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Charles Loiseau

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THÈSE DE DOCTORAT
DE L'UNIVERSITÉ PSL

**Méthodes pour évaluer la planification de l'espace marin :
La gestion des pressions anthropiques au cœur de la
conservation des écosystèmes**

Soutenue par
Charles Loiseau
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Méthodes pour évaluer la planification de l'espace marin : La gestion des pressions anthropiques au cœur de la conservation des écosystèmes

CHARLES LOISEAU



École doctorale Systèmes intégrés,
environnement et biodiversité
Thèse pour l'obtention du grade de docteur

Dirigée par
Joachim Claudet



À mes parents.

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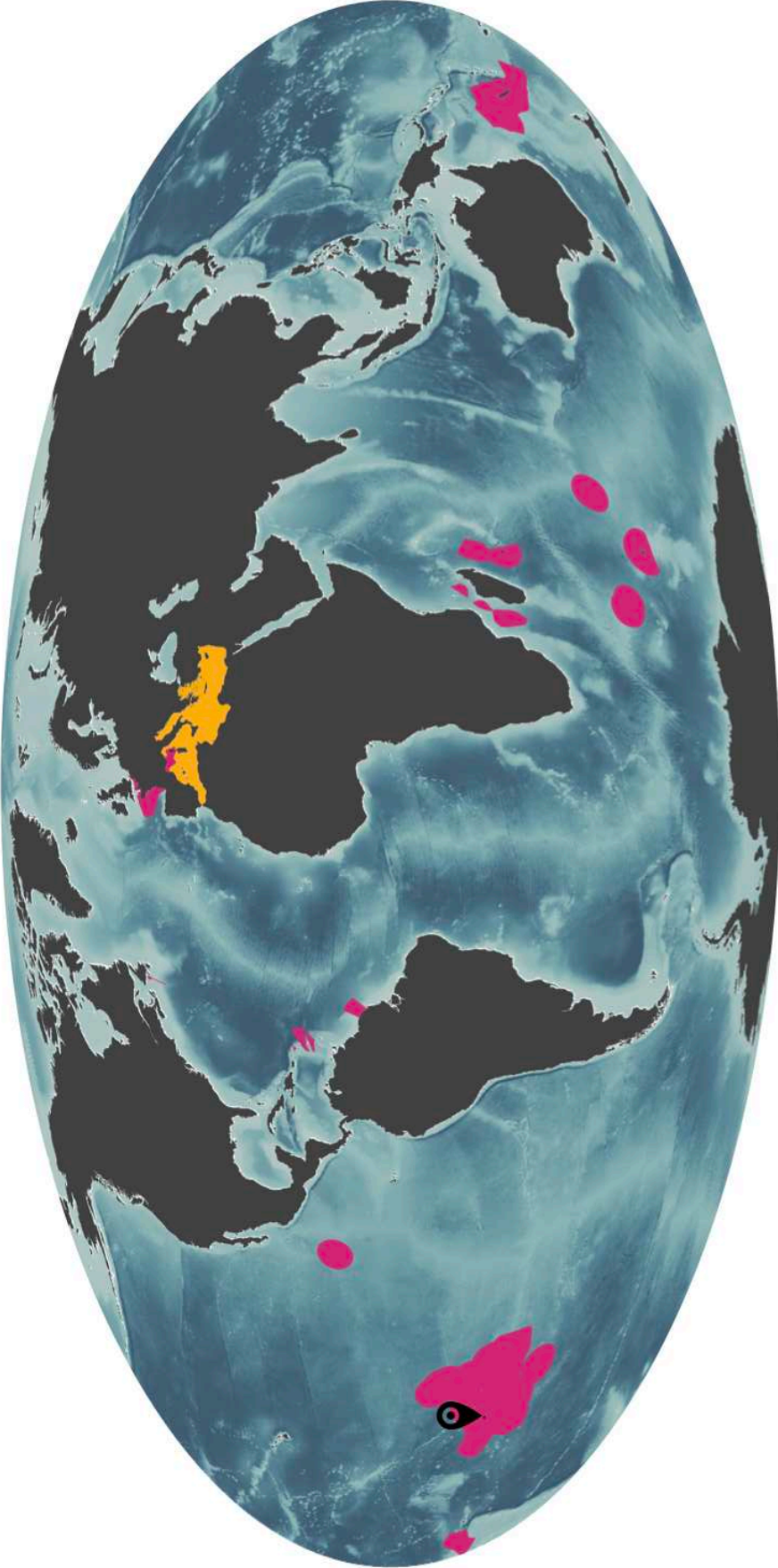
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Carte des cas d'études présentés dans cette thèse



Zone Economique Exclusive française
Chapitre 4



Mer Méditerranée
Chapitre 3 et 5



Moorea, Polynésie Française
Chapitre 1 et 2



Résumé




Objectif général | **Gérer les écosystèmes, c'est avant tout gérer les activités humaines qui les influencent**

Sous-objectifs | **L'évaluation de l'impact cumulé des activités humaines au coeur de la gestion écosystémique**

Opérationnalisation pour éclairer la gestion

Des aires marines protégées suffisamment fortes pour répondre aux enjeux de conservation des écosystèmes marins

Analyse des niveaux de protection et de leurs facteurs explicatifs

Cas d'études | **Moorea** *Polynésie française* 
France *Métropole et outre-mer* 
Mer Méditerranée 

L'évaluation de l'impact cumulé des activités humaines au coeur de la gestion écosystémique

Chapitre 1 | **Comment l'évaluation de l'impact cumulé des activités humaines peut éclairer une gestion écosystémique à l'échelle locale ?**
Utilisation de données locales sur les différentes activités dans le lagon de Moorea, et mise en évidence d'impacts significatifs des activités terrestres, à prendre en compte pour une gestion effective

Chapitre 2 | **Quelle potentialité pour l'outil d'évaluation de l'impact cumulé des activités humaines vers une évaluation de la planification spatiale maritime ?**
Application des réglementations de différents plans de gestion à l'impact cumulé des activités humaines pour évaluer les objectifs de conservation de la planification spatiale maritime

Des Aires Marines Protégées suffisamment fortes pour répondre aux enjeux de conservation

Chapitres 3 et 4 | **Quels niveaux de protection dans les aires marines protégées dans les eaux françaises France et en Mer Méditerranée ?**
Analyses des niveaux de protection de 1062 AMP en Mer Méditerranée et 524 AMP françaises, entre protection intégrale et aucune protection

Chapitre 5 | **Quels facteurs expliquent la présence de hauts niveaux de protection dans les aires marines protégées ?**
Cas d'étude en Mer Méditerranée, identifiant les facteurs préférentiels pour l'implémentation de plus hauts niveaux de protection

Résultats | **L'évaluation de l'impact cumulé des activités humaines à une échelle locale:**
- apporte aux gestionnaires une **évaluation transparente** du risque posé par l'éventail d'activités humaines présentes dans un territoire et des actions permettant d'organiser l'espace marin pour aller vers une **diminution des pressions** issues de ces activités (*chapitre 1*)
- permet de **quantifier la contribution de différents scénarios de planification à réduire les pressions** qui agissent sur les écosystèmes par des régulations adaptées, et apportent aux gestionnaires un outil pour replacer les objectifs de conservation au coeur du processus de planification (*chapitre 2*)

L'analyse des niveaux de protection des aires marines protégées en Mer Méditerranée (*chapitre 3*) et dans les eaux françaises (*chapitre 4*) démontre **l'échec de l'atteinte des objectifs internationaux de protection fixés pour 2020**, que ce soit en terme de surface, de représentativité écologique, ou de niveaux de protection suffisamment élevés pour garantir des bénéfices socio-écologiques, **remettant ainsi en cause leur rôle au sein de la planification spatiale maritime, dont elles sont un outil préférentiel pour la conservation des écosystèmes.**

Les plus hauts niveaux de protection en Mer Méditerranée semblent être opportunistes et ne répondent ni à des enjeux écologiques, ni à la nécessité de réduire les pressions issues des activités humaines (*chapitre 5*)

Conclusion | **Les résultats des différents chapitres mettent en lumière le manque de considération des objectifs de conservation, que ce soit au sein de la planification de l'espace marin, par un manque d'approches dédiées localement à l'évaluation de l'impact cumulé des activités humaines et d'évaluation de différents scénarios de planification au regard des enjeux de conservation, et au sein des aires marines protégées, pourtant dédiées à l'atteinte de ces objectifs de conservation mais qui manquent de mesures spécifiques visant à réduire les pressions préalablement identifiées. Les différentes approches exposées ici proposent de replacer la conservation des écosystèmes au coeur du processus de planification.**

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INTRODUCTION GENERALE

L'Océan a façonné la vie sur Terre, et celle des populations humaines. Les glaciations successives du Pléistocène ont poussé *Homo sapiens*, qui émerge il y a environ 300 000 ans en Afrique centrale, vers les côtes de ce continent, à la recherche de sources de nourriture abondantes et durables. Cette disponibilité en ressources a permis des installations pérennes et l'apparition des premières pratiques culturelles (Marean et al., 2007; McBrearty and Stringer, 2007). Il est probable que la dispersion d'*Homo sapiens* tout autour du globe ait pour point de départ ces petites populations côtières de réfugiés climatiques, profitant de conditions plus favorables pour sortir d'Afrique, il y a 70 000 ans. Leurs dispersions successives a pu bénéficier d'écosystèmes productifs le long des côtes, pourvoyeur de ressources abondantes dans des milieux qu'ils découvraient alors (Rito et al., 2019). Une théorie à propos du peuplement du continent américain, il y a environ 15 000 ans, propose d'ailleurs que ce peuplement a pu avoir lieu grâce aux forêts de macroalgues présentes le long des côtes du Pacifique, écosystèmes parmi les plus productifs de la planète, que Darwin compara plus tard avec les forêts tropicales terrestres dans son journal de bord lors de son voyage à bord du Beagle (Erlandson et al., 2007).

Des millénaires d'existences humaines ont ainsi été sous influence de l'Océan, au gré des périodes géologiques marquées par des variations climatiques et environnementales qui ont agi comme principal vecteur de changement sur la planète. Jusqu'à aujourd'hui, où les rôles semblent s'inverser, et où c'est désormais l'Homme qui agit comme principal force de changements: c'est l'ère de l'Anthropocène.

1.1 Accélération Bleue et conséquences sur l'Océan

1.1.1 L'expansion des activités humaines

On considère la période entre la fin du 18^{ème} siècle et le début du 19^{ème} comme un premier tournant dans l'intensification de l'utilisation de l'espace et des ressources environnementales. L'anthropocène, une ère dont le début coïncide avec l'invention de la machine à vapeur par James Watt en 1784, et qui sera marquée par la révolution industrielle (Crutzen, 2002). Le 20^{ème} siècle est quant à lui celui de la grande accélération, terme utilisé pour décrire des ruptures radicales dans différents compartiments sociaux, économiques et environnementaux (Steffen et al., 2015). A partir de 1950, de nombreux indicateurs socio-économiques et environnementaux marquent une forte croissance des activités humaines et de l'utilisation des ressources naturelles dans le monde entier : doublement de la population mondiale en 50 ans, développement des réseaux de transports, augmentation de l'utilisation en eau et en fertilisants, augmentation de gaz à effet de serre dans l'atmosphère, augmentation globale des températures, de l'acidification des océans, de la conversion des sols ou des pêcheries totalement exploitées (Steffen et al., 2015). Bien que des millénaires de présence humaine aient contribué à forger notre environnement (Stephens et al., 2019), les transformations induites par les sociétés humaines n'ont jamais été aussi rapides et étendues que depuis 1950. Même si légèrement retardé, le constat est le même dans les mers et océans. La « Révolution marine », défini en 1970 comme étant « tout aussi importante que les révolutions agricoles et industrielles précédentes », marque le contraste entre l'émergence de technologies et de connaissances sur le milieu marin permettant un accroissement des activités en mer d'une part, et le manque de cadre légal institutionnel pour les réguler d'autre part, qui pourrait mener à la surexploitation, et à terme, à la destruction des ressources marines (Ray 1970).

Similairement à la grande accélération, l'accélération bleue pose le constat d'une intensification de la demande en espace et en ressources marines depuis les années 1970 (Jouffray et al., 2020). L'augmentation de la population mondiale et du niveau de consommation accroît la demande en ressources issues de l'Océan, notamment dans un contexte où le milieu terrestre est déjà fortement exploité. Par exemple, la diminution des métaux rares les plus accessibles sur les continents et leurs disponibilités relatives aux contextes géopolitique et commercial, couplée à des besoins grandissants en ces matériaux pour le numérique, la transition vers des énergies renouvelables ou l'électrification des réseaux de transport, ont déclenché l'intérêt de nombreux pays et entreprises qui se tournent aujourd'hui vers l'Océan pour y exploiter ces ressources (Levin et al., 2020). Des secteurs historiques comme la pêche ou le transport maritime sont désormais rejoints en mer par des activités liées à la production d'énergie ou à la prospection minière. Ces augmentations combinées de la demande en ressources et en espace, par des secteurs historiques et des secteurs émergents, peut entraîner une compétition et le risque d'appropriation par l'un des secteurs, désigné par le terme « *ocean grabbing* » (Bennett et al., 2015), risque souvent déséquilibré compte-tenu du niveau de concentration observés dans le secteur privé ayant des intérêts dans l'Océan (Viridin et al., 2021). Par exemple, le secteur de la pêche perd souvent des zones, que ce soit au large, au profit de concessions d'exploitations de gaz ou d'hydrocarbures, ou sur la côte, à cause de développement d'infrastructures favorisant le tourisme ou l'aquaculture (Queffelec et al., 2021). L'Océan subit une pression grandissante issues des besoins humains en nourriture, énergie, matériel et en espace, chacun de ces besoins engendrant des pressions qui se superposent sur les écosystèmes abritant les ressources utilisées, résultant en de nombreux impacts qui mettent en péril la durabilité de ces ressources pour l'Homme, tout en générant des compétitions intra- et intersectorielles.

1.1.2 Conséquences des activités humaines en mer

L'Océan représente 70% de la surface de la Terre, avec une profondeur moyenne de 4 000 mètres. Malgré son gigantisme et l'apparente inaccessibilité de certaines zones qui ont longtemps laissé à penser que l'Océan et ses ressources étaient inexhaustibles, les activités humaines se répartissent dorénavant sur toute sa surface (Halpern et al., 2008b) et même récemment dans presque tout son volume, jusqu'aux abysses (Ramirez-Llodra et al., 2011). Déjà profondément transformé par l'Homme (Steneck and Pauly, 2019; Yasuhara et al., 2012), 97.7% de l'océan est aujourd'hui soumis aux effets cumulés des activités humaines, dominés par le changement climatique (anomalies de température de surface, acidification, augmentation du rayonnement ultraviolet) et la pêche, effets qui continuent d'augmenter sous l'impulsion de l'accélération bleue (Halpern et al., 2015; Jouffray et al., 2020). La pêche industrielle occupe 55% des océans (Kroodsma et al., 2018), avec un effort de pêche qui a triplé depuis 1970 (Bell et al., 2017). Malgré l'augmentation constante de l'effort de pêche et les améliorations techniques qui permettent de pêcher dans des zones de plus en plus lointaines et profondes (Tickler et al., 2018), les captures par unité d'effort (*catch per unit of effort*, CPUE) n'ont cessé de diminuer depuis 1950, suggérant une surexploitation de certaines pêcheries (Rousseau et al. 2019). Entre 40 et 50% des écosystèmes tropicaux et tempérés sont pêchés au-delà des limites permettant d'assurer la durabilité des populations de poissons ciblés (Link and Watson, 2019).

L'ensemble de l'océan est également soumis indirectement aux actions humaines responsables du changement climatique qui ont, depuis la révolution industrielle, rejetés en forte quantité des

gaz à effet de serre dans l'atmosphère, notamment le dioxyde de carbone et le méthane. Cette augmentation de gaz à effet de serre dans l'atmosphère entraîne de profondes modifications en cascade dans les océans : la hausse des températures atmosphériques augmente la température globale de l'océan, qui absorbe 90% de l'excès de chaleur accumulé dans l'atmosphère. Cette augmentation globale de la température, bien que différenciée dans l'espace et dans le temps, augmente la dilatation des masses d'eaux et accélère la fonte des calottes polaires, entraînant dans certaines régions du monde une élévation du niveau de la mer qui expose les zones côtières à des événements climatiques de plus en plus fréquents (IPCC, 2022). L'océan agit également comme une pompe à carbone, et a ainsi absorbé 41% du carbone d'origine anthropique, mais ce rôle de régulation est amené à diminuer dans les prochaines décennies (McKinley et al., 2016). L'augmentation du carbone dissous dans les océans entraîne une diminution du pH, et donc une acidification, qui perturbe la balance chimique des composés carbonés, notamment le carbonate de calcium (CaCO_3). La diminution du pH entraîne la diminution du taux de saturation du CaCO_3 , qui, dans ces conditions, se dissout et n'est plus disponible pour les organismes calcifiants, des algues calcaires microscopiques aux récifs coralliens. Le fonctionnement de nombreux écosystèmes pélagiques ou côtiers est profondément perturbé, mettant en péril les services écosystémiques associés tels que les pêcheries, l'aquaculture, la protection côtière ou les services culturels (Doney et al., 2020; He and Silliman, 2019). Le changement climatique a également des conséquences sur la productivité des pêcheries, et peut augmenter la vulnérabilité de certaines populations côtières exposées à une baisse de production (Thiault et al., 2019).

A l'échelle mondiale, 85% des pays possèdent un accès direct à la mer, représentant plus de 1 600 000 km de côtes (soit 400 fois le tour de la Terre) le long desquelles 40% de la population mondiale vit, un pourcentage atteignant plus de 80% dans la moitié des pays côtiers (Martínez et al., 2007). Les écosystèmes côtiers représentent une faible fraction des océans, mais concentrent la majorité des services écosystémiques marins (Barbier, 2017; Barbier et al., 2011), les rendant particulièrement exposés aux pressions issues des multiples activités humaines s'y déroulant, sur terre comme en mer (Bellwood et al., 2004; Lotze et al., 2006). Ainsi, la surface couverte par herbiers marins diminue chaque année de 7% depuis 1990 (Waycott et al., 2009) ; un tiers des mangroves ont été détruites depuis 1950 (Alongi, 2002) ; les côtes sableuses pourraient reculer de 7% d'ici 2025 et 25% d'ici la fin du siècle en fonction des scénarios climatiques (Vousdoukas et al., 2020) ; les écosystèmes coralliens sont de 30 à 80% dégradés dans le monde entier (Pandolfi et al., 2003) et peuvent localement atteindre des taux de couverture inférieurs à 10% (Bellwood et al., 2004) sous l'effet de multiples pressions (Hughes et al., 2017). La biogéochimie générale des zones côtières est modifiée, menant à l'émergence de zones hypoxiques et des cas d'eutrophisation, avec des effets délétères sur l'abondance et la diversité des organismes marins (Breitburg et al., 2018; Doney, 2010). Non seulement les écosystèmes côtiers (récif coralliens, mangroves, herbiers, récifs rocheux) sont les écosystèmes marins les plus soumis aux effets des activités humaines, mais ce sont aussi ceux qui subissent une augmentation de l'exposition à ces effets parmi les plus importantes entre 2003 et 2013 (Halpern et al., 2019).

1.2 D'une gestion sectorielle à l'approche écosystémique

1.2.1 Stratégies mono-spécifiques/sectorielles

La question d'une exploitation durable des ressources naturelles est antérieure à l'accélération bleue. La gestion des ressources naturelles s'est d'abord concentrée sur des composants restreints de la biodiversité, associée à des objectifs de production économique, comme par exemple la gestion des pêcheries, basée sur des statistiques historiques pour les espèces d'intérêt commerciale, afin de maximiser leur exploitation (DeFries and Nagendra, 2017). Déjà, en 1902, était créé l'ICES (*International Council for the Exploration of the Seas*), pour faire face aux déclin des populations des espèces de poisson pêchées dans les mers du Nord et Baltique (Smith 1992). Ces stratégies de gestion reposent sur l'idée d'équilibre stable au sein des écosystèmes, et ont pour but de maintenir un rendement maximal durable sur des ressources d'intérêt (DeFries and Nagendra, 2017). De ce fait, la gestion des ressources naturelles s'est compartimentée par secteur (agriculture, transport, eau, pêche, tourisme), et seulement sur certains composants de l'écosystème, suivant une vision utilitariste de la conservation des écosystèmes (Mace, 2014). Dans les océans, cette gestion sectorisée s'est construite sur les bases du *Mare liberum*, la liberté des mers pour quiconque, dans le but d'assurer que les océans restent accessible à tous (essentiellement les pays d'Europe de l'Ouest) pour le commerce, l'expansion coloniale, l'équilibre international, et même la gestion des stocks de pêche (Couper 1992).

Cependant, l'océan est soumis à de nombreuses pressions issues de secteurs économiques distincts, eux-mêmes gérés au travers de différentes agences et réglementations, à des échelles locales, nationales et internationales (Salomon and Dross, 2018). Il a été relevé aux États-Unis pas moins de 20 agences responsables de 140 statuts réglementaire et légaux relatifs à l'océan, et dans l'Union Européenne, plus de 200 législations s'appliquent à la gestion du domaine marin, compartimentées là aussi par secteur, avec le risque de réponses fragmentaires qui ne permettraient pas de résoudre des conflits ou d'agir sur les effets cumulés qui en sont issus (Boyes and Elliott, 2014; Crowder et al., 2006). Ainsi, malgré des décennies de stratégies de gestion sectorielles et/ou mono-spécifiques, le déclin de la biodiversité et des services écosystémiques associés persiste, et s'est même accéléré (IPBES, 2019), suggérant que les réponses apportées sont insuffisantes par rapport aux forces qui les engendrent (Johnson et al., 2017). Par exemple, 90% de la mortalité annuelle du marlin blanc (*Kajikia albida*) est due à des captures accidentelles par la pêche palangrière, et une gestion mono-spécifique centrée sur les stocks des espèces visées par cette technique (principalement les thons), ne permet pas le maintien du marlin (Pikitch et al., 2004). Autre exemple avec des tortues de mer dans l'Océan Pacifique (*Caretta caretta*, la couanne, et *Dermochelys coriacea*, la tortue luth) qui se répartissent sur tout le bassin, mais dont les prises accidentelles sont gérées seulement dans les eaux américaines et pas dans les pays frontaliers (Crowder et al., 2006). La gestion mono-spécifique des pêcheries n'a pas empêché la surexploitation de nombreux stocks de poissons dans le monde, car ce type de gestion ne parvient pas à répondre simultanément à des objectifs économiques, écologiques et sociaux (Jennings, 2006). De plus, ces objectifs, communs à différents secteurs, peuvent entrer en collision et entraîner des conflits entre usagers de plus en plus fréquents au fur et à mesure que la demande humaine en espace et en ressources marines augmente et que les différents usages se superposent, dans l'espace et le temps (Delacámara et al., 2020; Hegland, 2018; Jouffray et al., 2020; Klinger et al., 2018). Comme le résume Larry Crowder dans *Science* en 2006 :

«Historiquement, la gestion de l’Océan s’est concentrée par secteur. [...] C’est comme si plusieurs médecins spécialisés dans leurs domaines et sans réelle communication tentaient de traiter un patient avec de multiples symptômes. Plutôt que de résoudre le problème, le traitement combiné peut même l’exacerber » (Crowder et al., 2006).

1.2.2 *Vers une gestion holistique de l’Océan*

Afin de répondre aux manques illustrés précédemment, la gestion des ressources naturelles s’est progressivement tournée vers une approche plus holistique, adoptant une échelle qui permet d’intégrer l’ensemble des composants biotiques et abiotiques d’un lieu donné (une plage, une embouchure de rivière, un bassin océanique, etc.), ainsi que les interactions qui les lient : l’écosystème. Chacun des composants de l’écosystème remplit des fonctions indispensables à son fonctionnement, dynamique dans le temps et l’espace, et sont en interactions les uns avec les autres, formant un système complexe. Les sociétés humaines ont depuis toujours modifié et géré les écosystèmes afin d’en tirer des ressources alimentaires, matérielles, et des valeurs culturelles et récréatives. Comme vu plus haut, la gestion de ces ressources s’est historiquement centrée sur la ressource elle-même, dans le but de maximiser son rendement. Ces stratégies de gestion reposaient sur l’hypothèse que lorsqu’un écosystème atteint son équilibre théorique final et stable (i.e. climax), sa productivité peut être gérée indéfiniment par un contrôle permettant la diminution des variations pouvant perturber cet équilibre. Or, les avancées scientifiques réalisées dans le domaine de l’écologie lors des années 1970-80 ont démontré que ce n’était pas le cas : non seulement la notion de climax a été remise en cause, mais il a été montré que plus le niveau de contrôle exercé sur les variations naturelles d’un écosystème est élevé, moins le système est résilient à des perturbations externes : par exemple, dans le sud-ouest des Etats-Unis, les rivières le plus soumises à des contrôles de leurs flux par des barrages abritent des populations de poissons moins résilientes à l’arrivée d’espèces exotiques (Holling and Meffe, 1996). De plus, ces stratégies ignorent les interconnexions et interdépendances entre les différents composants de l’écosystème, pouvant mener à des effets imprédictibles, inattendus et indésirables (DeFries and Nagendra, 2017).

Les changements opérés par les activités humaines au sein d’un écosystème, par l’exploitation d’une espèce, la dégradation d’habitats physiques ou la modification des conditions environnementales comme celles engendrées par le réchauffement global, induisent donc des changements dans les composants biotiques et abiotiques d’un écosystème ainsi que dans les liens qui les unissent, perturbant la structure et les fonctions du système. Par exemple, l’introduction des dragues pour récolter des huîtres dans la baie de Chesapeake, aux États-Unis, a conduit à une destruction quasi-totale des récifs d’huîtres, avec des répercussions allant au-delà de la perte de cette espèce d’intérêt commerciale : sans ces vastes récifs biogéniques constitués d’organismes filtrant la colonne d’eau, l’apport en nutriments d’origine terrestre, en constante augmentation depuis la déforestation commencée dès l’arrivée des colons européens, ne peut plus être jugulé par ces organismes, résultant à une eutrophication de la baie, à de l’hypoxie et à des épidémies sur les récifs d’huîtres restant (Jackson et al., 2001) Les écosystèmes montrent différents degrés de résilience face à des changements abrupts, c’est-à-dire différentes réponses qui permettent de plus ou moins conserver leurs structures et fonctions après des perturbations, jusqu’à un point de non-retour où la structure et les fonctions changent drastiquement, pouvant conduire à leurs remplacements par d’autres fonctions. Ces notions sont

fondamentales pour la gestion des ressources naturelles puisque ce sont ces structures et fonctions des écosystèmes qui fournissent aux sociétés humaines ce qui est défini par le terme « services écosystémiques » et sur lesquels s'appuient l'existence humaine : ressources alimentaires, abri, matériaux, valeurs culturelles, activités de loisirs, etc.... La nécessité de réfléchir une gestion du milieu marin prenant en compte l'effet des multiples activités sur l'ensemble de l'écosystème considéré est d'autant plus important puisque comme décrit plus haut : la demande en ressources et espaces maritimes augmente sans discontinuer depuis les années 1970 (l'accélération bleue) et va continuer à le faire, sous l'impulsion de l'exploitation des énergies renouvelables en mer, par exemple ; les technologies associées aux activités maritimes, historiques et émergentes, continuent également de se développer, et permettent de mieux tirer parti des ressources disponibles (Markus, 2018), suggérant que la révolution marine définie par Carleton Ray en 1970 n'est pas terminée et que le risque de décalage entre l'exploitation des ressources maritimes et les cadres institutionnels permettant une gestion durable de ces ressources est toujours présent, comme l'écrivent les philosophes de l'environnement Catherine et Raphaël Larrère « Nos techniques sont devenues si puissantes qu'elles débordent nos capacités de prévoir et d'imaginer leurs effets à long terme. Les risques s'amplifient donc en fonction de notre puissance technique et du caractère cumulatif de techniques dont nous sommes devenus dépendants » (Larrère and Larrère, 2015)

Le concept de gestion basé sur les écosystèmes (*Ecosystem-Based Management*, EBM) apparaît dans les années 1980 et est défini comme « une approche intégrée de la gestion qui considère l'écosystème dans son entièreté, humains inclus. L'objectifs de l'EBM est de maintenir un écosystème sain, productif et résilient, afin qu'il puisse pourvoir aux besoins et désirs humains. L'EBM diffère des approches qui se concentrent sur seulement une espèce, un secteur, une activité ou une problématique ; il considère l'impact cumulé de différents secteurs » (McLeod et al., 2005). Deux raisons principales sont soulevées pour une adoption internationale des principes de l'EBM : 1) les interactions entre les différents composants des écosystèmes sont cruciaux pour leur résilience et leur bonne santé. Lorsque ces interactions disparaissent ou sont sévèrement dégradés, par la disparition d'espèces où la dégradation d'habitats, entre autres, la capacité des écosystèmes à se régénérer décline en même temps que les services qu'ils produisent; 2) les stratégies de gestion des océans appliquées jusqu'alors ne permettent pas de maintenir la durabilité de ces services (Leslie and McLeod, 2007). L'EBM consiste donc en une approche holistique de gestion des ressources naturelles dans leur contexte socio-environnementale, et le cadre des systèmes socio-écologiques tel que défini par Ostrom est particulièrement adapté pour caractériser ses composants (les ressources, la gouvernance et les usagers) et leurs interactions, et identifier les facteurs menant à la durabilité du système (Ostrom, 2009). On retrouve la notion d'EBM depuis les années 1980 dans des conférences et traités internationaux sur le milieu marin, puis au sein de l'organisation des Nations Unies pour l'alimentation et l'agriculture (FAO, *Food and Agriculture Organization*), à propos des pêches, jusqu'à aujourd'hui, dans la politique intégrée maritime de la Commission européenne (Enright and Boteler, 2020; Gee, 2019), ainsi que dans des exemples bien connus d'applications réussies, comme dans la baie de Chesapeake, aux Etats-Unis (Lefcheck et al., 2018), ou sur la grande barrière de corail, en Australie (Olsson et al., 2008).

Il existe un consensus pour réorienter les stratégies de gestion vers l'EBM, mais son opérationnalisation reste cependant un défi. Bien que les concepts directeurs de l'EBM soient reconnus dans la littérature scientifique, un manque de clarté subsiste quant à une approche standardisée de leurs applications, et de fait, les concepts de l'EBM ne se retrouvent pas – ou très peu – en termes opérationnels, notamment dans les plans de gestion (Arkema et al., 2006; Delacámara et al., 2020; Long et al., 2015; O'Higgins et al., 2020). De plus, les cadres de gouvernances existants sont issus d'une gestion traditionnelle, sectorielle et/ou centrés sur des ressources particulières, et permettent difficilement d'implémenter l'EBM (Leslie and McLeod, 2007; O'Higgins et al., 2020). Un autre challenge concerne le monitoring d'une telle stratégie, à l'aide d'indicateurs qui doivent incorporer les objectifs écologiques, économiques, sociétaux et institutionnels (Katsanevakis et al., 2011; Leslie and McLeod, 2007). Enfin, malgré de nombreuses données et observations scientifiques récoltées les dernières décennies à propos de la complexité des écosystèmes et de leur variation dans le temps et l'espace, leurs caractérisations et leurs réponses aux interventions humaines restent un défi (Arkema et al., 2006). La gestion des écosystèmes, pouvant être qualifiée de « *wicked problem* » (problème pernicieux, sans solution clairement identifiable) peut s'appuyer sur les concepts de l'EBM pour éviter la mise en place de solutions trop simplistes au détriment du système géré, où au contraire le maintien d'un *statu quo* face à une complexité pouvant paraître inextricable (DeFries and Nagendra, 2017).

1.3 Application de la gestion écosystémique en mer

1.3.1 La planification spatiale maritime

La planification spatiale maritime (désignée ensuite par son acronyme anglais MSP pour *Marine Spatial Planning*) est un processus permettant l'opérationnalisation des principes de l'EBM en mer, concentré sur un espace clairement défini (Domínguez-Tejo et al., 2016; Ehler, 2020; Katona et al., 2017). Ses origines sont attestées dès 1976, par la création du Parc de la Grande Barrière de Corail, en Australie, avec la mise en place de zones visant à contrecarrer la dégradation observée du milieu, causée par des activités humaines multiples, sur terre comme en mer (Ehler et al., 2019; Olsson et al., 2008). Le développement des activités humaines lors des dernières décennies a engendré une forte compétition pour l'espace, génératrice de conflits entre usagers. Ainsi, à partir des années 2000, le MSP est vu comme un outil préférentiel pour une implémentation effective de l'EBM en mer, pouvant être perçu comme trop vague, abstrait et complexe pour les gestionnaires, et permettant de résoudre les conflits usagers-usagers et usagers-ressources (Arkema et al., 2006; Douvère, 2008; Schubert, 2018). Cependant, alors que l'EBM se concentre sur l'écosystème et ses différentes unités, et considère les activités humaines comme autant d'impacts potentiels sur la durabilité des ressources issues des écosystèmes, le MSP est un « processus de planification qui vise à résoudre la fragmentation liée à l'approche sectorielle, et à analyser et organiser les activités humaines en mer pour atteindre des objectifs économiques et sociétaux tout en sauvegardant l'intégrité écologique » (le Tissier, 2020). Trois objectifs principaux sont donc définis pour les MSP, que l'on retrouve dans la directive européenne de 2014 sur la planification spatiale maritime : promouvoir le développement économique d'activités durables (la croissance bleue), prévenir et réduire les conflits entre usagers, et appliquer une approche écosystémique pour assurer le maintien des biens et services fournis par les écosystèmes (Ehler et al., 2019). Aujourd'hui, la moitié des pays côtiers ont officiellement mis en place des initiatives de MSP pour la gestion de leurs territoires marins,

couvrant 17% des zones économiques exclusives (ZEE), un pourcentage qui pourrait s'élever à 50% d'ici 2030 (Ehler, 2020). La majorité de ces initiatives ont été soulevé par le besoin de prévenir les conflits entre usagers, la nécessité d'une approche intégrée pour la gestion, des préoccupations sur la conservation du milieu marin, la gestion des activités humaines nouvelles, et les conflits entre usages et environnement (Ehler, 2020).

Mais face aux objectifs économiques et sociétaux, l'environnement n'est généralement pas considéré comme une priorité absolue lors de la mise en pratique de MSP, notamment lors des phases de participation et de consultation (Foley et al., 2010; Schubert, 2018). Ainsi même si la conservation des écosystèmes par le biais de la création d'aires marines protégées est l'un des principal facteur déclencheur de la planification spatiale maritime, ces objectifs initiaux se retrouvent en conflit avec les intérêts des différents secteurs lors du processus de planification et rendent difficile l'atteinte d'objectifs comme le « bon état environnemental » des eaux européennes (qui au contraire, se détériore), ou la surface d'aires marines protégées à atteindre (Jones et al., 2016). Au contraire, la planification peut être menée par le besoin d'organiser dans l'espace des secteurs économiques stratégiques, laissant les enjeux de conservation des écosystèmes au second plan. Par exemple, en Belgique, le processus de planification débuté en 2003 a eu pour but premier d'identifier des zones propices à l'extraction de granulats et à l'implémentation de fermes marines éoliennes, et les questions liées à l'environnement sont venues lors d'une seconde phase ; même chose pour la planification maritime du Golfe de Gascogne, où la priorité a été donnée à la promotion des énergies utilisant les courants et marées ; et également pour la mer de Barents, où c'est cette fois-ci l'exploitation de gaz et d'hydrocarbures qui a pris le pas sur les objectifs de conservation (Douvere et al., 2007). Plus généralement, et malgré ses objectifs cités plus hauts, différents exemples de MSP montrent qu'ils se concentrent généralement sur un secteur particulier, lié à des priorités nationales d'importance stratégiques, et que les considérations économiques prennent le pas sur les objectifs de conservation (Jones et al., 2016). Plusieurs études ont établi les progrès à faire vers une application du MSP intégrant l'EBM, incluant notamment une meilleure intégration de la dimension humaine et de données sociales, permettant une évaluation de l'impact cumulé des activités humaines qui pourrait servir de base pour l'adoption de compromis entre les objectifs économiques, sociétaux et environnementaux (Ansong et al., 2017; Frazão Santos et al., 2018; Menegon et al., 2018),

1.3.2 *Challenges liés à l'opérationnalisation du MSP*

L'évaluation de l'impact cumulé des activités humaines est requise dans le cadre de l'EBM et du MSP (Halpern et al., 2008a; Leslie and McLeod, 2007). Puisque les mesures de gestion vont en particulier agir sur les activités et les usages, il est crucial de disposer des meilleures données disponibles sur leur répartition, leur intensité, et le lien entre les pressions exercées par ces activités sur chaque composants de l'écosystème. La gestion des écosystèmes marins, qui sont par nature complexes, dynamiques et historiquement moins étudiés que les écosystèmes terrestres, reposent donc sur la gestion des activités humaines et de leurs pressions, et non pas sur la gestion directe des écosystèmes eux-mêmes (Ehler and Douvere, 2009; Leslie and McLeod, 2007). L'évaluation de l'impact cumulé des activités humaines (ci-après désignée sous son acronyme anglais CIA, pour *Cumulative Impact Assessment*) a pour ambition de cartographier les effets additifs, synergiques et antagonistes des multiples pressions issues d'activités humaines

à un endroit donné, sur différents composants de l'écosystème. Plus spécifiquement, il est nécessaire : 1) d'identifier les activités humaines responsables de pressions sur les écosystèmes, sachant qu'une activité peut être responsable de plusieurs pressions (par exemple la pêche au chalut de fond prélève un grand nombre d'individus et d'espèces marines, perturbant l'ensemble du réseau trophique, et modifie ou détruit le substrat et les communautés benthiques associées), et qu'une même pression peut-être issue de différentes activités (par exemple, la perte d'habitats benthiques peut être engendrée par le chalutage de fond, mais aussi par des activités de dragage pour nettoyer un chenal ou l'entrée d'un port, ou par l'extraction de granulats marins servant de matériaux de construction); 2) d'identifier les composants de l'écosystèmes qui seront utilisés pour refléter l'ensemble du système : il peut s'agir d'espèces, comme les hippocampes (Zhang and Vincent, 2019), de communautés, comme les récifs coralliens (Andrello et al., 2022) ou les récifs coralligènes de Mer Méditerranée (Bevilacqua et al., 2018), des habitats benthiques, comme ceux listés par la directive européenne cadre sur le milieu marin (Quemmerais-Amice et al., 2020), ou bien d'une combinaison de ces différents composants (Korpinen et al., 2021) ; 3) de lier pressions et impacts sur chacun des composants de l'écosystème considéré, c'est à dire de traduire comment un certain niveau de pression va agir négativement sur l'écosystème ou elle s'exerce, généralement estimé par un jugement à dire d'expert, ou par une analyse de la littérature scientifique (Halpern et al., 2007; Korpinen and Andersen, 2016). La question de la disponibilité et de la collecte de données sur l'écosystème, sur les activités humaines et sur comment ils interagissent devient donc central dans le processus de planification (Ban et al., 2013; Foley et al., 2010; Levin et al., 2014), et le manque de données appropriées peut devenir un frein lors de la mise en place d'un MSP, notamment à l'échelle locale (Mahadeo, 2022). Bien que considéré comme un outil indispensable au MSP et plus largement à l'approche écosystémique, l'évaluation de l'impact cumulé est peu utilisé à l'échelle locale, et rarement intégré dans les concertations et les prises de décisions au cours du processus de planification (Halpern et al., 2008a; Hodgson et al., 2019), du fait du manque d'outils et d'approche standardisée, de la nécessité d'informer sur les incertitudes qui y sont liées et du manque d'intégration du domaine terrestre (Domínguez-Tejo et al., 2016).

Un autre challenge concernant le MSP est son évaluation. La planification s'appuie sur un besoin, institutionnel ou sociétal, et définit des objectifs économique, sociétaux et environnementaux. L'évaluation de la planification est définie comme une évaluation systématique de politiques publiques aux objectifs multiples, et fournit des informations indispensables sur l'efficacité d'un MSP et sa capacité à atteindre ses objectifs (Varjopuro, 2019), notamment comme indiqué plus haut lorsque différents secteurs entrent aux conflits sur l'espace et les ressources. Elle peut-être : *a priori*, c'est-à-dire qu'elle peut aider à identifier le meilleur scénario parmi différentes propositions avant leur implémentation ; formative, évaluant le progrès durant le cycle de planification lui-même ; rétrospective, s'appuyant sur des résultats mesurables permettant d'identifier les conséquences de la planification et leur causalité, afin de d'adapter le prochain cycle de planning (Laurian et al., 2010; Varjopuro, 2019). L'évaluation peut donc porter sur le cycle de planification lui-même, sur l'implémentation des politiques de gestion, mais aussi sur les résultats dus à l'implémentation du MSP, au travers d'indicateurs économiques, sociologiques et écologiques qui permettent de suivre les changements dans le système dues au MSP, qu'il s'agisse de scénarios de MSP ou d'indicateurs mesurés directement sur le terrain (Day, 2008; Piet et al., 2020; Stelzenmüller et al., 2021). Étant donné l'éventail d'activités humaines et de

composants de l'écosystème, de la variété des échelles spatiales et temporelles, et du caractère dynamique de l'océan, rendant extrêmement complexe le système socio-écologique auquel le MSP s'applique, la formalisation d'une méthode d'évaluation est difficile, et peu d'outils existent pour mener à bien une telle entreprise (Carneiro, 2013). Deux problématiques sont particulièrement soulevées : la première est l'attribution de la causalité, c'est-à-dire dans quelles mesures les changements observés dans le système sont à mettre au crédit des mesures de gestion mises en place (Carneiro, 2013; Laurian et al., 2010). Cette question est cruciale, notamment dans un milieu dynamique en trois dimensions, soumis à de nombreux et différents facteurs, d'origines locales ou lointaines (Rocha et al., 2015), et où une situation contrefactuelle est difficile à établir (Carneiro, 2013; Laurian et al., 2010). La deuxième est le décalage temporel entre la mise en place de ces mesures de gestion et des changements observables dans le système (Carneiro, 2013). Par exemple, si l'on cherche à évaluer les objectifs environnementaux d'un MSP par le biais de l'efficacité de ses aires marines protégées à augmenter la biomasse et l'abondance d'espèces de poissons, ou à permettre le maintien ou la restauration d'un habitat particulier, un tel changement peut survenir des années, voire des dizaines d'années après la mise en place des AMP (Claudet et al., 2008; Edgar et al., 2014; Fraser et al., 2019).

1.4 Les Aires Marines Protégées

1.4.1 Définition et concepts

Les aires marines protégées (AMP) sont antérieures aux principes de l'approche écosystémique et de la planification spatiale en mer, mais se fondent parfaitement dans leurs concepts, comme outil de zonage en faveur de la protection des océans et de l'utilisation durable de leurs ressources, notamment depuis que la convention sur les droits de la mer attribue des droits souverains aux États au sein de leur ZEE, de pair avec le devoir de protéger l'environnement marin (Humphreys and Clark, 2019). Malgré la pluralité des formes qu'une AMP peut prendre, elles peuvent être définies comme « un espace géographique clairement défini, reconnu, spécialisé et géré par des moyens légaux ou d'autres moyens efficaces, visant à assurer la conservation à long terme de la nature et des services écosystémiques et valeurs culturelles qui y sont associés. » (IUCN, 2008). Plusieurs concepts et hypothèses soutiennent les AMP comme outil pour la conservation des écosystèmes marins (Claudet, 2012; Lubchenco et al., 2003) : 1) par l'arrêt des activités prélevant des espèces et/ou causant la détérioration des habitats marins, la mortalité des populations de poissons diminue, leur nombre et leur taille augmentent, et la qualité de leurs habitats s'améliore ; 2) la structure des populations pêchées, en terme d'âge et de taille, revient à des conditions d'avant leurs exploitations ; 3) la pêche a historiquement ciblés les espèces les plus hautes dans le réseau trophique, et a graduellement réorienté ses efforts vers des espèces de plus en plus bas au fur et à mesure du déclin des espèces (phénomène décrit sous les termes « *fishing down the marine food web* »), entraînant un changement de structure dans tout le réseau trophique et donc dans l'écosystème (Pauly et al. 1998 ; Steneck & Pauly 2019). Par l'arrêt des activités de prélèvement, des effets indirects en cascade à tous les niveaux du réseau trophique sont attendus, par le biais des relations proie/prédateurs entre chacun des niveaux ; 4) les espèces ciblées par la pêche sont plus abondantes et de plus grande taille à proximité de l'AMP, par effet de débordement (ou *spillover*); 5) les AMP peuvent jouer le rôle de source pour les larves de poissons à une échelle régionale. Plus qu'un outil de gestion des pêcheries, une AMP vise à protéger l'ensemble des espèces, des habitats et des conditions

environnementales en leur sein, avec des effets direct et indirects, à court et moyen/long terme, sur les écosystèmes mais également sur les humains.

Ces hypothèses ont été validées empiriquement, par l'étude des nombreuses AMP mises en places dans les dernières décennies. Les AMP permettent effectivement d'augmenter la densité et la taille (donc la biomasse) d'espèces de poissons auparavant ciblés par la pêche (Gill et al., 2017; Lester et al., 2009), jusqu'à 5 fois plus importante dans les AMP qu'à l'extérieur (Edgar et al., 2014). Les effets sont aussi constatés sur l'ensemble du réseau trophique : une augmentation du nombre de prédateurs supérieurs, comme les requins, a été constaté (Edgar et al., 2014) ; des écosystèmes côtiers, transformés par une surabondance d'oursins en lande désolée (*sea urchin barren*, landes à oursins) sont revenus à des états plus complexes grâce à l'augmentation du nombre de prédateurs de ces herbivores, favorisant ainsi le développement de forêts algales, habitats complexes abritant une biodiversité plus riche (Guidetti and Sala, 2007; Micheli et al., 2005).

Au-delà de ces effets directs, les AMP apportent indirectement d'autres bénéfices au-delà de leur périmètre, notamment par deux mécanismes : un réapprovisionnement conséquent en larves de poissons, due à la présence en plus grand nombre et de plus grande taille d'individus reproducteurs, et ce jusqu'à des milliers de kilomètres de distance (Manel et al., 2019), et l'export d'individus à proximité des AMP, ou la biomasse peut-être 1.5 fois importante que comparée à des zones plus distantes (Di Lorenzo et al., 2020). Les activités de pêche sont alors positivement impactées par la proximité d'une AMP malgré la perte initiale de zones de pêches, par une augmentation des captures par unité d'effort (Goñi et al., 2010; Russ et al., 2004). Cependant, les activités de pêche ne sont pas les seules concernées par la présence d'AMP : la conservation des écosystèmes marins les rend particulièrement attractifs et de nombreuses activités touristiques peuvent se développer à proximité (Leenhardt et al., 2015). Plus généralement, l'amélioration des conditions écologiques observées dans les AMP renforce leur capacité à délivrer des biens et services écosystémiques profitables à tous (Ban et al., 2019), comme la protection côtière, par la conservation d'espèces fondatrices (Leenhardt et al., 2015) ou le stockage du carbone, par la protection des mangroves et des herbiers (Jacquemont et al., 2022).

1.4.2 La situation des AMP aujourd'hui

Pensées à leurs débuts comme le penchant des réserves terrestres, ou toutes les activités humaines sont exclues, elles suivent le changement de paradigme observés depuis l'émergence des concepts de la gestion écosystémique et du MSP, passant d'une conservation pour la nature elle-même (des réserves intégrales, dont l'humain est absent), à une conservation de la nature pour l'Homme (des réserves ou les activités humaines sont gérées de manière à assurer la durabilité des ressources, notamment pour faire face aux déclinés des pêcheries observés dans les années 1990), puis idéalement à une conservation de la nature avec l'humain, sans être centrée sur les ressources (Mace, 2014). En parallèle, la convention pour la diversité biologique (CBD), issue du sommet de Rio de 1992 et faisant suite au corpus scientifique grandissant sur les bénéfices apportés par les AMP décrits précédemment, fixe pour la première fois un objectif global de surface à atteindre : 10% en 2010 (Humphreys and Clark, 2019), reconduit en 2020 mais en spécifiant que les AMP doivent être gérées de manières efficaces et équitables, qu'elles soient écologiquement représentatives, et connectées entre elles (CBD, 2010). L'intégration des

différents usages humains, facilitant l'acceptation des AMP par les différents usagers, et ces objectifs de surface à atteindre par les États ont entraîné une augmentation considérable de la couverture globale en AMP, de 0.1% en 1980 (Lubchenco and Grorud-Colvert, 2015) à plus de 8% aujourd'hui (<https://mpatlas.org>, consulté le 20 juillet 2023). Cette course aux 10% a entraîné la création de grandes AMP qui ne régulent peu ou pas les activités humaines en leur sein, et qui ne sont peu ou pas représentatives écologiquement (Agardy et al., 2016; Visconti et al., 2019). De plus, afin de répondre au changement de paradigme dans la conservation des écosystèmes et à la demande croissante en espace et en ressources (l'accélération bleue), les AMP présentent une forte hétérogénéité dans la réglementations des activités dans leur périmètre, souvent découpé en de multiples zones avec chacune leur régulation propres, allant de l'exclusion quasi-totale des activités humaines à aucune restriction des activités (Watson et al., 2014).

De ce fait, toutes les AMP ne se valent pas, et les bénéfiques listés précédemment dépendent de nombreux facteurs, comme la taille et l'âge des AMP (Claudet et al., 2008; Edgar et al., 2014), et d'autres liés à la gouvernance et aux moyens humains et financiers, comme la définition d'objectifs clairs et le niveau d'application des réglementations pour leurs mises en œuvre (di Franco et al., 2016; Edgar et al., 2014; Gill et al., 2017), l'engagement des usagers dans leur création et dans la prise de décision pour leur gestion (di Franco et al., 2016; Gill et al., 2017), ou la capacité technique et financière pour assurer leur gestion et leur suivi (Gill et al., 2017). Mais un prérequis fondamental pour que les AMP apportent les bénéfices attendus est leur capacité à réguler les activités humaines par des réglementations efficaces permettant de diminuer l'impact cumulé en leur sein (Halpern et al. 2010; Edgar et al. 2014; Gill et al. 2017; Zupan et al. 2018b).

Comme indiqué précédemment, 8.2% de l'océan est actuellement protégé par des AMP, mais seulement 2.9% régulent ou interdisent les activités de pêche (<https://mpatlas.org>, consulté le 20 juillet 2023). Ces AMP, dénommées dans la littérature *no-take zones*, ou *fully protected areas*, sont celles qui offrent le plus de bénéfices socio-écologiques (Lester et al., 2009; Sala and Giakoumi, 2018), comparées aux AMP partiellement protégées, qui offrent moins (Giakoumi et al., 2017; Sciberras et al., 2013; Zupan et al., 2018b) voire aucun bénéfices socio-écologiques (Turnbull et al., 2021). Afin de pouvoir rendre compte de la capacité des AMP à délivrer ces bénéfices via des réglementations sur les activités extractives et non-extractives, un nouveau système de classification a été proposé, pour pallier à celui jugé défaillant de l'UICN (Dudley, 2008) portant sur les objectifs d'une AMP, qui sont souvent peu définis, et qui s'appliquent mal aux AMP multizones et multi-usages (Horta e Costa et al., 2016). Cette classification s'appuie sur l'identification des activités de pêche, d'aquaculture, d'exploitation et d'extraction minières, ou touristiques, autorisées ou non au sein de chacune des zones qui se voient attribuer un score (Figure 8.1 dans *Supplementary materials for chapter 3 and 4*). Les scores par zones sont ensuite combinés en fonction de la surface de la zone par rapport à la surface totale de l'AMP, afin d'obtenir le score total de l'AMP. Les AMP sont ainsi classées par leur capacité à réduire ou supprimer les impacts dus aux activités humaines en leur sein, liée à leur potentiel à délivrer des bénéfices écologiques (Zupan et al., 2018b). L'identification des activités humaines se déroulant au sein ou à proximité des AMP, et la quantification des impacts dus à ces activités à l'échelle très locale est crucial pour déterminer la capacité des AMP à atteindre leurs objectifs de conservation, mais aussi pour informer leur processus de création, les prises de décisions pour

orienter leur gestion, et la priorisation de zones à renforcer (Halpern et al., 2008a; Zupan et al., 2018a).

1.5 Objectifs et présentation des travaux de thèse

La gestion des écosystèmes marins et de leur durabilité est à un tournant : la transition d'une gestion sectorielle vers une gestion écosystémique fait consensus, mais de nombreux freins ralentissent son opérationnalisation. Les objectifs de conservation, mis en balance avec des objectifs économiques et sociaux, ne sont pas atteints. Des moyens de rationaliser les principes de la gestion écosystémique et de mesurer les progrès vers la conservation des écosystèmes au sein de la planification spatiale maritime sont requis. Puisque la gestion basée sur l'écosystème concerne avant tout la gestion des activités humaines, vecteurs de pressions mettant en péril la capacité de l'écosystème à délivrer des biens et services, l'évaluation de l'impact cumulé des activités humaines est plébiscitée dans la littérature scientifique comme prérequis à l'EBM et au MSP. Par ailleurs, les aires marines protégées, qui s'intègrent dans le cadre de la planification de l'espace marin comme outil préférentiel pour la conservation, ne semble pas renverser le déclin de la biodiversité, et ce malgré la course aux AMP ayant mené à une rapide augmentation de leur surface lors des dernières décennies. Les indicateurs utilisés pour mesurer le progrès vers une protection des océans ne reflètent pas le niveau de protection réel actuel, et des nuances sont à préciser quant au rôle des AMP dans la conservation des écosystèmes marins

Au travers de différents cas d'étude, les travaux de cette thèse ont pour objectifs l'opérationnalisation des concepts de la gestion écosystémique au sein de la planification spatiale maritime, en se concentrant sur les objectifs de conservation. L'évaluation de l'impact cumulé est placée au cœur du MSP, par la démonstration de son applicabilité à un contexte locale dans un processus de planification spatiale maritime, comme outil de diagnostic et d'évaluation de la capacité d'un plan de gestion à effectivement réduire des pressions issues des activités humaines en mer. Le rôle des AMP dans cette diminution des pressions issues des activités humaines est ensuite étudié, au regard des critiques soulevées quant au manque de régimes de protection suffisamment efficaces pour effectivement conserver les écosystèmes marins, et par les conditions optimums pour une protection forte permettant l'atteinte des objectifs de conservation.

1.5.1 L'évaluation de l'impact cumulé au cœur du MSP

La première partie s'appuie sur Moorea, en Polynésie française, comme cas d'étude. Le chapitre 1 démontre l'opérationnalisation des principes de l'EBM par l'application du cadre d'évaluation de l'impact cumulé des activités humaines à une échelle locale et pertinente pour une gestion basée sur l'écosystème. Par l'utilisation des meilleures données disponibles afin de limiter les hypothèses liées à l'utilisation de données globales, inadaptées pour informer la gestion, et l'utilisation de modélisation dans un contexte de déficience en données, la cartographie à fine échelle de l'impact cumulé sur l'écosystème corallien de Moorea fournit un support indispensable à l'EBM, permettant d'identifier les activités responsables d'effet délétères sur le milieu marin et de cibler des mesures de gestion qui permettrait de réduire leurs effets. Moorea est déjà soumis à un plan de gestion, qui ne répond pas aux objectifs de conservation souhaités et rencontre des difficultés d'acceptation au niveau locale. Un processus de révision du plan de gestion a été lancé, en consultation avec la population, et un nouveau zonage a été proposé. Le

chapitre 2 propose d'évaluer les deux plans de gestion de l'espace marin par le prisme de l'impact cumulé, en quantifiant la contribution de chacun des deux scénarios vers une éventuelle réduction de l'impact cumulé engendrée par leurs différentes réglementations.

1.5.2 La capacité des AMP à répondre aux objectifs de conservation

La réduction des pressions est centrale pour les objectifs de conservation, et les aires marines protégées ont démontré leur capacité à répondre à cet enjeu. Cependant, toutes les AMP ne se valent pas, et la question de protection forte au sein de la planification spatiale maritime est aujourd'hui posée, au regard de l'état actuel de l'environnement marin et de l'hétérogénéité des AMP. La deuxième partie des travaux de cette thèse porte sur le rôle des AMP dans la conservation des écosystèmes marins, en Méditerranée et dans les eaux françaises, en métropole et outre-mer. Le chapitre 3 porte sur l'analyse des niveaux de protection des 1062 AMP de Mer Méditerranée, classées selon les régulations autorisées en leur sein, et compare les progrès des pays méditerranéens vers l'objectif 11 d'Aichi. Le chapitre 4 applique cette classification aux AMP dans la ZEE française, la deuxième plus vaste du monde après celle des États-Unis. Suite à l'annonce du président Emmanuel Macron d'atteindre les 30% de protection, avec 10% en protection forte, d'ici à 2030, cette étude s'inscrit dans l'actualité et apporte des informations au débat sur la nécessité de renforcer le niveau de protection des AMP françaises. Enfin, le chapitre 5 explore les facteurs qui expliquent la présence des plus hauts niveaux de protection (ceux pour lesquels des bénéfices socio-écologiques sont attendus) dans les AMP de Méditerranée, facteurs se rapportant aux pressions humaines, à la biodiversité, aux caractéristiques intrinsèques des AMP, et aux pays auxquelles elles appartiennent.

Cumulative impact assessments can show the benefits of integrating land-based management with marine spatial planning

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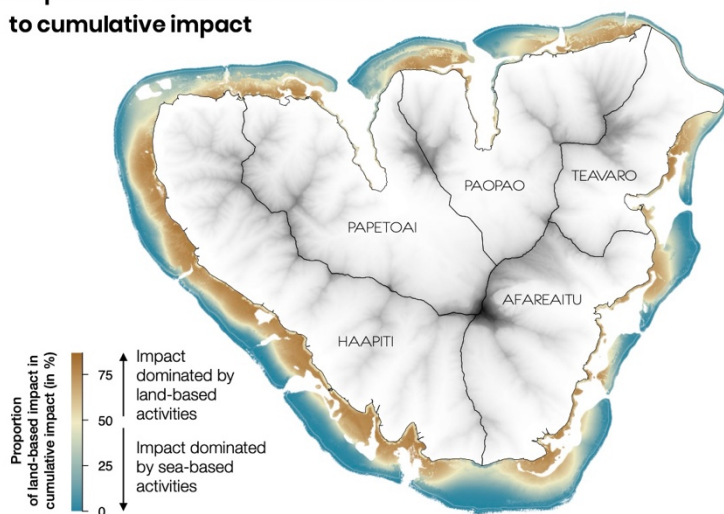
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Cumulative impact assessments can show the benefits of integrating land-based management with marine spatial planning

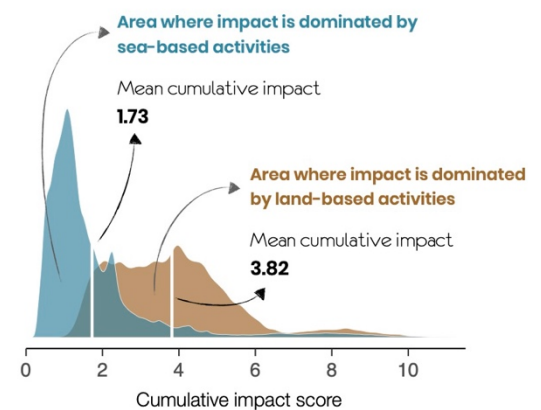
2.1 Abstract

Cumulative impact assessments can inform ecosystem-based management by mapping human pressures and assessing their intensity on ecosystem components. However, its use to inform local management is scarce, largely due to the need for fine-grained spatial data representing ecosystem threats that can assess impacts at a local scale. Here, we applied the cumulative impact assessment framework to Moorea's coral reef, French Polynesia to inform the ongoing revision of the island-wide marine spatial management plan. We combined high spatial resolution data on 11 local anthropogenic pressures and four ecological components with expert vulnerability assessments. Results revealed that the entire reef is impacted by at least four pressures: coral reef fisheries, agriculture, land use change and urbanization. These activities together contribute to 87% of the overall cumulative impact. Most importantly, land-based activities contribute to more than half (52%) of the overall impact. Other high-impact activities, such as reef-based tourism, remain very localized and contribute little to the overall human impact. These findings show that by focusing solely on reef-based activities, the current management plan misses critical sources of impact. Not considering land-based activities in the management may lead to decisions that could fail to significantly lower cumulative human impact on the reef. This study demonstrates how operationalizing the cumulative human impact framework at a local scale can help managers identify key leverage points likely to yield improved ecological outcomes.

Map of land-based activities contribution to cumulative impact



Distribution of cumulative impact in area dominated by land-based or sea-based activities



Graphical Abstract

2.2 Introduction

Biodiversity in coastal zones provides highly valuable ecosystems and at times vital services to communities worldwide (Barbier et al., 2011; de Groot et al., 2012). With about 40% of the world's population living near the coast (CIESIN, 2012), human activities on both land and water exacerbate pressures on coastal ecosystems (He and Silliman, 2019). Overfishing, pollution, and other human activities have already seriously depleted highly productive coastal ecosystems such as salt marshes (Deegan et al., 2012), seagrass meadows (Waycott et al., 2009), mangroves (Polidoro et al., 2010) and coral reefs (Pandolfi et al., 2003), to a point that goods, services and cultural benefits provided in these areas are critically endangered (Barbier, 2017; Cardinale et al., 2012). Moreover, human activities and their associated pressures are often overlapping in space and time, creating hotspots of cumulative impact with no trends of decrease (Halpern et al., 2019).

In this context, specific interventions for conserving biodiversity are a cornerstone of ecosystem services protection (Duarte et al., 2020). Integrating a wide range of environmental, ecological and human factors, Ecosystem-Based Management (EBM) has emerged as a holistic and consensual approach to secure ecosystems' capacity to provide ecosystem services against local as well as distant human pressures (Enright and Boteler, 2020; Long et al., 2015; McLeod et al., 2005). Local management is key to both reducing exposure to local threats and improving resilience to global change (Harvey et al., 2018; Lefcheck et al., 2018; MacNeil et al., 2019; O'Leary et al., 2017; Roberts et al., 2017; Shaver et al., 2018; I. D. Williams et al., 2019; Wu et al., 2017). Local management is also essential to define tangible targets and empower local stakeholders.

In an EBM perspective, cumulative impact assessments can help practitioners set management priorities by weighing ecosystems' vulnerability against exposure to different human activities in a spatially explicit way (Kappel et al., 2012). Cumulative impact assessments remain one of the few comprehensive tools that allow quantifying how humans are affecting natural systems, and how actions targeting specific stressors may be expected to alter the overall impacts (Halpern et al., 2008a). While traditionally used in the marine environment to inform management and policy decisions on the global (Halpern et al., 2019, 2015, 2008b) and regional scales (Ban et al., 2010; Bevilacqua et al., 2018; Clarke Murray et al., 2015; Gissi et al., 2017; Halpern et al., 2009b; Korpinen et al., 2012; Micheli et al., 2013), key challenges remain to apply the cumulative impact framework on the local scale. These include the availability of fine-scale resolution data on human activities and associated pressures, especially in a land-sea continuum perspective (Clark et al., 2016), the outputs' representation in a comprehensive way for local planners and managers (Lombard et al., 2019), and the integration of the inherent uncertainty into decision-support (Stock et al., 2018).

