajectories in red on Figure III.6, closing the fore reef implies similar ecological consequences as compared to the SQ strategy. Within the fore reef, the piscivores density is a

slightly higher than in SQ and does not significantly affect the herbivore biomass. Regarding catches plotted in Figures III.9, the wideness of the lagoon and the refuge effect that emerges from the closing of the fore reef entails a high level of catches, notably higher than under the previous strategy. Therefore, stop fishing in the fore reef and reallocate this effort on the lagoon seems to be more sustainable both ecologically and economically than previous strategy 2R closing the lagoon.

4.4 Status-Quo (SQ): Ecologically and socially sustainable

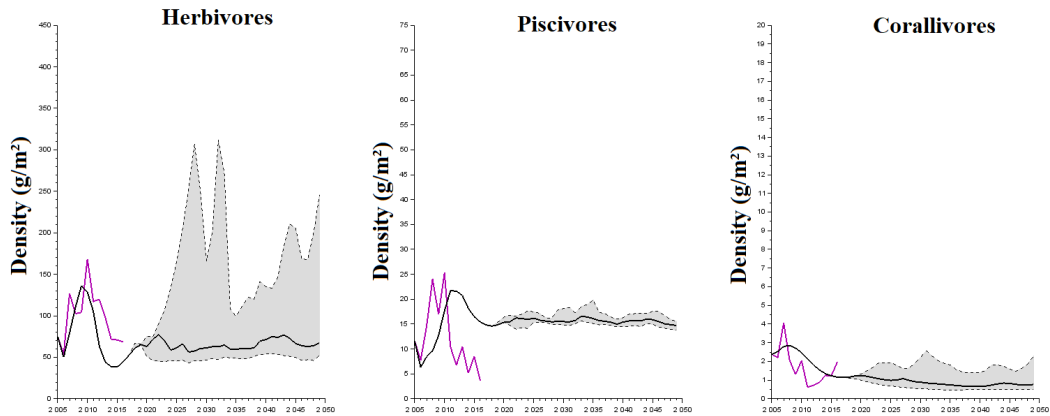


Figure III.7: Status-Quo strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.

Plotted in grey on Figure III.7, the Status-Quo strategy displays a relative diversified ecosystem. The density of commercial species remains at sustainable values. From the economic and fishing viewpoints as displayed by Figures III.9, catches increase throughout the time horizon. Thus, the potential demographic growth of human populations does not seem to alter too much the socio-ecosystem structure, sustainability and resilience.

4.5 SMMSY: Ecologically and socially sustainable

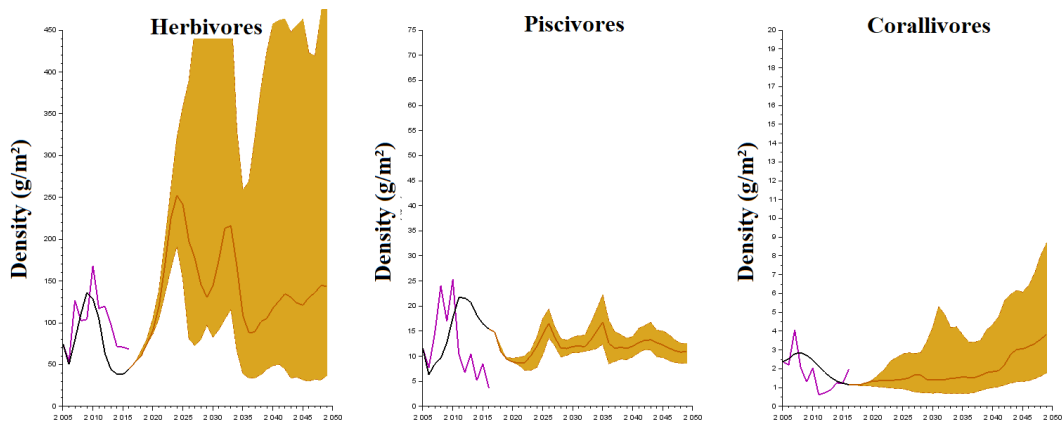


Figure III.8: SMMSY strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon - Fore Reef) fish densities. The 100 simulated trajectories are represented by the brown area. The solid lines shows the average of these 100 trajectories.

As highlighted by trajectories in brown on Figure III.8, the SMMSY strategy displays ecological paths similar to the previous strategy 2La but with a lower density of piscivores. From the economic and social viewpoints, figures III.9 show that catches are higher than all other strategies which is intuitive because it is the strategy maximising catches. Interestingly, strategy SMMSY performs at ecological and socio-economic levels close to the strategy 2La.

In Table III.1, we detail the SMMSY fishing effort evolution structure by type of fishing. The effort are gathered and averaged per geographical units underlying the case study. Furthermore, its evolution is here presented per capita¹. We clearly observe than the fishing effort increases.

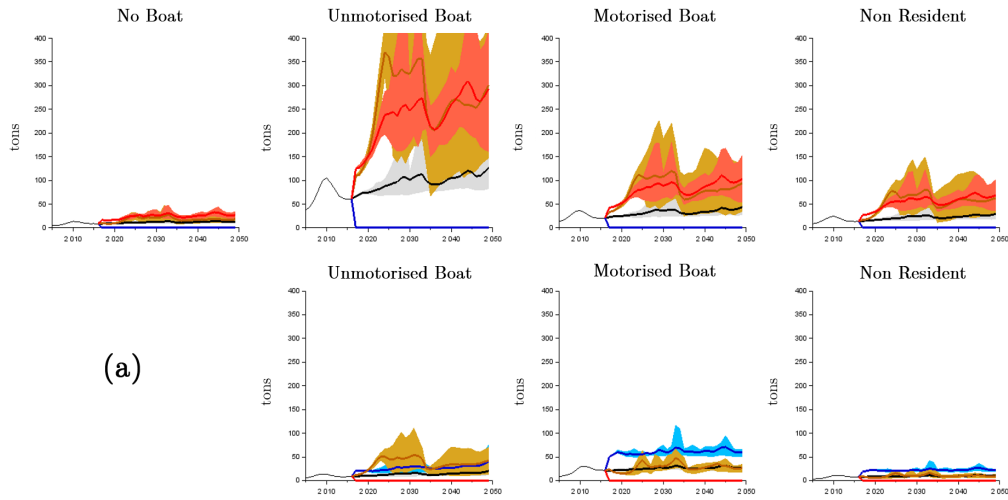
		Comité	Haapiti	Papetoai	Paopao	Tcavaro	Afarcaitu	
Lagoon	No Boat		39%	91%	-3%	22%	64%	43%
	Unmotorised Boat		51%	104%	49%	75%	89%	73%
	Motorised Boat		18%	97%	72%	46%	47%	56%
	Non Resident		50%	87%	-27%	72%	49%	46%
Reef	Unmotorised Boat		30%	75%	15%	47%	60%	45%
	Motorised Boat		68%	109%	-28%	93%	12%	51%
	Non Resident		56%	100%	8%	39%	60%	53%
			45%	95%	12%	56%	54%	

Table III.1: SMMSY Fishing Effort Evolution per comity Δe_{fc}

If we examine more closely the fishing effort structure, notably for the type of fishing, we notice that fishing effort is higher for unmotorised boat on the lagoon compared to other types of fishing. Such an outcome could be explained by a lower mobility (representing in this model by a lower catchability) of this fleet on the fore reef mainly because the fore reef is smaller and further (from the coast) than the lagoon. Thus, it is more profitable to increase the effort of this fleet in a patch where it has a better catchability i.e the lagoon.

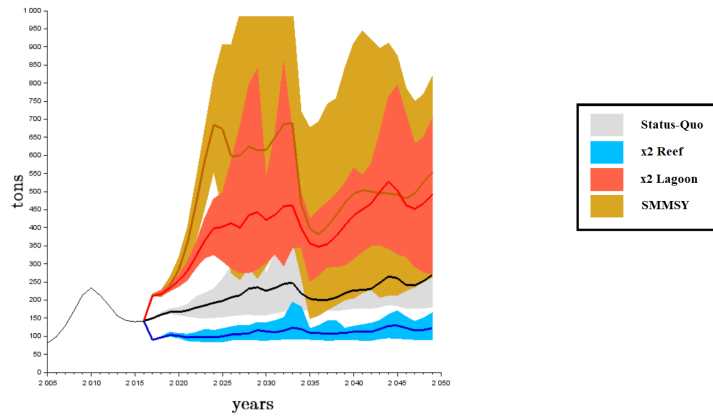
¹ $\Delta e_{fc} = \frac{e_{fc}(T) - e_{fc}(t_0)}{(1+l_f)^{T-t_0}}$ where $e_{fc}(T)$ stands for the estimated fishing effort per fleet f and per comity c in 2049 and l_f the demographic growth rate.

Catches per Fleets



(a)

Catches



(b)

Figure III.9: (a) Catches per fleets Trajectories: The 100 simulated trajectories are represented by the coloured areas depending on the management strategy. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef). (b) Aggregated Catches Trajectories: The 100 simulated trajectories are represented by the coloured areas depending on the management strategy. The solid lines display the average of these 100 trajectories.

5. Discussion

5.1 MICE modelling to operationalize ecosystem based fishery management

The model proposed in the paper is in line with Models of Intermediate Complexity (MICE) advocated in Plagányi et al. (2014); Doyen et al. (2017) to deal with ecosystem based fishery management. Indeed, as MICE, this model is question-driven, and aim to limit the complexity by restricting the focus to the minimum components and interactions needed to address the main effects of the management question under consideration. MICE are thus more focused than whole-of-ecosystem models, such as Atlantis (Fulton et al., 2011) or Ecopath with Ecosim (EwE - Christensen and Walters, 2004) although they integrate complex marine ecosystem dynamics under global changes, the economic processes underlying marine fisheries, and their interactions with marine ecosystem services and human well-being. They remain simple enough to allow easy adaptation and facilitate communication between disciplines and stakeholders, including non-scientists. In that respect, the model developped in our paper takes into account the major ecological complexities of coral reef ecosystems by articulating habitat dynamics, trophic interactions between fishes and environmental perturbations such COTS outbreaks. By putting emphasis on fishing outcomes, it also account for the economic and social drivers and the management issues. In that regard, the comparison between the different fishing strategies and the Status-quo gives important insights into potential improvements of the current situation, states and management.

5.2 SMMSY to promote ecological-economic resilience

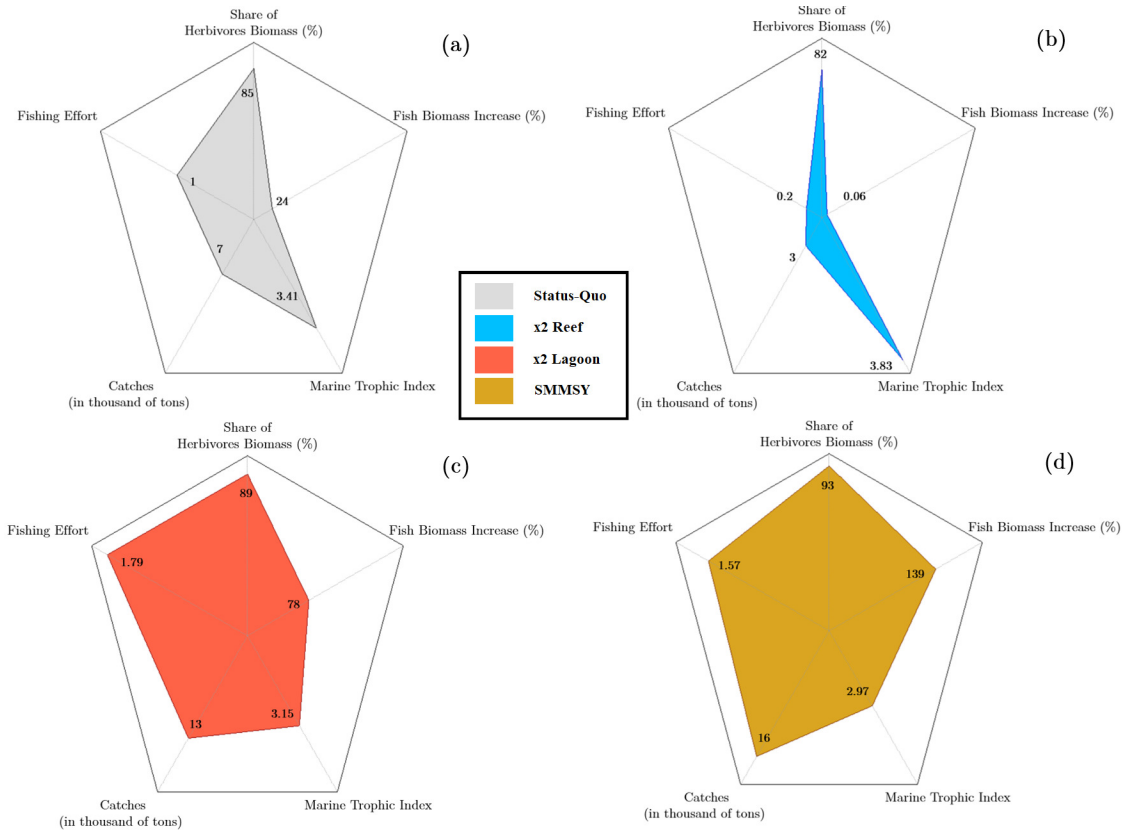


Figure III.10: Radar plot - The four radar plots are coloured according to the management strategies and the five components scales (SHB: 0-100%, Fishing Effort: 0-2, Catches: 0-20, Marine Trophic Index: 0-4, Fish Biomass Increase: 0-200%). Numbers within the radar plot represent the mean of the 100 simulations relating to its component. The five components are further described in Table III.5.

The normative strategy SMMSY turns out to be very informative in terms of ecological-economic reconciliation and resilience. By maximizing (expected) catches, such a strategy indeed sustains commercial species in the long run and also induces the persistence of the whole ecosystem components. Commercial species indeed represent a major share (>80%) of the entire ecosystem and play an essential ecological function for coral reefs such as grazing by herbivores (grazers). Therefore, the SMMSY emerges as an EBFM relevant to manage both large-scale (Lagarde et al., 2018) but also small-scale fishery such as coral reefs. In Figure III.10, we synthesize the ecological-economic scores (Share of Herbivores Biomass, Catches, Marine Trophic Index, Fish Biomass Increase) and display the average and global fishing effort evolution of the five¹ management strategies. If we use the surface of the radar plots (fishing effort excluded, so set at one to allow the comparison of surface), we can then rank the strategies in terms of these ecologico-economic results:

$$2R < SQ < 2La < SMMSY$$

In addition, we can notice that weighting the 5 index presented above according to a substitutability factor could improve the ranking's relevance. Nevertheless, implementing such a factor

¹The Closure scenario displays an empty radar plot except for the Share of Herbivores Biomass (51%) and as a consequence is not plotted here.

generates strong assumption of weak sustainability (Hartwick, 1977) which seem not really relevant nowadays (Stiglitz et al., 2009). Thus, we assume that the displayed scale, described in Figure III.10 commentaries and further addressed in Table III.5, are enough transparent to legitimize such a ranking with some realistic and reasonable upper and lower limits².

