Interactions entre écosystèmes marins et pressions anthropiques : Applications au suivi et à la gestion des eaux côtières de la mer Méditerranée

Florian Holon

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Spécialité : **Ecologie Marine**

Présentée par **Florian Holon**

Interactions entre écosystèmes marins et pressions anthropiques
Applications au suivi et à la gestion des eaux côtières de la mer Méditerranée

Soutenue le 10 décembre 2015 devant le jury composé de

Mr David Mouillot, Pr, Université de Montpellier	Président du jury
Mr Carlo Nike Bianchi, Pr, Università di Genova	Rapporteur
Mme Sylvie Gobert, Pr, Université de Liège	Rapportrice
Mr Martin Daufresne, Dr, Irstea	Examinateur
Mr Nicolas Mouquet, Dr, Cnrs	Directeur de thèse
Mme Julie Deter, Dr, Université de Montpellier	Co-Directrice de thèse

UNIVERSITÉ
DE MONTPELLIER

THÈSE
de
DOCTORAT DE L'UNIVERSITÉ DE MONTPELLIER

Interactions entre écosystèmes marins et pressions anthropiques
Applications au suivi et à la gestion des eaux côtières de la mer Méditerranée

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Thèse préparée au sein de l'école doctorale GAIA
filière - Écologie, Evolution, Ressources Génétiques, Paléobiologie

Accueillie par l'Institut des Sciences de l'Evolution
de Montpellier

Soutenue par le Labex CeMEB
la société Andromède Océanologie

Sous la direction de Mr Nicolas Mouquet, Dr, CNRS Montpellier
et Mme Julie Deter, Maître de conférence associé, Université de Montpellier

Résumé

Au cours du siècle dernier, le développement des activités anthropiques a engendré une augmentation des pressions sur les écosystèmes marins côtiers. Les données scientifiques pour aborder ce phénomène sont fragmentaires et d'une précision insuffisante (de l'ordre du kilomètre) pour la prise de décisions locales. La gestion de ces menaces multiples et simultanées exige pourtant des données fiables et précises sur la répartition des pressions et la localisation des écosystèmes les plus sensibles. Ces constations ont été à la base de la rédaction de deux directives européennes (la DCE, directive cadre eau et la DCSSM, directive cadre stratégie sur le milieu marin).

Afin de répondre à ces directives, les objectifs de ce projet doctoral ont été de caractériser la vulnérabilité et d'aborder la sensibilité de deux écosystèmes marins prioritaires : les herbiers de posidonie et les communautés coralligènes. Ce projet focalisé sur la méditerranée française a permis la quantification et la localisation de ces écosystèmes et des pressions anthropiques auxquelles ils sont soumis, ainsi que l'estimation de leur état écologique à une résolution fine (de l'ordre de 20 mètres). Le déclin des herbiers a également pu être modélisé en relation avec les pressions et la profondeur et a permis de définir des seuils de résistance de cet habitat aux différentes pressions et d'identifier des zones prioritaires de gestion.

Ce travail devrait contribuer à améliorer l'élaboration d'indicateurs pour l'évaluation de l'état de conservation des systèmes côtiers. Il pourrait également aider à mieux appliquer et coordonner les mesures de gestion à une échelle pertinente pour la conservation de la biodiversité.

Abstract

During the last century, human activities were at the origin of increasing pressures impacting marine coastal ecosystems. Scientific data concerning this phenomena are fragmentary and the grid scale (1 x 1 kilometer pixel) is insufficient to permit relevant local decision making. The management of these multiple and simultaneous threats necessitates reliable and accurate data concerning the distribution of pressures and the localization of the most sensitive ecosystems. This is at the origin of two European directives: the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD). In response to these directives, the objectives of this work were to characterize the vulnerability and sensitivity of two priority ecosystems: *Posidonia oceanica* seagrass beds and coralligenous reefs. Focusing on the French Mediterranean coastline (depth ranging from 0 to – 100 m), results were: the fine-scale (20 x 20 m pixel size) quantification and localization of the encountered anthropogenic pressures, the quantification and localization of these two priority ecosystems, and the assessment of their ecological state. Seagrass beds decline was modelled in link with depth and pressures; resistance thresholds to pressures, but also management priority areas were defined.

This work should contribute to improve the development of indicators for the ecological state of coastal ecosystems. It should also help to better apply and coordinate management actions at a relevant scale for biodiversity conservation.

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Introduction

1 Spécificités de la mer Méditerranée

1.1 La Méditerranée, une mer semi-fermée

La *Mare medi terraneum*, en latin, décrit la Méditerranée comme une mer au milieu de la terre. Ce bassin est le plus grand (2,969,000 km²) et le plus profond (moyenne 1460 m, maximum 5267 m) des mers fermées de notre planète (Coll et al., 2010). Elle représente 0,8 % de la superficie marine mondiale (Defant, 1961) et pourtant elle contient presque 18 % de la biodiversité marine connue (Bianchi and Morri, 2000; Coll et al., 2010). Cette mer semi fermée est située entre Europe, Afrique et Asie. Elle se connecte *via* le détroit de Gibraltar à l'océan Atlantique à l'ouest, à travers le détroit de Bosphore à la mer de Marmara et la mer Noire dans le Nord-Est, tandis que dans le Sud-Est, elle est reliée par le canal de Suez à la mer Rouge et l'Océan Indien. Le détroit de Sicile divise la mer en deux bassins distincts, occidental (0,85 millions de km²) et oriental (1,65 millions de km²) (Coll et al., 2011) (Figure 1).

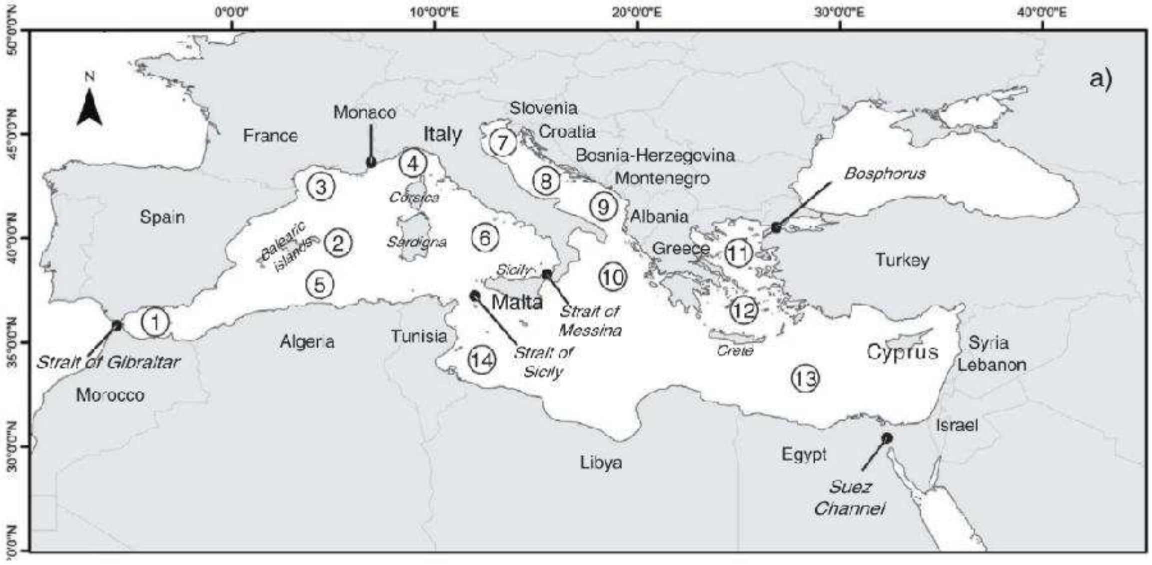


Figure 1: Différentes régions biogéographiques, bassins et divisions administratives. (1) Mer d'Alboran, (2) Mer des Baléares, (3) Golfe du Lion, (4) Mer de Ligurie, (5) Eaux Tunisienne et Algérienne, (6) Mer Tyrrhénienne, (7) (8) (9) Mer Adriatique, (10) Mer Ionienne, (11) (12) Mer Egée, (13) Mer Egée, (14) Golfe de Gabès. (Coll et al., 2011).

Le climat méditerranéen, caractérisé par de très faibles précipitations en été, induit un bilan hydrique négatif (Réunion des comités nationaux pour le programme hydrologique international des pays riverains de la Méditerranée, 1978). Ainsi les apports des fleuves et précipitations ne suffisent pas à compenser les pertes par évaporation. Ce sont les 1700 km^3 d'eau apportés par l'Atlantique, *via* le détroit de Gibraltar, qui viennent combler ce déficit (Oliver, 2003; Ribera d'Alcalà, 2003).

La Méditerranée a une histoire géologique variée, y compris son isolement de l'océan mondial, qui a conduit à son quasi-assèchement durant la crise messénienne (il y a 5,96 millions d'années) et à des changements drastiques dans le climat, le niveau de la mer, et la salinité (Garcia-Castellanos et al., 2009; Maldonado, 1985). L'histoire, la biogéographie, l'écologie, la géologie, et l'histoire humaine ont aussi contribué à la grande diversité culturelle et biologique de la Méditerranée (Boudouresque, 2004; Danovaro and Pusceddu, 2007; Fredj et al., 1992a; Myers et al., 2000).

1.2 Circulation des eaux en mer Méditerranée et apports nutritifs

L'eau de surface provenant de l'Atlantique entre par le détroit de Gibraltar et poursuit son chemin jusqu'en Méditerranée orientale. Durant cette traversée, l'eau chaude de surface (moyenne de 16°C) est appauvrie en nutriments (qui sont consommés) et, par évaporation, se concentre en sel (salinité pouvant atteindre 38,5 $g.l^{-1}$) (Bernardet et al., 2014; Lazzari et al., 2012). Cette masse d'eau devenue plus dense plonge alors pour former un courant froid (moyenne de 13°C) et profond, ressortant vers l'Atlantique (Millot, 1999).

La Méditerranée est reconnue pour sa diversité biologique (Bianchi and Morri, 2000; Coll et al., 2010). Pourtant, de manière globale, il s'agit aussi d'une région oligotrophe, c'est à dire très pauvre en nutriments comparée aux autres mers tempérées (Estrada, 1996; Krom et al., 1991; Stergiou et al., 1997). La concentration en nutriments est inégale et croissante en allant de l'ouest vers l'est (Moutin and Prieur, 2012; Pujo-Pay et al., 2011), les éléments nutritifs nécessaires au développement du plancton, base de la chaine alimentaire, provenant du déversement des fleuves (Béthoux et al., 1998; Ribera d'Alcalà, 2003). En hiver, les nutriments nécessaires à la vie sont apportés en surface par le mélange des eaux, puis consommés au printemps par le plancton (Mann and Lazier, 2013). Leur concentration diminue progressivement jusqu'à devenir très faible dans la couche supérieure de la masse d'eau, qui reste très stratifiée en été (Pasqueron de fommervault et al., 2015).

1.3 La Méditerranée, un point chaud de la biodiversité

Les courants d'eau de différentes compositions sont à l'origine d'une grande diversité d'habitats et d'espèces (Garibaldia and Caddya, 1998). La Méditerranée est en effet considérée comme un haut lieu de biodiversité compte tenu du nombre et de la diversité d'espèces abritées pour une mer tempérée (Myers et al., 2000). Elle héberge 4 à 18 % de la biodiversité mondiale (Bianchi and Morri, 2000), soit environ 17 000 espèces marines (Coll et al., 2010).

La biodiversité suit le gradient de production primaire, c'est-à-dire qu'elle est la plus importante au Nord-Ouest et décroit progressivement en allant vers le Sud-Est (Coll et al., 2010). Elle est généralement plus élevée à proximité de la côte et sur le plateau continental puis décroit avec la profondeur (Coll et al., 2010). Ces eaux du plateau continental représentent 20 % des eaux méditerranéennes et sont donc considérables comparées aux 7,6 % que l'on trouve dans les océans (Pinardi et al., 2006).

La Méditerranée est aussi un réservoir d'espèces endémiques : plus d'un quart des espèces qui y vivent lui sont propres (Fredj et al., 1992b). En effet, durant la crise messinienne datant d'il y a environ 5 millions d'années, la Méditerranée s'est refermée. Les espèces qui vivaient dans cette région y sont restées et se sont progressivement adaptées aux conditions environnementales, qui ont elles aussi changé avec le temps (Robba, 1987). Ces adaptations ont fait des espèces originaires de l'Atlantique et de l'Indo-Pacifique, des espèces aujourd'hui propres à la Méditerranée.

La Méditerranée orientale est considérée comme la région marine abritant le plus d'espèces exotiques envahissantes au monde (Edelist et al., 2013). Neuf cent quatre-vingt-six espèces marines non-indigènes ont été recensées dans le bassin méditerranéen, le groupe le plus représenté (215 espèces) étant les mollusques (Zenetos et al., 2012). Des centaines d'espèces, appelées lessepsiennes (Por, 1971), arrivent de la Mer Rouge depuis la création du Canal de Suez en 1869 (Galil, 2009; Zenetos et al., 2012). Le réchauffement climatique induit une accélération de l'invasion de la Méditerranée, à la fois par les espèces venues de Mer Rouge et par celles provenant des basses latitudes de l'Atlantique (Ben Rais Lasram and Mouillot, 2009).

L'abondance d'espèces exotiques envahissantes dans les zones côtières de Méditerranée aurait doublé ces vingt dernières années (Edelist et al., 2013). La migration de la Mer Rouge vers la Méditerranée serait favorisée par les eaux tempérées de Méditerranée moins abondamment peuplées que celles de Mer Rouge, avec des conditions propices (température et salinité élevées) dans la Méditerranée de l'Est (Por, 1971; Zenetos et al., 2010). Ces espèces sont en concurrence avec les espèces autochtones limitant leur propagation (Bax et al., 2003; Por, 1971). En parallèle, le changement global est à l'origine de disparitions de populations locales et de réduction des niches écologiques entrainant une diminution de la diversité génétique, la perte de fonctions et augmentant le risque de déclin ; tout ceci conduit à une homogénéisation biotique (Edelist et al., 2013; Galil, 2007). Cette menace écologique et économique a des répercussions globales sur la perte de la biodiversité, l'équilibre des écosystèmes, la pêche et la dépréciation du tourisme (Occhipinti-Ambrogi and Savini, 2003; Occhipinti-Ambrogi, 2007). Le réseau trophique méditerranéen est donc constitué d'espèces originaires de l'océan Atlantique, mais aussi d'espèces Indo-Pacifiques, qui constituent plus de 90 % des captures dans des régions de l'Est de la Méditerranée (Bariche, 2012; Edelist et al., 2011; Goren and Galil, 2005). La combinaison de facteurs anthropiques et naturels tels que la migration lessepsienne, les introductions d'espèces, le réchauffement de l'eau et les flux en provenance de l'Atlantique a conduit à une « tropicalisation » de la Méditerranée (Bianchi, 2007).

1.4 Changements globaux en Méditerranée

La Méditerranée, mer à l'interface entre climat semi-aride et continental, est qualifiée de principal hot-spot selon Giorgi (2006), suivie par les hautes latitudes de l'hémisphère Nord et l'Amérique centrale. Son environnement et les activités qu'elle héberge seront donc particulièrement impactés par le changement climatique (Giorgi, 2006). Les circulations d'eau pourraient aussi être profondément modifiées (Romanski et al., 2012) avec des flux sortants plus chauds et plus salés (Somot et al., 2006; Thorpe and Bigg, 2000) et un affaiblissement de la circulation thermohaline.

D'ici la fin du 21ème siècle, certaines modélisations envisagent une réduction de la circulation thermohaline de 40 % pour les eaux intermédiaires et de 80 % pour les eaux profondes (Somot et al., 2006). Le changement climatique impacterait donc l'écosystème marin entier. Les trois conséquences principales seraient l'élévation du niveau marin (et donc une augmentation de la superficie marine), l'élévation de température et enfin l'augmentation de la concentration en CO_2 (Laubier, 2003).

L'augmentation de la surface marine n'impacterait que peu les organismes puisque la surface gagnée serait colonisée par les mêmes espèces que celles déjà en place (Laubier, 2003). Cependant les organismes marins, et principalement les organismes fixés, devraient faire face à l'augmentation de température, induisant des épisodes de mortalité plus ou moins élevée. C'est ce qui s'est produit par exemple à la fin de l'été 1999 sur les côtes méditerranéennes françaises : une mortalité massive d'invertébrés a été révélée suite aux températures exceptionnelles de l'eau (23°C-24°C pendant plus d'un mois) (Perez et al., 2000; Romano et al., 2000). Le réchauffement des océans a été désigné comme un des principaux facteurs influençant les communautés marines (Pörtner and Peck, 2010), avec des effets sur leur distribution, taille, poids, période de ponte, mortalité, etc. (Pecl et al., 2014).

Les dernières simulations (Adloff et al., 2015) estiment, suivant les scénarios et pour la période 2070-2099 (par rapport à la période 1961-1990), une augmentation de la température de surface de la mer entre 1,73 et 2,97 °C (Figure 2) et une augmentation de la salinité entre 0,48 et 0,89 $g.l^{-1}$. Enfin, la réponse de l'augmentation du niveau de la mer varie de 34 à 49 cm (2070-2099 vs 1961-1990), principalement en fonction des eaux de l'Atlantique.

En effet, les facteurs influençant le plus le devenir de la Méditerranée, établis selon ces différents scénarios, seraient l'évolution des eaux de surface de l'Atlantique, suivie des choix socio-économiques faits par les pays côtiers (Adloff et al., 2015).

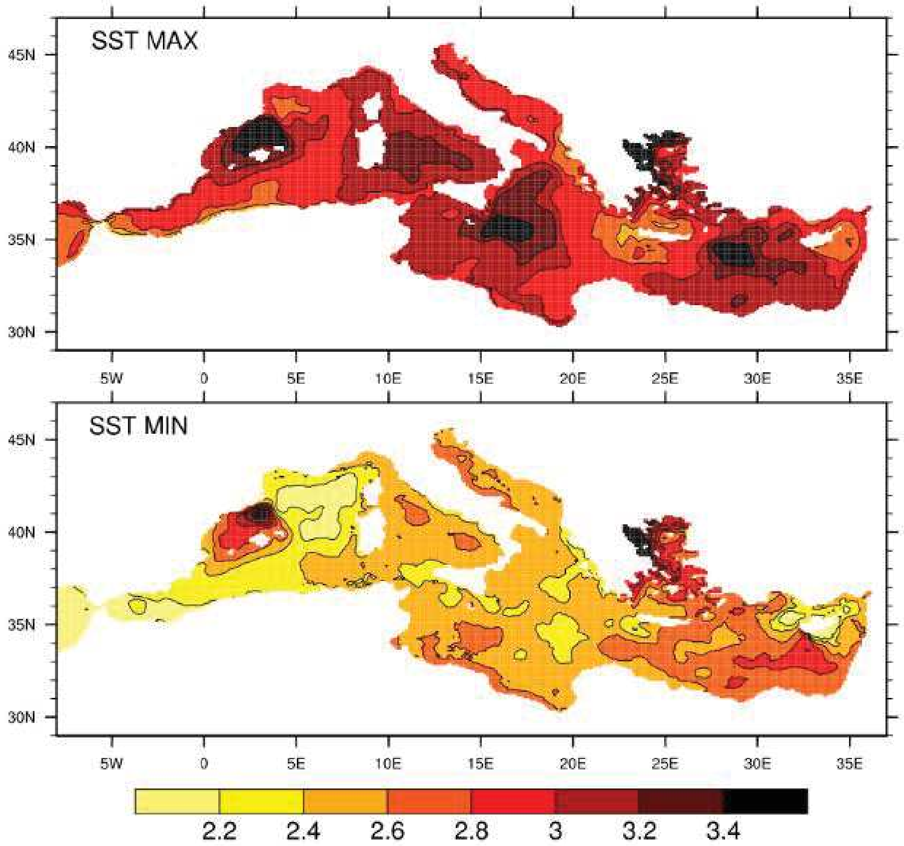


Figure 2 : Estimation des anomalies de température de surface, maxima (en haut) et minima (en bas) pour la période de 2070 à 2099 (par rapport à 1961-1990). Le plus grand (maxima) ou la plus petite (minima) anomalie sur les simulations de six scénarios est estimé à chaque point de grille. Les unités sont en °C. (Adloff et al., 2015).