Coral reefs are among the most threatened marine ecosystems in the world and are affected by a broad range of activities and pressures (Halpern et al., 2019; Harborne et al., 2017; G. J. Williams et al., 2019), jeopardizing the provision of key ecosystem services such as coastal protection, food provisioning and cultural heritage (Cinner, 2014; Hicks and Cinner, 2014). While mitigating distal threats like climate change require global action (Morrison et al. 2020), local management of activities such as fishing, shipping, tourism and coastal development can help reduce impacts

and support resilience to global stressors (Anthony et al., 2015). Here, we use and extend the cumulative impact model framework to characterize and explore the impact of human activities on scales relevant to most community-based decisions and management interventions, using the coral reef ecosystem of Moorea, in French Polynesia, as case-study. Specifically, we operationalized the cumulative impact model framework to produce a map of cumulative impact that would enable planners, resource managers, and communities to examine spatial variation in human pressures and explore different sources of impact at a fine spatial scale to inform local decision-making.

2.3 Materials and Methods

2.3.1 Study site

Located in French Polynesia, Moorea is a volcanic island surrounded by a 49 km² semi-enclosed coral reef, shallow (less than 30 m deep) and relatively narrow (maximum 1500 m long) lagoon, connected to the Pacific Ocean by 12 passes (Figure 2.1). While the fore reef is directly exposed to oceanic conditions, the lagoon is more sheltered and includes a shallow barrier reef (less than three meters deep), a fringing reef adjacent to land, and a channel (> 10 m deep) that separates the two. In the three past decades, Moorea's fore reefs has been exposed to numerous acute disturbances including two cyclones, one crown-of-thorns starfish outbreak, and five bleaching events (Adjeroud et al., 2018; Vercelloni et al., 2019). Until now, coral cover always managed to reach the pre-disturbance levels following many of these events. However, coral assemblages have changed significantly (Adjeroud et al., 2018; Pratchett et al., 2011; Trapon et al., 2011), with cascading effects on fish assemblages (Lamy et al., 2016) leading to a deficit in biodiversity and ecosystem functioning, a phenomenon called "recovery debt" (Dubois et al., 2019).

Moorea's population increased from 7,059 people in 1983 to 17,463 in 2017, becoming the second most populated island of French Polynesia after Tahiti (ISPF - INSEE, 2017). Human population and activities are mostly concentrated in a narrow land strip adjacent to the coast, due to the steep island topography. In the 90's, the "Progress Pact", an economic plan promoting local economic self-sufficiency based on tourism, agriculture, fishing and pearl farming, led to a shift from a subsistence economy (dominated by small-scale fisheries and farming) to a cash economy (relying mostly on tourism and services sector) (Leenhardt et al., 2016; Walker et al., 2014). Agriculture, which is now largely based on large pineapple plantations, has become more intensive and intended to the export market. As the third most visited island in French Polynesia after Tahiti and Bora-Bora (ISPF, 2019), tourism is an important economic sector of Moorea. It is supported by international resorts construction and relies largely on reef-based activities. However, while Moorea inhabitants exhibit a lower dependency to marine resources for food provisioning compared to other Pacific Islands (Rassweiler et al., 2020), between 35% and 50% of its population are still engaged in a fishing-related activity and feel strongly connected to the reef (Leenhardt et al., 2016; Thiault et al., 2017).

Following tourism development around Moorea, a marine spatial plan (*Plan de Gestion de l'Espace Maritime*, PGEM) - the first in French Polynesia - was implemented in 2005 following more than 10 years of consultation processes (Hunter et al., 2018). The PGEM encompasses the whole lagoon and the fore reef down to 70 m deep, and specifically regulates fishing activities

through 8 marine protected areas (MPA) covering 17% of its waters, of which 5 are fully protected and 3 moderately protected (Thiault et al., 2019). Other regulations include delineation of areas for recreational activities including swimming, ray and shark provisioning and boating, while coral extraction and embankments are prohibited island-wide. However, no explicit connections were made between this marine spatial plan and the pre-existing land planning framework (*Plan Général d'Aménagement*, PGA) during the planning process, and issues such as land-based pollution and sedimentation largely overlooked.

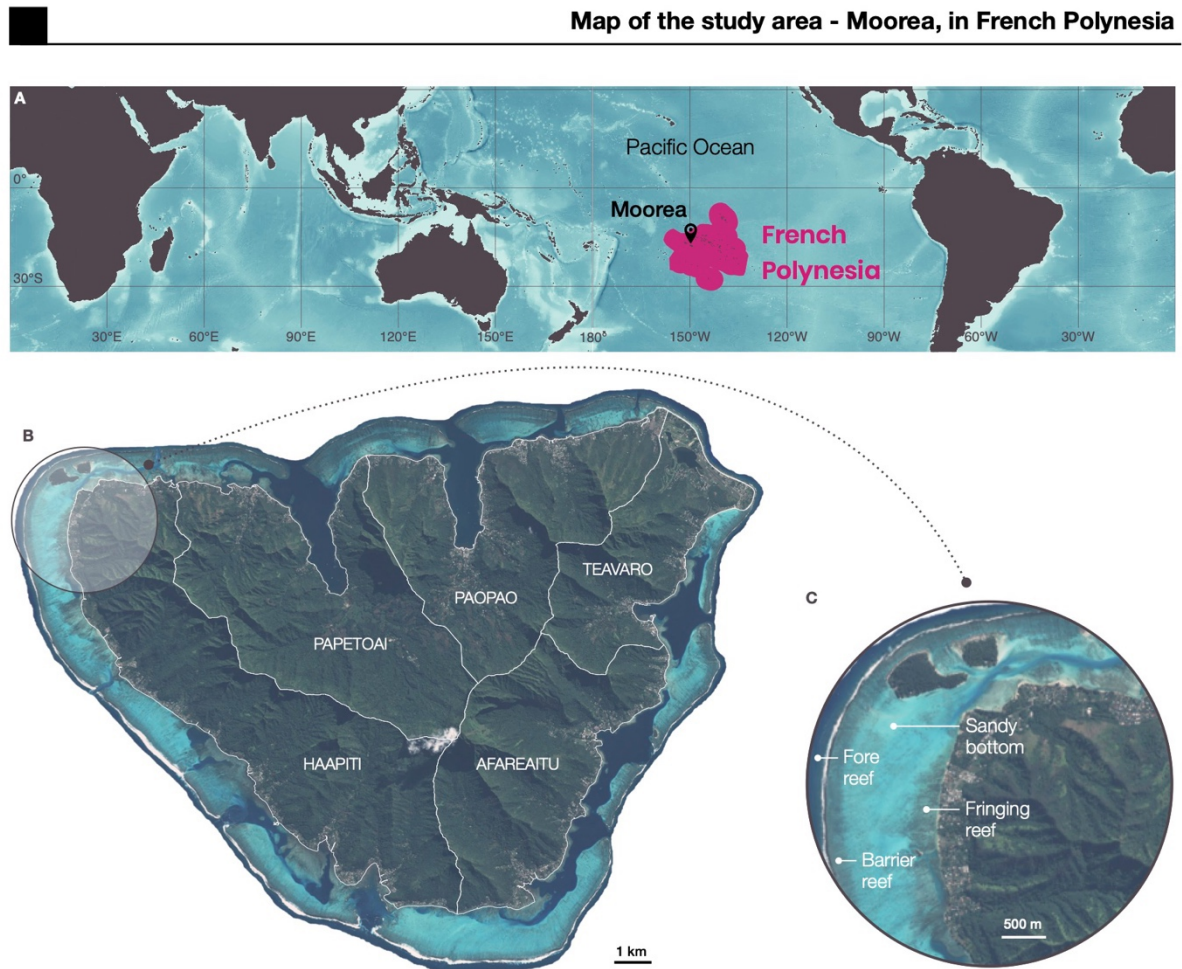


Figure 2.1. French Polynesia is located in the South Pacific (A).

Moorea is the second largest island after Tahiti (B). For the cumulative impact assessment we focus here on four marine habitats (C): fringing reef, barrier reef, fore reef and sandy bottom.

2.3.2 Cumulative impact assessment

2.3.2.1 Human activities and associated pressures

We reviewed the primary and grey literature and identified 11 human activities that can have a direct impact on Moorea's marine ecosystem (Table 2.1 in 2.7. Supplementary material for chapter 1). Activities were grouped into two broad categories: land-based activities and sea-based activities. For each activity, we used fine-scale data to model the spatial extent and

intensity of their associated pressures. The supplementary material for chapter 1 at the end of this chapter provides a detailed explanation of each pressures' models and a review of impacts.

We considered three land-based activities: agriculture, land conversion, and urbanization. Pressures from agriculture and land conversion were considered as point-source pollutions originating from the watersheds and associated river mouths, whereas pollution from urbanization was considered as originating from individual houses all along the coast. These activities result in different pressures on the adjacent coral reef ecosystem, such as eutrophication due to fertilizers and increase in sedimentation rates, and pollution from pesticides and sewage (Fabricius, 2005; Magris and Ban, 2019). Point-sources pressures were modelled using data on land occupation inside each watershed. Values indicating the intensity of the pressure were computed, associated to the corresponding river mouth, and extended spatially into the lagoon using a diffusion model. Sewage discharge from urbanization was based on houses density in the area (see supplementary materials 2.7 for more details on land-based activities impacts and modeling). To model the diffusion of pressures from agriculture and land conversion, we adapted a linked land-sea modeling framework designed for local management, using a cost-path surface model built from three factors known to affect diffusion: depth, distance from the shore and wind speed (Jade M. S. Delevaux et al., 2018).

Eight activities take place directly into the lagoon and along the fore reef. Six are related to tourism (i.e., boat traffic, mooring, anchoring, beach attendance, diving, wildlife provisioning), and the two remaining are coral reef fisheries and shoreline modification. Moorea's shoreline evolution is monitored since 1977 (Madi Moussa et al., 2019), allowing the modeling of the resulting pressure on near-shore habitats. Pressure coming from fishery in Moorea is acknowledged as very challenging to assess given the complexity of the artisanal coral-reef fishery and the absence of stock monitoring (Leenhardt et al., 2016). As a proxy for fishing pressure, we thus use a fishing effort model based on socioeconomic data (households' dependence on fishing) and participatory mapping of the fishermen's spatial preference for fishing grounds (Thiault et al., 2017). This model is the most recent attempt to describe the fishing pressure in Moorea in a spatially explicit way.

Moorea's tourism sector has rapidly grown in the last decades. It relies on a variety of activities such as diving, wildlife provisioning, and an increase in boat traffic and beaches attendance. The increase in boats and people use of the lagoon for touristic activities can be detrimental to coral reef ecosystem, through direct pressures (e.g, boat anchors, divers fins, trampling) causing physical damages to the reef, and through different types of pollution (e.g, noise production, release of pollutants, sewage) generated by the use of engines (Davenport and Davenport, 2006). Outside boat traffic and beach use, activities are restricted to specific places for different reasons: for example, wildlife provisioning is partially regulated by the PGEM, allowing the activity to only take place in six sites, even if one supplementary site has been observed. Similarly, mooring and the related anchoring are only allowed inside the lagoon at five specific sites. Diving tends to also take place at specific sites around Moorea, some of them being included in the PGEM, while others appearing on websites of diving operators, confirmed by face-to-face interviews.

To assess beach attendance, we combined data on household density along the coast, tourism infrastructures and presence of public beaches which allow to model the intensity of this pressure inside the lagoon, taking into account the distance to the shore and depth. Finally, our boat traffic intensity model was based on least-cost distance between boats' departure and arrival sites, which can be specific sites in the lagoon (wildlife provisioning sites, diving sites) or sites used for regular trips (from cruise ship parking to harbor, ferries to Tahiti, jet-ski tour, and dolphin and whale watching). The intensity of each of these six sub-activities was calculated based on the frequency of their trip. The supplementary materials 2.7 provides a detailed explanation of each pressure modeling.

2.3.2.2 *Habitats*

We considered four reef habitats, each characterized by a unique geomorphology, and specific benthic and fish assemblages: (1) the fringing reef, directly adjacent to the shore and less than two meters deep; (2) sandy bottoms, which include mainly the channel that separates the fringing and the barrier reefs, but also some sand banks and areas inside the lagoon; (3) the barrier reef, less than three meters deep and intersected by the passes, exhibiting similar coral cover compared to fringing reef but a greater specific richness; and (4) the fore reef, outside the lagoon, separated by the reef crest, from 0 to 70 m deep and directly influenced by the open ocean. It exhibits the greatest coral cover and species richness.

We derived a spatial layer of these habitats substrate map from very-high resolution satellite imagery (Collin and Hench, 2015). Cells for which substrate was dominated by sediments were classified as sandy bottom. We used the channel to discriminate spatially coral reef patches belonging to the fringing reef from those belonging to the barrier reef, while the reef crest discriminates the barrier reef from the fore reef ([Figure 2.7](#) in supplementary materials 2.7).

2.3.2.3 *Vulnerability coefficients*

Vulnerability coefficients, used to translate pressure intensities on habitats into impacts, were obtained using an expert judgement process. We interviewed 15 experts on French Polynesia coral reef ecosystems and management to discriminate how each activity impacts each habitats, based on five criteria: the spatial scale of the pressure, its frequency, its functional impact, the ecosystem resistance and the recovery time (Halpern et al., 2007). For example, the criteria about the scale of a pressure is evaluated against 7 categories (no occurrence of the pressure, occurrence in less than 1 kilometer, between 1 and 10 kilometers, between 10 and 100, between 100 and 1000, 1000 and 10000 and finally in more than 10 000 kilometers), and the certainty is evaluated against 4 categories (none, low, medium, high). Each category was then ranked (in this case from 0 to 6 for the scale, and from 0 to 4 for the certainty), and experts assigned a rank in each criteria, for each pressure, and on each ecological habitats along with the certainty rank. Mean values for these five criteria weighted by the certainty were then used to assign a single weighted vulnerability coefficient for each habitat facing each pressure ([Table 2.4](#) in supplementary materials 2.7).

To assess the cumulative impact of human activities on Moorea's lagoon and fore reef, we used an additive model combining the location and intensities of human pressures on different habitats

(Halpern et al., 2008b). We weighted these pressures based on vulnerability coefficients derived from experts' assessment as:

$$I_c = \sum_{i=1}^n \sum_{j=1}^m P_i E_j \mu_{i,j} \quad (1)$$

where I_c is the cumulative impact score for a given pixel c , P_i is the log-transformed and normalized value of anthropogenic pressures ($n = 11$), E_j is the presence or absence of habitat ($m = 4$), and $\mu_{i,j}$ is the vulnerability coefficients of habitat j to pressure i (Figure 2.2).

Uncertainty in the model's outputs was estimated using a method adapted from (Stock and Micheli, 2016). In this study, the authors identified nine model assumptions or data quality issues (referred to as "factor" hereafter) that could generate uncertainty, five of which were potentially present in our model: (1) missing pressure data, (2) type of transformation for pressure layers, (3) errors in vulnerability coefficients, (4) nonlinear responses, and (5) multiple pressure effects.

We quantified our model's uncertainty by bootstrapping these five factors using Monte-Carlo simulations (1000 runs). In each simulation run, factors were set to values randomly assigned from their ranges and cumulative impact was calculated. We then recorded how often pixels was in the first (low-impact area) or the last quartiles (high-impact area) of cumulative impact distribution to compare the robustness of low and high impacted areas with our original model. We assumed a low- or high-impact pixel is robust when it is confirmed by more than 75% of simulations (i.e. a pixel belonging to the first or last quartiles of original cumulative impact and belongs as well in the first or last quartiles of cumulative impact score in more than 75% of simulations).

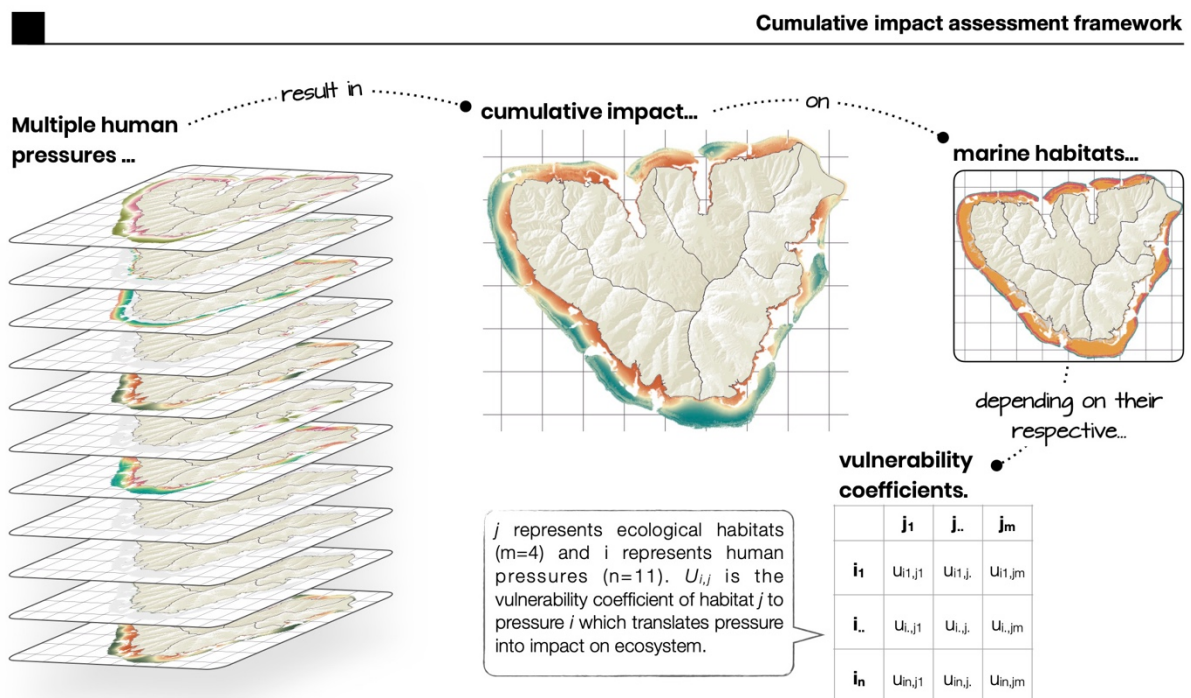


Figure 2.2. Schematic representation of the cumulative impact assessment framework used.

All analyses were conducted in QGIS software v.3.4.8 (QGIS Development Team, 2020) and R software v.4.0.3 (R Core Team, 2020) using the raster (Hijmans, 2022), sf (Pebesma, 2018) gdistance (van Etten, 2017) and SpatialEco (Evans et al., 2020) packages for spatial analyses, as well as ggplot2 (Wickham, 2016) and tmap (Tennekes, 2018) for figures and maps production.

2.4 Results

2.4.1 Cumulative impact assessment overview

The entire system studied is impacted by human activities, although the reef system is heterogeneously affected around Moorea (mean = 2.6; [Figure 2.3 A-B](#)). Impacts are generally

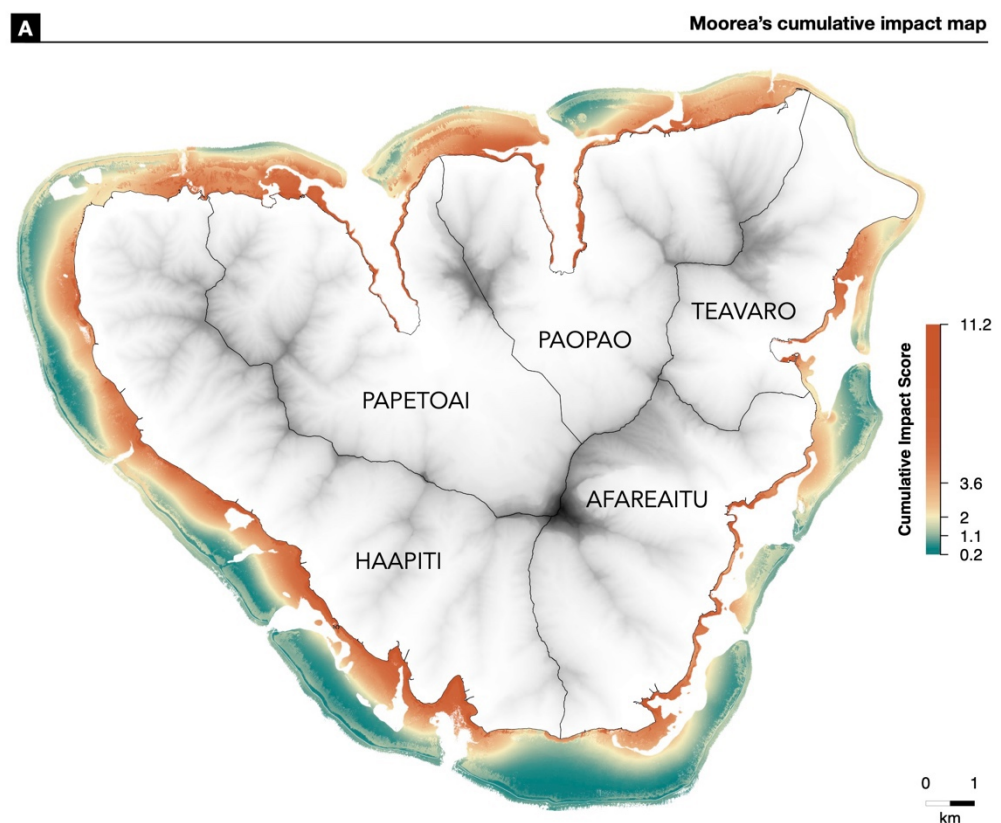
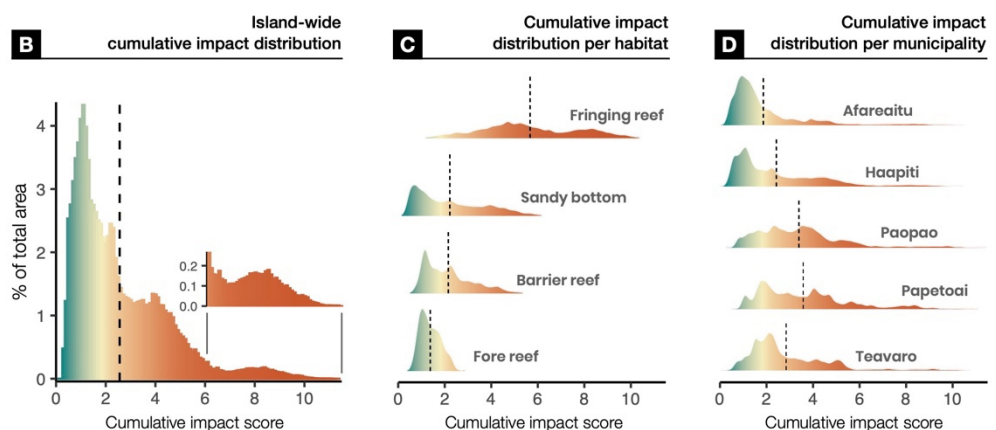


Figure 2.3.
Cumulative impact in Moorea.

Map of the cumulative impact on marine habitats around Moorea (A) and overall cumulative impact score (B) distributed per habitat (C) and municipality (D). The dashed vertical lines in B, C and D indicate the average cumulative impact score.



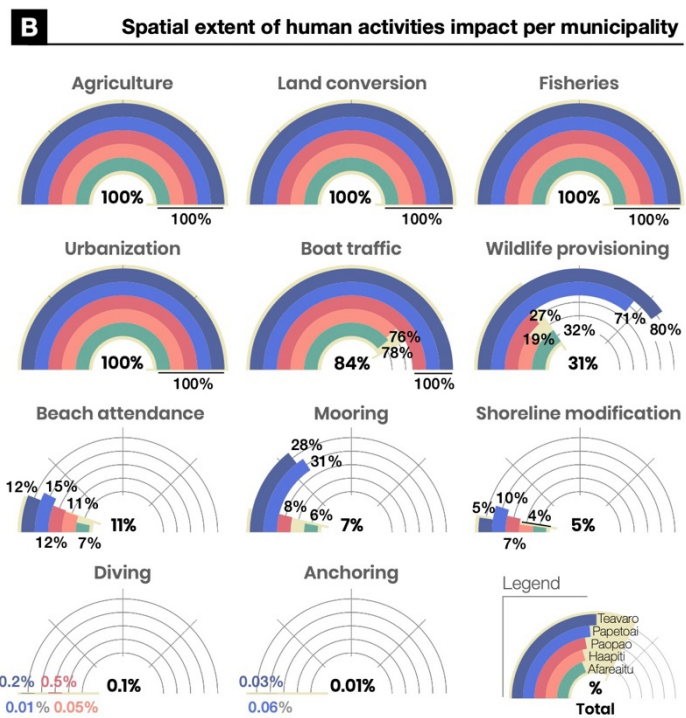
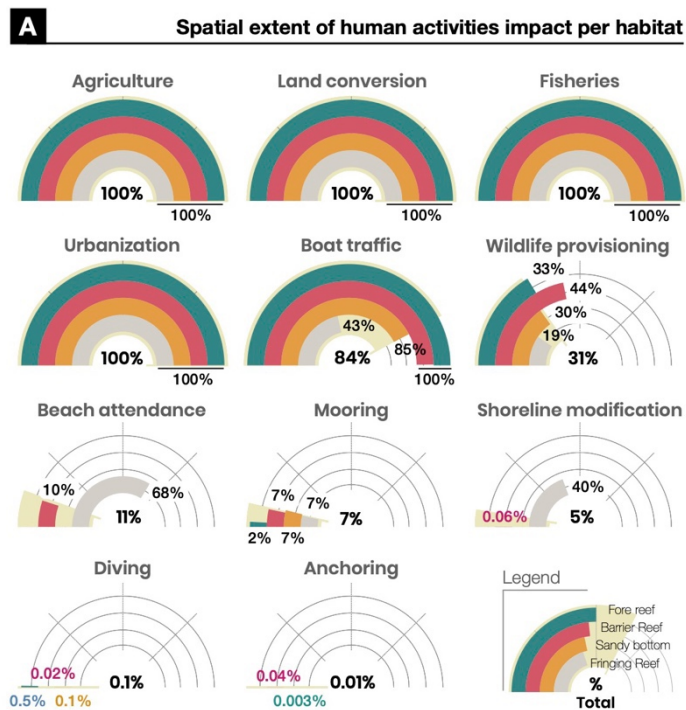


Figure 2.4. Spatial extent of human activities in Moorea.

Per habitat (A) and per municipality (B). Percentages in the middle of semi-circles indicate the spatial extent of each activities in the entire study area (related to the brown colored large bar). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

larger close to shore, with fringing reefs displaying the highest levels of impact overall (mean = 5.7). The barrier reef and sandy bottoms both exhibit lower cumulative impact scores, while the fore reef is the least impacted (Figure 2.3C). Cumulative human impact is particularly high off the municipalities of Papetoai and Paopao (mean = 3.6 and 3.4, respectively), while it remains close to average for Teavaro and Haapiti. Afareaitu has the lowest impact score with a mean of 1.9 (Figure 2.3D).

2.4.2 Spatial extent of human activities

Agriculture, fisheries, land conversion and urbanization impact the entire reef ecosystem across all five municipalities (Figure 2.4). Two other activities impacting a large portion of the reef are boat traffic (84% of the study area), and wildlife provisioning (31%), located mainly in the north municipalities of Papetoai and Teavaro, and on the fore reef. The five other activities (i.e., beaches attendance, mooring, shoreline modification, diving and anchoring) are distributed around the island in very specific locations, hence impacting a much smaller proportion of the different habitats (Figure 2.4A). Papetoai and Teavaro are the most concerned by impacts extent, especially for wildlife provisioning and mooring (Figure 2.4B). Boat traffic and wildlife provisioning have a higher impact on offshore habitats. However, the fringing reef is impacted by all activities, except for diving and anchoring.

2.4.3 Human activities contributions to cumulative impact

Fisheries are consistently the largest contributor to the overall cumulative human impact and, when added with land-conversion and urbanization, contribute to more than 80% of the overall cumulative impact (Figure 2.5A). There is only one habitat where fisheries are outpaced precisely by land conversion and urbanization, the fringing reef (Figure 2.5B). However, land-based activities contribute together to more than half of the cumulative impact around Moorea. In the barrier reef and the fore reef, impact is largely driven by fisheries and boat traffic. While heterogeneity between contributors is observed for habitats, municipalities have the same three activities as those observed at the island scale (Figure 2.5C). The fourth most impacting activity is agriculture in Afareaitu and Haapiti, while it is boat traffic in Paopao, Papetoai and Teavaro.

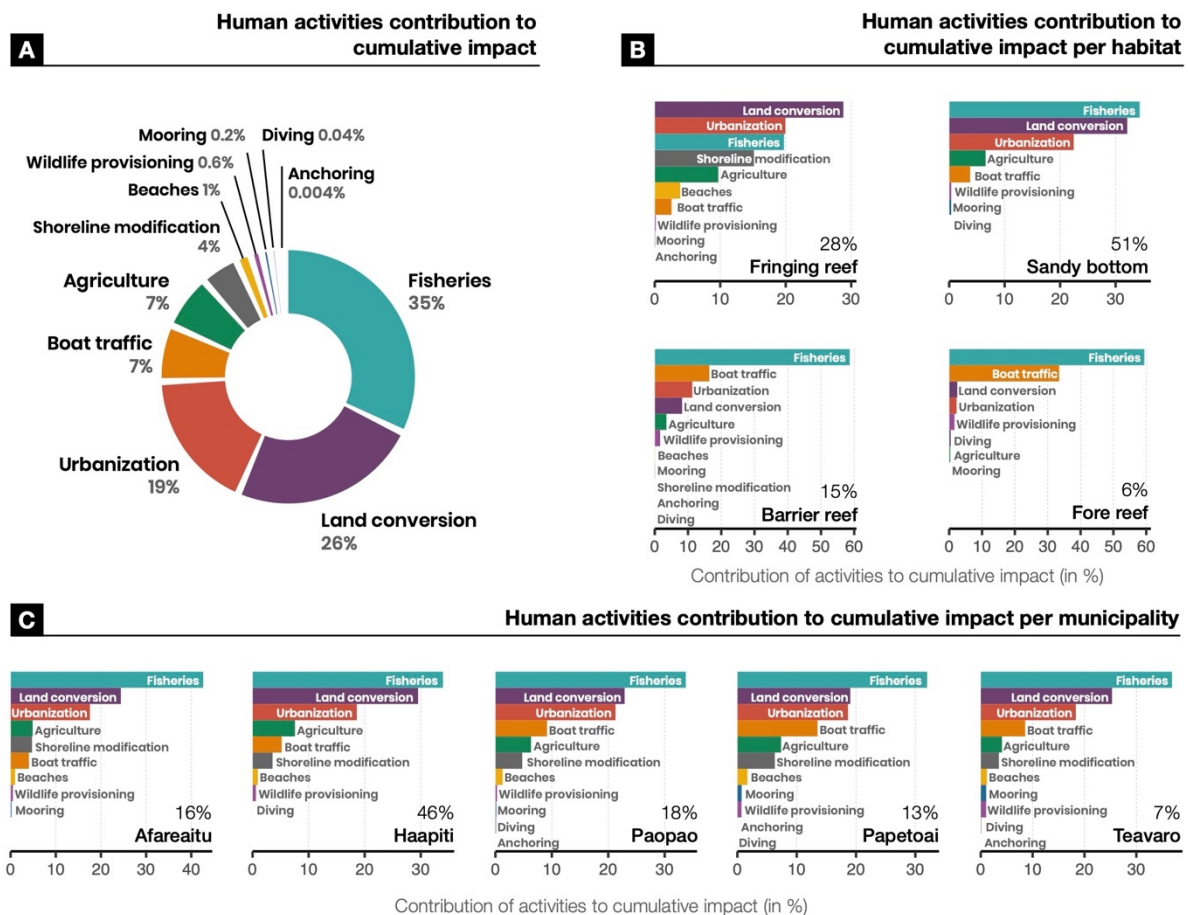


Figure 2.5. Contribution of individual human activities to cumulative impact in Moorea.

At the whole island scale (A), per habitat (B) and per municipality (C). Percentages above habitats and municipalities represent the proportion of overall cumulative impact in those habitats or municipalities.

2.4.4 Uncertainty analysis

Comparison between low- and high-impact area in the 1000 Monte-Carlo simulations (*i.e.* pixels being in first or last quartiles of cumulative impact score in more than 75% of simulations) and our original model indicates that 35% of original low-impact area and 23% of high-impact area are robust to factor variations (Figure 2.6 A-B). High-impacted habitat fringing reef (43% of this habitat is highly impacted) exhibits 45% of robust high-impact area, while other habitats high-

impact area are poorly robust. Low-impact areas, mostly on sandy bottoms, are robust at 15% (Figure 2.6C). Municipalities located in the north of Moorea display between 7% and 12% of robust high-impact areas, while municipalities in the south (mainly Haapiti and Afareaitu) host robust low-impact areas (both 12%). Results are similar to those observed in global studies on cumulative impact uncertainty (Stock et al., 2018; Stock and Micheli, 2016).

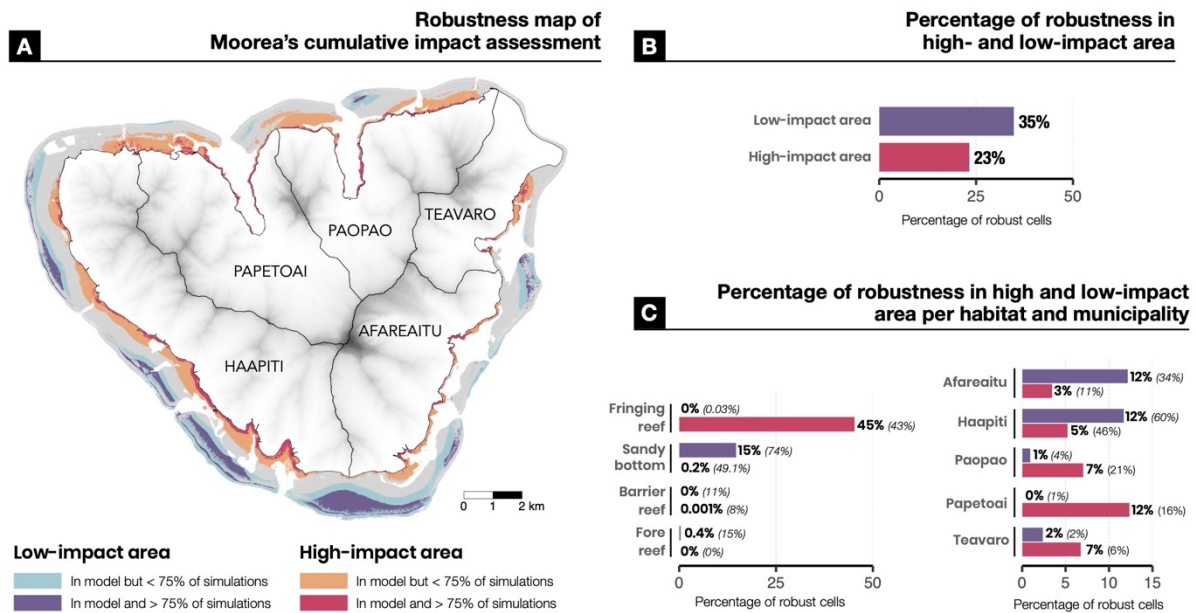


Figure 2.6. Robustness of Moorea's cumulative impact assessment.

Robustness mapped around Moorea (A), robustness expressed as proportion of robust pixels in low- and high-impact areas (B), and robustness per habitat and municipality (C). Percentages inside brackets in C indicate the proportion of original low- or high-impact area per habitat or municipality.

2.5 Discussion

Our cumulative impact assessment provides, to our knowledge, the most comprehensive fine-scale mapping of cumulative human impact in a marine environment to date. We found that over half of the overall cumulative impact on Moorea's reef is due to land-based activities: agriculture, land conversion and urbanization. In addition to having large impacts in intensity, the footprint of land-based activities is widely distributed throughout the reef and across all municipalities. These findings underline the critical role that integrated "ridge-to-reef management" should play (Álvarez-Romero et al., 2011; Brown et al., 2019; Carlson et al., 2019; Wenger et al., 2020). However, the current marine spatial plan does not address land-based pressures, focusing instead exclusively on regulating marine activities, hence failing to protect its waters from its key pressures. Our results highlight the large benefits that can be derived from focusing management attention towards more transformative, deeper leverage points (Abson et al., 2017; Fischer and Riechers, 2019) that might be more difficult to attain but potentially yielding more ecological benefits. These may require, among others, a greater collaboration between relevant agencies to better integrate both terrestrial and marine systems and threats, and effectively mitigate land-based pressures on the reef (Thiault et al., 2020). In addition, putting land-sea interactions at the core of the management plan is also critical to avoid the current management mismatches

between the land-sea continuum as perceived by traditional Polynesian management styles (Leenhardt et al., 2017). Moreover, mitigation of land-based pressures is likely to have the greatest positive impact on the intrinsic ecological resilience of the reef, especially inside the lagoon, and avoid a focus on restricting fishing activities, which can be detrimental when associated with high social vulnerability (Thiault et al., 2020).

The second largest contributor to the overall cumulative impact is fishing. It is the activity that alone has the most profound and widespread impact, affecting the entire reef and contributing to more than a third to the overall impact. These findings may appear surprising given the low impact on habitats the highly selective gears (mostly spearguns, lines and nets) used in Moorea has (Rassweiler et al., 2020). However, this may reflect the ubiquitous nature of fishing within the community (high livelihood dependency on fishing leading to high fishing capacity) (Leenhardt et al., 2016; Thiault et al., 2017), as showed by the fact that it ranked first in all municipalities. In addition, the relatively small lagoon width, the twelve passes and the convenience to fish on the outer slope under good weather conditions allow fishermen to access all reef components, although distance to the shore remains a key driver of fishing pressure (Thiault et al., 2017). Currently, the marine spatial plan around Moorea combines spatial (e.g., marine protected areas) and gear-based restrictions. While the existing Moorea MPA network appears effective, ecological benefits are weaker than expected (Thiault et al., 2019a). Weak compliance and enforcement have previously been put forward to explain their lower effectiveness, but the above-mentioned widespread impacts of land-based activities may also play a role.

Tourism-based activities, which were a major concern for stakeholders during and after the implementation of the marine spatial plan (Gaspar and Bambridge, 2008; Walker, 2001), were found to contribute very little to the overall impact. Nevertheless, when such activities do occur, the mean impact is high, highlighting that managing tourism activities remains key to reduce human impact on the reef. This is even more critical given that jet-skiing and other new activities such as flyboard are becoming increasingly popular. Boat traffic, which here reflects the daily operations of tourism operators, was the only tourism-related activity whose impact remained relatively widespread. Improving the management of this activity may require strategically channeling boat traffic away from vulnerable habitats (i.e. barrier reef and fore reef) to lower its contribution to cumulative impact or other sea-based activities (i.e., fishing) to reduce conflicts between stakeholders (Noble et al., 2019).

Our study advances the application of cumulative impact assessments to local contexts, based on empirical spatial data. By design, the cumulative impact model focuses on simple interactions between pressures and ecological components. However, such interactions are often complex, and relatively simple representations of relationships and feedbacks between ecological components and threats may obscure information that is important to understand system dynamics, key drivers, and opportunities for effective intervention (Halpern and Fujita, 2013). Future work building on new data and models could expand the set of activities and incorporate more complexity. This would further improve the ability of stakeholders and decision-makers to identify sources of impact on ecosystems and discover a wider range of options for managing coral reefs for the benefit of the people who depend on them.

2.6 Conclusion

Cumulative human impact assessments offer an approach for planners and decision makers to identify which areas are the most and least impacted, which activities are responsible for these impacts, and can thus help target and prioritize management actions. More generally, it can underpin improved and balanced consideration of the human dimension in place-based environmental management initiatives. Because of the standardized nature of the cumulative impact model, this approach enables direct and fine-scale comparisons that allow decision makers and stakeholders to measure and map change through time (Halpern et al., 2015). Such information may provide additional indications on the nature of changes (key sources of impact affected), and its direction (positive or negative) in specific locations. This approach is likely to be applicable in a variety of other places where human activity represents both an important threat to the ecosystem and an invaluable source of benefits for local communities.

2.7 Supplementary materials for chapter 1

2.7.1 General approach

This study focuses on coastal waters of Moorea, in French Polynesia, including a semi-enclosed lagoon and the surrounding fore reef for a total area of 49 km². We compiled local data on 11 human land-based and sea-based activities and associated pressures ([Table 2.1](#), [2.2,2.3](#)) to assess cumulative human impact on 4 distinct reef habitats (fringing reef, sandy bottoms, barrier reef, fore reef) at the fine resolution of 5 m ([Figure 2.7](#)). Each pressure layers is log[X+1]-transformed, rescaled between 0 and 1, and translated into impacts using vulnerability coefficients of habitats to individual pressure ([Table 2.4](#)) using an additive model (Halpern et al., 2008b).

2.7.2 Modelling human pressures

2.7.2.1 Land-based activities

Agriculture

Impacts review: Agriculture releases nutrients in coastal waters that can lead to eutrophication and a bioaccumulation of pesticides in the food web (Brown et al., 2017; Lu et al., 2018; Wenger et al., 2015). Eutrophication affects coral reef ecosystems by reducing corals' immunity to pathogens, accelerating bioerosion, and reducing their reproduction capacity, resulting in an overall loss of biodiversity and coral cover in exposed areas (Duprey et al., 2016). Traces of herbicides have been found at different levels of the food web in French Polynesia' reefs, with negative impacts on fish recruitment success (Besson et al., 2017) and coral fertilization, settlement and zooxanthellae's functions (Richmond et al., 2018; Salvat et al., 2016).

Modeling: To model the reef's exposure to agriculture, we first calculated an index of agricultural intensity at the watershed level, combining land use data with local estimates of fertilizers and pesticides inputs ([Table 2.3](#)). To do so, we first attributed a weight to each type of crops (w_C), depending of the total fertilizers and pesticides inputs, as:

$$w_C = \frac{It_C}{It_{min}} \quad (2)$$

Where It_C is the total input for agriculture category and It_{min} is the minimum of input among all crop types (i.e. coconuts plantations in our case). Then, the index of agricultural intensity I_A is obtained as the weighted score at the watershed level:

$$I_A = \sum_{i=1}^n S_{CA} \times w_C \quad (3)$$

where S_{CA} is the cultivated surface of crop type in the watershed A. Finally, values were reported at their corresponding stream mouth and dispersed onto the lagoon using a diffusion simulation modelling (see section [Diffusion simulation](#) below).

Land conversion

Impacts review: Given the geomorphology, the pristine land cover (tropical rainforests) and the climatic conditions of Moorea, land conversion generates substantial sediment run-offs in coastal waters and affect dramatically the coral reef ecosystem by increasing water turbidity and sedimentation with detrimental consequences on benthic species (Brown et al., 2017; Lu et al., 2018; Magris and Ban, 2019; Wenger et al., 2015). Sedimentation increases affect reproduction, larval behavior and recruitment success of corals (Richmond et al., 2018), and have been linked to decreases in coral cover and diversity (Déath and Fabricius, 2010; Magris and Ban, 2019). Such increases also resulted in changes in reef fishes ecology and behavior, especially herbivory fishes (Wenger et al., 2015).

Modeling: We used the Revised Universal Soil Loss Equation (Renard et al., 2012) to estimate intensity of potential sedimentation increase for each watershed basins and its associated stream mouth (see Dumas, 2015 for an application in Tahiti). This approach provides an estimation of average annual soil loss rate using rainfall, soil, topography, land cover and management practices as inputs. We performed two RUSLE models: one using the most up-to-date data on land use (“true use model”), and a second that assumed no human uses (“pristine model”) where land occupied by human uses (crops and urban area) are replaced by forest. The process resulted in soil loss maps, which we used to estimate the total loss of soil per watersheds. For each watershed, we then subtracted values of the “true use model” from the “pristine model” to obtain an estimate of excess sedimentation caused by human activities. These values were reported at their corresponding stream mouth and dispersed onto the lagoon using the diffusion simulation modeling.

Urbanization

Impacts review: Land-based sewages, which contain a mix of inorganic nutrients, pathogens, endocrine disrupters, suspended solids, heavy metals and hundreds of others different compounds, is a major driver of change in coral reef ecosystems (Wear and Thurber, 2015). A negative relationship between coral cover and sewage discharge has been shown in Hawaii’s coral reef ecosystems (Abaya et al., 2018), as well in Moorea where recent increases in macroalgae have been associated with nutrient enrichment coming from urban areas, among others factors (Adam et al., 2020). Those stressors were shown to increase the development of diseases, alter coral calcification, growth and ultimately distribution, abundance, and diversity (Abaya et al., 2018). Sewage discharge was also shown to hamper reef fish reproduction and behavior, and to profoundly disrupt the food web, due to its role in the development of algal outbreaks competing with corals (Wenger et al., 2015).

Modeling: In Moorea, only 3 % of households have their waters processed at a treatment plant, 94 % are equipped with their own treatment system (a basic sump), and the rest has no treatment system (ISPF-INSEE, 2012). As most of the population lives within 100 m from the coastline, sewage discharges directly into the lagoon via runoff, sewer networks, or individual pipes. Thus, household density was weighted by their corresponding sewage system (0: connected to treatment plant; 1: sump; 2: no treatment system) and was then extrapolated onto the reef using a weighted Gaussian kernel density estimation to map exposure to urban sewage (Evans 2019). Finally, we multiplied the resulting layer with the cost-path layer used in the diffusion simulations

(see section [Diffusion simulation](#) below), to more accurately model the spreading of urban pollution.

Diffusion simulation

We simulated land-based pressures dispersion for 2 out of 3 land-based activities, considered here as point-source pollution : the agricultural and the sediment runoffs, coming from stream mouths. We adapted a model (J M S Delevaux et al., 2018) designed for locations where data on local hydrodynamics are not available. This dispersion model used a cost-path surface based on depth, distance to the shore and wave power ([Table 2.2](#)). Because Moorea is surrounded by a lagoon and a reef crest, protecting from waves and swell, we assumed that wave power has no incidence on dispersion. So, first, for each river mouth, we created a cost-path surface quantifying the least accumulative cost-distance of moving between cells. Layers used to create it (by rescaling them between 1 and 10 and then adding them) are depth and distance to the shore, assuming that the deeper and the closer from the shore cells are, the less the diffusion cost is. Then, the spread of agricultural or sediments runoff into the lagoon was modelled using a decay function based on the cost-path layer, for each river mouth to a specific distance threshold specific to each watersheds as :

$$S_i = s_p \times e^{-c^2/D_c} \quad (4)$$

where S_i is the cell value for dispersed pollutants or sediment for river mouth and associated watershed i , s_p is the initial pressure value at each watershed pour-point (i.e. stream mouth), c is cost-path surface (unitless) and D_c the distance threshold for linear decay from each pour point (i.e. the size of the plume from each stream mouth to the lagoon). D_c is computed as a composite index derived from watershed's area, elevation and precipitation, which are drivers of plume size (Nezlin et al., 2005; Robin Burgers et al., 2014). Without field data on different river plume, we were based on the most studied in Moorea, the Opunohu river, where the plume spread to 2200 meters from the river mouth, to deduce each plume size of each stream with a linear regression.

2.7.2.2 Sea-based activities

Shoreline modification

Impact review: Shoreline modification can result in a direct destruction of near-shore habitats essential for coral reef fish ecology (Beck et al., 2001; Hamilton et al., 2017), and may result increased turbidity and sedimentation to modified hydrodynamics (Dugan et al., 2018). Moorea's shoreline has evolved over the last decades due to direct human transformation (Madi Moussa et al., 2019).

Modeling: We used long-term field data (Madi Moussa et al., 2019) on shoreline state gathered in two categories : anthropogenic shoreline or natural shoreline. Pressure intensity was then modelled by a linear decay from the anthropogenic shoreline to a 25 meters distance, to reflect the direct intensity of human constructions as embankments, very common in Moorea (Aubanel, 2016), and also for the modification of near-shore hydrological condition.

Fisheries

Impacts review: In Moorea coastal waters, fishing activities pertain to coral reef fisheries. Exploitation of coral reef fisheries can lead to a restructuration of fish species composition, loss of critical ecosystem functions, disrupting the entire food web (Houk et al., 2018; Morais et al., 2020; Ruttenberg et al., 2011). In Moorea, between 35% and 50% of people in Moorea are engaged at least partly in a fishing-related activity (Leenhardt et al., 2016; Thiault et al., 2017).

Modeling: We used a predicting model of fishing effort in Moorea, which combine participatory approach to spatially modelled fishing suitability into the lagoon, “the spatial preference for fishing grounds”, and socioeconomic approach to model the fishing capacity, “the ability of the fleet to extract resources” (Thiault et al., 2017).

Wildlife provisioning

Impact reviews: Wildlife provisioning alters natural behaviors of those species, affecting the surrounding food web (Milazzo et al., 2005; Orams, 2002). Swimming with sharks and ray became a popular activity on the island, and local studies reported altered ray and sharks behaviors at feeding sites (Brena et al., 2015; Clua et al., 2010; Mourier et al., 2020; Vignon et al., 2010).

Modeling: Wildlife feeding is allowed in 7 sites, where ray and shark occur on a regular basis. In Moorea, the behavioral effects of feeding site were observed within a 2 km radius (Gaspar et al., 2008). We thus applied a linear decay with a 2 km radius from ray and shark feeding sites to model wildlife provisioning pressure on coral reef ecosystem.

Mooring

Impacts review: Moorings tend to concentrate sailboats in specific locations, risking an increase in the amount of pollutants like hydrocarbons, sewage and antifouling compounds in those areas (Davenport and Davenport, 2006). The resulting effects are similar to those caused by urbanization (Richmond et al., 2018).

Modeling: There is one marina in Moorea with a capacity of 120 boats, located in the bay of Teavaro. Additionally, the current MSP defines five mooring areas for which we monitored a weekly boats density between February and April 2017. We modelled pollution’s intensity from each mooring zone using the diffusion model (see section [Diffusion simulation](#) above), the initial pixel for the diffusion simulation being the centroid of each zone, set with a value corresponding to the mean boats density observed until a distance of 150 meters.

Anchoring

Impacts review: Anchoring can cause direct physical damages to the seafloor and benthic organisms such as corals (Davenport and Davenport, 2006; Dinsdale and Harriott, 2004).

Modeling: Between March and April 2017, we counted the number of sailboats located in areas previously identified as mooring areas ($n = 8$). Then, we assigned to each zones an anchoring intensity index A_i as:

$$A_i = \frac{N_b}{S} \quad (5)$$

where N_b is the mean number of anchored boats in a zone and S is the zone area. Then the vector layer was rasterized using the index as values for the concerned pixels.

Beach attendance

Impacts review: The use of beach by tourists and local people results in higher trampling around those areas. Trampling can affect coral reefs both directly (e.g., resuspension of particulates in the water body, breaks of corals, especially branching morphology) and indirectly (e.g., loss in fish abundance and diversity, loss in coral cover area, smaller coral colony sizes) (Juhasz et al., 2010; Williamson et al., 2017).

Modeling: To spatially model trampling pressure around Moorea, we combined data on houses density, tourism infrastructures, and frequentation monitoring data of high-use leisure areas such as public beaches. Specifically, we extrapolated the average beach attendance onto shallow reef cells (1.8 m deep or less, where trampling are likely to occur) located within a 200m radius using a kernel density estimation. Kernel estimation was completed using `sp.kde` function implemented in “SpatialEco” package (Evans, 2020) in R v.4.0.3 (R Core Team, 2020).

Diving

Impacts reviews: SCUBA divers can accidentally damage corals and other benthic organisms and increase turbidity and sediment deposition through flapping, and collecting or handling organisms (Davenport and Davenport, 2006). Visitation intensity at a diving site have been associated with lower coral cover, higher number of injured and dead corals, and increased disease prevalence (Hawkins et al., 1999; Jones et al., 2004; Lamb et al., 2014; Tratalos and Austin, 2001).

Modeling: We performed interviews with diving operators ($n = 4$) complemented by online information to locate diving sites around Moorea ($n = 13$) and estimate diving intensity in trips per week for each diving sites (from 1: low use; to 5 high use). Diving intensity was represented spatially as a 100 m radius buffer around each site location, excluding reef cells shallower than 1.8 m where diving is not practical.

Boat traffic

Impacts review: Boat traffic can affect coral reefs in many ways, including accidental strikes with large animals (essentially air-breathing marine vertebrates) and corals, noise production, or release of pollutants (Davenport and Davenport, 2006; Jägerbrand et al., 2019; Whitfield and Becker, 2014).

Modeling: We considered six main sources of boat traffic around Moorea: boat tours, dolphin and whale-watching tours, diving boats, jet-ski tours, cruise ship charters and ferries linking Moorea to Tahiti. First, we identified start and stop points and the number of trip per week for each source, based on information found on operators' website, interviews with operators, and field observations. An exception was dolphin and whale watching tours, for which only the starting points were known due to varying presence of animals around the island. Therefore, we randomly generated points on the fore reef, where dolphins and whales are observed. Second, we modelled each operator's boat route by calculating the minimum cost-distance route from their start point to their stop point and back. Our model considers geophysical constrains such as channels, reef passes and low depths. We calculated the number of boats passing within a 100 m radius of each reef cell and we took into account the frequency across the 6 different types of shipping. For the boat tours, jet-ski tours and cruise ship charters, we assigned a frequency of 1 because these activities are daily. For ferries between Moorea and Tahiti, we assigned a frequency of 10 because this is the number of daily trips. And for the diving boats, we assigned a frequency of 3/12, because one operator does 3 trips a day, distributed on 12 possible sites of diving which. We used the aggregated 6 layers to map boat traffic around Moorea, although we acknowledge that other sources of boat traffic inherent among fishers and other local reef users were missing due to a lack of empirical data.

Table 2.1. Input data used to model pressures resulting of human activities

Human activities	Input data	Type of data	Source	Year
Agriculture	Watershed boundaries	Spatial (polygons)	Derived from Digital Elevation Model (DEM) using QGIS-GRASS plugin "r.watershed"	2017
	Outlet points	Spatial (points)	Derived from satellite imagery and rivers layer from urban planning office	2015 and 2012
	Land occupation	Spatial (polygons)	Urban planning office	2012
	Fertilizers and pesticides use	Quantitative	Technical report <i>Analyse éco-régionale marine de Polynésie française, 2010</i>	2007
Land-use changes	Digital Elevation Model	Spatial (raster, 5 m resolution)	Urban planning office	2006
	Land occupation	Spatial (polygons)	Urban planning office	2012
	Mean annual rainfall	Spatial (raster, 1 km resolution)	WorldClim 2.0 (Fick and Hijmans, 2017)	1970-2000
	C factor for RUSLE equation	Quantitative	Peer-reviewed publication (Dumas, 2015)	2015

	Outlet points	Spatial (points)	Derived from satellite imagery and rivers layer	2017
Urbanization	Households	Spatial (points)	Urban planning office	2012
	Households accommodation for sewage treatment	Qualitative (type of treatment)	ISPF	2012
Shoreline modification	Shoreline type	Spatial (lines)	Peer-reviewed publication (Madi Moussa et al., 2019)	2009
Fisheries	Fishing effort	Spatial (raster, 5m resolution)	Peer-reviewed publication (Thiault et al., 2017)	2015
Mooring areas	Mooring areas	Spatial (polygons)	Direct observation	2017
	Boats density	Quantitative	Direct observation	2017
Beaches attendance	Households	Spatial (points)	Urban planning office	2012
	Touristic infrastructures	Spatial (points)	Urban planning office	2012
	Beaches	Spatial (polygons)	Derived from satellite imagery	2015
	Beaches frequentation	Quantitative	Technical report (Pascal & Leport, 2013)	2013
Boat traffic	Starting and ending points of tourism operators	Spatial (points)	Tourism operator websites, local informants and direct observations	2017
	Delimitating buoys for channel emplacement	Spatial (points)	Urban planning office	2012
Diving	Diving sites	Spatial (points)	Local informants and diving operators' websites	2017
	Site frequentation	Qualitative	Local informants	2017
Wildlife feeding	Feeding sites	Spatial (points)	Reglementary data and direct observations	2017
Anchoring	Mooring areas	Spatial (polygons)	Direct observations	2017
	Boats density	Quantitative	Direct observation	2017

Table 2.2. Input data for diffusion simulation

Input data	Type of data	Source	Year
Bathymetry	Spatial (raster, 5 m resolution)	Peer-reviewed publication (Collin and Hench, 2015)	2015
Plume size	Quantitative	Peer-reviewed publication (Adjeroud and Salvat, 1996)	1996
Digital Elevation Model	Spatial (raster, 5 m resolution)	Urban planning office	2006
Watershed areas	Quantitative	Derived from watershed boundaries	2017
Mean annual rainfall	Spatial (raster)	WorldClim 2.0 (Fick and Hijmans, 2017)	1970-2000

Table 2.3. Uses of fertilizers and pesticides in the Winward Islands in 2007

Type of agriculture	Surface (ha)	Fertilizers (in Tons)	Pesticides (in Tons)	Total input	Input in Tons/ha	Weight
Market gardening	156	917.5	126	1043.5	6.69	334.5
Fruit production	755	114.7	25.2	139.9	0.19	9.5
Subsistence crops	239	32.5	4.8	37.3	0.17	8.5
Coconuts	403	5.4	0.8	6.2	0.02	1
Flowers crops	94	Unknown	Unknown	Unknown	Unknown	1
Aromatic crops	53	Unknown	Unknown	Unknown	Unknown	1
Pastures	612	Unknown	Unknown	Unknown	Unknown	1

Table 2.4. Vulnerability coefficients of habitats to pressures coming from human activities for Moorea's cumulative impact assessment

Human activities	Fringing reef	Sandy bottom	Barrier reef	Fore reef
Agriculture	2.177	2.007	2.092	0.5
Land-use changes	2.727	2.86	1.4	0.85
Urbanization	2.177	2.007	2.092	0.5
Shoreline degradation	2.603	0	1.497	1.927
Fisheries	1.471	1.281	2.093	1.76
Mooring areas	2.177	2.007	2.092	0.5
Beach attendance	2.179	0	1.75	0
Boat traffic	1.513	0.65	1.893	1.693
Diving	1.513	0.65	1.893	1.693
Wildlife feeding	1.513	0.65	1.893	1.693
Anchoring	2.179	0	1.75	0

Map of habitats used for cumulative impact assessment

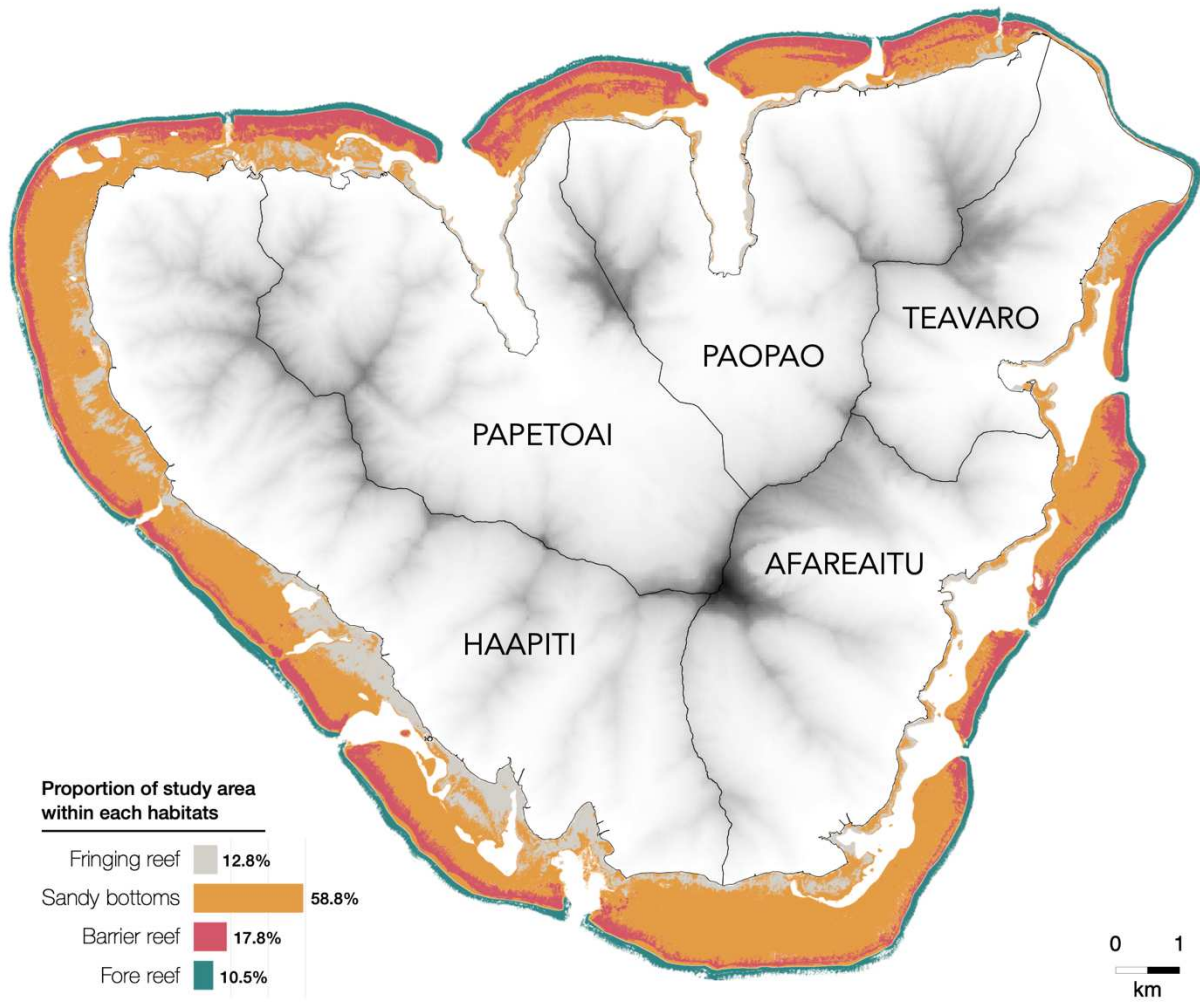


Figure 2.7. Map of habitats used in Moorea's cumulative impact assessment

Chapter 2

Evaluating conservation outcomes of marine spatial planning through the lens of cumulative impact

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Evaluating conservation outcomes of marine spatial planning through the lens of cumulative impact

3.1 Abstract

To face the increasing pace of human uses in the Ocean, marine spatial planning (MSP) is proposed as a holistic approach integrating economical, societal and ecological objectives to sustain human activities and healthy ecosystems. However, the lack of clear objectives, indicators and fit-for-purpose monitoring, along with the unbalanced representation of strategic economic sectors make evaluating MSP conservation outcomes a real challenge. Here, we proposed to address this gap by evaluating MSP conservation outcomes through the lens of cumulative impact. Using a cumulative impact assessment all around the island of Moorea, in French Polynesia, as a baseline, we compare alternative MSP scenarios according to the modeling of their respective cumulative impact assessment outcomes. Proposed MSPs reduce the cumulative impact by almost 60% in more than 20% of the study area, but they fail to mitigate cumulative impacts in originally highly impacted areas. Generally, the results suggest a mismatch between MSP conservation measures and the current situation of human pressures. Our approach allows to compare and prioritize alternative management scenarios in their capacity to mitigate human impact through the use of impact reduction targets.