Regarding ranking's analysis, the SMMSY strategy performs better ecologically with a higher increase in fish biomass but also economically (or socially) with a stronger amount of catches and a lower effort than the 2La strategy which is the second best. With a greater share of herbivores, the grazing function is also preserved thus promoting the resilience of the ecosystem. Nonetheless, a lower Marine Trophic Index (MTI) indicates a potential tipping point where some high trophic level could be at risk. A spatially explicit management strategy such as the 2La scenario, could also deal with this issue (further addressed in 5.4). More globally, SMMSY strategy emerges as a resilient management strategy.

Using the resilience analysis and 3Rs metrics proposed in (Grafton et al., 2019), we can claim that the system is resilient because it **resists** and retains its functional and diversified identity ; it is also **robust** because it does not fall below a critical threshold where a functional group could be extinct ; and its **recovery time** is very low for coral (few years) and nul for herbivores and piscivores.

5.3 Fishing as a driver of resilience

Exemplified by the Closure scenario, a too strong decrease in fishing effort, within a coral reef can generate a high increase in top-predator abundance. Therefore, this entails a decline in preys notably in the herbivores group whose grazing function is an essential process driving the resilience of coral reef ecosystems (Martin et al., 2017). As it has been shown by Dubois et al. (2017), it could be beneficial to keep fishing during some specific period of time such as COTS outbreak notably because fishing leads to reduce abundance of predatory species, especially piscivores (Roberts, 1995). Thus, by maintaining a certain level of fishing notably by targeting top-predators, this enhances the grazer biomass and sustains the resilience of coral thanks to algae feeders. Thus, fishing can be a driver of resilience and may compensate the negative effect of natural disturbances on the coral reef ecosystem.

5.4 Diffusion and spatial management

Regarding spatial issues, the major role of the diffusion process between patches for mobile species and the interest of spatial management are emphasized by the performances of the 2La and 2R strategies. These fishing scenarios indeed rely on a spatially explicit management. Closing the lagoon (2R) and its 90% cover is obviously not sustainable economically for the fishery but also ecologically as shown by the blue radar plot Figure III.10. Closing the lagoon induces the same ecological conclusion than under the closure scenario: increase of top-predator abundance due to fishing ban in the lagoon resulting in a decrease in herbivores (Figure III.10 (b)).

Conversely, the 2La strategy displays better economic outcomes that even compete with the SMMSY, highlighted by the magnitude of the radar plots in Figure III.10. This latter strategy seems to generate a refuge effect in the fore reef which benefits to the top-predators. This is of course highly dependent on the assumed dispersion rates between the fore reef and the lagoon. Indeed, its Marine Trophic Index (the mean trophic level of catches) is higher than for the SMMSY ($3.15 > 2.98$). This patched-management strategy could then mitigate the “fishing

²We have here a likely and realistic maximum of 200% in fish increase, a MTI with a maximum representing the higher trophic level (4) in the ecosystem, a upper limit for catches which is sustainable even for catches as showed by Figures III.9 and an effort that is limited by fishing capacity. Furthermore, the fishing effort is displayed here only as an indicative component.

down” effect (Pauly et al., 1998) that seems to locally occur in Moorea with a decrease in catch sizes (Leenhardt et al., 2012).

5.5 Policy recommendations for Moorea Island

For many reasons, most of the MPAs implemented in Moorea have not produce biological benefits or in some zones, they even had a negative impact on the biomass (Lauric and Claudet, 2014). Thus, the Moorea socio-ecosystem’s does not need to rely on MPAs for most of the coastal zone. Nevertheless, there is a need to set up a management plan for a sustainable fishing activity. This has entailed the creation of a new Maritime Space Management Plan that will be applied in 2019/2020 (Figure III.1) where most of fully protected AMPs will be re-opened but with new more flexible restrictions.

This new management plan is in line with our work mainly because we found that the fishing effort can be increased notably by re-opening the MPAs but with some new regulatory standards which seem undoubtedly necessary. Indeed, we need to precise that uncertainty among calibration, data collection and (non-fully understood) environmental and ecological processes, shade the outcomes in many aspects. The wide colored fields in the trajectories’ figures attest, for instance, this environmental issue. However, the displayed eco-tendancies³ and management methods, discussed and analysed in this article, indicate some likely sustainable paths to follow. We put forward a new and more adaptative way of managing the fishery not only by zone but also by patches and in fonction of the type of fishing. As well as a management plan by patches could operate, a fishing effort decreased in some smaller zones and increased in bigger zones could generate the same positive effect, as long as we have a dispersal process between both.

More globally, this paper puts forward the interest of an adaptive management strategy relating to the socio-environmental context and in particular for coral reef social ecological systems. We argue that fishing strategies might be more efficient by being more diversified in terms of temporality, gears, location and targets and might promote resilience facing environmental uncertainties. Longer and more precise data sets will surely give us more precise informations on the small-scale fisheries management.

³Standing for ecological-economic tendancies.

Acknowledgment

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Appendix

	Groups	Index (i)	Number of Species
Habitats	Coral (C)	1	24
	Algae (A)	2	10
Fish	<u>Herbivores (H)</u>	3	64
	<u>Piscivores (P)</u>	4	48
	<u>Corallivores (Co)</u>	5	7

Table III.2: The five functional groups. Underlined groups stand for commercial and mobile species.

Groups Interaction Matrix (s_{ij})	Coral	Algae	Herbivores	Piscivores	Corallivores
Coral	0.009	0.006	0	0	0
Algae	0	0	0.0005	0	0
Herbivores	0	0.022	0	0.001	0
Piscivores	0	0	6.10^{-5}	0.019	0.008
Corallivores	0.0045	0	0	0.0055	0

Species	Intrinsic Growth Rate (r_i)	Diffusion Rate ($d_{ip'p}$)	
Coral	0.25/0.92	Lagoon to Reef	Reef to Lagoon
Algae	-	-	-
Herbivores	0.01	0.25	0.5
Piscivores	0.40	0.45	0.75
Corallivores	0.01	-	-

	Carrying Capacity (K_p)	
	Lagoon	Fore Reef
Coral/Algae	0.38	0.98

Table III.3: Calibrated parameters of the model, respectively (top) interactions matrix s_{ij} , (middle) species intrinsic growth rates r_i and the diffusion rates $d_{ip'p}$ of species i between patch p and p'. We also display (bottom) the carrying capacity K_p of the habitat i.e the total area where algae and coral can evolve.

Fisherman per Km^2 $L_f(t)$ in Maatea	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
No Boat	9	9	9	10	10	10	10	10	11	11	11	11
Unmotorised Boat	16	17	17	18	18	18	19	19	20	20	20	20
Motorised Boat	4	4	4	4	4	4	4	4	5	5	5	5
Non Resident	10	10	10	11	11	11	11	11	12	12	12	12
No Boat	0	0	0	0	0	0	0	0	0	0	0	0
Unmotorised Boat	9	9	9	10	10	10	10	10	11	11	11	11
Motorised Boat	9	9	9	10	10	10	10	10	11	11	11	11
Non Resident	6	6	6	7	7	7	7	7	7	7	7	7

Catchabilities q_{ifp}	Lagoon		Fore Reef		
	Fleets	Herbivores	Piscivores	Herbivores	Piscivores
No Boat		7.10^{-7}	3.10^{-5}	-	-
Unmotorised Boat		4.10^{-6}	4.10^{-5}	4.10^{-6}	2.10^{-5}
Motorised Boat		5.10^{-6}	4.10^{-5}	-	4.10^{-5}
Non Resident		5.10^{-6}	4.10^{-5}	-	4.10^{-5}

Table III.4: Fishing Parameters: (bottom) Catchabilities (top) q_{ifp} of fleet f on species i in patch p; (top) Fisherman per Km^2 in Maatea $L_f(t)$. The share of labor involved in each fleet as well as the global density of fishermen in all areas are extrapolated from this distribution and also regarding the surface of the corresponding zone.

Components	Equation and Remarks
Marine Trophic Index	$\frac{\sum MTL_i * H_i(t)}{\sum H_i(t)}$ where MTL_i is the Marine Trophic Level of group i and $\bar{H}_i = \sum_f \sum_p H_{ifp}$. Only commercial species (Piscivores, Herbivores) are included in the calcul.
Fish Biomass Increase	$\frac{\sum_i \sum_p \bar{x}_{ip} - \sum_i \sum_p \bar{x}_{ip}^{hist}}{\sum_i \sum_p \bar{x}_{ip}^{hist}}$ where $\bar{x}_{ip}^{hist} = \frac{\sum_{t=t_0}^{t_h} x_{ip}^{hist}(t)}{t_h - t_0}$ and $\bar{x}_{ip} = \frac{\sum_{t=t_h+1}^{t_n} x_{ip}(t)}{t_n - (t_h+1)}$ with $t_n = 2049$ This evolution index stands for the mean biomass increase of the simulated years (2017-2049) comparated to the historical average biomass (2005-2016) and only concerns commercial species.
Share of Herbivores Biomass	$\frac{\sum_p \bar{x}_{3,p}}{\sum_i \sum_p \bar{x}_{ip}}$. This index calculates the average share of herbivores (2017-2049) based on the simulated average biomass of fish (2017-2049) and concerns all 3 fish groups.
Fishing Effort	$\frac{\sum_{t=t_h+1}^{t_n} \sum_n \sum_f \sum_p e_{fp}(t,n)}{PF(t_n - t_0)N}$ where N is the number of simulation, F the number of fleet and P the number of patches. The fishing effort is the overall and temporal mean of the 100 simulations, 4 Fleets (F) and 2 patches.
Catches	$\sum_{t=t_h+1}^{t_n} \left(\sum_i \sum_f \sum_p H_{i,f,p}(t) \right)$ Catches are the temporal sum of the simulated years (2017-2049).

Table III.5: Radar Plot Components' equations and remarks.

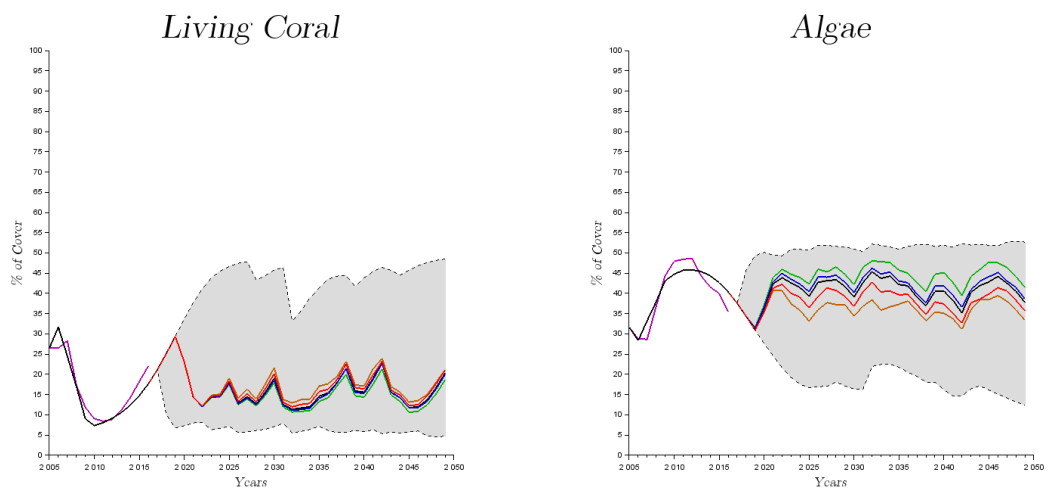


Figure III.11: Habitats Trajectories - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the colored area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.

Chapitre IV

To what extent fishing viability strategies promote ecological-economic resilience of coral reef facing climate change

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Abstract

This paper examines the ecological-economic viability of coral reef ecosystems under fishing and environmental drivers. This analysis draws on a dynamic, spatially explicit, multi-species, multi-fleet model accounting for stochastic environmental shocks altering coral cover and consequently the entire coral reef socio-ecosystem. The model is calibrated using ecological, economic and environmental data in French Polynesia. Then, a Status-Quo fishing scenario, a SMMSY (Stochastic Multi-Species Maximum Sustainable Yield) and three other goal-seeking strategies based on the viability approach (an ecological viability, a socio-economic viability and an eco-viability strategy) are compared in terms of projective ecological-economic outcomes, viability and resilience at the horizon 2050. The eco-viability strategy turns out to be the most relevant by promoting robustness and resistance. Interestingly, the SMMSY strategy also provides robust outcomes similar to the eco-viability. Nevertheless, it turns out that a too strong increase in natural shock occurrence, potentially induced by climate change, would mitigate the ecological-economic resilience of the fishery by endangering the recovering dimension of coral and generating a regime-shift from a coral to an algae dominated-state. Thus, the results question the choice of thresholds, in particular for coral.

Key words: ecological-economics, viability, resilience, sustainability, small-scale fisheries, ecosystems, scenarios, French Polynesia

1. Introduction

Indo-pacific coral reefs are among the most complex and diversified ecosystems worldwide. Most of them are currently undergoing rapid changes (or degradations) due to an increase in both anthropogenic and natural disturbances (Nyström et al., 2000; Bellwood et al., 2004).

In particular, natural disturbances play an important role in the long-term dynamics of coral reefs, often causing periods of decline in coral cover and periods of recovery (Wilson et al., 2007; Adjeroud et al., 2010). Resulting of a giant human footprint, it is now assumed that their number might increase in intensity and occurrence within a near future (Cinner et al., 2013). Therefore, climate change and environmental shocks (e.g. invasive species or cyclones) complicates and exacerbates the problems by creating, or intensifying existing risks, uncertainties and vulnerabilities (Cheung et al., 2010; Sumaila et al., 2011; FAO, 2018). As a consequence, ensuring the socio-economic sustainability of fisheries and preserving the marine biodiversity and ecosystems while facing uncertainty have become a major challenge for regional, national and international agencies¹. To implement these objectives, multidimensional and integrative management models are emerging, and that is why a wide number of scientists advocate the use of ecosystem approach or ecosystem-based fishery management (EBFM) in order to take into account the various ecological and economic complexities at play (Pikitch et al., 2004; Doyen et al., 2017). These new ecosystem models are well described by Plagányi (2007) and are now used as a tool for building ecosystem scenarios². With the conception of ecosystem scenarios, we can then evaluate the ecological-economic outcomes that go along with them and identify the sustainable, viable or resilient strategies.

The viability framework developed mathematically by Aubin (1991), is now widely studied by a growing number of researchers and is recognized as a relevant tool for EBFM (Doyen et al., 2017). In the context of dynamic systems, the aim of the viability approach is to explore states and controls that ensure the “good health” and safety of the system (Aubin, 1991; De Lara and Doyen, 2008). By identifying the viability conditions that allow constraints to be fulfilled over time, considering both present and future states of a dynamic system, the viability approach conveys information on sustainability (Baumgärtner and Quaas, 2009) and resilience (Béné et al., 2001; Deffuant and Gilbert, 2011). It accounts for dynamic complexities, uncertainties, risks and multiple sustainability objectives. With the integration of ecological, social and economic reference points, such an approach is now named *eco-viability* as in Cissé et al. (2015) and Doyen et al. (2017). However, this eco-viability framework, as many ecological-economic models, generates outcomes that might be altered by climate change effects and environmental shocks. Furthermore, the links between viability and resilience still deserve to be reinforced, although some scientists have found strong connections between dynamic system resilience and viability (Martin, 2004; Hardy et al., 2013; Doyen, 2018; Béné and Doyen, 2018).