Allant de pair avec les changements climatiques (Gorham, 1991; Harley et al., 2006), si les émissions de CO_2 ne diminuent pas, le pH des océans pourrait accuser une baisse de 0.4 (Riebesell, 2001). Cette baisse de pH affecterait la formation du squelette calcaire des certaines algues (comme les Coccolithophoridés) ainsi que la fixation du carbone par le phytoplancton. Ces organismes se situant au bas de la chaine trophique, tous les autres organismes pourraient alors en subir les conséquences (effet « Bottom up ») (Richardson and Schoeman, 2004).

1.5 Une population concentrée sur le littoral méditerranéen

Environ 7 % de la population mondiale vit dans les pays côtiers méditerranéens, soit 460 millions d'habitants auxquels il faut ajouter environ 200 millions de touristes internationaux.

Avec près de la moitié de la population méditerranéenne vivant près des côtes, le littoral méditerranéen figure parmi les secteurs les plus densément peuplés et les plus fortement urbanisés de la planète. D'ici 2025, ce pourcentage d'occupation des côtes devrait passer de 50 % à 88 %, et l'artificialisation de son littoral de 40 à 50 % (PNUE/PAM, 2009). Les côtes méditerranéennes et les écosystèmes qu'elles abritent sont donc soumis à des pressions humaines parmi les plus importantes de la planète (Costello et al., 2010).

L'influence humaine est présente en Méditerranée depuis des millénaires mais devient de plus en plus importante avec l'avancée technologique actuelle (Lotze et al., 2010). Cet impact est visible notamment sur le secteur de la pêche. L'évaluation des stocks de poissons en Méditerranée révèle que la plupart des espèces démersales et pélagiques sont pleinement exploitées voire surexploitées (Colloca et al., 2013; GFCM, 2013). La surexploitation et la destruction d'habitats marins sont les deux principaux facteurs humains ayant entrainé des changements de la biodiversité en Méditerranée (Coll et al., 2010).

La Méditerranée est aussi un lieu de trafic maritime intense. A chaque instant, environ 2000 bateaux traversent ses eaux. Les bateaux pétroliers effectuent environ 200 000 traversées par an, ce qui représente 28 % du trafic pétrolier mondial en mer (Laubier, 2003).

La Méditerranée connait d'autres menaces d'origine humaine telles que la pollution, la dégradation d'habitats, l'eutrophisation, et l'aquaculture (Coll et al., 2011).

2 Cadre réglementaire pour la surveillance du milieu marin

2.1 Contexte environnemental et DCE

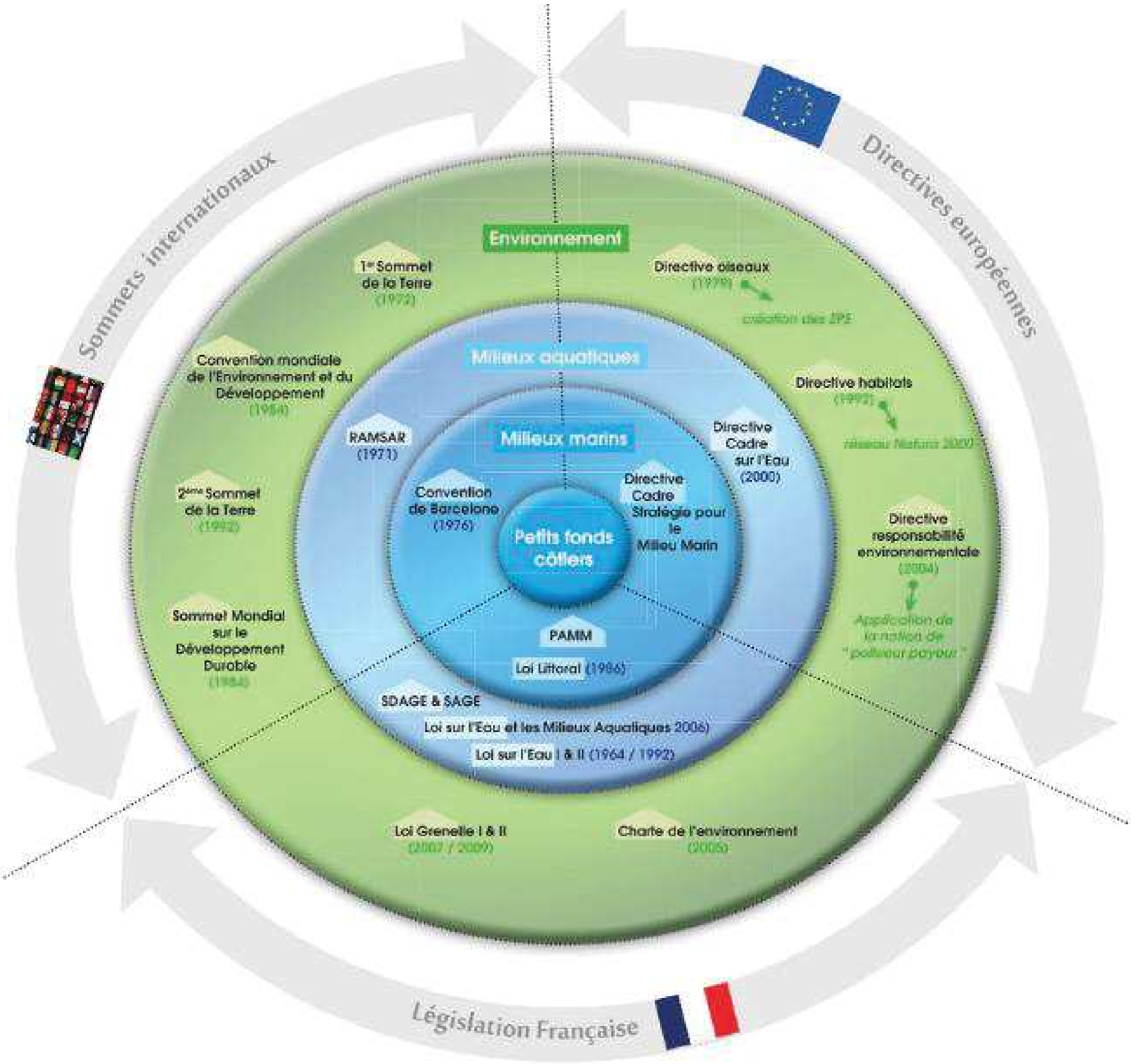


Figure 3 : Chronologie de la réglementation concernant l'environnement, le milieu marin et les petits fonds côtiers (Lenfant et al., 2015).

Les préoccupations françaises en matière de préservation de l'eau ont vu le jour dès les années 1950 sous la forme d'actions locales de dépollution de la qualité des eaux de surface (Vandenberghe, 2010). Pendant plusieurs années le principal secteur abordé reste celui de l'eau potable (Loi 64-1245 du 16 décembre 1964 relative au régime et à la répartition des eaux et à la lutte contre leur pollution), et dans les années 1970 émergent les premières mesures concernant les rejets toxiques (loi 76-663 du 19 juillet 1976). L'intérêt grandissant porté à la protection des aquifères s'est traduit par plusieurs initiatives comme la convention de Barcelone relative à la conservation de la Méditerranée, signée en 1976 ou encore la Convention de Paris pour la prévention de la pollution marine d'origine tellurique, entrée en vigueur en 1978 (Fig. 3).

En octobre 2000, l'Union Européenne instaure la Directive Cadre sur l'Eau (2000/60/CE, DCE). Cette directive vise à définir une politique commune quant à la gestion de l'eau, en se basant sur le fait que *« Les ressources en eau de surface ou souterraines, indispensables à la vie, doivent être protégées ».* Trois objectifs principaux sont définis: (1) Atteindre un bon état des eaux d'ici 2015 ; (2) Réduire progressivement les rejets, émissions, et pertes pour les substances classées prioritaires ; (3) Supprimer les rejets de substances prioritaires dangereuses avant 2021.

De manière plus générale cette directive incite à la création de plans de gestion intégrant des actions (i.e. le programme de mesures de la DCE) devant aboutir au bon état de l'eau (à la fois douce, souterraine, lagunaire et côtière). Ces plans d'action visent l'ensemble de la gestion de l'eau et des activités qui y sont liées. Ils définissent les actions prioritaires devant être menées, notamment concernant les prélèvements et rejets en mer. L'objectif majeur est d'utiliser durablement les ressources en eau. La DCE impose un objectif et une méthodologie commune, mais chaque état membre définit ses propres objectifs et moyens d'action. C'est une directive d'obligation de résultat et non plus de moyens, comme par exemple la directive ERU sur la gestion des eaux usées urbaines (1991).

L'Europe est divisée en grands bassins hydrographiques et la gestion de l'eau se fait au niveau de chaque bassin. La France compte 12 bassins hydrographiques (6 en métropole et 6 en outre-mer) (Figure 4).
Au sein de chaque bassin, les eaux littorales sont découpées en masses d'eaux côtières. Ce découpage se fait à partir de leurs caractéristiques hydrodynamiques et sédimentologiques. Depuis la loi sur l'eau de 1992, (*loi* n° 92-3 du 3 janvier *1992* sur l'*eau*) chaque bassin hydrographique français est géré selon un Schéma Directeur

Figure 4 : Les 12 bassins hydrographiques en France.

d'Aménagement et de Gestion des Eaux (SDAGE) sur 6 ans. Depuis 2000 ils se doivent d'intégrer les objectifs de la DCE fixés pour 2015. L'Agence de l'eau Rhône Méditerranée Corse (RMC) est le principal organisme en charge de la surveillance du milieu marin en Méditerranée française (Figure 5).

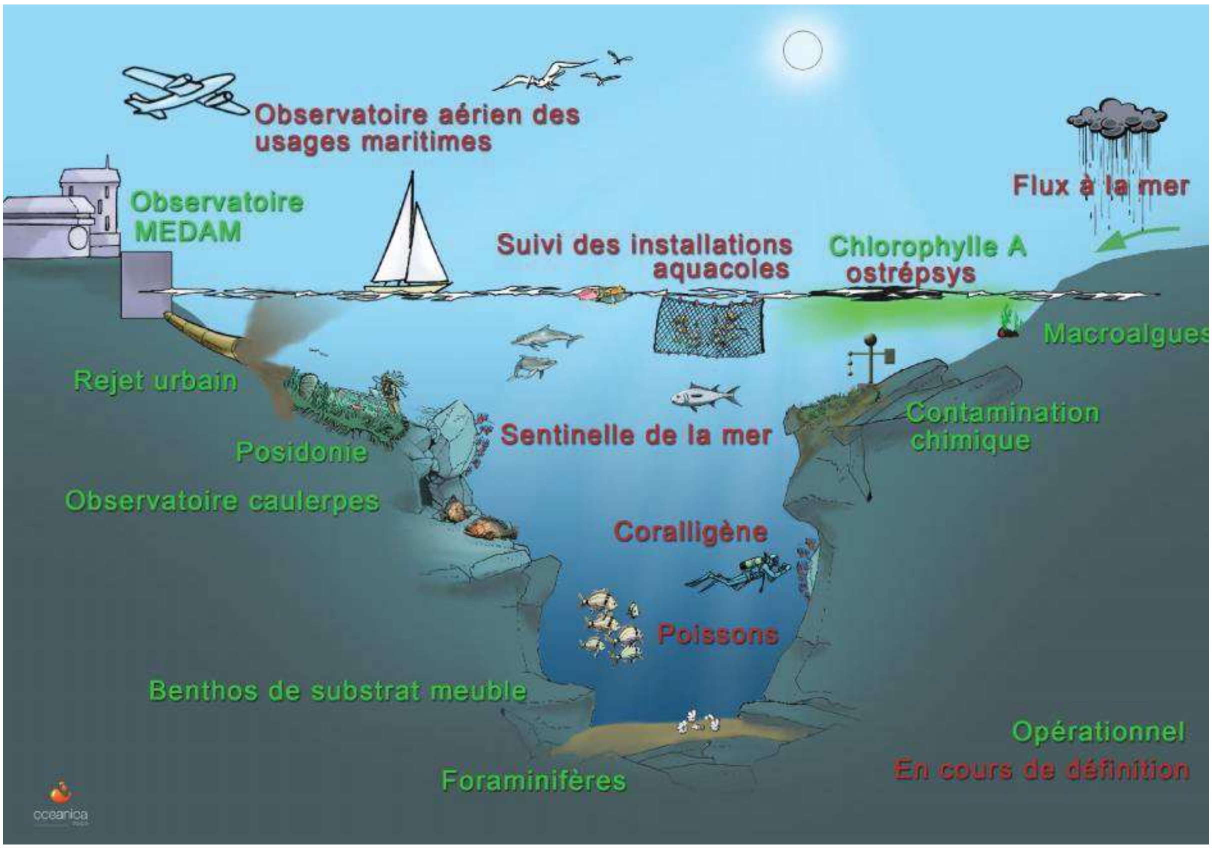


Figure 5 : Coordination de la surveillance du milieu marin en Méditerranée française par l'Agence de l'eau Rhône Méditerranée Corse.

2.2 Application spécifique au milieu marin - DCSMM

L'application de la DCE induit la mise en œuvre d'autres directives, de manière à gérer plus spécifiquement chaque secteur lié à l'eau. C'est dans ce cadre que paraît en juin 2008 la Directive Cadre Stratégie pour le Milieu Marin (2008/56/CE, DCSMM). Il s'agit d'une directive de l'Union Européenne qui établit un cadre et des objectifs communs pour la protection et la conservation de l'environnement marin en Europe d'ici à 2020.

L'objectif majeur est défini comme la lutte contre les « nombreuses menaces qui pèsent sur le milieu marin, tels que l'appauvrissement ou la dégradation de la diversité biologique et les modifications de sa structure, la disparition des habitats, la contamination par les substances dangereuses et les substances nutritives, et les répercussions du changement climatique ».

La notion d'« eaux européennes » est créée et implique une harmonisation des efforts pour la protection de la biodiversité marine. Afin d'atteindre cet objectif commun, chaque état membre doit évaluer ses besoins, élaborer et mettre en œuvre des plans de gestion cohérents, puis en assurer le suivi. Ainsi, en 2009, la France établit un Plan d'Action pour le Milieu Marin (PAMM) géré par les préfets maritimes des quatre régions (Manche-Mer du Nord, mer Celtique, Golfe de Gascogne et mer Méditerranée occidentale) (Fig. 6).

Le PAMM détaille les caractéristiques d'un bon état écologique pour chaque région et les actions à mettre en œuvre pour l'atteindre.

Figure 6 : Schéma récapitulatif des cinq étapes du PAMM (Plan d'Action pour le Milieu Marin).

Le Bon Etat Ecologique (BEE) est atteint lorsque « les masses d'eaux conservent la diversité écologique et le dynamisme d'océans et de mers qui soient propres, en bon état sanitaire et productifs et que l'utilisation du milieu marin soit durable, sauvegardant ainsi le potentiel de celui-ci aux fins des utilisations et activités des générations actuelles et à venir » (arrêté du 17/12/2012, conformément à l'article R. 219-6 du code de l'environnement).
La mise en œuvre de ces directives nécessite une évaluation ou un inventaire préalable des écosystèmes marins et l'utilisation de dispositifs de surveillance.

2.3 Mise en place de réseaux de surveillance

Le programme de surveillance, 4ème élément du PAMM (Figure 7), décrit l'ensemble des dispositifs de suivi à mettre en œuvre. Son élaboration est sous la responsabilité des préfets coordonnateurs de la région sous marine (dans le cas de la région marine Méditerranée il s'agit du préfet de la région Provence Alpes Côte d'Azur (PACA) et du préfet maritime de Méditerranée).

Le programme de surveillance doit permettre de déterminer si le programme de mesures répond aux objectifs environnementaux de BEE. Pour cela, des réseaux de surveillance liés à 13 programmes de surveillance thématiques sont mis en place (Figure 7). Ces réseaux permettent de récolter des données relatives aux 11 descripteurs de l'état écologique et ainsi de caractériser l'état écologique global d'un site.

Figure 7 : Les 11 descripteurs du BEE et les différents programmes et réseaux de surveillance qui leurs sont associés.

A plus long terme, les réseaux de surveillance ont pour but d'accroître les connaissances concernant les écosystèmes marins.

La France a choisi de mettre en œuvre le programme de surveillance selon une approche systémique. Cette stratégie de surveillance prend en compte :

- l’approche spatiale : zone d’étude, couverture géographique d’un descripteur ;
- l’approche temporelle : période d’acquisition des données ;
- le lien état/pression : concordance avec les mesures prises en termes de gestion ;
- l’organisation matérielle et la capacité à faire : utilisation d’opérateurs pérennes et facilement mobilisables, émergence de nouveaux acteurs.

Les données obtenues grâce aux réseaux de surveillance doivent ensuite être mutualisées et comparées. Pour cela, les données obtenues pour chaque élément de qualité sont comparées aux valeurs de référence pour cette région. Le rapport entre les deux définit le Ratio de Qualité Ecologique (RQE) compris entre 0 (mauvais état) et 1 (très bon état).

3 Protéger les écosystèmes marins des impacts des pressions anthropiques

3.1 Empreinte de l'Homme

L'empreinte de l'Homme sur les océans est si répandue qu'aucune région de l'océan mondial ne peut être considérée comme vierge (Jackson and Sala, 2001; Stachowitsch, 2003; Halpern et al., 2008). Il est ainsi admis que tous les écosystèmes marins sont affectés par la présence et les activités humaines (Glover and Smith, 2003). La Méditerranée, plus particulièrement, est considérée comme étant en « état de siège » (Coll et al., 2011) de par les nombreuses pressions qu'elle subit et les conflits entre usagers qu'elle héberge (Airoldi and Beck, 2007; Abdulla et al., 2008; Ferretti et al., 2008; Claudet and Fraschetti, 2010) : « *La mer Méditerranée doit faire face à de nombreuses menaces : densité de population très élevée, taux élevé d'urbanisation, érosion côtière, surexploitation des ressources marines et propagation des espèces envahissantes* » (PNUE/PAM-Plan Bleu, 2009). En France, les densités de population dans les régions côtières sont environ trois fois supérieures aux moyennes nationales (INSEE et SOes, 2004). Durant les 70 dernières années, poussé par la révolution industrielle, l'accroissement de la population a généré de nouvelles pressions le long du littoral : un développement urbain rapide, la construction de nouvelles stations balnéaires, de nouvelles marinas ainsi que l'extension de ports existants (Small and Nicholls, 2003). Ces multiples pressions s'additionnent dans un espace restreint et perturbent les écosystèmes (Crain et al., 2008; Darling and Côté, 2008; Doak et al., 2008; Halpern et al., 2008).

3.2 Conservation et gestion

L'efficacité des Aires Marines Protégées (AMP) sans prélèvements a été démontré comme des outils de conservation et de gestion importants avec des effets positifs sur la biomasse, la densité numérique, la richesse en espèces et la taille des organismes (Lester et al., 2009). Cependant ces AMP ne sont souvent pas suffisantes face aux pressions anthropiques (Agardy, 1994; Montefalcone, 2009). En effet, elles couvrent rarement les zones adéquates pour protéger les différents habitats, et sont bien souvent d'une superficie trop restreinte pour être efficaces (Mora et al., 2006). De plus, les AMP ne protègent pas de l'ensemble des pressions : elles ne peuvent, par exemple, prévenir de la pollution côtière ou de l'arrivée d'espèces invasives (Agardy, 1994; Halpern, 2003).

Une approche complémentaire de gestion (« management environnemental ») parle d' « Ecosystem-based Management » (EBM) en soulignant que les pressions multiples et leurs interactions avec les écosystèmes doivent être prises en compte de manière globale dans un espace défini (Ruckelshaus et al., 2008; Thrush and Dayton, 2010).