3.2 Introduction

Humans are using ocean resources at an increasing rate (Jouffray et al., 2020). Together with climate change, the multiple pressures coming from human activities erode marine biodiversity (O'Hara et al. 2021), which compromise the ability of the ocean to provide goods and services for millions of people (Díaz et al., 2019a) and potentially lead to an increasing social inequity in resources access (Bennett et al. 2021). Ocean sustainability management must embrace holistic, multisectoral and integrated approaches (Claudet, 2021; Winther et al., 2020).

Ecosystem-based management (EBM) has been promoted to support sustainable use of marine resources (Lubchenco et al., 2020). Marine Spatial Planning (MSP), sometimes seen as the operationalization of EBM at sea (Katona et al., 2017), produces management plans for human activities that can achieve objectives related to ecosystems, economy, and society (Ehler et al., 2019). In the past decade, MSP initiatives have flourished globally and have been implemented or required by policies at sub-national, national and sometimes at international levels (Ehler, 2020). Those plans are expected to cover more than 50% of world's exclusive economic zones (EEZs) by 2030 (Ehler, 2020).

While MSP processes are now well documented, challenges remain, especially in terms of evaluating their outcomes with regard to their stated objectives (Carneiro, 2013). Indeed, objectives are not always defined, therefore it is not possible to conduct a structured evaluation of MSP. Moreover, even when MSP objectives are stated and indicators to assess progress towards them are used, many MSP are not properly evaluated, due to the lack of fit for purpose monitoring data, or because the links between objectives and indicators are too fuzzy (Stelzenmüller et al.,

2021). It is also well-known that MSP objectives are unbalanced, and conservation dimension takes the “backseat” behind strategic economic and societal interests, leading to a MSP evaluation based on those (Flannery, 2023; Frazão Santos et al., 2021). Finally, attribution, or causality, of a MSP to carry out changes in an environment is complicated to assess, especially in the marine environment subject to many drivers of changes, local or distal, and where counterfactual (i.e. a control situation) is difficult to establish (Carneiro, 2013). Moreover, socio-environmental data used in MSP processes are collected prior to the process, and can only be indirectly linked to MSP objectives (Stelzenmüller et al., 2021).

Conservation objectives of MSP are not attained by managing ecosystems, but by managing pressures potentially harmful to them (Ehler and Douvère, 2009). Therefore, cumulative impact assessments (CIAs), which produce standardized distribution of cumulative human impact on ecosystems (Halpern et al., 2008b; Korpinen and Andersen, 2016), are suggested to become a key element of MSP processes (Halpern et al., 2008a; Stelzenmüller et al., 2021). CIA outcomes allow to discriminate areas along a gradient of human impacts and could provide essential information for managers, prior, during and at each iteration of a MSP process. However, CIA is rarely linked with expected action outcomes, i.e. the consequences of management actions on cumulative impact distribution (Menegon et al., 2018; Tulloch et al., 2015).

Target setting for MSPs could thus be based on the capacity of MSPs to act on cumulative impact through regulations on human activities. MSP process involves stakeholders through consultations and participatory approaches to co-design areas dedicated to specific sectors, activities or interests. Such process leads to several scenarios of zoning schemes where conservation objectives are often left behind (Reimer et al., 2023). Data collected for the pre-requisite CIA could be re-used to compare alternative management scenarios regarding their environmental implications, with a relatively small supplementary cost in time and resources but adding the potential to highlight conservation issues during the MSP process, making it more conservation ready (Reimer et al., 2023).

Here, we use the cumulative impact assessment approach to compare and prioritize alternative management scenarios in their capacity to mitigate human impact through the use of impact reduction targets. More specifically, we assessed the potential conservation impact of a current and a planned MSP, measured as the reduction in cumulative human impact on coral reef ecosystems surrounding the island of Moorea, French Polynesia (Figure 1). We set out to answer two related main questions: How much of the predicted cumulative human impact is reduced by the new MSP?; and how does it compare with the previous management plan?

3.3 Methods

3.3.1 MSPs in Moorea

Resulting from 10 years of concertation, Moorea became in 2005 the first island of French Polynesia to implement a MSP all around the island (referred hereafter as ‘current MSP’). In 2014, a revision of the MSP was launched and a revised version of the MSP with subsequent modification in zoning and rules (referred hereafter as ‘proposed MSP’) was proposed in 2020 to the government, and is pending on approval. At the core of the current MSP is a network of eight

MPAs that focus mainly on regulating fishing, sewage discharge and boat speed. Five of those MPAs are fully protected (*sensu* Horta e Costa et al. 2016), while four are partially protected (Zupan et al. 2018). The current MSP also manages wildlife provisioning, mooring areas and anchoring outside the MPAs. While the current MSP zoning covers almost 40% of the study area, the proposed MSP covers 64%, with designated zones with specific regulations increased about threefold (Table 3.1 and figure 3.1). The proposed MSP will add management of scuba-diving to the activities managed, but it will have less MPAs, in particular less fully protected area.

Table 3.1. Proportion of study area affected by MSPs regulations, in total and per activities

	Current MSP	Proposed MSP
Percentage of regulated area	37.7%	63.1%
Number of zones/sites managed	31	82
Area where activities are affected by regulations		
Fisheries (% of no-take)	35% (11%)	50% (8%)
Shipping	17%	18.5%
Mooring/Anchoring	0.3%	1.5%
Diving	0%	1.3%
Wildlife feeding	Allowed in 6 sites	Allowed in 4 sites
Urbanization	17%	2.7%

3.3.2 Baseline CIA model

The baseline model for our analyses was produced by not accounting for existing management regulations such as MPAs and other spatial regulations. This model was built using local spatial data on 11 human activities impacting four marine habitats (Loiseau et al., 2021). We used a CIA framework that translates modelled pressures from each activity into impact on marine habitats through expert-derived vulnerability coefficients (Halpern et al., 2008b).

3.3.3 Modified CIA models for MSP

The original pressure layers used in the baseline CIA model was modified to incorporate changes due to regulations of each MSPs. Fishing pressure was totally or partially removed inside zones where fisheries are regulated, and, following the “fishing the line” effect (Kellner et al., 2007), redistributed along their borders based on fishing capacity (i.e. density of potential fishers) and fishing suitability (i.e. fisher’s spatial preferences) (Thiault et al., 2017). Sewage discharge coming from urbanization being forbidden in some zones, discharge from households on the coast along these zones were removed from kernel density estimation analysis used to model this pressure. Boat traffic was modelled using a least-cost path analysis to simulate traffic between start points and sites of interests around Moorea, such as diving or wildlife provisioning sites. For the current MSP, boat traffic is regulated through speed limitation in particular zones, where we assumed a linear relationship between the speed reduction and the decrease of pressure intensity. In the proposed MSP, in addition to particular area where speed is limited, diving and wildlife provisioning sites have been added or displaced. We therefore modified boat traffic pressure according to this new configuration of some reef-based activities around Moorea. Wildlife

provisioning pressure layer has been modified to reflect the displacement or removal of sites in the proposed MSP. Boats mooring is only allowed in spatially defined areas, so boats density actually observed outside these zones were re-allocated inside, depending of the distance of the closest allowed mooring zones. For diving, we used a kernel density estimation based on the attendance of known sites to assign a weight for the new ones. Lastly, anchoring is only allowed on sandy bottom, we therefore excluded anchoring pressure from others habitats inside zones dedicated to mooring. For more explanations on pressure layer modifications, see supplementary materials 3.7 at the end of this chapter.

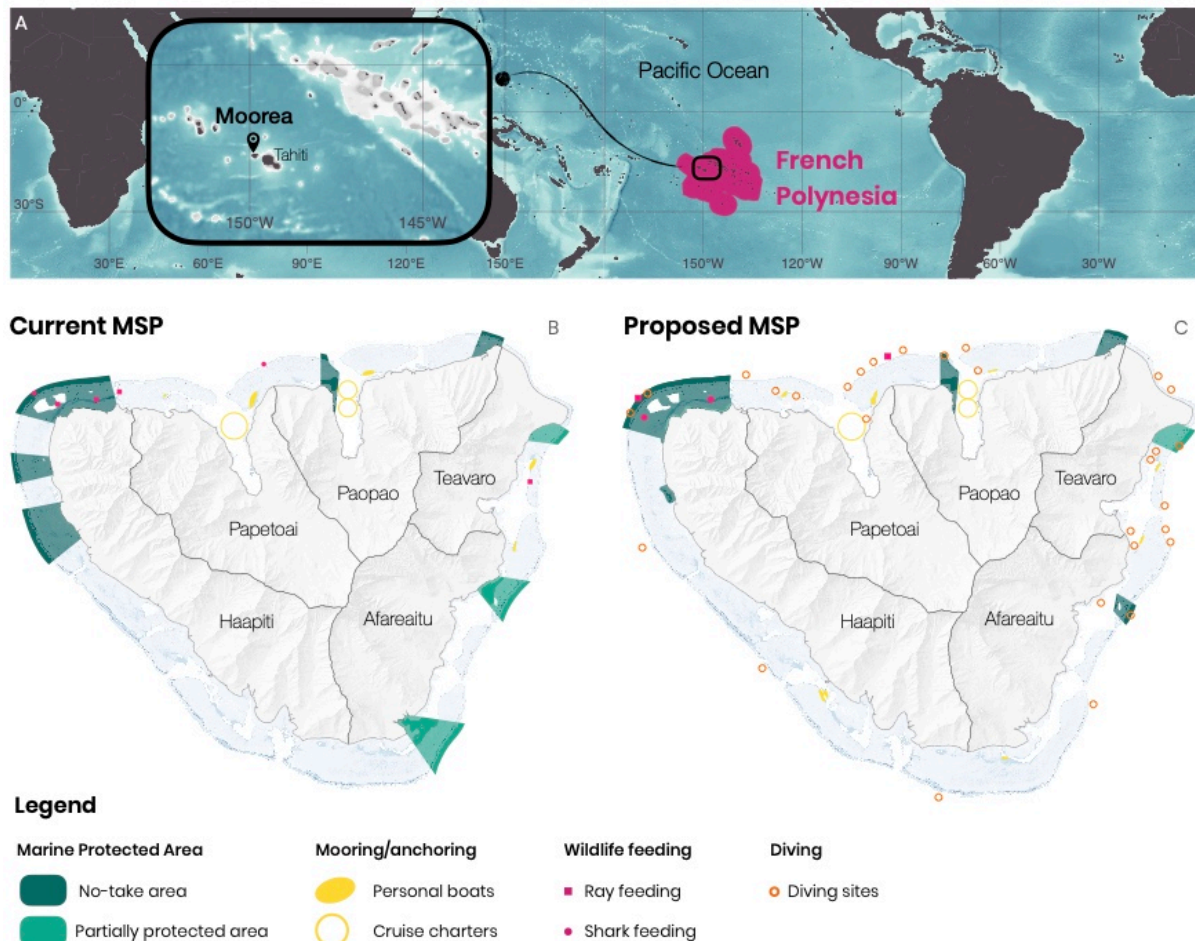


Figure 3.1. Location of Moorea, in French Polynesia, Pacific Ocean (A).

Map of the current (B) and proposed (C) Marine Spatial Planning around Moorea, indicating zones where activities are regulated and incorporated in our study.

3.3.4 Investigating MSP conservation outcomes through CIA

For each MSP, pressure layers associated with activities were translated into potential impact on habitats through experts-derived vulnerability coefficients (Loiseau et al., 2021). They were then summed to obtain two MSP-derived cumulative impact assessments. To assess the ability of MSPs to reduce cumulative impact, we compared MSP-derived CIA against the baseline (i.e. management-free CIA) by computing the log-ratio of cumulative impact inside each 5m grid cell. We assumed that a change in CI between -5% and 5% could result from the sensitivity of the method (e.g., data uncertainty or modelling method selected). Changes in cumulative impact

were classified into three classes: impact reduction (< -5%), no change (between -5 and 5%) and increase (> 5%). Translation between changes in log-ratio and percentage is computed by

$$C_{\%} = (e^{C_{log}} - 1) * 100 \quad (6)$$

where $C_{\%}$ is the cumulative impact change in percentage, and C_{log} is the cumulative impact change in log-ratio.

To describe changes in cumulative impact following MSPs, spatial aggregation of changes was performed for each class at different scales: for the entire study area, for ecological habitats and for municipalities. We also investigated how MSPs can influence areas of low and high cumulative impact. The first and the last quartiles of the cumulative impact score distribution from the baseline CIA model was used as area of originally low and high impact, and overlaid with layers of cumulative impact changes to identify the proportion of low and high impact area concerned by a decrease, an increase or a *statu quo* (unchanged) in impact following MSP regulations.

3.4 Results

Both MSPs reduce the cumulative impact by almost 60% in more than 20% of Moorea's coral reef ecosystems (Figure 3.2 B,E). While sandy bottom habitats receive most of the decrease in cumulative impact in the current MSP (in 25% of its the area), the barrier and fore reefs benefit the most in terms of magnitude of reduction, with a cumulative impact decreased by 76% for both. The proposed MSP would act mainly on distant habitats (barrier and fore reef), both in terms of area experiencing a decrease and the magnitude of the decrease (Figure 3.2 D,E,F). Under the two MSPs, fringing reef is the habitat having the smallest decrease in cumulative impact (Current MSP: -28%; proposed: -24%) in less than 20% of its surface.

The redistribution of some activities leads to an increase of cumulative impact in comparison to the baseline situation in 3.5% and 17% of the overall area, for the current and proposed MPS, respectively (Figure 3.2). The majority of the increase in impact resulting from the proposed MSP is found on the sandy bottom, where 23% of its area would undergo a mean impact increase of 74%. When MSP outcomes are assessed by municipalities, Haapiti benefits from both MSPs with a high magnitude of decrease (-67% and -71%, respectively) for different proportion of its area (20% and 28%). While Teavaro experiences a mean decrease of 70% in almost the half of its area, Afareaitu experiences a mean decrease of 63% in only 6% of its area, indicating that an accurate targeting of pressures responsible of cumulative impact can have the same effect in a wide or a small area. The current MSP increases cumulative impact in no more than 5% of municipalities area, but the magnitude of increase is high, especially for Haapiti (138%). Area experiencing an increase is much larger for the proposed MSP, with a mean increase varying between 40 and 107%. Papetoai exhibits the same proportion of area experiencing a decrease (30%) and an increase (28%), but while the mean decrease of cumulative impact is -22%, the mean increase is 42%, showing the ability of MSP to create winners and losers regarding its benefits.

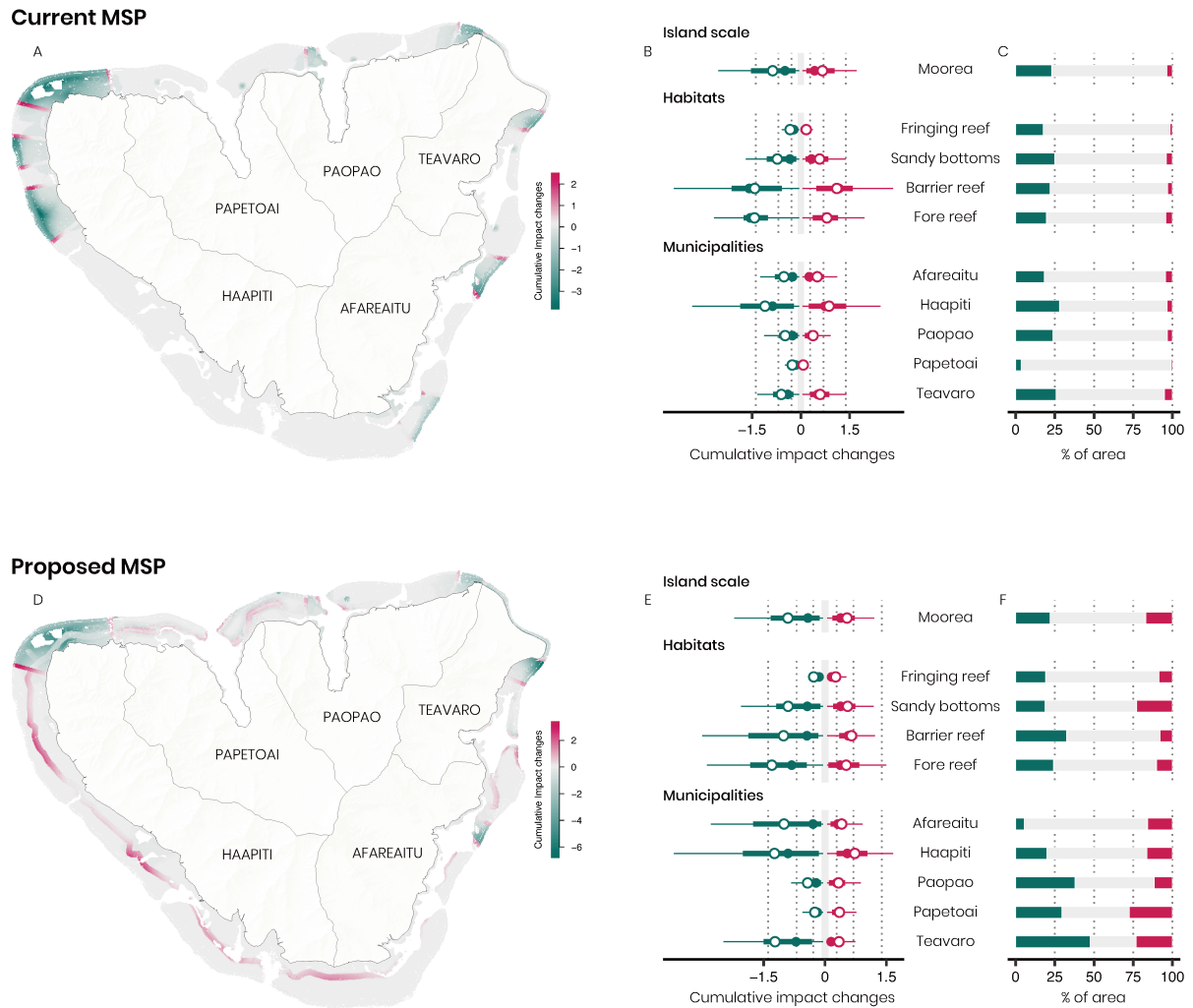


Figure 3.2. Cumulative impact changes under the current and the proposed MSP regulations.

Maps of cumulative impact changes (A and D), boxplots of cumulative impact changes distribution island-wide, per habitat and per municipality (B and E). Hollow circles represent the mean, solid circles the median, thicker lines spans the interquartile range (IQR), while thinner lines extend to the last observation within $1.5 \times$ IQR. Proportion of area affected by an increase, a decrease, or remaining unchanged following MSP regulations (C and F).

Both MSPs fail to fully mitigate cumulative impacts in originally highly impacted areas. Only less than 23% of them experience a decrease in cumulative impact (Figure 3.3). Moreover, the proposed MSP induces an increase in cumulative impact for 14% of already highly impacted area, even if cumulative impact reduction occurs in particularly highly impacted habitats (the fringing reef and sandy bottom). Regarding municipalities, the same reduction occurs mostly in Paopao, which contains a large proportion of originally highly impacted areas, the first being Haapiti where decrease of highly impacted area is less pronounced. The current MSP leads to a decrease in 15% of originally low impacted area, whereas the proposed MSP drives an increase of cumulative impact in 9% of these areas, mainly located in the fore reef, the more distal habitat relatively spared by high cumulative impact.

3.5 Discussion

In Moorea, although the proposed MSP regulations would cover twice the area of the current MSP regulations, their potential effect on reducing the overall cumulative impact is similar. While

the new plan would have more zones regulating fisheries, those zones would have lower protection levels, although fully protected zones offer high benefits for biodiversity conservation (Sala and Giakoumi, 2018). Also, sewage discharge would drop in the area where it would be forbidden despite the detrimental role of land-based pollutants in Moorea's coral reef ecosystem (Adam et al., 2020; Loiseau et al., 2021). Together, these changes could be perceived as negative towards the conservation objectives of MSP, but they rather reflect the management situation in Moorea under the current MSP. First, MPAs monitoring revealed weak responses of protection for harvested and non-harvested fishes inside both fully and partially MPAs (Thiault et al. 2019; Stoner-Osborne 2020). Although a crown-of-thrones sea stars invasion and several cyclones occurred during the period of monitoring, it is likely the lack of compliance and enforcement that are responsible for the limited ecological benefit from MPAs, suggesting that adding more fully protected zones would not add more ecological benefits. Second, the current MSP aims to forbid sewage discharge inside the 8 MPAs, but without dedicated public policies and resources. Even if the necessity to integrate land-based pressures in MSP regulations is recognized, the lack of effective limitations of waste water discharge could have detrimental effects and lead to the ineffectiveness of MPA, including no-take area. This can lead to undesirable effect inside MPAs, supposed to be more resistant to disturbance but which could have detrimental effect inside MPAs (Wenger et al., 2016). Besides, sea and land planning offices work in silos, illustrating the need of a better integration of the ridge-to-reef perspective in planning (Álvarez-Romero et al., 2015a; Carlson et al., 2019).

The first MSP process was data deficient and poorly engaged with the public, leading to a weak compliance of many local community members (Walker, 2001). The proposed MSP engaged stakeholders over 3 years, leading to what appears to be a less ambitious plan for biodiversity protection, but one that could generate more support from local stakeholders. For example, the proposed MSP aimed to improve data on fishing by conducting participatory mapping that identifies key fisheries areas, as well as others important areas for tourism-related activities. Also, the term MPA, not formally recognized in the Polynesian jurisdiction, disappeared in the proposed MSP, to better reflect local policy and to allow an efficient enforcement by existing laws. Altogether, stakeholders engagement and a better integration of human and social dimensions give a better chance to gain public trust towards the management plan (Frazão Santos et al., 2021). Moreover, we demonstrate here that the proposed MSP appears to have a similar capacity to reduce cumulative impact than current MSP, but this reduction may be not equivalent in terms of conservation benefits. The proposed MSP induces more managed zones and area, but with less restrictive regulations on human activities compared to the current MSP, resulting in a diluted cumulative impact reduction compared to the current MSP, which focus its reduction in key areas. Such a spatial dilution of the cumulative impact reduction may be not sufficient to protect marine biodiversity in key areas, and may lead to an illusion of protection (Turnbull et al., 2021), enlightening the imbalance between socio-economic development and environmental objectives (Frazão Santos et al., 2014).

Generally, our results suggest a mismatch between MSP conservation measures and the current situation of human pressures in the study area. In both MSPs, strong regulations inducing a strong decrease in cumulative impact act on no more than 20% of highly impacted areas. Conflicts with users are often avoided to enhance the acceptability across sectors (Kuempel et al., 2019;

Stevenson et al., 2020) but here such strategy leads to strong decrease in cumulative impact (as a result of strong regulations) only where levels of cumulative impact were initially low. Our approach can provide a baseline to identify areas of low and high cumulative impact, to better allocate conservation efforts in MSP. Moreover, it can help set targets on the proportions of low and high impacted areas an MSP could have, to reduce cumulative impact in these areas or to ensure that low level of impact will remains. The reallocation of human activities following MSP regulations led to a decrease in cumulative impact in some places but also to an increase in others. While the current MSP is lightly affected by an increase in impact, the proposed MSP suffers from an increasing impact of 22.5%, in areas already highly or lowly impacted, which emphasizes the need of clearly identify zones of low or high human pressures prior to allocate different regimes of regulations.

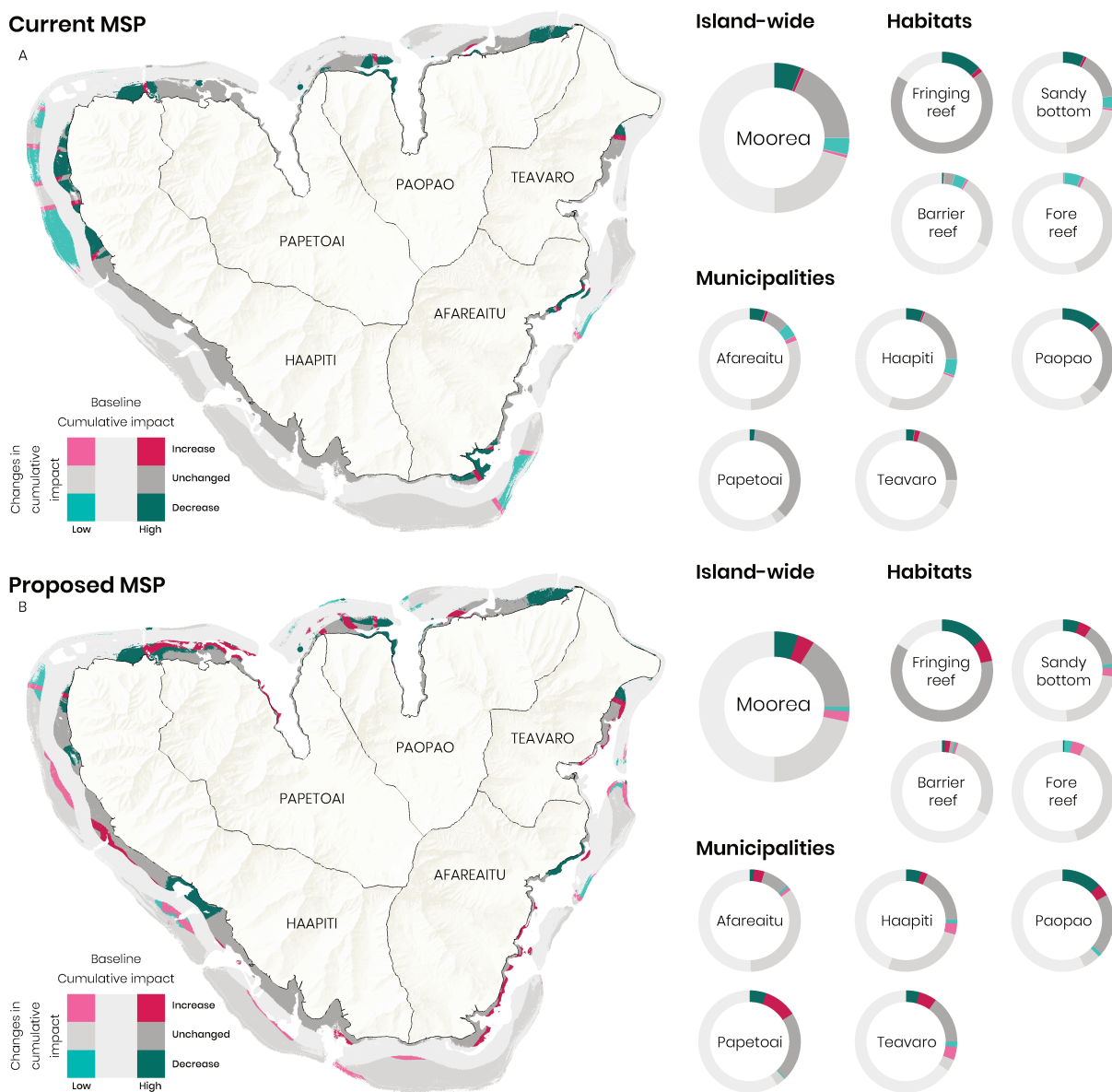


Figure 3.3. Combination of changes in cumulative impact (green color scale for a decrease, red color scale for an increase) under current (A) and proposed (B) MSP and area of originally low (lighter color scale) or high cumulative impact (darker color scale), at the island scale, per habitats and per municipalities (circles on the right of panels A and B)

Two main challenges hinder the development and use of standardized frameworks to evaluate MSP outcomes. First, evaluating potential MSP outcomes has a low-priority compared to day-to-day needs and use of management resources. Second, there is too little practical experiences in MSP to conduct the collection and the analyses of sufficient data necessary for a multi-objectives evaluation (Carneiro, 2013). Indeed, a-posteriori evaluation of MSP conservation outcomes requires a lot of resources for the selection of appropriate indicators, data collection and analyses (Gleason et al., 2013), with uncertainty due to the difficulty to attribute MSP causality in the observed changes (Carneiro, 2013). However, suitable spatial data on human activities and on ecosystems are required in marine spatial planning, being essential during stakeholder consultations, for systematic prioritization tools and for communication towards stakeholders (Flower et al., 2020). Based on those data layers collected at the planning process, CIA constitutes a transparent and adaptable tool to analyze current situation and possible consequences of MSP scenarios on ecosystem components, helping assess MSP outcomes on the environment. Moreover, the same input data on current and future human activities locations and intensities can be used in a Maritime Use Conflict (MUC) analysis, allowing the evaluation of MSP capacity to prevent conflicts for space between uses through an assessment of their interaction, expressed as a matrix of potential conflict score between each activity (Menegon et al., 2018).

The CIA approach itself imply a well-known suite of assumptions and uncertainties at multiple levels (Halpern and Fujita, 2013). The building of pressures coming from human activities is dependent of the available data and their quality which can differ in terms of temporal scale of acquisition and their spatial resolution. Moreover, it often relies on proxy data and intermediate modelling technics, adding more uncertainty. Others assumptions, such as the additivity of pressures or the assumed linear responses of ecosystems to pressures, contribute to an accumulation of uncertainty which however can be evaluated through identification of robust area of low or high cumulative impact for more transparency in decision making (Stock et al., 2018). A second set of limitations emerges when using CIA approach to assess management scenarios: regulations act on human activities in particular zones, and the assumptions of how regulations reduce or displace pressures coming from these activities can have strong influence on the final model (Hammar et al., 2020). Finally, setting target in cumulative impact reduction needs dedicated thresholds to identify tipping points beyond which the capacity of ecosystems to provide services is jeopardized. Such empirical thresholds are being developed, for example for seagrass meadows in Australia (Rees et al., 2023) or coralligenous assemblages in the Mediterranean Sea (Bevilacqua et al., 2018), but it needs to be extended for more benthic habitats and keystone species across the world's ocean, and therefore help managers to set effective target in cumulative impact reduction.

Our approach assessed MSP environmental outcomes integrated within the planning cycle, in a procedure that can be done before the MSP is finalized and that requires no supplementary data than those gathered in the first steps of the planning cycle. It allows a prospective evaluation of MSP alternatives, and may help planners and stakeholders select a final MSP by setting targets in reduction of impacts. For example, conservation objectives for MSP in France require the protection of 30% of its coastal waters using MPAs, with 10% of them being "strongly protected zones", recently defined as an area where human impact has to be removed, or at least significantly reduced (DCSMM, 2018). Our approach can help estimate how a given MSP

scenario could reduce impacts in a given area compared to alternative scenarios, providing valuable insights for prioritizing protected zones. Because management is not about managing ecosystems but managing human activities, impact reduction targets can be set to assess the potential of MSP benefits for the marine environment while being aware of existing challenges to link human impact reduction with the state of biodiversity (Pressey et al., 2017).

3.6 Conclusion

Marine Spatial Planning, and more largely Ecosystem-Based Management, are promoted around the world as tools for balancing human uses of marine resources and ecosystems conservation. Cumulative impact assessment (CIA) proved to be a useful tool to integrate the variety of activities and pressures to assess impacts on different ecosystem components, and to provide valuable insights on spatial distribution of human uses of the sea. We recommend CIA is systematically used in MSP process to guide conservation priorities and assess environmental outcomes of MSP scenarios in a prospective way. We demonstrated here the feasibility and the necessity of our approach, to fill the existing gap in evaluating MSP outcomes in an environmental perspective. Combined with others method such as a Maritime Use Conflict analyses, a social-ecological vulnerability assessment and an evaluation of MSP process, this work constitute a step towards a full evaluation of MSP cycle, integrating environmental, cultural, social and economic dimension of MSP objectives.

3.7 Supplementary materials for chapter 2

3.7.1 General approach

This study focuses on coastal waters of Moorea, in French Polynesia, including a semi-enclosed lagoon and the surrounding fore reef for a total area of 49 km². Here, we used a previously developed cumulative impact assessment (CIA) model (Loiseau et al., 2021) (hereafter “baseline model”) against we assessed the environmental outcomes of two Marine Spatial Planning (MSP), one currently implemented (hereafter “current MSP”) and the second planned, pending for government approval (hereafter “proposed MSP”). To do so, we modified the baseline model following current and proposed MSP zoning and rules, to incorporate changes in activities repartition and intensity, and to model cumulative impact for each of MSP. Therefore, we compared the modified CIA models against the baseline model to assess the capacity of each MSP to decrease cumulative impact.

3.7.2 Cumulative impact assessment models

3.7.2.1 Baseline model

We compiled local data on 11 human land-based and sea-based activities and associated pressures to assess cumulative human impact on 4 distinct reef habitats (fringing reef, sandy bottoms, barrier reef, fore reef) at the fine resolution of 5 m (Figure 2.7). Each pressure layers is rescaled between 0 and 1, and translated into impact through vulnerability coefficients of each habitats to each pressures (Table 2.4) using an additive model (Loiseau et al., 2021). However, we did not log-transform pressure intensities to avoid over-estimating low-intensity pressures when comparing modified CIA model against baseline model.

3.7.2.2 Modified pressure layers for current and proposed MSP

The current MSP manages 5 human activities which are taking into account in our baseline model: fishing, urbanization, boat traffic, mooring and wildlife feeding. The proposed MSP will manage the same 5 plus scuba-diving and anchoring. We modified each concerned pressure layers from the baseline model following zones and/or regulations addressing these activities.

Fishing pressure layer: Fishing pressure layer is derived from a predicting model of fishing effort in Moorea, which combine participatory approach to spatially modelled fishing suitability into the lagoon (i.e. “the spatial preference for fishing grounds”), and socioeconomic approach to model the fishing capacity (i.e. “the ability of the fleet to extract resources”) (Thiault et al., 2017). Both MSP include no-take zones where no fishing activities are allowed, and partially protected zones, where some fishing gears are forbidden. We hypothesis a “fishing the line” effect, meaning that fishers will preferentially report their effort along the borders of restricted zones (Kellner et al., 2007), while keeping preferences for particular fishing grounds (i.e. the fishing suitability component of predicted fishing effort). To simulate this behavior, we created a cost layer assuming that further a pixel from the borders and lower the fishing suitability are, lower the fraction of fishing effort reported is, and then calculated how much of the total fishing effort inside a zone is reported inside each pixel outside the zone as:

$$F_{R(x,y)} = F_{T \text{ or } P_i} \times \left\{ (d_T - d_{(x,y)}) + F_{S(x,y)} \right\} \quad (7)$$

Where $F_{R(x,y)}$ is the reported fishing effort inside a pixel of coordinates (x,y) , $F_{T \text{ or } P_i}$ is the summed (total or partial, depending of the zone regime) fishing effort inside a regulated zone i , d_T is a distance threshold from the zones borders until fishing effort is no more reported, $d_{(x,y)}$ is the distance of the pixel from the zones borders, and $F_{S(x,y)}$ is the fishing suitability for each pixels. For partially protected zones, where use of particular gears (lines, nets or spearguns) are prohibited, we used a fraction of total fishing effort inside a zone, based on the proportion of fishers using each types of gears in Moorea as:

$$F_{P_i} = F_{T_i} * p_g \quad (8)$$

Where F_{P_i} is the summed partial fishing effort inside a partially protected zone, F_{T_i} is the total fishing effort inside this zone and p_g is the summed proportion of fishers using preferentially spearguns (0.83), nets (0.11) or lines (0.06) in Moorea (Rassweiler et al., 2020), depending on which gears are allowed inside the zone. The fishing suitability component is also disaggregated per gears, to take into account the spatial preference of fishers depending on gears they used. Additionally, we report fishing effort inside the lagoon only along the borders also inside the lagoon, and fishing effort outside only on the borders outside the lagoon.

Urbanization pressure layer: Both MSP defined zones where sewage discharge from urbanization is prohibited. Urbanization pressure is modelled using households density along the shoreline weighted by their corresponding sewage system (connected to a treatment plant, sump or no treatment system), and extrapolated onto the reef using a weighted Gaussian kernel density estimation (sp.kde function in the R package SpatialEco) (Evans, 2020). To model the prohibited sewage discharge into particular zones according to both MSP regulations, we assigned a weight of 0 (corresponding to an eventual connection of households to treatment plant) for households susceptible to discharge sewage into these zones and performed two weighted Gaussian kernel density estimation for each MSP.

Boat traffic pressure layer: Boat traffic pressure was originally modelled using a least-cost path analysis (R package gDistance ; van Etten 2017) to simulate traffic between start points and sites of interests around Moorea, such as known diving or wildlife provisioning sites, and restricted only within the channel, where water depth is enough for shipping. For the current MSP, boat traffic is regulated through speed limitation (from 20 to 5 knots) in particular zones, where we assumed a linear relationship between the speed reduction and the decrease of pressure intensity. We therefore decreased by 75% the pressure associated with boat traffic in regulated zones. In the proposed MSP, in addition to particular area where speed is limited in the same order of magnitude, diving sites and wildlife provisioning sites have been added or displaced. We therefore modified boat traffic pressure layer according to this new configuration of some reef-based activities around Moorea, resulting in changes in start points and sites of interests for the least-cost path analysis.

Mooring pressure layer: Mooring pressure, representing a potential increase in pollutants like hydrocarbons, sewage and antifouling compounds in areas where boats are concentrated, was originally based on observed location and frequentation of mooring areas around Moorea. Both MSP spatially define zones where long-term mooring is allowed, we therefore redistributed boats observed outside them, depending of the distance of the closest allowed mooring zones.

Wildlife provisioning pressure layer: Wildlife provisioning is one of the oldest and popular tourism attraction in Moorea, long before the implementation of the current MSP. The current MSP define wildlife provisioning sites matching with those in place, then no modifications have been added to the original pressure layer. However, the proposed MSP spatially defined wildlife provisioning sites resulting in displacement or removal of known sites, we therefore modified wildlife provisioning pressure by incorporating these changes in location. Pressure was modelled as an inverse function of the distance from the site until a threshold of 2000 meters, where experimental studies shown fishes behavioral consequences of these sites (Gaspar et al., 2008).

Anchoring pressure layer: Both MSP spatially define zones for long-term anchoring, and restrained anchoring outside these zones only on sandy bottom, where the associated vulnerability is null (see Table S). The baseline pressure was estimated for each zones, using direct observations of boats relative to the surface of the zone. For the current MSP, boats observation match with the regulated zones, thus there are no changes between the baseline pressure layer and the pressure layer for the current MSP. However, the proposed MSP add new mooring zones and associated capacity (in number of boats). We therefore modified the layer by incorporating these new zones and by estimating the pressure intensity using their capacities and areas.

3.7.2.3 Modified pressure layers only for proposed MSP

Scuba-diving pressure layer: Baseline pressure layer for scuba-diving was created using interviews and online information to locate sites (n = 13) and estimate their visitation intensity (from 1 for a low use to 5 for a high use) as a proxy for pressure intensity. We then represented scuba-diving pressure as a 100 m radius buffer around each sites, weighted by visitation intensity. The proposed MSP spatially defines 29 diving sites, some matching with those previously identified, and some being new. To assign a weight for each new sites, we used the original 13 scuba-diving sites and their visitation intensities to create a layer of potential visitation intensity for new sites using a weighted Gaussian kernel density estimation (sp.kde function in the R package SpatialEco) (Evans, 2020). Therefore, the pressure intensity for each new site is based on the distance of the surrounding scuba-diving sites and their visitation intensities. We then performed a radius buffer of 200 m around each sites to model the pressure coming from scuba-diving activity.

Underprotected Marine Protected Areas in a global biodiversity hotspot

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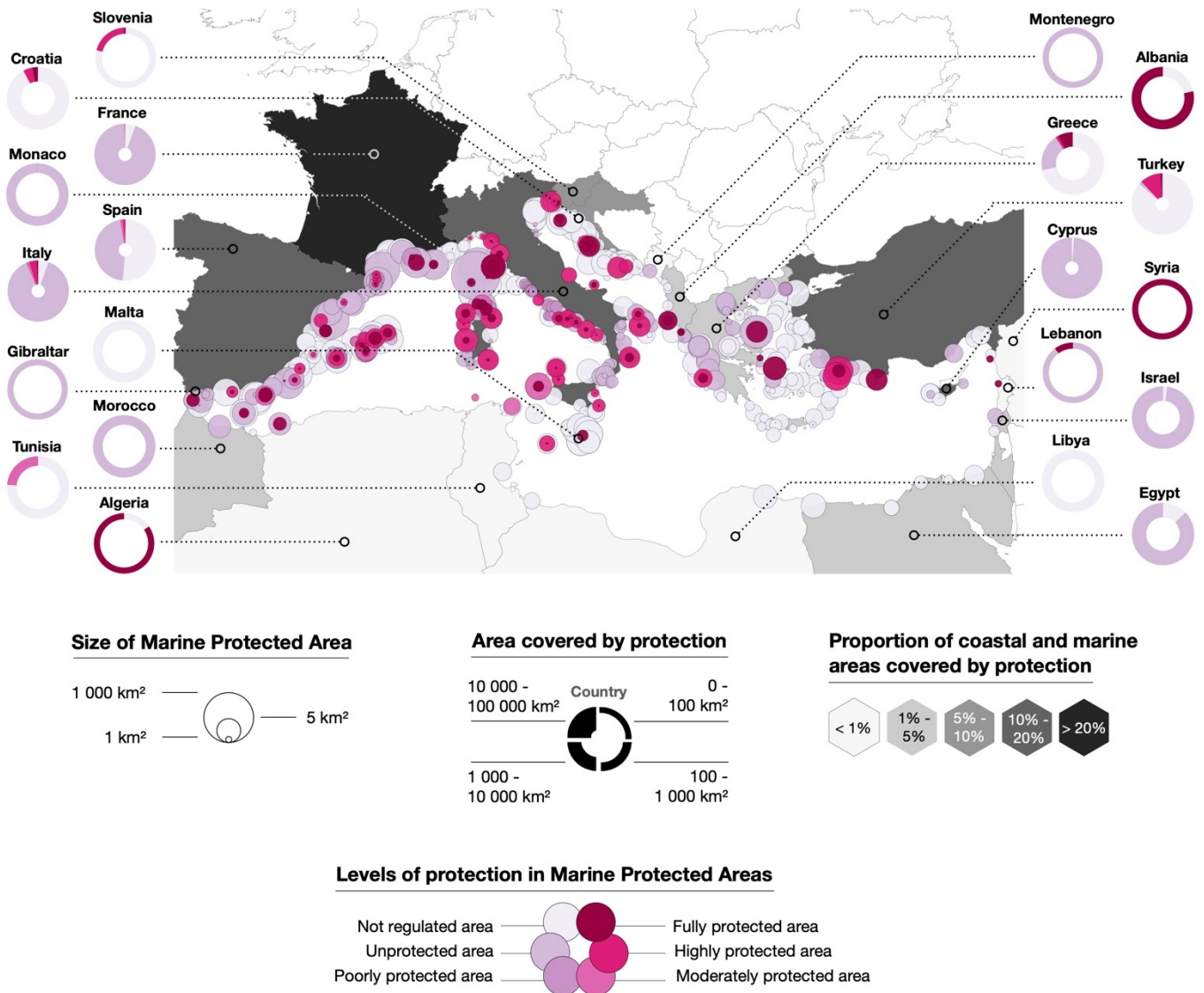
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Underprotected Marine Protected Areas in a Global Biodiversity Hotspot

Graphical abstract

Levels of protection in the Mediterranean Sea



4.1 Abstract

Ocean health is critical for human wellbeing but threatened by multiple stressors. Parties to the Convention on Biological Diversity agreed to protect 10% of their waters by 2020. The scientific evidence supporting the use of marine protected areas (MPAs) to conserve biodiversity stems primarily from knowledge on fully protected areas but most of what is being established is partially protected. Here, we assess the protection levels of the 1062 Mediterranean MPAs. While 6.01% of the Mediterranean is covered by protection, 95% of this area show no difference between the regulations imposed inside the MPAs compared with those outside. Full and high

levels of protection, the most effective for biodiversity conservation, represent only 0.23% of the basin and are unevenly distributed across political and eco-regions. Our current efforts are insufficient at managing human uses of nature at sea and protection levels should be increased to deliver tangible benefits for biodiversity conservation.

4.2 Introduction

A healthy ocean is critical for human well-being. Many Sustainable Development Goals (SDGs) may not be met without achieving SDG 14 for ocean conservation and sustainable use (Singh et al., 2018). However, oceans are threatened by multiple stressors, with fishing as the most important driver (Díaz et al., 2019a). While there is an urgent need to modify human behavior to allow sustainable development pathways (Butchart et al., 2010; Nash et al., 2017), mitigation strategies still need to be put into practice. Within this context, marine protected areas (MPAs) are an effective spatial, ecosystem-based management tool (Lubchenco and Grorud-Colvert, 2015) and Member States Parties to the Convention on Biological Diversity (CBD) agreed to cover 10% of their coastal and marine areas with MPAs by 2020 (CBD Aichi target 11) (CBD, 2010). This areal target is shared by target 5 of SDG 14 and should only be considered a milestone as current research suggests that at least 30% of the ocean should be protected to meet global conservation goals (O’Leary et al., 2016). Here we ask whether CBD Aichi target 11 led to effective conservation strategies or if Member States endeavors deviated from the original aim of the target, which is to deliver conservation outcomes.

Most of the science in support of MPAs has been based on fully protected areas (Claudet et al., 2008; Lester et al., 2009), where all extractive activities are forbidden, yet in order to meet the CBD Aichi target 11, most of the recently established MPAs are only partially protected (Claudet 2018; Sala et al. 2018a). Although partially protected areas can be effective in some instances, they have significantly less conservation benefit than fully protected areas (Lester and Halpern, 2008; Zupan et al., 2018b). Partially protected areas are often preferred over fully protected areas because a broader range of users can still access those areas. However, allowed uses, even if regulated, often concentrate inside such areas (Mazaris et al., 2019; Zupan et al., 2018a) with potentially higher detrimental impacts on biodiversity (Dureuil et al., 2018).

A recently developed regulation-based classification system for MPAs (see [Figure 8.1](#) in supplementary material for chapter 3 and 4) allows MPAs to be grouped according to the potential impacts on species and habitats of allowed uses (Horta e Costa et al., 2016). When applied to a range of published literature on MPA effectiveness it showed that, on average, only fully and highly protected areas, which only allowed infrequent use of some types of non-industrial, highly selective, low impact, recreational, commercial or subsistence fishing gear, could deliver ecological benefits (Zupan et al., 2018b). Protection levels are therefore a good indicator of MPA performance.

In this study, we focused on the Mediterranean Sea, which is both a global hotspot for biodiversity and for human pressure (Coll et al., 2010; Micheli et al., 2013; Myers et al., 2000), and is an area that features an extensive system of MPAs (Amengual and Alvarez-Berastegui, 2018). Our assessment took a critical look at whether conservation efforts are appropriately strategized to deliver ecological benefits.

4.3 Results and Discussion

We compiled information from MAPAMED (MedPAN/SPARAC-MAPAMED, 2018), the most complete database for MPAs in the Mediterranean. For multiple-zone MPAs, we worked at the zone level and reviewed the management plans and legal texts for the 1062 existing MPAs (or 1346 zones) to classify them using the regulation-based classification system (Horta e Costa et al., 2016). All 1062 MPAs included in our study are approved by countries or focal points of the Barcelona Convention (UNEP Regional Sea Convention), and thus count toward international biodiversity conservation targets. When several zones (or MPAs, or designations) overlapped, only the one which conferred the strongest level of protection was kept.

We found that 6.01% of the Mediterranean Sea is covered by an MPA. Interestingly, this percent cover is similar to the global cover with the United Nations Environment Program's World Conservation Monitoring Centre (WCMC) and the International Union for the Conservation of Nature (IUCN) reporting 6.97% of global ocean protection as of 2017 (Sala et al. 2018a). In the Mediterranean Sea, more than a fifth of this coverage is neither established nor managed as no management plan or legal text could be found, and two thirds lack restrictions on activities that can impact biodiversity ([Figure 4.1](#)). Hence, for 95% of the total protected area in the Mediterranean Sea (72.6% of the MPAs) no differences exist between the regulations imposed inside the MPA compared with those outside.

Full and high levels of protection, known to deliver ecological benefits (Zupan et al., 2018b), cover only 0.23% of the Mediterranean Sea and represent only 3.42% of what is being protected. As the CBD's 10% target of countries' coastal and marine areas was designed to achieve conservation outcomes, most of the MPAs, if not all, should fall within these levels of protection. The conservation effort is greatly unbalanced across political boundaries since close to 97% of total marine protection, and 80% and 63% of full and high protection, respectively, lay in the European Union's waters ([Figure 4.2A](#)). This striking unbalance between Mediterranean European and non-European countries can be due to differences in governance frameworks, institutional structures, wealth distribution, social capital, or knowledge on the environment (Abdulla et al., 2009). Such a pattern can also be observed globally, where advanced economies account for two thirds of the global system of MPAs (Marinesque et al., 2012). In the European Union, full and high protection cover 0.15% of countries' coastal and marine areas while it is less than half in non-European countries. Countries that protect a large part of their coastal and marine areas generally harbor large MPAs with low levels of protection ([Figure 4.2B](#)).

The CBD Aichi target 11 stipulates that protected areas have to be "ecologically representative" (CBD, 2010). In the Mediterranean, marine ecoregions (Spalding et al., 2007) are not equally protected ([Figure 4.2C](#)). The Western Mediterranean is by far the most protected (8.62%), but only 1.89% of what is being protected is done so by full or high levels of protection. The Adriatic and Alboran Seas are the second most protected marine ecoregions. Aegean and Ionian Seas have very similar percent cover of protection but full and high protection coverage vary up to

three orders of magnitude. The Levantine Sea and the Tunisian plateau are the least protected ecoregions in the Mediterranean.

Coverage of the different levels of protection in the Mediterranean Sea

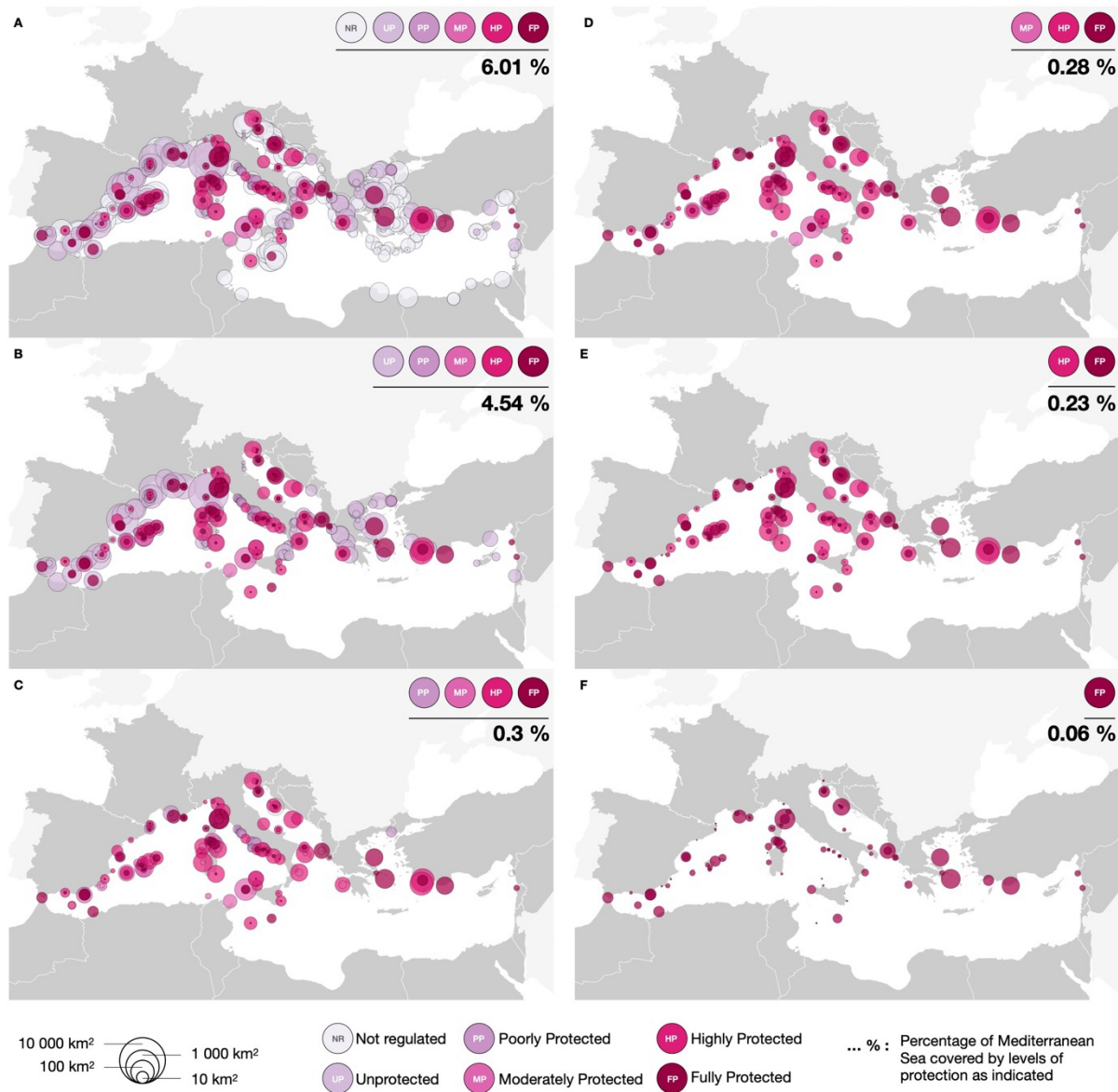


Figure 4.1. Coverage of the different levels of protection in the Mediterranean Sea.

Each dot represents the centroid of a marine protected area (MPA), or a zone within an MPA in the case of multiple-zone MPAs. The size of the dots is proportional to the size of the MPA on a log scale. The color of the dots corresponds to the level of protection of the MPA. The percentage in the top right of each panel represents the cumulative percentage of the Mediterranean Sea covered by the displayed levels of protection in the panel. In each panel (A–F), MPAs from the lower level of protection from the previous panel are sequentially removed.

Our results suggest that much of the Mediterranean Sea is not protected, and more than 95% of what is supposed to be protected does not convey regulations strict enough to confer any ecological benefit (Zupan et al., 2018b). As in other parts of the world, where weak regulations cannot deliver ecological outcomes (Cramp et al., 2018; Dureuil et al., 2018; Magris and Pressey, 2018), or where protected areas are not properly resourced or managed (Gill et al., 2017; Rife et

al., 2013), it is important to ensure that the race to meet key biodiversity targets does not lead us to a false sense of security about appropriate actions being undertaken (Agardy et al., 2016; Lemieux et al., 2019). We believe that classifying MPAs according to their protection levels as we did here is necessary in order to shed light on the fact that current efforts are insufficient with respect to managing human uses of nature at sea (Costello and Ballantine, 2015). We hope this will translate into more action by policy-makers to establish and appropriately manage MPAs with protection levels that are able to deliver tangible benefits for biodiversity conservation.

Distribution of the different levels of protection in the Mediterranean Sea

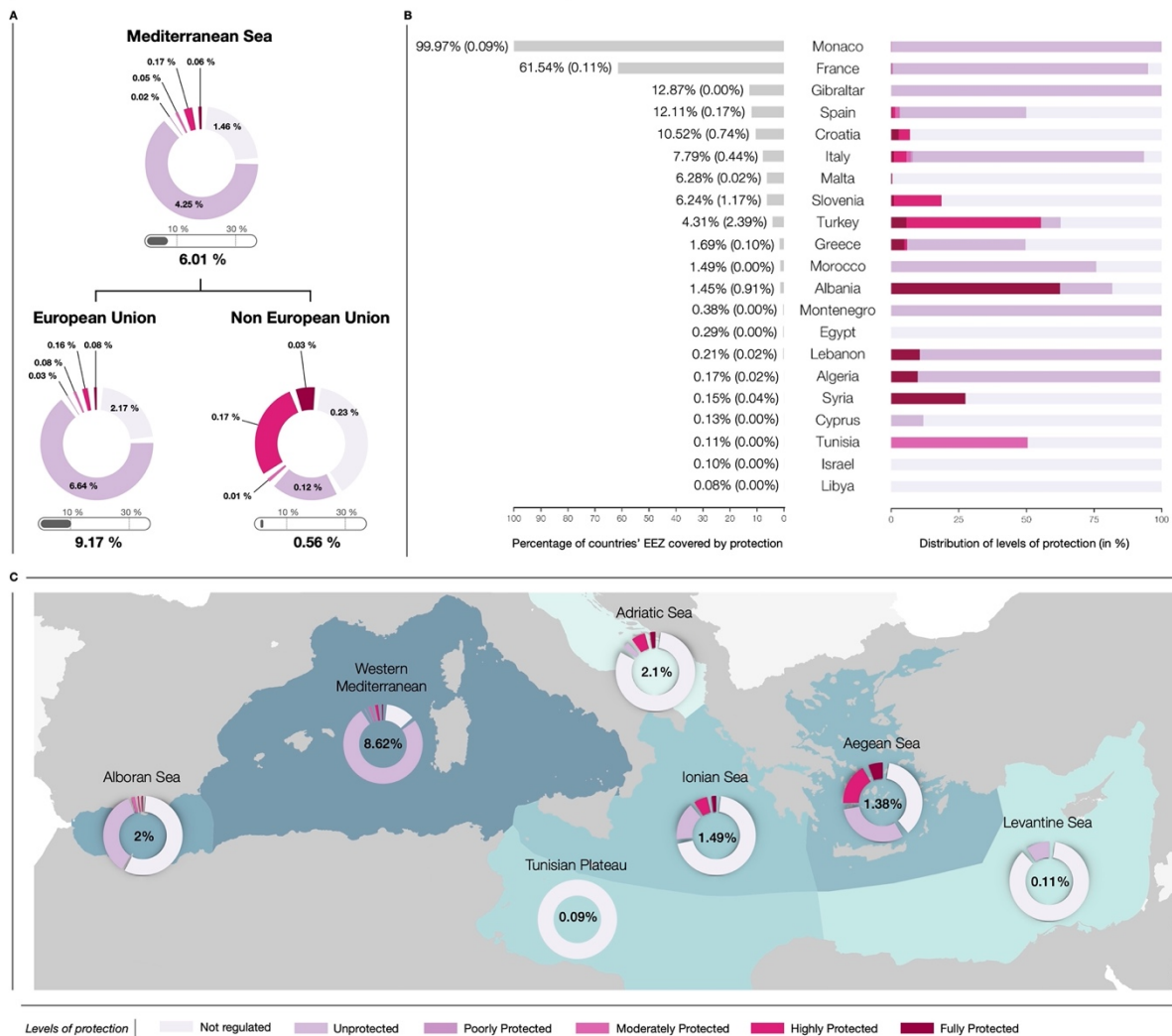


Figure 4.2. Distribution of the different levels of protection in the Mediterranean Sea.

The proportion and distribution of the different levels of protection are displayed at different scales: (A) the entire Mediterranean Sea, European Union, and non-European Union countries (percentages below the progress bars indicate the overall percentage cover of protection in the corresponding grouping, percentages in the colored pie charts show how the different levels of protection are distributed in the corresponding grouping); (B) at the country level (gray bars on the left show the percentage cover of the country's coastal and marine areas under protection for all cumulated levels of protection, percentages inside brackets show the percentage cover of only full and high levels of protection, colored bars on the right show how the levels of protection are distributed inside each country's coastal and marine areas); and (C) at the ecoregion level (colored pie charts show the distribution of the levels of protection inside each ecoregion and percentages indicate the percentage cover of the ecoregion under protection).

4.4 Experimental Procedures

4.4.1 Marine protected areas classification

Legally binding MPAs were collected from MAPAMED (MedPAN/SPARAC-MAPAMED, 2018). Fishing Restricted Areas (n=7), Specially Protected Area of Mediterranean Importance (n=34) and Particularly Sensitive Sea Area (n=1) were removed. In the case of non-strictly marine MPAs (n=46), only the marine part was kept. In the case of multiple-zone MPAs (n=75), MPAs were considered at the zone level. We then collected information on allowed or prohibited activities from legal texts, management plans and personal communications with MPA managers in local languages. Specific information from Natura 2000 sites was also obtained from the European Environment Agency official website (<https://www.eea.europa.eu/data-and-maps/data/natura-10>), but we cross-referenced it as in many cases it was outdated. We then classified all MPAs, or zones in the case of multiple-zone MPAs, using the regulation-based classification system (Horta e Costa et al., 2016). We thus obtained a protection level for each of the 1062 MPAs (or 1346 zones). In the case of MPAs with no legal text or management plan where regulations were established, we assigned the MPAs to a non-regulated category.

4.4.2 Data analysis

Existing georeferenced information in MAPAMED was used. When missing, in multiple instances, and for almost all zoning schemes in the case of multiple-zone MPAs, additional information was obtained as detailed above for the regulations. To avoid overestimating the total area covered by protection we removed overlapping area, keeping only those that conferred the strongest levels of protection for each overlapping layer. Exclusive Economic Zones were retrieved from Flanders Marine Institute, Maritime Boundaries Geodatabase, version 10 (2018); Available online at <https://doi.org/10.14284/319>. Mediterranean eco-regions were retrieved from Spalding et al. 2007. All analyses were conducted using QGIS v.2.18.0 and R (R Core Team, 2021). Areas in km² of the levels of protection levels per country and ecoregions can be found in annex 8.3 “supplementary data for chapter 3”.

Chapter 4

Critical gaps in the protection of the second largest exclusive economic zone of the world

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Critical gaps in the protection of the second largest exclusive economic zone of the world

5.1 Abstract

A healthy Ocean is critical for achieving sustainable development goals but the Ocean is threatened by multiple stressors. There is a global call to increase the coverage of marine protected areas (MPAs) from 10% to at least 30% by 2030. France, a major actor for marine conservation with the second largest exclusive economic zone (EEZ) in the world, with territories in all oceans but the Arctic, aims at reaching the 30% by 2022, for which one third shall include a strong protection status. However, the strategy to reach this twofold target faces two challenges. First, while some standards exist to classify the levels of protection, France is currently using a case specific, loose approach to define strong protection. Second, there is no criteria that addresses the representativeness of the protection across French Ocean basins. Here, we assess the protection levels of the 524 French MPAs and their distribution across territories and habitats. While 33.7% of France's waters are covered by an MPA, 12.5 % of these areas do not impose regulations stronger inside than outside. Full and high levels of protection, the most effective for biodiversity conservation, represent only 1.6% of French waters and are unevenly distributed across Ocean basins and habitats, with 80.5% concentrated in a single territory. To fill this gap in protection for the second largest exclusive economic zone in the world, it is critical that France's high ambition is both qualitatively and quantitatively deployed in each Ocean basin to protect our ocean, its biodiversity and to sustain the livelihood of millions of people.

5.2 Introduction

The Ocean is critical for human society as a whole. Many Sustainable Development Goals (SDGs) benefit from achieving SDG 14 for a healthy ocean (Claudet et al., 2020a; Singh et al., 2018). However, local and global assessments consistently show that Ocean stressors are increasing (Jouffray et al., 2020), with direct exploitation of organisms, and fishing as the most impactful stressor (Díaz et al., 2019a). While new social norms should be developed to foster sustainable development pathways (Butchart et al., 2010; Nash et al., 2017), adaptation and mitigation strategies are still needed. Marine protected areas (MPAs) are an effective area-based management tool in this respect and can help to reach both ecological and social outcomes (Halpern et al., 2010; Lubchenco and Grorud-Colvert, 2015; Reimer et al., 2021). Member States Parties to the Convention on Biological Diversity (CBD) agreed to cover 10% of their coastal and marine waters with MPAs by 2020 (CBD, 2010). While this target has not been met (Maxwell et al., 2020), Member States Parties to the CBD are now discussing options to raise their ambition and to cover at least 30% by 2030.