Introduced by Holling (1973), the concept of resilience, in ecology, has since been defined in many ways (Mumby and Anthony, 2015). Grafton et al. (2019) attribute 3 dimensions to the resilience. Entitled the 3Rs, it refers to the Robustness, the Resistance and the Recovery. Following an adverse event, the system is ecologically resilient if it insures not to cross a critical point (robustness), if it actively preserves its identity or passively maintain its performances (resistance) and if the time to recover its desired features is very low or nul (recovery). In simpler words, a resilient socio-ecological system is a system that can withstand to a series of socio-economic shocks and whose the 3Rs are still maintained. These three concepts are particularly relevant for coral reef, and fisheries as a whole. In the case of coral reef, the herbivory process is considered as the key of resilience (Mumby et al., 2007; Adam et al., 2011; Holbrook et al., 2016; Martin et al., 2017).

¹For instance, an agenda with several Sustainable Development Goals at the 2030 horizon has been adopted by the United Nations (FAO, 2018).

²The creation of the IPBES (International Panel for Biodiversity and Ecosystem Services) at the interface between decision support and scientific knowledge aims at supporting these issues.

In this chapter, we attempt to evaluate to what extent fishing viability strategies, facing environmental stochastic shocks, can promote resilience within a complex, diversified and already resilient coral reef. To achieve this, a dynamic, spatially explicit, multi-species, multi-fleet model is developed. Four probabilities of natural shock occurrence are implemented. These disturbances are assumed to alter coral cover on the fore reef along with the entire socio-ecosystem. We are focusing on Moorea's coral reef by comparing and analyzing the ecologico-economic outcomes that emerge from the implemented stochastic management strategies through the viability approach and an assessment of the fishery's eco-resilience.

The paper is organized as follows: Section 2 presents the study case of Moorea, Section 3 describes the ecological-economic model along with the fishing scenarios, Section 4 analyses the ecological-economic results and Section 5 discusses and concludes.

2. The Moorea Island small-scale fishery

Moorea Island (17°30'S, 149°50'W ; French Polynesia) is surrounded by a barrier reef characterized by a rich of its coral and fish diversity. Long of 61 km and wide of 750 m, the coral reef of Moorea has been subject to a long and steady historical monitoring and therefore displays one of the longest data set among the world coral reefs beginning in the early eighties (Galzin et al., 2016). Thus, it is a witness laboratory attesting the whole state of polynesian marine ecosystems.

All around the world, environmental shocks affect coral reef system. Depending on many physical and geographical features, the impacts differ from Pacific to Caribbean reefs (Nyström et al., 2000). However, small Pacific Islands globally share the same anthropogenic and environmental disturbances based on their isolated locations (Carlos et al., 2008). Thus, the following assumptions can then be further developed for other islands over the Pacific. Moorea island experienced a great number of disturbances through decades affecting the tridimensional structure of the reef (Lamy et al., 2016). Cyclones (1991, 2010) and crown-of-thorns seastar (COTS) outbreaks in particular (1979-1984, 2006-2009) and some bleaching event (1983, 1987, 1991) but not all caused recurrent declines and recoveries of coral cover and changes in the most abundant coral genera (Adjeroud et al., 2010). Despite these perpetual shift in coral assemblage along with changes in fish assemblages, fish functional diversity remains stable. Natural shocks plays a fundamental role in the dynamics of Moorea's reef as well as human population growth (Bellwood et al., 2004; Wilson et al., 2007). However, despite all the recurrent natural shocks experienced by Moorea's coral reef, it remains one of the most resilient ecosystem worldwide without any regime shift from a coral dominated-state to an algae dominated-state (Adjeroud et al., 2009; Traçon et al., 2011; Adam et al., 2011, 2014).

Regarding the demography, the growth in French Polynesia over the past decades, estimated to 2% by year, is important (Statistics Institute of French Polynesia , ISPF) although a slow-down in population growth is observed¹. Economically and socially, fishing activities are of key importance to local populations of Moorea as in most small islands of the Pacific (Bell et al., 2009; Leenhardt et al., 2012). However, the extent of the pressures entailed by fishing on coral reef ecosystems of Moorea, and the ensuing sustainability of observed fishing activities remains poorly known. Indeed, the strength of this anthropogenic pressure and in particular landing levels remains uncertain: for many reasons, the few analysis (Aubanel, 1993; Yonger, 2002; Kronen et al., 2006; Brenier, 2009) attempting to assess the global fishing catches of Moorea suggest very different results over the years whereas fishermen ascertain to harvest the same amount of fish than before (Leenhardt et al., 2012). A large proportion of the production is self-consumed² but the rest is sold by the so-called 'tui'³ to contribute to cash of households or communities. The weight of a tui is about 3 kilos and is sold around 1500 XPF⁴. At the outset of the 90s, Aubanel (1993) observe the same price than nowadays (pers. observation). Therefore, the socio-economic system relating to fishing appears to display a very stable pattern. It is only recently that more detailed analysis of tui composition and prices has been undertaken (Moussa, 2010), showing the potential for some heterogeneity in the prices of individual species, with implications for the changes in the composition of landings over time, following environmental perturbations and impacts of fishing. Furthermore, we must consider the Moorea Island as a closed system: importations and exportations of lagoon fish are negligible (Brenier, 2009).

In this study, we are focusing on the thirteen areas defined by the previous Maritime Space Management Plan⁵ (2005-2019) now gathered in five committees (Figure IV.1). Biological data

¹Less than 1% between 2007 and 2017 (ISPF, 2017).

²More than 50% in average according to social surveys.

³A tui is "A wreath of fish consisting of one or more species, tied together with plant fibre drawn through their gills and then suspended on a metal holder, which forms the sales unit." (Leenhardt et al., 2012).

⁴Around 12€.

⁵PGEM (Plan de Gestion de l'Espace Maritime) in french - (Salvat et al., 2008)

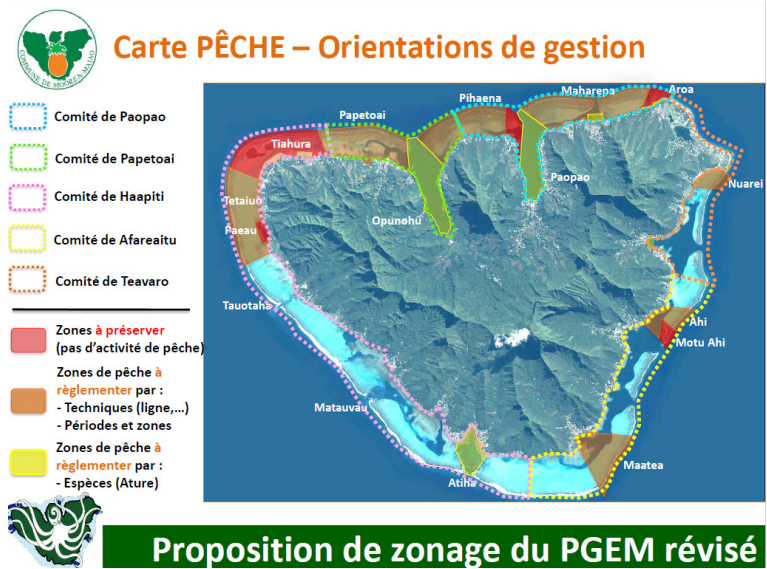


Figure IV.1: Moorea Island and its new former PGEM decomposed in five committees

such as fish density, % of cover of habitats, area in km² stem from the long-term monitoring carried out by the CRIOBE⁶ within the framework of the PGEM from 2005 to 2016. Economic data (capturability coefficients, fishing effort, fleets composition) are obtained from the PROC-Fish report (2010), through ANR ACROSS project and surveys which have been done in Maatea (One of the Moorea's commune).

⁶<http://www.criobe.pf/>

3. The ecological-economic model

To address the issues in the introduction, a stochastic, dynamic, spatially explicit, multi-groups and multi-fleet model is developed in the vein of Lagarde et al. (2020). Environmental adverse events are also assumed to alter coral cover and consequently the entire coral reef socio-ecosystem dynamics.

3.1 Conceptual model

The conceptual ecological-economic model of Figure IV.2 illustrates the main interactions between the ecological, economic and social components of the fishery. The Table IV.7 in the Appendix informs on the number of species underlying the different functional groups considered in the model. It also provides the notations for these groups hereafter.

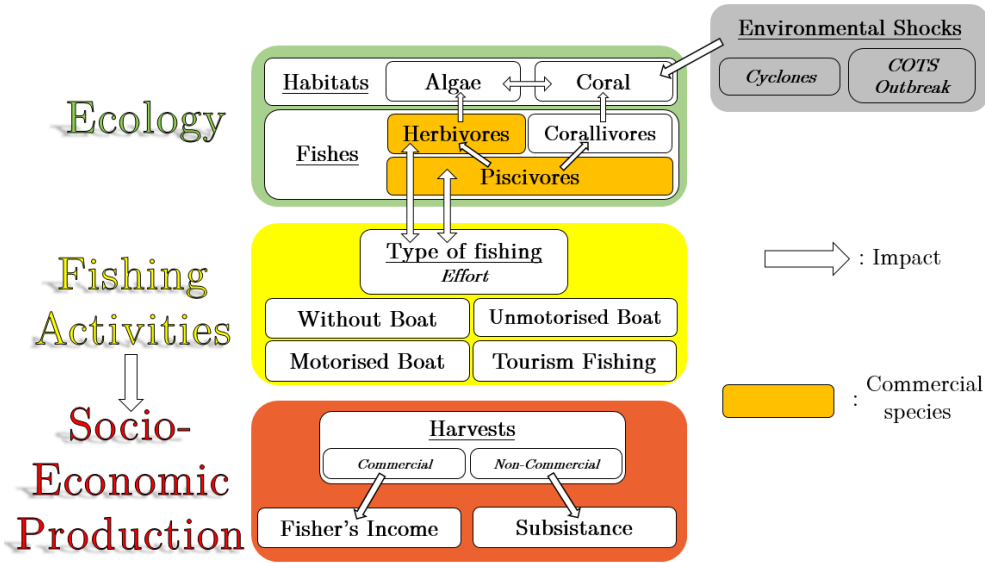


Figure IV.2: Conceptual Model

3.2 Mathematical model

We now describe the ecological-economic model in mathematical terms. It is inspired by Mumby (2006); Doyen et al. (2007); Dubois et al. (2017) and Lagarde et al. (2020). The dynamics of the five functional groups are assumed to be governed by Lotka-Volterra type interactions including trophic (for fishes) or competition (for habitats) mechanisms. The model is spatially explicit with two patches to account for the movements of fishes between the Lagoon and the Fore Reef. In each patch p (Lagoon or Fore Reef), we now describe three distinct dynamics: the first one for the Coral (Eq. (IV.1)), the second for Algae (Eq. (IV.2)) and the third one for the three functional groups of fish (Eq. (IV.4) (IV.6) (IV.7)) namely the piscivores, the herbivores and the coralivores. The stochasticity in the model arises from shocks (cyclones, cots outbreaks) damaging the coral cover on the fore reef.

Habitat dynamics

The dynamics of coral cover $x_{1,p}(t)$ in patch p at time t reads as follows:

$$x_{1,p}(t+1) = x_{1,p}(t) \left(1 + r_1 - s_{1,1}x_{1,p}(t) - s_{1,2}x_{2,p}(t) - s_{1,5}x_{5,p}(t) \right) (1 - \Delta r_p(\omega(t))) \quad (IV.1)$$

where r_1 stands for the intrinsic growth rate of the coral cover, $s_{1,1} > 0$ relates to carrying capacity of coral while $s_{1,2} > 0$ represents the competition between coral and algae cover $x_{2,p}$ on the same patch p and $s_{1,5} > 0$ corresponds to the trophic interaction between corallivore biomass $x_{5,p}$ in patch p and coral. The variable rate $\Delta r_p(\omega(t))$ captures the environmental disturbances (cots outbreak, cyclones) on coral cover in patch p depending on stochastic scenarios $\omega(t)$. We assume that shocks only impact the coral group on the fore reef as it has been observed during these historical perturbations described in section 2.

The dynamics of algae cover $x_{2,p}(t)$ in patch p at time t reads as follows:

$$x_{2,p}(t+1) = x_{2,p}(t) (1 - s_{2,3}x_{3,p}(t)) \quad (\text{IV.2})$$

where $s_{2,3} > 0$ relates to the trophic interaction between herbivore biomass $x_{3,p}$ in patch p and algae. $x_{2,p}(t)$ is also constrained by a surface inequality :

$$x_{2,p}(t) \leq g (K_p - x_{1,p}(t)) \quad (\text{IV.3})$$

where the parameter g stands for the percentage of reef grazed (only algae) as in Mumby (2006) and K_p represents the carrying capacity for habitats in patch p i.e the entire available area which can be colonized by algae and coral (canal and sandy floor/depth excluded¹), in other words K_p is carrying capacity for habitats in patch p .

Fish dynamics

The dynamics of corallivores $x_{5,p}(t)$ in patch p at time t reads as follows:

$$x_{5,p}(t+1) = x_{5,p}(t) \left(1 + r_5 - \gamma_p(x_{1,p}(t))s_{5,4}x_{4,p}(t) + s_{5,1}x_{1,p}(t) \right) \quad (\text{IV.4})$$

where parameter $s_{5,1} > 0$ stands for the consumption of coral by corallivores. Parameter $s_{5,4} > 0$ stands for the predation of piscivores on corallivores and $\gamma_p(x_{1,p}(t))$ represents the refuge effect as explained in equation (IV.5). In that regard, following Doyen et al. (2007), we capture the effect of habitat refuge for predation intensity by the difference between the carrying capacity of Coral (maximum of Coral cover depending on patches) and the percentage of Coral cover at time t such as:

$$\gamma_p(x_{1,p}(t)) = K_p - x_{1,p}(t) \geq 0 \quad (\text{IV.5})$$

where $K_p - x_{1,p}(t)$ also presented above in eq. (IV.2) stands basically for the non-coral area (where finding a refuge is harder for grazers/small herbivores).