Comme prescrit par la Directive Cadre Stratégie pour le milieu marin (MSFD 2008), tous les états européens devaient évaluer l'état écologique de leurs eaux territoriales pour juillet 2014, et élaborer des stratégies pour atteindre le bon état écologique en 2020. Au niveau régional, les 21 pays méditerranéens ont ratifié le Plan d'Action Méditerranéen du PNUE (PAM) permettant d'orienter la gestion marine de la Méditerranée vers une approche écosystémique (ECAP), et de développer dans le bassin méditerranéen non-UE les mêmes mesures de conservation et de gestion mises en œuvre dans les eaux de l'UE (Micheli et al., 2013a). Ainsi afin de protéger les écosystèmes marins, de nombreux pays développent la mise en place de gestion écosystémique ou de planification spatiale marine (Crowder and Norse, 2008; Ruckelshaus et al., 2008), parfois même obligatoire (Rosenberg et al., 2009).

Une gestion écosystémique, pour être mise en place, nécessite d'identifier, de quantifier et de localiser les menaces pesant sur les écosystèmes (Last et al., 2010). Des études permettant de classer les menaces en fonction de leurs impacts sur les écosystèmes et de prioriser les actions à mener pour leur conservation ont vu le jour (Giakoumi et al., 2015; Olson and Dinerstein, 1998). Des cartographies de l'impact des pressions humaines ont été réalisées sur différents écosystèmes, comme les récifs coralliens (Bryant and Spalding, 1998), les herbiers à posidonie, le coralligène ou les grottes marines (Giakoumi et al., 2013). Ces premières cartographies, en localisant les menaces, ont constitué une aide à la gestion et à la conservation des écosystèmes marins.

3.3 Etude de l'impact des pressions anthropiques sur les écosystèmes marins

3.3.1 A échelle large : monde et Méditerranée

Les approches basées sur l'opinion d'experts sont souvent utilisées pour déterminer les impacts sur les habitats (Halpern et al., 2008, 2007; Claudet and Fraschetti, 2010; Parravicini et al., 2012). Bien que peu objectives et qualitatives plutôt que quantitatives, elles évitent les coûts importants des études de terrain et la difficulté de les mener à grande échelle.

En 2005, Zacharias et Gregr proposent d'évaluer la vulnérabilité (probabilité d'exposition à un stress) et la sensibilité (degré de réponse au stress) de plusieurs sites marins. Leur étude se base sur des zones vulnérables identifiées à partir des surfaces de stress acoustique pour deux espèces de baleines et de la mise en place d'un modèle de prévision de l'habitat. Cette première approche méthodologique basée sur des mesures de terrain peut être appliquée à d'autres combinaisons : écologiques/facteurs de stress (Zacharias and Gregr, 2005).

Par la suite, Halpern et al. (2007) ont proposé la mise en œuvre d'une méthode quantitative pour caractériser la menace qui pèse sur un écosystème donné. Leur étude tient compte des espèces jusqu'à l'écosystème entier, ainsi que du niveau de certitude de la menace. Les résultats montrent que tous les écosystèmes sont menacés par au moins neuf menaces (= pressions) différentes et qu'un quart des 38 écosystèmes qu'ils ont étudiés sont menacés par plus de 90 % des menaces existantes. Les menaces ayant le plus fort impact sont l'augmentation de la température de l'eau, la pêche démersale et les sources ponctuelles de pollution organique (Halpern et al., 2007). Les écosystèmes les plus menacés sont les récifs rocheux, les récifs coralliens, les mangroves, les plateaux durs et les écosystèmes épipélagiques (Halpern et al., 2007). Ensuite, une méthode standardisée et quantitative a été développée pour évaluer l'impact de 17 pressions anthropiques sur des écosystèmes spécifiques. L'étude se base sur les données actuelles (dix dernières années), plus faciles à analyser, et aboutit à une cartographie mondiale des pressions exercées en mer (Halpern et al., 2008) (Figure 8).

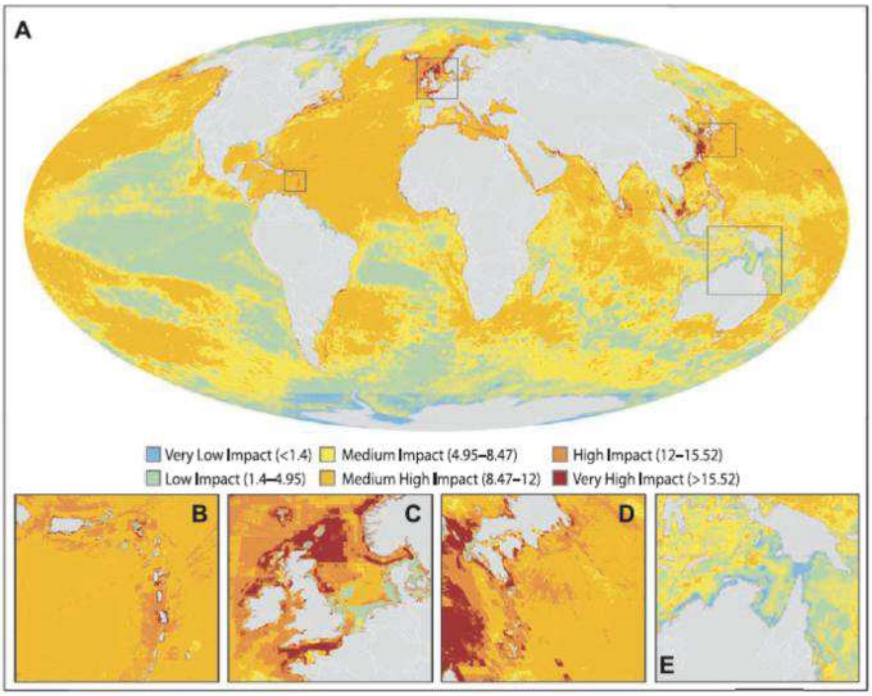


Figure 8 : ***Carte globale*** *(A)* ***de l'impact humain cumulé sur 20 types d'écosystèmes océaniques.*** **(Encadrés)** ***Régions hautement impactées dans l'Est des Caraïbes*** *(B)**,* ***la Mer du Nord*** *(C)* ***et les eaux japonaises*** *(D)* ***et une des régions les moins touchées, le détroit de Torres au nord de l'Australie*** *(E)**.*** **(Halpern et al., 2008)**

En utilisant une méthode similaire, l'étude de Coll et al. (2011) ciblée sur la Méditerranée a permis d'identifier les zones de forte biodiversité qui se trouvent le long des côtes, du centre et du nord du bassin. Le recouvrement de zones de haute biodiversité et de zones menacées se retrouve principalement sur les côtes espagnoles, dans le Golfe du Lion, au Nord-Est de la mer Ligure, en mers Adriatique et Egée, au Sud-Est de la Turquie, autour du delta du Nil et sur les côtes Ouest d'Afrique du Nord (Figure 9).

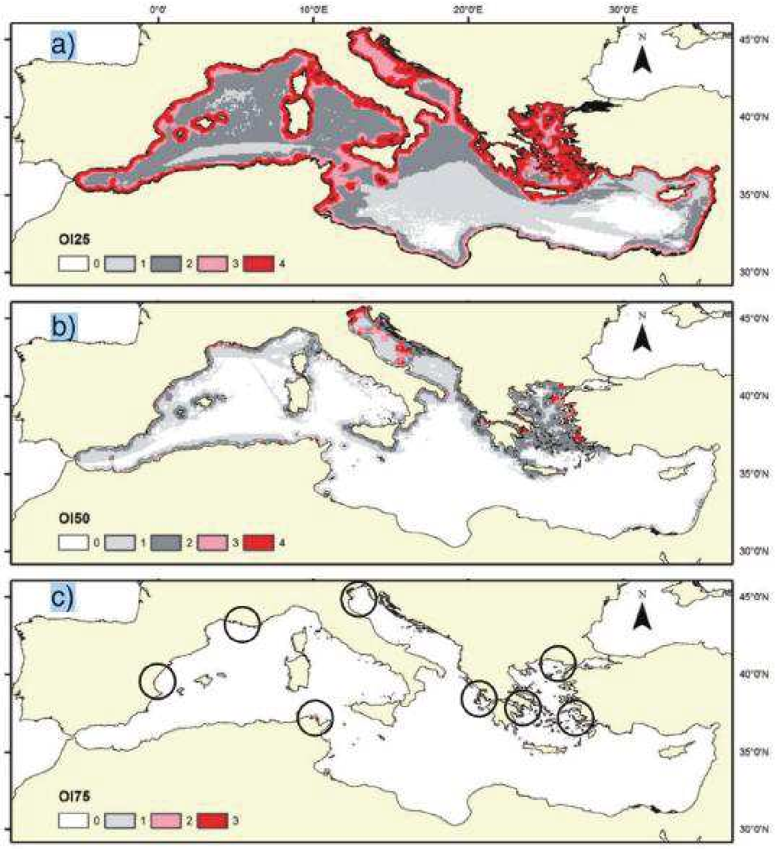


Figure 9 : Zones d'intérêt pour la conservation dans la mer Méditerranée où la diversité des invertébrés, poissons, mammifères marins, tortues et oiseaux de mer se chevauchent avec les menaces. L'indice de chevauchement (OI Overlap Index) indique les zones où à la fois la diversité des espèces et l'intensité cumulée des menaces étaient : (a) ≥25% (OI25), (b) ≥50% (OI50) et (c) ≥75% (OI75). 0 = pas de groupes (sur quatre groupements de biodiversité étudiées: les invertébrés, les poissons, les mammifères marins/les tortues et les oiseaux marins) montrant une forte diversité et des menaces cumulées élevées ; 1 = un seul groupe; 2 = deux groupes; 3 = trois groupes; et 4 = tous les groupes montrent une grande diversité et des menaces élevées. Les cercles noirs indiquent les cellules avec des données. (Coll et al., 2011)

Cette cartographie (Coll et al., 2011) basée sur des impacts cumulatifs appliqués à des taxons clés est complétée par une analyse globale sur les écosystèmes réalisée par Micheli et al. (2013) (Figure 10). Cette dernière étude, focalisée sur la mer Noire et la mer Méditerranée, prend en compte 22 pressions appliquées à 17 écosystèmes marins. Il en ressort que 20 % des eaux du bassin et que 60 à 99 % des eaux des états membres de l'UE sont fortement impactées. Il apparaît qu'une gestion coordonnée de certaines zones clés de la Méditerranée pourrait engendrer une amélioration considérable de l'état des écosystèmes marins (Micheli et al., 2013).

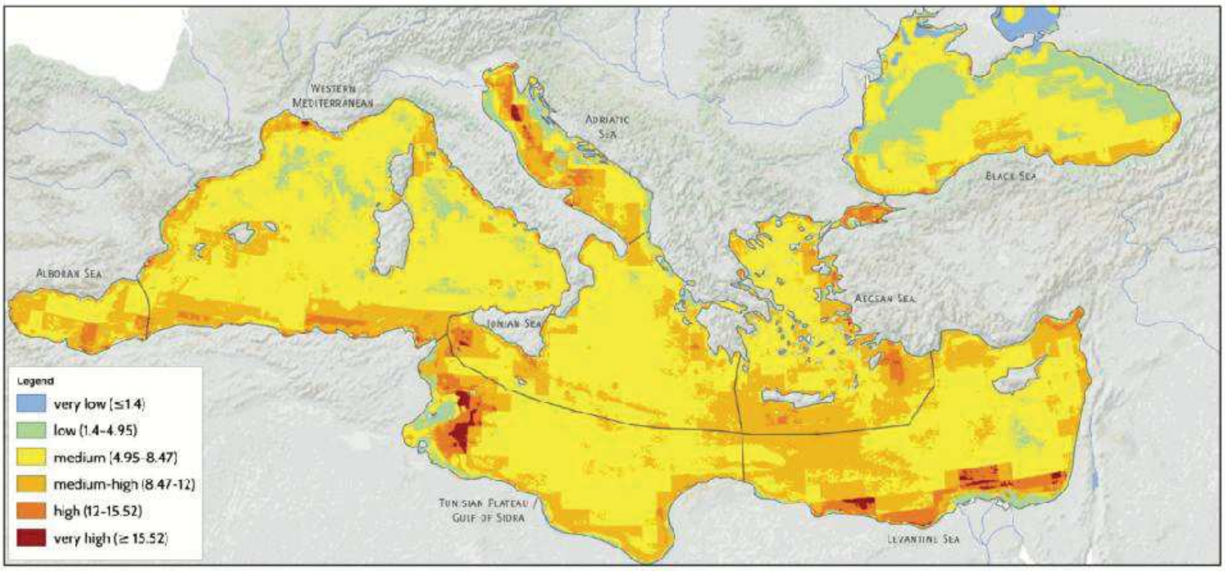


Figure 10 : Distribution spatiale des impacts cumulatifs sur les écosystèmes marins de la Méditerranée et de la mer Noire (Micheli et al., 2013).

Une autre étude de 2013 compile les informations disponibles sur la répartition des habitats : herbier à posidonie, formation à coralligène et grotte marine, à travers toute la Méditerranée, pour produire des cartes de distribution avec un pixel de 10 km de côté à l'échelle du bassin. Les objectifs de conservation pour chaque type d'habitat ont été établis selon les lignes directrices de l'Union Européenne (Giakoumi et al., 2013).

Le logiciel de planification de la conservation Marxan a été ici utilisé pour évaluer l'utilité comparative de deux scénarios de planification à l'échelle d'aire prioritaire ou selon des écorégions. Les conclusions suggèrent que la planification à l'échelle de l'écorégion assure une meilleure représentativité des caractéristiques de conservation sélectionnées et une protection adéquate des espèces, des fonctions, et de la diversité génétique dans le bassin (Giakoumi et al., 2013). Bien qu'il existe plusieurs initiatives qui identifient des aires prioritaires dans la mer Méditerranée, cette approche est nouvelle car elle combine trois problématiques: (a) la répartition des habitats et non des espèces, ce qui était rarement le cas dans les efforts précédents, (b) la variabilité spatiale des coûts autour de ce bassin socio-économique hétérogène, et (c) l'échelle « écorégion » comme le niveau le plus approprié pour la planification à grande échelle.

3.3.2 A l’échelle de la France

Le rivage méditerranéen français comprend 1700 km (dont environ 700 km pour la Corse) (Source : www.shom.fr) des 46 000 km de côtes méditerranéennes. Les précédentes études menées à l’échelle du bassin entier (Giakoumi et al., 2013; Micheli et al., 2013) montrant l’impact de l’homme sur les écosystèmes marins du littoral français sont à une échelle trop large (pixel supérieur au km²) pour permettre d’en tirer des informations utiles à une gestion locale sur la zone côtière française.

Le littoral français, comme le reste du littoral méditerranéen, est lui aussi victime de l’artificialisation du littoral : 4 800 000 habitants en région Provence-Alpes Côte d’Azur avec un taux d’artificialisation de 19,05 % et une région Languedoc-Roussillon (2 500 000 habitants) en pleine expansion (1,4 % de croissance annuelle, taux deux fois plus rapide que la moyenne métropolitaine) avec un taux d’artificialisation de 19,51 % (Sources : MEDAM et INSEE). A l’opposé la Corse, avec 300 000 habitants et deux millions de touristes par an, ne présente que 2,23 % de son trait de côte artificialisé (Sources : MEDAM et INSEE). L’urbanisation implique la présence de consommateurs, soit une demande accrue en ressources de toutes sortes (eau, énergie, matière première), d’accès aux espaces naturels pour des besoins récréatifs ainsi que des rejets divers dans les eaux, les sols et l’air. En 1990, il a été estimé qu'environ 10 % des fonds côtiers du sud de la France avaient été détruits par l'aménagement du littoral ce dernier siècle (Meinesz et al., 1991). Les aménagements (port, digue, plage artificielle, aéroport,...) entrainent une destruction irréversible des fonds infralittoraux par recouvrement ainsi que des perturbations considérables de l'équilibre hydrodynamique autour de ces constructions (Meinesz et al., 1991; Ruíz et al., 1993).

Les effluents urbains, après traitement, sont rejetés en mer *via* un émissaire afin d'en assurer la dilution. Il en résulte une grande variabilité d'influence spatiale du rejet suivant la dissipation des contaminants et leur dilution par les vagues et les courants (Bishop et al., 2002). Ces effluents ont clairement des effets négatifs sur les habitats marins et notamment sur les herbiers de posidonie (Balestri et al., 2004). A côté des villes, la croissance industrielle entraine une augmentation des contaminants aux effets négatifs sur l’environnement (Agunbiade et al., 2011). La pollution est impliquée dans la contamination de l'air, de l'eau et du sol par divers produits chimiques engendrant de fortes répercutions sur les écosystèmes entiers (Jijaescu, 2009).

L’agriculture représente également une part importante des impacts potentiels sur le milieu naturel et notamment marin côtier. L’accroissement incessant de la production pour les besoins alimentaires des populations a poussé l’activité agricole à utiliser des procédés artificiels d’optimisation : l’engrais et les pesticides. L’engrais, apporté en quantité supérieure à ce que les plantes peuvent absorber, est emporté par l’eau ou par le vent avant de pouvoir être absorbé. L’excès d’azote et de phosphate, lessivé, se retrouve alors dans le milieu marin (FAO, 2002). Les pesticides également sont retrouvés en grande quantité dans le milieu marin. La particularité de ces éléments est leur temps de dégradation très long et donc leur bioaccumulation dans la chaine alimentaire, avec des influences négatives sur la santé humaine (Van der Ooost et al., 2003 ; Vagi et al., 2007).

Enfin le tourisme, avec le doublement des touristes affluant vers les plages de la Méditerranée et les régions côtières attendu d'ici 2025 (Benoit and Comeau, 2005), joue un rôle crucial dans le développement intensif de nombreuses zones côtières, nécessitant ainsi la construction d'infrastructures importantes telles que les routes situées à moins d'un kilomètre et les parkings de bord de mer. Les impacts du tourisme sont en augmentation exponentielle depuis 1950 (Davenport and Davenport, 2006). Dans sa thèse, Peuziat (Peuziat, 2009) s'attache à identifier les impacts liés à la fréquentation nautique, dont les plus importants sont les suivants :

- Les pollutions engendrées par les bateaux (peinture anti salissures, anode sacrificielle, émission des gaz d'échappement,...) ;
- Les pollutions et nuisances dues aux pratiques et aux comportements des plaisanciers (mouillage forain, pollution organique et bactérienne *via* les rejets directs en mer, savons et détergents, élimination des macrodéchets, pêche, chasse et plongée sous-marine).

Aucun indice n'existe cependant pour mesurer l'impact du tourisme sur le milieu marin, leur quantification restant trop complexe (Peuziat I., 2009).

4 Deux écosystèmes méditerranéens particulièrement sensibles : les herbiers à posidonie et le coralligène

Deux écosystèmes contribuent à la majeure partie de la biodiversité, de la productivité et des services écosystémiques de la côte méditerranéenne : les herbiers à *Posidonia oceanica* (L.) Delile (Figure 11) et les récifs coralligènes (Figure 12) (PNUE/PAM, 2009). Menacés par les nombreuses activités humaines (PNUE/PAM, 2009), ces deux écosystèmes sont reconnus comme des habitats d'intérêt communautaire, dont la conservation requiert la désignation de zones de conservation spéciale au niveau européen (92/43/CEE Habitats Directive, habitat codes 1120 : *Posidonia* beds and 1170-14: Reefs, coralligenous assemblage).