While the science supporting the usefulness of MPAs was mostly based on fully protected areas (Claudet et al., 2008; Lester et al., 2009; Lubchenco et al., 2003), where all extractive activities are forbidden, the most recently established MPAs to meet the CBD Aichi target 11 are partially protected (Claudet 2018b; Sala et al. 2018a). Partially protected areas are often preferred over

fully protected areas as they can satisfy access to a broader range of users. However, partially protected areas often lead to a concentration of allowed uses, even if regulated (Mazaris et al., 2019; Zupan et al., 2018a), thus sometimes threatening biodiversity more than in unprotected areas (Barnes et al. 2018a; Dureuil et al. 2018a).

10,193,037 km² of French marine territories across the Ocean

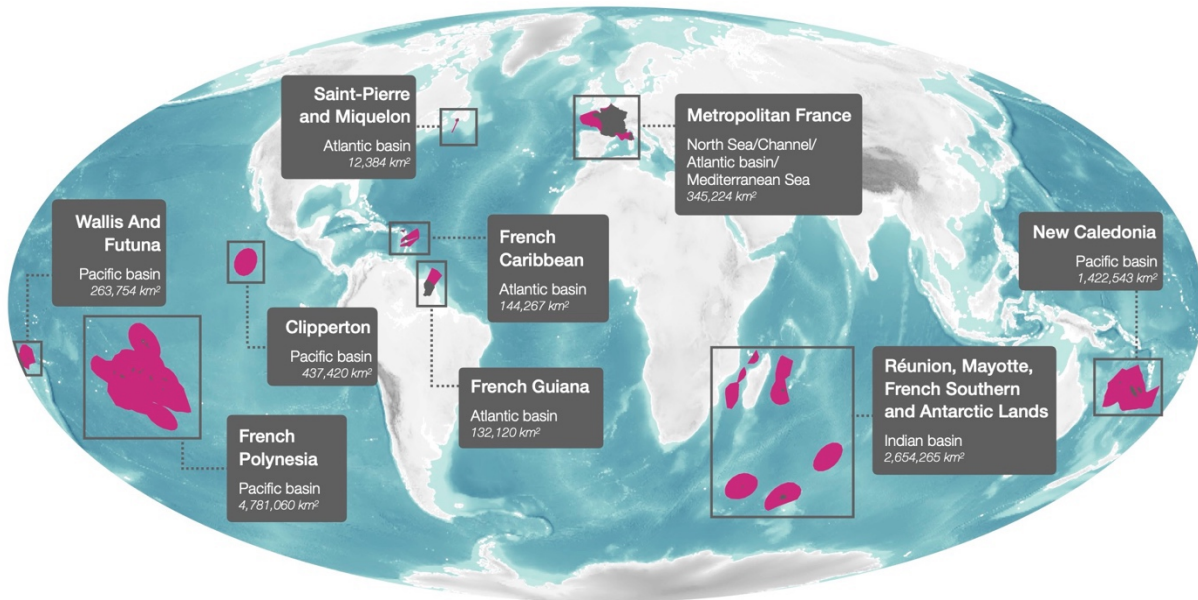


Figure 5.1. France's coastal and marine waters within its economic exclusive zone across the global Ocean

A recently developed regulation-based classification system for MPAs (see [Figure 8.1](#) in supplementary material for chapter 3 and 4), now part of the Blue Parks awarding system (<https://marine-conservation.org/blueparks/>) and integrated within the MPA Guide (www.protectedplanet/c/mpa-guide), allows for MPAs to be classified and compared according to the potential impacts on species and habitats authorized activities can have (Horta e Costa et al., 2016). An assessment of published literature on MPA effectiveness at the global scale with the application of this classification system showed that, on average, MPAs delivered ecological benefits only for full protection (where all forms of extractive activities are excluded) or high protection (allowing only infrequent use of some types of non-industrial, highly selective, low impact, recreational, commercial or subsistence fishing gears) (Zupan et al. 2018c). A regional assessment in Hawaii showed the same pattern (Friedlander et al., 2019). Protection levels are therefore a good indicator of MPA performance (Zupan et al., 2018b) and can thus be used to assess conservation strategies over large spatial scales (Claudet et al., 2020b).

France is a major actor for marine conservation. Due to its numerous Overseas Territories scattered across the Ocean (but the Arctic basin; [Figure 5.1](#)), France possesses the second largest exclusive economic zone (EEZ) in the world after the USA, covering 10,193,037 km², approximately 8% of the total surface of all countries' EEZs. Following the release of the IPBES Global Assessment Summary for Policymakers (Díaz et al., 2019a) in Paris on May 6th 2019, French President Macron called for an increase in MPAs, with respect to both coverage and levels of protection, in France's coastal and marine waters. This commitment was articulated at the first Ecological Defense Council held in Paris, on May 23rd, which aimed to set guidelines for the

ecological transition, including the preservation of biodiversity. France's target is now to reach 30% of protected territory by 2022, for which one third shall have a strong protection status (Claudet, 2019). In early 2019, France officially declared to the CBD 23.57% of its coastal and marine territory was under protection (French Agency for Biodiversity, 2019). The challenge is thus whether the new MPAs will help protect potential underrepresented territories and ecosystems and whether the levels of protection of the so-called "strong" protection will be sufficient enough to deliver the expected benefits to biodiversity.

Here, with a focus on France, which covers a large portion of global marine biodiversity within its EEZ (Costello et al., 2010; Tittensor et al., 2010), which hosts a range of human pressures (Halpern et al., 2015), and which features an extensive system of MPAs with more than 564 protected sites, we critically assess whether the country's current conservation efforts and its new quantitative and qualitative targets are appropriately strategized to deliver ecological benefits.

5.3 Methods

We compiled information on legally binding MPAs from the French Agency for Biodiversity portal (http://www.amp.afbiodiversite.fr/accueil_fr/ressources, 2020), the most complete database on French MPAs. In the case of protected areas non-strictly marine, only the marine part was kept. All identified 524 MPAs included in our study are reported to the CBD, thus counting toward international biodiversity conservation targets. In the case of multiple-zone MPAs (n=20), MPAs were considered at the zone level. We then collected information on allowed or prohibited activities from legal texts, management plans and personal communication with MPA managers. Specific information from Natura 2000 sites was also obtained from the European Environment Agency official website (<https://www.eea.europa.eu/data-and-maps/data/natura-10>), but we cross-referenced it as in many cases the information was outdated. We then classified all MPAs, or zones in the case of multiple-zone MPAs, using the regulation-based classification system (Horta e Costa et al., 2016). We thus obtained a protection level for each of the 524 MPAs (or 564 zones). Assigning a protection level to an MPA using the regulation-based classification system (Horta e Costa et al., 2016) consists in walking through a decision tree, answering a maximum of four questions about the authorized uses within the MPA. MPA protection levels range from Fully Protected, where no form of extractive activities are present, Highly Protected, Moderately Protected, Poorly Protected, to Unprotected, where activities with the greatest impact on species and habitats are not restricted inside the MPA. In the case of MPAs which lacked a legal text or management plan in which regulations were established, we assigned the MPAs to an additional non-regulated category.

Existing georeferenced information in the French Agency for Biodiversity portal was used. When missing, in multiple instances, and for almost all zoning schemes in the case of multiple-zone MPAs, additional information was obtained as detailed above for the regulations. To avoid overestimating the total area covered by protection we removed overlapping areas, keeping only those conferring the strongest levels of protection for each overlap. Exclusive Economic Zones were retrieved from Flanders Marine Institute, Maritime Boundaries Geodatabase, version 10 (2018; Available online at <https://doi.org/10.14284/319>). Marine habitat maps were retrieved

from the European Environment Agency European Nature Information System (EUNIS), Version 2007 (<http://eunis.eea.europa.eu/habitats.jsp>).

All analyses were conducted using QGIS v.2.18.0 and R (R Core Team, 2021). Maps were generated using `rworldmap` (South, 2011), `sf` (Pebesma, 2018) and `tmap` (Tennekes, 2018) packages. Figures were created using `tidyr` (Wickham, 2021) and `ggplot2` (Wickham, 2016) packages.

5.4 Results and Discussion

France has a long history of marine protection, with the first marine protected area (MPA) established in the 1940's (Figure 5.2). While early MPAs often contained a portion under full protection, these segments were typically few and small in size. A significant increase in number and size followed the creation of the French Marine Protected Area Agency in 2006 (Figure 5.2). This agency has now been integrated within the new French Agency for Biodiversity, established in 2020, and in charge of operationalizing the French strategy for MPAs.

Protection of metropolitan France and overseas waters through time

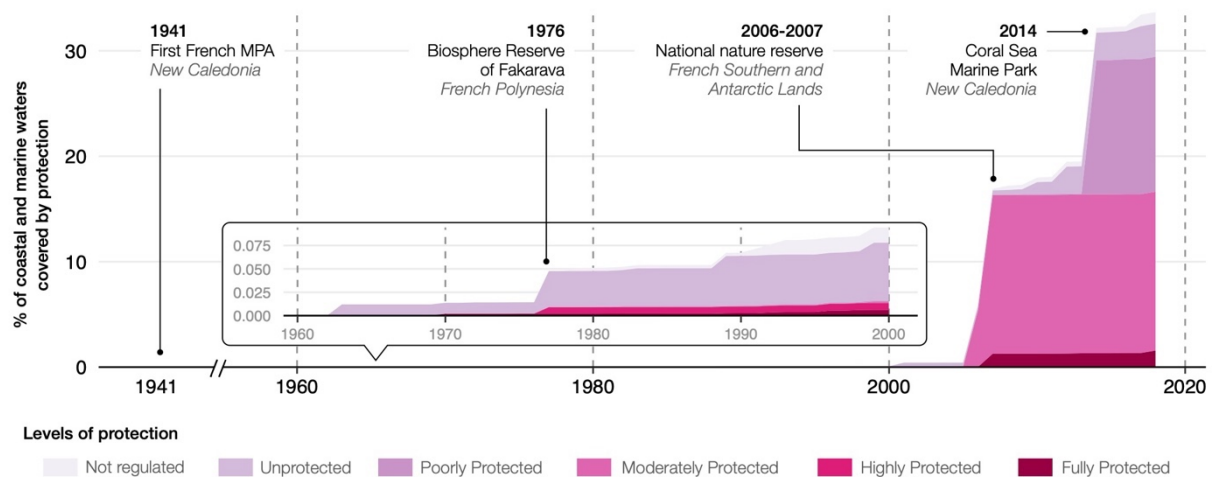


Figure 5.2. Evolution of the coverage and protection levels of French marine protected areas through time.

Colors correspond to the protection level within marine protected areas.

As of mid-2020, France has already reached its 30% target. Currently, 33.7% of France's coastal and marine waters are under some level of protection. This figure is much higher than the global level, with the United Nations Environment Program's World Conservation Monitoring Centre (WCMC) and the International Union for the Conservation of Nature (IUCN) reporting 6.97% of global ocean protection as of 2017 (Sala et al. 2018a). In France, contrary to other parts of the world where paper parks can represent a large portion of declared efforts (Claudet et al., 2020b; Gill et al., 2017; Pieraccini et al., 2017), only 3.3% of the areas designated as MPA cannot be considered implemented as no management plan or legal text could be found. However, more than 10% of the area declared as protected, although often actively managed, does not provide restrictions on activities that can impact biodiversity (Figure 5.3). Hence, for 13.5% of the French protected areas there is no difference in regulations between the inside and the outside.

Coverage of the different levels of protection in French metropolitan and overseas waters

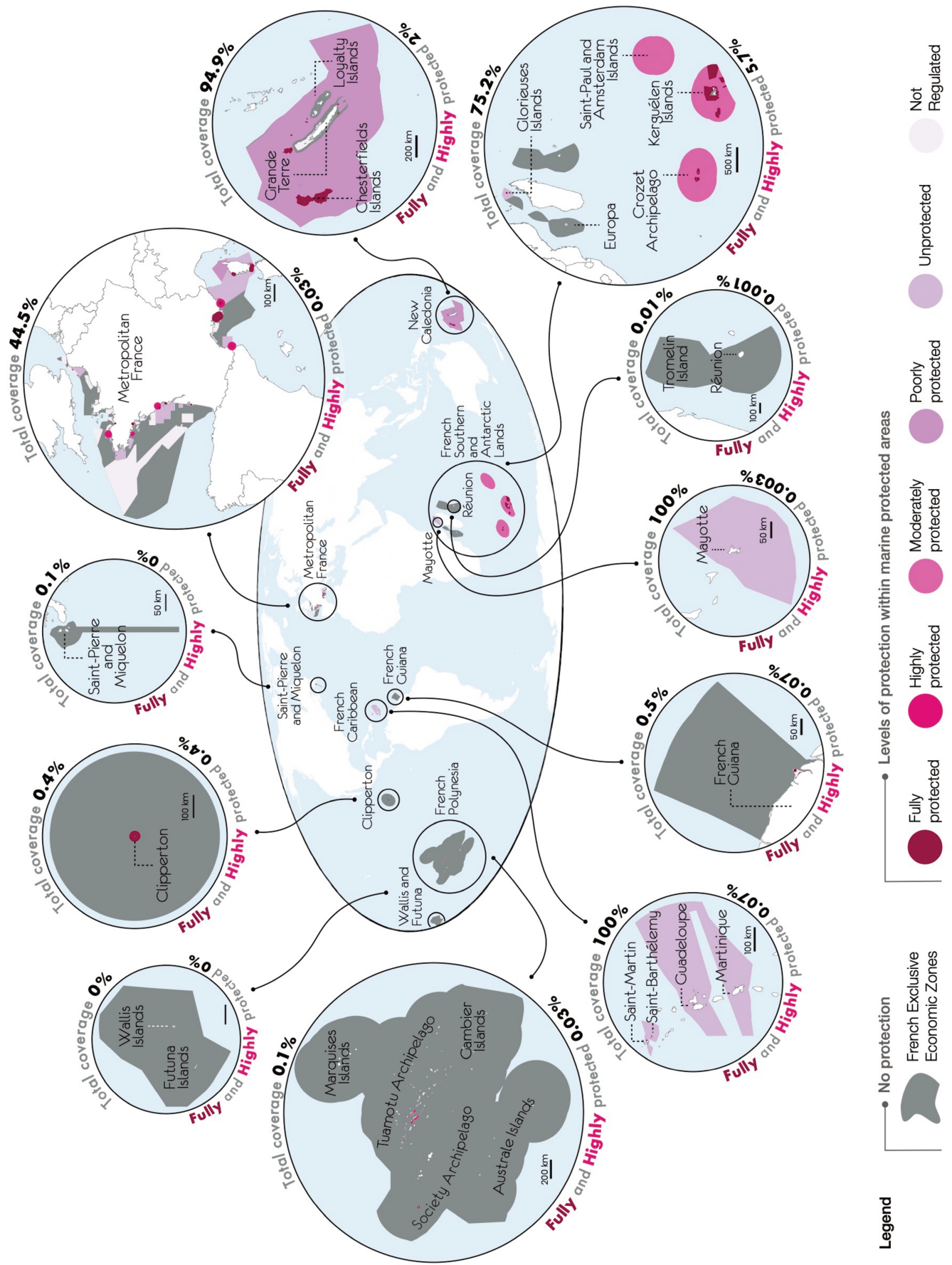


Figure 5.3. Coverage of protection in France's coastal and marine waters.

Colors correspond to the protection level within marine protected areas. Portions of exclusive economic zone not covered by a marine protected area are depicted in grey. For each territory or region, both the total coverage of marine protected area and coverage of only full and high protection are highlighted.

Full and high protection, known to deliver ecological benefits (Friedlander et al., 2019; Zupan et al., 2018b), cover only 1.58% of France's coastal and marine waters, close to an order of magnitude below the 10% target of "strong" protection for 2022.

One of the most striking results of our assessment is that the global figures (across France's global EEZ) hardly depict the situation in each specific ocean basin, emphasizing the need for France's integrated target to be locally or regionally deployed in each ocean basin. Further, the overall effort is greatly unbalanced since more than half of total marine protection, and more than 80% of full and high protection, lay in French Southern and Antarctic Lands' waters ([Figure 5.4](#); [Figure 5.5](#)).

In Metropolitan France, while 59.1% of the French Mediterranean is covered by MPAs, full and high protection represent only 0.09% and 0.01% of the basin, respectively. The pattern is similar in the other Metropolitan France basin, the French Atlantic - Channel - North Sea basin, where while MPAs represent up to 39.5% of its surface, only 0.005% and 0.003% are under full and high protection, respectively. If France's 10% target of "strong" protection was designed to only capture full and high levels of protection, and was only adopted per Ocean basin rather than at the global EEZ scale, France's two metropolitan Ocean basins would fall short of meeting the target by two to three orders of magnitude. In other words, full and high protection of the Mediterranean and Atlantic - Channel - North Sea basins should be increased by 100 and 1,000 fold, respectively.

In France, the EEZ of some territories may be embedded at 100% within an MPA (e.g. Mayotte in the Indian Ocean and all of the French Caribbean Islands), but with protection levels conferring only poor protection ([Figure 5.5](#)). Others, like Clipperton, in the Pacific Ocean, are covered only by full protection ([Figure 5.5](#)). It should be noted that the strongest levels of protection occur in those territories that are both the most remote from urban centers and where France sovereignty is disputed with other countries, such as Clipperton with Mexico and some islands of the French Southern and Antarctic Lands with Madagascar. While these areas host remarkable ecosystems that clearly deserve full and high protection (Friedlander et al., 2019; Quétel et al., 2016), the overrepresentation of the strongest levels of protection in those areas might reflect geopolitics (Leenhardt et al., 2013) and ease of establishment (Devillers et al., 2015).

The CBD Aichi target 11 stipulates that protected areas have to be "ecologically representative" (CBD, 2010). In France, we have seen that marine ecoregions (or Ocean basins) are not equally nor evenly protected ([Figure 5.4](#); [Figure 5.5](#)). The pattern is the same within Ocean basins. Among marine habitats, coral reefs are by far the most protected. France aims to protect 75% of its coral reefs by 2021 and 100% by 2025 (Plan Biodiversité). As of 2020, 65.2% of French coral reefs falls within an MPA ([Figure 5.6](#)). However, the protection levels vary greatly from one territory to another with, for instance, 86.9% and 83.9% of coral reefs within full and high protection in Saint-Barthélemy and Saint-Martin, respectively, while only 1.3% and 2% are under full and high

protection in Martinique or Mayotte, respectively (Figure 5.6). The larger the coral reef extent in a given territory, the lower the protection level.

Distribution of the different levels of protection in France across Ocean basins

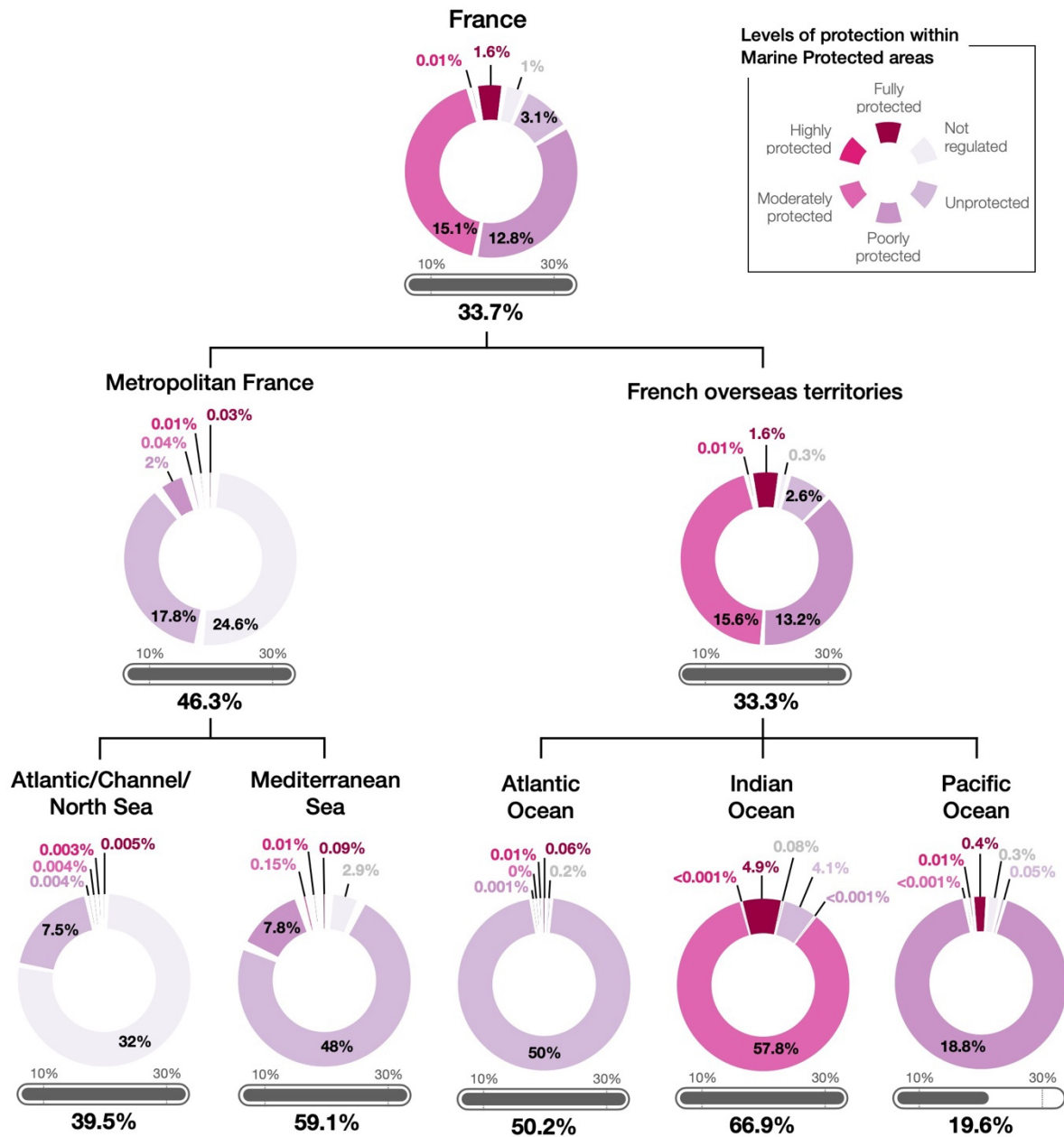


Figure 5.4. Distribution of the different levels of protection in France’s coastal and marine waters across France’s Ocean basins.

Percentages below the progress bars indicate the overall percent cover of protection in the corresponding grouping, percentages in the colored pie-charts show how the different levels of protection are distributed in the corresponding grouping.

In the French Mediterranean Sea, rocky substrates are the marine habitats that receive the most, if not all, full and high protection (Figure 5.7). However, the fraction of these habitats protected under such regimes remains low, with 2.7% of rocky substrates under full or high protection status. Only 1.43% of subtidal soft bottoms are covered by full or high protection. Deep habitats,

while particularly vulnerable and valuable (Davies et al., 2007), never receive similar full or high levels of protection. In the French Atlantic – Channel – North Sea basin, only intertidal habitats (mostly sandy habitats) receive some full or high protection (Figure 5.7). None of the other habitats harbor differences in regulations for potentially impactful use between inside and outside the protected areas.

Proportion and distribution of levels of protection in French waters

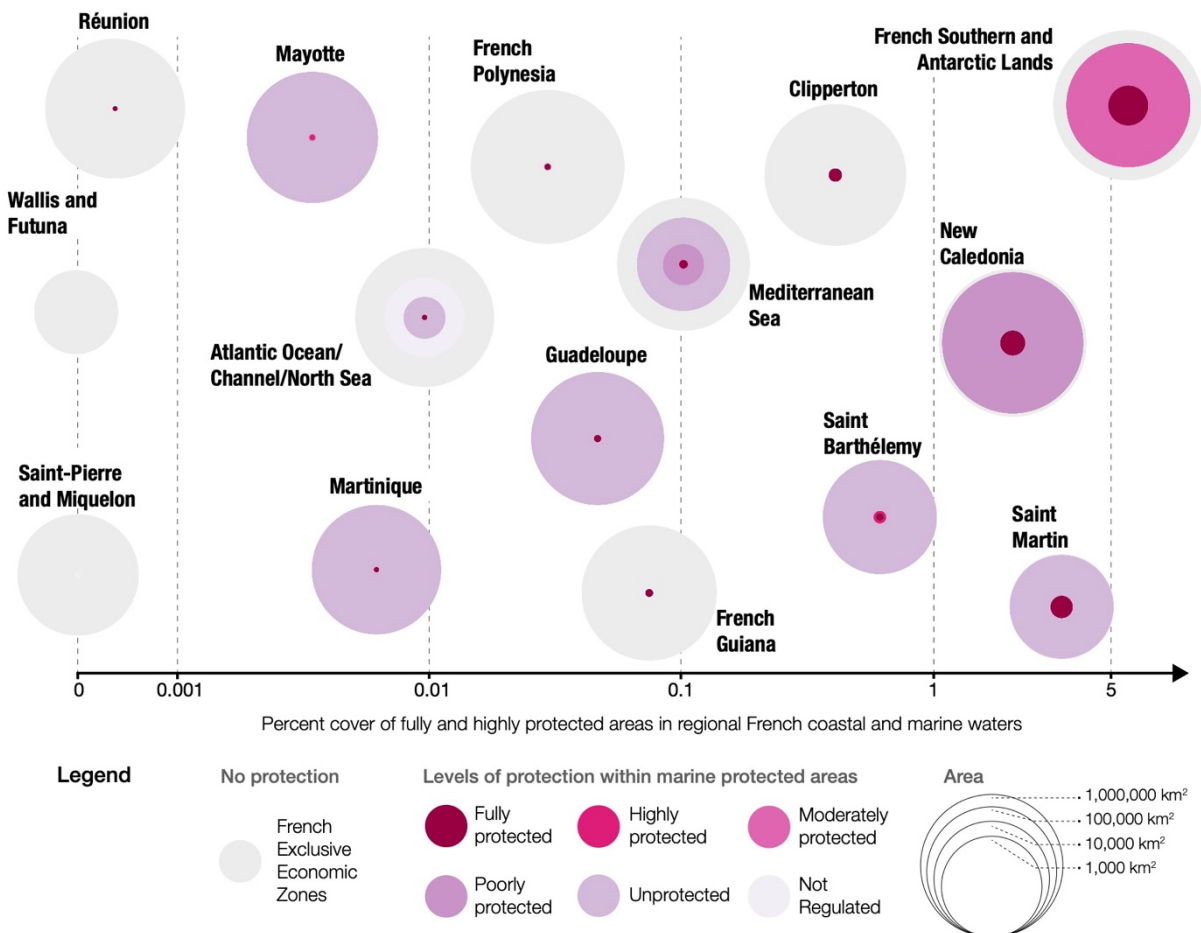


Figure 5.5. Distribution of protection levels per French territory.

The area of the coastal and marine waters of each French territory or region is depicted by a grey circle in a logarithmic scale, within which the proportion of the different levels of protection (relative to the corresponding coastal and marine waters surface area) are depicted by circles of different colors. Circles are ranked horizontally along an increasing gradient of the proportion of fully and highly protected area against the coastal and marine waters area in each region/territories.

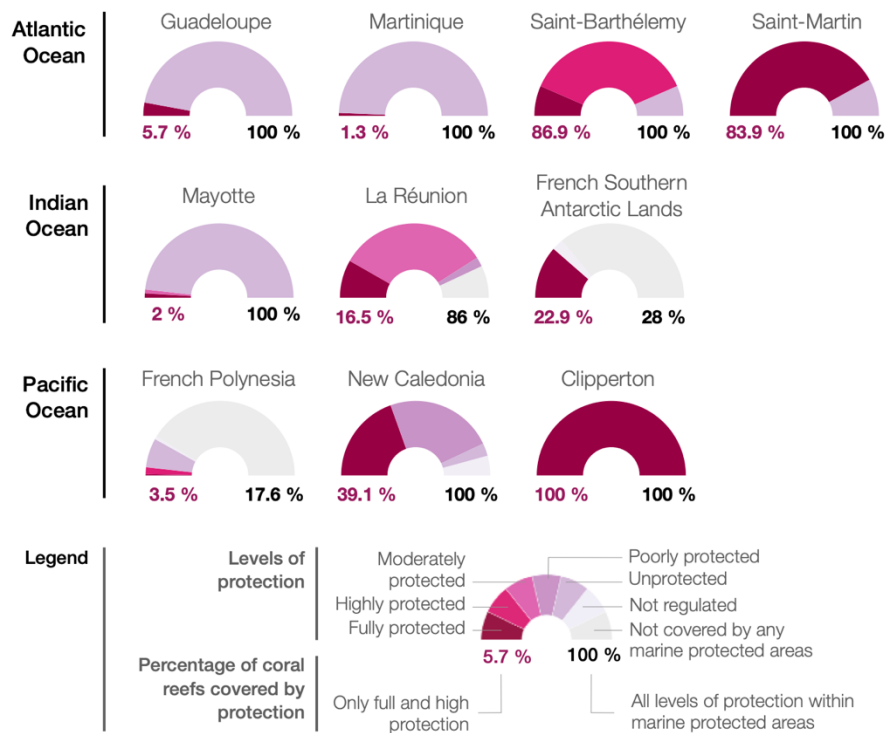
Here, we mostly assessed the representation component of the CBD Aichi target 11 and the new French strategy for MPAs. Other aspects such as effective and equitable management, or connectivity, are also critical in the delivery of ecological and social outcomes (Ban et al., 2019; Gill et al., 2019; Harrison et al., 2020). Such information is still lacking in many regions of the world (Meehan et al., 2020). While close to half of the MPAs globally are not implemented but only proposed or designated (Sala et al., 2018), we identified only 1% of France’s coastal marine waters within MPAs without legal or management texts. While this does not reflect the effectiveness of management, nor the compliance levels, France’s determination to achieve effective management can be observed through the increasing number of French MPAs on the

IUCN Green List or receiving a Blue Parks award (formally Global Ocean refuge System) (Morgan et al., 2018; Wells et al., 2016).

Levels of protection in French coral reefs

Figure 5.6 Proportion of coral reefs within marine protected areas, by protection level, in France's overseas territories.

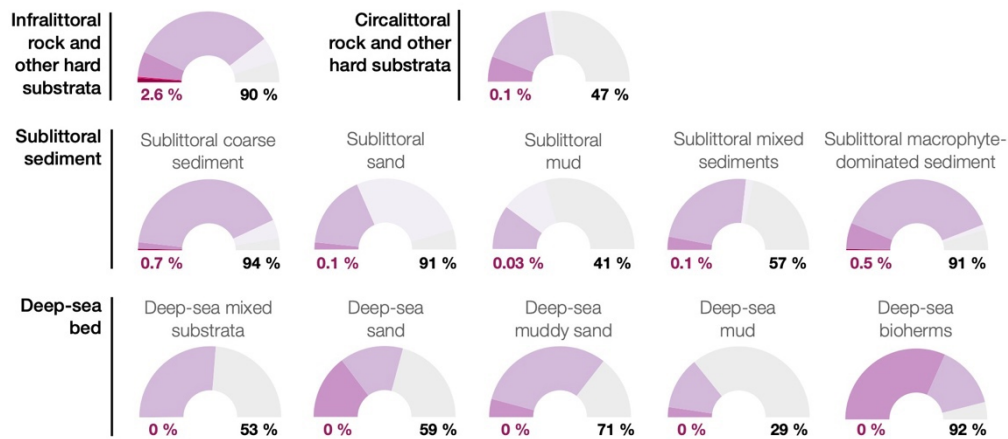
Black percentages show how much of the corresponding habitat is protected by any protection level. Colored percentages show how much of the habitat is protected by full and high protection levels.



President Macron's ambitions to further develop and expand France's MPAs is very promising news for biodiversity conservation. However, to translate this desire into the delivery of tangible benefits depends directly on the sub-targets and clear definitions that are attached to these protection levels. First, and most importantly, in order to avoid to repeat the deep gaps between MPA coverage and effective protection levels such as in the under-protected Mediterranean Sea (Claudet et al., 2020b), it is of utmost importance to ensure the actual protection levels behind the 10% "strong" protection can lead to a sea use change that is meaningful for biodiversity conservation (Claudet, 2019). In this respect, we advocate that France adopts the recently developed international standards for MPA classification, such as the regulation-based classification system for MPAs (Horta e Costa et al., 2016), now integrated in the soon to be released MPA Guide (www.protectedplanet.net/c/mpa-guide), where full and high protection are the levels recognized to deliver the benefits MPAs are expected to deliver (Claudet et al., 2008; Di Lorenzo et al., 2020; Edgar et al., 2014; Lester et al., 2009; Lubchenco and Grorud-Colvert, 2015; Roberts et al., 2017; Zupan et al., 2018b). France is currently using case specific criteria to define strong protection (called Measure M003), which is typically not more restrictive than the simple definition of an MPA according to the standards of the International Union for the Conservation of Nature (<https://www.iucn.org/commissions/world-commission-protected-areas/our-work/marine/marine-protected-areas-global-standards-success>).

Second, unless we consider that the French 2022 30% target is already met, as is the case when computed over the whole French coastal and marine waters, this quantitative areal target (together with its “strong” protection counterpart) should only be considered met if reached for each French ocean basin. Only this would ensure ecological representativeness (O’Leary et al., 2016) and would help to distribute the socio-economic benefits of conserving biodiversity more evenly across French territories (Ban et al., 2019)

Levels of protection in French Mediterranean Sea habitats



Levels of protection in French Atlantic Ocean, Channel and North Sea habitats

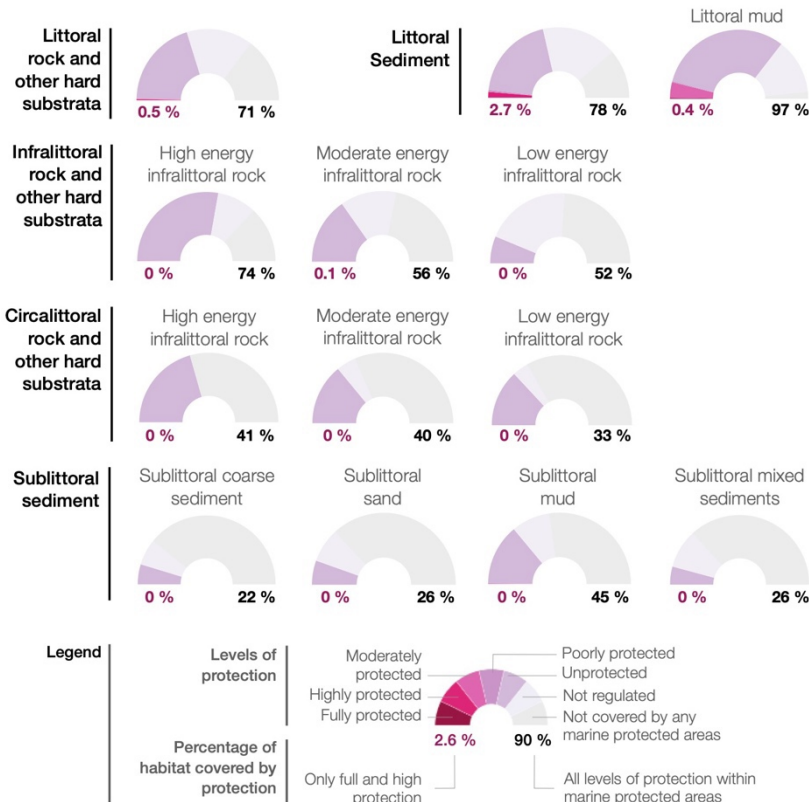


Figure 5.7. Proportion of French Mediterranean, Atlantic, Channel and North Sea basin habitats within marine protected areas, by protection level.

Black percentages show how much of the corresponding habitat is protected by any protection level. Colored percentages show how much of the habitat is protected by full and high protection levels.

As in other parts of the world, where weak regulations cannot deliver ecological outcomes (Cramp et al., 2018; Dureuil et al., 2018; Magris and Pressey, 2018), or where protected areas are not properly resourced or managed (Gill et al., 2017; Rife et al., 2013), it is important to

ensure that the race to meet key biodiversity targets does not lead us to a false sense of security about appropriate actions being undertaken (Agardy et al., 2016; Devillers et al., 2019; Lemieux et al., 2019; Visconti et al., 2019). We believe that classifying MPAs according to their protection levels, such as we did here, is a much needed step towards shedding light on our actual insufficient efforts at managing human uses of nature (Costello and Ballantine, 2015). We hope this will translate into a greater desire by our policy-makers to establish and appropriately manage MPAs with protection levels that are able to deliver tangible benefits for biodiversity conservation (Adams et al., 2019). To truly fill the gaps in the protection of the second largest exclusive economic zone of the world, appropriate levels of protection in all Ocean basins are critical to protect our ocean, its biodiversity and to sustain the livelihood of millions of people.

Drivers of full and high protection in Marine Protected Areas

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Drivers of full and high levels of protection in Marine Protected Areas

6.1 Abstract

Marine Protected Areas (MPA) are a preferential tool for marine conservation, and recent international and national commitments are triggering an expansion of their numbers and surface. MPA are known to deliver various socio-ecological outcomes, but only under particular enabling conditions. A primary one is the level of protection. The multiple benefits of MPAs are maximized under the highest levels of protection, where potentially harmful activities are banned and others are regulated. However, the majority of MPA are harboring the lowest levels of protection and cannot address the most pervasive threats on biodiversity. Understanding the social-ecological drivers that are associated with the highest level of protection can help facilitate the implementation of stronger protection levels as this is now part of national and regional targets. Here, we use data on biodiversity, human pressures, MPA characteristics and country to explore which drivers influence the presence of highest levels of protection. (i.e. the levels restricting extractives activities in MPAs) in 172 MPA of the Mediterranean Sea. Among the six parameters explaining 64% of the model deviance (in order of importance: MPA age, level of land-based pressures, distance from another MPA with strong protection, integrated dependency of countries to marine resources, level of unmanageable pressure, and proportion of Gross Domestic Product spent in environmental concerns), nor biodiversity or level of abatable pressure are retained. Our results demonstrate that presence of high levels of protection inside MPAs is weakly related with conservation objectives such as the decrease of human pressures or the protection of areas with high biodiversity value. There is a need to a deeper understand of the leverage for higher protection in MPA, in order to ease its implementation and ensuring conservation objectives.

6.2 Introduction

To address biodiversity crisis, tackle acceleration of human expansion in the ocean and ensure delivery of marine ecosystem services (Díaz et al., 2019b; Jouffray et al., 2020; O'hara et al., 2021), last decades have seen an exponential growth of Marine Protected Areas (MPAs) coverage worldwide, from 0.1% of the ocean in the 70's to more than 8% today (Lubchenco and Grorud-Colvert, 2015; MPA Atlas, 2022). International commitments for conservation based on area targets, such as Target 3 of the Kunming-Montréal Global Biodiversity Framework have exacerbated this acceleration, but concerns have been raised about these targets and their potentially perverse outcomes (Barnes et al., 2018; Visconti et al., 2019). Among these preoccupations is the capacity of MPAs to effectively reduce pressures within their borders (Zupan et al., 2018a), since the majority of them are partially protected and therefore do not prevent harmful effects from unregulated activities (Sala et al. 2018a).

MPAs capacity to deliver social-ecological outcomes rely on their levels of protection: MPAs where extractive activities are forbidden and others activities are regulated have a higher capacity to deliver various outcomes (Costello and Ballantine, 2015; Sala and Giakoumi, 2018), from the

conservation of biodiversity and ecosystems to the reduction of poverty in adjacent livelihoods (Mcleod et al., 2022). Partially protected MPAs, those where extractive activities are allowed in different degrees, although being the dominant model for ocean conservation, provide fewer, if any, positive outcomes for both people and biodiversity (Turnbull et al., 2021; Zupan et al., 2018b). Moreover, they can even exacerbate and concentrate fishing effort and other impactful activities within their borders (Dureuil et al., 2018; Zupan et al., 2018a).

To incorporate the multiple management schemes of MPAs, proposals have been made to complete the UICN classification and to classify MPAs according to their capacity to reduce potential impacts on biodiversity through regulations on human activities (Horta e Costa et al., 2016). Using this regulation-based classification, Mediterranean Seas MPAs have been assessed and results showed that only 0.23% of MPA coverage are under full and high levels of protection (Claudet et al., 2020b), levels for which ecological benefits are guaranteed (Zupan et al., 2018b). A similar analysis demonstrates a similar trend for French waters, the second largest Economic Exclusive Zone (EEZ) of the world, with only 1.6% of MPAs under full and high levels of protection (Claudet et al., 2021). Recently, a new framework has been proposed, taking into account levels of protection and stage of establishment, to “improve design, implementation and management of future and existing MPAs”, putting into exergue the central question of what an effective protection means (Grorud-Colvert et al., 2021).

Knowing that, new objectives for the protection of ocean have been discussed and the Target 3 of the Kuming-Montreal Global Biodiversity Framework has set a new goal of 30% of protected marine waters by 2030, following researchers guidelines for conserving biodiversity and conservations organizations calls at the World Conservation Congress in 2016 (O’Leary et al., 2016). Moreover, the European Union Biodiversity Strategy adopted in 2020 specify that “at least one-third of EU protected areas – representing 10% of EU land and 10% of EU sea – should be strictly protected” (EC, 2021), as well as in the “Coalition for an exemplary Mediterranean in 2030”. In France, the national strategy for biodiversity specifies also a 10% target for higher levels of protection in MPA, added to the international objectives of 30% by 2030 (JORF, 2022, 2021).

To address the 10% objective towards higher levels of protection, countries will have to rely on their already existing MPA in which stronger regulations could take place, especially for countries where the 30% target of MPA coverage is already reached. For protected areas in land, it has been demonstrated the benefits of investing in already existing protected areas versus creating new one, in order to ensure adequate management of human pressures and to therefore maximize their potential towards ecosystems conservation (Adams et al., 2019). But if we know at which degree of protection benefits are expected and what drivers favor MPA effectiveness (di Franco et al., 2016; Edgar et al., 2014; Gill et al., 2017), few is known about what parameters influence the presence of full and high levels of protection inside MPAS. Previous studies have focused on the presence/absence of MPAs all levels of protection cofounded (Kuempel et al., 2019), or on what factors influence the MPAs’ UICN categories (Stevenson et al., 2020), but none address the central question of what drivers explain the presence of full and high levels of protection inside MPAs, levels at which ecological benefits are ensured.

It has been demonstrated that the locations of many MPAs around the world are driven by the avoidance of areas of high economic values for human activities, or areas with high levels of human pressures (Devillers et al., 2015; Kuempel et al., 2019; Stevenson et al., 2020), therefore resulting in a “favoring ease of establishment over need of protection” principle (Devillers et al., 2015). Moreover, protection strategies differ across countries, depending on their socio-economic characteristics such as their level of dependence to the extraction of marine resources (Marinesque et al., 2012). Regarding this, we gathered data on Mediterranean MPAs, biodiversity, human pressures and countries to identify what drivers currently favor the ease of full and high level of protection establishment within MPAs and to discuss their reliability for the future outbreak of higher levels of protection.

6.3 Methods

6.3.1 Marine Protected Areas

We focused on 172 nationally designated MPAs spread in 20 countries across the Mediterranean Sea, in which 89 contain at least one zone under full or high level of protection *sensu* (Horta e Costa et al., 2016) (Figure 6.1). We only selected nationally designated MPAs because strict regulations on human activities allowing a strong protection is enabled by local government policies (indeed, others MPAs of the Mediterranean Sea designated by regional or international convention do not exhibit full or high levels of protection, except for 0.2% of Natura 2000 sites but excluded from our analysis for more consistency).

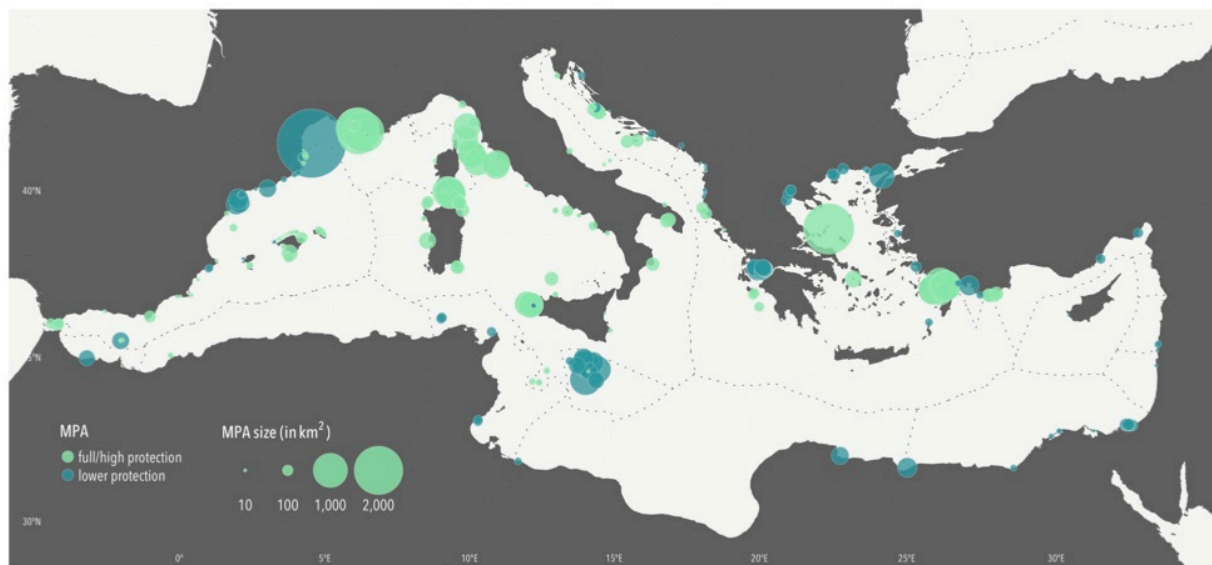


Figure 6.1. Nationally designated Marine Protected Areas in the Mediterranean Sea with or without full/high levels of protection

6.3.2 Drivers of full and high levels of protection

To explore in which extent full and high levels of protection is driven by biodiversity, ease of establishment and countries characteristics, we selected 3 different groups of variables (variables related to biodiversity, human pressures, countries characteristics), plus a fourth group related to MPA characteristics itself.

6.3.2.1 *Biodiversity-related data*

We used two types of data to describe marine biodiversity in Mediterranean MPAs: data on distribution of species from IUCN (IUCN 2021) and AquaMaps (Kaschner et al. 2019), and location of Key Biodiversity Area in the Mediterranean Sea. We combined IUCN maps of species distribution based on expert knowledge of occurrence and habitat preferences, and AquaMaps 0.5° grids based on environmental envelopes of species matched against real environmental conditions, to obtain a layer of species richness in the Mediterranean Sea (Roberson et al., 2021). We selected resident marine species from the Mediterranean Sea using IUCN classification (n = 1930), for which 710 maps were available. Each species layers was rasterized at 1 x 1 km resolution and summed up to obtain a number of species contained in one cell. AquaMaps data for Mediterranean Sea were provided as data frame containing species (n = 2344), geographical coordinates and probability of occurrences. We only kept cells for which probability of species occurrences was higher than 0.5 and removed *Protozoa* and *Chromista*, giving us a total of 2186 species. We then combined IUCN and Aquamaps datasets by excluding 496 common species across both, identified by names or synonyms. IUCN data were preferred over AquaMaps because they are expert reviewed. The final layer contains number of species in each cell of 1 x 1 km, for a total of 2400 marine species (Figure 6.2A).

Biodiversity predictors

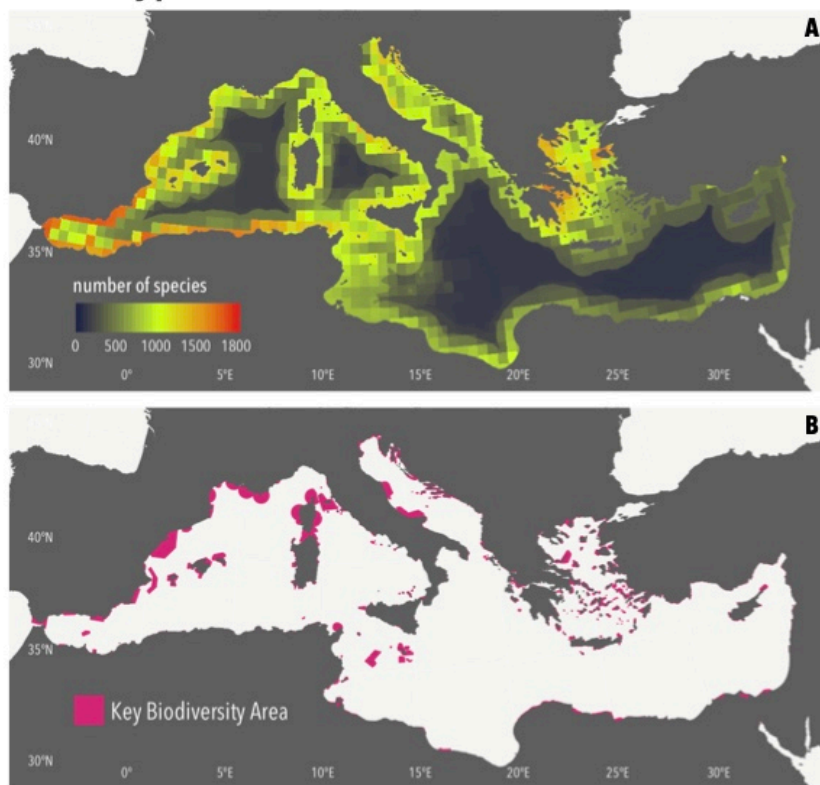


Figure 6.2. Biodiversity-related predictors used in our analysis

Map of species richness from IUCN and Aquamaps dat (A); map of Key Biodiversity Areas in the Mediterranean Sea (B)

In complement to species richness, we used Key Biodiversity Areas (KBA) as drivers for strong protection. KBA are defined as “sites that contribute significantly to the global persistence of biodiversity”, and are standardly defined with a set of indicators and thresholds (IUCN, 2016). Spatial data were downloaded from KBA website and intersected with Mediterranean Sea, giving us 484 marine KBAs (Figure 6.2B). To facilitate computation, the spatial layer was rasterized at the resolution of 1 x 1 km, used to compute percentage of KBA inside each MPAs.

Human pressure predictors

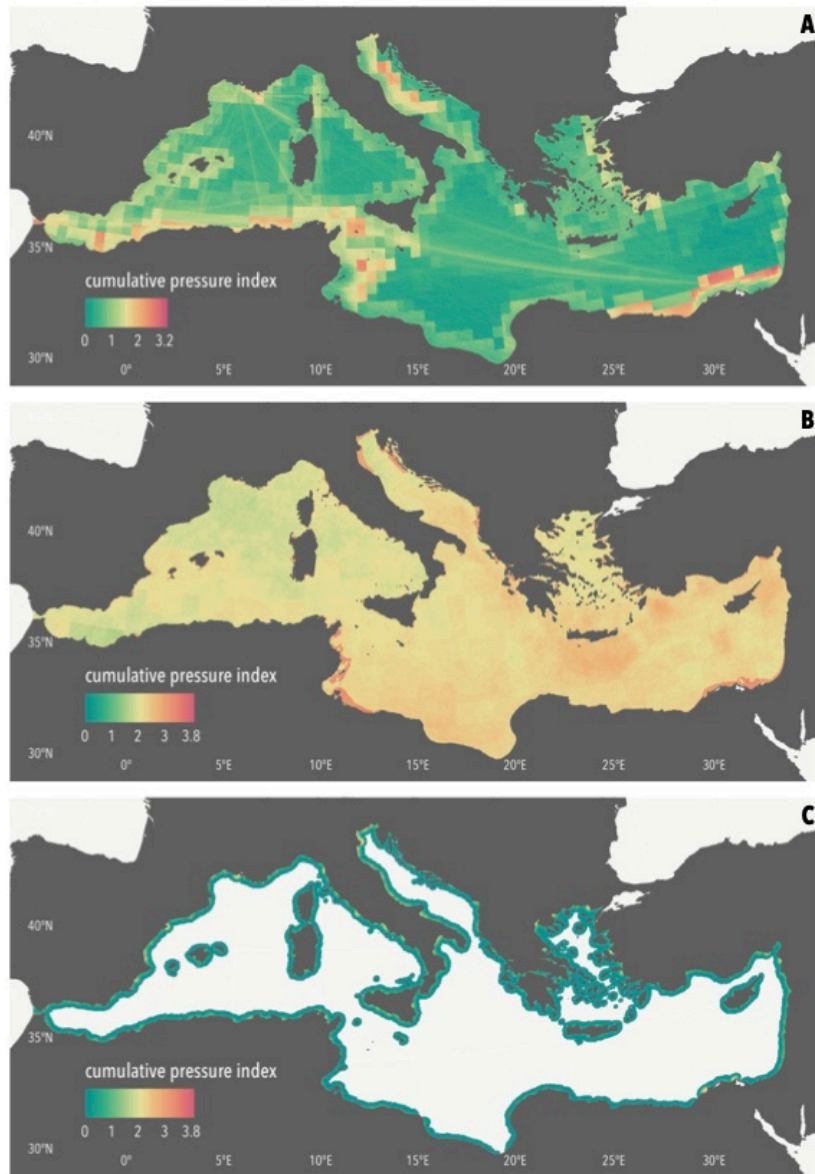


Figure 6.3. Human pressures predictors used in our analysis

Maps of abatable pressures (A); unmanageable pressures (B); land-based pressures (C).

6.3.2.2 Human pressures data

We used a dataset of 18 human-induced pressures on marine environment mapped for the Mediterranean Sea (Micheli et al., 2013). Pressures were gathered in three groups: abatable pressures are pressures coming from activities occurring at sea and directly manageable through regulations where they take place (like fishing activities or shipping) (Figure 6.3A); unmanageable pressures are related to climate change (sea surface temperature, ocean acidification) and require international coordination at the global scale (Figure 6.3B); land-based pressures are those coming from activities on land and require integrated land-sea management (pollutants inputs from agriculture, or urbanization) (Figure 3C). Individual pressure layers consist of log-transformed and 0 to 1 normalized rasters, at a 1km² resolution.

6.3.2.3 MPA-related data

Three variables were retained for MPAs: area, age and distance of the closest MPAs with strong protection. Area is expressed in square-meters and log-transformed for a better reading of results, as well as for distance in kilometers.

6.3.2.4 *Countries related data*

We collected data on Mediterranean countries to express their dependency to the sea for resources and their willingness to protect their waters. We summed for each country the total aquaculture production and fish catch between 1999 and 2019, using the FAO database for zone 37 (FAO, 2021). We choose this time range because the mean age of our MPA dataset is around twenty years. The annual fish catch was also used to compute a trend in catch per country across this period, using the slope coefficient of linear regression models. Added to these raw indicators of marine resource dependencies, we used a composite index of human dependence on marine ecosystems, expressed as the mean of the economic and nutritional dependence (Selig et al., 2019). For countries' commitment towards protection, we used public expenditures for environment expressed as the percentage of the gross domestic product as yearly mean between 2005 and 2018 (IMF, 2018), the percentage of countries' EEZ under protection, the number of MPAs per country, the oldest MPA per country and the trend of MPA cover, expressed as the area covered by MPA across time (See [table 8.5](#) in supplementary data for chapter 5).

6.3.3 *Data analyses*

We used boosted regression trees (BRT) to examine the occurrence of full/high levels of protection inside MPAs in relation to biodiversity, human pressures, MPA characteristics' and countries to which MPA belong. BRT is built on a large succession of relatively simple trees by iteratively fitting each new tree to the residuals from the previous one to optimize predictive performance (Elith and Leathwick, 2015).

First, we assessed pairwise collinearity between our predictors using Spearman coefficients and removed predictors for which coefficients was greater than $|0.7|$, leading to the exclusion of aquaculture production, EEZ area, and trend in MPA coverage. We completed the collinearity analysis with the Variance Inflation Factor, for which no estimates were greater than 3.3. Secondly, we introduced a random predictor in our dataset (random value between 0 and 100) to test later the significance of relative influence of others predictors. Third, we fitted the BRT model with the *gbm* package (Hijmans et al., 2021) in R statistical environment (R Core Team, 2021) using a Bernoulli distribution (presence or absence of full/high levels of protection inside MPA). BRT requires three parameters to be specified: the shrinkage s (or learning rate), which controls how quickly the algorithm proceeds down to the gradient descent and minimizing the residuals at each step; the interaction depth ic , controlling the complexity of the boosted ensemble; the bag fraction (bf), which determine the proportion of original dataset randomly selected to build trees at each iteration. In order to identify the best set of parameters, we performed a hyperparameters tuning by exploring all combinations of the parameters to be set ($ic=[1; 2; 3; 4; 5; 6; 7; 8; 9]$), $s=[0.001; 0.005; 0.01; 0.05; 0.1; 0.15; 0.2]$, $bf=[0.5; 0.6; 0.7; 0.8; 0.9; 1]$, using ten-fold cross-validation, and the best set of parameters maximizing cross-validated Area Under the Curve (AUC) was retained ($ic=8$, $s=0.15$, $bf=0.6$).

Final BRT model explains 28% of the cross-validated variance with high predictive performance (cross-validated AUC = 0.83) and low positive spatial autocorrelation (Moran's index = 0.14, p -value < 0.001). We calculated the mean relative influence of the 10 predictors for which their relative influence was superior to the random predictor influence from 1,000 bootstrap replicates

of the original dataset. Based on the same bootstrap replicates, we obtained partial dependency plots to visualize the relationships between the most influential predictor variables and the response (probability of having full/high level of protection between 0 and 1), while keeping all other predictors constant.

6.4 Results

Five predictors have a mean relative influence of 58% on the variability of presence of full/high levels of protection inside MPAs ([Figure 6.4A](#)): MPA age (14%), levels of land-based pressures (12%), distance to another fully/highly MPA (12%), integrated dependency of countries to marine resources (10%), levels of unmanageable pressures (10%). When looking per group of predictors (human pressures related, biodiversity related, country related or MPA related), countries and MPA characteristics explain together more than 60% of the variability in the presence of full/high protection in MPAs (62%), leaving 22% for human pressures predictors and only 9% for biodiversity predictors. Finally, the abatable pressure predictor was not kept in the final model since its relative influence was smaller than the one for the random predictor, as well as for two country-related predictors: oldest MPA and percentage of EEZ protected.

The parabolic shape of the relationship between full/high levels of protection probability and MPA age suggests a lower probability for these levels of protection in MPA older than 40 years and younger than 20 ([Figure 6.4B](#)). The distance from a MPA with full/high levels of protection to another one with full/high levels of protection is the third more important predictor but do not demonstrate particular pattern until a distance of approximately 150 km, from which the probability of having full/high levels of protection in MPAs decreases.

Probability of full/high levels of protection in MPAs is higher for low levels of land-based pressure, and steadily decreases when values of land-based pressures approximately exceed the mean of land-based pressures in our dataset ([Figure 6.4B](#)). Pattern is reversed for unmanageable pressures, suggesting a higher probability of having full/high levels of protection when levels of unmanageable pressures (pressures related to climate change) are high.

Country predictors have a large influence in the variability of the model, dominated by the integrated dependency of countries to marine resources, with countries having a higher dependency exhibit a lower probability of setting up full/high protection in their MPAs ([Figure 6.4B](#)). On the other hand, the percentage of GDP used for environment protection and management (expenditures predictor) is positively correlated with the probability of having higher levels of protection in MPAs. Even if having a little relative explicative power, predictors about countries fisheries show a higher probability of full/high levels of protection in MPAs for countries having large amount of catch, but when catches are not decreasing over time.

Finally, despite the fact that biodiversity related predictors have a weak relative influence in our model, relationships between the two predictors and the probability of full/high protection in MPAs is positively shaped, showing a slight trend in having higher levels of protection in MPAs with higher levels of biodiversity ([Figure 6.4B](#)).

6.5 Discussion

Higher levels of protection in new or existing MPAs are required to efficiently protect the ocean. This study explores what drivers explain the presence of full/high levels of protection inside MPAs, using predictors describing biodiversity, human pressures, MPAs and countries characteristics. The resulting identification of levers and barriers towards higher levels of protection can inform their future implementation in new or existing MPAs, and bring insights on how effectively protect the marine environment.

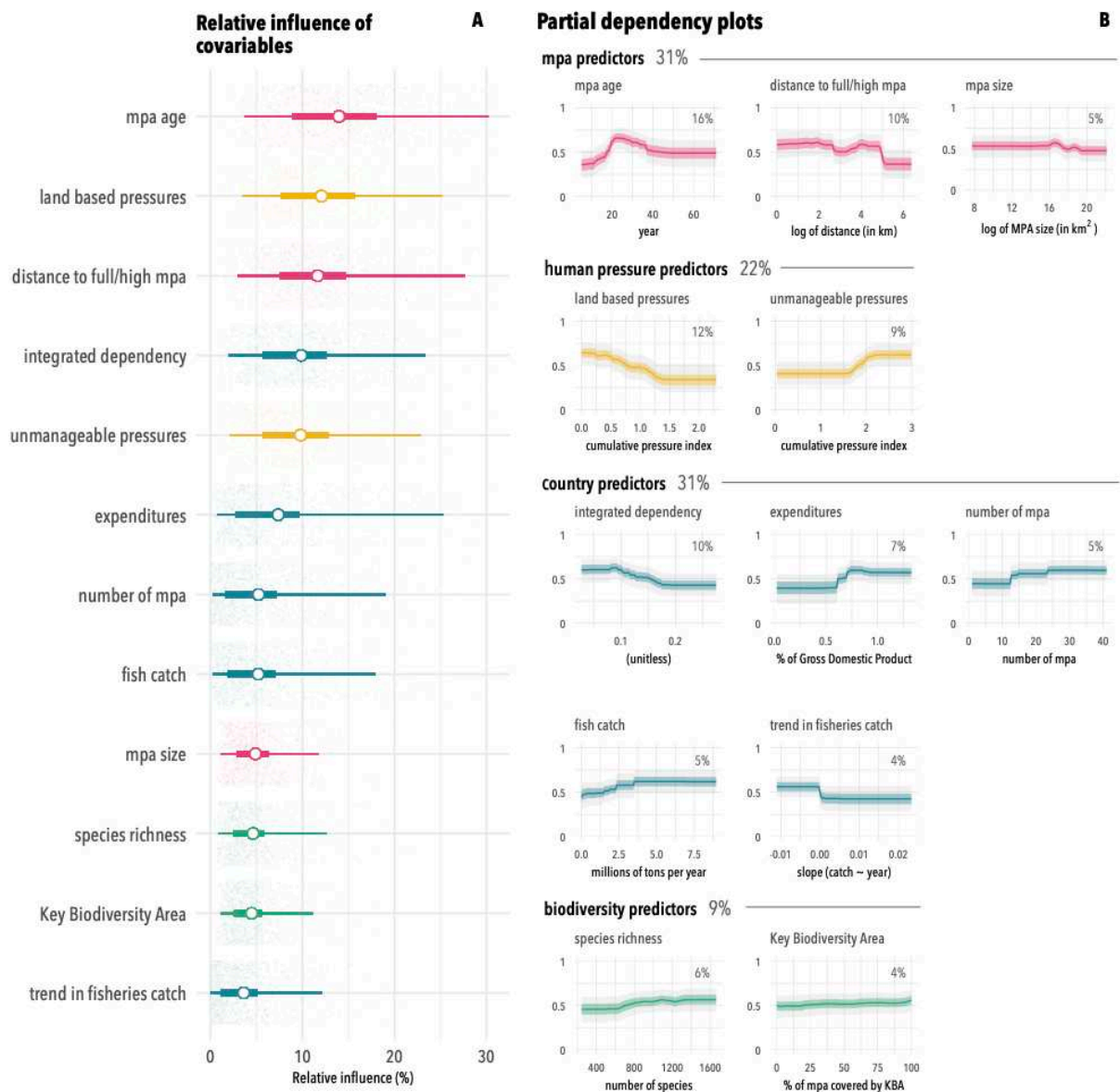


Figure 6.4. Outputs of the final boosted regression trees model, with colors according to the type of predictors (MPA related: pink; human pressures related: yellow; country related: blue; biodiversity related: green)

Panel A shows the relative influence of predictors (dots represent the mean relative influence over the 1,000 bootstrap replicates, large bars represent the median quartiles distribution, thin bars represent the 95% confidence interval). Panel B shows the partial dependency plots with the 50% confidence intervals colored and the 95% confidence interval in grey.