Regarding the dynamics of mobile and harvested fish groups, namely herbivores $x_{3,p}(t)$ and piscivores $x_{4,p}(t)$ in patch p at time t , we first write it without accounting for dispersion mechanisms between the lagoon and the fore reef as follows:

$$\begin{cases} \widetilde{x_{3,p}}(t+1) = x_{3,p}(t) \left(1 + r_3 + s_{3,2}x_{2,p}(t) - \gamma_p(x_{1,p}(t))s_{3,4}x_{4,p}(t) \right) - H_{3,p}(t) \\ \widetilde{x_{4,p}}(t+1) = x_{4,p}(t) \left(1 + r_4 + \gamma_p(x_{1,p}(t))s_{4,3}x_{3,p}(t) - s_{4,4}x_{4,p}(t) + \gamma_p(x_{1,p}(t))s_{4,5}x_{5,p}(t) \right) - H_{4,p}(t) \end{cases} \quad (\text{IV.6})$$

where $H_{3,p}(t)$ and $H_{4,p}(t)$ are the fishing harvests of herbivores and corallivores respectively. Again r_3 and r_4 stand for the intrinsic growth rates of the herbivores and piscivores. Parameter $s_{3,2} > 0$ stands for the consumption by herbivores of algae. Symetric parameters $s_{3,4} > 0$ and $s_{4,3} > 0$ stand for the predation of herbivores by piscivores and $s_{4,4} > 0$ for the intra-specific competition i.e cannibalism within piscivores group. Again $\gamma_p(x_{1,p}(t))$ models the refuge effect generating by coral $x_{1,p}(t)$.

¹ K_{lagoon} equals 50% for the lagoon and K_{reef} is set at 98% for the fore reef.

Spatially explicit dynamics

In order to capture the spatial movements of fishes between the lagoon and the fore reef, a diffusion factor depending on the mobile species and on the patch is implemented as in Costello and Kaffine (2016). The parameters $d_{i,p,p'}$ stand for the share of the biomass of group i which moves from patch p to patch p' . We assume that herbivores and piscivores are mobile species while the corallivores remain within its initial patch.

$$\begin{cases} x_{3,p}(t+1) = \sum_{p'=lagoon}^{reef} d_{3,p',p} \widetilde{x_{3,p'}}(t+1) \\ x_{4,p}(t+1) = \sum_{p'=lagoon}^{reef} d_{4,p',p} \widetilde{x_{4,p'}}(t+1) \end{cases} \quad (IV.7)$$

Fishing

The fishery production is represented by a standard Schaefer function where harvest is a linear function of both fishing efforts and resource biomass as follows:

$$H_{i,f,p}(t) = q_{i,f,p} e_{f,p}(t) L_f(t) x_{i,p}(t) \quad i = 3, 4 \quad (IV.8)$$

Here the parameter $q_{i,f,p}$ measures the catchability of group i by fishing pattern/fleet f in patch p . The fishing effort $e_{f,p}(t)$ is the number of hour per year spent by one fisherman of the fleet f in patch p while $L_f(t)$ stands for labor² engaged within fleet f (in number of fishers per km²).

Uncertainty

We assume that shocks only impact the coral group on the fore reef as it has been observed during these historical perturbations described in section 2. In other words,

$$\Delta r_{lagoon}(t) = 0.$$

We also assume that the environmental shocks $\Delta r_{reef}(t)$ follow a Bernoulli distribution of parameter μ and p . Therefore, the probability of the natural disturbance μ ($0 < \mu < 1$) on the fore reef for the coral is described by:

$$\mathbb{P}(\Delta r_{reef}(t) = \mu) = p, \quad \mathbb{P}(\Delta r_{reef}(t) = 0) = 1 - p \quad (IV.9)$$

where μ is the magnitude of shocks on coral reef. The Bernoulli distribution is supposed to be independant and identically distributed (i.i.d).

Uncertainty is here expressed by a probability of occurrence of a natural disturbance. We assume that COTS outbreak and cyclone (which are the two main shocks that affect coral cover) have the same impact based on historical observations and in order to simplify the model.

3.3 Calibration

We calibrate the model using ecological, economic and environmental data from French Polynesia and in particular Moorea island (Kronen et al., 2006; Lamy et al., 2016). The ecological parameters to estimate include the intrinsic growth rates r_i of groups, the interaction coefficients $s_{i,j}$ between groups and the diffusion rate $d_{i,p,p'}$ of mobile species between patches for each species i . The method to estimate the ecological parameters consists in minimizing the mean square error

²Historical Labor (2005-2016) is presented in Table IV.9. Estimated labor is assumed to grow linearly following an estimated demographic rate noted l_f : $L_f(t+1) = l_f * L_f(t_h)$ with $t_h = 2016$.

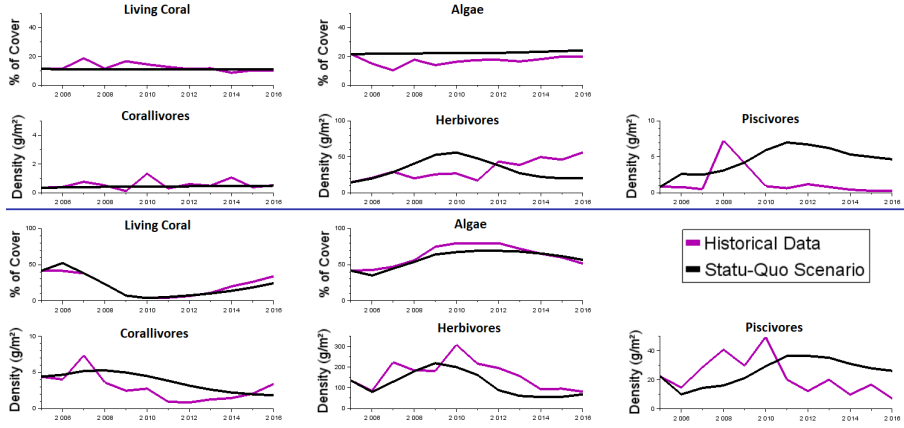


Figure IV.3: Calibration - The five first figures (top) account for the lagoon and the five following figures (bottom) stand for the fore reef.

between the annually observed $x^{hist}(t)$ and simulated states $x(t)$ by the model. We rely on data from different zones z along the coast in Moorea as follows:

$$\min_{r,s,d} \left[\sum_{t=t_0}^{t_h} \sum_i \sum_p \sum_z \left(x_{i,p,z}^{hist}(t) - x_{i,p,z}(t) \right)^2 \right] \quad (\text{IV.10})$$

with $t_0 = 2004$ and $t_h = 2016$ stands for first and the last year of observed data. The nonlinear optimization problem in Equation (IV.10) was solved numerically using the Scilab³ software and algorithms based on genetic optimization (optim_ga).

From Moorea's monitoring, we must consider that external shocks such as cyclones, COTS outbreak or bleaching event have a major impact only on coral cover on the fore reef⁴.

In Appendix, Table IV.8 summarizes the different biological parameters of the model while Table IV.9 displays the fishing parameters.

The stochasticity parameters relating to the coral dynamics and equation (IV.9) are $\mu = 0.7$ and $p = [10\%, 20\%, 30\%, 50\%]$.

Regarding calibration dynamics, the deviation from the collected data is better on the fore reef than on the lagoon where a inertia is observed. This may be explained by the fact that other ecological drivers apply in the lagoon. In particular, other human activities (terrigenous inputs, physical development, navigation and tourism, ...) whose nature is not known negatively impact the piscivores notably in the second part of the historical period.

Furthermore, the smaller size of the fore reef implies a higher overall density (fishes) and cover (habitats), it is then difficult to compare them on the same scale. Moreover, collected data (notably for herbivores and piscivores) on the fore reef are likely to be more biased given the very high density and diversity in this particular patch.

3.4 Viability assessment

The viability approach focuses on the safety and feasibility of controlled dynamics of the system with respect to constraints or targets representing the safety, good health, or sustainability

³<https://www.scilab.org/>

⁴The biological dimension embeds these phenomena through a forcing of the coral cover in 2006, 2007, 2008 (COTS) and 2010 (cyclone OLI) to its historical and observed % of cover

$$x_{1,reef}(t_{shock}) = x_{1,reef}^{hist}(t_{shock}) \quad \text{with} \quad t_{shocks} = [2007, 2008, 2009, 2010]$$

of the socio-ecosystem. These constraints can involve ecological thresholds as in the case of an extinction threshold in population viability analysis (PVA - Morris et al., 2002). Socio-economics constraints (guaranteed rent, food security, ...) can also be integrated as recently reviewed in Schuhbauer and Sumaila (2016); Oubraham and Zaccour (2018), thus allowing for multicriteria and bio-economic analyses. Such integrated viability objectives generally refer to a mix of the following ecological and economic constraints defined below.

3.4.1 Ecological viability

Firstly, ecological thresholds are here considered the minimum levels of living coral, herbivores and piscivores required for the conservation of respectively the refuge effect/barrier reef, the grazing fonctionnality, the top predators fonctionnality and denoted by x_i^{lim} with $i = [coral, herb, pisc]$. This reads:

$$x_{coral}(t) \geq x_{coral}^{lim} \quad \forall t \quad (IV.11)$$

$$x_{herb}(t) \geq x_{herb}^{lim} \quad \forall t \quad (IV.12)$$

$$x_{pisc}(t) \geq x_{pisc}^{lim} \quad \forall t \quad (IV.13)$$

where $x_{coral}(t)$, $x_{herb}(t)$, $x_{pisc}(t)$ are respectively the coral cover, herbivore biomass and piscivore biomass at time t aggregated for both patches and all zones. x_{coral}^{lim} , x_{herb}^{lim} , x_{pisc}^{lim} are respectively set at 5%, 35g/m², 5g/m².

These assumptions on ecological thresholds derive from many recent studies and historical observations carried on Moorea's coral reef (Lamy et al., 2016; Galzin et al., 2016; Dubois et al., 2017; Moritz et al., 2018).

Regarding the coral threshold x_{coral}^{lim} , setting a high value seems to be irrelevant in the case of Moorea's reef. Indeed, even with a very low level after the most severe disturbance that occurred in 2006/2009 (COTS outbreak), biologists observed an astonishing recovering of coral cover from 5 to 10 years (Lamy et al., 2016). It currently displays a wider coral cover than before the perturbation reaching 70% on the fore reef. This figure is still creating on-going researches. A question that arise here is: which level of shock occurrence endangers the resilience of Moorea coral reef ?

Given the historical resilience of Moorea's coral reef, we already observed that at a 35g/m² level (on an aggregated scale for both patches), the herbivore biomass can recover even with a low coral cover and is able to sustain the ecosystem viability as a whole. Above this limit, the essential grazing function of the herbivore group should be maintained.

Top predators (piscivores) density is set at 5g/m² mainly for similar reasons. At this level, despite environmental perturbations, the fonctionnality of this group is maintained and the ecosystem structure is likely not to be endangered.

Ecological Viability Probability (PVA)

Following the Population Viability Analysis (PVA) (Morris et al., 2002), we can calculate the probability⁵ of satisfying the ecological viability thresholds introduced previously, according to uncertainties $\Delta r_{reef}(t)$ and $\omega(t)$ such as:

$$\mathbb{P}((IV.11) (IV.12) (IV.13) \text{ holds for } t = t_h + 1, \dots, t_n) = \mathbb{E} \left[\prod_{t=t_h+1}^{t_n} \mathbf{1}_{PVA}(x(t), e(t), \omega(t)) \right] \quad (IV.14)$$

⁵Ctrs stands for Constraints. A confidence rate β is prescribed set at 90%.

with the indicator function:

$$\mathbb{1}_{PVA}(x, e, \omega) = \begin{cases} 1 & \text{if constraints (IV.11) (IV.12) (IV.13) are satisfied} \\ 0 & \text{otherwise} \end{cases}$$

and where $x(t)$ is the vector that contains all habitat and fish densities of Moorea's ecosystem, $e(t)$ corresponds to the vector which includes all fishing effort for all fleet operating in both patches.

3.4.2 Socio-economic viability

Secondly, a socio-economic viability is also considered in order to capture an objective of food security for insular populations. Accounting for both patches, we globally have 7 distincts fleets⁶. We here assume that 6 among the 7 types of fishing must maintain their catches per capita (i.e per fishermen) above the mean historical levels defined by H_f^{lim} in eq. (IV.16). The socio-economic threshold is then defined such as:

$$H_f(t) \geq H_f^{lim} \quad \text{for } f \geq 6 \quad \forall t \quad (\text{IV.15})$$

where $H_f(t)$ are the catches per capita of fleet f at time t computed as follows:

$$H_f(t) = \frac{1}{N_f(t)} \sum_{i,p} H_{ifp}(t) \quad \forall t$$

with $N_f(t)$ is the number of fishermen at time t .

Sociological surveys, conducted through the Across project, show a relative stability of catches among surveyed fishermen even though the estimation remains complex, uncertain and really different among studies (Leenhardt et al., 2012). In order to take into account the demographic context and to respect a social intergenerational equity, we set the thresholds of catches per capita. The socio-economic viability threshold⁷ denoted by H_f^{lim} is then defined by:

$$H_f^{lim} = \frac{1}{t_h - t_0} \sum_{t=t_0}^{t_h} H_f(t) \quad (\text{IV.16})$$

where t_h refers to the last historical year i.e 2016 and t_0 corresponds to the first historical year i.e 2005.

Socio-Economic Viability probability (EVA)

Following the Economic Viability Analysis (EVA) (Gourguet et al., 2013; Doyen et al., 2017), we can then calculate the probability⁸ of satisfying the socio-economic viability thresholds introduced above, according to uncertainties $\Delta r_{reef}(t)$ and $\omega(t)$ such as:

$$\mathbb{P}(Ctrs \text{ (IV.15) is fulfilled for } t = t_h + 1, \dots, t_n) = \mathbb{E} \left[\prod_{t=t_h+1}^{t_n} \mathbb{1}_{EVA}(x(t), e(t), \omega(t)) \right] \quad (\text{IV.17})$$

with the indicator function being:

⁶The “No boat” fleet operating only in the lagoon and the “unmotorised boat”, “motorised boat” and “non-resident” fleets operating in both patches. Thus, we have 4 fleets on the lagoon and 3 on the fore reef.

⁷We need to notice that we use the mean of the historical reconstructed catches (2005-2016) mainly because it aims better socio-economic performances than just crossing the historical minimum level of catches. We then attempt to generate better outcomes than the “no regrets” strategy i.e “at least better than before” (Doyen and Gajardo, 2019).

⁸*Ctrs* stands for Constraints. A confidence rate β is prescribed set at 90%.

$$\mathbb{1}_{EVA}(x, e, \omega) = \begin{cases} 1 & \text{if constraints (IV.15) are satisfied} \\ 0 & \text{otherwise} \end{cases}$$

This constraint aims at keeping the fishery active, and at sustaining the related social benefits of maintaining the activity among each fleet. Given that the fleets are distributed across two patches, this also ensures the maintenance of fishing operations and activities all along the coral reef. The objective, thus defined, is in fact akin to a social constraint, as it essentially requires that levels of economic incomes achieved by fleets allow these different types of fishing to continue participating in the fishery. In this sense, it is similar to the participation constraint defined by Pereau et al. (2012) in their bio-economic analysis of the effects of ITQ regulations on fisheries.

3.4.3 Eco-Viability (EcoVA) strategy:

Following Béné et al. (2001); Gourguet et al. (2013); Cissé et al. (2015); Doyen et al. (2017), we consider a viability framework embedding both ecological and socio-economic constraints presented above.