Figure 11 : Paysage d'un herbier de posidonie (Dans la réserve de Tavolara (Sardaigne) © Laurent Ballesta)

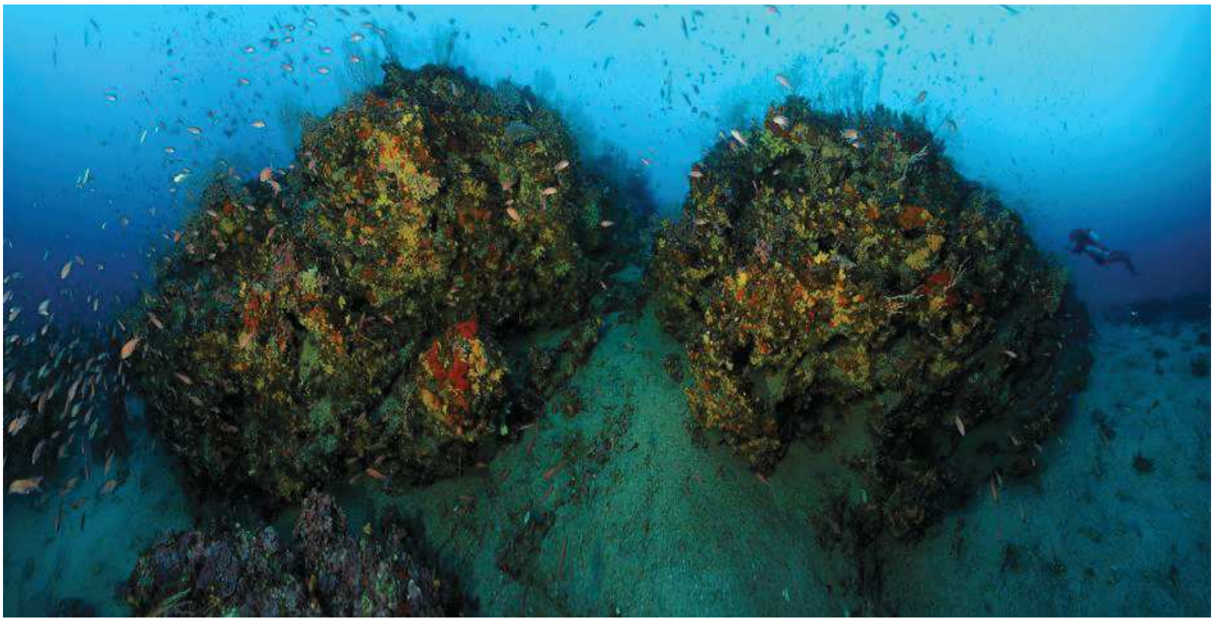


Figure 12 : Paysage de formations coralligènes (Au large de Fréjus (France)- © Laurent Ballesta)

4.1 Vulnérabilité et sensibilité des herbiers à *Posidonia oceanica*

4.1.1 Biologie de *Posidonia oceanica* (L.) Delile

La Posidonie, *Posidonia oceanica* (L.) Delile, est une phanérogame endémique de la mer Méditerranée qui peut constituer de véritables prairies sous-marines. Elle est présente sur une grande partie de la façade méditerranéenne, exceptée sur les côtes égyptiennes, palestiniennes, libyennes, israéliennes, le nord de l'Adriatique et certaines zones littorales des autres pays (Boudouresque and Meinesz, 1982; Por, 1978; Zalokar, 1942) (Fig. 13).

Cette plante angiosperme est constituée de faisceaux de 4 à 8 feuilles, longues de 20 à 80 cm en moyenne. Ces feuilles sont supportées par des rhizomes, tiges rampantes ou dressées, généralement enfouis dans le sédiment (Boudouresque et al., 2006). Les sédiments se déposent et s'accumulent entre les rhizomes, les obligeant à croître verticalement (rhizomes orthotropes) ce qui engendre une élévation du fond de la mer (Boudouresque et al., 2009). On nomme « matte » l'ensemble constitué par les rhizomes, les écailles (gaines des feuilles caduques), les racines et par le sédiment qui remplit les interstices (Boudouresque et al., 2006).

Exceptées les feuilles, les parties mortes de la plante sont peu putrescibles, ce qui explique leur longue conservation à l'intérieur de la matte qui peut persister pendant plusieurs millénaires (Boudouresque et al., 2006). Les feuilles tombent toute l'année et se retrouvent sur les plages sous forme de pelotes marrons. La floraison a lieu à l'automne mais ne se produit pas tous les ans. Elle serait provoquée par des températures élevées en été qui se prolongeraient jusqu'en octobre autour de 20°C (Caye and Meinesz, 1984; Pergent et al., 1989). Après six à neuf mois, les fruits mûrs se détachent et flottent pour disperser l'unique graine qu'ils contiennent (Boudouresque and Meinesz, 1982). De récentes études ont démontré que les jeunes plantules de posidonies colonisaient mieux les substrats rocheux que les étendues de sables (Alagna et al., 2015). En effet, les poils adhésifs présents sur leurs racines leur confèrent une force d'ancrage allant jusqu'à 5,23 N (Badalamenti et al., 2015) leur permettant ainsi de résister aux courants et tempêtes susceptibles de les déraciner. Cependant, la reproduction de *Posidonia oceanica* est majoritairement végétative et se fait par bouturage (Arnaud-Haond et al., 2007; Molinier and Picard, 1952).

Les herbiers à *P. oceanica* sont présents entre 0 et 40 mètres de profondeur. Ils occupent donc une surface totale restreinte (1 à 2 %) (Pasqualini, 1997; Rico-Raimondino and Pergent, 1995). La zone totale connue des herbiers à *Posidonia oceanica* dans la mer Méditerranée a été estimée à 1 224 707 ha (12 247 km²), 510 715 ha dans la partie ouest du bassin et 713 992 ha dans la partie orientale (Figure 13) (Telesca et al., 2015).

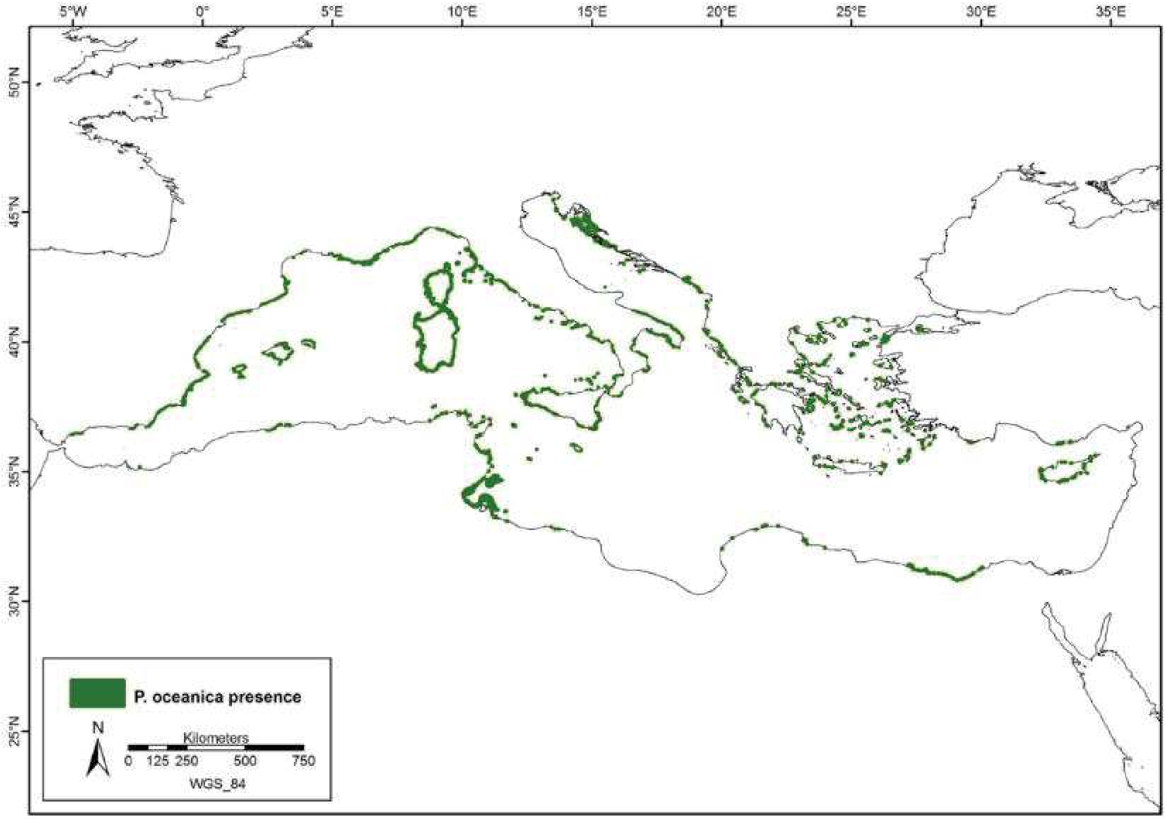


Figure 13 **: Position actuelle des herbiers à posidonie (Telesca et al., 2015).**

Cette plante marine est sensible à la salinité et ne se développe que dans des milieux de salinité moyenne (Fernández-Torquemada and Sánchez-Lizaso, 2005, 2003; Sánchez-Lizaso et al., 2008). Son absence, dans certaines zones, peut être liée aux concentrations en CO_2 trop faibles pour la réalisation de la photosynthèse, nécessaire à son développement (Romero, 2004). La lumière est aussi un facteur très important qui limite sa répartition en profondeur (Elkalay et al., 2003). Les exigences écologiques importantes de *P. oceanica* en font une espèce sentinelle : tout changement de sa répartition spatiale (par exemple une réduction de la limite maximale de profondeur ou une perte de zones couvertes), traduit un changement de l’état de l’environnement (Orth et al., 2006).

4.1.2 Rôles écologiques et économiques des herbiers à posidonie

Cette phanérogame constitue la biocénose la plus complexe de Méditerranée (Boudouresque and Meinesz, 1982; Molinier and Picard, 1952; Peres and Picard, 1964) et est à la base de la richesse des eaux littorales (Boudouresque and Meinesz, 1982; Cinelli et al., 1974; Molinier and Picard, 1952). Son rôle écologique est majeur : production primaire benthique, production d’oxygène (Bay, 1978; Bedhomme et al., 1983; Caye, 1980; Drew and Jupp, 1976), transparence des eaux par le piégeage des particules en suspension (Blanc and Jeudy De Grissac, 1984), base de nombreuses chaînes alimentaires (Velimirov, 1984), lieu de frayère, nurserie, abris vis-à-vis des prédateurs ou habitat permanent pour des milliers d’espèces animales et végétales (Boudouresque et al., 2006). Les herbiers à posidonie régulent, directement ou non, la disponibilité des ressources pour d’autres espèces et sont, à ce titre, qualifiés d'ingénieurs écosystémiques (Lawton, 1994). De ces rôles écologiques découle une quantité de services dont les Hommes bénéficient : protection du littoral, piégeage des sédiments, nourriture, etc. (Borum et al., 2004; Boudouresque et al., 2012; Campagne et al., 2015).

4.1.3 Conservation et déclin des herbiers : les pressions mises en jeu

Les préférences écologiques (eaux peu profondes et claires, courant faible) et les caractéristiques biologiques (croissance très lente de quelques centimètres par an) de *P. oceanica* rendent cette espèce très vulnérable aux pressions anthropiques (Boudouresque et al, 2012). C'est pourquoi *P. oceanica* et l'habitat qu'elle constitue sont protégés par la législation européenne (Directive Habitats, 92/93/CEE), par les conventions de Berne et Barcelone, par la législation nationale (Loi du 10 juillet 1976 et son décret d'application du 25 novembre 1977 (officialisée par l'arrêté interministériel du 19 juillet 1988 renseignant la liste des espèces végétales marines protégées)). De plus, *P. oceanica* est classée préoccupation mineure sur la Liste rouge de l'IUCN (Pergent et al., 2010).

Comme de nombreux habitats constitués par des phanérogames marines (Selig et al., 2014; Short and Wyllie-Echeverria, 1996; Spalding et al., 2003; Waycott et al., 2009), les herbiers à *P. oceanica* ont connu un déclin généralisé au cours des dernières décennies (Boudouresque et al., 2009). Ce déclin est caractérisé par un recul des herbiers peu profonds et/ou par une remontée des limites plus profondes, et donc une perte de l'étendue spatiale.

Les déclins d'herbiers sont principalement situés à proximité des zones urbaines (Boudouresque et al., 2012; Tomas et al., 2005) et sont surtout liés aux activités humaines, même s'ils peuvent parfois être dus à des processus naturels (par exemple, la colonisation et l'érosion dynamique, le changement climatique, le changement du niveau de la mer, les événements météorologiques, les événements exceptionnels tectoniques ou les maladies) (Duarte, 2002; Boudouresque et al., 2009; Pergent et al., 2014; Tuya et al., 2013). Une revue récente a montré que la régression de *P. oceanica* est pour les deux tiers (67,6%) attribuée aux impacts physiques d'origine humaine (Marbà et al., 2014b). En effet, la distribution des phanérogames marines étant fortement liée aux conditions de luminosité, la turbidité engendrée par les constructions entraine leur déclin (Ruiz and Romero, 2003). Les rejets des émissaires sont également mis en cause en raison notamment de l'apport en matières organiques et nutriments qui rendent l'eau turbide et qui s'accumulent au niveau sédimentaire (Kress et al., 2003).

Les fermes aquacoles entrainent le relargage dans les eaux de surface de nutriments et de matière organique qui affectent les écosystèmes d'une façon similaire à l'eutrophisation. L'impact environnemental des fermes aquacoles a fait l'objet de nombreuses études (Cannac et al., 2006; Porrello et al., 2005). Il a ainsi été montré que l'excès en matière organique et nutriments diminue la luminosité disponible par eutrophisation et que la dégradation de ces composés avec formation de sulfure entrainait une hypoxie sédimentaire, présentant un effet toxique sur les phanérogames marines (Holmer et al., 2008). Cet effet négatif se maintient et se propage après l'arrêt de l'exploitation en raison de la déstabilisation et de l'accumulation de matière organique au niveau sédimentaire (Delgado et al., 1998). Il a été montré que l'impact sur les herbiers et notamment sur la croissance verticale était observable jusqu'à 800 mètres des cages (Marbà et al., 2006), certains auteurs préconisant un placement des cages à une distance minimale de 400 mètres des herbiers (Holmer et al., 2008).

Les herbiers marins sont aussi connus pour souffrir des dommages mécaniques causés par l'ancrage des bateaux (Milazzo et al., 2004). La chaine des ancres entraine des dégradations importantes notamment dans les zones profondes et lorsque l'herbier est moins dense (Montefalcone et al., 2008).

De manière générale, d’autres causes de régressions sont recensées : les déchets solides et liquides (Boudouresque et al., 2012; Moreno et al., 2001; Pergent-Martini et al., 2002), les bateaux de plaisance et le tourisme de croisière (Boudouresque et al., 2012; Montefalcone et al., 2006; Okudan et al., 2011) et l'introduction d'espèces exotiques (Boudouresque et al., 2012; Marbà et al., 2014a). Cependant, l'influence quantitative relative de chacune de ces causes sur la baisse globale reste inconnue.

4.1.4 Quantifier le déclin des herbiers à posidonie

La tendance au déclin des herbiers marins, si elle se poursuit, sera un enjeu majeur dans le futur (Boudouresque et al., 2009). La régression des herbiers à *P. oceanica* en Méditerranée peut être étudiée *via* la persistance de la matte morte (Leriche et al., 2004). Cependant, l’étude de la matte morte (par datation au C^{14}) ne renseigne pas toujours sur la date de mort de l’herbier puisque les couches supérieures de la matte peuvent être érodées naturellement ou par l’action humaine. Cette érosion conduit à une estimation de la date de mort plus ancienne que ce qu’elle est réellement (Boudouresque et al., 2009). Par exemple, dans le golfe de Marseille, une bande de matte morte située entre 22 et 30 mètres de profondeur a été estimée au C^{14} comme datant de 573-885 alors qu’elle était considérée comme due à la pollution moderne (puisque la limite inférieure peut descendre jusqu’à -39 mètres au nord-ouest de la Méditerranée) (Gravez et al., 1992).
Une autre manière de quantifier les changements spatiaux de l’herbier est la comparaison de cartes au cours du temps (Boudouresque et al., 2006; McKenzie et al., 2001). Cependant, les cartes anciennes sont très rares et peu fiables (carte de 1883 par exemple à Marseille (Marion, 1883)). Les modifications spatiales de l’herbier observées sur les cartes au cours du temps doivent donc être étudiées avec précaution d’autant plus si les méthodes de cartographie diffèrent d’une date à l’autre (Leriche et al., 2004). .

Le déclin global (perte de surface) généralement accepté pour *P. oceanica* au cours des 100 dernières années est de dix pour cent (Boudouresque et al., 2012), mais une étude récente affirme une réduction de 50 % de la densité ou de la biomasse au sein de la Méditerranée au cours des 20 dernières années (Marbà et al., 2014a), avec une possible extinction fonctionnelle en 2059 selon d'autres (Jordà et al., 2012; Marba et al., 1996).

En 2015 une synthèse des études récentes estime que 124 091 ha de *P. oceanica* ont été perdus au cours des 50 dernières années, ce qui correspond à une régression moyenne de 10,1 % de la superficie totale connue (bassin méditerranéen) (Telesca et al., 2015). Si l'on considère uniquement les zones pour lesquelles il existe des informations historiques (368 837 ha), la perte estimée de *P. oceanica* est de 33,6 %. L'étendue des phénomènes de régression des herbiers à *P. oceanica* dans les 50 dernières années, sur la base de la comparaison des cartes historiques et actuelles disponibles , est montré sur la Figure 14.

La réalité reste difficile à évaluer en raison d'un manque de données historiques, des études se concentrant souvent sur de petites échelles spatiales et temporelles et/ou en utilisant des cartes anciennes incertaines ainsi que des études utilisant des méthodes de cartographies différentes (Bonacorsi et al., 2013; Montefalcone et al., 2013).

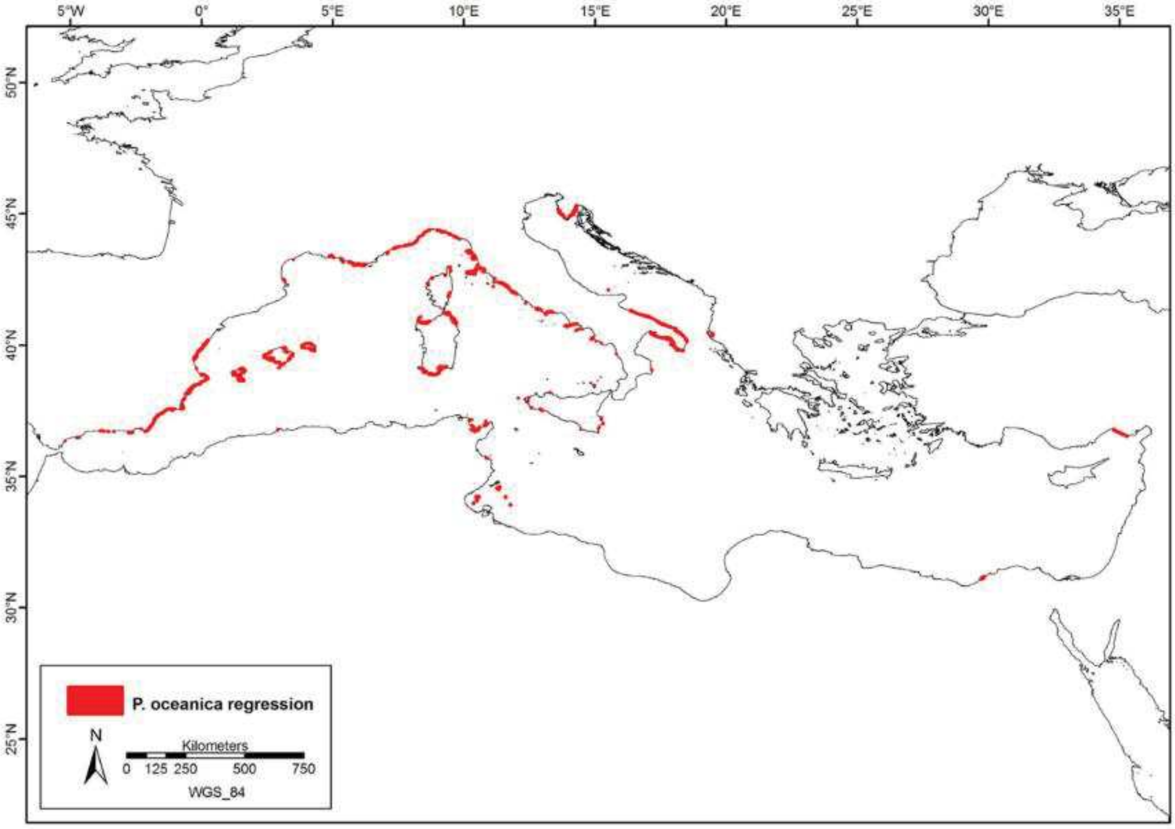


Figure 14 : Zones côtières avec régression des herbiers à *Posidonia oceanica* (Telesca et al., 2015).