MPAs usually avoid threats and target areas of low human uses (Devillers et al., 2015; Kuempel et al., 2019), especially abatable threats such as fishing or shipping (Stevenson et al., 2020). The threat avoidance pattern is different for full/high levels of protection: contrary to global distribution of MPAs, where land-based pressures are positively related with MPA presence (Stevenson et al., 2020), full/high levels of protection is negatively related to such pressures. When looking at the global distribution of MPAs, the majority of them are coastal and therefore receive various sources of land-based pressures (Partelow et al., 2015). The avoidance of such pressures by full/high levels of protection can be explained by the fact that MPAs are not well designed to deal with them, the recognition for a land-sea integrated management is recent and requires a coordination between planning agencies solely dedicated to terrestrial, freshwater or marine realms (Álvarez-Romero et al., 2015a; Brown et al., 2019). Implementing full/high levels of protection in MPAs where places strongly impacted by land-based activities could be risky at providing short term benefices, putting in peril the credibility of managers and scientific towards public involved during the planification process, but such places cannot be left behind. Land-based pressures and the lack of coordination in the land-sea continuum shouldn't be a barrier towards higher levels of protection.

The probability of full/high levels of protection is positively related to unmanageable pressure, related here to pressures coming from climate change. MPAs can have a positive role in mitigating climate change effects, by protecting carbon-storage habitats or by constituting refuge area where ecological communities will have better chances to adapt and develop (Jacquemont et al., 2022; Roberts et al., 2017). However, climate change effects is rarely take into account in MPA planning, and MPA monitoring is not well designed to distinguish between adverse effects from local activities and those coming from climate change (Rilov et al., 2020). Moreover, the "protection provides resilience" paradigm is still debated, and the "protection paradox", defined as the fact that vulnerable species who would recover following protection measures against particular pressure is more sensitive to climate-change effects, provides a good example of the combined negative effects of high levels of protection and high exposure to climate change pressures (Bates et al., 2019). Our results highlights that full/high levels of protection in the Mediterranean Sea are subject to the highest level of climate change pressures in the area. Even if these levels of protection have been demonstrated effective to locally enhance the resilience of species and habitats, there is a need to assess and monitor if the current high and the future higher levels of climate change pressure (the mean sea surface temperature is projected to increase of 2.8°C by 2100 in world's MPAs) could jeopardize this capacity, in order to avoid a "Protection Paradox". (Bates et al., 2019; Bruno et al., 2018).

Country predictors together have a large relative influence, mainly driven by the integrated dependency of marine resources. The integrated dependency is an index built on three types of dependency: nutritional, economic and coastal protection (Selig et al., 2019). Coastal protection was not quantified for Mediterranean countries, and due to the high collinearities between the economic and the integrated dependency (Spearman coefficient of 0.76), we only kept the integrated dependency index in our analysis, which is therefore mainly driven by economy. Countries with higher dependency exhibits lower probability of implementing higher levels of protection within their MPAs. This result highlights the predominance of strategic economic sectors and blue growth in marine spatial planning compared to conservation objectives, even if

those sectors rely on healthy ecosystems that can be ensured by full and high levels of protection in MPAs (Reimer et al., 2023).

The previous area-based target for ocean protection such as the Aichi target 11 lead to a race for MPA declaration in the last decades. This is highlighted by our results, MPA age being the first explicative predictor with MPA younger than 20 years exhibit less probability to benefit from full and high levels of protection. Even if the recent target 3 of the Kuming-Montreal Global Biodiversity Framework formalize the need of higher levels of protection, perverse consequences still come along with area-based objectives (Barnes et al., 2018). Time is come to shift the focus for MPA implementation, from quantity to quality (Barnes et al., 2018; Visconti et al., 2019).

Our model explains only 28% of the probability of having full/high levels of protection within MPA, reflecting the lack of others considerations taken into account when designing full and high protection in MPAs. For example, abatable pressure predictor was not selected in the final model due to its poor explanation strength. While it is negatively correlated with the global presence of MPAs (Kuempel et al., 2019; Stevenson et al., 2020), intensity of manageable pressure brings no information on full/high levels of protection in our analysis. Fully and highly levels of protection, either within zone(s) inside MPAs or within an entire MPA, are small and mainly coastal, while data layers of abatable pressures are large scaled and represent mostly fishing effort and commercial shipping in distal waters. Our analyze potentially do not capture well the local context in human uses of the sea space at which the MPA planning has been done. Likewise, the weak observed positive relationship between biodiversity predictors and probability of full/high levels of protection also reflects the lack of finer spatial resolution in data, to take into account local biodiversity challenges. More generally, design of MPA involves local data on biodiversity (for example occurrence of emblematic species or particular habitats) and on the socio-economical context, and the process engages many stakeholders resulting in choices and trade-offs that the data we used in our analyses cannot reflect (Horta e Costa et al., 2022).

6.6 Conclusion

Even if the current coverage of national MPAs is insufficient to attain global targets and there will need the support of larger MPAs, there are the best tool for marine conservation since full and high levels of protection are only present in their borders. Full and high protection alone will prevent potential impact from human activities on marine environment, therefore addressing conservation objectives and ensuring delivery of ecosystem services for people. In an era of increasing demand for space and resources at sea, the identification of drivers guiding full and high levels of protection in MPA is necessary to understand what can favor or curb their implementation, provide insights for a better management of MPAs.

DISCUSSION GÉNÉRALE

Face à l'intensification des pressions et de l'impact exercée par les activités humaines sur les écosystèmes marin, une organisation rationnelle de ces activités est nécessaire pour endiguer la dégradation des écosystèmes sur lesquels reposent l'approvisionnement en biens et services des sociétés humaines. Les concepts récents de l'approche écosystémique (EBM, pour *Ecosystem-Based Management*) et leur prometteuse application à la planification spatiale maritime (MSP, pour *Marine Spatial Planning*) apportent des réponses à ce défi (Olsson et al., 2008). Cependant, la complexité du tissu formé par l'océan, dynamique et riche en écosystèmes variés, et les usages humains, qui interagissent avec chacun des composants de l'écosystème, à des échelles spatiales, temporelles et administrative différentes, rend difficile l'opérationnalisation de ces approches (Arkema et al., 2006; Curtin and Prellezo, 2010; Delacámara et al., 2020; Leslie and McLeod, 2007). De plus, les intérêts stratégiques de secteurs économiques historiques, comme la pêche, et émergents, comme le développement des énergies renouvelables marines, prennent le pas sur les enjeux de conservation lors de l'implémentation de MSP, démontrant la difficulté rencontrée par les différents gestionnaires à aller au-delà de la gestion traditionnelle orientée par secteur (Österblom et al., 2017; Reimer et al., 2023). Les objectifs de croissance économique (la croissance bleue, citée comme l'un des objectifs du MSP) entrent ainsi en contradiction avec d'autres objectifs liés à la conservation des écosystèmes, notamment la mise en place d'Aires Marines Protégées (AMP) présentant des niveaux de régulation suffisamment forts pour permettre la délivrance de bénéfices socio-économiques. Ainsi, les dernières décennies ont vu une forte croissance de la surface couverte par les AMP, pouvant être considérées comme résiduelles ou opportunistes, c'est-à-dire profitant de l'absence de secteurs économiques stratégiques pour être mise en place (Devillers et al., 2020), ou bien autorisant la présence d'activités humaines qui ne sont pas compatibles avec les enjeux de conservation, et qui ainsi n'apportent pas les bénéfices socio-écologiques attendus (Giakoumi et al., 2017; Turnbull et al., 2021; Zupan et al., 2018b).

Les travaux de cette thèse ont pour vocation à proposer des outils d'intégration de l'EBM dans le MSP, à évaluer les progrès du MSP vers ses objectifs de conservation, et à identifier le rôle des AMP actuel et futur dans l'atteinte de ces objectifs. L'évaluation de l'impact cumulé des activités humaines (CIA, pour *Cumulative Impact Assessment*) en est un point central, puisque la gestion des écosystèmes ne peut être centrée sur les écosystèmes eux-mêmes mais sur la gestion des activités humaines qui les impactent, pouvant compromettre l'approvisionnement en biens et services écosystémiques qui découlent de leurs fonctionnements (Ehler and Douvère, 2009; MacLeod and Leslie, 2009). La démonstration de l'applicabilité du CIA à une échelle locale, intégrant des données spécifiques au contexte socio-écologique, apporte aux gestionnaires une évaluation transparente du risque posé par l'éventail d'activités humaines présentes dans un territoire et des actions permettant d'organiser l'espace marin pour aller vers une diminution des pressions issues de ces activités, et donc une amélioration de l'état de fonctionnement des écosystèmes (Chapitre 1). Une fois ce constat posé, cette même méthode peut alors être utilisée pour répondre à un challenge fondamental du MSP : l'évaluation de ses capacités à répondre à ses objectifs de conservation. Au cours du processus de planification, les objectifs économiques, ceux de régulations des potentiels conflits entre usagers, et ceux liés à la conservation des écosystèmes sont mis en balance par différentes consultations et concertations impliquant les différents usagers, gestionnaires et scientifiques, en s'appuyant sur des données écologiques et socio-économiques (Flower et al., 2020; Horta e Costa et al., 2022; Jones et al., 2016). Ces

processus mènent à plusieurs propositions de scénario de planification spatiale maritime parmi lesquels différents compromis peuvent être adoptés, souvent – comme on l’a vu plus haut - au détriment des objectifs de conservation. Nous proposons ici d’évaluer *a priori* ces différents scénarios, en se concentrant sur ces objectifs de conservation, par leur capacité à réduire l’impact cumulé au sein du territoire considéré, en s’appuyant sur l’évaluation initiale qui ne prend aucune mesure de gestion en compte comme point de référence (Chapitre 2). Les résultats permettent de quantifier la contribution de différents scénarios de planification à réduire les pressions qui agissent sur les écosystèmes par des régulations adaptées, et apportent aux différents gestionnaires un outil pour replacer les objectifs de conservation au cœur du processus de planification.

La planification spatiale maritime repose sur la définition de zones définies géographiquement, et les AMP s’y intègrent en tant que réponses aux objectifs de conservation. Cependant, et ce malgré l’engagement des différents états à travers le monde à des surfaces minimales à atteindre, la prédominance des secteurs économiques et/ou de considérations sociales et/ou culturelles entraînent la mise en place d’AMP partiellement protégées, où différents types d’activités peuvent être autorisées et ainsi compromettre les bénéfices attendus de la protection. Par l’utilisation d’un système de classification des AMP basé sur les activités autorisées ou non en leur sein, nous avons évalué le niveau de protection réel des AMP en Mer Méditerranée (Chapitre 3) et dans les eaux françaises, en métropole et outre-mer (Chapitre 4) . Ce système de classification, dont les différents niveaux sont reliés aux potentiels bénéfices écologiques attendus (Zupan et al., 2018b), nous a permis de démontrer l’échec de l’atteinte des objectifs fixés pour 2020, que ce soit en terme de surface, de représentativité écologique, ou de niveaux de protection suffisamment élevés pour garantir des bénéfices socio-écologiques, remettant ainsi en cause leur rôle au sein du MSP dont elles sont un outil préférentiel pour la conservation des écosystèmes. La quinzième conférence des parties de la Convention sur la Biodiversité Biologique (COP 15) a adopté en 2022 le cadre mondial pour la biodiversité (le *Kumming-Montreal Global Biodiversity Framework*), constitué de 4 grandes orientations pour 2050 et de 23 objectifs basés sur des actions concrètes, dont l’objectif 3 qui stipule que 30% des océans doivent être efficacement conservés et gérés par des régulations portant sur des activités compatibles avec les enjeux de conservation (Convention on Biological Diversity, 2022). Dans cette même optique, l’Union Européenne a adopté sa stratégie pour la biodiversité d’ici à 2030, qui explicite également l’objectif de 30% d’AMP en 2030, mais en ajoutant un objectif de 10% de zones strictement protégées (European Commission, 2020). Enfin, la France s’est engagée dans la même voie, par une première version de sa stratégie nationale pour la biodiversité qui devrait être finalisée à l’automne 2023, qui prévoit également de déployer 30% d’AMP sur son territoire marin, incluant 10% de protection forte. Ainsi, à travers le monde, les objectifs de surface en AMP pour la prochaine décennie intègrent les notions de niveaux de protection élevés, dont nous avons démontré leur large manque au travers de nos cas d’études. Afin d’élucider quels sont les facteurs qui influencent la présence de niveaux de protection suffisamment élevés dans les AMP, garant de bénéfices écologiques, nous avons conduit une analyse en Mer Méditerranée qui intègre des variables relatives à la biodiversité, aux pressions engendrées par les activités humaines, aux caractéristiques propres des pays dans lesquels l’AMP est présente et aux caractéristiques propres à l’AMP elle-même (Chapitre 5). Il a été démontré qu’à l’échelle mondiale, les AMP évitent les zones fortement utilisées par les activités humaines, que ce soit spatialement, par leur

implémentation dans des zones distantes, ou réglementairement, par la mise en place de régulations qui permettent un maintien des activités potentiellement impactantes (Devillers et al., 2015; Kuempel et al., 2019; Stevenson et al., 2020). Nos résultats en Mer Méditerranée indiquent que la présence de niveaux de protection élevés semble également être opportuniste et ne répond ni à des enjeux écologiques, ni à la nécessité de réduire les pressions issues des activités humaines. Considérés ensemble, les résultats des différents chapitres mettent en lumière le manque de considération des objectifs de conservation, que ce soit au sein du MSP, par un manque d'approches dédiées localement à l'évaluation de l'impact cumulé des activités humaines et d'évaluation de différents scénarios de planification au regard des enjeux de conservation, et au sein des AMP, pourtant dédiées à l'atteinte de ces objectifs de conservation mais qui manquent de mesures spécifiques visant à réduire les pressions préalablement identifiés. Au-delà d'apporter ce constat, les différentes approches proposées ici permettent de replacer la biodiversité au centre de la planification de l'espace marin.

7.1 Apports méthodologiques

Les différentes approches utilisées dans ces travaux ont pour but l'opérationnalisation des principes de l'EBM au sein du MSP, centrées sur l'identification et l'évaluation du risque posé par les activités humaines aux écosystèmes. L'évaluation de l'impact cumulé permet de cartographier la distribution spatiale des pressions issues des activités humaines, et d'estimer leurs impacts sur les différents composants de l'écosystème (habitats physiques, espèces, communautés). La collecte des données écologiques et socio-économiques est un processus clé dans la planification spatiale maritime en général, puisque c'est la disponibilité de ces données qui va permettre la définition du contexte local, l'évaluation de l'impact cumulé et l'analyse de potentiels conflits entre usagers (Ban et al., 2013; Foley et al., 2010; Menegon et al., 2018). La qualité et la quantité de ces données influent sur la qualité des analyses et de leurs produits, et sur l'incertitude des résultats liée à la méthodologie (Hodgson and Halpern, 2019; Stock and Micheli, 2016). Par exemple, un facteur de qualité des données utilisées est leur résolution, qu'il s'agisse des données écologiques ou des données permettant la modélisation spatiale des pressions humaines (Halpern and Fujita, 2013). L'évaluation de l'impact cumulé des activités humaines telle que pratiquée (Korpinen and Andersen, 2016) informe sur les sources et la répartition de l'impact dans l'océan à des échelles mondiales ou régionales, mais peu d'exemples existent à échelle locale, échelle à laquelle les décisions de gestion sont discutées et appliquées. L'essence du CIA, intégré au MSP et à l'EBM, est de donner des informations sur la localisation et l'intensité des pressions humaines, appliquées sur des composants de l'écosystème plus ou moins vulnérable à ces pressions, afin d'informer la gestion. Si la résolution spatiale des composants écologiques est trop faible, l'utilité de l'information pour les gestionnaires et usagers est discutable (Halpern and Fujita, 2013). Par exemple, une récente analyse mondiale de l'impact cumulé sur les récifs coralliens utilise une résolution de 5 km², apportant ainsi des informations globales sur l'impact humain dans les récifs coralliens, mais ne pouvant informer une gestion locale (Andrello et al., 2021). D'autres travaux, dans les eaux portugaises et dans la mer Baltique, utilisent une résolution de 25 km², qui peut s'avérer suffisante pour des écosystèmes situés au large du plateau continental, mais ne capturent pas l'hétérogénéité ni des écosystèmes ni des activités humaines en zones côtières (Fernandes et al., 2017; Korpinen et al., 2012). Même une résolution de 0.04 km² (200 x 200 m), utilisée pour caractériser l'impact humain sur les récifs de coralligènes, ne reflètent pas la complexité de ces écosystèmes et des pressions spécifiques

pouvant agir dessus, comme la plongée récréative (Bevilacqua et al., 2018). Ces chiffres sont corroborés par un autre : seulement 14% de l'information citée dans les plans de gestion proviennent de sources scientifiques, suggérant un décalage entre les productions scientifiques actuelles et le besoin des gestionnaires (Devillers et al., 2020). Bien qu'informatrice et utile pour une communication publique à large échelle sur les enjeux de biodiversité, les analyses cartographiques globales portant sur comment prioriser les efforts de conservation sont critiquables dans l'information qu'elles apportent pour soutenir des efforts de gestion locale (Wyborn and Evans, 2021). L'évaluation de l'impact cumulé à Moorea, à haute résolution spatiale et utilisant des sources de données locales démontre la faisabilité d'approches locales, pertinentes pour les usagers et les gestionnaires, et pouvant agir comme support dans les prises de décisions. A l'inverse, le relativement faible pouvoir explicatif de l'analyse portant sur la présence des niveaux de protection intégral et haut au sein des AMP de Méditerranée appuie la nécessité de disposer de données locales pour caractériser les efforts actuels de conservation.

Malgré le caractère cumulatif des pressions qui s'exercent sur les écosystèmes côtiers, exacerbées en nombre et en intensité par la proximité avec les populations humaines, la gestion des écosystèmes marins s'est focalisée sur certains secteurs, par des régulations sur la pêche au sein d'aires marines protégées, par exemple, tout en se déconnectant des espaces terrestres et des activités y prenant place (Álvarez-Romero et al., 2011). Cependant, les écosystèmes côtiers, riches en biodiversité, productifs, et supportant de nombreuses économies, subissent directement des pressions issues des activités humaines sur terre, à plus ou moins grande distance, et ces pressions sont largement ignorées dans le processus de planification (Álvarez-Romero et al., 2011). Une intégration optimale des pressions issues des activités terrestres dans l'évaluation de l'impact cumulé est requise afin de pouvoir les prendre en compte dans la gestion (Brown et al., 2019). De récentes approches ont permis la modélisation de plus en plus fiables, même en étant limitée en données, de la décharge en sédiments et nutriments dans le milieu marin, permettant de passer d'une diminution linéaire de la pression issue des embouchures en fonction de la distance (Ban et al., 2010; Halpern et al., 2008b) à une modélisation prenant en compte des paramètres physique et environnementales qui, même statique, représente mieux les processus de diffusion des sédiments et polluants dans les zones côtières (Álvarez-Romero et al., 2015b; Brown et al., 2019; Jade M. S. Delevaux et al., 2018). L'adaptation de ces approches au cas de Moorea a permis de cartographier finement les pressions et l'impact issues des activités terrestres, ce qui dans ce cas s'avère crucial puisque la majorité de l'impact reçu par les différents habitats du lagon est dominé par ces activités. La combinaison de données locales sur l'occupation des sols, sur l'utilisation d'intrants chimiques, d'images satellites et d'études scientifiques portant sur les principaux bassins versants et leurs embouchures ont permis de simuler la diffusion autour de Moorea de la pression issue des activités humaines sur chaque bassin versant, confirmé par des échantillonnages in-situ opportunistes (Adam et al., 2020).

7.2 Limites méthodologiques

Pouvoir comparer les résultats d'impact cumulé estimé par une approche de modélisation avec des données empiriques sur l'état réel des composants de l'écosystème considéré est rarement décrit dans les applications existantes de l'évaluation de l'impact cumulé (Korpinen and Andersen, 2016). Dans notre cas, nous avons eu la chance (et c'est pourquoi j'emploie plus haut le terme opportuniste) de pouvoir confronter notre modélisation de l'enrichissement en

nutriments dans le lagon avec des mesures empiriques. Lorsqu'on considère l'enrichissement en nutriments du à l'agriculture et aux rejets d'eaux usées, on pose l'hypothèse que cet enrichissement peut favoriser le développement de macroalgues dans des zones où il est élevé, au détriment des coraux, résultant ainsi en un fort impact sur les communautés coralliennes (Lesser, 2021). Seulement, notre modélisation ne repose ni sur des données de concentration en nitrogène dans la colonne d'eau, dans le sédiment ou dans des tissus biologiques, ni sur un modèle hydrodynamique en trois dimensions de la circulation des masses d'eaux dans le lagon, mais repose sur des proxys telles que les surfaces cultivées et le type de cultures par bassins versants, la densité de population, et d'autres facteurs influençant la circulation de l'eau dans le lagon comme la profondeur et le vent. Nos résultats, exprimés sans unité entre 0 et 1, sous-entendent que plus la valeur est élevée, plus l'impact est fort, c'est-à-dire plus les coraux sont désavantagés par rapport aux macroalgues, avec une relation linéaire entre pression et impact. Cependant, il est possible que même dans des zones où l'indice de pression est élevé, aucun impact ne soit observable, la pression réelle, c'est-à-dire la concentration en azote dissous dans ces zones, n'étant pas assez élevée pour induire un changement de régime corail-algue. Par ailleurs, il est admis que la relation pression-impact n'est généralement pas linéaire, et que cette hypothèse sous-estime ou surestime l'impact réel sur l'écosystème (Halpern and Fujita, 2013). Le fait de pouvoir disposer de données empiriques sur l'impact réel de cet enrichissement permet de traduire effectivement un niveau de pression en impact réel sur l'écosystème, et de déterminer des seuils ou des points de bascule à partir desquels le niveau de pression devient tel que l'écosystème est effectivement impacté, dans ce cas par un changement de régime corail-algues. Malheureusement, ces données n'existent pas pour la grande majorité des couples pressions-composants écologiques. Malgré quelques exemples existant (Bevilacqua et al., 2018; Rees et al., 2023), la translation de pression vers l'impact est généralement déterminée à dire d'experts (Korpinen and Andersen, 2016), et reste soumise à de nombreuses incertitudes (Jones et al., 2018). De nombreuses études utilisent le terme « effet cumulé » ou « risque d'effet cumulé », plutôt qu'« impact cumulé », l'effet étant les conséquences d'une pression sur un composant de l'écosystème, mais qui ne sous-entend pas forcément un impact qui compromettraient sa structure et/ou son fonctionnement (Judd et al., 2015). Cette adaptation terminologique, bien que pouvant clarifier les implications de cette approche, ne permet pas de répondre aux challenges qui demeurent sur le manque de données explicitant les relations pressions-impacts, et sur l'interactions des différentes pressions entre elles.

Notre approche permettant d'évaluer la capacité de différents scénarios de planification à diminuer l'impact cumulé propose de modéliser les changements attendus dans la distribution et l'intensité des différentes activités suites aux différentes propositions de réglementations et de zonages issues des scénarios, et de comparer l'impact cumulé qui en résulte par rapport au modèle de référence, sans régulation. Même si la redistribution des activités présentent plusieurs challenges, comme par exemple le comportement des pêcheurs affectés par la fermeture de zones, il est possible de s'appuyer sur plusieurs hypothèses comme le *fishing the line effect*, c'est-à-dire le fait que les activités de pêche se déplacent de préférence le long des bordures de l'AMP (Cabral et al., 2017; Gell and Roberts, 2003), ou bien sur la préférence spatiale des pêcheurs en fonction de paramètres locaux (Thiault et al., 2017). Cependant, certaines régulations sont plus spécifiques: la diminution de la vitesse maximale autorisée pour les engins motorisés par exemple, ou les quotas imposés sur certaines espèces pêchées. Ce type de régulation est plus

difficile à traduire en terme de réduction de l'impact cumulé. Si on continue sur l'exemple de la pêche, la pression engendrée est définie dans notre cas par une indice composite qui exprime la probabilité relative qu'une zone soit pêchée. Bien que cette approche intègre quel type d'engin est utilisé, il ne prend pas en compte ni la saisonnalité de la pratique, ni les espèces visées ou capturées. De la même manière, en Europe, l'annexe 3 de la Directive Cadre sur le Milieu Marin (DCSMM) définit la pression de pêche par des « perturbations biologiques » et son impact par « l'extraction sélective d'espèces », sans notion de temporalité ou d'espèces particulières (European Commission, 2008). Bien que les définitions des couples pressions-impacts issues de la DCSMM soient largement utilisées dans différentes évaluations de l'impact cumulé (Korpinen et al., 2021; Quemmerais-Amice et al., 2020), il est difficile d'y intégrer des mesures de gestion tels que des quotas, des tailles limites de capture ou des changements dans la taille de maille des filets, portant sur des espèces particulières, ou bien sur des fermetures temporaires de certaines zones, sauf en utilisant une nouvelle fois une approche à dire d'expert qui ajoute de l'incertitude (Hammar et al., 2020). La situation est similaire lorsqu'on classe les AMP selon les réglementations autorisées ou non en leur sein : seule compte l'autorisation ou l'interdiction des activités, et des mesures de gestion plus spécifiques au contexte socio-écologique de l'AMP dans laquelle elles sont mises en places n'entrent pas dans le champ de cette classification tout en pouvant avoir des effets bénéfiques sur les écosystèmes (Dudley et al., 2017; Sletten et al., 2021). Le manque de données permettant d'explicitier les couples pressions-impacts le long d'un gradient de pressions, allant d'interdiction totale des activités génératrices d'impact à des réglementations portant sur des quotas, des limitations de vitesses ou de limitations du nombre d'utilisateurs, est encore un frein à l'évaluation des mesures de conservation au sein du MSP et des AMP. Compte tenu des objectifs fixés par la CBD, par l'UE et au sein de chaque état, la planification spatiale maritime et les AMP sont vouées à se développer tout en intégrant la présence d'activités humaines compatibles avec les enjeux de conservations, entraînant un découpage de l'espace marin en plusieurs zones soumises à différentes réglementations et régulations, résultats de compromis entre les différentes parties prenantes (Gleason et al., 2013; Horta e Costa et al., 2022). La collecte d'informations sur comment les écosystèmes vont répondre à ces mille-feuilles réglementaires est cruciale pour évaluer leurs capacités à réduire l'impact des activités humaines qui s'y exercent et à maintenir leur structure et leur fonctionnalité.

7.3 Implications pour la gestion

Le chapitre 1 expose la large contribution des activités terrestres dans l'impact cumulé subi par l'écosystème coraliens de Moorea, et le chapitre 2 démontre la faiblesse relative de différents scénarios de gestion à diminuer cet impact cumulé. En effet, les réglementations agissent principalement sur les activités basées en mer, comme la pêche et les activités liées au tourisme, alors que très peu concernent les activités terrestres, qui auraient pourtant un fort potentiel dans la diminution de l'impact et pourrait maximiser les bénéfices apportées par la gestion (Halpern et al., 2009a; Tulloch et al., 2020). La continuité entre terre et mer, notamment pour les écosystèmes côtiers et insulaires, est mise en avant dans les initiatives de gestion, mais sont souvent déconnectée car leur gestion est assurée par différentes entités administrative, avec des objectifs écologiques et socio-économiques différents et parfois aux objectifs divergents (Álvarez-Romero et al., 2015a). Ces résultats sont corroborés par ceux du chapitre 5, qui suggèrent que les plus hauts niveaux de protection sont placés là où les niveaux de pression issus des activités terrestres sont les plus faibles, soulignant un évitement de ces pressions, au détriment d'enjeux

de biodiversité ou de la gestion d'autres pressions directement gérables par des actions de gestion locales. Deux stratégies semblent être ainsi dégagées : l'une, à Moorea, où les pressions issues du milieu terrestre sont prises en compte dans le premier plan de gestion de manière explicite et concrète (« Le rejet de tout système de traitement des eaux usées qu'elles soient domestiques ou industrielles est interdit dans les aires marines protégées », article 53 de l'arrêté n°410 du 21 Octobre 2004 rendant exécutoire le plan de gestion de l'espace maritime de Moorea), mais sans moyen d'opérationnaliser les mesures, ce qui peut mener à terme à des AMP inefficaces, comme démontrés dans la Grande Barrière de Corail où la couverture corallienne est plus faible dans des AMP soumises à des forts événements de décharge d'eaux issus des bassins versants comparés à des zones pêchées mais moins soumises à ces événements (Wenger et al., 2016) ; l'autre, en Mer Méditerranée, où même si les pressions terrestres ont pu être discutées localement, elles sont globalement évitées par les plus hauts niveaux de protection, constituant un aveu d'échec des gestionnaires face à ces sources potentielles d'impact, et ce malgré le fait que la gestion du continuum terre-mer est plébiscitée dans la littérature scientifique (Brown et al., 2019; Crain et al., 2009). Un exemple bien connu de ce modèle de gestion est celui de Chesapeake Bay, aux Etats-Unis, où à partir des années 1970, l'état environnemental s'est grandement détérioré notamment à cause de l'enrichissement en nutriments, provoquant la disparition progressive des végétaux subaquatiques (incluant les herbiers marins) et l'effondrement des pêcheries associés au système (Leslie, 2018). Après plus de 30 ans d'efforts de déploiement d'une gestion écosystémique, impliquant une coordination entre 16 agences fédérales et six états fédéraux américains, tout en intégrant les citoyens, les scientifiques, les gestionnaires et les décideurs politiques, les résultats parlent d'eux-mêmes : la réduction progressive de la concentration en nutriments dans la colonne d'eau a permis une augmentation de la surface des végétaux subaquatiques de plus de 300%, allant de pair avec une augmentation du nombre d'espèces présentes sur ces habitats restaurés (Lefcheck et al., 2018), et ce malgré un doublement de la population humaine vivant sur les bassins versants alimentant la baie (Leslie, 2018). Cet exemple démontre : 1) la complexité de la tâche à accomplir pour coordonner une telle gestion à l'échelle de plusieurs milliers de kilomètres carré, que ce soit en terme administratif, sociétal et scientifique, puisque ce travail s'appuie sur un programme d'échantillonnage et d'analyses sur le long terme et à grande échelle ; 2) que la tâche n'est pas impossible, et que lorsque des moyens sont mis en œuvre vers l'objectif commun d'un environnement qui permet la délivrance de services écosystémiques identifiés, les bénéfices sont indiscutables.

De plus, le manque de considération des pressions locales, et plus particulièrement de celles issues d'activités basées sur terre, peut-être vu comme une forme d'impuissance des autorités locales face au narratif du changement climatique, où seul ses effets sont responsables de la dégradation des récifs coralliens à travers le monde (Abelson, 2020). Ce discours, appuyé par une multitude d'études scientifiques, peut avoir pour effet pervers de désengager les communautés locales dans la protection de leurs écosystèmes par une forme d'aquoibonisme (« à quoi bon s'échiner à adopter des mesures de gestion locales coûteuses tant que l'on n'aura pas réglé le problème du changement climatique ? ») (Abelson, 2020). A Moorea, de nombreuses études se focalisent sur la pente externe, dont les communautés coralliennes sont plus soumises aux effets du changement climatiques ou d'événements climatiques extrêmes, comme des cyclones (Adjeroud et al., 2018; Lamy et al., 2016; Penin et al., 2007; Viviani et al., 2019). Cependant, les études intégrant des données à l'intérieur du lagon montrent une réponse moindre

des communautés qui y sont présentes à ces pressions (Carroll et al., 2017; Moritz et al., 2021; Pratchett et al., 2011), alors qu'elles répondent négativement à d'autres sources de stress, comme un changement de la qualité de l'eau ou la pêche (Adam et al., 2021; Gil et al., 2016). Le constat est similaire dans d'autres lieux : les effets du changement climatique ne sont pas les seuls responsables de la dégradation des récifs coralliens à travers le monde, et un discours centré sur ce narratif n'encourage pas une gestion des pressions directes (Abelson, 2020). A l'échelle locale, que ce soit dans le lagon de Moorea ou au sein des AMP de Mer Méditerranée, une meilleure coordination entre les entités responsables de la gestion des différents domaines (terrestre, eaux douces, marin) est nécessaire pour atteindre les objectifs de conservation des écosystèmes côtiers, par des actions de gestion courageuses et difficiles à mettre en place, mais qui localement pourront apporter davantage de bénéfices écologiques et sociétaux que la seule gestion des activités maritimes ou l'attente d'une éventuelle diminution des émissions de carbone qui limiterait les effets du changement climatiques.

Le processus de planification spatiale maritime favorise largement les secteurs économiques stratégiques, comme la pêche ou le développement des énergies renouvelables. Dans le chapitre 2, on propose de placer l'évaluation de l'impact cumulé au centre du processus. S'appuyant sur des données représentant le mieux possible le contexte socio-écologique local, cette méthode présente l'opportunité d'évaluer les objectifs de conservation du MSP par les effets possibles attendus sur les réductions de pressions sur les composants écologiques identifiés lors de l'évaluation et ouvre la voie à vision critique de différents scénarios de planification envisagés. A notre connaissance, une seule autre analyse de ce genre a été menée jusqu'à aujourd'hui, portant sur le domaine maritime suédois, à bien plus grande échelle, donc, et utilisant des données régionales (Hammar et al., 2020). De ce fait, les auteurs conseillent d'appliquer leurs résultats comme un support de décision à un niveau stratégique, nationale ou internationale, mais pas pour une évaluation ou une gestion locale. Notre approche complète donc le travail mené en Suède et opérationnalise cette approche à l'échelle locale, donnant des clés aux gestionnaires pour replacer les objectifs de conservation au cœur des discussions.

L'application en mer des principes de l'EBM par la planification spatiale maritime devrait concerner 50% des ZEE mondiales d'ici 2030 (Ehler, 2020), mais son opérationnalisation est complexe, due au large éventail des contextes locaux dans lesquels elle s'applique, au caractère dynamique et complexe des écosystèmes marins, et aux trois objectifs visés : social, économique, environnemental. L'évaluation de l'impact cumulé présente une opportunité d'évaluer les objectifs de conservation d'un MSP *a priori*, par sa capacité à diminuer l'impact dans une zone donnée. Si effectivement intégrée dans le processus de planification, l'évaluation des objectifs de conservation d'un MSP par l'impact cumulé ne présente pas de coût supplémentaire en financement et en moyens techniques, qui peuvent représenter un frein à sa mise en place (Frazão Santos et al., 2021). Idéalement, le processus de planification, dans ses étapes préliminaires, fournit plusieurs scénarios de gestion à travers différents outils (comme la planification spatiale systématique, ou l'intensité d'activités humaines peut être considéré comme le « coût » à protéger (Ban and Klein, 2009; Flower et al., 2020), qui seront discutés entre usagers et décideurs. Les scénarios retenus peuvent alors être évalués *a priori*, par leur capacité à réduire l'impact cumulé via des régulations portant sur certaines activités. La modification des couches d'informations géographiques utilisées pour l'évaluation de l'impact cumulé, en fonction des

différentes régulations issues des scénarios de gestion, permet des analyses alternatives et fournit de nouvelles cartographies de réorganisation de l'impact cumulé en fonction des régulations. La transparence et la répétabilité de la méthode est un atout pour ceux qui l'utiliseront et pour ceux qui recevront les résultats (usagers, décideurs), qui seront familiarisés avec cette méthode. Les différentes parties prenantes peuvent ainsi s'engager ensemble dans la planification, autour d'un outil transparent et connu de tous qui ne résoudra pas tous les conflits potentiels mais qui a le mérite de les poser à plat, élément indispensable dans un processus de planification (Frazão Santos et al., 2021)

Les résultats obtenus sur les deux scénarios de plan de gestion, à Moorea, différents en terme de processus de création, de régulations et d'activités concernées par ces régulations, montre que dans les 2 situations, 20% de la zone est concernée par une diminution de l'impact cumulé de 60%. Le plan de gestion actuel, critiqué par de nombreux usagers et dont la mise en place a soulevé de nombreux débats (Gaspar and Bambridge, 2008; Walker, 2001), est en passe d'être remplacé par un nouveau MSP, issu de concertations entre usagers et gestionnaires et impliquant l'ensemble des habitants de Moorea, à travers différents ateliers de cartographie participative, de réunions animées par les gestionnaires, des scientifiques et des représentants des différents groupes d'usagers, indiquant une meilleure probabilité de son application et de son respect (Wencélius et al., 2022) Par exemple, le terme d'aires marines protégées, fortement discuté par les pêcheurs lors du premier MSP et dont l'application n'a pas apporté les bénéfices attendus (Thiault et al., 2019a), a disparu lors des discussions autour du deuxième MSP, montrant la volonté des gestionnaires d'impliquer les différents comités de pêche locaux, opposés à l'idée d'AMP en faveur de pratiques traditionnelles comme le *rāhui* (Wencélius et al., 2022). L'évaluation des deux MSP par leur capacité à réduire l'impact cumulé montre que le premier plan, restrictif et peu discuté, apporte les mêmes résultats que le deuxième, issu d'un processus engageant tous les usagers et potentiellement mieux accepté. L'intégration de cette évaluation dans le processus de gestion, sur des scénarios, permettrait de dégager rapidement des préférences pour les gestionnaires, basée sur la capacité d'un MSP à répondre aux objectifs de conservation par la diminution de l'impact cumulé dans certaines zones. Cependant, les zones les plus impactées autour de Moorea sont faiblement visées par des régulations permettant de diminuer cet impact, notamment sur le récif frangeant, qui est soumis aux pressions issues des activités terrestres que les MSP successifs régulent peu. Ainsi, l'évaluation d'un MSP par l'impact cumulé permet également d'identifier les endroits où il y a un déséquilibre entre l'impact et les régulations, c'est-à-dire des zones où des régulations fortes s'appliquent alors que l'impact y est faible, situation qui est parfois observée pour les AMP, qui peuvent être placées à des endroits où elles évitent les pressions, afin de faciliter leur acceptation (Devillers et al., 2015; Kuempel et al., 2019; Stevenson et al., 2020).

Les AMP sont préférentiellement utilisés pour la conservation des écosystèmes marins, et s'intègrent dans la planification spatiale maritime comme dispositif de zonage, où la restriction totale ou partielle de certains usages permet de réduire les pressions exercées sur les écosystèmes d'intérêt. La course aux AMP provoquée par la définition d'objectifs exprimés en terme de surface de ZEE à couvrir par les États a abouti à une augmentation de la couverture en AMP dans le monde, mais de nombreuses préoccupations quant au véritable rôle des AMP dans la conservation de la biodiversité sont soulevées (Agardy et al., 2016; Barnes et al., 2018; Golden

Kroner et al., 2019; Visconti et al., 2019). En effet, seules les AMP imposant des régimes de protection strictes (pas d'usages extractifs) fournissent des bénéfices écologiques, desquels découlent des bénéfices socio-économiques (Claudet, 2012; Lester et al., 2009; Sala and Giakoumi, 2018). Les résultats en Mer Méditerranée et dans la ZEE française, la deuxième plus grande du monde et présente sur tous les océans (sauf Arctique) démontrent la tendance générale des États à déclarer de nombreuses AMP, mais dont une faible partie est effectivement protégée, avec des régulations interdisant les activités extractives et destructives. Malgré cette augmentation en nombre et en surface, la plupart des pays n'ont pas atteint l'objectif de 10% d'AMP en 2020, mais se sont tout de même engagés à atteindre 30% en 2030, avec 10% en protection stricte (selon l'Union Européenne) ou protection forte (selon la France). Le choix de ces termes – stricte ou forte – n'est pas anodin, et ne reflète pas la littérature scientifique sur le sujet, qui utilise plutôt les termes « intégrale », « totale » ou « *no-take* » pour désigner les niveaux de protection au sein desquels des bénéfices socio-écologiques sont attendus. Le cas de la France est particulièrement intéressant : à l'heure actuelle, elle a déjà rempli l'objectif des 30% de sa ZEE en AMP, et s'est donc engagée à placer 10% des AMP déjà existantes sous le régime de protection forte. Ce régime est actuellement définie par la mesure M003-NAT1B (et par décret depuis avril 2022), dans le cadre de la Directive Cadre Stratégie pour le Milieu Marin (DCSMM), par cinq critères : la protection forte doit s'appliquer sur des enjeux écologiques identifiés ; elle doit prioritairement être mise en place au sein d'une AMP déjà existante ; elle doit proposer des régulations suffisantes pour permettre de diminuer significativement ou de supprimer les pressions qui s'exercent sur les enjeux écologiques identifiés; elle s'appuie sur un document de gestion ; elle est contrôlée par un dispositif de surveillance (JORF, 2022). Elle diffère peu de la définition donnée par l'UICN de sa catégorie la plus forte, les réserves strictes, notamment par les indications données sur la gestion des pressions humaines, qui doivent être réduites voire supprimées dans la mesure M003, ou contrôlées et limitées selon l'UICN. Dans les 2 cas, une gestion des écosystèmes s'appuyant sur la diminution des pressions qui s'y exercent nécessite une évaluation préliminaire de l'impact cumulé dans la zone concernée, qui fournira une base de travail pour ensuite évaluer une potentielle réduction de l'impact par des régulations appropriés. Ces zones, qui doivent donc être placées au sein d'AMP déjà existantes, présentent une opportunité pour les gestionnaires, les scientifiques et les usagers de mettre en pratique les concepts du MSP, puisque les AMP actuelles n'ont plus seulement vocation à protéger l'environnement, mais aussi à organiser l'espace et les usages selon des objectifs sociétaux, économiques et environnementaux. L'évaluation de l'impact cumulé est au cœur de la définition de ces futures zones destinées à répondre aux objectifs environnementaux : elle permettra d'identifier les lieux particulièrement impactés, et quels composants remarquables de l'écosystème subissent quelle magnitude d'impact, afin de proposer des zones candidates et d'élaborer des scénarios qui pourront être évalués par leur capacité à diminuer l'impact au sein de ces zones.

La mesure explicitant les zones de protection forte (ZPF) indique également une liste d'activités sur lesquelles une attention particulière devra être portée car potentiellement incompatible avec le statut de protection forte, compatibilité qui sera « appréciée en fonction du contexte local et des pressions induites par ces activités », qui pourront ainsi « ne pas faire l'objet d'une interdiction » (RF, 2023). Ce champ laissé libre à l'autorisation potentielle d'activité telles que le chalutage ou au développement des énergies renouvelables au sein des ZPF, couplés au discours

porté par l'État français à la commission européenne qui a voté contre l'interdiction du chalutage de fond dans les AMP européennes, laisse entrevoir que la France parviendra effectivement à déclarer 10% de ZPF d'ici à 2030, tout en préservant les activités de pêche à grande échelle et industrielles ainsi que le développement des énergies marines renouvelables sur lesquelles elle est aussi soumise à des objectifs, illustrant ainsi la prédominance de secteurs économiques stratégiques dans la planification spatiale maritime.

7.4 Perspectives

Cette thèse s'est concentrée sur le développement et l'application d'outils utiles aux gestionnaires et décideurs, qui font face à une demande accrue pour l'espace marin et ses ressources, et qui en même temps sont responsables de sa protection. La diffusion des principes du MSP et les objectifs internationaux de couverture en AMP comme outil préférentiel pour la préservation des écosystèmes marins offrent de nombreuses opportunités pour concilier usages humain et conservation, mais de nombreux défis restent à relever.

Premièrement, les principes du MSP et les étapes de son opérationnalisation, bien qu'admises dans la littérature, sont appliqués de manière hétérogène et sont difficiles à appréhender pour les gestionnaires (Trouillet and Jay, 2021). Les outils présentés ici et basés sur l'évaluation de l'impact cumulé peuvent être le cœur du cycle de planification, une méthode autour de laquelle s'articule les différentes étapes du processus de planification (Ehler and Douvère, 2009). Les acteurs du milieu marin doivent s'engager vers une clarification de la planification, par une collecte de données locales et pertinentes, axée sur la spatialisation des composants de l'écosystème et des usages humains. Par exemple, en Mer Méditerranée, la disponibilité en données physiques et environnementales est fortement hétérogène, les pays du sud et à l'est disposant de très peu de données comparées aux pays situés à l'Ouest, et avec des résolutions spatiales qui n'autorisent pas une planification basée sur la spatialisation de l'écosystème (Levin et al., 2014). Les données disponibles et/ou collectées sont à la base de différents outils qui vont ensuite permettre de fournir des scénarios de planification, en concertation avec les différents usagers, mais elles sont aussi à la base d'outils permettant d'évaluer en amont (avant leur implémentation) ces différents scénarios sur les trois objectifs du MSP : la comparaison de l'impact cumulé entre une situation présente et celles présentées dans les scénarios permet d'évaluer les objectifs de conservation du MSP ; une analyse potentielle des conflits entre usagers, utilisant les cartographies des pressions issues des activités humaines permet d'évaluer les objectifs sociétaux ; et des analyses en cours de développement sur les bénéfices halieutiques dégagés par des restrictions totales ou partielles de certains engins de pêche permet d'évaluer les objectifs économiques (Séve, 2023).

Deuxièmement, la méthodologie d'évaluation de l'impact cumulé présente des limites bien connues, notamment la relation entre l'impact et l'état réel de l'écosystème. La majorité des publications sur le sujet utilise les seuils déterminés par l'étude globale de Benjamin Halpern pour catégoriser l'impact de faible à fort, basé sur la condition des récifs coralliens sous différents régimes d'impact à travers le monde de manière linéaire (Halpern et al., 2008b). Des études empiriques sont nécessaires pour déterminer localement la capacité de charge de chaque composant de l'écosystème, qui pourraient ainsi aider à fixer des seuils critiques d'impact à ne pas dépasser pour ne pas compromettre la structure et le fonctionnement de l'écosystème. La définition d'une capacité de charge contexte-dépendante, c'est-à-dire à la fois liée au contexte

socio-économiques, duquel sont issues les activités qui génèrent des pressions particulières, et du contexte écologique, qui intègre les particularités à fine échelles des différents compartiments de l'écosystème considéré, permettrait d'identifier des seuils pour les différentes pressions que peut accepter l'écosystème sans perdre ses fonctionnalités. Les activités responsables de ces pressions pourraient alors être visées par des mesures de régulations, afin de ne pas dépasser la capacité de charge de l'écosystème, qui compromettrait son fonctionnement et ainsi les services sur lesquels les activités humaines s'appuient.

Enfin, et de manière complémentaire avec les deux points précédents, les AMP sont centrales dans la planification de l'espace marin, comme zonage destiné à préserver les écosystèmes et à assurer la durabilité des services écosystémiques qu'ils délivrent. Les objectifs de surface fixés pour la prochaine décennie mettent l'accent sur des niveaux de protection suffisamment forts pour effectivement réguler les activités humaines qui menacent le fonctionnement des écosystèmes. La question de l'évaluation de l'impact cumulé des activités humaines à des échelles spatiales adaptées aux AMP généralement côtières et de petite taille est également centrale dans la priorisation à venir des futures zones fortement ou strictement protégées, puisqu'elles devront s'appuyer sur une identification fine des activités et de leur potentiels impacts, dans le but de délivrer des régulations qui permettront la restauration du fonctionnement des écosystèmes marins. Intégré au cours du processus de planification dans des exercices de planification spatiale systématique, ces données permettront de mettre en exergue différents scénarios d'évitement de zones à fortes pressions, ce qui constitue globalement la situation actuelle, ou au contraire de ciblage de ces zones en fonction des enjeux écologiques, mettant l'accent sur les potentiels bénéfiques socio-écologiques attendus de ce type de scénarios. Le couplage entre l'évaluation de la planification de l'espace marin et l'évaluation de l'impact économique (positif et négatif) sur les différents secteurs permettraient de mettre en avant des compromis gagnants-gagnants, entre secteurs économiques eux-mêmes, et vis-à-vis de la protection des écosystèmes marins.

ANNEXES

8.1 Article de vulgarisation sur les chapitres 1 et 2

Concilier développement humain et conservation des écosystèmes : le cas de Moorea, en Polynésie française.



1. Moorea, au milieu de l'océan Pacifique
2. Vue de Moorea en 3D, avec Tahiti au second plan, la plus grande île de Polynésie française
3. Activité de « ray-feeding » et « shark-feeding » dans le lagon de Moorea
(on attire les raies et les requins à l'aide de nourriture afin que les touristes puissent les observer au plus près)
4. Paysage corallien
5. Moorea est réputée pour la production de ses ananas qui nécessite la conversion de forêt en terres arables
6. Pêcheur sous-marin
7. Exemple de paysage sous-marin où les algues entrent en compétition avec le corail, favorisées par l'apport de matière organique et de sédiments

.....
Laboratoire CRIOBE
par Charles LOISEAU
Doctorant
et Joachim CLAUDET
Directeur de recherche CNRS
.....

Les animaux aquatiques, et tout spécialement les animaux marins (...) sont protégés de la destruction de leur espèce par l'homme. Leur multiplication est tellement rapide, et leurs moyens d'échapper à la poursuite ou aux pièges sont si grands, qu'il est invraisemblable que l'homme soit capable d'exterminer complètement l'une quelconque de ces espèces. - Jean-Baptiste Lamarck, *Philosophie zoologique* - 1809. Et pourtant à l'échelle mondiale, les derniers rapports de WWF et de l'IPBES montrent que le milieu marin n'est pas invulnérable et qu'il y a un déclin général de la biodiversité : depuis 1970, les populations de vertébrés ont diminué de 60 %, avec comme principales causes la perte d'habitats naturels, la surexploitation d'espèces, la pollution et la prolifération d'espèces envahissantes.

Même si nos modes de vie peuvent amener à penser que nous sommes désormais indépendants de notre environnement, c'est pourtant bien lui qui les rend possible, par la fourniture de ce qu'on appelle des biens et services écosystémiques, supportant des fonctions d'approvisionnement (en nourriture, matières premières, etc.), des fonctions de régulation (du climat, de la qualité de l'air, etc.), des fonctions de support (recyclage des nutriments, formation des sols, etc.) et les cultures humaines (valeurs esthétiques, loisirs, etc.).

Mais la dégradation des écosystèmes menace ces biens et services fournis par la

nature. Il est donc essentiel de concilier le développement des activités humaines avec la durabilité de ces ressources. Les écosystèmes coralliens, sur lesquels le CRIOBE travaille, sont parfaitement appropriés pour explorer la dépendance de l'Homme vis-à-vis de son environnement : on estime qu'ils abritent 32 % de la biodiversité marine, qu'ils fournissent sécurité alimentaire, revenus financiers et protection contre la submersion marine pour des millions de personnes autour du globe.

.....
" La dégradation des écosystèmes menace les biens et services fournis par la nature. Il est donc essentiel de concilier le développement des activités humaines avec la durabilité de ces ressources. Les écosystèmes coralliens, sur lesquels le CRIOBE travaille, sont parfaitement appropriés pour explorer la dépendance de l'Homme vis-à-vis de son environnement. "
.....

Moorea, petite île (134 km²) située en Polynésie française, où le CRIOBE est installé depuis 1971, appartient à ce type d'écosystème. Isolée géographiquement, elle n'échappe pourtant pas à l'intensification des activités humaines et ses conséquences : en 40 ans, la population est passée de 7 000 habitants à 18 000. Les activités économiques se développent. Des hôtels ont vu le jour sur les côtes, avec une grande variété d'activités touristiques proposées. Sur terre aussi, on convertit des sols en terres arables, les zones urbaines s'étendent et le littoral est transformé. Dans ce contexte, la pression de pêche s'intensifie, l'urbanisation et l'agriculture entraînent une pollution dans le lagon due aux rejets d'eaux usées et à l'utilisation d'engrais et de pesticides. Les activités touristiques peuvent causer des dommages directs aux coraux

et modifier le comportement d'espèces importantes dans l'équilibre de cet écosystème.

Afin de quantifier l'impact dû aux activités humaines, les chercheurs du CRIOBE ont récolté des données sur la localisation de ces activités et leur intensité. Le résultat est représenté sous forme de carte, qui offre d'un seul regard des informations sur l'impact de l'Homme sur son milieu, et qui peut, dans l'idéal, informer des prises de décision autour de projets de nouvelles infrastructures ou sur la réglementation de la pêche, par exemple. Mais il est également possible d'utiliser cet outil pour évaluer un plan de gestion dans sa capacité à réduire l'impact cumulé. Les premiers résultats à Moorea indiquent une réduction de 7 %, presque entièrement basée sur les réglementations des huit aires marines protégées (AMP) autour de l'île. Le constat est donc simple : si les règles ne sont pas respectées dans les AMP, alors la réduction de l'impact cumulé est quasi nulle et c'est tout le plan de gestion qui devient inefficace.

Il existe de nombreux exemples dans le monde où les AMP et les plans de gestion en général, ne sont pas respectés, mettant en péril la durabilité des ressources. C'est le cas à Moorea, où l'on observe du braconnage dans les zones protégées. Ce manque de conformité peut s'expliquer par plusieurs raisons : une mauvaise communication de la part des décideurs et gestionnaires et donc une mauvaise perception de la part des usagers du lagon, mais aussi un manque de moyens sur le terrain au quotidien. La révision du plan de gestion de Moorea a permis d'impliquer tous les acteurs dans les processus de réflexion, d'élaboration et de décision, avec comme conséquence attendue que les réglementations soient mieux respectées, autorisant ainsi des bénéfices en termes écologiques et sociétaux.

A NEW GLOBAL CLASSIFICATION SYSTEM FOR MARINE PROTECTED AREAS

A regulation-based classification for MPAs to assess how well they can perform.
A system that classifies MPAs as well as each MPA zone individually, is globally applicable and unambiguously discriminates the impacts of uses.