Thus, given the stochastic nature of the model (i.e. uncertainties on coral cover and direct/indirect trophic interactions as a result), the performance of management strategies must be assessed in terms of the probability for these constraints to be met by the fishery at any point in time. This probability⁹ is defined such as:

$$\begin{aligned} & \mathbb{P}(Ctrs \text{ (IV.11) (IV.12) (IV.13) (IV.15) are fulfilled for } t = t_h + 1, \dots, t_n) \\ &= \mathbb{E} \left[\prod_{t=t_h+1}^{t_n} \mathbb{1}_{EcoVA}(x(t), e(t), \omega(t)) \right] \end{aligned} \quad (\text{IV.18})$$

with the indicator function being:

$$\mathbb{1}_{EcoVA}(x, e, \omega) = \begin{cases} 1 & \text{if constraints (IV.11) (IV.12) (IV.13) (IV.15) are satisfied} \\ 0 & \text{otherwise} \end{cases}$$

3.5 Fishing strategies

We here consider different fishing strategies in order to compare them in terms of simulated trajectories over the period $t_h = 2016$ to $t_n = 2049$. Following Doyen (2018) and IPBES (2018), we distinguish between exploratory and goal-seeking strategies for fishing.

Status-quo (SQ) strategy: The first exploratory fishing strategy entitled Status-Quo (SQ) maintains fishing efforts at the current level (of year $t_h = 2016$) throughout the period of interest for every fleet f and every patch p :

$$e_{f,p}^{SQ}(t) = e_{f,p}(t_h) \quad \forall t = t_h + 1, \dots, t_n \quad \forall f, p$$

Stochastic Multi-species Maximum Sustainable Yield (SMMSY) strategy: This strategy differs from the exploratory fishing strategies (such as SQ) in the sense that it is a goal-seeking (or normative) strategy aiming at achieve specific goals in the future. The objective of this SMMSY strategy is to reach a maximum sustainable yield over all harvested species, that is to say, to maximize long-term landings induced by the different types of fishing f . To account for

⁹*Ctrs* stands for Constraints. A confidence rate β is prescribed set at 90%.

the uncertainty of shocks $\Delta r_{reef}(t)$ affecting coral reef dynamics in equation (IV.1), we consider the expected value of the sum of the landinds over time as follows:

$$\max_{e(\dots)} \mathbb{E} \left[\sum_{t=t_h+1}^{t_n} \left(\sum_i \sum_f \sum_p H_{i,f,p}(t) \right) \right] \quad (\text{IV.19})$$

In this strategy, we assume that only two fishing effort choices e_0 et e_1 are identified at the start of two periods t_h and t_1 . Moreover, to deal with feedback controls and uncertainties, the choice of these two efforts relies on a ‘non-anticipative’ control strategy (Shapiro et al., 2009; Doyen et al., 2017).

Ecological Viability (PVA) strategy:

Following the Population Viability Analysis (PVA) (Morris et al., 2002), this strategy consists in maximizing the probability of satisfying the ecological viability thresholds (eq. (IV.11) (IV.12) (IV.13)) according to uncertainties $\omega(t)$. We then attempt to find the viable fishing effort noted e^{PVA} that maximizes the metrics defined in eq. (IV.14) as follows:

$$\mathbb{P}_{PVA}(e^{PVA}) = \max_e \mathbb{P}_{PVA}(e) \quad (\text{IV.20})$$

It only relates here to the ecological dimension of the socio-ecosystem. It gives us insights regarding conservation issues and shows the need to embed the socio-economic dimension in order to fully analyse the complexity of Moorea’s reef socio-ecosystem.

Socio-economic Viability (EVA) strategy:

Following the Economic Viability Analysis (EVA) (Gourguet et al., 2013; Doyen et al., 2017), this strategy consists in maximizing the probability of satisfying the socio-economic viability thresholds (eq. (IV.15)) according to uncertainties $\omega(t)$. We then attempt to find the viable fishing effort noted e^{EVA} that maximizes the metrics depicted in eq. (IV.17) such as:

$$\mathbb{P}_{EVA}(e^{EVA}) = \max_e \mathbb{P}_{EVA}(e) \quad (\text{IV.21})$$

Eco-Viability (EcoVA) strategy:

Following Doyen et al. (2017); Cissé et al. (2015); Gourguet et al. (2013), this strategy consists in maximizing the probability of satisfying the ecological and socio-economic viability thresholds according to uncertainties $\omega(t)$. We then attempt to find the viable fishing effort noted e^{EcoVA} that maximizes the metrics depicted in eq. (IV.18) such as:

$$\mathbb{P}_{EcoVA}(e^{EcoVA}) = \max_e \mathbb{P}_{EcoVA}(e) \quad (\text{IV.22})$$

Such a formulation points to the fact that the viability approach, in a stochastic context, consists in minimizing ecological-economic risk or vulnerability.

4. Results of shock scenarios

Hereafter, we compare the ecological-economic scores of the different fishing strategies SQ, SMMSY, PVA, EVA and EcoVA under different probabilities p of natural shocks $\Delta r_{reef}(t)$. We examine adverse events on coral with four probabilities being $p = 10$, $p = 20$, $p = 30$, $p = 50$ every year.

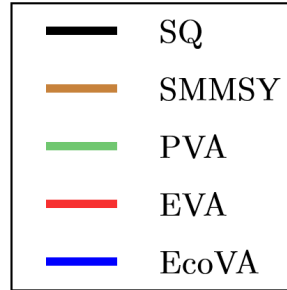


Figure IV.4: Graph Legend - Colors showed here are the reference for all the following figures.

4.1 $p = 10\%$: A viable and resilient ecosystem

Fishing Strategies	PVA (in %)	EVA (in %)	EcoVA (in %)
SQ	50	100	50
SMMSY	92	100	92
PVA	100	100	100
EVA	86	100	86
EcoVA	100	100	100

Table IV.1: Case 10% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints described by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.

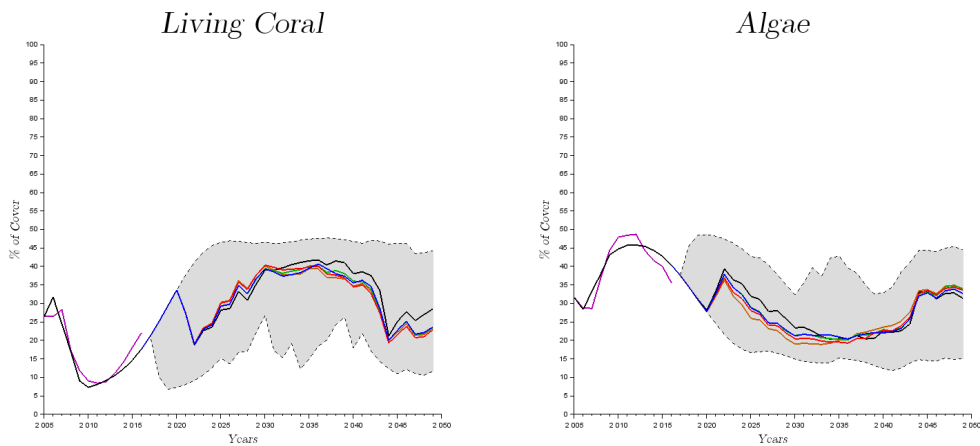


Figure IV.5: 10% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.

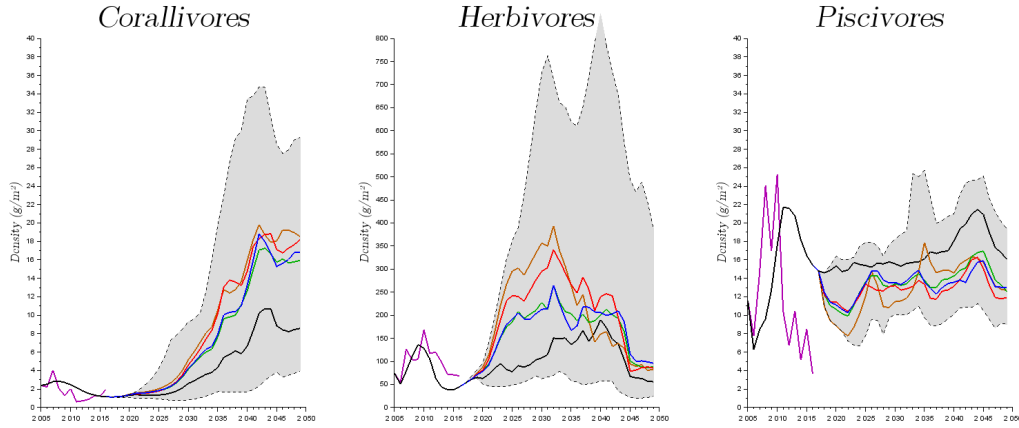


Figure IV.6: Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.

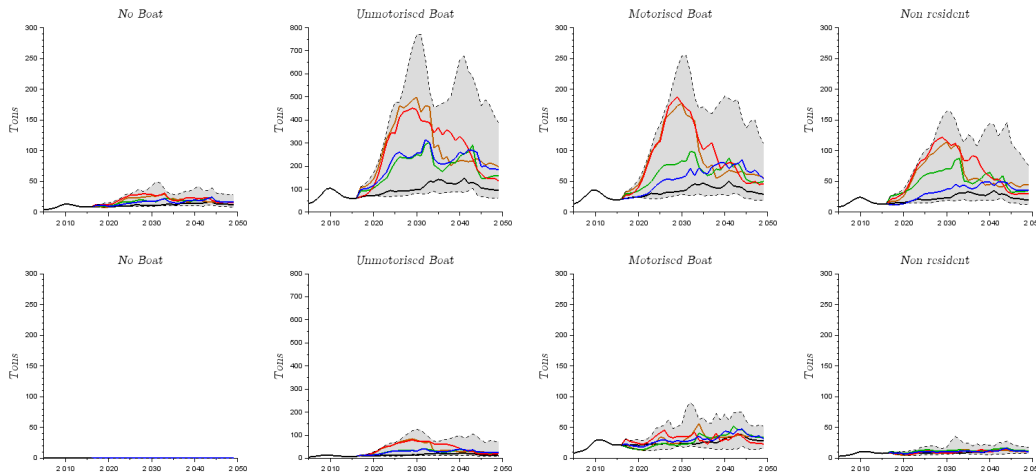


Figure IV.7: Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).

At a $p = 10\%$ shock occurrence, the system is viable for most of the simulations, regardless of the strategies. Only the SQ, SMMSY and EVA strategies which are economically viable do not satisfy all ecological constraints (Table IV.1). Indeed, regardless of the fishing strategy, the dynamics, in average, for habitats and fishes remain viable, displaying an increase in fish densities comparatively to the historical period throughout the temporal horizon (Figures IV.5 and IV.6). However, under the SQ, SMMSY and EVA strategies, some trajectories/simulations (notably for coral and herbivores) cross the ecological thresholds at the end of the temporal horizon reaching respectively 50%, 92%, 86% of PVA and EcoVA percentages for these fishing strategies.

Regarding fishing effort structure, given the SQ ecological results, even at a 10% shock occurrence, fishing effort needs to be changed in order to improve the ecologico-economic performances of the system. Indeed, Table IV.6 indicates that the too strong increase of the effort on both patches under the EVA strategy mitigates the ecological outcomes. It, of course, generates a better number of fish caught but it endangers the herbivore biomass at the end of the temporal

horizon. Conversely, the PVA and EcoVA strategies do not increase the effort as much as the EVA generating a lower level of catches.

Figure IV.7 shows the catch structure per fleets and reinforces the previous analysis.

More Globally, unless under SQ strategy, none of the groups might collapse. Therefore, Resistance is maintained, the Recovering time is really low or tend to zero, and the Robustness occur (the system remains diversified and do not a reach a critical point). In other words, according to 3Rs of the ecological resilience depicted by Grafton et al. (2019), the resilience of Moorea’s coral reef is maintained for the four goal-seeking strategies (SMMSY, PVA, EVA, EcoVA). Then, if we add the socio-economic dimension, the PVA and EcoVA strategies are the two most viable strategies with a 100% Eco-Viability probability, ensuring an ecological-economic viability and promoting resilience.

4.2 $p = 20\%$: A viable and resilient ecosystem

Fishing Strategies	PVA (in %)	EVA (in %)	EcoVA (in %)
SQ	45	100	45
SMMSY	79	100	79
PVA	100	100	100
EVA	96	100	96
EcoVA	100	100	100

Table IV.2: 20% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.

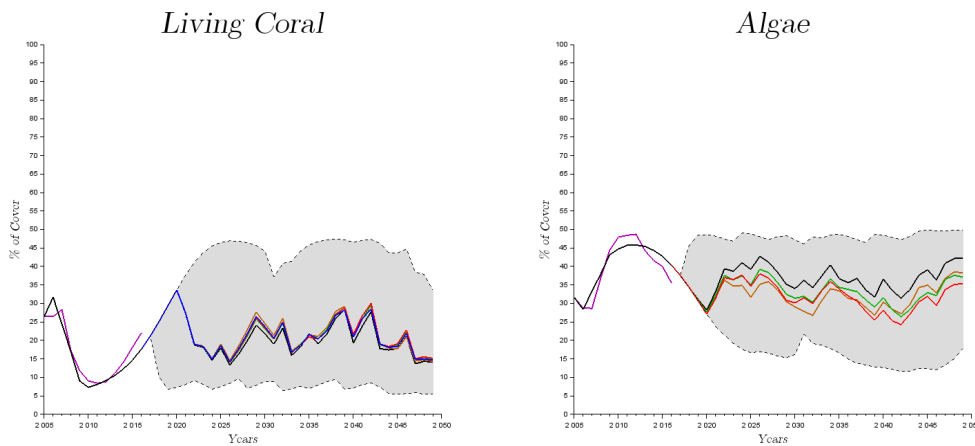


Figure IV.8: 20% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.

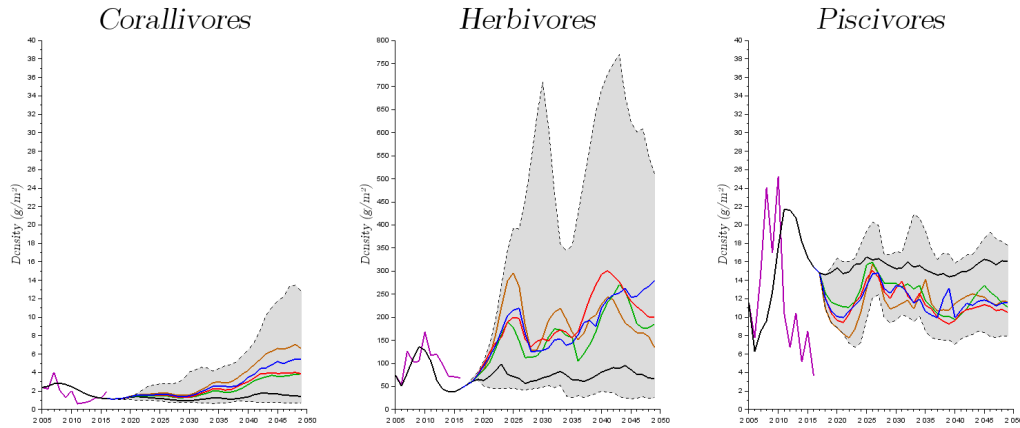


Figure IV.9: 20% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.