4.1.5 La surveillance des herbiers à posidonie

Plusieurs programmes de surveillance ont été développés ces dernières décennies dans de nombreux pays comme en Espagne, Italie, ou Grèce (Boudouresque and Pergent, 1992). En France, le « Réseau de Surveillance Posidonies « (RSP) a été mis en place en 1984 en région PACA (Boudouresque et al., 2007). Depuis sa création en 2008, le réseau de suivi des herbiers de posidonie par télémétrie acoustique (TEMPO) (Holon et al., 2013) a été créé et s'étend sur l'ensemble de la façade méditerranéenne française. Les réseaux de surveillance ont deux buts principaux : (1) évaluer l'état de santé des herbiers à *Posidonia oceanica* et (2) utiliser ces herbiers comme indicateur de la qualité de l'environnement (Pergent et al., 2015). Le réseau TEMPO cartographie les limites inférieures de 84 sites en France méditerranéenne et évalue leur qualité écologique. Plusieurs protocoles développés récemment afin de calculer des indices de qualité écologique (à partir de mesures collectées à une profondeur intermédiaire de -15 mètres) sont utilisés : EBQI (Ecosystem-Based Quality Index) (Personnic et al., 2014), BiPO (Lopez y Royo et al., 2010), PREI (Gobert et al., 2009).

Suite à la création du programme européen COST 647 («Coastal Benthic Ecology Project », programme de la Commission des Communautés Européennes), d'autres pays ont initié un suivi de l'herbier comme l'Algérie (2000), la Grèce (2003) ou encore la Slovénie (Sartoretto et al., 2009).

4.2 Vulnérabilité et sensibilité des assemblages coralligènes

4.2.1 Caractéristiques biologiques du coralligène

Alors que certains groupes taxonomiques, comme les poissons et l'herbier de posidonie, ont été largement étudiés dans le bassin méditerranéen (Mouillot et al., 2011; Ruíz et al., 2009) et utilisés comme indicateurs pour les politiques environnementales européennes (Devlin et al., 2007; Gobert et al., 2009; Personnic et al., 2014), les écosystèmes marins plus profonds (au-delà de -40 mètres) restent peu connus (Cartes et al., 2004) et les efforts de conservation moindres (Abdulla et al., 2008).

En mer Méditerranée les systèmes littoraux profonds sont colonisés par des assemblages coralligènes qui débutent, selon la transparence de l'eau, entre -12 et -50 mètres et s'arrêtent entre -40 et -120 mètres (Ballesteros, 2006). Le coralligène se compose de blocs de concrétions organiques, principalement d'algues calcaires (Ballesteros, 2006; Laborel, 1961; Laubier, 1966; Sartoretto et al., 1996) et d'animaux bioconstructeurs comme les bryozoaires, les serpulidés, les cnidaires, les mollusques, les éponges, les crustacés et les foraminifères (Hong, 1980; Ros et al., 1985). Ces blocs représentent un substrat pour la fixation et la croissance d'autres organismes comme les colonies de corail rouge ou de gorgones. Les assemblages coralligènes sont donc un patchwork complexe de micro-habitats et d'espèces dont la richesse, la biomasse et la productivité équivalent à celles des assemblages de récifs tropicaux (Bianchi, 2001). Le coralligène constitue, après les herbiers à posidonie, le deuxième écosystème-clé de Méditerranée en termes de biodiversité (Boudouresque, 2004). Cependant, à surface égale, le coralligène pourrait être le premier écosystème benthique clé.

Les concrétions coralligènes ont un taux de croissance lent (1 à 4 mm / an) mais une grande longévité (estimation de l'âge moyen entre 25–200 ans) (Garrabou and Ballesteros, 2000; Teixidó et al., 2011). Elles sont dépendantes de l'équilibre entre leur bio-construction et la bio-érosion, équilibre lié à leur environnement et notamment aux conditions climatiques (Cerrano et al., 1997).

4.2.2 Rôles écologiques et économiques du coralligène

Selon des estimations prudentes, plus de 1800 espèces sont associées aux communautés à coralligène (15-20 % des espèces de la Méditerranée), parmi lesquelles figurent plusieurs espèces protégées et commerciales (Ballesteros, 2006; Bianchi and Morri, 2000; Coll et al., 2010). Elles font aussi partie des communautés écologiques les plus riches, comparables à des assemblages de récifs coralliens (Ballesteros, 2006; Bianchi and Morri, 2000). Egalement connues pour leur grande valeur esthétique, les structures à coralligène constituent des zones privilégiées pour le tourisme sous-marin et la plongée loisir. Ces assemblages sont parmi les plus importants en mer Méditerranée en raison de divers atouts : large distribution, complexité structurelle, diversité en espèces, rôle dans le flux d'énergie et le cycle du carbone, valeur économique (Ballesteros, 2006; Gili and Coma, 1998).

La carte réalisée par Giakoumi et al. (2013) (Figure 15) permet de répertorier à l'échelle de la Méditerranée la large répartition des concrétions coralligènes, leur conférant un rôle écologique majeur.

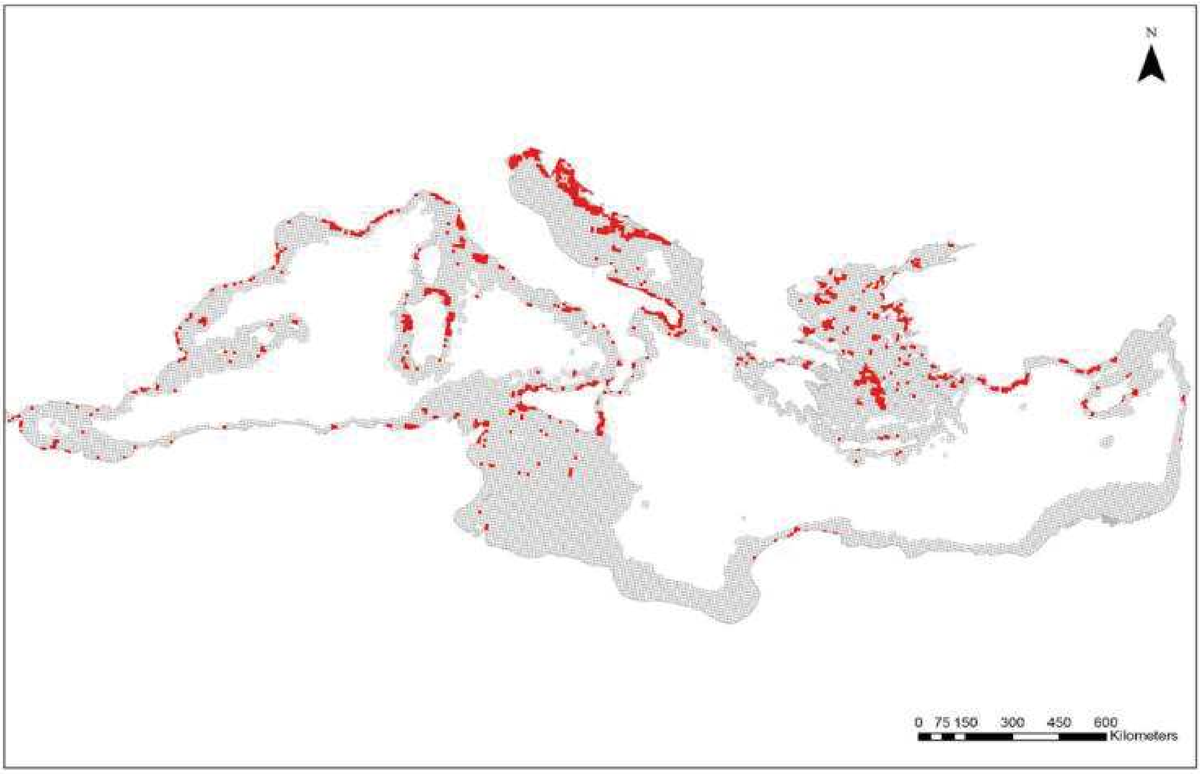


Figure 15 : Distribution des formations coralligènes en Méditerranée (Giakoumi et al., 2013).

4.2.3 Conservation du coralligène et pressions mises en jeu

Le coralligène est répertorié par la directive Habitats de l'Union Européenne comme habitat générique type des « récifs » et doit être protégé par les états signataires de la convention de Barcelone (Unep-Map-Rac/Spa, 2008). Cependant les surfaces de cet habitat dans les espaces d'Aires Marines Protégées françaises restent relativement faibles (Giakoumi et al., 2013). La distribution en taches de ces assemblages sur les fonds côtiers amplifie sa sensibilité à la fragmentation provoquée par l'Homme (Piazzi et al., 2012, 2011). Peu de données existent sur les capacités de dispersion des 1800 espèces constituant les assemblages coralligènes. Les quelques espèces longévives et inféodées à cet habitat qui ont été étudiées montrent de faibles distances de recrutement (de l'ordre de la dizaine de centimètres pour le corail rouge *Corallium rubrum*) comme de plus larges (160 km pour les gorgones rouges *Paramuricea clavata* (Ledoux et al., 2010; Mokhtar-Jamaï et al., 2011)).

Actuellement, les assemblages coralligènes sont parmi les écosystèmes les plus menacés. En effet, les espèces constructrices clés de ces assemblages présentent des taux de croissance lents, tandis que l'équilibre dynamique entre les processus de construction et de bio-érosion est particulièrement sensible aux changements environnementaux (Cerrano et al., 2001; Garrabou and Ballesteros, 2000; Sartoretto and Francour, 1997; Teixidó et al., 2011). Les perturbations d'origine anthropique directe ou indirecte auxquelles ils sont soumis comprennent principalement les dommages mécaniques causés par des pratiques de pêche destructrices, la pollution, la sédimentation, la fréquentation par les plongeurs, les invasions biologiques, les foyers de mortalité de masse liée à des anomalies de température, et les effets synergiques de ces facteurs de stress (Ballesteros, 2003; Cebrian et al., 2012; Piazzi et al., 2012, 2005; Teixidó et al., 2013).

Dans le bassin méditerranéen, la plupart des activités humaines sont liées à la profondeur. Par exemple la pêche récréative, la navigation et la plongée ont lieu dans les eaux peu profondes (moins de 50 mètres). Cependant, d'autres sources de perturbations telles que la pêche professionnelle, la destruction mécanique (par exemple le placement de câbles sous-marins) et la pollution provenant de sources terrestres ou marines peuvent impacter les zones plus profondes (Grall and Hall-Spencer, 2003; Meinesz and Blanfuné, 2015). Les conséquences, à plus ou moins long terme, des activités humaines sur le coralligène ont généralement été étudiées en se basant sur des espèces cibles (Bavestrello et al., 1997; Cupido et al., 2009; Garrabou et al., 2009, 2001; Giuliani et al., 2005; Linares et al., 2005). Une des premières menaces concerne la pollution. Une étude menée en 1983 dans le golfe de Fos-sur-Mer a mis en évidence l'influence des diverses sources de pollution, comme les eaux usées, sur l'appauvrissement de la faune coralligène, en nombre et en diversité (Hong, 1983). Il a aussi été observé une diminution du nombre d'espèces infralittorales dans le coralligène au profit d'espèces à large distribution écologique (Hong, 1983, 1980). De plus, les observations ont révélé que les zones polluées sont plus abondamment peuplées par des espèces bio-érosives (Hong, 1980). De manière générale, la pollution empêche la bio-construction et accélère la destruction des écosystèmes coralligènes (CAR/ASP, 2003). L'impact des polluants à proprement parler reste peu connu, il a toutefois été démontré que les ions orthophosphates ($H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}), présents dans certains engrais utilisés en agriculture par exemple, empêchaient la calcification (Simkiss, 1964).

La pêche au chalut de fond est aussi une des activités humaines ayant d'importantes et durables conséquences sur les communautés épibenthiques comme les récifs (Halpern et al., 2007). Cette menace est à l'origine de la destruction de grandes zones coralligènes par destruction directe des récifs ou par augmentation de la turbidité et de la sédimentation (Boudouresque et al., 1990). Cet impact augmente avec la profondeur car plus les organismes vivent loin de la surface, plus leur taux de croissance est faible et leur temps de récupération est long (Thrush and Dayton, 2010; Watling and Norse, 1998). La pêche du corail rouge a aussi été la cause de grandes destructions de récifs coralligènes, comme cela a été rapporté par Ortiz (1986) qui décrit que pour 15 kg de corail rouge pêchés, 50 kg d'organismes benthiques (principalement des gorgones) ont été prélevés en mer d'Alboran.

De même, les pêches traditionnelles et de loisirs ont des conséquences négatives sur les écosystèmes coralligènes. La pêche de certaines espèces cibles entraine une diminution de leur nombre et ainsi des changements dans les communautés coralligènes et les poissons associés (Bell, 1983; Garcia-Rubies and Zabala, 1990). Les récifs coralligènes les plus profonds restent moins touchés par les activités de pêche de par leur localisation plus discrète (Garcia-Rubies, 1999).

Les récifs coralligènes font partie des sites les plus appréciés par les plongeurs de loisirs en Méditerranée, notamment pour leur richesse et leur diversité. L'important développement de la plongée ces dernières années a entrainé une augmentation des interactions entre touristes et organismes marins (Luna-Pérez et al., 2009; Semeniuk et al., 2009). Certaines espèces fragiles comme le bryozoaire *Myriapora truncata* (Pallas, 1766) sont utilisées pour évaluer l'impact de la plongée loisir sur les communautés coralligènes en Méditerranée (De La Nuez-Hernández et al., 2014). L'étude de cette espèce avant et après la saison touristique a révélé que le nombre d'individus et la surface colonisée par cette espèce étaient significativement réduits dans les zones exposées aux plongeurs. Une autre étude a démontré que l'affluence de plongeurs entrainait une diminution de la densité des colonies d'un autre bryozoaire érigé, *Pentapora fascialis,* de 50 % en un an, après ouverture aux plongeurs d'un site jusqu'alors protégé aux îles Mèdes (Garrabou et al., 1998). Dans cette même région il a été observé qu'une fréquentation importante de plongeurs, de l'ordre de 1000 par site et par an, multipliait par trois la mortalité des gorgones rouges adultes *Paramuricea clavata,* déstabilisées ou renversées par les plongeurs (Coma and Polà, 1999; Coma et al., 2001). D'autres études ont mis en évidence l'impact de la plongée loisir sur les gorgones (Linares et al., 2005), les bryozoaires (Sala et al., 1996), les ascidies (Luna-Pérez et al., 2011, 2010; Naranjo et al., 1996) ou les poissons (Milazzo et al., 2006).

C'est la présence sur le long terme de nombreux plongeurs cumulés sur une période estivale restreinte qui engendre un impact significatif sur les communautés coralligènes (Plathong et al., 2000; Tratalos and Austin, 2001), par la remise en suspension d'argile limoneux ou par contact physique (fragmentation ou dommages tissulaires) (Barker and Roberts, 2004; Luna-Pérez et al., 2009; Uyarra and Côté, 2007). En présence de pressions de fortes intensités, les communautés coralligènes pourraient être dominées par des organismes incrustants ou massifs au détriment des espèces érigées trop exposées aux mouvements des plongeurs (Garrabou et al., 1998).

Les activités humaines sont aussi à l'origine de l'entrée de nouvelles espèces en Méditerranée, que ce soit par le trafic maritime, l'ouverture du canal de Suez, etc. Certaines espèces d'algues introduites envahissent le coralligène et le menacent. C'est le cas par exemple de la petite algue rouge *Womersleyella setacea* (Athanasiadis, 1997) qui recouvre les algues constituant le concrétionnement coralligène en formant un tapis dense. Cette espèce exotique envahissante bloque ainsi l'accès à la lumière, accroit le piégeage sédimentaire (Airoldi et al., 1995) et entre en compétition avec d'autres espèces autochtones (Ballesteros, 1998; Piazzi et al., 2002). D'autres algues peuvent envahir les récifs coralligènes comme *Caulerpa taxifolia* qui a envahi des communautés coralligènes au Cap Martin (Meinesz, 1999), ou encore *Asparagopsis taxiformis* (Ballesteros and Rodriguez-Prieto, 1996) et *Lophocladia lallemandii* (Patzner, 1998) aux Baléares.

Enfin, même si des particules sédimentaires qui atteignent les récifs coralligènes peuvent être incluses dans les concrétions, prélevées par les espèces brouteuses ou remises en suspension (CAR/ASP, 2003), des apports sédimentaires trop élevés peuvent être nocifs pour les récifs coralligènes. Ces excès de sédiments recouvrent le thalle des algues encroûtantes, limitant ainsi leur accès à la lumière (Laborel, 1961) ou limitant le recrutement de nouveaux individus (Sartoretto, 1996).

4.2.4 Impact du changement climatique sur le coralligène

Le changement climatique d'origine humaine affecte tous les écosystèmes, jusqu'aux communautés marines profondes ; en modifiant la physiologie des individus et en augmentant l'incidence de maladies, il agit sur l'abondance et la répartition des espèces et l'organisation du réseau trophique (Harley et al., 2006; Hoegh-Guldberg and Bruno, 2010; Rosenzweig et al., 2007). Cependant, le changement climatique est aussi à l'origine de grands déphasages et évènements climatiques violents (Folke et al., 2004; Carpenter and Brock, 2006; Scheffer et al., 2001) qui, selon les prédictions, sont amenés à se multiplier dans les années à venir (Easterling et al., 2000; IPCC, 2007). Ces évènements restent cependant rares en Méditerranée, ce qui explique le peu de données collectées concernant leur impact sur les communautés benthiques méditerranéennes telles que les concrétions coralligènes. Une étude, menée par Teixidó et al. (2013) sur une tempête qui a touché les eaux des îles Mèdes (Espagne) en 2008, a mis en évidence une perte sur certains sites de 22 à 58 % des espèces benthiques.

Le changement climatique induit aussi une augmentation de la température des océans. Un épisode climatique très chaud a été observé en été 1999 sur une période prolongée. Suite à cet épisode, une vague de mortalité importante a été relevée chez plusieurs communautés coralligènes, soumises à une anomalie de température positive de plus d'un mois (Ben Mustapha and El Abed, 2001; Perez et al., 2000; Romano et al., 2000).

Seules formations calcaires d'origine biogénique de Méditerranée (Ballesteros, 2006), les récifs coralligènes sont extrêmement vulnérables à l'acidification des océans (Martin et al., 2013) ainsi qu'à la fragmentation provoquée par l'Homme (Piazzi et al., 2012, 2011).

4.2.5 Surveillance du coralligène

En dépit de leur valeur économique et écologique, les récifs coralligènes restent peu connus (Ballesteros, 2006; Casellato and Stefanon, 2008). En effet, leur localisation profonde plus difficile d’accès, mais aussi leur diversité et leur dynamique lente rendent cet écosystème difficile à étudier (Kipson et al., 2011).

Les études concernant le coralligène sont souvent limitées en termes d’échelle spatiale et utilisent des méthodologies différentes, ce qui ne permet pas d’effectuer des démarches de comparaisons quantitatives (Balata et al., 2005; Ferdeghini et al., 2000; Garrabou et al., 2002; Piazzi et al., 2004; Virgilio et al., 2006). En termes de surface réelle et de distribution spatiale, les récifs coralligènes sont mal connus (Virgilio et al., 2006) et seulement à de très petites échelles, sans doute en raison des études scientifiques antérieures principalement portées sur la description des espèces et faciès au niveau local (Ballesteros, 2006). Par conséquent son importance relative surfacique par rapport à d'autres habitats est inconnue pour le moment.

Quelques indices ont été récemment proposés pour suivre et évaluer l’état de santé des communautés coralligènes. Le COARSE (COralligenous Assessment by ReefScape Estimation) (Gatti et al., 2015) est un indice permettant l’évaluation de l’état du coralligène grâce à des observations et des mesures en plongée. Il tient compte de la couverture biotique, de la richesse des espèces visibles et de la structure tridimensionnelle du récif pour évaluer l’état du récif coralligène en tant qu’indicateur de l’intégrité du fond marin (Gatti et al., 2015). Cependant, comme il est fondé sur une méthode visuelle directe, l'indice ne peut être calculé au-delà de -50 mètres pour des raisons de sécurité des plongeurs, est difficilement mesurable sur un grand nombre de sites en raison du temps limité sous l’eau et peut être influencé par la variabilité entre les observateurs affectant à la fois les estimations visuelles du recouvrement (Meese and Tomich, 1992) et l'identification correcte des espèces (Thompson and Mapstone, 1997).