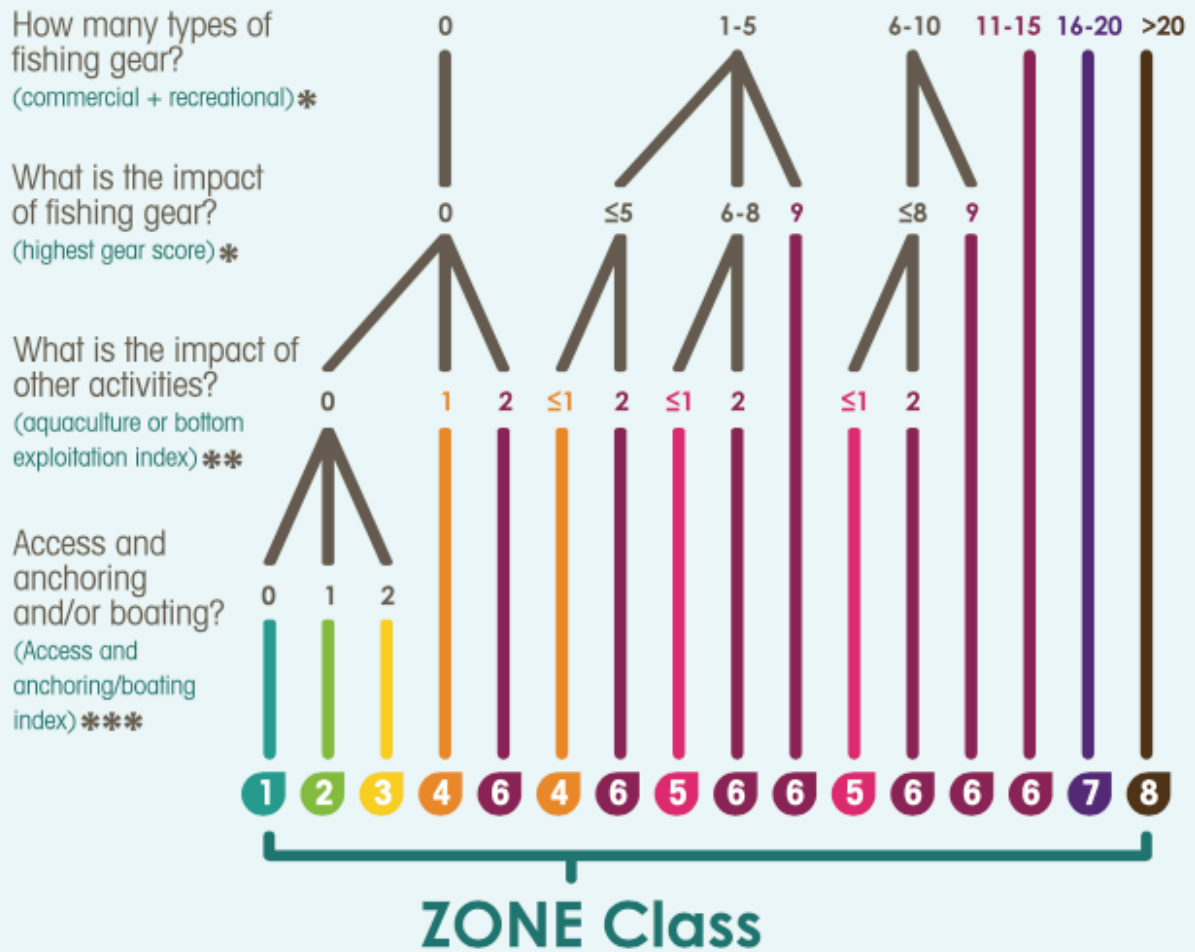
Classify your MPA at classifypmas.org

Figure 8.1. The regulation-based classification for MPAs

A Regulation-Based Classification System for Marine Protected Areas (MPAs)

Based on: Harfa e Costa et al. Marine Policy. DOI: <http://dx.doi.org/10.1016/j.marpol.2016.06.021>

Classification System of Zones within MPAs (a decision tree)



ZONE Classification

- 1** No-take/No-go
- 2** No-take/Regulated access
- 3** No-take/Unregulated access
- 4** Highly regulated extraction
- 5** Moderately regulated extraction
- 6** Weakly regulated extraction
- 7** Very weakly regulated extraction
- 8** Unregulated extraction

A Regulation-Based Classification System for Marine Protected Areas (MPAs)

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Classification System of MPAs

ZONE Class 

Next stage: how to classify MPAs

An MPA index is calculated based on the area each ZONE Class occupies within the MPA

$$\text{MPA index} = \text{SUM} \left(\text{ZONE}_i \text{ Class} \times \frac{\text{Area ZONE}_i}{\text{Area MPA}} \right)$$

Example of a multiple-use MPA with 3 zones (and corresponding zone classes) occupying different areas



EXAMPLE

MPA with 100 ha of total area

15ha class 1 + 35ha class 5 + 50ha class 8

$$\text{MPA index} = \left(1 \times \frac{15}{100}\right) + \left(5 \times \frac{35}{100}\right) + \left(8 \times \frac{50}{100}\right) = 5.9$$

MPA index

MPA Classification

1 to 3 *incl.*



FULLY PROTECTED

3 to 5 *incl.*



HIGHLY PROTECTED

5 to 6 *incl.*



MODERATELY PROTECTED

6 to 7 *incl.*



POORLY PROTECTED

7 to 8



UNPROTECTED

A Regulation-Based Classification System for Marine Protected Areas (MPAs)

Based on: Horta e Costa et al. Marine Policy. DOI: <http://dx.doi.org/10.1016/j.marpol.2016.06.021>

Indices of Activities (supporting information for the decision tree)

* Highest fishing gear score

Fishing gear (commercial or recreational)	Gear score
Beach seines	8
Cast nets	3
Dredges (bivalves)	7
Drift nets	5
Fish traps	6
Fixed fish traps "madraque"	6
Gillnets	6
Hand dredges (bivalves)	5
Intertidal hand captures	3
Lines (jigs, hooks, pole and line, rod, troll)	5
Longlines (bottom)	5
Longlines (pelagic)	4
Purse seining (bottom)	9
Purse seining (pelagic)	5
Spearfishing/diving	3
Surrounding nets near shore	8
Trammel nets	8
Traps (lobster/octopus/crab)	4
Trawl (bottom)	9
Trawl (pelagic)	5

≤ 5 corresponds to highly selective and low impact gears (e.g. lines, octopus traps);
 6-8 medium impacting gears (e.g. fish traps, bottom longlines, pelagic towed gears);
 9 to the most destructive gears affecting biodiversity and ecosystems (e.g. bottom trawling, bottom purse seining)

** Aquaculture or bottom exploitation index

Activities allowed	Index
Aquaculture and bottom exploitation not allowed	0
Aquaculture OR bottom exploitation allowed, but not mining/oil platforms/sand extraction/detonations	1
Both aquaculture AND bottom exploitation allowed with no restrictions (or if aquaculture is not allowed but mining/oil platforms/sand extraction/detonations are)	2

*** Recreational Access index

Activities allowed	Index
No access	0
Fully regulated access including boating and/or anchoring allowed but fully regulated, restricted to particular areas or mooring buoys	1
Boating and/or anchoring allowed but partially regulated or unregulated	2

Contact us: classifympas@gmail.com

8.3 Supplementary data for chapter 3

Table 8.1. Protected area (km²) by protection level and per country

Country	Member of EU	coastal and marine area	Protected area								
			Number of MPAs	Number of zones	Total	Fully Protected	Highly Protected	Moderately Protected	Poorly protected	Unprotected	Not Reported
Albania	No	11152.0	9	14	161.9	101.07	0	0	0	31.15	29.67
Algeria	No	128905.5	4	6	220.5	21.82	0	0	0	197.11	1.52
Croatia	Yes	55866.5	267	304	5876.8	161.63	250.2	0	0.54	0	5464.42
Cyprus	Yes	98118.5	8	8	129.7	0	0	0	0	15.49	114.17
Egypt	No	169411.6	6	6	485.0	0	0	0	0	0	485
France	Yes	88565.8	85	102	54505.0	79.127	15.27	122.27	101.86	51459.761	2726.76
Gibraltar	No	426.3	1	1	54.9	0	0	0	0	54.85	0
Greece	Yes	494594.5	149	160	8382.8	408.264	75.293	0	18.13	3655.89	4225.26
Israel	No	27753.2	10	10	26.5	0	0	0	0	0	26.5
Italy	Yes	539059.9	279	414	42002.9	431.75	1957.65	644.03	303.31	35922.01	2744.17
Lebanon	No	19265.9	2	2	39.9	4.17	0	0	0	35.69	0
Libya	No	357297.6	2	2	300.3	0	0	0	0	0	300.32
Malta	Yes	55417.4	30	30	3479.6	12.98	0	0	0	0	3466.61
Monaco	No	283.2	3	3	283.1	0.02	0.23	0	0	282.889	0
Montenegro	No	7466.1	2	2	28.1	0	0	0	0	28.1	0
Morocco	No	18776.7	2	2	278.9	0	0	0	0	211.5	67.39
Slovenia	Yes	193.2	15	17	12.1	0.12	2.14	0	0	0	9.8
Spain	Yes	261168.8	173	226	31628.1	121.723	323.42	524.68	45.066	14742.8167	15870.43
Syria	No	10194.2	2	2	15.4	4.24	0	0	0	0	11.14
Tunisia	No	100550.9	4	4	110.0	0	0	55.47	0	0	54.52
Turkey	No	72414.9	12	31	3123.7	175.93	1557.66	0	0	222.1	1168.04
TOTAL		2516882.5	1062	1346	151145.1	1522.8	4181.9	1346.5	468.9	106859.4	36765.7

Table 8.2. Protected area (km²) by protection level and per eco-region

Ecoregion	Ecoregion area	Number of MPAs	Number of zones	Protected area							
				Total	Fully Protected	Highly Protected	Moderately Protected	Poorly protected	Unprotected	Not Reported	
Adriatic Sea	353081.5	340	388	7416.8	168.03	461.57	0	0.54	202.19	6584.47	
Aegean Sea	610233.3	127	153	8394.51	576.18	1557.66	0	18.13	2844.33	3398.21	
Alboran Sea	352546.6	42	55	7048.45	51.553	23.16	96.52	0	2649.237	4227.98	
Ionian Sea	455011.8	107	128	6788.707	131.714	412.013	0	0	1120.21	5124.77	
Levantine Sea	1118761.7	32	32	1180.84	8.41	0	0	0	130.19	1042.24	
Tunisian Plateau/Gulf of Sidra	1077854.3	7	15	986.77	0.79	39.85	0	0	0	946.13	
Western Mediterranean	1383324.4	405	572	119234.483	565.567	1687.61	1249.93	450.236	99839.22	15441.92	

8.4 Poster « La mer Méditerranée et ses AMP pas tellement protégées... »

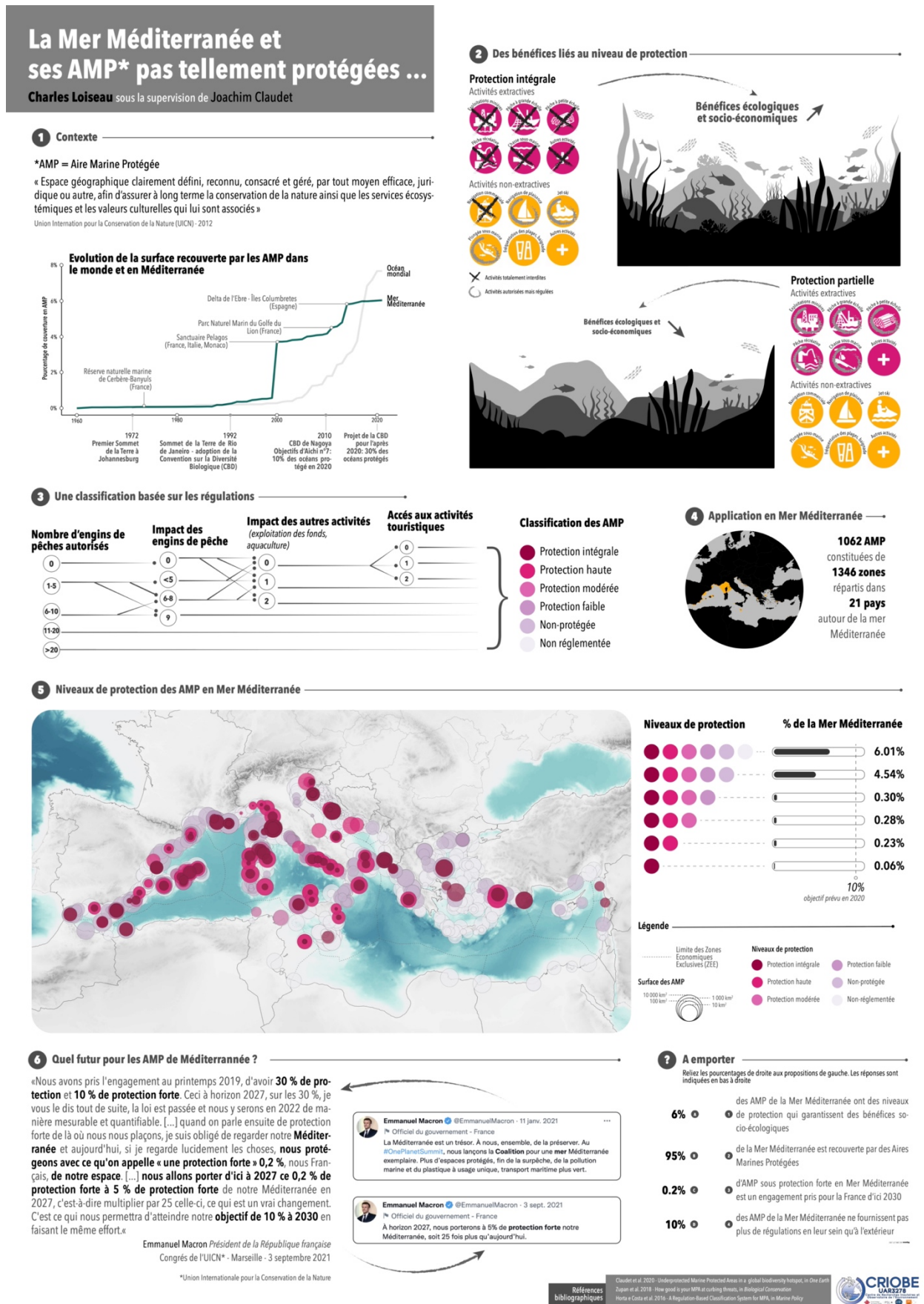


Figure 8.2. Poster « La Mer Méditerranée et ses AMP pas tellement protégées... »

Présenté à l'occasion du Colloque de Collioure « Méditerranée, climat et risques environnementaux », du 22 au 25 juin 2022

8.5 Supplementary data for chapter 4

Table 8.3. Protected area (km²) by protection level and per French Territory

	Surface EEZ (km ²)	Number of MPA	Number of zones	Protected area							Not Regulated (km ²)
				Total (km ²)	Fully (km ²)	Highly (km ²)	Moderately (km ²)	Poorly (km ²)	Unprotected (km ²)	318 137.9	
Total	10 193 037.00	524	564	3 430 387.3	160 015.9	993.8	1 534 725.2	1 304 751.4	318 137.9	111 763.2	
Metropolitan France	345 224.00	340	358	153 598.5	94.5	19.9	140.5	6 921.0	61 443.4	84 979.2	
Atlantic/Channe/North Sea	257 636.00	256	259	101 860.7	12.4	7.9	11.4	10.6	19 393.8	82 424.6	
Mediterranean Sea	87 588.00	84	99	51 737.8	82.1	12.1	129.1	6 910.4	42 049.6	2 554.5	
France overseas	9 847 813.00	184	206	3 276 788.9	159 921.3	973.8	1 534 584.8	1 297 830.3	256 694.5	26 784.0	
Atlantic Ocean	288 771.00	61	71	144 969.3	172.4	25.8	-	2.6	144 202.2	566.3	
Guadeloupe	91 013.00	33	33	91 012.9	41.9	-	-	-	90971	-	
Martinique	47 958.00	10	12	47 958.0	1.1	3.2	-	2.6	47 951.1	-	
Saint-Barthélemy	4 193.00	2	4	4 193.1	2.5	22.6	-	-	4 168.0	-	
Saint-Martin	1 103.00	9	9	1 102.7	33.7	-	-	-	1 069.0	-	
French Guiana	132 120.00	9	12	688.5	93.20	-	-	-	43.1	552.2	
Saint-Pierre And Miquelon	12 384.00	1	1	14.1	-	-	-	-	-	14.1	
Pacific Ocean	6 904 777.00	105	112	1 356 629.3	30 194.9	945.8	10.2	1 297 820.0	3 580.4	24 077.9	
French Polynesia	4 781 060.00	47	54	4 142.8	391.1	908.5	10.2	-	2 766.6	66.4	
New Caledonia	1 422 543.00	57	57	1 350 675.1	27 992.5	37.3	-	1 297 820.0	813.8	24 011.5	
Clipperton	437 420.00	1	1	1 811.4	1 811.4	-	-	-	-	-	
Wallis and Futuna	263 754.00	-	-	-	-	-	-	-	-	-	
Indian Ocean	2 654 265.00	18	23	1 775 190.3	129 554.0	2.2	1 534 574.6	7.7	108 911.9	2 139.9	
Mayotte	66 990.00	11	11	66 990.0	-	2.2	28.0	-	66 956.0	3.8	
La Réunion	315 982.00	1	4	35.1	2.0	-	9.6	7.7	15.8	-	
French Southern and Antarctic Lands	2 271 293.00	6	8	1 708 165.2	129 552.0	-	1534537.00	-	41 940.1	2 136.1	

Table 8.4. Protected area (km²) by protection level and per marine habitats

EUNIS	Habitats area	Protected area						Not regulated	
		Total	Fully	Highly	Moderately	Poorly	Unprotected		
Metropole									
French Atlantic, Channel and North sea									
A1	Littoral rock and other hard substrata	386.47	272.40	-	2.05	0.53	0.19	153.65	115.97
A2	Littoral sediment	1 253.95	972.20	1.43	32.11	2.88	8.13	492.35	435.30
A2.3	Littoral mud	358.73	346.71	1.48	-	26.28	1.63	225.18	92.14
A3.1	Atlantic and Mediterranean high energy infralittoral rock	795.38	588.30	-	-	1.84	-	440.06	146.40
A3.2	Atlantic and Mediterranean moderate energy infralittoral rock	1 020.60	573.55	-	0.94	-	-	310.20	262.41
A3.3	Atlantic and Mediterranean low energy infralittoral rock	589.24	305.19	-	-	0.00	-	73.70	231.50
A4.1	Atlantic and Mediterranean high energy circalittoral rock	405.26	167.07	-	-	-	-	167.07	-
A4.2	Atlantic and Mediterranean moderate energy circalittoral rock	1 370.04	506.67	-	-	-	-	383.96	122.72
A4.3	Atlantic and Mediterranean low energy circalittoral rock	2 531.52	846.38	-	-	-	-	662.09	184.29
A5.1	Sublittoral coarse sediment	54 318.18	11 731.06	-	0.05	-	-	5 102.20	6 628.81
A5.2	Sublittoral sand	56 693.69	14 874.41	-	0.69	-	-	6 156.17	8 705.56
A5.3	Sublittoral mud	5 478.09	2 476.91	-	-	11.99	-	1 515.79	948.89
A5.4	Sublittoral mixed sediments	44 074.73	11 597.10	-	-	12.23	-	3 690.35	7 906.75
Mediterranean Sea									
A3.1	Atlantic and Mediterranean high energy infralittoral rock	104.78	94.30	2.05	0.65	0.08	11.98	68.00	11.55
A4.2	Atlantic and Mediterranean moderate energy circalittoral rock	260.73	121.46	0.37	-	-	29.64	84.15	7.29
A5.1	Sublittoral coarse sediment	106.06	100.16	0.70	0.01	0.02	3.12	87.39	8.92
A5.2	Sublittoral sand	763.79	693.63	0.02	0.52	0.79	23.54	256.99	411.77
A5.3	Sublittoral mud	6 787.30	2 809.59	-	1.76	-	2.13	1 366.64	1 439.05
A5.4	Sublittoral mixed sediments	12 955.38	7 366.08	10.26	2.67	0.98	758.44	6 152.11	441.63
A5.5	Sublittoral macrophyte-dominated sediment	897.39	813.01	4.71	0.07	1.09	105.88	680.29	20.97
A6.2	Deep-sea mixed substrata	2 143.49	1 128.36	-	-	-	6.59	1 119.30	2.46
A6.3	Deep-sea sand	3 928.39	2 297.42	-	-	-	1 156.11	1 137.44	3.87
A6.4	Deep-sea muddy sand	5 567.90	3 955.62	-	-	-	464.07	3 491.55	-
A6.5	Deep-sea mud	113 598.78	32 433.14	-	-	-	5 262.06	27 095.33	75.76
A6.6	Deep-sea bioherms	43.43	40.00	-	-	-	27.47	12.52	-
Overseas									
/	Guadeloupe	113.30	113.30	6.50	-	-	-	106.80	-
/	Martinique	72.00	72.00	0.90	-	-	-	71.10	-
/	Saint-Barthelemy	10.70	10.70	1.40	7.90	-	-	1.40	-
/	Saint-Martin	6.20	6.20	5.20	-	-	-	1.00	-
/	Mayotte	294.26	300.00	5.60	0.30	4.60	-	283.76	-
/	Reunion	12.12	12.12	2.00	-	-	0.50	-	-
/	French Southern an Antarctic lands	113.9	31.86	26.04	-	-	-	-	5.82
/	New-Caledonia	4 576.83	4 574.80	1 786.00	3.40	-	2 143.50	252.70	389.20
/	Clipperton	4.30	4.30	4.30	-	-	-	-	-
/	French Polynesia	3072.09	540.40	16.20	91.00	2.00	-	388.70	42.50
/	Wallis & Futuna	410.99	0.00	-	-	-	-	-	-

Marine Protected Areas in metropolitan France and overseas territories

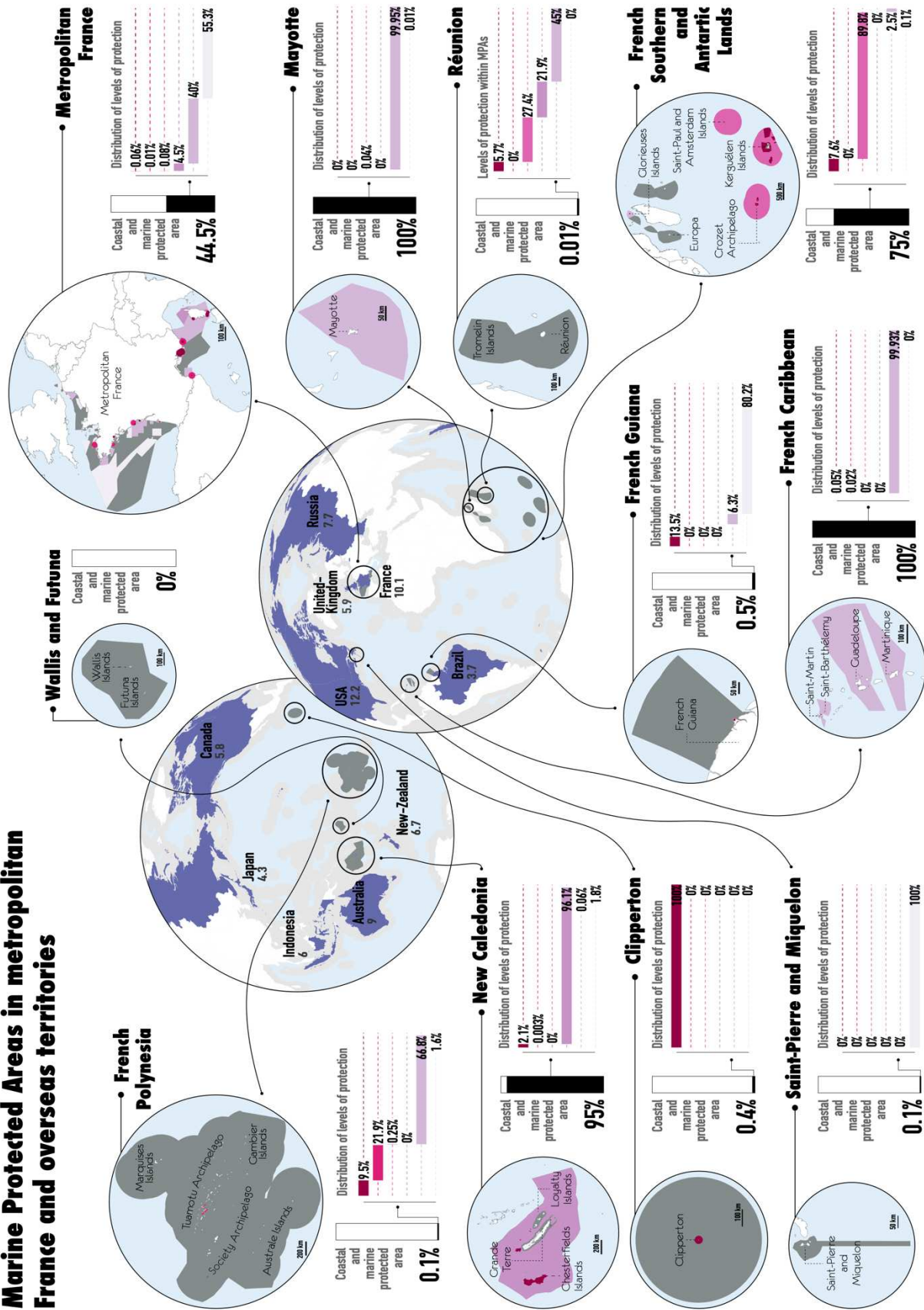


Figure 8.3. Visualisation scientifique de la protection marine en France et Outre-mer (voir chapitre 5)

Les pays en bleu foncé sur la mappemonde centrale sont les 10 pays ayant les plus grande Zones Économiques Exclusives (ZEE), dont la taille est indiqué sous leur nom (en millions de km²). Le code couleur pour les niveaux de protection est le même que celui des chapitre 3 et 4.

Protéger fortement ses océans, un défi pour la France

De vastes zones maritimes devront être classées pour atteindre l'objectif fixé par le gouvernement

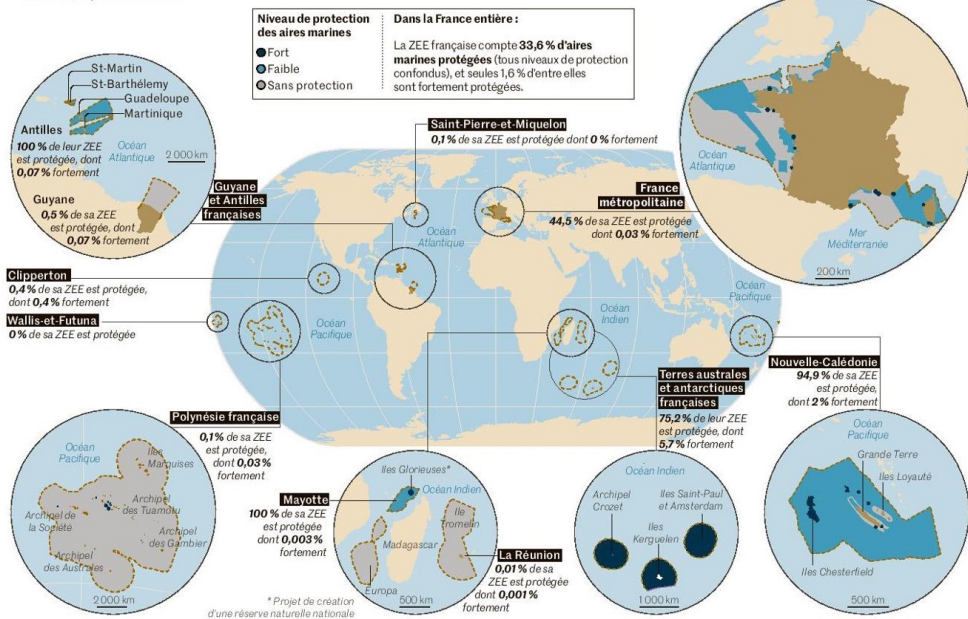
Emmanuel Macron l'avait dit sur le perron de l'Élysée dès le printemps 2019, puis répété sur la scène internationale: la France ambitionne de doter au moins 30 % de son territoire et de ses eaux sous juridiction nationale d'un statut destiné à protéger leur biodiversité à partir de 2022. La stratégie nationale 2030 rendue publique en janvier précise qu'une aire protégée se définit comme « un espace géographique clairement défini, reconnu, consacré et géré, par tout moyen efficace, juridique ou autre, afin d'assurer à long terme la conservation de la nature ». Son objectif est global: les 30 % concernent les terres et les mers ensemble. Mais il faut surtout s'attendre à y trouver de vastes portions d'océan, les terres étant davantage sous l'emprise des activités humaines.

Aujourd'hui parcs, réserves, zones Natura 2000, sites classés à des titres divers couvrent à peu près 1,5 % de terres, tandis que, selon les données officielles de 2019, les aires marines protégées (AMP) représentent 23,5 % de la zone économique exclusive (ZEE) comprise entre le littoral et 200 milles marins au moins). Une étude dans la revue *Marine Policy* (février) datée de février estime même que, depuis 2020, un tiers des eaux françaises (33,7 % précisément) ont officiellement un statut d'AMP. En outre, le plan d'action 2021-2023 qui accompagne la stratégie nationale annonce la création prochaine d'une réserve naturelle nationale de près de 47 000 kilomètres carrés autour de l'archipel des Glorieuses – qui n'avait qu'un simple statut de parc marin jusqu'à présent. Et évoque aussi l'extension de la réserve des Terres australes, dont la superficie actuelle atteint déjà 672 000 km².

« Rendez-vous manqué »
Toute la question est maintenant de déterminer quelles zones, au sein de ces espaces immenses, seront régies par une réglementation stricte qui les preserve des activités humaines néfastes pour leurs écosystèmes, interdisant donc toute forme de pêche. Car le président de la République a mis la barre haut: dans le futur réseau des aires protégées tel qu'il l'envisage, un tiers des 30 % devra bénéficier d'une protection forte. « Pour ces 10 % là, la marche est plus haute à franchir, mais nous allons y arriver, assure au Monde Béatrice Abba, la secrétaire d'État à la biodiversité. J'entends le débat autour de l'idée de protection "intégrale". Il faut sortir cette vision de la nature sous cloche. Nous voulons la réconciliation avec les activités humaines, même économiques. » Dans les faits, la France est très loin du compte. L'étude de *Marine Policy* a passé en revue 524 AMP et en a conclu

Des aires marines faiblement protégées

● Zone économique exclusive (ZEE) française (domaine maritime français et des collectivités d'outre-mer) et niveau de protection associé



seulement 1,6 % du domaine maritime français bénéficie d'un statut de protection haute ou intégrale, avec un plan de gestion, une réglementation exigeante et des moyens pour la faire appliquer. Pour 3 % d'entre elles, il n'existe même pas davantage de contraintes à l'intérieur qu'à l'extérieur de la zone concernée. Enfin, 80 % des espaces strictement réglementés se trouvent dans les Terres australes et antarctiques françaises (TAAF).

Classer de vastes zones situées à l'écart des routes maritimes et des zones de pêche les plus fréquen-

Selon une étude, seulement 1,6 % du domaine maritime français bénéficie d'un statut de protection haute ou intégrale

tes est plus simple que de convaincre l'ensemble des usagers de la mer de créer un sanctuaire près d'un littoral surpeuplé. Du coup, « les niveaux de protection sont extrêmement faibles en métropole, avec notamment 0,1 % de protection intégrale et haute en Méditerranée [alors que plus de la moitié des eaux sont incluses dans une AMP]. Et 0,01 % en Manche-Atlantique-mer du Nord », souligne Joachim Claudet (CNRS), principal auteur de cette recherche.

Les ONG dénoncent de leur côté les insuffisances de la France. Le 26 janvier, France Nature Environnement (FNE) a rendu public son bilan des campagnes de pêche dans les différents AMP du golfe de Gascogne. Elle y a comptabilisé 415 504 heures de 2015 à 2018, usant de méthodes destructrices pour les écosystèmes marins (chats de fond et pélagique, filets maillants...), soit une pêche presque aussi intense à l'intérieur qu'à l'extérieur de ces espaces « préservés ». « L'État crée des réserves dans les TAAF où ça ne gêne personne, proteste Elodie Martini-Cousty,

responsable du réseau océans de FNE. Il ne prend pas la main sur ses façades maritimes alors que les trois quarts de nos habitats côtiers sont en très mauvais état. » FNE, ainsi que Bloom, Greenpeace, MedReact, Oceana, Pew et la Fondation Bertarelli ont fait part de leur déception commune face à la stratégie nationale qualifiée de « rendez-vous manqué avec l'océan ». Elles auraient souhaité la désignation de sites sur chaque façade maritime.

Résistances
Si la France est autant soumise aux critiques, c'est parce qu'elle est très attré: elle dispose du deuxième plus vaste domaine maritime du monde, fort de plus de 10 millions de kilomètres carrés. Elle est présente dans tous les océans, sauf l'Arctique. À partir des années 2000, elle a d'ailleurs participé à l'effort mondial visant à enrayer la dégradation du monde marin. Des réserves plus gigantesques les unes que les autres ont vu le jour aux États-Unis, au Royaume-Uni, au Chili, aux Kiribati et

« L'Etat crée des réserves dans les TAAF, où ça ne gêne personne »
ÉLODIE MARTINI-COUSTY
responsable du réseau océans de FNE

aux Palaos, entre autres. Paris s'est pour sa part tournée vers le formidable réservoir d'espaces TAAF et a pu se réjouir aussi de voir aboutir le parc naturel de la mer de Corail en Nouvelle-Calédonie, l'un des plus vastes du monde.

Mais, au-delà de leur degré réel de protection, la constitution de ces espaces se heurte à bien des résistances. La cour administrative d'appel de Paris a ainsi annulé, le 13 octobre 2020, pour des questions de forme, le décret de 2018 érigeant les zones de Chesterfield, Bellona, Entrecasteaux, Astrolabe et Petri au rang de réserves naturelles intégrales. À la

suite d'une plainte d'une société locale habituée à pêcher les holothuriers (concombres de mer) à l'intérieur du parc de Corail, le gouvernement calédonien a dû se résoudre à mener une nouvelle procédure de classement.

Dans le Pacifique encore, les parages de la petite île de Clipperton abritent le poisson ange *Holocentrus limbaughi*, une espèce endémique extrêmement rare. Prisée des aquariophiles, elle est menacée d'extinction. Paris y a décidé la création d'une AMP en 2016, mais seulement dans la limite des eaux territoriales, jusqu'à 12 milles marins des côtes. Les pêcheurs sud-américains, entre autres, tiennent à continuer à capturer le thon obèse dans cette région.

Des questions diplomatiques entrent en jeu. C'est le cas avec Madagascar, par exemple, au sujet de la création de la réserve des îles Glorieuses à la place d'un simple parc marin, qui aura un impact sur les pêcheurs malgaches, mais aussi avec les collectivités d'outre-mer. Ainsi en 2018, le président de la Polynésie française, Edouard Fritch, a choisi d'englober l'intégralité de sa ZEE en une « aire marine gérée » de près de 5 millions de kilomètres carrés, remisant au passage deux projets d'AMP aux Marquises et dans l'archipel des Australes dont le dossier, soutenu par Paris, était bien avancé. Cette classification, propre au code de l'environnement polynésien, vise un développement durable: elle n'empêche pour l'heure ni l'exploitation des fonds marins ni celle des ressources halieutiques que Tahiti entend valoriser.

Edouard Fritch voudrait malgré tout obtenir une reconnaissance internationale pour son espace maritime et a demandé à Paris de porter sa demande auprès des Nations unies. En superficie, ces eaux feraient grandement progresser les objectifs mondiaux de préservation de l'océan. Les ONG craignent, elles, un dévoiement de la raison d'être des aires marines protégées. ■

M. V. MARTINE VALO

La réserve des Glorieuses abritera une biodiversité exceptionnelle

EN BRAS DE CHEMISE sur une plage paradisiaque de Grande Glorieuse, Emmanuel Macron avait annoncé en octobre 2019 la transformation du parc marin de cet archipel de l'océan Indien en réserve naturelle nationale. La stratégie française pour les aires protégées annoncée en janvier confirme cette mutation. Il ne manque plus que le décret l'officialisant.

L'intégralité des eaux territoriales des Glorieuses, situées dans le canal du Mozambique, est concernée, soit 46 077 kilomètres carrés autour de 4,3 km² de terres émergées. Le dossier est prêt, il a été soumis à une consultation publique en 2020. S'appuyant sur l'état des lieux dressé par les scientifiques, il préconise de placer près du quart de la future réserve, 10 960 kilomètres carrés précisément, sous un statut de protection intégrale, c'est-à-dire à l'abri de toute forme d'extraction, de pêche en particulier.

Les Glorieuses forment un espace naturel remarquable dans une région elle-même considérée comme l'un des trente-cinq hauts lieux de la biodiversité mondiale. On y compte des centaines de mammifères marins, d'oiseaux, de céphalopodes différents. Mais, sur les 2567 espèces qui ont été inventoriées dans ses eaux, 532 sont considérées par l'Union internationale pour la conservation de la nature comme menacées à des degrés divers.

Patrimoine mondial de l'Unesco
L'archipel est un site exceptionnel de reproduction des tortues marines vertes et des tortues imbriquées: 1 500 à 2 500 femelles y pondent chaque année, tandis que près de 300 000 couples de sternes fuligineuses y nichent. Ses plus de 450 kilomètres carrés de coraux et d'herbiers marins figurent parmi les principaux enjeux de conservation de la future réserve. Ils abritent une biodiversité qui se dif-

fuse dans toute cette région océanique. La stratégie nationale veut en outre protéger 100 % des récifs coralliens présents dans les eaux françaises d'ici à 2025.

Par le passé, des bateaux ont apporté sur l'archipel rats, chats, musaraignes, lapins. L'arrivée d'autres espèces invasives, voire de pathogènes, n'est pas exclue sous l'effet du changement climatique, mais aussi de la plaisance non déclarée. Les écosystèmes pâtissent surtout de la pêche. Même si les thoniers n'ont pas le droit de poser des dispositifs de concentration de poisson (généralment fabriqués avec des restes de filets de pêche) dans cette zone, on en retrouve échochés. Surtout, des pilotes venus de Madagascar ou des Comores y pêchent des holothuriers (concombres de mer) et des poissons de récifs dont plusieurs sont menacés d'extinction.

La gestion des Glorieuses est du ressort de l'administration qui a en charge les

Terres australes et antarctiques françaises, les TAAF. Cette entité dotée de l'autonomie administrative et financière veille sur l'archipel des Eparses – qui comprend, outre les Glorieuses, les îles Juan de Nova, Tromelin, Europa et Basas da India –, ainsi que sur Crozet, Kerguelen, Saint-Paul et Amsterdam.

L'ensemble de leurs espaces terrestres respectifs et 40 % de leurs eaux – près de 673 000 kilomètres carrés au total – sont inscrits sur la liste du patrimoine mondial de l'Unesco. Ils constituent la réserve naturelle nationale des terres et mers australes françaises, où le travail d'inventaire scientifique se poursuit afin d'identifier les frayères, lieux de nidification, et autres habitats naturels remarquables. Une étape indispensable pour préfigurer une extension de l'aire protégée et désigner en son sein de nouveaux sites de protection renforcée. ■

8.7 Supplementary data for chapter 5

Table 8.5. Country related predictors used in chapter 5. Grey shaded columns are predictors not retained in the final model.

country	fish catch (in millions of tons per year)	trend in fisheries catch (slope of linear models)	number of MPA	Expenditures (in % of GDP)	integrated dependency	proportion of EEZ protected (in %)	age of oldest MPA (in year)	MPA trend (in % of supplementary cover per year)	EEZ area (in km ²)	aquaculture production (in millions of tons per year)
Albania	87388	2.66E-03	8	0.08161153	0.101205	1.32	45	0.03	12165.4857	44854
Algeria	2439376.63	-1.82E-04	1	0.10554304	0.16188	0.012	19	0.011	131194.4601	0
Croatia	1039592.52	2.63E-04	15	0.64707262	0.149379	1.1	71	0.016	55403.3428	184397.639
Cyprus	35606	-1.06E-02	1	0.26711991	0.106333	0.006	33	0.0002	98449.1315	81355.07
Egypt	1364821	NA	5	0.10035066	0.130138	0.3	37	0.008	171290.1767	0
France	552970.75	-4.30E-04	14	0.89909575	0.102209	7.23	59	0.125	87588.2621	488953.41
Gibraltar	NA	NA	0	NA	NA			0.014	388.2777	NA
Greece	1666498.75	-3.10E-04	9	0.93001962	0.151298	0.63	45		482898.2588	2234453.2
Israel	47247	-6.61E-03	9	0.58089853	0.0284	0.11	58	0.002	25717.6398	49608
Italy	4897678.95	-1.26E-04	32	0.81507913	0.084732	0.57	38	0.015	536647.3838	1585975.688
Lebanon	71456.8	-7.67E-03	2	0.02523632	0.130735	0.2	30	0.007	20183.6237	0
Libya	795522.3	-4.88E-04	2	NA	0.162843	0.09	11	0.009	364538.4439	1834
Malta	37553.7	7.81E-03	11	1.32596482	0.183548	6.6	32	0.212	52924.6392	71528.414
Monaco	NA	NA	2	NA	NA	0.08	44	0.002	287.7111	NA
Montenegro	11851.8	2.33E-02	1	NA	0.027601	0.4	43	0.01	6383.301	4198
Morocco	699940.49	-4.47E-04	1	NA	0.275022	1.13	18	0.066	18943.3072	7781.193
Slovenia	15940.19	-1.10E-02	3	0.75271118	0.0563	1.05	32	0.034	214.3709	7917.5
Spain	2128534.05	-2.43E-04	41	0.89158453	0.198635	0.82	40	0.021	260918.9552	543699.506
Syria	52462	-5.93E-03	1	NA	0.051717	0.04	22	0.002	10269.2111	0
Tunisia	2161097.23	4.74E-04	3	0.35035865	0.231634	0.24	49	0.005	99688.4669	4362
Turkey	8957300	-5.12E-05	11	0.33681363	0.075967	3.75	56	0.068	80171.683	2035561.1

COLLABORATIONS

Operationalizing risk-based cumulative effect assessment in the marine environment

Vanessa Stelzenmüller, Marta Coll, Roland Cormier, Antonio Mazaris, Martua Pascual, Charles Loiseau, Joachim Claudet, Stelios Katasnevakis, Elena Gissi, Athanasios Evagelopoulos, Bob Rumes, Steven Degraer, Henn Ojaveer, Tiia Moller, Joan Giménez, Chiara Piroddi, Vasiliki Markantonatou, Charalampos Dimitriadis

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Review

Operationalizing risk-based cumulative effect assessments in the marine environment



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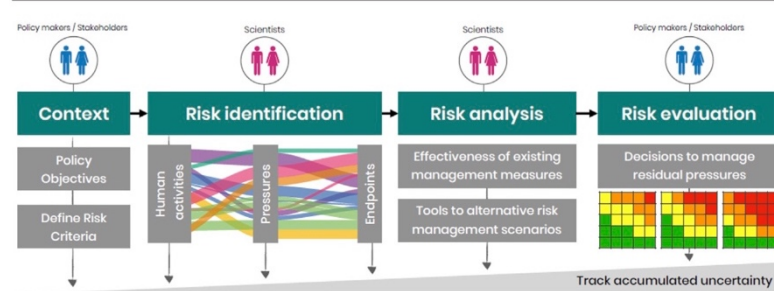
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HIGHLIGHTS

- Eleven contrasting case studies reveal key issues for the operationalization of a risk-based CEA framework
- General need to better define CEA context, risk criteria and the roles of scientists, managers and stakeholders
- Customized tools for the communication of uncertainty and trade-offs of knowledge and data are demonstrated
- Need to differentiate CEA by purpose informing either governance advice, marine spatial planning or regulatory processes
- Well-framed CEA as a strategic tool to integrate ecosystem considerations across sectors

GRAPHICAL ABSTRACT



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ABSTRACT

Ecosystem-based management requires an assessment of the cumulative effects of human pressures and environmental change. The operationalization and integration of cumulative effects assessments (CEA) into decision-making processes often lacks a comprehensive and transparent framework. A risk-based CEA

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framework that divides a CEA in risk identification, risk analysis and risk evaluation, could structure such complex analyses and facilitate the establishment of direct science-policy links. Here, we examine carefully the operationalization of such a risk-based CEA framework with the help of eleven contrasting case studies located in Europe, French Polynesia, and Canada. We show that the CEA framework used at local, sub-regional, and regional scales allowed for a consistent, coherent, and transparent comparison of complex assessments. From our analysis, we pinpoint four emerging issues that, if accurately addressed, can improve the take up of CEA outcomes by management: 1) framing of the CEA context and defining risk criteria; 2) describing the roles of scientists and decision-makers; 3) reducing and structuring complexity; and 4) communicating uncertainty. Moreover, with a set of customized tools we describe and analyze for each case study the nature and location of uncertainty as well as trade-offs regarding available knowledge and data used for the CEA. Ultimately, these tools aid decision-makers to recognize potential caveats and repercussions of management decisions. One key recommendation is to differentiate CEA processes and their context in relation to governance advice, marine spatial planning or regulatory advice. We conclude that future research needs to evaluate how effective management measures are in reducing the risk of cumulative effects. Changing governance structures takes time and is often difficult, but we postulate that well-framed and structured CEA can function as a strategic tool to integrate ecosystem considerations across multiple sectorial policies.

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1. Introduction

Over the last decade scientific effort on the categorization and description of human pressures on marine ecosystems has increased (Borgwardt et al., 2019; Knights et al., 2015) together with a better understanding of the dynamics of global patterns of human activities at sea (Allan et al., 2013; Halpern et al., 2019; Halpern et al., 2015). However, in light of increasingly rapid changes of direct and indirect pressures on biodiversity and ecosystem services, pathways towards a sustainable future still remain uncertain (Harrison et al., 2019; Lindegren et al., 2018; O'Neill et al., 2017). In particular, an improved detection of tipping points in social-ecological systems is key to prevent the coupled human-nature systems to shift into undesirable states (Bates et al., 2018; Hodgson and Halpern, 2019; Rilov et al., 2019).

Management frameworks exist that aim to explicitly avoid such undesired changes in marine socio-ecological systems. Marine ecosystem-based management (EBM) (Katsanevakis et al., 2011) or integrated marine management (Stephenson et al., 2019) can effectively inform policies to meet sustainable development goals. A sound understanding of cause-effect pathways describing the link from human pressures causing potential state changes of ecosystem components, processes or functions should form the backbone for such management frameworks (Stelzenmüller et al., 2018). Thus, despite uncertainty such a knowledge base can help to implement appropriate programs, measures, procedures, and control actions (Cormier et al., 2017; Stephenson et al., 2019).

Cumulative effects assessments (CEAs) aim to explore these causal pathways and should deliver advice for the implementation of management measures for human uses to maintain or restore ecosystem states while balancing conservation and restoration with social and economic

objectives (Cormier et al., 2019). CEAs are defined as holistic evaluations of the combined effects of human activities and natural processes on the environment and constitute a specific form of environmental impact assessments (Jones, 2016). As a consequence, CEA results can, therefore, directly inform regulatory processes (Willsteed et al., 2017), marine spatial planning (MSP) (Liversage et al., 2019; Menegon et al., 2018; Stelzenmüller et al., 2010) or the implementation of environmental policies such as the European Union Marine Strategy Framework Directive (MSFD; EC, 2008/56/EC). Although the numbers of CEA case studies are increasing in the marine realm (Korpinen and Andersen, 2016; Menegon et al., 2018; Murray et al., 2015), the formal uptake of CEA results in management processes is yet to be evidenced (Willsteed et al., 2018).

Thus, this underlines the need on guidance and best practices for the operationalization of CEA in a management context. Stelzenmüller et al. (2018) suggested a risk-based CEA framework (dividing the process into risk identification, risk analysis and risk evaluation), which structures complex analyses and facilitates the establishments of direct science-policy links, highlighting the fact that CEAs should not only be scientifically driven (see also Cormier et al., 2018). By applying standardized risk analysis along with a unified glossary and terminology (Stelzenmüller et al., 2018; Appendix A), the framework outcomes should allow, independent of the context, to address the likelihood of exceeding accepted risk of ecosystem state changes together with the potential effectiveness of new management measures. Thus, this risk-based CEA framework can support the operationalization of CEA as a strategic tool in ecosystem-based management, being an integral part of the management process, where the roles of scientists and decision-makers are clearly defined.

Here, we shed light on the challenges and opportunities of the operationalization of such a risk-based CEA at different spatial scales and in diverse settings. We identified eleven case studies in Europe, French Polynesia, and Canada at local, sub-regional, and regional management scales. In each case study, we used the framework described by Stelzenmüller et al. (2018) to identify the main outcomes and challenges for a better uptake of CEA into management and decision-making. Furthermore, we developed and applied a structured evaluation of uncertainty in CEA outcomes and facilitated its application for management advice. Based on the here compiled knowledge base, we derived some key recommendations on how to overcome the main challenges for the operationalization of the risk-based CEA framework. Ultimately, these recommendations will help scientists and managers alike to foster the dialog between key players at the science-policy interface.

2. Comparative analysis of CEA case studies

The precondition for a case study to be included in our analysis was the capability to either conduct a risk based CEA or decompose an existing CEA with the help of the risk-based CEA framework. We primarily included CEA case studies that participated in the EU MARCONS program to account for the contrasting northern and southern European conditions. Furthermore, we enlarged the geographical scope and included the island of Moorea and the Gulf of St. Lawrence cases since these cases add to the variation of context. Hence, we conducted a qualitative comparison among eleven CEA case studies (Fig. 1, Appendix A), which either used the risk-based CEA framework to structure a subsequent assessment or used it as a lens for evaluating existing CEA. For this we designed a standardized questionnaire containing thirteen open questions (see Appendix B), which were answered by each of the eleven case studies. In the following sections we present a synthesis of observed key outputs in relation to the context, knowledge, data, approaches, and outcomes of the case studies and provide corresponding

recommendations and solutions to advance the operationalization of CEA.

2.1. CEA drivers and assessment endpoints

Our comparison revealed that in most of the cases, scientists had initiated CEAs with the aim of producing meaningful results to inform a respective management context. Thus, with only one exception, management bodies or governance institutions have not commissioned such an analysis as part of e.g. marine spatial planning process. Hence, only the Canadian case was initiated by a management body in the course of the implementation of an integrated management plan. Across case studies, the CEA management context spanned from regional policies such as the MSFD, marine spatial planning processes to sectoral regulations (Fig. 2). Based on the case studies, we also observed that targeted assessment endpoints were broad, and comprised of biological entities such as species (e.g. sea turtles, dolphins), ecosystem types (e.g. coral reefs), ecosystems state (e.g. ecosystem health), and ecosystem services (e.g. sustainable resource use) (Fig. 2). The diversity of assessment endpoints and assessment scales demonstrated clearly that the proposed risk-based CEA framework is flexible and context independent.

In addition to biological components, case studies also reported assessment endpoints in relation to the effectiveness of conservation and management measures such as marine protected areas. The capacity, functioning and the achievement of a Good Environmental Status (GES) of marine waters, as requested by the MSFD, was targeted by two case studies (see Fig. 2). These CEA examples integrated an ecological state assessment with an evaluation of cost-effective management processes. Thus, choosing GES as the endpoint of an assessment requires the consideration of policy context, thresholds, and ecological state assessment. Two case studies targeted the broader effects of sectoral plans of the energy sector. Another example from the Adriatic Sea highlighted that management needs to bring together not only the

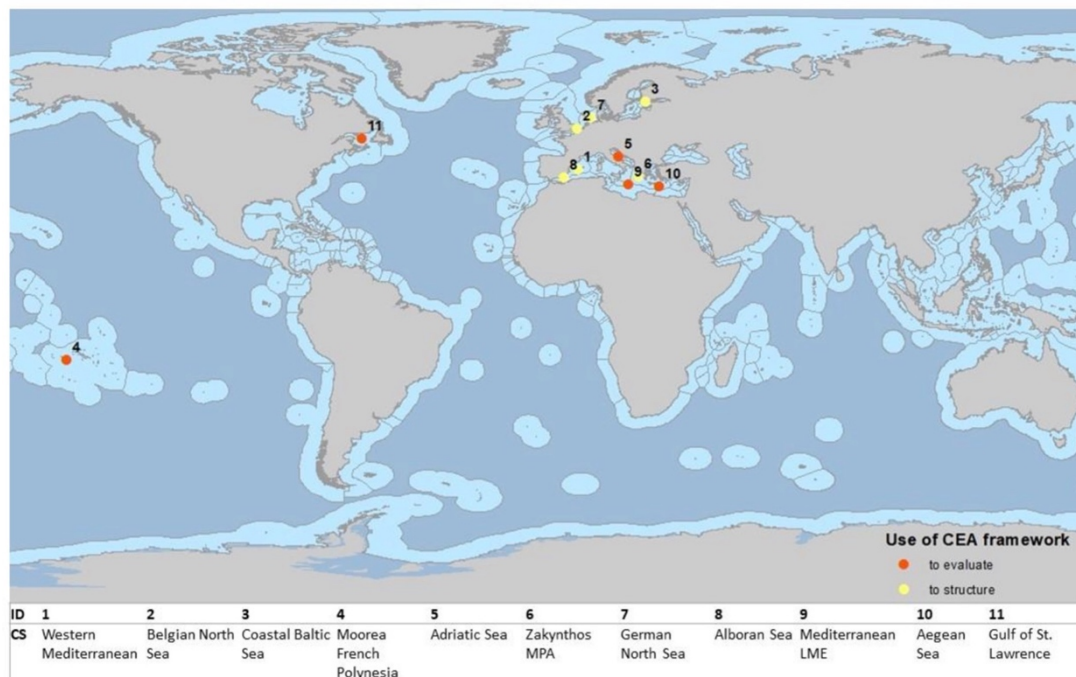


Fig. 1. Spatial distribution of the eleven local, sub-regional and regional case studies applying the risk based cumulative effects assessment framework (Stelzenmüller et al., 2018) either to evaluate an existing CEA process or to structure a CEA. Note that the exact spatial expansion of respective case study area is not shown.

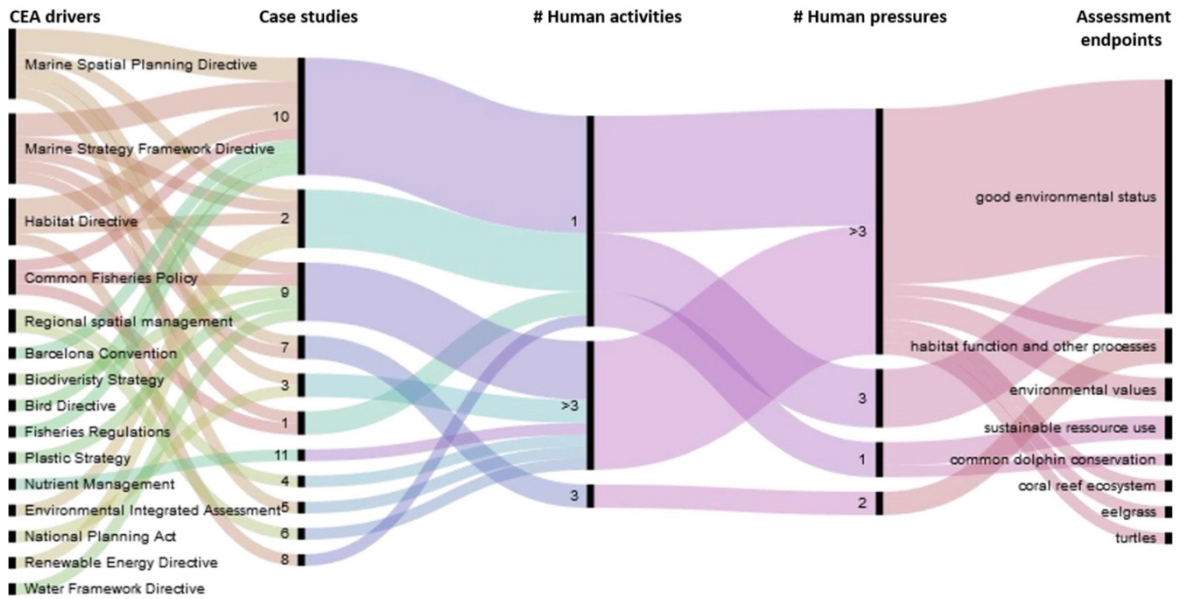


Fig. 2. Alluvial plot showing the frequencies of the relationships between the CEA drivers, number of human activities, pressures, and the assessment endpoints considered by the eleven cases studies. The width of the back nodes and colored lines is proportionally to the flow quantity (produced with RAWGraphs Visualization Platform; Mauri et al., 2017).

biological components and the relevant human pressures with their management measures, but also has to take into account the inherent complexity of the responsible authorities and sector policies (Gissi et al., 2017). The studied cases exemplified the breadth of the drivers for CEA, assessment endpoints and their envisioned strategic setting in specific management processes. This underlines the need for an integrative setting of the risk based CEA framework application between policy and science.

2.2. Establishing cause-effect pathways

Applying the risk-based CEA framework entails the initial establishment of the linkages between human activities, the corresponding pressure categories, and the effects on the respective ecosystem components, processes and functions (Stelzenmüller et al., 2018) (Fig. 3). As described by the framework this is part of the risk identification entailing also an assessment of the degree of spatial and temporal

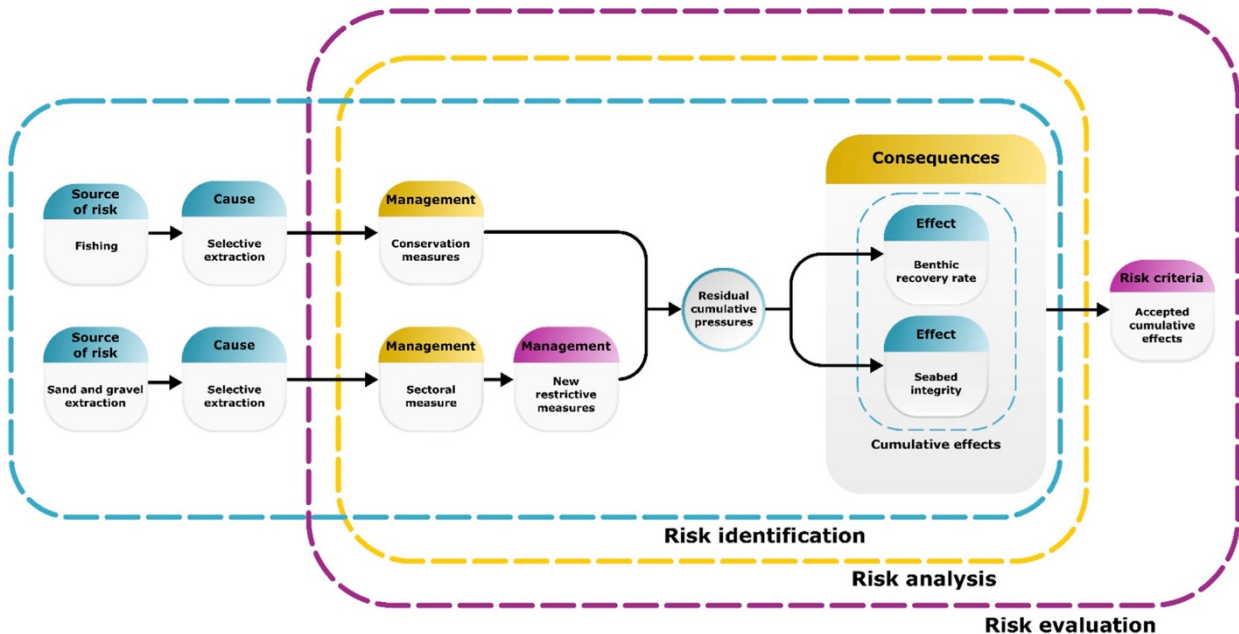


Fig. 3. General illustration of the elements and interlinkages of the steps of risk identification, risk analysis and risk evaluation. Two theoretical cause-effect pathways showing the links between two human activities (e.g. fishing and aggregate extraction), their common pressure (e.g. selective extraction), sector specific management measures (e.g. conservation measures, sectoral measure), cumulative residual pressures (total selective pressure load in the system despite management measures) and measurable state change of ecosystem components (e.g. benthic recovery, seabed recovery).

overlap of the assessment endpoint with a certain pressure. A generic pressure such as e.g. abrasion or siltation describes the actual mechanism of change or alteration to the ecosystem component (Elliott et al., 2017). Further, more than one human activity can cause the same type of pressure (Menegon et al., 2018). The general linkages between human activities and respective pressures is an established concept in environmental effects assessment (Borgwardt et al., 2019).

The total pressure load to which an ecosystem component is exposed to will contribute to its overall vulnerability. Vulnerability is a function of the exposure of the ecosystem component to a given pressure and the susceptibility of the ecosystem component to that specific pressure (Piet et al., 2015; Stelzenmüller et al., 2015a). The overall susceptibility or sensitivity of an ecosystem component to a pressure depends on its general resilience and adaptive capacity (Alliance, 2010). The assessment of vulnerabilities is the foundation for defining cause-effect pathways that in turn are necessary to prioritize human activities requiring regulation to prevent an increase of adverse effects on the system. As illustrated in Fig. 3, measurable cumulative effects are caused by the amount of pressures (referred to as residual pressures), that still exists despite management measures or restrictions implemented within management boundaries (Cormier et al., 2018). This implies that, those implemented measures can technically not reduce the pressure loads to levels, which are not causing adverse effects on ecosystem components. Despite differences in the definitions of vulnerability, we noted that this general concept has been embraced by most case studies. Further, we observed that all case studies identified general cause-effect pathways between an ecosystem component at risk and the respective pressures generated by human activities or at least identified the link to relevant human activities (see Fig. 2).

When establishing cause-effect pathways, case studies encountered difficulties with regard to both the quality of the human pressure data and the confidence in the assumed causality. For instance reported data gaps related to the description of ecosystem components and functions (e.g. species richness, pelagic compartment, benthic habitats, non-commercial species, movement patterns), occurrence of human activities (e.g. aquaculture and energy extraction plans) and their pressures (e.g., plastic pollution, noise, climate change). Furthermore, data limitations (e.g. water circulation, high resolution habitats, artisanal and recreational fisheries) held up the application of modelling tools and

introduce uncertainty in the representability of data (e.g. available information not capturing well inter-annual variation or different spatial scales). In fact, reconciling data of different geographic scales (local to regional), seasonal dimensions (spawning, secondary production) and temporal resolutions (past and current dynamics) seemed to be the main challenge for most case studies. Further, several cases have been very explicit about data needs and identified knowledge gaps that should be addressed by future monitoring schemes, research programs and initiatives aiming to provide standardized and accessible data systems.

In addition, the consideration of connectivity among the terrestrial, freshwater and marine realms, and cross-realm pressures to ecosystems was also highlighted as a challenge due to data requirements from different sectors (e.g. agricultural use of pesticides and fertilizers). This underlines the recently described gap on knowledge and research on connectivity across realms (Pascual et al., 2016). Further, not including climate change was also mentioned frequently as an important limitation of CEA case studies contributing to the increase of uncertainties in the results. These observed challenges correspond well with recent work that highlighted that uncertainty in the data resolution on human-induced pressures can have significant effects on the interpretation of cause-effect pathways and respective vulnerability assessments (Amoroso et al., 2018; Stock et al., 2018).

The key requirements for establishing cause-effect relationships are knowledge on the causality and data to support such conclusions. Both aspects introduced uncertainties in the interpretation and communication of CEA results. However, uncertainty is part of any decision making process and therefore requires a transparent and explicit handling of knowledge and data. We developed and applied a confidence matrix that facilitates a general communication of uncertainty with regard to knowledge on the causality and quality of pressure data (Fig. 4). The four quadrants of the matrix allow for a quick ranking of the produced CEA outcomes in relation to their potential use in a specific management context. We work from the premises that science advice of a CEA that underpins a regulatory process requires the highest confidence, as opposed to scientific advice for policy processes (Fig. 4). There is a greater need for confidence in the established causal relationships between activity, pressure and effect at a regulatory process since this entails technical advice on how to regulate human activities or

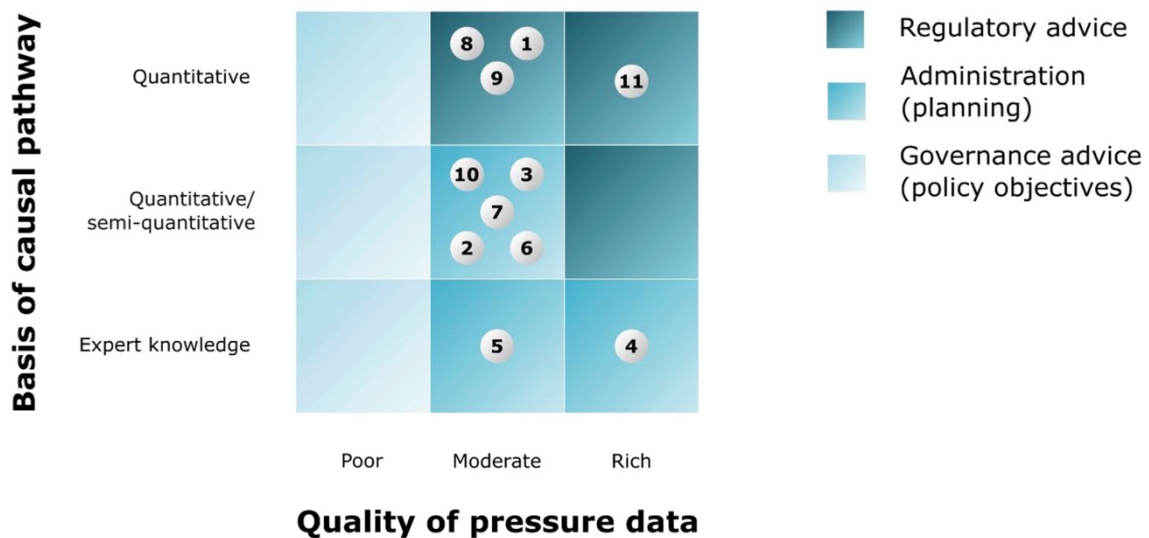


Fig. 4. Confidence matrix which ranks the quality of the pressure data as poor (spatiotemporal resolution showing a mismatch with spatiotemporal data on ecosystem components), moderate (spatiotemporal resolution showing a partial overlap with spatiotemporal data on ecosystem components), and rich (spatiotemporal resolution showing a sufficient overlap with spatiotemporal data on ecosystem components). Causal pathways can be derived from expert knowledge, semi-quantitative, or quantitative assessments; numbers correspond to the cases studies shown in Fig. 1.

requirements such as environmental quality standards. Less confidence may be sufficient in a marine spatial planning context when developing planning objectives for multiple activities. When data on pressures are of poor quality in terms of e.g. a mismatch between spatiotemporal resolutions of pressure and ecosystem components data, CEA outcomes should at most underpin strategic processes, such as the development of policy objectives. Hence, dark blue implies that a rather low level of uncertainty of scientific evidence should be provided to a regulatory process, middle blue implies that a medium level of uncertainty in scientific evidence could still underpin a planning process and that scientific results with a rather high level of uncertainty (light blue) would still be sufficient when advising the implementation of e.g. environmental policies. Fig. 4 shows how the eleven cases mapped their confidence across the quadrants. Interestingly, none of the cases studies reported a poor quality of human pressure data, indicating that most of the case studies can inform spatial planning processes.

2.3. Understanding the need of risk criteria

We noticed that most case studies were confined to the risk identification stage, where the human activities, their pressures and the respective vulnerable ecosystem components are being described. This is also in accordance with the majority of the CEA published over the last years (Korpinen and Andersen, 2016; Menegon et al., 2018; Murray et al., 2015), which mainly identified priority areas of concern for management processes. However, from the risk-based CEA framework perspective, all these cases were missing the essential ingredients that allow moving from risk identification to the analysis of effectiveness of management measures and then risk evaluation processes. Hence, they were missing the identification of the level of risk of adverse effects that would be tolerated in a given management setting. The tolerated risk should be reflected in risk criteria, such as definitions of effect sizes (e.g. defining the degree of change of an ecosystem state due to a certain amount of exposure) or thresholds in relation to acceptable levels of pressures remaining within management boundaries after considering existing management measures (see also Fig. 3). Risk criteria should not only define different levels of ecosystem state change, they should also enable assessing in the risk evaluation the overall risk of not achieving policy objectives. Hence, the CEA should be founded on established risk criteria reflecting the selected policy objectives to maximize the advice a CEA can deliver in the management process. Further, in alignment to the procedures of classical risk assessments, risk criteria should be developed prior to initiating the CEA within the context and the scope of the policies involved and in consultation with stakeholders (Rozmus et al., 2014). Without risk criteria individual personal objectives and values become the basis of debate of what is risky given the different perceptions of the level of risk and individual tolerances to risk when making a decision (Cormier and Lonsdale, 2020).

We found that, in general, case studies responded to national policies, informed marine spatial planning processes and have been well framed in the context of regional policies (Fig. 2). Some cases referred to the European Blue Growth policy (Buhl-Mortensen et al., 2017) and designed the CEA to assist the allocation of new uses while managing conflicts between them, and between uses and the environment, according to the MSP Directive (EU, 2014/89/EU). However, most of them did not mention specific risk criteria.

2.4. Accounting for the effectiveness of management measures and trade offs

Risk analysis means determining the actual consequences of cumulative effects; thus the consequences that will occur when a state change of an ecosystem component, function or process has occurred. This entails an analysis of the effectiveness of management measures that exist to regulate the pressures (Fig. 3). The case studies showed that with only a few exceptions, the existence and effectiveness of management

measures have not been considered as part of the CEA (Appendix A). Further, we detected quite some confusion across case studies on how to assess the effectiveness of management measures and how to incorporate it within the respective studies. One exception is the Western Mediterranean case study, which did consider the effectiveness of management measures. With a help of an ecosystem model the effects of different management measures were tested. Those management measures corresponded to categories of marine protected areas reflecting different levels of protection (Horta e Costa et al., 2016). The assessment of potential management measures revealed that only a high level of protection will likely be effective in achieving management objectives (Zupan et al., 2018).

A few case studies also mentioned the importance to acknowledge the complex social-ecological dimensions in a CEA, hence pointing to the fact that conflicts and trade-offs between human activities need to be analyzed in relation to the risk of cumulative effects. Hence, trade-off analysis might need to consider both positive and negative effects of pressures since some human activities may counter-balance the effects of pressures, while others may amplify them. Mechanistic models can be used to quantitatively identify such trade-offs (Christensen and Walters, 2004; Coll et al., 2008).

When cumulative effects are occurring in a given area, there are several factors at play that cannot be managed by measures taken locally. Hence, cumulative effects can also be driven by natural variability, the effect of climate change or pressures that are generated from outside the planning or management area. In the latter case, regulatory options that can address these external factors require cross-jurisdictional or cross-boundary coordination in the implementation of management measures to reduce the pressures in each jurisdiction equivalently. This makes a strong case for the recognition of climate change induced effects and their contributions to cumulative effects in regulatory frameworks for human activities and their pressures. This comprises the consideration of such external effects in marine spatial planning processes, therefore complementing conservation and restoration efforts.