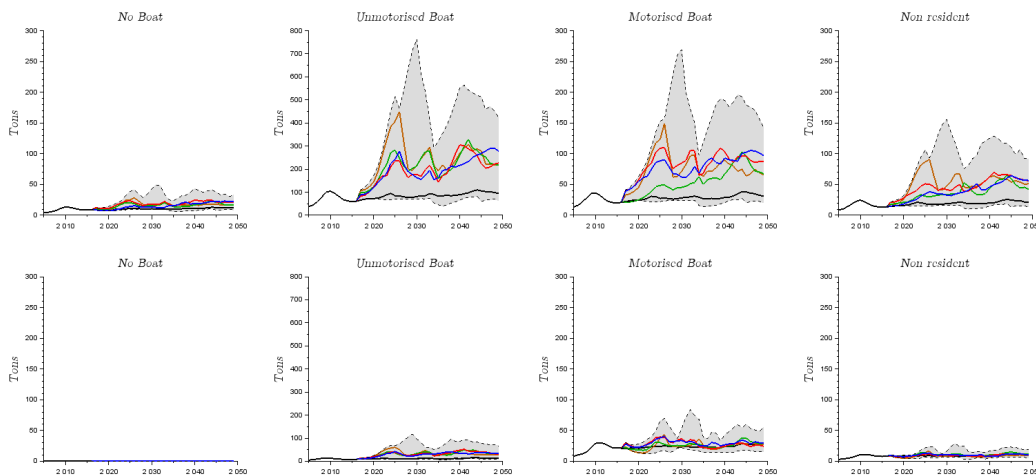


Figure IV.10: 20% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).

At a $p = 20\%$ shock occurrence, the system is viable for the PVA, EVA and EcoVA strategies. Only the SQ, SMMSY and EVA strategies which are economically viable do not satisfy all ecological constraints (Table IV.2). Indeed, regardless of the fishing strategy, the dynamics, in average, for habitats and fishes remain viable but lower than during a 10% shock occurrence (Figure IV.8). Apart from the SQ strategy, trajectories display a higher density of herbivores comparatively to the historical period throughout the temporal horizon (Figure IV.9) and appears steadier as compared to the case of a 10% shock occurrence. However, under the SQ, SMMSY and EVA strategies, some trajectories/simulations (notably for coral and herbivores) cross the ecological thresholds at the end of the temporal horizon reaching respectively 45%, 79%, 96% of PVA and EcoVA percentages for these fishing strategies.

Regarding fishing effort structure, given the SQ ecological results, even at a 20% shock occurrence, fishing effort needs to be modified in order to improve the ecologico-economic performances of the system. Indeed, Table IV.6 indicate that the too strong increase of the effort on

both patches under the EVA strategy mitigates the ecological outcomes. It, of course, generates a better number of fish caught but lower than during a 10% shock occurrence because of a mitigation of the refuge effect (Lower Coral cover). The PVA and EcoVA strategies do not increase the effort as much as the EVA, generating a lower level of catches. The EcoVA strategy increases more the effort of the fleet with the strongest capturability (motorised boats on the lagoon) which is compensated by a lower effort increase on the unmotorised boat fleet. Conversely, the PVA strategy does the exact opposite by applying the smallest increase of the most productive fleet (again motorised boats on the lagoon) and reports the effort on the unmotorised boat fleet.

Figure IV.10 shows the catch structure per fleets and reinforces the previous analysis.

More Globally, unless under SQ strategy, none of the groups might collapse as during a 10% shock occurrence. Therefore, Resistance is maintained, the Recovering time is really low or tend to zero, and the Robustness occurs (the system remains diversified and does not reach a critical point). In other words, according to 3Rs of the ecological resilience depicted by Grafton et al. (2019), the resilience of Moorea’s coral reef is maintained for the four goal-seeking strategies (SMMSY, PVA, EVA, EcoVA). Then, with a 100% Eco-Viability probability, the PVA and EcoVA strategies are still the two most viable strategies, ensuring an ecological-economic viability and promoting resilience.

4.3 $p = 30\%$: A viable and less resilient ecosystem

Fishing Strategies	PVA (in %)	EVA (in %)	EcoVA (in %)
SQ	22	100	22
SMMSY	90	100	90
PVA	100	100	100
EVA	92	100	92
EcoVA	100	100	100

Table IV.3: 30% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.

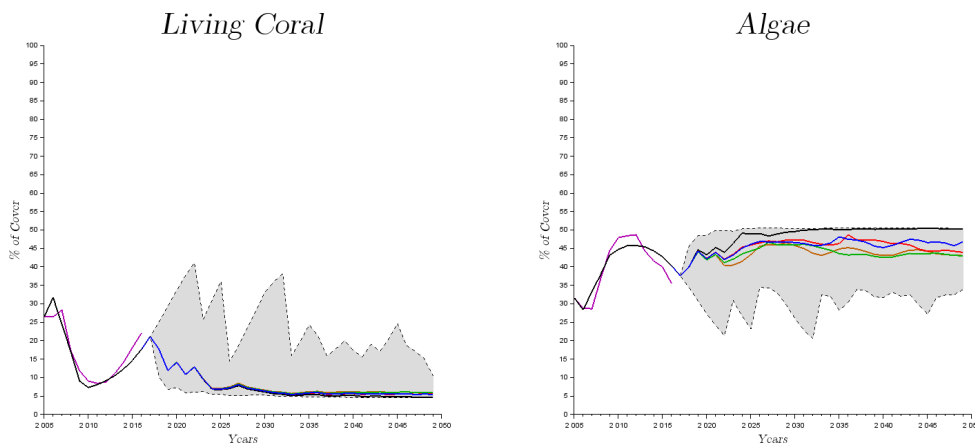


Figure IV.11: 30% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.

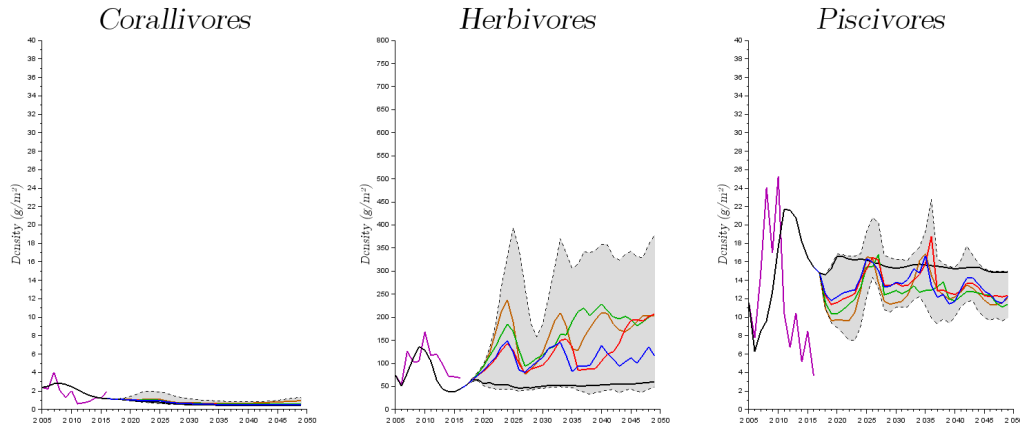


Figure IV.12: 30% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.

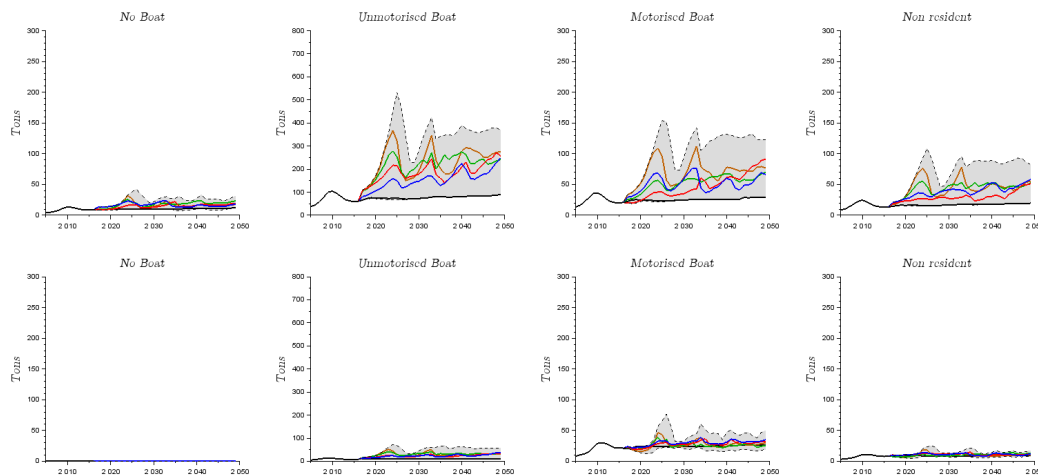


Figure IV.13: 30% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).

At a $p = 30$ shock occurrence, the current management system (i.e SQ) becomes strongly unviable while the four other fishing strategies appear viable for more than 90% of the simulations. Table IV.3 shows that the two most viable strategies are the PVA and the EcoVA. Ecologically, most of the trajectories for habitats remain above the the viability thresholds however there is now a recuperation time which seems infinite. Indeed, with a 30% shock occurrence, the coral recuperation does not occur. The mean dynamics stabilize just above 5% of cover (Figure IV.11). Regarding fish densities, with a smaller density for herbivores, and un narrower uncertainty envelope, at a 30% shock occurrence, the system is much more impacted by creating a lack of refuge (i.e coral) for preys. This is profitable for the top predators (i.e piscivores) which display higher average trajectories. (Figure IV.12). Under the SQ, SMMSY and EVA strategies, some trajectories/simulations (notably for coral and herbivores) cross the ecological thresholds at the end of the temporal horizon reaching respectively 22 %, 90%, 92% of PVA and EcoVA percentages for these fishing strategies.

Regarding fishing effort structure, given the SQ ecological results, with a 30% shock occurrence, fishing effort needs to be modified in order to support the resilience of Moorea’s reef and improve the ecologico-economic performances of the system. Indeed, Table IV.6 highlights a choice for the EVA and EcoVA strategies towards a lower increase of the effort homogeneously allocated among fleets. The PVA strategy which surprisingly generates the largest number of catches, maintain the same structure than under a 20% shock occurrence but at a tiny higher level. It appears to be the most viable ecologico-economic strategy.

Figure IV.13 shows the catch structure per fleets and displays a narrower uncertainty field attesting that an increase in shock occurrence mitigates the ecological outcomes as explained but as well as the socio-economic performances of the fishery. This is evidenced by the weaker number of catches generated by the fishing strategies (Table IV.6).

More Globally, unless under SQ strategy, none of the groups might collapse but we observe a likely regime shift, starting from a dominated coral area to a dominated algae area. Nevertheless, the coral does not recover to the previous level. Therefore, Resistance is maintained (the 5% cover of coral needed for its future and likely recuperation is still observed), the Robustness occurs (the system remains diversified notably for fishes) however the Recovering time for the coral group tends towards infinity. With a future decrease in shock occurrence, the system may recover but there is no evidence beyond the historical data. In other words, according to 3Rs of the ecological resilience depicted by Grafton et al. (2019), the resilience of Moorea’s coral reef is not maintained for all strategies. Nevertheless, the PVA and EcoVA still remain the two most viable strategies with a 100% Eco-Viability probability, ensuring an ecological-economic viability but with a weaker resilience capacity on the recuperation time.

4.4 $p = 50\%$: A viable and non-resilient ecosystem

Fishing Strategies	PVA (in %)	EVA (in %)	EcoVA (in %)
SQ	0	100	0
SMMSY	100	100	100
PVA	100	100	100
EVA	75	100	75
EcoVA	100	100	100

Table IV.4: 50% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.

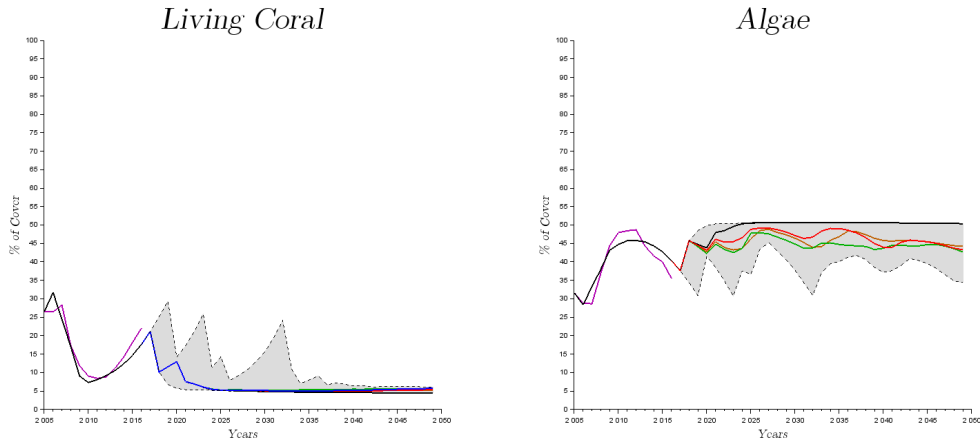


Figure IV.14: 50% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.

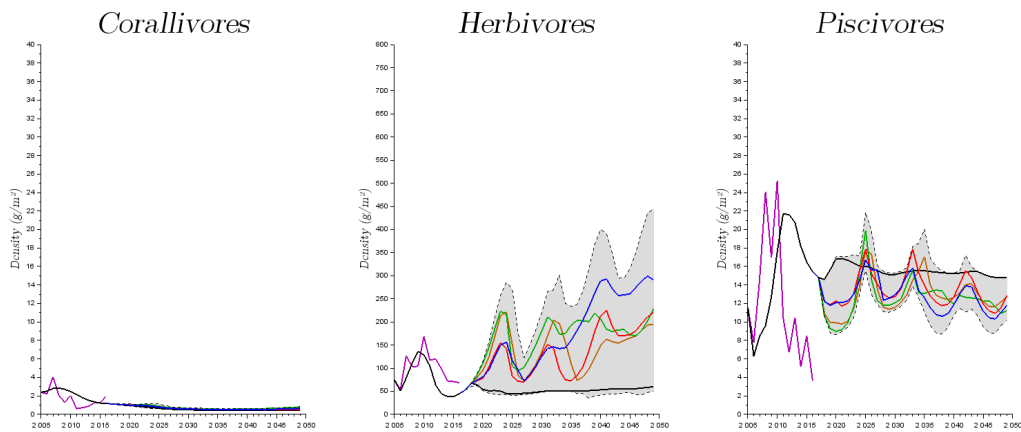


Figure IV.15: 50% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.