D’autres indices basés sur des analyses photographiques s’affranchissent de ces contraintes en ajoutant les avantages de l’archivage : l’ESCA (Ecological Status of Coralligenous Assemblages) (Cecchi et al., 2014) ou le CAI (Coralligenous Assemblage Index) (Deter et al., 2012). Ces méthodes non destructives présentent toutefois l’inconvénient d’une identification des espèces limitée au visuel (Deter et al., 2012). Les méthodes d’échantillonnage photographique (Deter et al., 2012; Kipson et al., 2011) nécessitent des temps d’interprétation par des experts très longs. Une alternative à ces indices pourrait être obtenue par un traitement d’image par analyse spectrale. Initialement développée pour mesurer la complexité physique de l'environnement (Proulx and Parrott, 2008), une telle méthode a récemment été utilisée sur les récifs coralliens, pour analyser la biodiversité benthique (Lambert et al., 2013; Mellin et al., 2012; Tanner et al., 2015) en établissant un indice d’hétérogénéité, lié à la biodiversité.

5 Spécificités de la cartographie des habitats en milieu marin

5.1 Intérêts de la cartographie des habitats marins benthiques

Depuis des décennies, la cartographie des fonds marins est un outil essentiel pour la connaissance et la gestion des écosystèmes (Bianchi et al., 2012, 2004; Parravicini et al., 2012). A une large échelle spatiale, la connaissance des fonds marins permet de surveiller des changements environnementaux (Kostylev et al., 2001) et de comprendre des phénomènes globaux comme les pressions anthropiques sur les écosystèmes marins (Anderson et al., 2008; Bers et al., 2013; Bianchi et al., 2012; Catherine et al., 2013; Halpern et al., 2008; Kostylev et al., 2001; Micheli et al., 2013; Parravicini et al., 2012).

La cartographie des habitats (et des espèces) à grande échelle apporte une contribution à la caractérisation de l'environnement (répartition et étendue de la biodiversité marine) et constitue donc un outil essentiel pour la gestion des espaces littoraux et la mise en place de programmes de conservation de la biodiversité (Barberá et al., 2012; Bianchi et al., 2012; Godet et al., 2009; Halpern et al., 2008; Mumby and Harborne, 1999; Pasqualini et al., 1997; Reshitnyk et al., 2014; Vela et al., 2008). Elle représente un support pour la planification des réseaux d'aires protégées (Barberá et al., 2012; C. N. Bianchi et al., 2012; Godet et al., 2009; Mumby and Harborne, 1999), la modélisation cartographique de différentes alternatives de gestion (Parravicini et al., 2012), ou facilite la conciliation entre conservation de l'écosystème et développement social et économique dans le cadre d'une gestion écosystémique (Bianchi et al., 2012; Parravicini et al., 2012).

5.2 Méthodes de réalisation des cartographies d'habitats marins benthiques

En Europe, la cartographie des habitats benthiques a été étudiée dans les eaux danoises dès les années 1900 (Brown et al., 2001; Godet et al., 2009; Petersen, 1915, 1913, 1918; Thorson, 1952) puis par différents auteurs en Manche orientale et sur les côtes françaises (Cabioch, 1968; Holme, 1950). En Méditerranée, il existe une importante tradition de la cartographie marine principalement en raison de l'œuvre de scientifiques français tels que Molinier et Picard (1952) (Bianchi et al., 2004).

Les techniques utilisées dans les premières études de cartographie étaient essentiellement des méthodes de dragage et de sondage (Brown et al., 2001; Cabioch, 1968). Ils informaient sur la structure et la répartition spatiale des assemblages benthiques (Godet et al., 2009). Celles-ci restaient ponctuelles et éloignées géographiquement (Brown et al., 2001). Récemment, la production de cartes continues des habitats marins a été rendue possible avec le développement de techniques acoustiques abordables et de bonne résolution, et dont l'utilisation est croissante depuis les années 1990 (Brown et al., 2001). Les techniques acoustiques sont appropriées pour cartographier les habitats marins, là où l'identification visuelle directe est limitée par la transparence de l'eau. Le couplage de ces techniques acoustiques à des techniques optiques, adaptées pour les faibles profondeurs, permet la réalisation de cartes des habitats benthiques présentant une couverture spatiale de 100 % (Marbà et al., 2006; Sánchez-Carnero et al., 2012a).

Les techniques pour cartographier une zone peu profonde, généralement inférieure à 20 mètres (Andréfouët, 2008; Pasqualini et al., 1997), intègrent les technologies d'imageries comme la photographie aérienne (Pasqualini et al., 1998). L'utilisation de photographies aériennes est une méthode ancienne d'interprétation visuelle des habitats marins (Godet et al., 2009). Leur résolution, leur couverture et disponibilité, ainsi que leur fréquence de prises de vue font que les photographies aériennes sont des données indispensables pour cartographier les habitats marins dans les petits fonds (Ekebom and Erkkilä, 2003; Maccarrone, 2010). La photographie aérienne a l'avantage d'offrir un degré de résolution (taille du pixel) qui peut être adapté par l'utilisateur en fonction des objectifs spécifiques et des résultats finaux des travaux (Bianchi et al., 2004; Pasqualini et al., 1997). Leur utilisation permet de suivre l'évolution de la distribution des habitats sur de longues périodes, à différentes échelles spatiales, en particulier pour les herbiers de phanérogames marines. Citons par exemple les études sur l'évolution de la distribution de l'algue *Zostera marina* depuis 1924 sur les côtes atlantiques françaises (Godet et al., 2008) ou depuis 1987 au Cap Feto (Italie, île de Sicile) (Maccarrone, 2010).

5.2.1 Outils d'extrapolation pour mesurer bathymétrie et texture des fonds

Concernant la cartographie des habitats dans la zone subtidale, de nombreux outils acoustiques sont disponibles et utilisés dans le domaine marin depuis quelques décennies (Anderson et al., 2008; Brown et al., 2011). Ces méthodes produisent des « images » des fonds marins par la transmission d'ultrasons dans l'eau. Les principaux outils acoustiques sont les échosondeurs mono ou multifaisceaux (SMF) (Brown and Blondel, 2009; Kostylev et al., 2001), les sonars à balayage latéral (SBL) (Montefalcone et al., 2013; Pasqualini et al., 2000, 1998; Rovere et al., 2010; Sánchez-Carnero et al., 2012b; Van Overmeeren et al., 2009), les systèmes acoustiques de classification automatique des fonds (SACLAF) comme RoxAnn (Bers et al., 2013; Brown et al., 2005; Greenstreet et al., 1997) ou QTC-View (Freitas et al., 2008; Reshitnyk et al., 2014).

Chaque méthode acoustique est caractérisée par son champ d'action (quelques centaines de m² jusqu'à plusieurs milliers de m²), sa résolution, et son coût d'utilisation. Seule la combinaison de plusieurs systèmes permet d'accéder à un niveau de confiance suffisant pour définir, au final, les délimitations des peuplements benthiques et obtenir une cartographie pertinente des habitats. Les moyens de détection acoustique sous-marine restituent en premier lieu les propriétés acoustiques du substrat et fournissent un préalable à la cartographie des habitats. L'utilisation de sondeurs multifaisceaux (SMF) et de sonar à balayage latéral (SBL) permet d'accéder à une connaissance exhaustive de la bathymétrie et de la réflectivité du fond contrairement aux systèmes acoustiques de classification automatique des fonds. Les SMF fournissent une bonne bathymétrie et également des informations de texture qui sont devenus incontournables pour caractériser les fonds marins (Lamarche et al., 2011; Sánchez-Carnero et al., 2012a).

Le développement combiné de ces techniques et des outils informatiques tels que les Systèmes d'Information Géographique (SIG) facilite la production de cartes à de larges échelles (kilomètres) (Bers et al., 2013), détaillées et géo-référencées (Telesca et al., 2015). Combinée à des moyens d'observation directe, l'utilisation de ces outils optiques et acoustiques permet une connaissance précise et sur de grandes étendues de la topographie sous-marine et de la nature des fonds (Freitas et al., 2008; Godet et al., 2009; Kostylev et al., 2001).

5.2.2 Vérités-terrain

Les combinaisons des systèmes d'extrapolation peuvent être multiples mais demandent toutes, pour valider les données, le couplage à un système d'observation (= vérité-terrain) (Bianchi et al., 2004). Ces observations doivent utiliser des techniques d'échantillonnage adaptées à la nature du fond (Bianchi et al., 2004). Les méthodes d'échantillonnage indirect (prélèvements à la benne, à la drague, voire par chalutage) sont plus adaptées aux fonds meubles, alors que pour les fonds durs les vérités-terrain peuvent être réalisées directement dans le milieu (Bianchi et al., 2004). A grande échelle, l'utilisation d'une caméra tractée (Ardizzone et al., 2006) ou d'un ROV (Remotely Operated Vehicle), petit sous-marin téléguidé depuis la surface, permet d'obtenir une multitude d'images transmises en direct. Le système de guidage du ROV est très précis et rend l'exploration des zones profondes possible, mais la méthode est lente et coûteuse (Bianchi et al., 2004). L'utilisation de la plongée sous-marine est préférable pour les fonds durs car les observations sont de qualité supérieure. Cependant, le caractère ponctuel de celle-ci (faible rayon d'action d'un plongeur autonome) peut être un facteur limitant. Quelle que soit la technique d'observation directe utilisée (vidéo et plongeur) se pose également le problème du géo-référencement des données sous-marines. Le transect plongeur audio est une méthode de vérités terrain qui permet d'améliorer la fiabilité et la qualité des cartographies biocénotiques. Le plongeur est équipé d'un matériel de plongée classique hormis un masque facial pourvu d'écouteurs et de micros. Le plongeur est tracté par le bateau et transmet ses informations en temps réel à l'opérateur qui suit son parcours sur l'ordinateur relié au GPS. Les informations transmises par le plongeur sont positionnées avec une précision métrique grâce à l'utilisation d'un module USBL (du type Micronav de Tritech) porté par le plongeur. Cette méthode de « transect plongeur audio » permet la validation terrain de plusieurs milliers de points par un plongeur biologiste le long de transects pouvant atteindre une dizaine de kilomètres par jour d'acquisition (Descamp et al., 2010).

5.3 Echelles et cartes existantes

Le choix de la méthode et des outils de cartographie est dicté par le choix de l'échelle qui affecte grandement la représentation finale des cartes ; le détail des informations collectées change en fonction de l'échelle de représentation. Les échelles doivent être choisies en fonction des unités environnementales à cartographier (Bianchi et al., 2004).

Les grandes échelles (> 1 : 50 000) sont utilisées pour les représentations synoptiques des principales biocénoses d'un vaste territoire. Par exemple, Bianchi et al. (1996) ont réalisé une carte bionomique pour la mer de Toscane avec une échelle de 1 : 250 000 (Bianchi et al., 2004). Ces échelles ne permettent pas de montrer certains assemblages tels que ceux vivants sur fonds durs et, plus généralement, dans les habitats strictement côtiers.

Les échelles intermédiaires sont préférées pour les grandes zones qui gardent une certaine unité physiographique (Bianchi et al., 2004). Ces cartes nécessitent une simplification de symboles et un regroupement des biocénoses. Navone et al. (1992) ont utilisé une échelle de 1 : 25 000 pour cartographier l'ensemble de l'Aire Marine Protégée de Tavolara – Punta Coda Cavallo (nord-est de la Sardaigne) et qui a servi à la réalisation d'un ensemble de cartes pour la gestion de l'AMP (Rovere et al., 2013).

Les petites échelles sont adéquates pour cartographier des zones de manière détaillée dans lesquelles les fonds rocheux sont dominants. Ces ambitieuses échelles nécessitent un degré élevé de précision qui implique un échantillonnage assidu et fréquent et une représentation schématique assez compliquée au vu du nombre élevé de symboles. Gili & Ros (1985) par exemple ont choisi une échelle de 1 : 2 000 pour cartographier les communautés benthiques dans une AMP de 2 km² au large de la côte catalane : les îles Medes. Sardá et al. (2012) ont réalisé une cartographie des habitats benthiques du Cap de Creus (Espagne, nord-ouest de la mer Méditerranée) au 1 : 5 000. Piazzi et al. (2000) ont réalisé des cartes à une échelle de 1 : 10 000 sur l'île d'Elbe en Méditerranée occidentale.

La cartographie des fonds marins représente un enjeu pour la connaissance et la gestion des écosystèmes marins quelle que soit l'échelle spatiale considérée (compréhension de phénomènes globaux à large échelle, gestion d'un territoire, d'une aire marine protégée, ou d'un habitat spécifique à plus fine échelle). Afin d'améliorer la connaissance des liens entre pressions humaines et statut des écosystèmes côtiers des modèles spatiaux ont été développés (Parravicini et al., 2012b). Ces modèles sont basés sur la cartographie des fonds marins, et permettent une approche prédictive du statut écologique résultant de différentes alternatives de gestion. L'information sur la répartition des pressions anthropiques par l'intermédiaire de substituts (par exemple présence / absence d'activités humaines pertinentes ou la distance pondérée de ces activités) a été utilisée avec succès pour la cartographie des risques potentiels de l'impact humain (Eastwood et al., 2007; Petrosillo et al., 2010; Stelzenmüller et al., 2010).

La connaissance des points de référence historiques (l'état de conservation des écosystèmes marins antérieur aux impacts humains à grande échelle), et l'observation des conséquences des pressions passées sur leur état actuel reste la meilleure approche pour réduire les impacts humains et s'engager dans la voie du développement durable, mais ces connaissances nous font défaut (Micheli et al., 2013; Pauly, 1995; Underwood, 1992).

Objectifs

La vulnérabilité, considérée comme la probabilité d'exposition à un stress et la sensibilité comme le degré de réponse à ce stress, sont des composantes essentielles à la compréhension et donc à la conservation des écosystèmes. Les objectifs de ce projet doctoral sont de caractériser cette vulnérabilité et d'aborder la sensibilité de deux écosystèmes marins remarquables car d'importance écologique majeure : les herbiers de posidonie et les assemblages des communautés à coralligène.

Les objectifs visés sont donc en adéquation avec ceux de la DCSMM (Ministère de l'Ecologie du Développement Durable et de l'Energie 2012) concernant ces deux écosystèmes :

« *A. Maintenir ou rétablir la biodiversité et le fonctionnement des écosystèmes des fonds côtiers (...), A2. Renforcer la conservation des zones de coralligène et des zones d'herbiers [...] A4. Maîtriser la pression des usages maritimes sur le milieu en développant l'organisation spatiale des usages [...] A6. Identifier les sites présentant des habitats naturels dégradés et engager la restauration de la moitié de ces sites* »

La mise en application de ces mesures nécessite des évaluations préalables. Celles-ci font suite à la dynamique de la surveillance biologique mise en place sur le littoral méditerranéen et nécessitent (1) de cartographier et quantifier les surfaces couvertes actuellement par ces écosystèmes sensibles, (2) quantifier et cartographier leur diversité ainsi que leur état écologique, (3) cartographier et quantifier les pressions côtières agissant sur ces écosystèmes, (4) définir des états de référence des écosystèmes côtiers, et (5) mettre en avant les liens entre état des habitats et pressions anthropiques côtières.

Dans le cadre de ce projet doctoral, ces points constituent des réflexions auxquelles j'ai souhaité apporter ma contribution.

Chapitre 1 / Analyse spatiale

Ce chapitre s'articule autour de :

- Une publication tirée des actes du 5ème Symposium Méditerranéen sur la Végétation Marine à Portoroz (Slovénie) en octobre 2014 et intitulée : « First continuous seabed map in France used for the creation of a management tool protecting *Posidonia oceanica* »
- Un Article publié dans Estuarine, Coastal and Shelf Science en mai 2015 sous le titre : « The impact of 85 years of coastal development on shallow seagrass beds (*Posidonia oceanica* L. (Delile)) in South Eastern France : A slow but steady loss without recovery »
- Un article publié dans Ecological indicators en novembre 2015 sous le titre : « Mapping biodiversity in three-dimensions challenges marine conservation strategies: the example of coralligenous assemblages in North-Western Mediterranean Sea »

1 Articulation entre les articles

1.1 Une première cartographie continue des fonds marins côtiers en Méditerranée française

Mon projet doctoral fait suite à huit années d’activité professionnelle principalement consacrée à l’étude et la cartographie des fonds marins côtiers en Méditerranée française. Aussi, un des premiers objectifs a été, avec le soutien de l’Agence de l’eau RMC, de réaliser une synthèse des différents travaux menés depuis 30 ans par les différents acteurs du milieu marin (université, bureau d’études, établissement public). Je me suis ensuite attaché à compléter les zones inconnues avec l’appui du bureau d’études Andromède Océanologie. L’aboutissement de ce travail est la première cartographie continue des habitats marins en Méditerranée française. Les habitats marins y sont décrits jusqu’à la profondeur de -80 mètres avec une échelle au 1:10 000ème (1 mm sur la carte représente 10 m dans la réalité). La donnée est géoréférencée, intégrable à tout Système d’Information Géographique (format « shapefile ») et disponible sur la plateforme cartographique MEDTRIX (www.medtrix.fr) dans le projet DONIA Expert.

Figure 16 *:* Poster présenté lors de la « Journée mer sur la surveillance du milieu marin » en juillet 2014 organisée par l’Agence de l’eau RMC à Marseille.

Si ce travail n’a pas été à ce jour publié dans une revue scientifique, il a fait l’objet d’un rapport (Holon et al., 2014), d’un poster présenté à la « Journée mer sur la surveillance du milieu marin » à Marseille en juillet 2014 (Figure 16) et d’une communication lors du 5ème Symposium Méditerranéen sur la Végétation Marine à Portoroz (Slovénie) en octobre 2014. Un court article fait ainsi partie des actes de ce symposium (voir Article 1 dans ce chapitre) sous le titre « *First continuous seabed map in France used for the creation of a management tool protecting Posidonia oceanica* ». Dans cet article, je décris succinctement la méthodologie de réalisation ainsi que les liens avec l'application mobile DONIA pour aider les plaisanciers à ancrer en dehors des herbiers de posidonie. La cartographie combinée à cette application ouvre de nouvelles perspectives pour la gestion de la zone côtière.

Cette cartographie est l'aboutissement de plusieurs années de travail, mobilisant de nombreux acteurs du littoral. Elle pose aussi toutes les bases nécessaires à la suite de mon projet doctoral.

1.2 Dynamique spatio- temporelle des herbiers à posidonie

En 2010, le CRIGE PACA (Centre Régional de l'Information Géographique de Provence-Alpes-Côte d'Azur) a mis en ligne une mosaïque de photographies aériennes de la région datant de 1922 à 1944. La mer y est d'une clarté étonnante, on y voit par transparence les herbiers avant le développement urbain et touristique que connaîtra cette région. C'était surtout une formidable opportunité de retour dans le temps et une possibilité d'étudier l'impact des pressions anthropiques côtières sur l'environnement marin.

Ce fut également un travail cartographique chronophage, auquel a pris part l'ensemble des co-auteurs de l'article publié dans Estuarine, Coastal and Shelf Science en mai 2015 sous le titre « The impact of 85 years of coastal development on shallow seagrass beds (*Posidonia oceanica* L. (Delile)) in South Eastern France : A slow but steady loss without recovery ».

La méthode consiste (1) en des analyses comparatives des photographies aériennes (anciennes, intermédiaires et actuelles) pour suivre les changements de répartition de l'herbier à *Posidonia oceanica* en limite supérieure (0-15 mètres de profondeur) et (2) à la superposition de ces cartographies avec les pressions anthropiques côtières visibles sur ces photographies (artificialisation du littoral, aquaculture, conduits sous-marins, activités militaires, mouillage).

Cette étude conclut sur la faible progression des herbiers (les trous des bombes de la seconde guerre mondiale y sont toujours visibles) et permet d'apporter des chiffres intéressants concernant le déclin de l'herbier en fonction des différents types d'aménagements.