Other aspects concerning the assessment of the effectiveness of implemented measures is the level of conformity to the implementation specification of the measure, the compliance of those that have to implement the measures and the reliability of the measures to perform adequately over time (Cormier et al., 2019). Due to the lack of studies and research designed to determine by how much a given measure contributes to the reduction of a specific pressure, the quantification of effectiveness remains challenging.

An analysis of the effectiveness of management measures should allow defining the amount of residual pressure as an undesirable outcome of a measure or measures (e.g. level of contaminant reduction in an effluent, the reduction of the spatial extent or frequency of sedimentation, etc.). From a methodological perspective, for instance, modelling tools can be used to simulate different levels of effectiveness of an action linked to different pressure levels and compare prediction with observational data (Coll et al., 2008; Piroddi et al., 2015). Further, Cormier et al. (2018) presented a modelling framework which permits to quantify the residual pressure and how it contributes to the management effectiveness. A sound understanding of the cause-effect pathways should then help assessing the contribution of the reduction of the pressure to achieve the desired ecosystem state.

2.5. Providing scientific evidence for risk evaluation

The risk-based CEA framework considers risk evaluation as a process where management and stakeholders evaluate what could be done to reduce the detected risks of cumulative effects. In the risk evaluation step the decision is taken to maintain or improve existing measures or implement additional ones. Hence, the decision in risk evaluation is about choosing the management strategy that would reduce the risks as low as reasonably practicable given that risk can never be zero (Baybutt, 2014). Risk evaluation is where the results of the risk analysis

are brought into the policy realm of decision-making, which is actually the interface between the science and the policy (Cormier et al., 2018). Up to this point, risk identification and risk analysis have primarily a scientific and technical role in the provision of independent scientific advice without any value judgement such as e.g. “serious”, “harmful”, “impacting” or “severe” (Fig. 5; left). Thus, only the levels of the likelihood of the effect occurring and the magnitude of consequences are discussed in relation to the source of the risk, as outlined by the cause-effect pathways. In risk evaluation, the scientific advice is provided to the managers and stakeholders to underpin their decisions as to what to do in terms of management measures required to reduce the risk considering the severity of those risks. Given that visualization is a key communication tool to non-technical managers and stakeholders (Stelzenmüller et al., 2018), risk matrices are typically used in risk evaluation as a graphical representation of the likelihood and consequence combinations that are less to more tolerable given the policy context (Cormier and Lonsdale, 2020). In risk evaluation, tolerable refers to the likelihood or risk of not achieving stated management objectives. Hence, it is important to note here that the use of such matrices goes beyond the simple identification of the severity of the risk as commonly presented in ecological risk assessments (Astles and Cormier, 2018). As shown in Fig. 5, in a CEA context risk matrices should show how the combinations of the levels of the likelihood of the effect of cumulative residual pressures occurring and the severity of consequences are mapped to different risk tolerance levels (e.g. high, moderate, low). In the examples of Fig. 5, the likelihood of the consequence of the existing management measures (P_{EM}) is compared to the likelihood of the consequence for the proposed improvements to existing measures (P_{EP}). Thus, improvements and additional measures should reduce the likelihood of the cumulative residual pressures and/or the severity to a level that is tolerable in terms of reaching stated objectives given the scientific, management and operational uncertainties. The different color scheme of the three example matrices simply reflects different levels of risk tolerance by the managers. For instance, there are more red boxes for cases of low tolerance to risk compared to matrices reflecting higher tolerances. This requires prior definitions; red could mean that the likely consequences are not tolerable because the management measures are not effective enough to reach defined objectives, while orange or yellow could mean that there are uncertainties as to whether management measures will lead to the achievement of targets, which would imply for instance extensive monitoring and review. Finally, green would imply that the management measures are considered effective in the sense that policy objectives would be reached. Matrices should avoid using qualifiers such as high, medium or low or 1, 2, 3 because they do not explicitly convey the severity of the risks to managers and stakeholders (Baybutt, 2018). If the risk of cumulative residual pressures is assessed for more than one assessment endpoints (e.g. species, functions, processes), each causal concern should also

have its own matrix because decisions regarding such risks would weight different combinations of likelihood and consequences.

Given the iterative aspects of CEA, decision makers, managers and stakeholders could submit new management options to risk analysis that would then be analyzed by scientific and technical experts. As explained above, technically, scientists should not be part of the risk evaluation, but in practice, they are often consulted when it comes to actual decision making. Across our case studies, we identified examples where there have been processes to clarify the roles of science and management (e.g. Gulf of St. Lawrence case study) up to cases where roles have been mixed. Therefore, scientists should be prepared to develop and deploy tools in risk identification and risk analysis to determine the effectiveness of various management options. In risk evaluation, scientists can only provide insight into uncertainties and assumptions involved in determining the likely consequences of various management scenarios. The decision about the tolerability of the risks not to meet the objectives should be left to the decision-makers, managers and stakeholders during the risk evaluation phase. Some ready to use tools to provide informative results to managers and to help them find alternatives or information about risk already exist (Stelzenmüller et al., 2018). This is linked to the “being proactive” in the assessment of alternative scenarios of management advice and “being prepared” to present results in a science-policy context.

3. Unfolding uncertainty in CEA

The spatial and temporal distribution of ecological components, multiple pressures operating at various scales, their potential effects upon assessment endpoints, and the effects of proposed management actions are fundamental pieces of information for a CEA. Like in any environmental impact assessment, many assumptions and predictions have to be made, thus making it difficult to estimate the overall uncertainty of the analysis outcomes (Tenney et al., 2006). Hence, considering and treating the uncertainty that is inherent to the various steps of an environmental impact assessment is critical for conveying a comprehensive understanding of the limitations and accuracy of the generated outputs. Towards this direction, previous studies (Gissi et al., 2017; Stelzenmüller et al., 2015b; Stock and Micheli, 2016) offered insights on the potential sources of uncertainty linked with causality and data, and proposed technical solutions on how to deal with it.

Apart from dealing with uncertainty in risk identification and risk analysis, it is also critical to disclose the degree and sources of uncertainty associated with risk evaluation process. In risk evaluation as described above, proposed management actions are contrasted to the likelihood of achieving policy objectives. Here an additional layer of uncertainty is introduced, which could influence the transparency

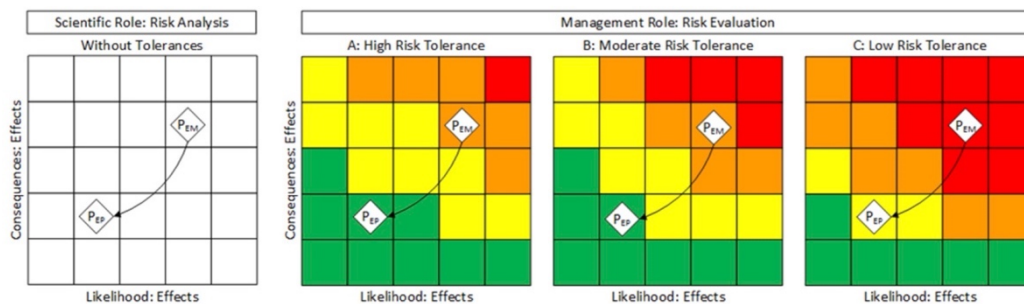


Fig. 5. A risk matrix without tolerance levels (left) derived from the results of the risk analysis of the effectiveness of the management measures in contrast to risk matrices colored by the tolerance levels in risk evaluation (right). The existing management measures (P_{EM}) are compared to the likelihood of a given consequence for the proposed improvements to existing measures or additional measures (P_{EP}).

throughout the decision-making process and therefore affect capitalization of the outputs (Leung et al., 2015; Tenney et al., 2006).

From our case studies, we observed that many cases acknowledged uncertainty, but often in an unstructured fashion. Only for the Adriatic case a substantial effort was undertaken to transparently assess uncertainty (Ansong et al., 2017). In an attempt to unfold the dimensions of uncertainty associated to the risk-based CEA framework and to offer a systematic guidance for improving the treatment of uncertainty, we followed the approach presented in Ansong et al. (2017) and Gissi et al. (2017) and elaborated a Walker-type matrix (Walker et al., 2003). We defined a total of eight uncertainty descriptors: context uncertainty, cause-effect model uncertainty, data input uncertainty, statistical uncertainty, scenario uncertainty, recognized ignorance, epistemic or knowledge-related uncertainty and inherent variability (Table 1 and Appendix C). These descriptors represent uncertainty in a structured way, synthesize sources, causes and needs across the three dimensions: location, level and nature (Walker et al., 2003). Such a comprehensive reporting of uncertainty allows for a better understanding of the overall level of uncertainty associated with the assessment outcomes and is key for an informed decision-making.

We found a great variation of sources of uncertainty across the eleven CEA cases (Appendix C). Hence, we often observed that policies and measures that were identified in the CEA context setting could often not be translated into clear operational objectives with explicit criteria and targets. Comparing the eleven cases showed that the efficiency and adequacy of policies and measures identified at local (e.g. through the managing authority responsible for the protection of sea turtle nesting habitats), national (e.g. Belgium national policies on blue growth and offshore renewable sources) or international scales (e.g. the CFP and the MSP in the case of fisheries in the western Mediterranean or as the potential determinants for protecting habitats and key ecosystems in the Adriatic-Ionian sea) were often defined both as the assessment endpoints and uncertainty of the case study context.

Factors contributing to the uncertainty of the cause-effect modelling processes include the limited knowledge of the spatiotemporal dynamics of ecological processes, the lack of a precise understanding of the mechanism of cumulative effects (e.g. additive, synergistic or antagonistic), and the gaps and incomplete information on the distribution, dynamics and magnitudes of pressures. To overcome these gaps, inputs have been generated from models, expert assessments, and extrapolations from patchy datasets. In some cases, efforts have been made to statistically quantify uncertainty by using sensitivity analyses (Ansong et al., 2017) or by incorporating variability to ensure that model structure is adequate, such as in the Western Mediterranean case. Still, in many of the case studies a further exploration and description of sources of uncertainty through e.g. statistical tools was missing. In relation to scenario uncertainty, identified sources of uncertainty comprised the management measures tested and the magnitude of their future effectiveness and reinforcement. Following the complexity of ecological, environmental and social-political dimensions involved in the CEA process and the focus and spatial scale of the cases, contributors identified uncertainty, which they did not further address. Examples of such recognized ignorance included the spatio-temporal variability of different pressures (e.g. noise pollution, fishing pressure, tourism activities) but also the cumulative effect of invasive species and climate change. Therefore, environmental variability, the multi-dimensional interactions at the ecosystems level or complex ecosystem responses due to climate change reflect sources of uncertainty that are often acknowledged, but rarely defined, quantified or addressed.

4. Recommendations for CEA operationalization

From our CEA case study comparison and analysis, we derive four key recommendations to strengthen the implementation of CEA into management through a risk-based CEA framework:

- 1) *Framing the context and setting risk criteria* – The operationalization of a CEA requires a well-framed context comprising the identification of the drivers, management objectives, and targets. In the absence of clear objectives that address the avoidance or mitigation of cumulative effects, the CEA process should still formulate or lay out the aspired objectives regarding the tolerance of cumulative effects. Only then, risk criteria can be defined. Risk criteria need to be set prior to the assessment, which requires the involvement of stakeholders and decision-makers.
- 2) *Defining the roles* – Throughout a CEA a clear separation and allocation of the roles and expected tasks of decision makers, various stakeholder groups and scientists is fundamental. This helps to build trust when sharing and interpreting data and knowledge.
- 3) *Reducing and structuring complexity* – CEAs are context-dependent, resource intensive and complex. There are unavoidable trade-offs among the level of complexity, available resources and timelines, but they should be reached in a transparent and well documented manner, as cause-effect pathways have to be assessed for each identified link between human activities, pressures, and assessment endpoints.
- 4) *Communicating assumptions and uncertainty* – A cross-cutting issue in successful CEAs is a clear communication of assumptions made throughout the process and types and levels of uncertainty. Emphasis should be put on the selection of tools to present the different dimensions of uncertainty, which accumulates along a CEA process.

5. Conclusions

Our analysis of the context, approaches, and implementation of eleven CEA case studies, which aligned their analyses or evaluation to a risk-based CEA framework, revealed the large variation in CEA drivers, objectives and assessment endpoints. A single recipe on how to conduct a CEA does not exist, but the application of a standardized framework facilitated a consistent and coherent comparison of the key issues to operationalize such complex assessments. Here we underline the urgent need to differentiate CEA in light of the different clients or processes such as governance advice, marine spatial planning or regulatory advice. Thus, laying out the context, assessment objectives and criteria, and roles of those involved, is fundamental to allow for the take up of CEA outcomes in management processes. We suggest that future CEAs should move towards this direction to maximize the advice a CEA can provide in an EBM context. Further, we conclude that it is crucial to communicate uncertainty throughout the various assessment steps in a transparent and structured manner, which helps build confidence and trust in the derived scientific evidence. One of the reasons why CEA have not been formally operationalized yet is their complexity and limitations of knowledge and evidence and the difficulty in identifying which human activity and pressure should be reduced. Applying the risk-based CEA framework together with a strategy of communicating uncertainty should help to overcome bemoaning of imperfect knowledge on the sensitivity of ecosystem components to distinct pressures, and embrace uncertainty around the scientific evidence. Our results underlined the need for further research on the effectiveness of management measures to improve current practices or to develop new ones to reduce the effects of specific human activities. Finally, risk evaluation comprises trade-off analysis of the cost and benefits of additional management measures. Here the final decision on management strategies should be left to the decision makers; scientists should only provide technical advice to such a process. We postulate that if the description and quantification of uncertainty and trade-offs becomes a routine in CEA, then decision makers will more likely understand the potential repercussions of their decisions. In summary, our study makes a strong case that CEA should be well framed and recognized as cross-cutting tools that could bridge different management objectives. We acknowledge that

Table 1

We developed a Walker-type matrix (Ansong et al., 2017; Gimpel et al., 2015; Walker et al., 2003) with eight uncertainty descriptors: context uncertainty, cause-effect model uncertainty, data input uncertainty, statistical uncertainty, scenario uncertainty, recognized ignorance, epistemic or knowledge-related uncertainty and inherent variability. This allows assessing the dimensions of uncertainty associated to the risk-based CEA framework and offers a systematic guidance for improving the treatment of uncertainty.

Uncertainty dimensions	Location - identifies where uncertainty establishes within the methodological approach applied for the risk-based CEA. Location can refer to the context, model and input			Level - encompasses statistical uncertainty, scenario uncertainty and recognized ignorance			Nature - the nature of uncertainty can be distinguished as knowledge related and variability related uncertainty	
Uncertainty descriptors	Context	Cause-effect model	Input	Statistical uncertainty	Scenario uncertainty	Recognized ignorance	Knowledge related	Variability related
	Policy drivers for CEA (e.g. problem framing stage or boundaries determined by policies, legislations) and defined risk criteria (i.e. benchmarks) against which the evaluation of cumulative effects is being performed	Uncertainty in assessing cause-effect pathways can relate to (i) the description of causal relationships, (ii) externalities outside the CEA context	The data input relates to pressures and their related effects. It also comprises the data used for assessing the effectiveness of management measures	Uncertainty that can be statistically quantified	The range of possible outcomes of the management measures being considered to reduce pressures and the risk of cumulative effects	A fundamental uncertainty about the mechanisms and functional relationships considered in the CEA	Uncertainty which refers to the imperfection of knowledge; which may be reduced by conducting more research	Uncertainty related to the variability inherent in the studied system

resolving mismatches in governance structures takes time and is often difficult, but we suggest that CEA can be one strategic approach to integrate ecosystem management considerations across multiple sectorial policies.

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CRedit authorship contribution statement

Vanessa Stelzenmüller:Conceptualization, Methodology, Resources, Formal analysis, Writing - original draft.**Marta Coll:**Conceptualization, Methodology, Resources, Formal analysis, Writing - original draft.**Roland Cormier:**Conceptualization, Methodology, Resources, Writing - original draft.**Antonios D. Mazaris:**Conceptualization, Methodology, Resources, Formal analysis, Writing - original draft, Formal analysis.**Marta Pascual:**Conceptualization, Methodology, Writing - original draft.**Charles Loiseau:**Conceptualization, Methodology, Visualization, Writing - review & editing.**Joachim Claudet:**Resources, Writing - review & editing.**Stelios Katsanevakis:**Resources, Writing - review & editing, Funding acquisition.**Elena Gissi:**Resources, Writing - review & editing.**Athanasios Evagelopoulou:**Resources, Writing - review & editing.**Bob Rumes:**Resources, Writing - review & editing.**Steven Degraer:**Resources, Writing - review & editing.**Henn Ojaveer:**Resources, Writing - review & editing.**Tiia Moller:**Resources, Writing - review & editing.**Joan Giménez:**Resources, Writing - review & editing.**Chiara Piroddi:**Resources, Writing - review & editing.**Vasiliki Markantonatou:**Resources, Writing - review & editing.**Charalampos Dimitriadis:**Resources, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts

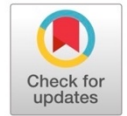
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9.2 Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts

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Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts

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Abstract. Nutrient pollution is altering coastal ecosystems worldwide. On coral reefs, excess nutrients can favor the production of algae at the expense of reef-building corals, yet the role of nutrients in driving community changes such as shifts from coral to macroalgae is not well understood. Here we investigate the potential role of anthropogenic nutrient loading in driving recent coral-to-macroalgae phase shifts on reefs in the lagoons surrounding the Pacific island of Moorea, French Polynesia. We use nitrogen (N) tissue content and stable isotopes ($\delta^{15}\text{N}$) in an abundant macroalga (*Turbinaria ornata*) together with empirical models of nutrient discharge to describe spatial and temporal patterns of nutrient enrichment in the lagoons. We then employ time series data to test whether recent increases in macroalgae are associated with nutrients. Our results revealed that patterns of N enrichment were linked to several factors, including rainfall, wave-driven circulation, and distance from anthropogenic nutrient sources, especially human sewage. Reefs near large watersheds, where inputs of N from sewage and agriculture are high, have been consistently enriched in N for at least the last decade. In many of these areas, corals have decreased and macroalgae have increased, while reefs with lower levels of N input have maintained high cover of coral and low cover of macroalgae. Importantly, these patchy phase shifts to macroalgae have occurred despite substantial island-wide increases in the density and biomass of herbivorous fishes over the time period. Together, these results indicate that nutrient loading may be an important driver of coral-to-macroalgae phase shifts in the lagoons of Moorea even though the reefs harbor an abundant and diverse herbivore assemblage. These results emphasize the important role that bottom-up factors can play in driving coral-to-macroalgae phase shifts and underscore the critical importance of watershed management for reducing inputs of nutrients and other land-based pollutants to coral reef ecosystems.

Key words: bottom-up; coral reef; $\delta^{15}\text{N}$; herbivory; macroalgae; nitrogen; nutrient pollution; phase shift; sewage; stable isotopes; top-down; *Turbinaria ornata*.

INTRODUCTION

Nutrient pollution is a major anthropogenic force altering coastal ecosystems worldwide (Carpenter et al. 1998, Howarth 2008). Anthropogenic inputs of nitrogen and phosphorous can facilitate blooms of harmful algae, reduce water clarity, alter food webs, and deplete oxygen leading to dead zones (Boesch 2002), ultimately disrupting fisheries and harming human health (Anderson et al.

2012). These adverse effects of excess nutrients on coastal ecosystems are likely to be exacerbated in coming decades by continued changes in land-use as well as altered precipitation regimes and ocean warming due to climate change (Altieri and Gedan 2015, Gobler et al. 2017, Sinha et al. 2017).

Excess nutrients can be especially harmful to oligotrophic systems such as coral reefs. High levels of nutrient loading can lead to persistent blooms of algae that can dramatically alter the structure and function of reef ecosystems (Naim 1993, Loya 2004, Lapointe et al. 2005). Even moderate nutrient loading can favor algae and heterotrophic filter feeders that compete with corals

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and other reef building taxa (Fabricius 2005, Rice et al. 2020), thereby impeding coral recovery following a disturbance (Graham et al. 2015, MacNeil et al. 2019). Nitrogen (N) loading can be particularly harmful to coral reefs, as excess N can negatively impact coral growth and survival in addition to fueling algal growth (D'Angelo and Wiedenmann 2014, Shantz and Burkepile 2014, Morris et al. 2019). Thus, understanding the sources and impacts of anthropogenic N is critical for conservation and management of coral reef ecosystems.

In some cases, N pollution can be traced to a specific point source (e.g., sewage outfall) (Smith et al. 1981, Dailer et al. 2010). However, most N input to coastal ecosystems is derived from diffuse nonpoint sources (Carpenter et al. 1998). On coral reefs, N from nonpoint sources tends to be delivered in surface water runoff during episodic rainfall events and via discharge of submarine groundwater. After reaching the ocean, N-rich water is then transported and mixed via local hydrodynamic features. Small-scale hydrodynamics also influence rates of nutrient uptake by modifying boundary-layer conditions of benthic organisms (Carpenter et al. 1991). Thus, the movement and ultimate fate of anthropogenically derived N in nearshore ecosystems are complex and will depend on many interacting factors, including the location of N sources, the timing of rainfall, and the hydrodynamic environment (Brocke et al. 2015).

A more complete understanding of the impacts of N pollution on nearshore ecosystems requires knowledge of the spatial and temporal patterns of enrichment and how the physical environment modulates these patterns. In many locations, water column nutrients are measured periodically in mandated water quality monitoring programs. These programs can provide valuable information on decadal-scale trends and large-scale spatial patterns of nutrient availability (De'Ath and Fabricius 2010, Duprey et al. 2016, Lapointe et al. 2019). However, the coarse temporal and spatial resolution of most water column monitoring programs precludes building a mechanistic understanding of the causes and consequences of nutrient enrichment in many coastal systems. This is particularly true on coral reefs, where researchers have used N content and isotopic signatures in benthic algae to resolve patterns of N enrichment at relatively small spatial scales (Umezawa et al. 2002, Lin and Fong 2008, Dailer et al. 2010). However, these studies generally have not linked enrichment patterns to benthic community dynamics or ecosystem processes (but see Lapointe 1997). As a result, there is widespread disagreement regarding the levels of nutrient enrichment that are harmful to coral reefs and how important nutrients are for impacting coral reef community dynamics (Lapointe 1999, Szmant 2002, D'Angelo and Wiedenmann 2014, Arias-González et al. 2017, Bruno et al. 2019).

Here we quantify the spatial and temporal patterns of N availability on coral reefs around the South Pacific island of Moorea, French Polynesia. The coral reefs

around Moorea have undergone multiple cycles of disturbance and recovery over the past four decades. While the oligotrophic fore reef has shown remarkable resilience to disturbances (Adjeroud et al. 2009, 2018, Holbrook et al. 2018, Kayal et al. 2018), the reefs within the lagoons have exhibited contrasting dynamics, with some reefs shifting from coral- to macroalgae-dominated communities (Schmitt et al. 2019). Nutrient input could be a major driver of algal proliferation within the lagoons, but to date, no studies have evaluated the spatial or temporal patterns of enrichment or how those patterns relate to benthic community dynamics. Here, we combine N content and stable isotopes ($\delta^{15}\text{N}$) in the tissue of the macroalga *Turbinaria ornata*, together with empirical models of nutrient discharge and time series data on the dynamics of benthic algae and coral to address the following questions: (1) What are the spatial patterns of N enrichment and how do these patterns vary across seasons with different levels of rainfall and wave forcing? (2) To what extent are spatial patterns of N enrichment related to anthropogenic inputs? and (3) Are recent increases in macroalgae related to N enrichment?

METHODS

Site description

Moorea, French Polynesia (17°30' S, 149°50' W) is a small (~135 km²) volcanic island located near the larger island of Tahiti in the central South Pacific. The ocean surrounding Moorea is oligotrophic, with levels of dissolved inorganic nitrogen (DIN) often at or below detection limits (nitrate plus nitrite in the top 25 m of the water column, $0.38 \pm 0.84 \mu\text{mol/L}$ [mean \pm SD]; Alldredge and Carlson 2019). The island is surrounded by a well-developed barrier reef located ~1 km offshore that is intersected by 12 reef passes that connect the shallow protected lagoons inside the barrier reef to the open ocean (Fig. 1a). Shoreward of the reef crest is a distinct back reef habitat that is less than 3 m deep and is dominated by patch reefs separated by sand, coral rubble, and reef pavement. Directly adjacent to land are shallow fringing reefs that in some locations are separated from shallow back reef habitats by a channel (>10 m deep) that connects to a reef pass.

Tidal amplitudes are small (<30 cm), and circulation within the lagoon is primarily driven by wave forcing across the reef crest, which creates flow into the lagoon and out of the reef passes (Hench et al. 2008, Leichter et al. 2013). Wave forcing is largely driven by long period swells originating from storms thousands of kilometers away. As a result, wave exposure (and thus circulation patterns) varies seasonally and among sides of the island concurrent with seasonal differences in storm activity in the northern and southern hemispheres. The west and east shores receive the most wave energy, with peak wave energy occurring during winter in the southern hemisphere (hereafter "austral winter") from May to October.

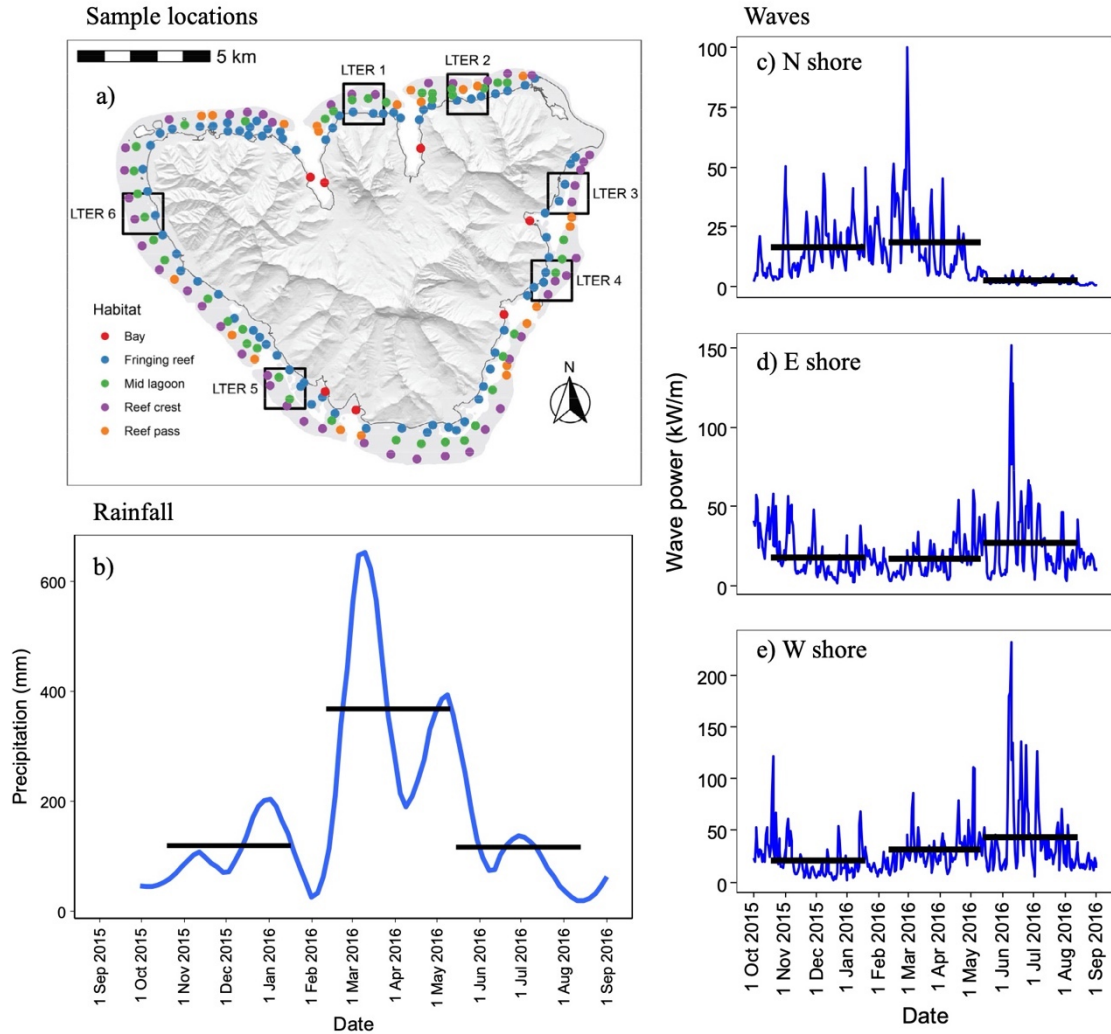


FIG. 1. (a) Map of Moorea showing the locations of the six MCR LTER sites and the sampling grid for the high-resolution nutrient maps with sample locations colored according to habitat type. Shallow lagoon habitat is shown in gray. White space within the lagoons represents non-reef area (e.g., deep water sandy areas). Land is displayed as a digital elevation model with a hillshade to show ridgelines and valleys. (b) Rainfall from October 2015 until September 2016 in Faaa, Tahiti ~15 km east of Moorea. Blue curve is the LOESS smoothed 30-d sum. Black lines are monthly means for each of the three 90-d periods preceding the January, May, and August sampling. Note that we expect the nutrient content of macroalgal tissue (*Turbinaria ornata*) to reflect the nutrient environment during the ~3-month period prior to its collection. Patterns of wave forcing from October 2015 until September 2016 on the (c) north, (d) east, and (e) west shores of Moorea. Blue lines are daily wave power calculated from ADCPs. Black lines are daily means for each of the three 90-d periods preceding the January, May, and August sampling. Note the highly seasonal pattern of wave forcing on the north shore, with minimal wave forcing during the austral winter (June–September). Also note the different scales on the y axes.

In contrast, peak wave energy on the north shore occurs from November to April during winter in the northern hemisphere (“austral summer”; Edmunds et al. 2010). Moorea’s climate is characterized by a warm, wet season from November to April and a cooler and drier season from May to October.

Due to the steep topography, the island’s ~17,000 residents are concentrated along its 60-km coastline, with

population centers located in several large valleys. Rapid population increases in recent decades have coincided with substantial land clearing for agriculture and urbanization (Walker et al. 2014). In contrast to ocean waters, DIN concentrations can be very high (>100 $\mu\text{mol/L}$) in groundwater and streams (Knee et al. 2016, Häbler et al. 2019). Major sources of N to the lagoons include treated and untreated human sewage, animal waste from

livestock, and fertilizer from agriculture and landscaping (Haßler et al. 2019).

What are the spatial patterns of N enrichment and how do these patterns vary across seasons with different levels of rainfall and wave forcing?

To quantify seasonal variation in nutrient delivery mechanisms, we obtained daily rainfall and wave data. Rainfall data were from a meteorological station in Faaa, Tahiti (Global Historical Climatology Network ID: FP000091938), ~15 km east of Moorea. We calculated a 30-d moving sum for October 2015 through August 2016, which corresponded with our high-resolution N sampling, and plotted the LOESS smoothed values together with the monthly means during the different sampling intervals using the R package *ggplot2* (Wickham 2016).

Wave data have been collected continuously at three sites on the exposed fore reef of Moorea (one site on each of the island's three sides) since 2005 as part of the MCR LTER project (Washburn 2019). At each site, data from bottom-mounted Wave & Tide Recorders (SBE 26plus; Sea-Bird Electronics, Bellevue, Washington, USA) and bottom-mounted Acoustic Doppler Current Profilers (Sentinel ADCPs; Teledyne RD Instruments, Poway, California, USA) were used to estimate significant wave height (H_s) and dominant wave period (T_p) at 2-h sampling intervals. Daily averages of H_s and T_p were then calculated and total wave power (P) per linear meter of reef was estimated using deep-water approximations as $P = \rho g^2 H_s^2 T_p / 32 \pi$ where ρ is seawater density (1,025 kg/m³) and g is the acceleration of gravity (9.81 m/s²).

We used ADCPs as the primary instrument to quantify patterns of wave forcing on the three shores of the island because ADCPs had fewer missing values compared to the SBE 26s during our study; to obtain a complete record we used data from the SBE 26s when ADCP data were missing. To ensure that data from the two instruments were comparable, we used the full time series of wave data in least squares linear regressions to estimate the site-specific log-log relationships between P as measured by the SBE 26s and ADCPs. The relationships were strong (r^2 ranging from 0.86 to 0.90), and missing ADCP data were gap filled with the predicted values based on these site-specific relationships. We then calculated daily mean wave power for each shore of the island during each 90-d period preceding our sampling efforts. We focused on a 90-d period assuming that N tissue content in the thalli of *Turbinaria ornata* (hereafter, *Turbinaria*), the species of macroalgae we used to map N availability, reflects the nutrient environment during a period of up to three months prior to its collection. This estimate is based on the growth rates and longevity of *Turbinaria* thalli (Davis 2018, Schmitt et al. 2019). However, because little is known about N storage and turnover in *Turbinaria* we also describe rainfall and wave conditions during the 45-d period prior to *Turbinaria* collection (Appendix S1:

Figs. S1 and S2) since these two time scales (45 and 90 d) are likely to bracket the period of influence.

To map N availability in the lagoons of Moorea, we collected samples of *Turbinaria* at ~180 sites around the island during three different sampling periods, corresponding with different rainfall and wave regimes (January 2016, May 2016, and August 2016; Burkepile and Adam 2020). Sites were at least 0.5 km apart and were spaced to maximize coverage of the different reef habitats within the lagoons, including the fringing reefs, mid-lagoon/back reef, reef crest, reef passes, and bays (Fig. 1a). Like other macroalgae, *Turbinaria* responds to N pulses by storing surplus N (Schaffelke 1999) and consequently N tissue content is believed to be an excellent time-integrated indicator of N availability (Atkinson and Smith 1983, Fong et al. 1994, Shantz et al. 2015). Sampling was conducted over ~3 weeks during each of the three sampling periods; due to logistic constraints, some sites were not sampled in all three sampling periods (January $n = 184$, May $n = 171$, August $n = 173$). At each of the sites, we collected thalli from 10 different patches of *Turbinaria* across an area of ~500 m². Samples were immediately placed on ice and transported to the laboratory. One blade from each of 10 thalli was sampled at 5 cm below the apical tip. Blades were scrubbed of epiphytes and rinsed with fresh water before being dried at 60°C to a constant mass and ground to a fine powder. Total N content was determined via elemental analysis using a CHN Carlo-Erba elemental analyzer (NA1500) at the University of Georgia, Center for Applied Isotope Studies.

In addition to analyzing N content of algal tissue, we also conducted stable isotope analyses to help determine the sources of N. The use of naturally occurring stable isotopes of N (¹⁵N:¹⁴N, expressed as $\delta^{15}\text{N}$) is particularly useful for distinguishing between natural and sewage-derived N because natural sources generally have low signatures while sewage-derived N is high in ¹⁵N (with $\delta^{15}\text{N}$ values ranging from ~5‰ to 20‰; Risk et al. 2009, Kendall et al. 2012). In the lagoons of Moorea, N likely comes from a mix of oceanic and terrestrial sources, the latter including synthetic and organic fertilizers, livestock, and human sewage. Because synthetic fertilizers tend to have $\delta^{15}\text{N}$ signatures that are similar to or lower than natural sources (generally ranging from -4‰ to 4‰; Dailer et al. 2010), elevated $\delta^{15}\text{N}$ values would indicate that human sewage or animal waste are important sources of N but would not rule out the importance of fertilizers or other sources. Isotopic analysis on dried and ground algal tissue was conducted using a Thermo Finnigan Delta-Plus Advantage isotope mass spectrometer with a Costech EAS elemental analyzer at the University of California, Santa Barbara, Marine Science Institute Analytical Laboratory.

Spatial patterns of N enrichment (total N expressed as percentage of dry mass) were visualized using ordinary kriging with a spherical variogram model as implemented in the R package "kriging" (Olmedo 2014). To

test whether enrichment patterns were consistent across sample periods at the scale of individual sites we used linear correlation, with separate correlations conducted for each side of the island. Finally, we used ANOVA to test whether N levels varied among our three sample periods. Because of significant spatial autocorrelation among sites, we analyzed the mean percent N on each of the island's three shores as a conservative test, using data only from the set of 155 sites that were surveyed in all three sample periods.

Visual inspection of N interpolations suggested that spatial patterns of enrichment were related to wave power (see *Results*). Therefore, we used the results of the correlation analysis described in the previous paragraph to test whether temporal consistency depended on variation in wave forcing. Specifically, we used Pearson's correlation to test whether the correlation coefficient describing the temporal correlation in N enrichment (percent N) was related to differences in mean wave power (specific to each shore) across the different sample periods.

To what extent are spatial patterns of nitrogen enrichment related to anthropogenic inputs?

To test whether spatial patterns of N enrichment were associated with anthropogenic N inputs into the lagoons, we modeled the distribution of N from sewage and agriculture based on spatially explicit data from the urban planning office of French Polynesia and the Institut de la Statistique de la Polynésie Française (ISPF). We mapped the relative input of nutrients from agriculture in the lagoon using a three-step procedure. First, for each catchment area ($n = 119$), we calculated the total farmed area based on data available from the urban planning office of French Polynesia, and used it as an estimate of "agriculture intensity." Second, we estimated the maximum diffusion potential of each catchment (MDP_w) using the following formula:

$MDP_w = a \times A_w + b$ where A_w is the size of the catchment w in square kilometers and a and b are the slope ($a = 0.0022$) and intercept ($b = 15.82$) of the linear relationship, obtained from Adjeroud and Salvat (1996). This approach is consistent with global (Burgers et al. 2014) and local (Brown et al. 2017) studies highlighting the positive correlation between A_w and nutrient discharge to the marine environment, and has been widely used in cumulative impact assessments (Halpern et al. 2008, Ban et al. 2010, Micheli et al. 2013). Third, we applied a kernel density function at each river mouth that linearly decays "agriculture intensity" of the associated watershed basin from the river mouth (maximum value) to the MDP_w (null value) to disperse these watershed-scale values onto Moorea's reefs. Then, values were rescaled from 0 to 1 in order to represent the relative level of nutrient input from agriculture with the final layer having 5 m resolution. Kernel estimation was completed using the Heatmap plugin in QGIS v2.18.14 (QGIS Development Team 2016).

We mapped the relative sewage discharge by combining household density along the coast and the water treatment system used in each individual household ($n = 8,614$). Each treatment system was assigned a value based on its overall environmental impact (no treatment = 2; treatment with a sump = 1; and treatment with a plant = 0; data from ISPF), which was subsequently used as a weighting component of household density. We then extrapolated onto the reef using linear decay, assuming that pollution from sewage discharge spread linearly into the lagoon from the source. Kernel estimation was completed using the Heatmap plugin in QGIS v2.18.14 (QGIS Development Team 2016).

To test whether spatial patterns of N enrichment were related to anthropogenic inputs we used a two-step approach. First, we tested whether the isotopic signature ($\delta^{15}\text{N}$) of *Turbinaria* was related to the modeled distributions of N from agriculture and sewage. We found that $\delta^{15}\text{N}$ signatures were well predicted by modeled sewage input, particularly during the rainy season, suggesting that $\delta^{15}\text{N}$ is a good proxy for anthropogenic nutrients (see *Results*). Thus, to test whether the overall N enrichment patterns were driven by anthropogenic N input we tested for a relationship between $\delta^{15}\text{N}$ and total N. Because we had no a priori expectation of the functional forms of the relationships between anthropogenic nutrient input, isotopic signature ($\delta^{15}\text{N}$), and total N, we used boosted regression tree (BRT) analyses; BRTs are a powerful machine learning approach to model fitting that use the data to characterize the relationships between variables (Elith et al. 2008, De'Ath and Fabricius 2010). BRT models were conducted using the dismo package in R (Hijmans et al. 2017) and parameterized according to Elith et al. (2008). Island shore was included in all models. In addition, to explore whether models had greater prediction power for some shores compared to others we also created separate models for each island shore. To aid in data interpretation we show both the fitted relationships from the BRT analyses as well as correlation plots of the raw data.

Are recent increases in macroalgae related to nitrogen enrichment?

To track long-term variation in N availability to benthic organisms, we collected data on N content of *Turbinaria* tissue annually between 2007 and 2013 as part of the Moorea Coral Reef Long-Term Ecological Research (MCR LTER) program (Carpenter 2018). This time frame captures the period when macroalgae increased markedly at some locations in the lagoons. We sampled 10 *Turbinaria* from 18 locations on six cross-shore transects (corresponding to the six core MCR LTER sites), with samples taken on the fringing reef, back reef, and reef crest during the last week of May or first week of June. Total N content was determined via CHN analysis on a model CEC 440HA organic elemental analyzer at the University of California, Santa

Barbara, Marine Science Institute Analytical Laboratory. To determine whether N enrichment varied consistently among sites, habitats, and years we used ANOVA and AIC_c to select among models with and without interaction coefficients (Burnham and Anderson 2002). For this and other linear models, we inspected model residuals to ensure the assumptions of the tests were met.

To characterize the dynamics of macroalgae and coral, we collected data annually in three habitat types (fringing reef, back reef, and fore reef) at six sites (two on each of the three sides of the island) from 2006 to 2018 as part of the Moorea Coral Reef Long-Term Ecological Research (MCR LTER) program (Carpenter 2019). Here we focus on the back reef habitat, where macroalgae increased in abundance in some locations (Schmitt et al. 2019). At each site, the dominant space holders are characterized using point contacts at 50 fixed locations on five permanent 10 m long transects ($n = 10$ 0.25-m² quadrats per transect). Macroalgae are identified to species in situ with other benthic space holders categorized to functional group (e.g., turf algae, scleractinian coral, etc.). In addition to benthic space holders, annual data on the abundance and biomass of fishes are collected at the same sites via visual surveys on four 50 m long by 5 m wide fixed transects (Brooks 2018).

To determine whether the persistent increases in macroalgae that occurred on the back reef were related to spatial patterns of N enrichment and/or herbivore biomass we tested for an association between the change in the mean percent cover of macroalgae at each of our six long-term sites and (1) mean N enrichment, and (2) mean biomass of herbivorous fishes during the time when the coral-to-macroalgae phase shifts occurred (between 2007 and 2013). Ideally, we would use a model selection framework to disentangle the multiple factors that may be driving increases in macroalgae. Yet, with only six sites this was not possible. Instead we used two separate Spearman's rank correlations to independently test for an association between N and macroalgae and herbivore biomass and macroalgae. A positive correlation between N and macroalgae would indicate that reefs that have been consistently enriched in N were more likely to experience increases in macroalgae than less enriched reefs but would not rule out the importance of other factors. Similarly, a negative correlation between herbivore biomass and macroalgae would indicate that reefs with the highest biomass of herbivores were least likely to experience increases in macroalgae.

RESULTS

What are the spatial patterns of N enrichment and how do these patterns vary across seasons with different levels of rainfall and wave forcing?

Rainfall varied among the three sample periods, reflecting typical seasonal patterns in Moorea. During

2016, mean monthly rainfall was more than threefold greater prior to the May sampling, compared to the January and August sampling (January, 121 mm; May, 372 mm; August, 110 mm) due to abundant rainfall during February, March, and April (Fig. 1b).

Wave power varied greatly among the three sides of the island and exhibited different seasonal patterns reflecting differences in exposure to waves generated from storms in the southern and northern hemispheres (Fig. 1c–e). Wave power was most variable on the north shore (among period CV = 70%), intermediate on the west shore (among period CV = 40%), and least variable on the east shore (among period CV = 28%). Mean daily wave power was consistently high on the north shore prior to the January and May sampling periods but dropped precipitously in May and remained low throughout the austral winter (Fig. 1c). The result was a sevenfold decline in wave power on the north shore in the three months leading up to the August sampling relative to the previous three months. Wave power was consistently high on the west shore, with wave power exceeding the north shore during all sampling periods (Fig. 1e). Wave power was especially high on the west shore during the austral winter, with wave power approximately twofold greater in the three months leading up to the August sampling compared to the three-month period preceding the January sampling (Fig. 1e). Mean wave power on the east shore was intermediate between the north and west shores. Similar to the west shore, wave power was greatest on the east shore during the austral winter and least during the austral summer (Fig. 1d).

Nitrogen content of *Turbinaria* tissue revealed evidence for enrichment hotspots on all three sides of the island (Fig. 2). Algae were enriched in N near bays and large watersheds and were also frequently enriched near reef passes. In contrast, algae tended to be lower in N near the reef crest and in the mid-lagoon (Fig. 2). Enrichment patterns varied through time and were related to seasonal changes in wave power and rainfall. Algae were more enriched in N in May following the rainy season compared to January and August, which followed drier periods (ANOVA, $F_{2,6} = 18.4$, $P = 0.003$; Post hoc Tukey tests comparing percent N in May to January and August, $P < 0.01$ for both). The spatial extent of enrichment appeared to be related to the wave regime. During periods of low wave forcing (e.g., the north shore during August), enrichment signals were constrained to nearshore fringing reefs and reef passes compared to periods of high wave forcing (e.g., the west shore during August) when enrichment signals extended farther into the lagoons (Fig. 2).

Enrichment patterns were more consistent throughout the year on some sides of the island compared to others. Spatial patterns of enrichment were highly consistent on the east shore, where wave forcing was least variable among seasons (Appendix S1: Fig. S3). In contrast, enrichment patterns were much more variable on the

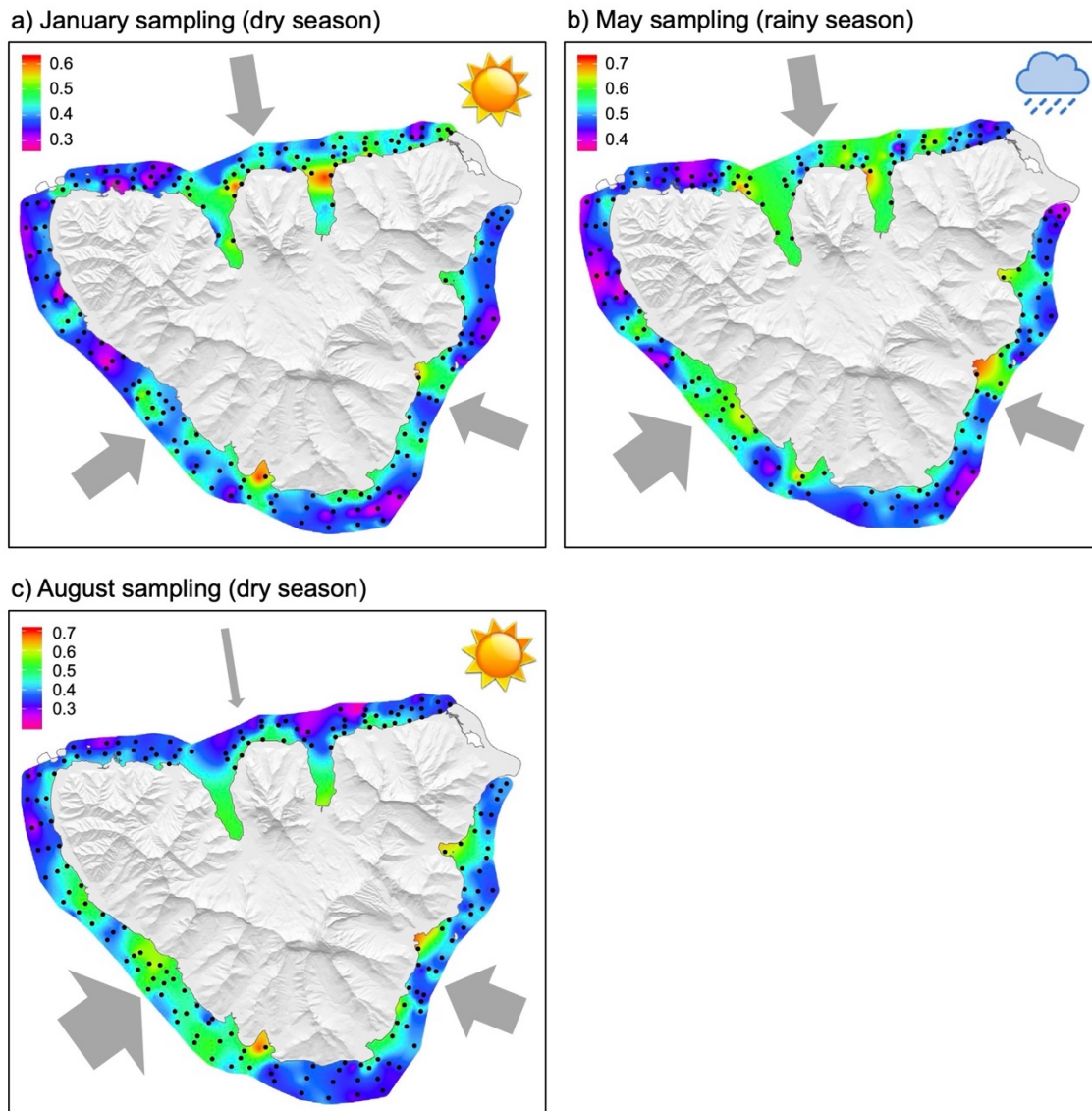


FIG. 2. Spatial patterns of nitrogen enrichment (percent N) in tissue from the macroalga *Turbinaria ornata* for the (a) January, (b) May, and (c) August sampling. Arrows are proportional to the mean wave power on each of the three sides of the island during the 90 d preceding each of the three sampling efforts. Note that the color scale for percent N differs among sample periods. Algae were consistently enriched in N at fringing reef sites and near reef passes compared to sites near the reef crest. Enrichment is especially strong near large watersheds, most notably the areas around the two large bays on the north shore of the island as well as the smaller bays on the east and west shores.

north shore where wave forcing was highly variable (Appendix S1: Fig. S4). Spatial patterns of enrichment on the west shore were intermediate in consistency compared to the other two shores (Appendix S1: Fig. S5). Pairwise correlation of algal N content among sites during different seasons was strongest when wave power was similar, as indicated by a negative association between the among-period correlation coefficient and the difference in mean wave power during the 90-d

period immediately preceding sampling (Pearson's correlation, $r = -0.59$, $P = 0.09$; Appendix S1: Fig. S6).

To what extent are spatial patterns of nitrogen enrichment related to anthropogenic inputs?

Isotopic signature ($\delta^{15}\text{N}$) of *Turbinaria* exhibited strong spatial patterns that varied considerably among sample periods (Fig. 3b, Appendix S1: Fig. S7). During

the rainy season, when anthropogenic inputs of N are likely to be highest, kernel density estimates of human sewage derived from data on household sanitation predicted well $\delta^{15}\text{N}$ values in *Turbinaria* (Fig. 3a–e) (BRT full model cross-validation correlation = 0.51; relative importance of modeled sewage = 75%). The fitted relationship between modeled sewage and $\delta^{15}\text{N}$ was positive and roughly linear across most of the range of the data (Appendix S1: Fig. S8). $\delta^{15}\text{N}$ was also positively related to modeled nutrients from agriculture (Appendix S1: Fig. S8), though agriculture had little predictive power compared to sewage (BRT relative importance = 7%). Kernel density estimates of human sewage were also correlated with $\delta^{15}\text{N}$ during the January and August sampling (Appendix S1: Figs. S8, S9), though the correlations were less strong (January, full model cross-validation correlation = 0.48, relative importance of modeled sewage = 59%; August, full model cross-validation correlation = 0.38, relative importance of modeled sewage = 30%) and the fitted relationships were highly nonlinear with $\delta^{15}\text{N}$ initially decreasing with predicted sewage before increasing at higher levels (Appendix S1: Fig. S8).

Model performance also varied for different sides of the island. For example, during the rainy season, modeled sewage predicted as much as 73% of the variation in $\delta^{15}\text{N}$ on the east shore of the island (BRT cross-validation correlation = 0.73; relative importance of modeled sewage = 100%). In contrast, modeled sewage during the rainy season only predicted 24% of the variation in $\delta^{15}\text{N}$ on the west shore (BRT cross-validation correlation = 0.32; relative importance of modeled sewage = 100%) and 53% of the variation in $\delta^{15}\text{N}$ on the north shore (BRT cross-validation correlation = 0.55; relative importance of modeled sewage = 97%). Modeled nutrients from agriculture consistently had little predictive power (BRT relative importance < 10%).

Nitrogen tissue content was positively associated with $\delta^{15}\text{N}$ during all sample periods, but the relationship was particularly strong (and roughly linear across most of the range of data) during the rainy season (Fig. 3f–h; Appendix S1: Figs. S10 and S11; BRT full model cross-validation correlation = 0.61; relative importance of $\delta^{15}\text{N}$ = 89%). The association between $\delta^{15}\text{N}$ and total N was strongest for the north and east shores of the island. For example, during the rainy season, $\delta^{15}\text{N}$ predicted 75% and 81% of the variation in total N on these shores, respectively (BRT cross-validation correlation north shore = 0.75, east shore = 0.81), while predicting a more modest 35% of the variation in total N on the west shore (BRT cross-validation correlation = 0.35).

Are recent increases in macroalgae related to nitrogen enrichment?

Annual sampling of *Turbinaria* from three habitats at our six long-term sites revealed strong spatial patterns that were consistent through time (model without

interactions outperformed all other models with a delta $\text{AIC}_c \geq 13$). Algal N content varied strongly among habitats (ANOVA, habitat $F_{2,108} = 23.3$, $P < 0.001$) and sites (ANOVA, site $F_{5,108} = 3.2$, $P = 0.009$; Fig. 4). Algae were enriched in N on nearshore fringing reefs, and to a lesser extent the back reef, relative to the reef crest (Fig. 4a). These differences mirror long-term differences in water column DIN over the same time period (Appendix S1: Fig. S12). In addition to these differences among reef habitats, algae also tended to be more enriched in N at the north shore sites compared to sites on the east and west shores (Fig. 4b).

Beginning at or shortly after the onset of our benthic time series in 2007, corals began to decline and macroalgae began to increase on the back reef at permanent study sites LTER 1, LTER 2, and LTER 3 (Fig. 5). By 2013 at LTER 1 and LTER 2, and 2014 at LTER 3, the cover of macroalgae exceeded coral cover (Fig. 5). Since 2014, coral cover at these sites has continued to decline or stabilize at low (<5%) levels while the cover of macroalgae has remained high. Species composition of algal assemblages varies somewhat among sites; dominant taxa include the brown algae *Turbinaria ornata*, *Sargassum pacificum*, and *Dictyota bartayresiana* as well as the red alga *Amansia rhodantha* (Fig. 5). Over the same time period, coral cover has remained high, and macroalgae cover low at permanent study sites LTER 4, LTER 5, and LTER 6, which experience lower nutrient conditions compared to LTER 1, LTER 2, and LTER 3 (Fig. 5). Increases in macroalgae were strongly correlated with the long-term nutrient environment, with higher N associated with greater increases in macroalgae (Spearman's rank correlation, $r_s = 0.94$, $P = 0.02$; Fig. 6a). In contrast, there was no significant association between increases in macroalgae and the mean biomass of herbivorous fishes (Spearman's rank correlation, $r_s = 0.20$, $P = 0.71$; Fig. 6b). Higher N was also associated with coral decline, although the correlation was only marginally significant (Spearman's rank correlation, $r_s = -0.77$, $P = 0.10$; Appendix S1: Fig. S13).

DISCUSSION

Patterns of N enrichment in the lagoons of Moorea were related to several factors, including rainfall, wave power, and distance from anthropogenic nutrient sources, particularly sewage. High-resolution maps of N enrichment revealed several patterns that were consistent among the three sample periods. During all sampling periods, algae had higher N on nearshore fringing reefs compared to the mid-lagoon and reef crest. In addition, algae were consistently enriched in N near large watersheds, where inputs of N from sewage and agriculture are high. Our long-term data indicate that areas of the lagoon that are near shore and close to major watersheds have been consistently enriched in N for at least a decade. During this time, corals have decreased and macroalgae have increased at back reef locations with

a) Predicted N from sewage

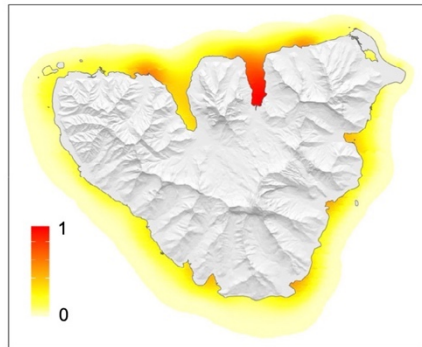
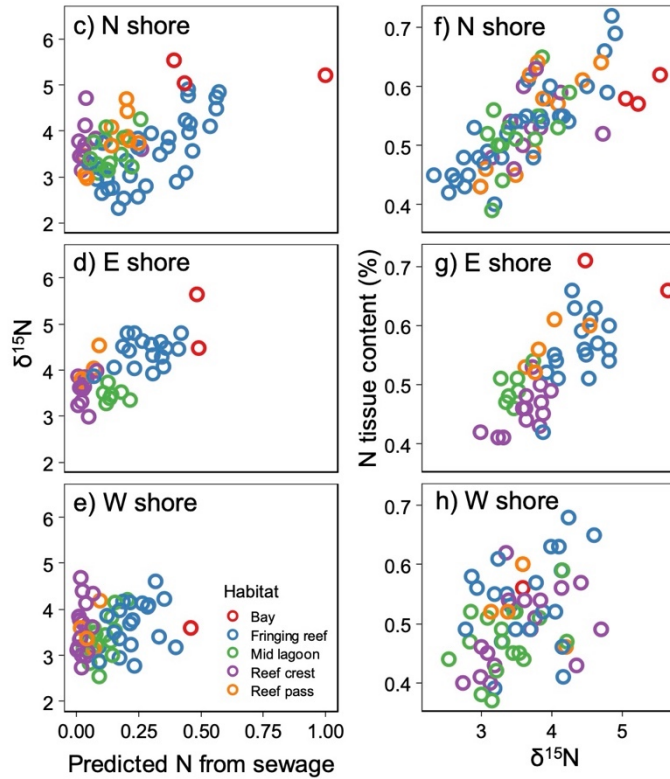
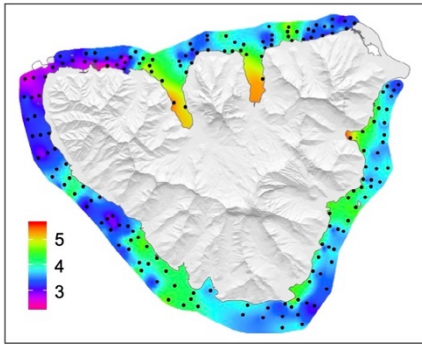
b) $\delta^{15}\text{N}$ during the rainy season

FIG. 3. (a) Modeled N enrichment based on locations of septic systems and sources of untreated sewage. (b) Empirical patterns of $\delta^{15}\text{N}$ in *Turbinaria ornata* tissue during the rainy season. Association between modeled N from sewage and $\delta^{15}\text{N}$ for the (c) north, (d) east, and (e) west shores of Moorea during the rainy season. Association between $\delta^{15}\text{N}$ and nitrogen content (percent N) of *Turbinaria ornata* tissue during the rainy season on the (f) north, (g) east, and (h) west shores of Moorea.

high N, while reefs with lower levels of N have maintained high cover of coral and low cover of macroalgae. Importantly, these patchy phase shifts to macroalgae have occurred despite island-wide increases in the biomass of herbivorous fishes. Similar coral-to-macroalgae phase shifts have been observed on many reefs globally (Hughes 1994, Graham et al. 2015, Bozec et al. 2019), and are often associated with reductions in the biomass of herbivorous fishes due to overfishing. Our observation that nutrient loading, and not herbivory, is strongly associated with phase shifts in the lagoons of Moorea emphasizes that bottom-up factors, such as nutrient pollution from sewage and agriculture, can also be important drivers of coral-to-macroalgae phase shifts on tropical reefs.

Influences of rainfall and wave-driven flow on seasonal patterns of nitrogen enrichment

Many of the enrichment patterns we observed in our high-resolution sampling were consistent among seasons, yet there were also differences that were related to

seasonal differences in wave forcing and rainfall. For example, overall levels of N in *Turbinaria* tissue were higher in May following the wet summer period compared to January and August, which followed periods with less rainfall. In addition, spatial patterns of enrichment were related to wave forcing, which varied asynchronously around the island due to seasonal differences in wave exposure. During periods of low wave energy, enrichment signals were constrained to nearshore fringing reefs and embayments, while during periods of higher wave energy enrichment signals extended farther into the lagoons. The precise mechanisms driving these spatial patterns are unknown but are likely to be related to circulation since waves are the primary driver of circulation in the lagoons (Hench et al. 2008, Leichter et al. 2013).

Irrespective of the precise mechanism, our finding that rainfall and waves can interact to determine spatial patterns of nutrient enrichment in a lagoonal coral reef system has important implications. For example, on the north shore of Moorea where the wave climate is highly seasonal, the rainy season corresponds with a period of

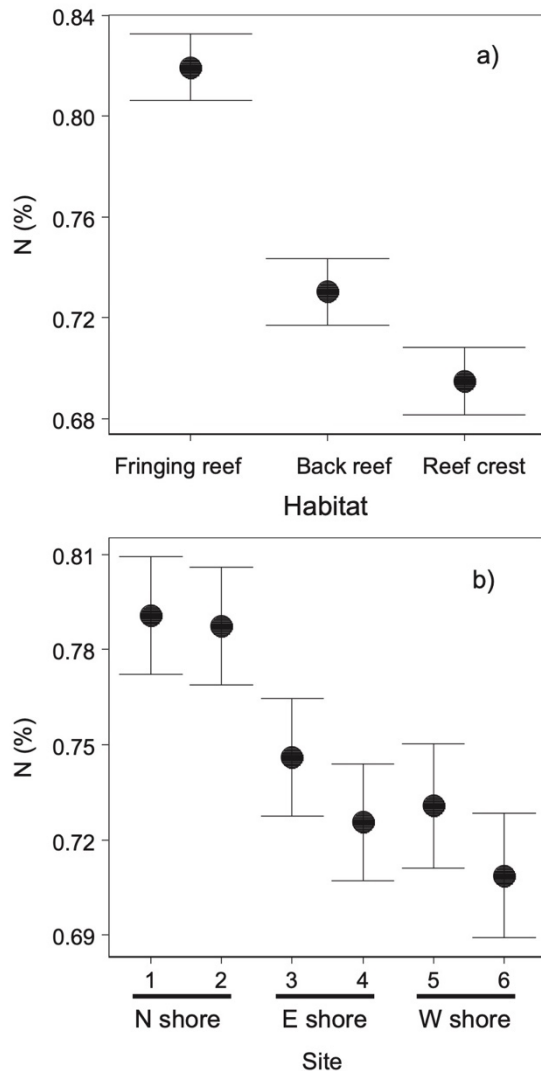


FIG. 4. N content (mean \pm SE) of *Turbinaria ornata* tissue in (a) three habitats (fringing reef, back reef, and reef crest) and (b) at six long-term MCR LTER sites (two sites on each of the three sides of the island) between 2007 and 2013. Note that N is consistently enriched at the fringing reef sites relative to the back reef and reef crest. Also note that LTER sites 1, 2, and 3 tend to be enriched relative to sites 4, 5, and 6.

high wave energy. Because increased rainfall results in N enriched conditions in the lagoons and waves appear to be important for delivering N to the reef crest, the coincidence of the rainy season with a period of large waves could result in nutrient pulses to wide reaches of the lagoon that otherwise might be restricted to the fringing reefs when waves are smaller. Given that waves drive circulation patterns on many tropical oceanic islands (Hench et al. 2008), seasonality in wave-driven circulation could be an important mechanism driving patterns of nutrient enrichment on many coral reefs.