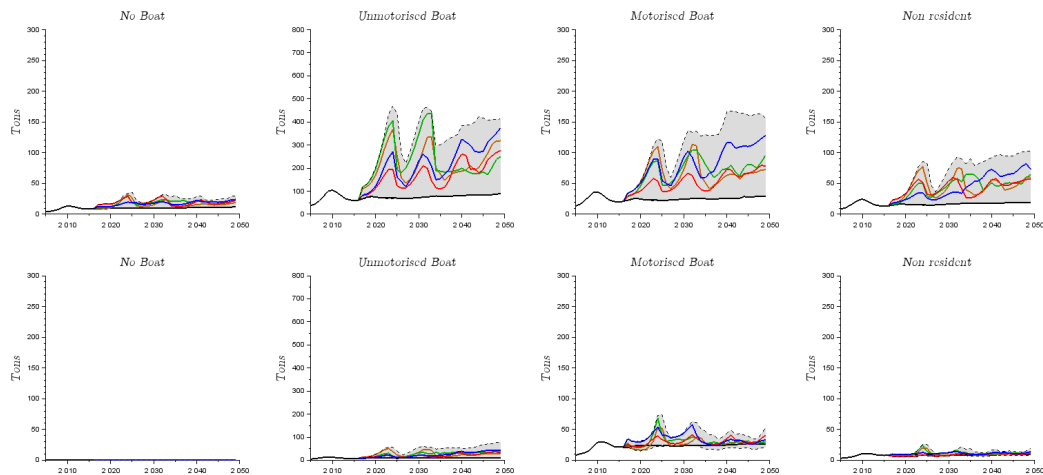


Figure IV.16: 50% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).

At a 50% shock occurrence, the current management system (i.e SQ) becomes fully unviable ecologically while the four other fishing strategies appear viable for more than 75% of the simulations. Table IV.4 shows that the three most viable strategies are the SMMSY, the PVA and the EcoVA. Regarding these three strategies, ecologically, all of the trajectories for habitats remain above the the viability thresholds however there is now a recuperation time which seems infinite as under a 30% shock occurrence. Indeed, with a 50% shock occurrence, the coral recovering does not occur. The mean dynamics stabilize just above 5% of cover (Figure IV.14). Regarding fish densities, we observe a higher number of herbivores than during a 30% shock occurrence. The lack of refuge (coral cover) is compensated by an increase in fishing effort on top-predators. Consequently, this group (piscivores) is a bit less numerous than during a 30% shock occurrence (Figure IV.15). Under the SQ, EVA strategies, some trajectories/simulations (notably for coral and herbivores) cross the ecological thresholds at the end of the temporal horizon reaching respectively 0 %, 75% of PVA and EcoVA percentages for these fishing strategies.

Regarding fishing effort structure, given the SQ ecological results, with a 50% shock occurrence, fishing effort needs to be modified in order to support the resilience of Moorea’s reef and improve the ecologico-economic performances of the system. Indeed, as said in the previous paragraph, Table IV.6 shows a stronger increase in fishing effort for the PVA and EcoVA strategies notably on the most efficient whose targets are mostly piscivores especially on the fore reef (motorised boats - Table IV.9). By increasing the effort on this particular fleet, this strategy accounts for the role of the coral (refuge) by reducing predation intensity and thus promotes herbivores density and the essential grazing function.

Figure IV.16 shows the catch structure per fleets and displays a better number of fish caught than with a 30% shock occurrence as evidenced in Table IV.6.

More globally, unless under the SQ strategy, none of the groups might collapse but we observe a full and certain regime shift, starting from a dominated coral area to a dominated algae area. Indeed, the coral does not recover to the historical level. Therefore, Resistance is maintained (the 5% cover of coral needed for its future and likely recuperation is still observed), the Robustness occurs (the system remains diversified notably for fishes) however the Recovering time for the coral group tends to be infinite. With a future decrease in shock occurrence, the system may recover but there is no evidence beyond the historical data. In other words, according to 3Rs of the ecological resilience depicted by Grafton et al. (2019), the resilience of Moorea’s coral reef is not maintained for all strategies. Nevertheless, the SMMSY, PVA and EcoVA are the three

most viable strategies with a 100% Eco-Viability probability, ensuring an ecological-economic viability but with a weaker resilience notably on the recuperation time.

5. Discussion

5.1 PVA and EcoVA: the most viable ecological-economic strategies

Shock Occurrence	Strategies	Specialization	Diversification	Resilience
10%	PVA	\	Spatial, Metier, Target	Maintained
	EcoVA	\	Spatial, Metier, Target	Maintained
20%	PVA	\	Spatial, Metier, Target	Maintained
	EcoVA	Metier (Motorised Boats) Target (Piscivores)	Spatial	Maintained
30%	PVA	Metier (Unmotorised Boats)	Spatial, Target	Not Maintained (High Recovery Time)
	EcoVA	\	Spatial, Metier, Target	Not Maintained (High Recovery Time)
50%	PVA	Target (Piscivores)	Spatial, Metier	Not Maintained (Infinite Recovery Time)
	EcoVA	Metier (Motorised Boats) Target (Piscivores)	Spatial	Not Maintained (Infinite Recovery Time)

Table IV.5: Type of strategies used by the PVA and EcoVA fishing scenarios and the consequences on Resilience according to shock occurrence.

In Table IV.5, we show the different effort strategies used by the two most optimal strategies (PVA and EcoVA) which are, furthermore, both viable regardless of shock scenarios.

We can observe that under a 10% shock occurrence, a diversification strategy remains the most viable, and promotes resilience.

With a 20% shock occurrence, the EcoVA strategy tends to be more specialized notably by concentrating the effort on the motorised boats (in the Lagoon mostly) which entails a specialization of the strategy towards the piscivores. This is explained by the higher capturability (Table IV.9) of the motorised boats on the piscivores. That choice generates a better abundance of herbivores.

Under a 30% shock occurrence, the resilience (through coral recovering) is damaged. To deal with that consequence, the PVA strategy consists in a specialization in the unmotorised boat fleet. If we look at the capturability of this fleet (Table IV.9), we observe that the PVA strategy allocates the fishing impact across both commercial fish group (herbivores, piscivores) and both patches. Thus, although this effort strategy appears as a strategy of metier specialization, it is in fact a strategy of fish target diversification.

Finally, when the shock occurrence reaches 50% (one shock every two years, in average), the resilience (for habitats notably) is highly affected. The coral cover will not recover its historical level and with a small (or nul) uncertainty envelope, the recovery time becomes infinite under this environmental context.

Firstly, in order to enhance the grazing function (which indirectly support the coral cover recovering), the PVA strategy relies on an overall increase in effort¹ similarly to the EcoVA strategy. Indeed, while the EcoVA allocates the effort towards the top predators i.e piscivores on both patches, the PVA strategy focuses its predator's fishing on the fore reef. By decreasing

¹By having a stronger increase of the effort (for almost all fleet), the PVA strategy attempts to fish through all exploited species. On the one hand, it generates a better number of catches by fishing more herbivores than the other strategies and on the other hand, it compensates that exploitation by decreasing the piscivore density. In other words, it partly takes over the rôle of predation mitigation generated by the coral.

the predators (as said, on the Fore reef for the PVA and on both patches for the EcoVA), both strategies compensate the loss of the refuge effect (generated by the living coral) and promotes herbivore biomass.

Secondly, the EcoVA strategy specializes in motorised boats on both patches to increase herbivores biomass mainly to generate more catches as a whole. Nevertheless, the effort of the EcoVA strategy is not high enough to achieve more catches than the PVA strategy because it mainly focuses on piscivores. Thus, this strategy, whose criterion is only ecological, surprisingly produces better socio-economic outcomes under a catastrophic environmental context.

The first intuition emerging from that analysis, is that: under a risky, uncertain and critical environmental context, focusing only on the ecological dimension may produce better ecological-economic results. Indeed, as the implied ecological consideration needed for the MMSY², the PVA strategy has an underlying socio-economic criterion (maintaining a high biomass produces better socio-economic outcomes i.e catches) which is a consequence of the biological viability purpose.

5.2 Viability \neq Resilience

As we showed in the previous section, the 100% probability of eco-viability that still occurs for the PVA et EcoVA strategies, during a 50% shock occurrence, does not necessarily generate resilience. In particular, this analysis depends on the ecological-economic thresholds we implement. We acknowledge that above a certain level, socio-ecosystem can recover, resist and stay robust (i.e be resilient) because historical data underpin this idea, such as Moorea's reef for 30 years and moreover recently after the COTS perturbation in 2006-2009 (Lamy et al., 2016). However, does this level or floor still hold when shocks increase in occurrence? What if the level of coral cover needed for its recovering changes over time? Or more globally, what is a true viable coral reef?

5.3 Is a viable coral reef, still resilient under a gradual increase of shocks ?

To answer the title of this sub-section, we need to look at the main key process which promotes resilience within a coral reef: grazers (small herbivores - Mumby et al., 2007; Adam et al., 2011; Holbrook et al., 2016; Martin et al., 2017). If we focus on this functional group, we can examine that even at higher level than the historical observations, a too strong frequency of natural shocks (starting from 30%) does not allow the ecosystem to be resilient anymore. However, if the shock occurrence decreases in the future, all things being equal, the coral is likely to recover and then to be resilient again given the current level of grazers. This is evidenced by the 10% and 20% (Figures IV.5, IV.8) shock scenarios where the coral goes back to its pre-shock level of cover.

Conversely, if the shock occurrence stabilizes between 30% and 50% after 2050 (or even worse, if the intensity rises), given the increase of human population (as well as fishing intensity), the coral reef may lose its resilience definitively as evidenced by the lack of uncertainty envelop in Figure IV.14. Moreover, with a lack of resilience capacity (through a increase in effort on herbivores and a lack of refuge effect without living coral), the viability of the coral reef, may turn out to be endangered or lost. This has already happened in the Caribbean (Mumby, 2009).

²The MMSY, even with a unique economic objective, needs a sustainable level of biomass to maximize catches (Lagarde et al., 2018).

5.4 Eco-viability: a toolbox for resilience assessment and management

This study highlights that socio-ecologico-economic viability thresholds need to be dynamic and function of the environmental context in order to be relevant in viability analysis. Furthermore, the conclusions that arised from these assumptions, could embedd resilience theory within an viability framework. Béné and Doyen (2018) explain why the viability approach is relevant in the case of resilience assessment.

First, resilience always concerns dynamic systems (whose state evolves throughout time) and thresholds, which are two of the main components of the viability approach.

Second, resilience assessments require the analysis of dynamic control and management action. The viability framework is in line with that in the sense that the approach is built around decision and control variables.

Finally, the most comun definition of resilience (regardless of the field of study) is about “system’s recovery” or the time taken by the system to fully recover or to maintain its identity and fonctionnalités. Here again, the viability approach is relevant because it can examine the “minimal time of crisis” i.e the time where the system is out the viability space. This temporal range would correspond to the recovery capacity so a proxy of resilience. The longer the time of crisis is, the less resilient the system is.

Therefore, by defining these dynamic levels of viability, resilience may be not only a theoretical concept whose implementation remains complex but as the eco-viability, may become a mathematical and quantitative framework where each dimension of the resilience (e.g the 6 points³ of Martin, 2004 , the 3Rs of Grafton et al., 2019, or the 5 factors⁴ of Béné and Doyen (2018)) is taken into account.

³The state of the system, the objectives (the property to be maintained), the anticipated types of disturbances, the cost associated with the effort necessary for restoring this property, the control measures that are available, and the time scale of interest (Martin, 2004).

⁴Resistance, Absorptive resilience (copying strategy), Adaptive resilience, Adaptive preference and Transformation. These 5 points are described in the Table 1 of Béné and Doyen (2018).

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Appendix

	Groups	Index (i)	Number of Species
Habitats	Coral (C)	1	24
	Algae (A)	2	10
Fish	<u>Herbivores (H)</u>	3	64
	<u>Piscivores (P)</u>	4	48
	Corallivores (Co)	5	7

Table IV.7: The five functional groups. Underlined groups stand for commercial and mobile species.

Groups Interaction Matrix (s_{ij})	Coral	Algae	Herbivores	Piscivores	Corallivores
Coral	0.009	0.006	0	0	0
Algae	0	0	0.0005	0	0
Herbivores	0	0.022	0	0.001	0
Piscivores	0	0	6.10^{-5}	0.019	0.008
Corallivores	0.0045	0	0	0.0055	0

Species	Intrinsic Growth Rate (r_i)	Diffusion Rate ($d_{ip'p}$)	
Coral	0.25/0.92	Lagoon to Reef	Reef to Lagoon
Algae	-	-	-
Herbivores	0.01	0.25	0.5
Piscivores	0.40	0.45	0.75
Corallivores	0.01	-	-

	Carrying Capacity (K_p)	
	Lagoon	Fore Reef
Coral/Algae	0.38	0.98

Table IV.8: Calibrated parameters of the model, respectively (top) interactions matrix s_{ij} , (middle) species intrinsic growth rates r_i and the diffusion rates $d_{ip'p}$ of species i between patch p and p'. We also display (bottom) the carrying capacity K_p of the habitat i.e the total area where algae and coral can evolve.

		Shock Occurrence : 10%				
		Strategies	SMMSY	PVA	EVA	EcoVA
Lagoon	No Boat		42%	51%	67%	37%
	Unmotorised Boat		73%	45%	60%	68%
	Motorised Boat		56%	46%	48%	24%
	Non Resident		45%	77%	65%	23%
Reef	Unmotorised Boat		45%	46%	67%	50%
	Motorised Boat		51%	28%	68%	44%
	Non Resident		53%	63%	58%	44%
Catches (in tons)			18 228	14 933	18 765	14 955

		Shock Occurrence : 20%				
		Strategies	SMMSY	PVA	EVA	EcoVA
Lagoon	No Boat		42%	51%	67%	32%
	Unmotorised Boat		73%	63%	42%	44%
	Motorised Boat		56%	24%	75%	85%
	Non Resident		45%	40%	51%	28%
Reef	Unmotorised Boat		45%	63%	55%	48%
	Motorised Boat		51%	55%	60%	58%
	Non Resident		53%	48%	45%	42%
Catches (in tons)			16 857	14 874	15 613	14 773

		Shock Occurrence : 30%				
		Strategies	SMMSY	PVA	EVA	EcoVA
Lagoon	No Boat		42%	57%	37%	50%
	Unmotorised Boat		73%	68%	65%	39%
	Motorised Boat		56%	33%	34%	48%
	Non Resident		45%	48%	27%	56%
Reef	Unmotorised Boat		45%	74%	57%	50%
	Motorised Boat		51%	56%	35%	43%
	Non Resident		53%	43%	53%	54%
Catches (in tons)			15 137	14 237	12 010	11 473

		Shock Occurrence : 50%				
		Strategies	SMMSY	PVA	EVA	EcoVA
Lagoon	No Boat		42%	59%	74%	49%
	Unmotorised Boat		73%	80%	50%	55%
	Motorised Boat		56%	53%	40%	71%
	Non Resident		45%	43%	64%	49%
Reef	Unmotorised Boat		45%	32%	47%	21%
	Motorised Boat		51%	71%	48%	63%
	Non Resident		53%	69%	7%	42%
Catches (in tons)			14 539	15 314	11 803	14 755

Table IV.6: Average Fishing Effort Evolution Structure of the four goal-seeking strategies under the four shock scenarios (10% (a), 20% (b), 30% (c), 50% (d)). Below the efforts are displayed the total catches produced by each fishing strategies.