1.3 Suivi des assemblages des communautés à coralligène (méthode RECOR)

Malgré sa forte valeur écologique et économique, les suivis spatio-temporels du coralligène sont rares. Porté par l'Agence de l'eau RMC et Andromède Océanologie, le réseau RECOR (Réseau Coralligène) s'étend sur l'ensemble de la façade méditerranéenne française bordée par les trois régions Corse, Provence-Alpes-Côte d'Azur (PACA) et Languedoc-Roussillon (LR). Initié en 2010, les objectifs de RECOR sont de recueillir des données descriptives de l'état et du fonctionnement de l'habitat Coralligène et de suivre leurs changements dans le temps et dans l'espace. L'ensemble de ces données complète l'estimation de la qualité écologique des masses d'eau côtières demandée par la Directive Cadre Eau (DCE), chaque masse d'eau comprenant au minimum un site RECOR.

Nous avons publié la méthodologie ainsi que les premiers résultats de ce réseau dans deux articles scientifiques antérieurs à mon travail de thèse (Deter et al., 2012a; Deter et al., 2012b). Ces deux articles sont donc reportés dans les annexes de ce manuscrit (annexes 1 et 2).

RECOR comprend actuellement 169 stations réparties entre 17 et 90 mètres de profondeur (Figure 17). Chaque année, une région est suivie à la fin du printemps (mai-juin) afin de couvrir l'ensemble de la façade méditerranéenne tous les trois ans.

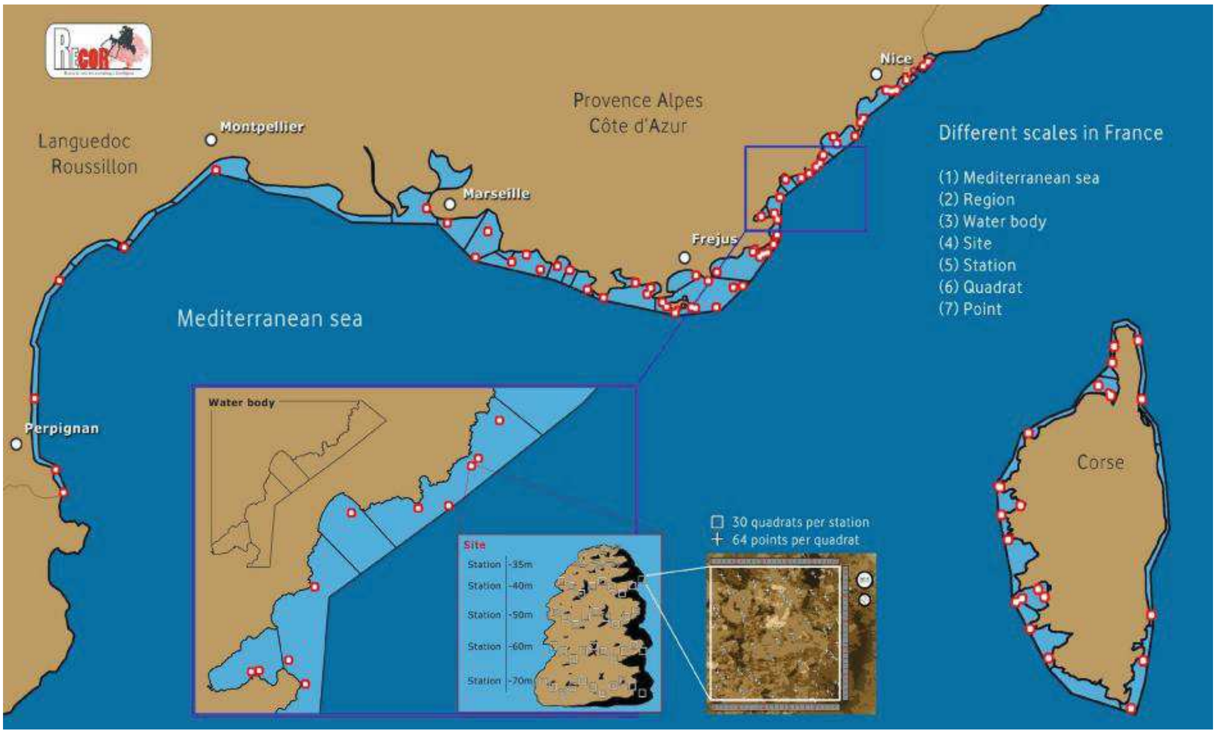


Figure 17 : Localisation des différents sites de suivi Coralligène dans le cadre du réseau RECOR.

La méthode RECOR consiste essentiellement en la réalisation de séries de quadrats photographiques (Figure 18) de 50 x 50 cm à une même profondeur (= une station) le long d'un transect de 20 mètres.

Figure 18 : Plongeur en recycleur circuit fermé réalisant des quadrats photographiques sur le coralligène.

Parmi ces photographies, 30 sont tirées aléatoirement puis analysées avec le logiciel CPCe 4.1 « coralligenous assemblage version ». Le logiciel répartit 64 points aléatoirement sur chacun des 30 quadrats photographiques, représentant ainsi 1920 points par station. L'identification de la nature des espèces ou du substrat sur lesquels sont disposés ces points est réalisée par un même observateur. Une centaine de variables quantitatives sont extraites de ces analyses tels que :

- Les pourcentages de recouvrement par le non vivant (vase, anfractuosités, débris) et le vivant (les actiniaires, les alcyonaires, les ascidies, les astérides, les algues brunes, les algues vertes, les algues rouges, les crinoïdes, les échinodermes, les bryozoaires érigés, les bryozoaires encroûtants, les grands foraminifères, les gorgonaires, les hydraires, les scléractiniaires, les vers sédentaires, les zoanthaires).

- Les pourcentages de recouvrement par des espèces particulières comme *Mesophyllum* sp., *Lithophyllum* sp. et *Peyssonnelia* sp. (bio-construction), le corail rouge *Corallium rubrum* (protection et intérêt commercial), les axinelles *Axinella* sp. (protection *d'Axinella polypoides* et fragilité due au port érigé), les algues *Caulerpa taxifolia* ou C. *racemosa* (potentiel invasif), les éponges *Cliona* sp. et l'oursin melon *Echinus melo* (bio-érosion), l'ascidie *Halocynthia papillosa* (sensible à la pression « plongée sous-marine »), les algues filamenteuses, les vers *Filograna* sp / *Salmacina* sp (potentiellement sensibles à la dégradation du milieu, Ballesteros, comm. pers.).

- Les regroupements d'espèces tels que les pourcentages de recouvrement par les principales espèces bioconstructrices (les algues *Mesophyllum* sp., *Litophyllum* sp. et *Peyssonnelia* sp., les coraux jaunes solitaires *Leptopsammia pruvoti*, les madrépores *Caryophyllia inornata,* les foraminifères *Miniacina miniacea*), les espèces bioérodeuses (les cliones *Cliona* sp., l'oursin melon *Echinus melo*, l'oursin violet *Sphaerechinus granularis*), les espèces dressées (les gorgones comme *Paramuricea clavata, Eunicella cavolinii, Eunicella singularis*, le corail rouge *Corallium rubrum*, les grandes espèces de bryozoaires et les éponges *Axinella* sp.), les espèces d'algues invasives (*Caulerpa* sp., *Womersleyella setacea*), les espèces d'intérêt patrimonial (*Spongia officinalis, Axinella polypoides, Savalia savaglia, Astroides calycularis, Paracentrotus lividus, Centrostephanus longispinus, Ophidiaster ophidianus, etc.*).

1.4 Etude multifacette de la diversité des assemblages à coralligène

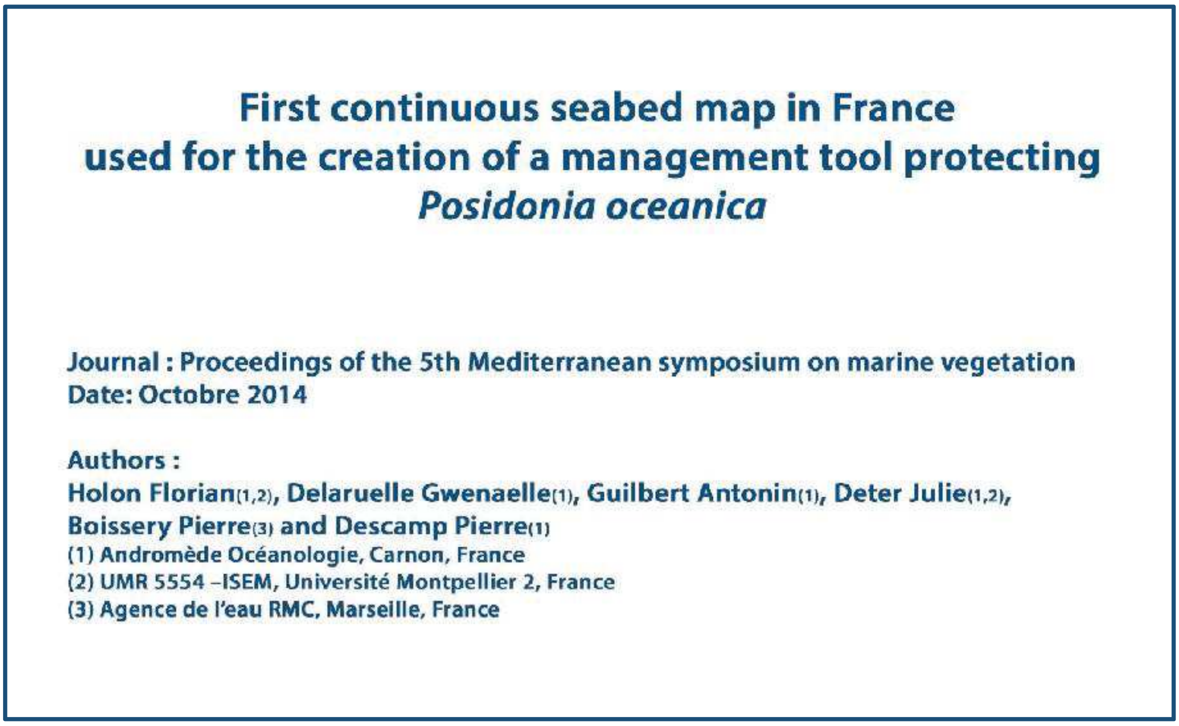
Si la mise en lumière des relations entre pressions anthropiques et assemblages d'espèces est essentielle à la mise au point d'indicateurs de la qualité écologique des masses d'eaux, tels que demandés par la DCE, la prise en compte des autres dimensions de ces assemblages n'en est pas moins nécessaire sous peine de mettre en avant de fausses relations. Ainsi, à l'échelle biogéographique, la distribution des espèces résulte des interactions biotiques, de la limitation par la dispersion, de la distribution des ressources dans le temps et l'espace, ainsi que de l'histoire évolutive des régions écologiques (Keddy, 1992). Depuis quelques années, les écologistes intègrent de nouvelles dimensions à la diversité telles que les relations phylogénétiques entre espèces (Webb et al., 2002) et la mesure de leurs traits fonctionnels (Mason et al., 2005; McGill et al., 2006) qui semblent essentielles à la compréhension de la diversité à différentes échelles (Harrison and Cornell, 2007; Mittelbach et al., 2007).

Si ces analyses multifacettes de la diversité sont de plus en plus utilisées dans l'écologie de la conservation (Bellwood et al., 2004; Devictor et al., 2010; Hooper et al., 2005; Mouillot et al., 2011), l'analyse des distributions horizontales et verticales dans un écosystème structuré en trois dimensions n'avait pas encore été abordée. Formant des structures complexes à la fois verticales et horizontales, comparables aux récifs tropicaux en termes de richesse, de biomasse et de production, les assemblages coralligènes nous ont semblé un bon modèle d'analyse.

Ainsi sur la base des espèces identifiées dans le cadre du RECOR, nous avons entrepris, dans un premier temps, la réalisation d'une base des traits fonctionnels afin de pouvoir comparer diversité taxonomique, phylogénétique et fonctionnelle des différentes stations (voir ci-après). Les valeurs des différents traits proviennent de la littérature et du savoir d'experts.

L'article scientifique, porté par Aggeliki Doxa (post doctorante sur ce projet d'étude), intitulé : « Mapping biodiversity in three-dimensions challenges marine conservation strategies: the example of coralligenous assemblages in North-Western Mediterranean Sea » (en révision pour Ecological indicators) révèle que les valeurs de diversité taxonomique diffèrent de celles de diversité fonctionnelle et phylogénétique. La diversité taxonomique seule est donc insuffisante pour fixer des objectifs de conservation des communautés à coralligène notamment afin de préserver son fonctionnement et son histoire évolutive. Nos résultats mettent en évidence la nécessité d'aborder la question des aires marines protégées sous un autre angle qu'une unique délimitation restreinte de cap à cap mais également de considérer différentes ceintures de profondeur à grande échelle dans les efforts régionaux de conservation.

2 Article 1 : Première cartographie continue des habitats marins en France, application à la création d'un outil de management pour la protection de *Posidonia oceanica*



First continuous seabed map in France used for the creation of a management tool protecting *Posidonia oceanica*

Journal : Proceedings of the 5th Mediterranean symposium on marine vegetation
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Authors :
Holon Florian(1,2), Delaruelle Gwenaelle(1), Guilbert Antonin(1), Deter Julie(1,2), Boissery Pierre(3) and Descamp Pierre(1)
(1) Andromède Océanologie, Carnon, France
(2) UMR 5554 –ISEM, Université Montpellier 2, France
(3) Agence de l'eau RMC, Marseille, France

2.1 Abstract

The Mediterranean Sea is a biodiversity hotspot facing numerous threats: high population density, high urbanization rate, coastal erosion, overexploitation of marine resources and spread of invasive species. Maps of marine habitats are an essential tool in order to appreciate the ecological and spatio-temporal heterogeneity of the environment, potential and real distribution of species, identify corridors but also propose relevant management measures and evaluate their effects. Different studies allowed the realization of high scale marine habitat maps. It resulted in a patchwork of more or less connected maps. Through the DONIA program we reviewed and completed 30 years of work in order to obtain a continuous map of the French coastline: 1 700 km and 47 water bodies.

We obtained a 1:10 000 continuous map of the seabed habitats using ten classes. The results are freely available (with login) on www.medtrix.fr in DONIA®expert. Besides the complete French Mediterranean coast, several Mediterranean islands are concerned: the Galite Archipelago in Tunisia, Zembra Island in Tunisia and Tavolara - Punta Coda Cavallo in Sardinia (Italy). A simplified database called DONIA® intended for the general public is also freely available on Medtrix without any login. The same regions are concerned but habitats were simplified into four classes (seagrass, dead matte, sand, rock) instead of ten. This simple dynamic map is freely accessible to all through the DONIA® application (App Store and Google play); it helps boats to safely anchor outside of sensitive habitats. The application opens new perspectives in terms of management of marine protected areas by facilitating the communication between site managers and visitors and providing an active management of mooring.

Key-words: anchoring management; marine habitat; cartography; seagrass conservation

2.2 Introduction

The Mediterranean Sea is a biodiversity hotspot facing numerous threats: a very high population density, high urbanization rate, coastal erosion, over-exploitation of marine resources and spread of invasive species (PNUE/PAM-Plan Bleu, 2009; Chevaldonné et al., 2010). A key-ecosystem particularly mistreated despite its protected status is the *Posidonia oceanica* seagrass meadow (Boudouresque et al., 2009). Indeed, its ecological preferences (shallow and clear waters, low current) and biological characteristics (very slow growth) makes this species very vulnerable to anthropogenic pressures (Boudouresque et al., 2012). For example, every day, by anchoring, thousands boat users may irreversibly destroy small pieces of meadow because they do not know this protected species and are not aware of its spatial distribution.

Maps of marine habitats are an essential tool in order to appreciate the ecological and spatio-temporal heterogeneity of the environment, potential and real distribution of species, identify corridors but also propose relevant management measures and evaluate their effects (Boström et al., 2011). Several high scale maps of marine habitat are produced each year. The outcome is a patchwork of more or less connected maps. Through the DONIA program, we reviewed and completed 30 years of work in order to obtain a continuous map of the entire French Mediterranean coast and of 47 water bodies. We then used this map for the development of a free mobile application helping boat owners to avoid anchoring within seagrass and thus preserve *Posidonia oceanica* beds.

2.3 Materials and Methods

After a bibliographic synthesis, we pooled and homogenized the habitat maps built in the framework of numerous programs: DONIA (Andromède Océanologie), CARTHAM - AAMP (G.I.S. Posidonie, Sintinelle, Stareso, Evemar, Andromède Océanologie), PIM (Conservatoire du littoral, Agence de l'eau RMC), MEDBENTH (IFREMER), EGIS EAU, Aire marine protégée de la côte agathoise (site Natura 2000), DREAL PACA, Ville de St Cyr-sur-Mer and Ville de Saint Raphaël. Campaigns were led between 2010 and 2014 using aerial photography (in order to see the spatial extent of the habitats in shallow waters), a multi-beam echo-sounder survey (to draw the bathymetry), a side-scan sonar survey (used in more turbid and deeper waters), sonar information post-treatment (in order to determine potential presence and covering of underwater habitat representation), production of a preliminary cartography which is then validated by 'ground truth' diving sessions (field work). The methodology is available at Andromède Océanologie (2014). A final continuous habitat map was realized considering ten habitat classes. This map was secondarily simplified (four classes) in order to be understood by the general public. A Smartphone / Tablet Application was then developed in partnership with the French company 3Liz in order to help yachtsmen and yacht captains to check the nature of the seabed and find a suitable location for anchoring.

2.4 Results

We produced a 1:10000 continuous map for seabed habitats using ten classes: Cymodocea nodosa seagrass, *Posidonia oceanica* seagrass, dead matte association, infralittoral shingle association, infralittoral soft bottoms, infralittoral algae, coralligenous habitat, offshore rocks, circalittoral soft bottoms, artificial habitats. The results are freely available (with login) on www.medtrix.fr in DONIA®expert. Besides the complete French Mediterranean coast (Languedoc-Roussillon, Provence and French riviera, Corsica being completed), several Mediterranean islands were concerned: the Galite Archipelago in Tunisia, Zembra Island in Tunisia and Tavolara - Punta Coda Cavallo in Sardinia (Italy). A simplified database (four classes = seagrass, dead matte, sand, rock) called DONIA® intended for the general public is also available on the so-called www.donia.fr website and on www.medtrix.fr without any necessary login. This simplified dynamic map is freely accessible to all through the DONIA® application (App Store and Google play), helping boats to anchor safely outside of sensitive habitats. Once the chosen maps are downloaded on a smartphone or a tablet, DONIA® is able to position the boat and show the nature of the underwater seabed. The boat appears colored in green or red depending if anchoring should be preferred or avoided (Fig. 1). The application also provides a prepaid panel of anti-skid and anti-collision anchoring alarms which increase safety onboard, even detecting the crossing of chains and preventing all risk of entanglement. DONIA® also offers users the ability to acquire high precision bathymetry 3D maps to accurately visualize the underwater topography of several sites and prepare for scuba dives for example. Officially launched at the 2013 Paris Boat Show, DONIA® had more than 1 000 users (and 8 113 views) in July 2014.

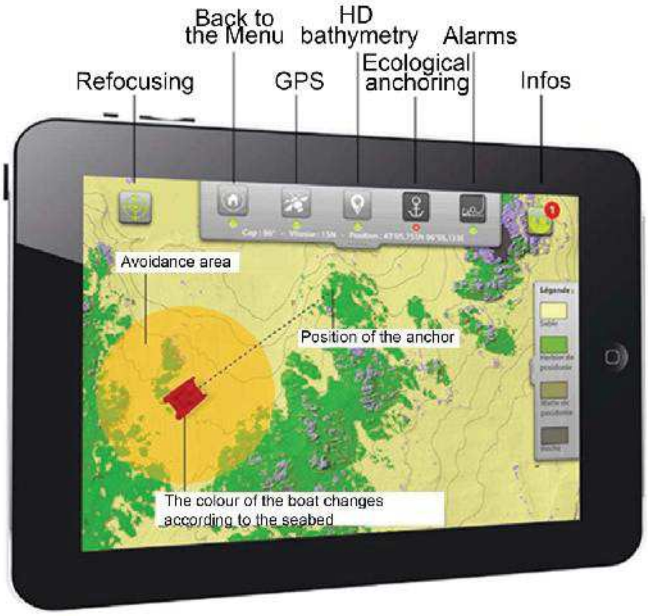

Figure 1: Main window of DONIA® application (currently only available in French). The legend displays color correspondence for the habitats: yellow = sand, green = *Posidonia oceanica* meadow, brown = dead matte and violet = rocky habitat.