Anthropogenic nitrogen sources

We observed the highest levels of N enrichment near areas with dense human populations and known nutrient sources, particularly sewage. In addition, isotopic signatures of N in areas of high enrichment were consistent with terrigenous rather than oceanic sources of N (Lin and Fong 2008, Dailer et al. 2010, Page et al. 2013). While it was not possible for us to quantify the relative contributions of all the possible N sources to the lagoons, the strong association between the predicted spatial distribution of human sewage and the $\delta^{15}\text{N}$ signature in *Turbinaria* tissue strongly suggests that algae are incorporating sewage-derived N. In addition, the fact that the $\delta^{15}\text{N}$ signature predicted well total N tissue content suggests that anthropogenic sources of N (including sewage) are driving spatial patterns of N availability across the lagoons.

In contrast to sewage, our model of N enrichment from agricultural fertilizer was not a good predictor of the isotopic signature or total N content in *Turbinaria* tissue. The lack of an isotopic signature is unsurprising, given that $\delta^{15}\text{N}$ values of fertilizers are often similar to natural sources (Kendall et al. 2012). Our ability to detect an effect of fertilizer on N tissue content also may have been limited by the assumptions we made when modeling fertilizer input to the lagoons. A key assumption of our predictive model was that N from fertilizer was delivered to the lagoons in surface water runoff via one of several dozen streams around the island. In contrast, we modeled sewage assuming diffusion from the source based on the assumption that sewage contamination would primarily reach the lagoons via groundwater. While little is known about the extent of submarine groundwater discharge in Moorea or the relative importance of groundwater versus surface water runoff for delivering different types of nutrients, recent studies in Moorea suggest that septic waste and animal manure in groundwater are likely an important source of N to the lagoons (Knee et al. 2016, Haßler et al. 2019). Given the potential importance of anthropogenic N input for shaping the structure and function of coral reef ecosystems, future work is needed to better characterize the sources and delivery pathways of anthropogenic N to the lagoons.

The role of nitrogen enrichment in driving coral-to-algae phase shifts

Patterns of N enrichment have important implications for understanding the dynamics of benthic algae, which can strongly influence overall ecosystem health. Nutrient enrichment can allow for the proliferation of algae that are physiologically unable to grow under nutrient poor conditions. In addition, excess nutrients can also facilitate algae indirectly through the loss of top-down control if adding nutrients causes algal production to outpace herbivory (Scheffer et al. 2008, Arias-González

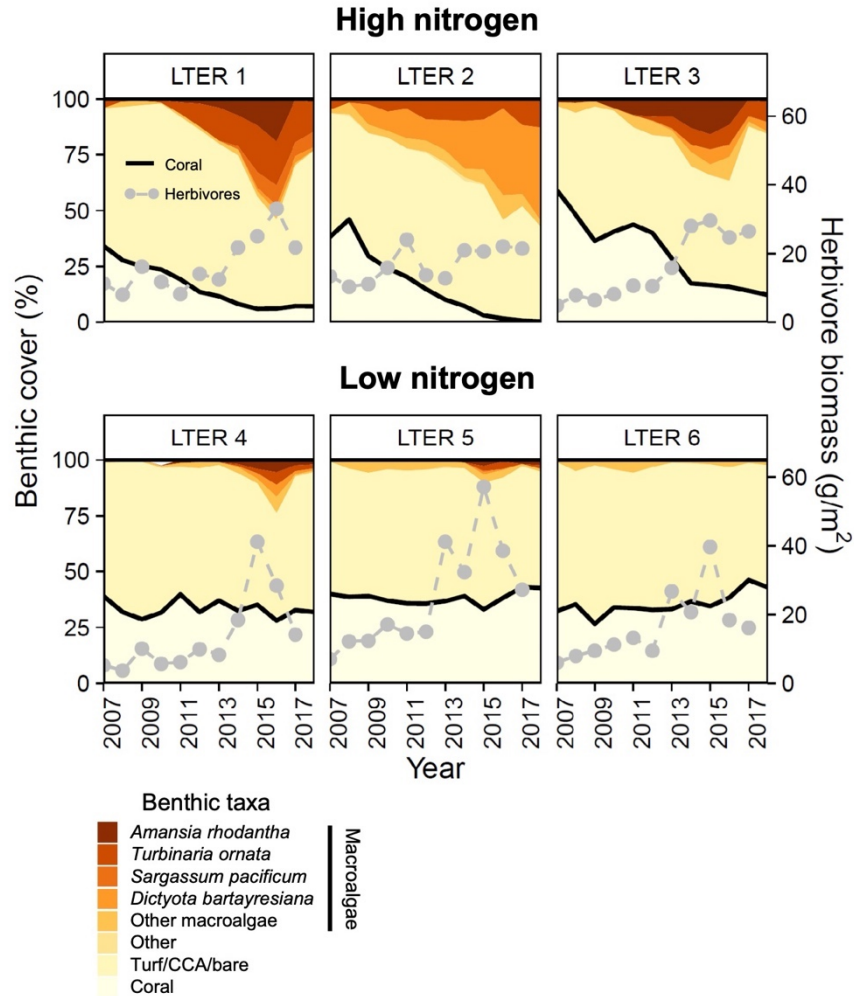


FIG. 5. Area plot showing the percent cover of coral and algae at the six MCR LTER back reef sites (three sites with relatively high nitrogen: LTER 1, LTER 2, and LTER 3 and three sites with relatively low nitrogen: LTER 4, LTER 5, and LTER 6) from 2007 through 2018. Also shown is the mean biomass of herbivorous fishes for the same time period. All values are from annual sampling.

et al. 2017, Briggs et al. 2018). In this study, we found that several species of macroalgae have recently increased in abundance on back reef habitats of Moorea, but only where levels of N are relatively high. Benthic algal blooms have been noted on some nearshore fringing reefs in Moorea in the past (Payri 1987, Adjerdou and Salvat 1996, Gattuso et al. 1997), and it had been hypothesized that those blooms could be related to waste water discharge (Wolanski et al. 1993, Gattuso et al. 1997). Our results support this hypothesis and also suggest that the impacts of nutrients may extend beyond nearshore fringing reefs to back reef habitats farther from shore.

Several factors in addition to nutrient enrichment could be contributing to the increases in macroalgae in

the lagoons of Moorea. Phase shifts to macroalgae frequently occur following large coral-killing disturbances that liberate space for the growth of benthic algae, particularly in systems with low levels of herbivory due to overfishing (Done et al. 1991, Hughes 1994, Graham et al. 2015). The reefs in Moorea experienced two large coral-killing disturbances in the late 2000's, including an outbreak of corallivorous crown-of-thorns seastars (*Acanthaster planci*; COTS) and a large cyclone that together reduced the cover of living coral to near zero on the fore reef (Adam et al. 2011, 2014, Kayal et al. 2012). While reefs in the lagoons were much less impacted by these disturbances, some reefs have experienced a period of protracted coral decline that appears unrelated to these recent disturbances (Han et al. 2016, Schmitt et al.

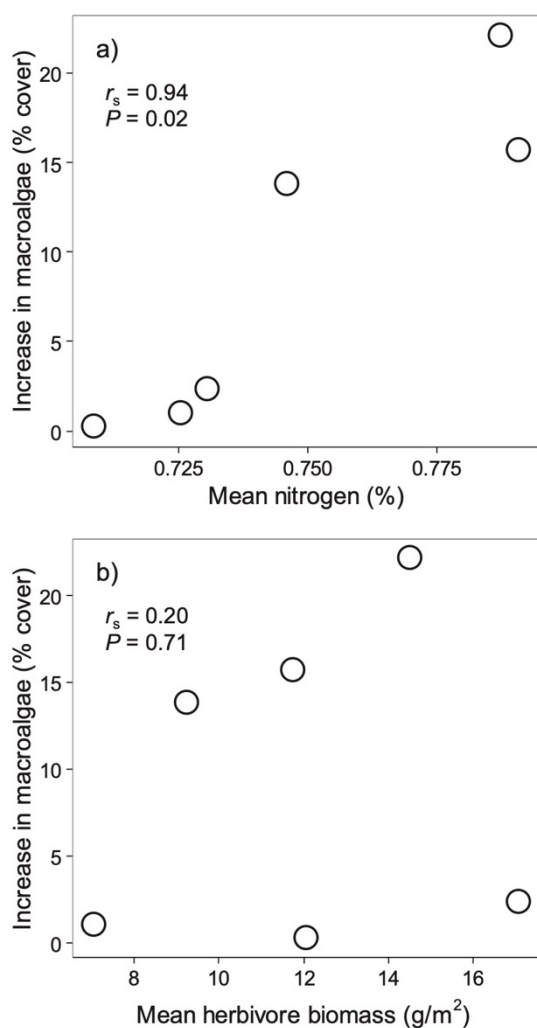


FIG. 6. Correlation between the change in the percent cover of macroalgae between 2007 and 2013 at each of the six MCR LTER back reef sites and the mean (a) nitrogen content of *Turbinaria ornata* tissue and (b) biomass of herbivorous fishes during the same time period.

2019, this study). Coral decline both on the fore reef and within the lagoons coincided with an increase in the biomass of herbivorous fishes (Adam et al. 2011, 2014, Lamy et al. 2015, Han et al. 2016, Dubois et al. 2019). Yet, unlike the fore reef, where herbivores controlled the proliferation of algae and corals have rapidly recovered (Holbrook et al. 2016, 2018), many reefs within the lagoons have become dominated by persistent algal assemblages, despite the presence of a diverse and abundant herbivore assemblage (Han et al. 2016, Edmunds et al. 2019). Higher nutrient loading in the lagoons is one possible explanation for why many reefs within the lagoons have become dominated by macroalgae while macroalgae have remained uncommon on the fore reef

despite increases in herbivore biomass in both habitats. Likewise, differences in nutrient enrichment among lagoons may help explain why reefs in some lagoons have undergone persistent phase shifts to macroalgae, while others remain coral dominated despite similar increases in the biomass of herbivorous fishes.

Patterns of N loading may have also influenced the heterogeneous decline in corals across the lagoons. In addition to the positive correlation between long-term N enrichment and increase in macroalgae, N enrichment was also related to the loss of coral cover. Part of the impact of N on corals could be mediated by the rise in macroalgae, as algae can outcompete juvenile and adult corals (Bulleri et al. 2013, Brown and Carpenter 2014) and inhibit coral recruitment (Kuffner et al. 2006, Mumby et al. 2016, Bulleri et al. 2018). But, excess nutrients also can directly negatively impact corals by increasing the prevalence and severity of coral disease (Bruno et al. 2003, Vega Thurber et al. 2014) and by exacerbating coral bleaching and mortality during thermal stress events (Wiedenmann et al. 2013, Zaneveld et al. 2016, Burkepile et al. 2020, Donovan et al. 2020). Human sewage may be particularly harmful to corals as it both delivers excess nutrients as well as pathogenic microbes that cause coral disease and mortality (Patterson et al. 2002, Sutherland et al. 2010). Nutrients from sewage and other land based sources of pollution are also commonly associated with toxins, organic carbon, and sediments, all of which have direct detrimental effects on corals (Fabricius 2005, Wear and Thurber 2015).

Conclusion

Spatial patterns of N availability in the lagoons of Moorea, a small oceanic island in the South Pacific, are shaped by terrigenous inputs even though lagoons are continuously flushed with oligotrophic ocean water. Long-term data indicate that reefs experiencing the highest levels of N enrichment have undergone persistent phase shifts to a macroalgae-dominated state despite the presence of a diverse and abundant herbivorous fish assemblage. These results indicate that anthropogenic nutrient pollution is likely an important driver of reef degradation in Moorea. Persistent phase shifts from coral to macroalgae are a problem on many reefs worldwide and are often associated with declines in the abundance of herbivorous fishes. Our results emphasize that bottom-up factors can also play an important role in driving these algal phase shifts. Future work is necessary to better characterize the sources and impacts of anthropogenic nutrients on Moorea, but sewage from septic systems appears to be an important contributor. Impacts of nonpoint sources of N are difficult to isolate yet are likely widespread in tropical countries with coral reefs, underscoring the critical importance of managing watersheds to reduce inputs of nutrients and other land-based pollutants to coral reef ecosystems.

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SUPPORTING INFORMATION

Additional supporting information may be found online at: <http://onlinelibrary.wiley.com/doi/10.1002/eap.2227/full>

DATA AVAILABILITY STATEMENT

Data are available on the EDI Data Portal (Burkpile and Adam 2020): <https://doi.org/10.6073/pasta/1197c058ca1672c6f21bdd25d536e25a>

Tracking changes in social-ecological systems along environmental disturbances with the ocean health index

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9.3 Tracking changes in social-ecological systems along environmental disturbances with the ocean health index

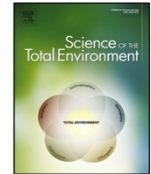
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Tracking changes in social-ecological systems along environmental disturbances with the ocean health index



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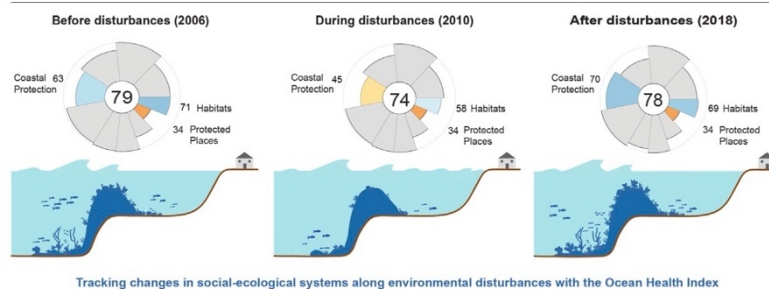
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HIGHLIGHTS

- We use Ocean Health Index (OHI) to track a social-ecological system through time.
- We apply OHI to a local system before, during and after environmental disturbances.
- Environmental disturbances affected mainly the provision of coastal protection.
- Ocean health, and the delivery of ecosystem services, recovered after disturbances.
- Applying OHI at a local scale can benefit decision-making and management.

GRAPHICAL ABSTRACT



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ABSTRACT

The well-being of coastal communities is intimately tied to a healthy ocean, but coastal social-ecological systems are among the most vulnerable to global change. Improving the resilience of coastal communities requires an understanding of how local social-ecological systems respond to shocks to better inform decision-making and adapt local management interventions. However, assessments of social-ecological changes throughout a disturbance regime are scarce at the local level, although critical for efficient natural resource management and sustainable use of ocean ecosystem services. Here, we apply the Ocean Health Index (OHI) to assess the status of the marine social-ecological system of a tropical island (Moorea, French Polynesia), and track changes of the system before, during and after a disturbance regime. Our results show that while there are signs of social-ecological recovery, coastal protection was most affected along the disturbance, and that there is room for improvement toward biodiversity conservation. In addition, our study highlights some context-specific challenges associated with local OHI assessments, particularly those driven by limited fisheries data and appropriate reference point selection for coastal protection. Our results demonstrate the value of localized, regular OHI assessments through time to track changes in marine social-ecological systems, while uncovering important data gaps, to inform management at appropriate scales for decision-making.

1. Introduction

Coastal ecosystems provide highly valuable benefits to people worldwide, such as food provision, coastal protection, carbon storage and a means to livelihood (Barbier, 2017; Bernhardt and O'Connor, 2019; Hoegh-Guldberg et al., 2019; Lau et al., 2019). Yet the condition of ecosys-

tems, particularly coral reefs, is declining due to increasing pressures, including overfishing, pollution, and climate change (Cinner et al., 2018, 2012; He and Silliman, 2019; Hughes et al., 2017b). Ocean health degradation is a major concern for Small Island Developing States (SIDS), whose dependence on healthy marine ecosystems is deepened by their cultural identity and socio-economic and environmental isolation (Balzan et al., 2018; Friedlander, 2018; Narayan et al., 2020). Given this, SIDS are particularly vulnerable to the impacts of climate change (Bennett et al., 2016; Cinner et al., 2018, 2012; Hughes et al., 2017b).

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In the context of global change, increasing resilience of coastal communities requires an understanding of how environmental disturbances alter ecosystem function and the provision of ecosystem services to better guide local, place-based management interventions (Allison and Bassett, 2015; Cardinale et al., 2012; Cinner et al., 2018; Delevaux et al., 2019; Halpern et al., 2019; Thiault et al., 2020). There is an urgent need for “conserving and sustainably using the oceans and marine resources” as stipulated by the United Nations Sustainable Development Goal 14: Life Below Water (SDG 14) (<https://sdgs.un.org/goals/goal14>). Sustainability targets, including SDG 14, have triggered a demand for analytical approaches to assess ocean health and adapt human activities to reflect the ocean's capacity to sustainably support natural resource use (Claudet et al., 2020).

Impact assessments of social-ecological systems are a challenging task (Leenhardt et al., 2015). Quantitative estimates of ocean health are a cornerstone of understanding the changing ocean (Halpern, 2020). Considering the complexity of the ocean as a coupled social-ecological system, composite indicators rather than specific ones might better reflect the state of the ocean (Halpern, 2020). The Ocean Health Index (OHI) is a holistic framework to assess the delivery of benefits obtained from a healthy ocean. It defines a healthy ocean as “one that sustainably delivers a range of benefits to people now and in the future” (Halpern et al., 2012). This framework facilitates transparent and reproducible assessments of ocean health in different contexts, at any spatial scale, and through time. It can contribute to understanding ocean health changes, their consequences for the social-ecological system, and which actions might improve ocean health (Halpern et al., 2017).

Frameworks aimed at informing management decisions need to use indicators obtained at the appropriate scales. One decade after its launch, nine global OHI assessments have been conducted and over 20 regional assessments are completed or in process (Halpern, 2020); however, global scale analyses are too coarse to guide local management and may need to be redefined at finer scales that reflect the scale at which decisions are made (Claudet et al., 2020; Daigle et al., 2017). Despite growing concern over the impacts of global stressors (e.g., increasing sea surface temperature) on the delivery of ecosystem services (Cardinale et al., 2012), and the potential of OHI to track ocean health over time (Halpern et al., 2017; O'Hara et al., 2020), OHI has not yet been used to assess changes in marine ecosystem health temporally across intense environmental disturbances.

Here, we use a tropical island (Moorea, French Polynesia) as a case study. Our objectives are twofold: while we conduct the first assessment of Moorea's social-ecological system using the OHI, we also aim to apply the OHI framework to assess how ocean health and, consequently, the delivery of ecosystem services, change over time through a disturbance regime. Moorea was recently affected by intense environmental disturbances (Lamy et al., 2016, 2015) and the ecosystem currently highlights a recovery debt (Dubois et al., 2019). Although the integrated social-ecological vulnerability of the fisher-fish system was assessed along this disturbance regime (Thiault et al., 2018a, 2018b), more comprehensive social-ecological assessments, accounting for the variety of ecosystem services delivered by coral reefs to communities (Leenhardt et al., 2017), are needed to better inform local management.

2. Methods

2.1. Social-ecological context

Moorea is a volcanic island located in the central South Pacific, 20 km northwest of Tahiti (17°30' S, 149°50' O) in French Polynesia. The island of Moorea is surrounded by a 53 km² shallow lagoon comprised of fringing reef, barrier reef, channels and bays, and a 6 km² fore reef (Fig. 1). Moorea's population is distributed primarily along the 60 km coastline and small valleys (Thiault et al., 2017).

The use of Moorea's coral reefs has changed over the last century from a subsistence economy of mainly artisanal fishing to a tourism-based economy (Leenhardt et al., 2016; Thiault et al., 2018b). Although fishing is not currently the main economic driver for the island, approximately 23 % of

the adult population depends on marine resources for their subsistence and/or income and 35 % of households are involved in a fishery-related activity (ISPF, 2017; Thiault et al., 2017). Polynesian societies are strongly attached to fishing as an essential part of their identity, culture, and lifestyle (Thiault et al., 2017, 2018b).

Coral ecosystems of Moorea have experienced two high-intensity disturbances since 2007: a cyclone in 2010 and an outbreak of the coral predatory starfish *Acanthaster planci* from 2007 to 2010 (Dubois et al., 2019; Kayal et al., 2012; Lamy et al., 2016, 2015). In the fore reef, these disturbances led to a marked decline of live coral and a shift in fish diversity, and have since shown signs of recovery (Adjeroud et al., 2018, 2009; Dubois et al., 2019; Kayal et al., 2018; Lamy et al., 2016, 2015; Thiault et al., 2018b). Moorea provides a unique case study to examine the dynamics and recovery of ecosystem health, and thus, the delivery of ecosystem services affected by drastic disturbances.

2.2. Adapting OHI calculations to local application

We provide here a brief overview of the methods for calculating the Ocean Health Index (OHI). The OHI framework and methods are detailed extensively in the literature (Halpern et al., 2017, 2012; Lowndes et al., 2015) and public sources (e.g., Ocean Health Index, 2016), along with its associated limitations and uncertainties (Branch et al., 2013; Frazier et al., 2016; Halpern, 2020; Halpern et al., 2017).

The OHI is comprised of 10 goals that encompass key ocean elements that people value and benefit from (Halpern et al., 2012). Each goal score is calculated as the average of the current status and the likely future status. The current status is estimated as its present status value relative to a reference point of the highest sustainable performance for that goal. The likely future status estimates the near-term future performance for the goal five years from the current status. It is calculated as a function of the current status, modified by variables that are expected to affect the future status of the specific goal: trend, calculated as the slope in the change of the status score of the previous five years, pressures, and factors enhancing resilience (Halpern et al., 2012).

For the Moorea OHI (MOHI), we focused on Moorea's lagoon and fore reef, rather than the entire exclusive economic zone (EEZ), given its essential role for the subsistence of Polynesians (Leenhardt et al., 2016; Thiault et al., 2018b). We assessed nine of 10 goals (Table A.2). We did not assess the natural products goal due to a lack of monitoring data and the prohibition of extractive activities (other than fisheries) and sea mining (Gabri  et al., 2006). The food provision goal was based on limited fisheries data, details for which, including data used, can be found in the Appendix. We calculated the overall MOHI score as the average of individual goal scores with equal weighting and adjusted by the area of Moorea afterwards. Applying different weighting would have required stakeholder consultations (Daigle et al., 2017; Halpern et al., 2014), which were beyond the scope of this study. The model equations used to estimate current status per goal or sub-goal were maintained from the methods used in global assessments (Table A.3) (Halpern et al., 2017, 2012; Ocean Health Index, 2018a, 2018b, 2018c), while the goals approach and the reference point for some goals were adapted to better suit the Moorea region (Table A.2). Overall, 52 % of data layers were local datasets (Table A.6). The remaining data were derived from 2018 global OHI assessments (Halpern et al., 2017; Ocean Health Index, 2018b). Data source for each data layer is provided (Table A.6), being most of them already available online. We also provided a detailed description of data processing and modifications made in our MOHI in the Appendix.

2.3. Calculating OHI scores over time

We based this analysis on two approaches. First, we calculated the current status for goals or sub-goals across years to explore long-term dynamics. The reference point for each data layer was held constant across years (Halpern et al., 2014). The models were as previously described for the most recent MOHI calculation, except for sub-goals with a moving temporal

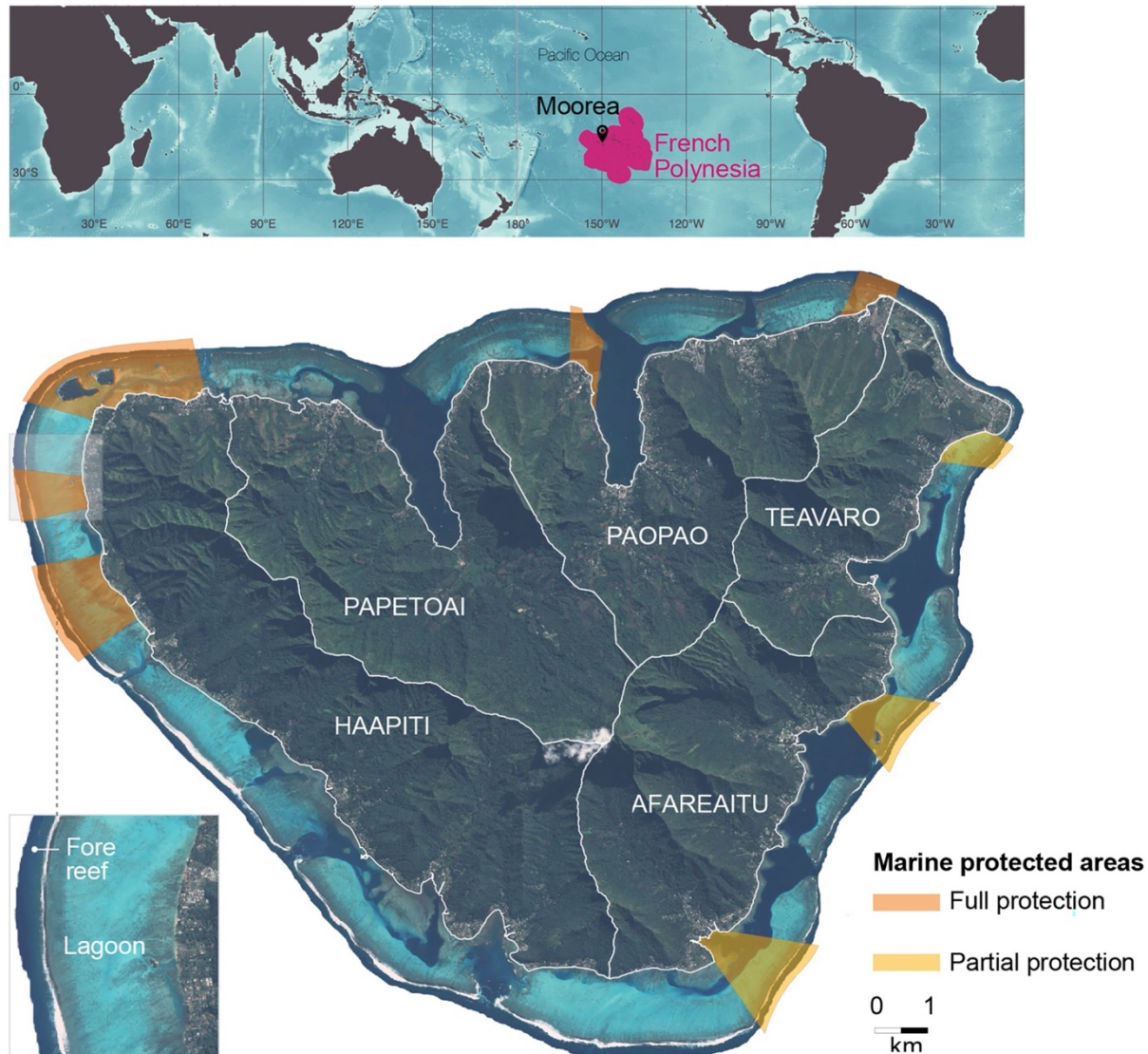


Fig. 1. Map of Moorea, French Polynesia, and marine protected areas of full and partial protection.

target, including economies and livelihoods. The reference point for these targets was an eight-year temporal window, instead of five, to compare the yearly state of the system to the existing condition prior to the cyclone (2009). Secondly, we calculated the whole OHI at specific one-year periods following Dubois et al. (2019): before the disturbances (2006), during the disturbances (2010), and after the disturbances (2018 or most recent available year).

For each one-year period, we calculated goal current status. Given data limitations for calculating the likely future status of the pre-disturbance and during-disturbance period following the OHI framework, we used current status as a proxy for the likely future status at the prior period. Thus, the current status of the post-disturbance period is used as the likely future status of the during-disturbance period. The same approach was used to estimate the likely future status of the pre-disturbance period. For OHI over time, we included only goals (sub-goals) for which enough data were available across the time series: artisanal fishing opportunities, food provision (artisanal fisheries), habitats, sense of place (iconic species and protected

places), coastal livelihoods and economies, tourism and recreation, coastal protection and carbon storage.

2.4. Reference point change exploration for coral data

Reference points have important influence over goal scores because they determine how all scores are scaled. Thus, setting reference points is a challenging but fundamental task for interpreting goal and index scores (Halpern et al., 2017; Halpern et al., 2012; Samhuri et al., 2012). Here, we explored how OHI scores might be sensitive to different reference points for coral-related goals (i.e., coastal protection and habitat). In line with previous OHI assessments, we used a temporal reference point for MOHI (Halpern et al., 2017; Selig et al., 2015). We tested how established targets for the coastal protection and habitat goals – 8 %, 10 % and 12 % hard coral cover following Darling et al., 2019 – would influence scores. Protection provided by coral reefs against sea level rise relies on the capacity of scleractinian corals to produce enough calcium carbonate to maintain

vertical reef growth, which is compromised by reef degradation (Carlot et al., 2021; Darling et al., 2019). The 10 % hard coral cover is considered a key threshold above which reefs are likely able to maintain net reef growth due to positive carbonate budget, and below which the carbonate budget is likely to be neutral or negative (Januchowski-Hartley et al., 2017; Perry et al., 2015), and might not provide defence against sea level rise (Darling et al., 2019; Perry et al., 2018). Following Darling et al. (2019), we also considered 8 % and 12 % hard coral cover thresholds. We thus calculated coral condition – used in habitat and coastal protection - using the above-mentioned reference points. For each of the reference points, the coral condition calculation is as follows:

$$h_{coral} = \frac{C}{C_{ref}}$$

where h_{coral} is coral condition. C_{ref} is the reference hard coral cover and can be either equal to 0.08, 0.10 or 0.12, respectively.

3. Results

3.1. Moorea's ocean health index (MOHI)

Overall, MOHI within lagoon and fore reef ecosystems scored 81 out of 100. The highest scoring goal was tourism and recreation (100) and the lowest scoring was sense of place (52), including the protected places sub-goal (34). Scores varied across goals as follows: tourism and recreation (100), artisanal fishing opportunities (99), clean waters (97), livelihoods and economies (95), biodiversity (75), food provision (73), coastal protection (70), and sense of place (52). The largest difference among sub-goals was found for sense of place, in which the protected places sub-goal scored 34 and iconic species scored 69 (Fig. 2, Table A.8). For nearly every goal or sub-goal, likely future status scores were similar to or higher than current status scores, except for the livelihood sub-goal, which was lower (Fig. 3, Table A.8).

The scores of the resilience dimension were on average 2.5 times higher than those of pressures. Regarding trends, four out of nine goals, namely carbon storage, artisanal fisheries, coastal livelihoods and economies, and sense of place, showed negative trends; three showed positive trends, including clean waters, coastal protection and biodiversity goals; and two showed no change, including artisanal fishing opportunities and tourism and recreation goals. There were disparities between sub-goal trends for biodiversity, sense of place, and coastal livelihoods and economies goals. Within the biodiversity goal, the habitat sub-goal showed a positive trend, while the species sub-goal trended negatively. For the coastal livelihoods and economies and sense of place goals, both the economies and protected places sub-goals exhibited no change, while iconic species and livelihoods sub-goals showed negative trends (Fig. 3, Table A.8).

The results of exploring three alternatives for established targets as reference points – 8 %, 10 % and 12 % cover of hard coral - used in the coastal protection goal and habitat sub-goal showed that MOHI scored 85 across alternatives. Coastal protection scored 99 and habitat scored 87 (Fig. A.1).

3.2. Changes in Moorea OHI over time

For goals and sub-goals with enough data to calculate yearly current status (status), we examined the change in goal current status over time. Habitat-based goals, notably coastal protection and habitat, exhibited large variability in current status. From 2006 to 2010, the during-disturbance period, the current status of coastal protection and habitats decreased 3 and 1.6-fold, respectively. From 2012 to 2018, the post-disturbance period, they increased 2.2 and 1-fold, respectively. The current status of the carbon storage goal declined 2-fold over the past 15 years. The artisanal fisheries goal revealed short-term fluctuations in status over the past 12 years, dropping by 1.2-fold in this period (Fig. 4).

The MOHI score for 2006, before the outbreak of predatory starfish and the cyclone, was found to be 79. In 2010, during such environmental disturbance events, the MOHI score was 74. In 2018 or more recently, after

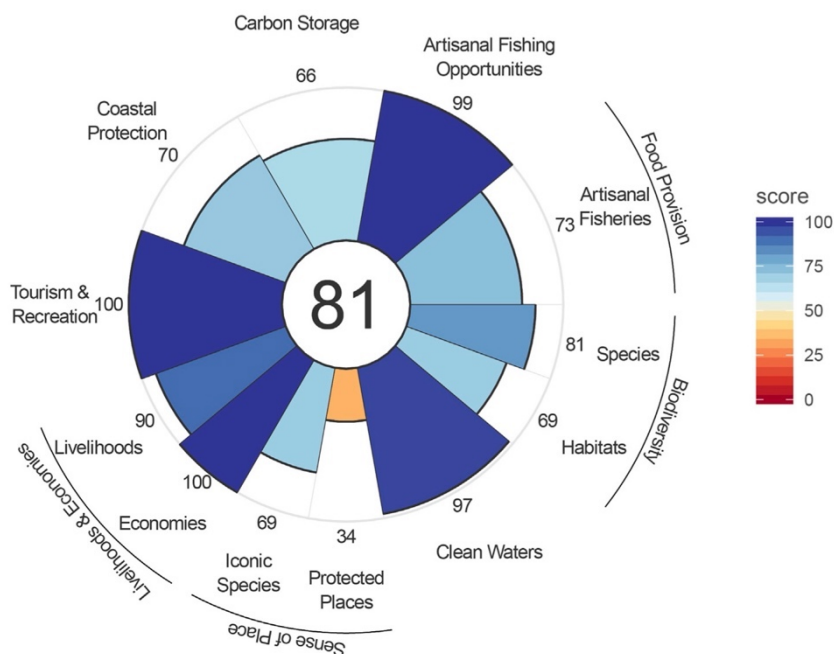


Fig. 2. Moorea Ocean Health Index scores for individual goals or sub-goals (coloured petals), and overall (central number) for 2018. Each petal represents the score (radius) and weight (width) for the goal or sub-goal. Scores range from 0 to 100, with colour ranging from red for low scores to blue for high scores.

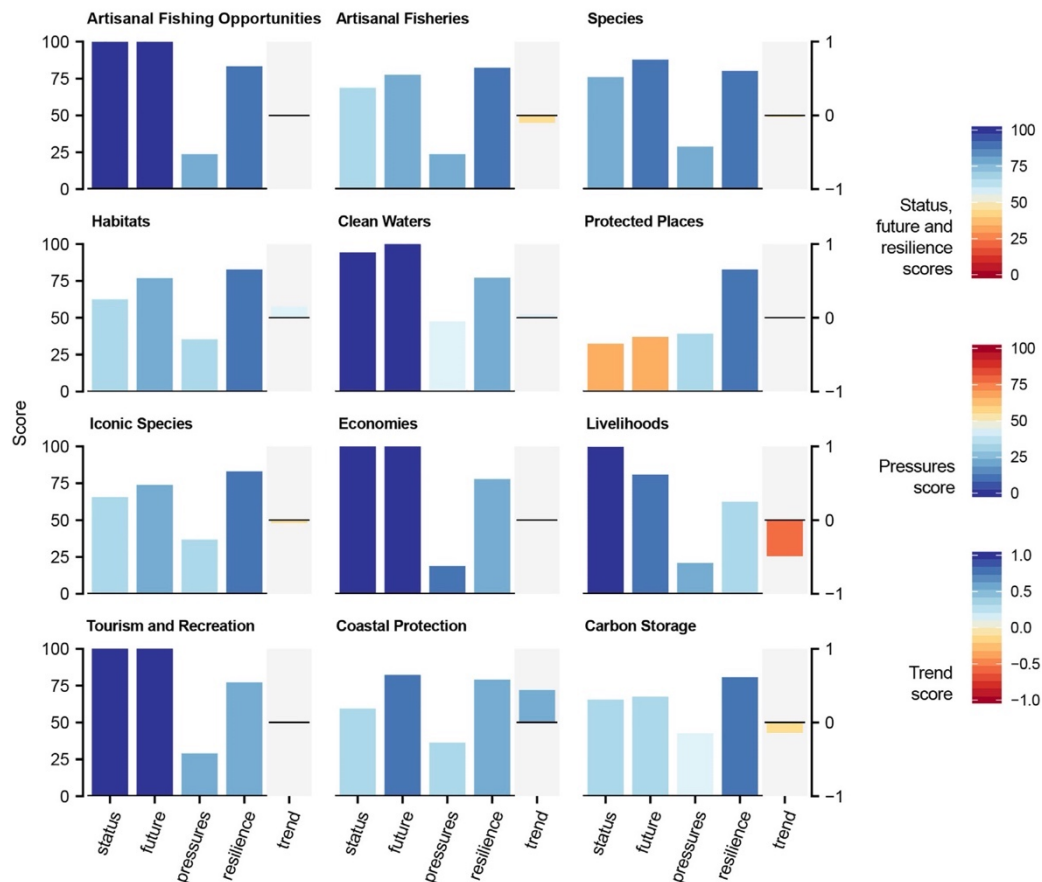


Fig. 3. Scores for current status (status), likely future status (future), pressures, resilience and trend per each goal or sub-goal. Trend ranges from -1 to $+1$, while status, future, pressures and resilience range from 0 to 100 . Colour ranges from red for lower scores and blue for higher scores, except for pressures where the colour scale is reversed, red meaning high pressure.

disturbances, the MOHI score rebounded to 78 (Fig. 5, Table A.9). Changes in scores ranged from approximately $+56\%$ to -28% , depending on the goal and disturbance period (Fig. 6, Table A.10). Similar to the current-status time series, habitat-based goals also showed greater change between 2006 and 2018. From 2006 to 2010, the coastal protection score dropped by 28% , the habitat goal by 19% , and the carbon storage goal by 12% . The opposite was observed during the post-disturbance period in 2018, in which the coastal protection score increased by 56% and habitats by 20% , though the carbon storage score still decreased, to a lesser extent, by 6% . For net changes to individual scores, coastal protection scored 12% higher, and habitat and carbon storage scored 3% and 17% lower, respectively, in the post-disturbance period than in the pre-disturbance period. For non-habitat related goals over the past decade, food provision and livelihoods scores decreased by 2% and 9% , respectively, while iconic species and tourism scores increased by 3% and 5% , respectively. Changes in scores of economies, protected places, and artisanal fishing opportunities were not observed in this study (Fig. 6, Table A.10).

4. Discussion

Here we present, to our knowledge, the first use of the OHI framework for tracking changes in ocean health through time before, during and after a regime of major environmental disturbances. We provided the first integrated and quantitative assessment of ocean health from a social-ecological perspective in Moorea. Our assessment of Moorea's Ocean Health Index (MOHI) revealed that overall ecosystem health declined during

disturbances, including cyclone and predatory starfish outbreak. While after the disturbance period the ecosystem recovered to the same overall OHI score as before the disturbances, we observed a reorganization of individual goal and sub-goal scores. Further, it showed that progress is needed to improve biodiversity conservation and better tailor the OHI fisheries model to data-poor fisheries to appropriately inform management.

Understanding how ocean degradation will affect its ability to deliver goods and services is key for securing them now and in the future (Cardinale et al., 2012; Halpern et al., 2019). While previous studies have tested the change in OHI scores over time in different regions (Halpern et al., 2015, 2017; O'Hara et al., 2020), as well as the sensitivity of OHI scores to different scenarios, such as habitat degradation (Longo et al., 2017) and management alternatives (O'Hara et al., 2020), our study is a novel application of the framework by incorporating the assessment of changes through major disturbances. We show that the overall MOHI score dropped 5 points from 2006 (before disturbances) to 2010 (during disturbances), representing a large change for a small period. Comparatively, global OHI assessment found that the greatest increase in OHI scores among regions was >1 point per year on average and the decrease was >-2 points per year on average (Halpern et al., 2017).

We found that the rate of decrease in current status was not constant across years (Fig. 6), and observed that major changes from current status in OHI score likely occurred from 2007 to 2008, following outbreak of the predatory starfish, and again from 2009 to 2010, following the cyclone. Although the overall MOHI score changed through major disturbance events, the post- and pre-disturbance MOHI scores were similar.

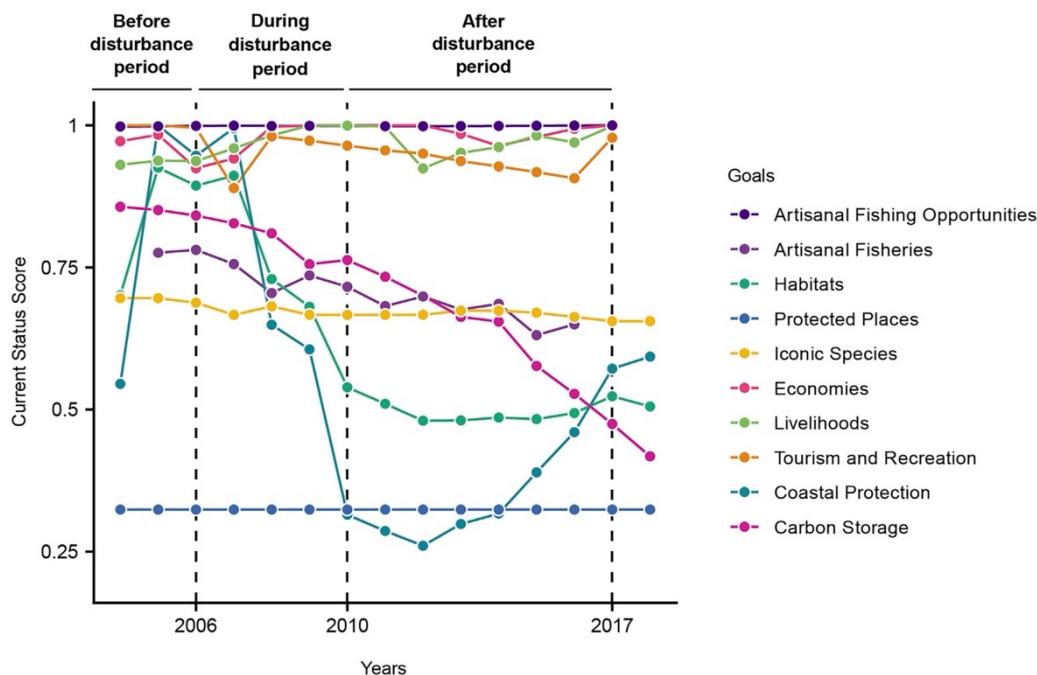


Fig. 4. Time series of current status (status) for goals and sub-goals with available historical data, ranging from 2004 to 2018 and capturing a period before, during, and after disturbance from a cyclone and an outbreak of coral predatory starfish.

Economies, livelihoods, tourism and artisanal fisheries opportunities scores were generally high and stable. However, no data were available to document the extent to which economic activities are environmentally sustainable. The observed changes in MOHI score were mostly driven by changes in live coral cover. These results reflect the high recovery potential of Moorea's coral reef social-ecological systems to cyclones and outbreaks of coral predatory starfish (Adjeroud et al., 2018, 2009; Dubois et al., 2019; Lamy et al., 2016, 2015; Vercelloni et al., 2019) and highlight the rebound potential of coral reefs (Graham et al., 2015; MacNeil et al., 2015; Perry et al., 2018). These outcomes are consistent with previous studies that revealed the capacity of coral reefs to recover from major climate disturbances, like cyclones (De'Ath et al., 2012; Guillemot et al., 2010; Perry et al., 2015).

We found that scores for habitat-related goals exhibited considerable changes over time. Carbon storage declined throughout the 10-year period, possibly due to coastal urban development and embankments that have led to a decrease in mangrove extent in Moorea (Meyer et al., 2021). Coastal protection and habitat scores declined over the course of the disturbances, driven mostly by loss of live coral cover. Coral reefs provide protection against natural hazards like cyclones, reducing wave energy by an average of 97%; however, this first line of defence is compromised by the degradation of coral reefs (Ferrario et al., 2014). In Moorea, the disturbances led to a significant decline in coral cover, changes in coral assemblages in the fore reef, and shifts in fish communities (Dubois et al., 2019; Lamy et al., 2016, 2015; Vercelloni et al., 2019), with potentially strong implications for community well-being (Harborne et al., 2017; Hughes et al., 2017a; Lau et al., 2019). Our results align with previous studies that revealed coral reef degradation following major disturbances (e.g., tropical cyclones, coral predation by crown-of-thorns starfish, and coral bleaching) (De'ath et al., 2012; Harborne et al., 2017; Hughes et al., 2017a).

We performed a sensitivity test on the reference point used for coral condition in coastal protection goal and habitat subgoal, and found that setting 10% hard coral cover as the reference point would result in the same score for 8% and 12% thresholds (Fig. A.1). The proportion of hard coral

cover decreased during disturbances, but was maintained above the 10% threshold across time (Fig. A.2), showing the potential of coral reefs in Moorea to sustain coastal protection benefits, similar to other coral reefs globally (Darling et al., 2019; Perry et al., 2015). A recent study of Moorea coral reefs showed that juvenile corals sustained carbonate production and accelerated reef recovery through rapid growth after disturbances (Carlot et al., 2021). Future work can improve the coastal protection model by considering the contributions of species' composition, structural complexity, and growth rate (Darling et al., 2019; Harris et al., 2018) for coastal protection benefits in the reference point, particularly since structural complexity is likely crucial for Moorea coral reefs to ensure coastal protection in the future (Harris et al., 2018).

The existing network of marine protected areas (MPAs) in Moorea underperforms (Stoner-Osborne, 2020; Thiault et al., 2019). Here, we show that iconic species and protected places scores were low but stable over time. The establishment of the MPA network provided positive, but limited, ecological benefits over nine years of protection, that were mostly detected in fully protected MPAs, while no positive effect was reported within moderately protected MPAs nor for non-harvested fishes (Thiault et al., 2019). The increase in biomass of harvested fishes inside fully protected MPAs were significant, but limited (Thiault et al., 2019). These limited ecological benefits were mainly explained by weak compliance and surveillance, though the response to protection might have been compromised by the high intensity of environmental disturbances after establishment of the MPA network (Thiault et al., 2019). The potential management failure of the MPA network in Moorea may reflect the local communities' opposition to the network. The current management plan does not adequately integrate important cultural ecosystem services, such as sense of place and traditional knowledge (Gill et al., 2019; Hunter et al., 2018). Further, the current management plan does not account for critical impacts from land-based activities (e.g., agriculture and urbanization), which contribute to more than half of the overall impact on the reef (Loiseau et al., 2021). Terrestrial, coastal, and ocean ecosystems should not be managed in isolation, but rather as a land-sea ('ridge to reef')

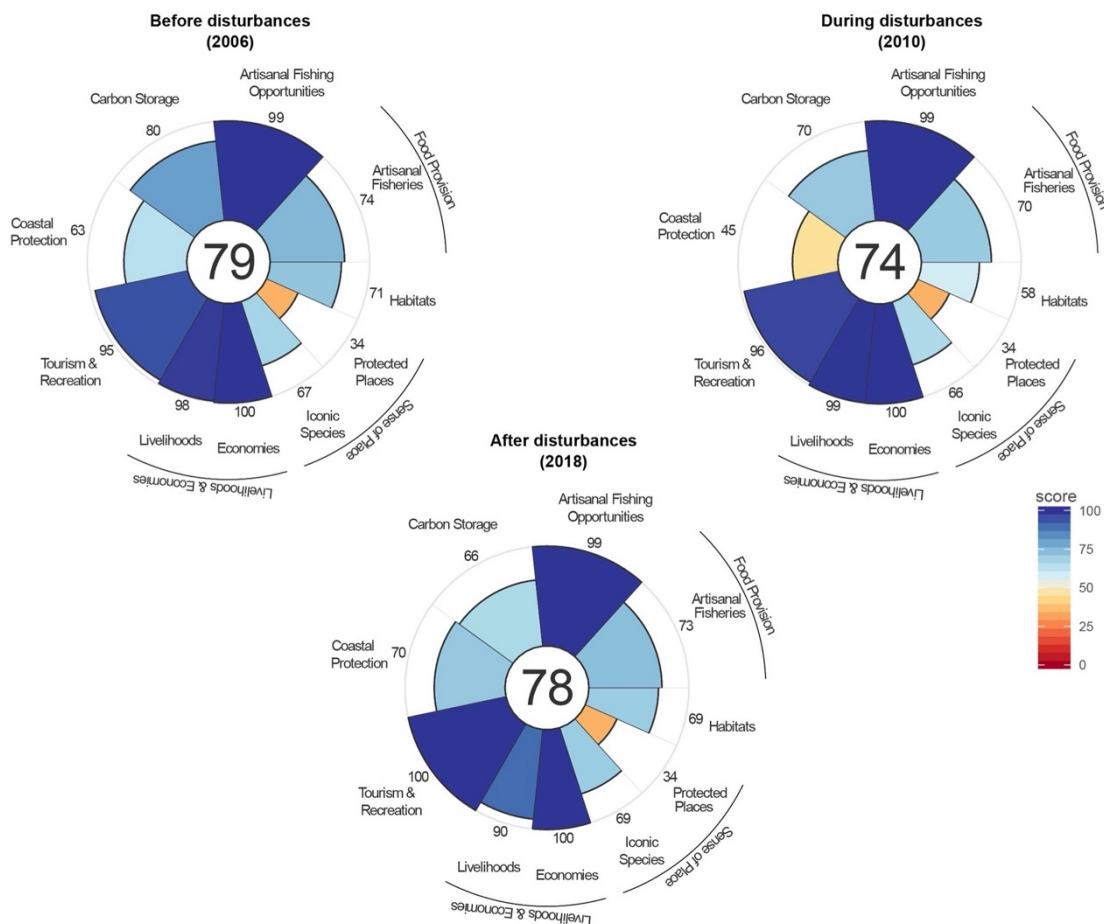


Fig. 5. Moorea's Ocean Health Index (MOHI) scores for individual goals or sub-goals (coloured petals), and overall (central number) along the disturbance regime period (MOHI is obtained for before, during, and after disturbance periods, respectively). Each petal represents the score (radius) and weight (width) for the goal or sub-goal. Scores range from 0 to 100, with colour ranging from red for low scores and blue for high scores.

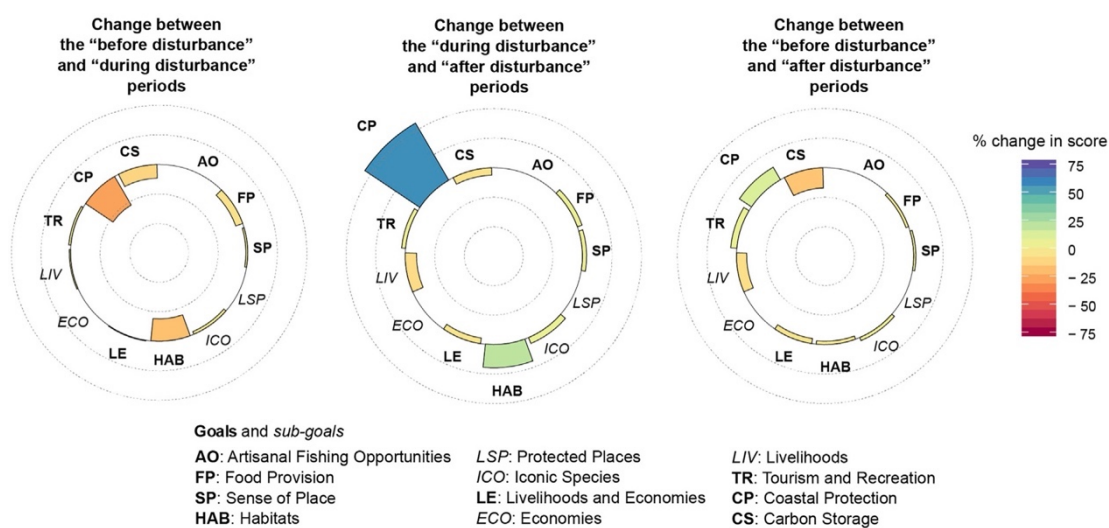


Fig. 6. Percent change in scores per goal or sub-goal between “before disturbance” and “during disturbance” periods, “during disturbance” and “after disturbance” periods, and “before disturbance” and “after disturbance” periods, respectively.

continuum. Considering land-based pressures in MPA management plans and efforts to connect marine spatial planning with current land-use planning are crucial to reducing cumulative impacts on the reef (Loiseau et al., 2021).

At present, the marine spatial management plan of Moorea is under review and shall integrate fewer MPAs, instead emphasizing co-management between local communities and land-sea authorities. Given this shift in approach, and the potential for locally managed areas to contribute to ocean sustainability targets (Reimer et al., 2020), the MOHI could be used in the future for MPA design or marine spatial planning in Moorea as a foundation for priority setting and evaluating trade-offs by modelling ocean health under varying management scenarios that are viewed as possible futures, rather than factual estimations (Halpern et al., 2014; Nguyen et al., 2018). Efforts to improve the MOHI for this purpose may include incorporating information about MPA network compliance and land-sea continuum integration into the coastal protection model, for example as pressures and resilience layers for the coastal protection goal, respectively, or as a component in the coastal protection status function.

Evaluating the stock status of artisanal fisheries, especially for coral reef fisheries, like those of Moorea, is often very challenging given the data-poor nature of most of these fisheries (Exeter et al., 2021; Machado et al., 2021; Rehren et al., 2021, 2022). The OHI fisheries model typically uses catch-MSY method to assess fish stock status (Halpern et al., 2012), though there is ongoing discussion about the use of catch-only methods to assess fish stock status (Afflerbach et al., 2019; Branch et al., 2011; Free et al., 2020; Froese et al., 2012; Pauly et al., 2013; Ovando et al., 2021). Afflerbach et al. (2019) assessed the uncertainty derived from the OHI fisheries goal model, and their findings support the use of catch-MSY. Further, the OHI fisheries model has been previously applied in data-poor fisheries (Selig et al., 2015; Hawaii, 2018). Here, our food provision goal relies on limited data derived from social surveys, with short time series and low yield (Dubois et al., 2019). Thus, our fisheries goal status presents a large degree of uncertainty (Afflerbach et al., 2019) that likely hinders the interpretation of food provision goal scores and fisheries status for Moorea. Quantifying the fish production and yields in Moorea is very complex, given that fishing takes place anywhere, at any time and there is no fishing market in Moorea (Leenhardt et al., 2012). These conditions make it difficult to determine the status of fish stocks and catch in Moorea's lagoon and fore reef. Surveys suggest that fish stocks are overexploited, but there is currently no sign of collapse in fish communities (Leenhardt et al., 2016).

Without overlooking the associated uncertainties and the biased related MOHI food provision goal, keeping this goal allows us to highlight the challenge that data poor-fisheries are facing to assess fisheries status and effectively manage fisheries. A more holistic OHI fisheries model through the incorporation of alternative data sources, such as self-reporting data by artisanal fisheries communities and local knowledge, could improve the application of the OHI framework in data-poor fisheries systems, reduce the uncertainty of outcomes, and better inform management (Van Gemert et al., 2021; Machado et al., 2021; Rehren et al., 2022).

The main caveats to the interpretation of scores are related to data quantity and quality, knowledge limitation and reference point selection (Halpern et al., 2017). Besides the earlier mentioned uncertainty associated with the food provision goal and reference selection for coastal protection, other sources of uncertainty pertain to gap-filling (Frazier et al., 2016), a widely used method to estimate missing values (Halpern et al., 2017, 2012, 2015). Details on data gap-filling and reference point selection are provided in the Appendix. Here, some datasets have been updated for the Moorea assessment and are specific to Moorea, while others were taken from the global OHI assessment for French Polynesia (Ocean Health Index, 2018b). The latter were mostly data layers linked to management and governance, which were not available at a local scale, such as strength of governance (see Appendix). While the use of exclusively local data would have been ideal for improving the reliability of management implications of MOHI, our study additionally serves as a gap analysis, highlighting where finer and more localized data collection and monitoring is needed.

The disturbance plus time series aspect of this study highlights the limitation of the likely future status of OHI to capture environmental disturbances, as it cannot anticipate them (Longo et al., 2017). For nearly all goals, the predicted future status was higher or similar to the current status. The model used for estimating the likely future status appears to overestimate the future scores for high-scoring regions, such as French Polynesia, implying an overestimation of goal scores (Halpern et al., 2017). Further, for all goals, resilience scored higher than pressures which may reflect a likely underestimation of the pressures dimension. Due to data availability, some reported pressures affecting Moorea's ecosystem were not included in this assessment, such as pig farming (Duane, 2006) or shark and ray feeding activities (Mourier et al., 2020).

Effective local management is crucial for sustaining ocean benefits and the people who depend on them (Cinner et al., 2020; Harvey et al., 2018; Jupiter et al., 2017; Shaver et al., 2018; Sterling et al., 2020). In this respect, OHI can be useful for assessing management effectiveness over time (Halpern et al., 2015; Lowndes et al., 2015). OHI can also support local management by providing insight into the condition of the marine social-ecological system, as well as through the OHI preparation phase beyond just the scores (Halpern, 2020). OHI assessments can reveal which goals should be prioritized (here, protection), where data collection efforts should be invested (here, fisheries data), or be used to raise awareness about linkages between environmental and socioeconomic states.

5. Conclusion

Applying the OHI framework in a marine social-ecological system across major disturbances provides a holistic understanding of the changing system and highlights the actions needed to improve its condition. For OHI to better inform local management, it is important to account for its limitations, derived uncertainties, and selected reference points. It offers the opportunity to explore how ocean health changes under environmental disturbances, which components are most affected, and the implications for the delivery of ecosystem services. Not only understanding, but also communicating, how disturbances alter ecosystem services is essential for mainstreaming the need for ocean protection and management.

CRedit authorship contribution statement

María D. Castro-Cadenas: Data curation, Formal analysis, Methodology, Visualization, Writing – original draft, Writing – review & editing. **Charles Loiseau:** Formal analysis, Visualization, Writing – review & editing. **Julie M. Reimer:** Writing – review & editing. **Joachim Claudet:** Conceptualization, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2022.156423>.

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RÉSUMÉ

Face à l'intensification des pressions exercées par les activités humaines sur l'environnement marin, une organisation rationnelle de ces activités est nécessaire pour endiguer la dégradation des écosystèmes sur lesquels reposent l'approvisionnement en biens et services des sociétés humaines. Les concepts récents de l'approche écosystémique et leur prometteuse application à la planification spatiale maritime apportent des réponses à ce défi, mais la complexité du tissu formé par les écosystèmes marins et les usages humains interagissant avec rend difficile l'opérationnalisation de ces approches. Les travaux de cette thèse proposent de replacer la conservation des écosystèmes marins au cœur du processus de planification, par l'intégration des principes de la gestion écosystémique au sein de la planification spatiale maritime, et par l'évaluation du rôle des aires marines protégées (AMP) comme outil préférentiel pour la conservation des écosystèmes au sein du processus de planification. L'évaluation de l'impact cumulé des activités humaines à une échelle pertinente pour la gestion à Moorea, en Polynésie française, a permis de mettre en évidence la forte contribution des activités terrestres dans l'impact cumulé à l'échelle de lagon, soulignant la nécessité d'une gestion intégrée du continuum terre-mer. Nous démontrons ensuite la valeur ajoutée de cette même approche afin d'orienter de manière rationnelle et transparente les décisions de gestion et notamment la contribution des AMP à effectivement réduire les pressions humaines en leur sein. En effet, les AMP sont un outil de zonage préférentiel pour la conservation des écosystèmes marins et s'intègrent ainsi dans la planification spatiale maritime, à condition de gérer efficacement les activités responsables de pressions délétères sur le milieu. Nous avons utilisé un système de classification des AMP basé sur les activités régulées ou non en leur sein pour évaluer le niveau de protection réel des AMP en France et en Mer Méditerranée. Les résultats indiquent que, même si certains pays ont atteint les objectifs de 10% d'AMP en 2020, la grande majorité des AMP ne présentent pas de niveaux de protection suffisants pour délivrer des bénéfices socio-écologiques. Afin de s'assurer que les AMP puissent remplir leur rôle dans la conservation des écosystèmes, les objectifs récemment fixés pour 2030 fixent désormais 10% de protection stricte ou forte. Dans cette optique, nous analysons quels facteurs expliquent la présence des plus hauts niveaux de protection en Mer Méditerranée, et les résultats montrent que les hauts niveaux de protection en Mer Méditerranée sont opportunistes et ne ciblent ni une biodiversité remarquable, ni des niveaux de pressions humaines élevés. Considérés ensemble, les résultats des différents chapitres mettent en lumière le manque de considération des objectifs de conservation, que ce soit au sein du MSP, par un manque d'approches dédiées localement à l'évaluation de l'impact cumulé des activités humaines et d'évaluation de différents scénarios de planification au regard des enjeux de conservation, et au sein des AMP, pourtant dédiées à l'atteinte de ces objectifs de conservation. Au-delà d'apporter ce constat, les différentes approches proposées ici permettent de replacer la biodiversité au cœur de la planification de l'espace marin.

MOTS-CLÉS

Gestion basée sur l'écosystème ; Planification Spatiale Maritime : Évaluation de l'impact cumulé ; Aires Marines Protégées

ABSTRACT

As the pressures exerted by human activities on the marine environment intensify, a rational organization of these activities is needed to stem the degradation of the ecosystems on which the supply of goods and services to human societies depends. Recent concepts of the ecosystem approach and their promising application to maritime spatial planning provide answers to this challenge, but the complexity of the tissue formed by marine ecosystems and the human uses interacting with them makes it difficult to operationalize these approaches. This thesis proposes to put marine ecosystem conservation back at the heart of the planning process, by integrating the principles of ecosystem-based management into maritime spatial planning, and by assessing the role of marine protected areas (MPAs) as a preferred tool for ecosystem conservation within the planning process. Assessing the cumulative impact of human activities at a scale relevant to management in Moorea, French Polynesia, has highlighted the strong contribution of land-based activities to cumulative impact at lagoon scale, underlining the need for integrated management of the land-sea continuum. We then demonstrate the added value of this same approach in guiding management decisions in a rational and transparent way, and in particular the contribution of MPAs to effectively reducing human pressures within them. Indeed, MPAs are a preferential zoning tool for the conservation of marine ecosystems, and can thus be integrated into maritime spatial planning, provided that the activities responsible for deleterious pressures on the environment are effectively managed. We used an MPA classification system based on regulated and unregulated activities to assess the actual level of protection afforded by MPAs in France and the Mediterranean Sea. The results show that, although some countries have achieved the target of 10% MPAs by 2020, the vast majority of MPAs do not have sufficient levels of protection to deliver socio-ecological benefits. In order to ensure that MPAs can fulfill their role in ecosystem conservation, the recently set targets for 2030 now set 10% of strict or strong protection. We therefore analyzed which factors explain the presence of the highest levels of protection in the Mediterranean Sea, and the results show that the high levels of protection in the Mediterranean Sea are opportunistic, targeting neither particular biodiversity components nor high levels of human pressure. Taken together, the results highlight the lack of consideration given to conservation objectives, both within the MSP, through a lack of approaches dedicated locally to assessing the cumulative impact of human activities and evaluating different planning scenarios with regard to conservation issues, and within the MPAs, which are nonetheless dedicated to achieving these conservation objectives. The various approaches proposed here make it possible to place biodiversity back at the heart of marine spatial planning.

KEYWORDS

Ecosystem-Based Management ; Marine Spatial Planning ; Cumulative Impact Assessment ; Marine Protected Areas