Fisherman per Km^2 $L_f(t)$ in Maatea	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
No Boat	9	9	9	10	10	10	10	10	11	11	11	11
Unmotorised Boat	16	17	17	18	18	18	19	19	20	20	20	20
Motorised Boat	4	4	4	4	4	4	4	4	5	5	5	5
Non Resident	10	10	10	11	11	11	11	11	12	12	12	12
No Boat	0	0	0	0	0	0	0	0	0	0	0	0
Unmotorised Boat	9	9	9	10	10	10	10	10	11	11	11	11
Motorised Boat	9	9	9	10	10	10	10	10	11	11	11	11
Non Resident	6	6	6	7	7	7	7	7	7	7	7	7

Catchabilities q_{ifp}	Lagoon		Fore Reef	
Fleets	Herbivores	Piscivores	Herbivores	Piscivores
No Boat	7.10^{-7}	3.10^{-5}	-	-
Unmotorised Boat	4.10^{-6}	4.10^{-5}	4.10^{-6}	2.10^{-5}
Motorised Boat	5.10^{-6}	4.10^{-5}	-	4.10^{-5}
Non Resident	5.10^{-6}	4.10^{-5}	-	4.10^{-5}

Table IV.9: Fishing Parameters: (bottom) Catchabilities (top) q_{ifp} of fleet f on species i in patch p; (top) Fisherman per Km^2 in Maatea $L_f(t)$. The share of labor involved in each fleet as well as the global density of fishermen in all areas are extrapolated from this distribution and also regarding the surface of the corresponding zone.

Liste des Figures

I.1	Suivi des pêcheries mondiales: (a) Quantités capturées par les pêcheries marines (en noir) et produites par l'aquaculture (en gris) ; (b) Pourcentage des stocks selon leur statut d'exploitation. Sources: FAO (a) et Sea Around Us Project (b).	18
I.2	Représentation schématique du "Fishing Down Effect" décrit par Pauly et al. (1998).	20
I.3	Tendances de l'indice trophique marin (Marine Trophic Index - Gauche) et de l'indice de soutenabilité des pêcheries (Fisheries Sustainability Index - Droite) (Kleisner and Pauly, 2011).	20
I.4	Modèle de Gordon-Schaefer où le revenu et les coûts sont fonction de l'effort de pêche. Le revenu égalise les coûts à l'équilibre bioéconomique (BE). (Christensen, 2010)	25
I.5	Illustration schématique d'un réseau trophique.	27
I.6	Les "17 objectifs de développement durable pour transformer le monde à l'horizon 2030" (UN, 2015).	30
I.7	Représentation mathématique du concept de viabilité où 2 états initiaux du système x_0 et x_1 sont présentés. Les évolutions temporelles du premier état x_0 violent toutes les contraintes dans un temps fini, tandis que l'autre x_1 affiche une évolution viable (qui demeure dans K) et une autre non viable qui n'appartient plus à K (Aubin et al., 2011).	31
I.8	Exemple conceptuel d'effets possibles d'un événement indésirable sur la Résistance, la Récupération et la Robustesse (Grafton et al., 2019). Les courbes colorées correspondent à trois scénarios différents ou comment un écosystème peut réagir face à un évènement indésirable.	33
I.9	Nouvelle représentation schématique du concept de viabilité écologico-économique où le viabilité de Kernel (<i>Viability kernel</i>) est l'objectif illustré pour le cas de seulement 2 contraintes faisant référence à la conservation écologique versus la sécurité alimentaire. La partie en vert/bleu représente les états viables du système dynamique selon le pourcentage de confiance β introduit dans l'éq. (I.25) tandis que les parties en blanc correspondent aux états non viables (Doyen et al., 2019).	37
I.10	Carte du Golfe de Gascogne et des divisions réalisées par le CIEM. La zone d'étude est en bleue.	42
I.11	Stratégie MMEY - Les trajectoires de SSB de la sole (en haut à gauche) et du merlu (en haut à droite) sont représentées pour les deux scénarios climatiques en milliers de tonnes. La ligne noire continue représente le seuil de précaution (du CIEM - Bpa) du stock de l'espèce. La troisième figure (en bas) représente les profits totaux de l'ensemble des treize sous-flottes. Les données historiques sont affichés en bleu. Les 500 trajectoires simulées sont représentées par les zones colorées en fonction du scénario climatique (vert: meilleur / RCP 2.6 et rouge: pire / RCP 8.5). Les courbes en gras dans les zones vertes et rouges affichent la moyenne de ces 500 trajectoires.	43
I.12	L'Index de Diversité de Simpson face à la Valeur Actuelle Nette (Pour les deux espèces, sole et merlu).	44

I.13	Carte de Moorea divisée en comités de gestion de pêche selon le nouveau Plan de Gestion de l'Espace Maritime (PGEM) débutant en 2019-2020.	45
I.14	Conceptual Model	45
I.15	Stratégie SMMSY - Trajectoires écologiques - Les cinq figures représentent les pourcentages de couverture des deux habitats et les densités agrégées (Lagoon - Fore Reef) des trois groupes fonctionnels de poissons. Les 100 trajectoires simulées sont représentées par la zone orangée. Les courbes au sein des champs colorés indiquent la moyenne de ces 100 trajectoires tandis que les courbes violettes attestent des trajectoires historiques et les courbes noires de la calibration.	47
I.16	Diagramme en radar - Les quatre radars sont colorés selon les stratégies de gestion et les cinq échelles des composants (SHB: 0-100%, Fishing Effort: 0-2, Catches: 0-20, Marine Trophic Index: 0-4, Fish Biomass Increase: 0-200%). Les nombres dans les radars représentent la moyenne des 100 simulations liées à chaque composant correspondant. Les 5 composants sont décrits en appendices au Chap. III.	48
II.1	Map of the Bay of Biscay and ICES divisions. The studying area is in blue.	78
II.2	Historical evolution of the spawning biomass for the common sole. The dashed line refers to the precautionary threshold (Bpa. (ICES, 2017)	78
II.3	Historical evolution of the spawning biomass for the European hake. The dashed line refers to the precautionary threshold (Bpa) estimated at 46200 tons (ICES, 2016)	78
II.4	Relations existing between environmental, biological and economic factors within the bio-economic model. Arrows stand for the interactions between variables while figures between brackets refer to the equations/models that link the various factors within the bio-economic model.	81
II.5	Historical trajectories of SST in the Bay of Biscay from 1991 to 2013 and projections of SST trajectories according to the two climate scenarios (RCP 2.6 and 8.5) from 2014 to 2100	84
II.6	Strategy Status-Quo - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.	86
II.7	Strategy MMSY - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.	88

II.8	Strategy MMEY - Sole (top left) and Hake (top right) SSB trajectories under the two climate scenarios in thousands of tons. The black solid line represents ICES precautionary threshold (B_{pa}) of the species' stock. The third Figure (bottom) represents the total profits over all thirteen sub-fleets. Historical paths are displayed in blue. The 500 simulated trajectories are represented by the colored areas depending on the climate scenario (green : best/RCP 2.6 and red : worst/RCP 8.5). The solid lines within green and red areas display the average of these 500 trajectories.	89
II.9	Simpson's Index of Diversity versus Average Net Present Value.	91
II.10	Fishing effort multipliers (left axis) and contribution to the overall fishing mortality of Sole in percentage (right axis). Sub-fleets (X-axis) are ranked by contribution to sole mortality with SGN=Sole gill-netters, MGN=Mixed gill-netters and MT=Mixed Trawlers. The first Figure accounts for the best climate scenario (RCP 2.6) and the second (bottom) for the worst climate scenario.	110
II.11	Numbers of sailors in the Atlantique facade (Aglia, 2014)	111
II.12	Contribution to the fishing mortality and dependence on sole of fleets of Bay of Biscay fishery in 2010 (Aglia, 2014)	111
II.13	Sales in volume and in value of the sole in all auction centres (histogram, left vertical axis) and price curve (right vertical axis). The percentages represent the part of sales in volume compared to sales in value. (data source : visionet.franceagrimer.fr)	112
II.14	Comparative evolution of the TAC and landings of the sole in the Bay of Biscay since 1984 (Lagiere, 2012)	112
III.1	Moorea Island and its new PGEM decomposed in five committees	121
III.2	Conceptual Model	122
III.3	Calibration - The five first figures (top) account for the lagoon and the five following figures (bottom) stand for the fore reef.	125
III.4	Closure strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon - Fore Reef) fish densities. The 100 simulated trajectories are represented by the green area. The solid lines display the average of these 100 trajectories.	127
III.5	2R strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the blue area. The solid lines display the average of these 100 trajectories.	128
III.6	2La strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the red area. The solid lines display the average of these 100 trajectories.	128
III.7	Status-Quo strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.	129
III.8	SMMSY strategy- Ecological Trajectories - The three figures account for the aggregated (Lagoon - Fore Reef) fish densities. The 100 simulated trajectories are represented by the brown area. The solid lines shows the average of these 100 trajectories.	129

III.9 (a) Catches per fleets Trajectories: The 100 simulated trajectories are represented by the coloured areas depending on the management strategy. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef). (b) Aggregated Catches Trajectories: The 100 simulated trajectories are represented by the coloured areas depending on the management strategy. The solid lines display the average of these 100 trajectories.	131
III.10 Radar plot - The four radar plots are coloured according to the management strategies and the five components scales (SHB: 0-100%, Fishing Effort: 0-2, Catches: 0-20, Marine Trophic Index: 0-4, Fish Biomass Increase: 0-200%). Numbers within the radar plot represent the mean of the 100 simulations relating to its component. The five components are further described in Table III.5.	133
III.11 Habitats Trajectories - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the colored area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.	154
IV.1 Moorea Island and its new former PGEM decomposed in five committees	160
IV.2 Conceptual Model	161
IV.3 Calibration - The five first figures (top) account for the lagoon and the five following figures (bottom) stand for the fore reef.	164
IV.4 Graph Legend - Colors showed here are the reference for all the following figures.	169
IV.5 10% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.	169
IV.6 Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.	170
IV.7 Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).	170
IV.8 20% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.	171
IV.9 20% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.	172
IV.10 20% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).	172
IV.11 30% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.	173

IV.12	30% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.	174
IV.13	30% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).	174
IV.14	50% - Habitats Trajectories: Coral vs Algae - Both figures account for the aggregated (Lagoon - Fore Reef) cover of Coral and Algae. The 100 simulated trajectories are represented by the grey area. The solid lines shows the average of these 100 trajectories depending on the fishing strategy.	176
IV.15	50% - Fishes Trajectories - The three figures account for the aggregated (Lagoon/Fore Reef) fish densities. The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories.	176
IV.16	50% - Catches per fleets Trajectories: The 100 simulated trajectories are represented by the grey area. The solid lines display the average of these 100 trajectories. The four first fleets (up) operate on the lagoon and the three other ones (down) operate on the fore reef (with “no boat” it is impossible to fish on the fore reef).	177

Liste des Tables

I.1	Definitions des 3 R d'une résilience socio-écologique selon Grafton et al. (2019)	32
I.2	Tableau d'indicateurs de biodiversité.	34
I.3	Probabilités d'éco-viabilité selon la stratégie de pêche et pour tous les scénarios de choc: p=10%, p=20%, p=30%, p=50%.	50
I.4	Type de strategies utilisées par les scénarios de pêche PVA et EcoVA et les conséquences sur la résilience selon l'occurrence des chocs.	50
II.1	Parameters and standard errors of the estimated Ricker model (equation II.13) accounting for temperature for sole. $t(17)$ stands for the Student test with 17 observations.	85
II.2	Parameters and standard errors of the estimated Ricker model (equation II.14) accounting for SST for hake. $t(62)$ stands for the Student test with 62 observations.	85
II.3	Fishing effort multipliers for MMEY and MMSY strategies with respect to the two climate scenarios. Numbers between brackets refer to the number of vessels in 2008 (Gourguet et al., 2013).	90
II.4	Type of Stock-Recruitment models with and without environmental factor (θ) affecting recruitment according to the specie ($i = 1, 2$ respectively Sole, Hake). Numbers between brackets correspond respectively to the AIC criterion and R^2 associated to the model. The underlined models do not satisfy one or several associated statistical tests (test de White, Chow, Jarque and Bera and Durbin and Watson).	113
II.5	Sole parameters, ($i = 1$), $t_h = 2008$. Source: ICES; Ifremer, SIH, DPMA.	113
II.6	Hake parameters, ($i = 2$), $t_h = 2008$. Source: ICES; Ifremer, SIH, DPMA.	113
II.7	Values of fishing mortality on Sole ($i = 1$): $F_{1,a,f}(t_0)$. Source: ICES; Ifremer, SIH, 2008.	114
II.8	Values of fishing mortality on Hake ($i = 2$): $F_{2,a,f}(t_0)$. Source: ICES; Ifremer, SIH, 2008.	114
II.9	Initial number of vessels $K_f(t_0)$, effort by vessel $e_f(t_0)$, rate of extra fishing income α_f and rate of extra fishing catches β_f of the thirteen sub-fleets. Source: Ifremer, SIH, DPMA, 2008	115
II.10	Mean reference costs of the thirteen sub-fleets. Source: Ifremer, SIH, DPMA, 2008	115
III.1	SMMSY Fishing Effort Evolution per comity Δe_{fc}	130
III.2	The five functional groups. Underlined groups stand for commercial and mobile species.	152
III.3	Calibrated parameters of the model, respectively (top) interactions matrix s_{ij} , (middle) species intrinsic growth rates r_i and the diffusion rates $d_{ip'p}$ of species i between patch p and p' . We also display (bottom) the carrying capacity K_p of the habitat i.e the total area where algae and coral can evolve.	152

III.4 Fishing Parameters: (bottom) Catchabilities (top) q_{ifp} of fleet f on species i in patch p; (top) Fisherman per Km^2 in Maatea $L_f(t)$. The share of labor involved in each fleet as well as the global density of fishermen in all areas are extrapolated from this distribution and also regarding the surface of the corresponding zone. .	153
III.5 Radar Plot Components' equations and remarks.	153
IV.1 Case 10% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints described by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.	169
IV.2 20% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.	171
IV.3 30% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.	173
IV.4 50% - Probabilities associated to combinations of effort obtained for each five fishing strategy to satisfy the constraints depicted by eq. (IV.14) for PVA, eq. (IV.17) for EVA and eq. (IV.18) for EcoVA.	175
IV.5 Type of strategies used by the PVA and EcoVA fishing scenarios and the consequences on Resilience according to shock occurrence.	179
IV.7 The five functional groups. Underlined groups stand for commercial and mobile species.	198
IV.8 Calibrated parameters of the model, respectively (top) interactions matrix s_{ij} , (middle) species intrinsic growth rates r_i and the diffusion rates $d_{ip'p}$ of species i between patch p and p'. We also display (bottom) the carrying capacity K_p of the habitat i.e the total area where algae and coral can evolve.	198
IV.6 Average Fishing Effort Evolution Structure of the four goal-seeking strategies under the four shock scenarios (10% (a), 20% (b), 30% (c), 50% (d)). Below the efforts are displayed the total catches produced by each fishing strategies.	199
IV.9 Fishing Parameters: (bottom) Catchabilities (top) q_{ifp} of fleet f on species i in patch p; (top) Fisherman per Km^2 in Maatea $L_f(t)$. The share of labor involved in each fleet as well as the global density of fishermen in all areas are extrapolated from this distribution and also regarding the surface of the corresponding zone. .	199