2.5 Discussion

We present here a very important step forward in marine ecology: the first continuous map of marine habitats along the French Mediterranean coasts. A fine knowledge of the presence, distribution and conservation status of the habitats of each area, and inparticular of *Posidonia oceanica* meadows, is the first essential step for an appropriate and effective long-term managing of marine resources. Moreover, these data are now available to the general public and more importantly to sea users. Free and easy to use, DONIA® is a resounding success. It received two awards in 2013: “Companies and biodiversity” from the French Ministry of Environment and “Bateau Bleu” from the French nautical industries.

A management tool named “Donia Manager Module” is under progress. The objective is for managers of marine protected areas to control and limit the impacts of tourism in their area, especially of leisure boating. This interactive interface will allow a real-time monitoring of the boat flow through the sending and receiving of information between managers and users, such as on anchoring, possible marine pollution, or on the presence of protected species.

2.6 Conclusion

The application opens new perspectives in terms of management of marine protected areas by facilitating the communication between site managers and visitors and providing an active management of mooring. It will also raise awareness and call for sustainable practices associated with open anchoring in sites of community interest.

The objectives of this tool are to decrease the impact of anchoring in marine protected areas by 50 %. This innovative tool is expected to widen its geographical scope and to be exported, first in Mediterranean countries such as Italy and Spain, then in all parts of the world where the impact of recreational boating on the marine environment is a major challenge for the conservation of marine biodiversity.

Acknowledgments

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3 Article 2 : Impact de 85 années de developpement côtier sur la limite supérieure de l’herbier à *Posidonia oceanica* (L. Delile) dans le Sud-Est de la France : un déclin lent mais sûr sans récupération.

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The impact of 85 years of coastal development on shallow seagrass beds (*Posidonia oceanica* L. (Delile)) in South Eastern France: A slow but steady loss without recovery

F. Holon [a], P. Boissery [b], A. Guilbert [a], E. Freschet [a], J. Deter [a, c, *]

[a] *Andromède Océanologie, 7 place Cassan, 34280, Carnon, France*
[b] *Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001, Marseille, France*
[c] *Université de Montpellier/Institut des Sciences de l'Evolution (ISEM) – UMR 5554 UM -CNRS – IRD, Campus de l'Université de Montpellier, 34095, Montpellier cedex 5, France*

3.1 Abstract

Shallow *Posidonia oceanica* beds (0 to -15 m), the most common seagrass in the Mediterranean, were mapped from aerial pictures dating from the 1920’s and from 2012 along 800 km of coastline in South-Eastern France (Provence-Alpes-Côte d’Azur region). Changes in P. *oceanica* beds spatial distribution (limits and areal extents) during these 85 years were analyzed in terms of concordance (remaining areas), positive discordance (expanding areas) or negative discordance (lost areas). Lost areas were linked with direct or indirect impacts of coastal development (artificialized coastlines (namely harbours, ports of refuge, landfills, artificial beaches, groynes and pontoons, submarine pipelines and aquatic farms) visible on the pictures. The comparison showed that 73 % of the shallow limits have declined. Considering areal extents, remaining seagrass meadows areas accounted for the major part (85 %), while lost areas accounted for 13 % and expanding areas for 1.1%. Lost areas were mainly linked with artificialized coastlines but 44 % remained with undetermined causes (invisible pressures and/or mixed effects). The analysis of 96 coastal facilities participating to the artificialized (namely man-made) coastlines showed that the highest impact over the longest distance (5 km) was caused by harbours. Only artificial beaches had such a distant impact. Pontoons were the least surrounded by lost seagrass meadows areas. These quantitative data offer important information for marine conservation.

Key words: seagrass meadows decline; consequences of urbanization; large-scale decrease of seagrass meadows; loss of marine meadows; human-driven impacts; anthropogenic pressures

3.2 Introduction

With more than seven billion people on Earth (United States Census Bureau 2014), human activities have global impacts on all oceans and seas (Jackson et al 2001; Stachowitsch 2003; Halpern et al 2008). Coastal areas and coastal ecosystems are particularly affected as they concentrate rich marine biodiversity, an important human population and a wide range of human uses (Halpern et al 2008). Population densities in coastal regions are now about three times higher than the average elsewhere, and the last seventy years with the industrial revolution and the population explosion were particularly demanding: rapid urban development, construction of new seaside resorts, marinas and extensions of existing ports (Small and Nicholls 2003). However, marine ecosystems provide important and valuable goods and benefits (i.e. contributions that humans derive or create from ecosystem services (Millennium Ecosystem Assessment MEA 2005; Haines-Young and Potschin 2013)). For example, more than half of the total value of the world natural capital and services are considered to be related to a single marine ecosystem: seagrass beds (Costanza et al 1997). In this context, marine conservation science needs to assess and understand the impacts of human beings on marine habitats in order to protect them. Approaches based on expert's opinions (Halpern et al 2007; Halpern et al 2008; Claudet and Fraschetti 2010; Parravicini et al 2012) are often used as a proxy for real impacts on habitats, but they are not as significant as quantitative assessments, and the critical lack of empirical knowledge about marine systems impedes the implementation of effective conservation measures (Claudet and Fraschetti 2010). The knowledge of historical reference points (the state of conservation of marine ecosystems prior to large-scale human impacts), and observation of the consequences of past pressures on their current state remains the best approach to reducing human impacts and moving along a sustainable development path, but we are lacking this knowledge (Underwood 1992; Pauly 1995; Micheli et al 2013).

Seagrasses are often considered as biological sentinels because any change in their distribution (e.g. a reduction in the maximum depth limit or a loss of covered areas) implies an environmental change (Orth et al 2006). *Posidonia oceanica* L. (Delile) is the most common seagrass species in Mediterranean Sea (Boudouresque et al 2012). It forms extensive meadows from the surface to 30-40 m depth (depending on water transparency and temperature). Over time, this long-lived plant builds up a set of rhizomes and roots which interstices are filled in by sediment; this structure is called "matte" (Boudouresque et al 2012). The plant can reproduce both sexually and asexually but its growth is very slow (a few centimetres per year). After the death of the plant, the deterioration of rhizomes is very slow, leading to a dead matte that may persist for millennia (Boudouresque et al 2012). Because of the important ecological (nursery, spawning, feeding, oxygenation) and economic roles (coastal protection and sediment trapping) (Borum et al 2004; Boudouresque et al 2012), P. *oceanica* is protected by EU legislation (Habitat directive), the Bern and Barcelona Conventions, national legislation and is classified Least Concern on the IUCN Red List (Pergent et al 2010).

Like numerous seagrass species (Short and Wyllie-Echeverria 1996; Spalding et al 2003; Waycott et al 2009; Selig et al 2014), P. *oceanica* meadows have known a widespread decline over the last decades

(Boudouresque et al 2009); a decline characterized by a decrease of shallow seagrass beds and/or by a reduction of the deeper limits and thus a loss of areal extent. Ten percent is the global decline (loss of area) generally accepted for P. *oceanica* over the last 100 years (Boudouresque et al 2012) but a recent paper claims a reduction by 50% of the density or biomass within the Mediterranean over the last 20 years (Marbà et al 2014a). Actually, the magnitude of the overall P. *oceanica* are a loss over the last century that ranges from 0 to50 % depending on the author(González-Correa et al 2007; Boudouresque et al 2009; Bonacorsi et al 2013) but could reach 8 % per year with possible functional extinction in 2059 according to others (Marbà et al 1996; Jordà et al 2012). The reality is difficult to assess because of a lack of reliable baseline data: quasi-absence of historical data, studies often only focusing on small spatial and temporal scales and/or using uncertain old maps (Montefalcone et al 2013; Bonacorsi et al 2013). These observed declines are mainly located near urban areas (Thomas et al 2005; Boudouresque et al 2012) and mostly associated with human activities even if they can sometimes be related to natural processes (e.g. colonization and erosion dynamics, climate change, sea level change, weather events, exceptional tectonic events or diseases) (Duarte 2002; Boudouresque et al 2009; Pergent et al 2014;Tuya et al 2014). A recent review of the literature showed that the responsibility of P. *oceanica*'s decline is attributed to human physical impacts by two thirds (67.6%) of the studies (Marbà et al 2014b). Main declines of P. *oceanica* meadows are related to coastline engineering (Ruiz and Romero 2003; Boudouresque et al 2012; Roca et al 2014), aquaculture (Pergent-Martini et al 2006; Holmer et al 2008; Rountos et al 2012), solid and liquid waste (Morena et al 2001; Pergent-Martini et al 2002; Boudouresque et al 2012), pleasure boats and cruise tourism (Montefalcone et al 2006; Okudan et al 2011; Boudouresque et al 2012) and to the introduction of exotic species (Boudouresque et al 2012; Marbà et al 2014a). However, the relative quantitative influence of each of these causes on the overall decline remains unknown.

The present work consists in estimating the changes that the shallowest part of P. *oceanica* meadows have undergone in connection with coastal human activities over a large spatial (800 km) and temporal (85 years) scale. The objectives are thus i) to make an assessment of old and present P. *oceanica* meadows (limits and areal extent) using a unique methodology, ii) to link the loss observed with human activities in order to estimate their direct and indirect impacts on the meadows and iii) to quantify the spatial scale of the impacts on adjacent seagrass meadows. Considering the available literature (see introduction above) and the plant characteristics (slow growth, long-term persistence, high sensitivity) we expect to observe a decline of a large part of the shallow limits (an average loss of 10 % of the initial area is expected) mostly located near urban areas, but also to highlight an overall stability of the meadows general areal extent and small expanded areas.

3.3 Materials and methods

Study area

This study was led along 800 km of coastline. It represents the coastline of Provence-Alpes-Côte-D'Azur (PACA), the French Mediterranean region where the highest reclamation area from the sea was observed between 1920 (1643.19 ha) and 2010 (3945.56 ha) (MEDAM 2014). The man-made (artificialized) coastline went from 45.10 km in 1920 (mainly harbours) to 156.39 km (=19.05 %, mainly harbours, landfills, artificial beaches and ports of refuge) in 2010 (MEDAM 2014). This region regroups three French departments (Bouches du Rhône, Var and Alpes Maritimes) and represents 26 coastal water bodies, namely geographical units of homogeneous waters according to the Water Framework Directive (WFD,2000/60/EC).

Pictures used

This study used several geo-referenced mosaics of historical pictures (1922, 1924, 1927 and 1944, depending on the area) made available by the "Région Provence-Alpes-Côte d'Azur © SHOM, IFREMER et Photothèque nationale (2008)" consortium. Only one picture (the oldest one) was kept per place with the following proportions: 6 % of the study area was based on pictures dating from 1922, 53 % from 1924, 34 % from 1927 and 7 % from 1944 (Fig. 1). All of these pictures were there after called "old pictures" without distinction in order to simplify the message. They were provided after undergoing geometric corrections allowing to eliminate image distortions with BD-ORTHO® ©IGN. Present aerial geo-referenced pictures were mostly (94 %) taken in 2012 (IGN, "Ortho Littorale V2 – MEDDE"). Four year older pictures (2008) were used when those taken in 2012 were not usable. Thus, according to the areas involved, this study considered a mean time frame of 85 years and a median time frame of 68 years. Pictures were exported with a 5 x 5 km grid into a CAD software at 1/20 000 with a 1000 dpi resolution. They were then processed for quality improvement: colors, contrast, sharpness and noise filtration.

Posidonia oceanica meadows charts

Aerial pictures generally permit a mapping of P. *oceanica*'s distribution up to 20 m depth (Pasqualini et al 1998). Shallow seagrass beds (0 to -15 m) of P. *oceanica* were mapped from old and present aerial pictures along the PACA coastline (Fig. 2). The present coastline geographical informations were provided by IGN and SHOM; it was modified according to the old aerial pictures in order to draw the old coastline. The deep delimitation was based on the SHOM -15 m isobath improved by fine-scale bathymetric data obtained from a multi-beam echosounder (Andromède Océanologie 2013). Seagrass meadows were interpreted from sudden changes in hue and lightness in a semi-automatic way. At a 1:5000 scale, the image was automatically segmented and the lab technician validated every single segment of seagrass patch within polygons. Additional polygons were sometimes manually delineated when they were visible but not recognized by the segmentation tool. The more or less good quality of pictures (objects, paper defaults, bad digitalization of silver shots) and of the shooting conditions (reflections, luminosity, waves, silver shot quality) makes the pictures more or less suitable for use. Three levels of certainty were thus defined in order to qualify our confidence in the interpretation of the old pictures. Level 1 of certainty qualified seagrass beds with distinct contours, growing on identified substrates. Level 2 qualified areas with a more difficult but still reliable interpretation (water turbidity, swell, shadows) solved thanks to the lab technician's experience of the area and the help of external data. Level 3 concerned subjective interpretations and/or an absence of data. Ground truth points (observations from a rubber dinghy with an aquascope, one-off scuba dives and transect dives) were performed between 0 and -15 m at 3 861 points of questionable interpretation identified on the present pictures (Fig. 1).

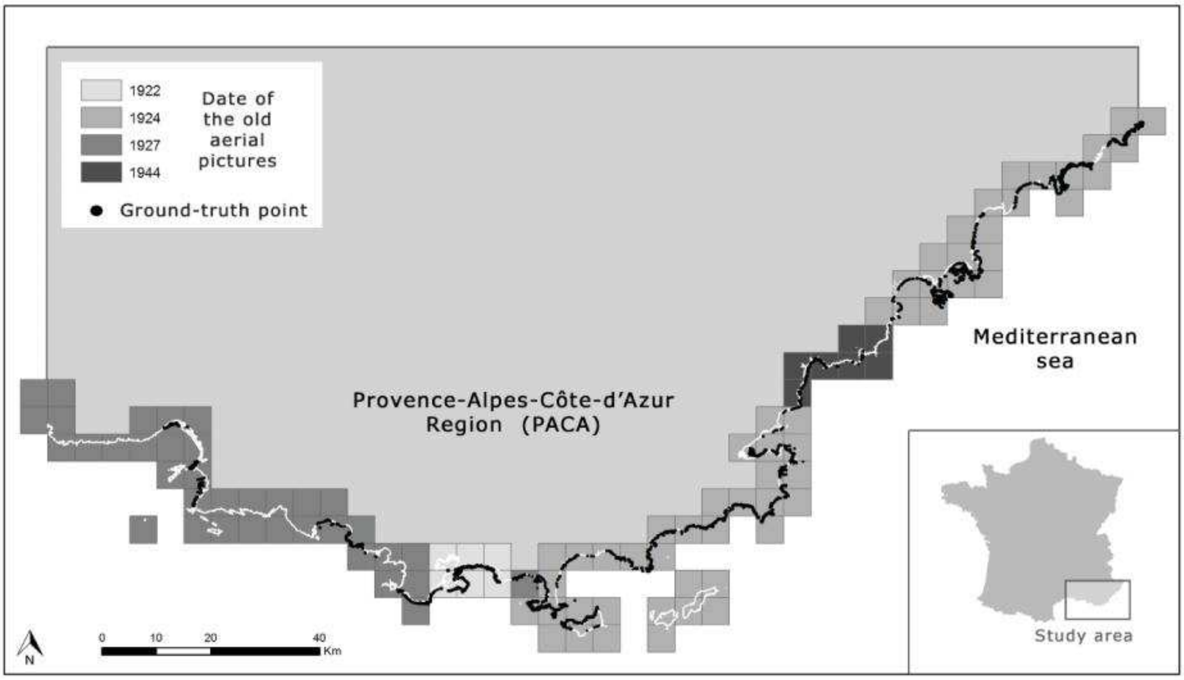

Figure 1. Location of the old pictures and of the ground-truth points (observations from a rubber dinghy with an aquascope, one-off scuba dives and transect dives) used to map *P. oceanica* beds along the Southern-East (Provence-Alpes-Côtes-d'Azur region) coastline (in white) at a 0 to -15 m depth.

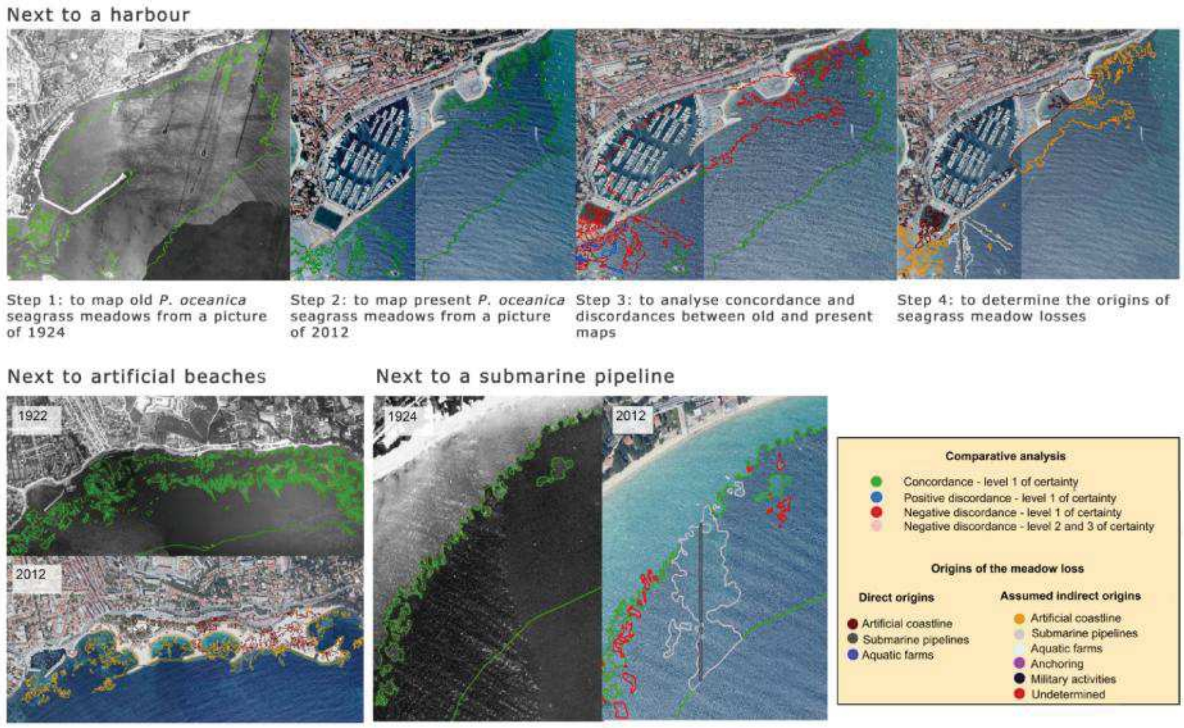

Figure 2. Three examples are taken in order to illustrate the methodology. Old and present aerial pictures were used to map old and present P. *oceanica* seagrass beds (steps 1 and 2), then the two maps were compared and analyzed in terms of concordances and discordances (step 3) and finally the negative discordances were attributed to the coastal settlements (step 4).Three levels of certainty were defined in order to qualify our confidence in the interpretation of the old pictures. Level 1 of certainty qualified seagrass beds with distinct contours, growing on identified substrates. Level 2 qualified areas with a more difficult but reliable interpretation (water turbidity, swell, shadows) resolved thanks to the lab technician's experience of the area and the help of external data. Level 3 concerned subjective interpretations and/or an absence of data. Only the most reliable maps (level 1 of certainty) were used to analyze direct and indirect origins of lost meadows (areas in negative discordance).

Comparative analysis and origins of meadow loss

Comparative maps were obtained after superposition of the layers containing the old and present P. *oceanica* beds distributions using a CAD software (Fig. 2). They were then vectorized within a GIS software. Polygons were automatically drawn from the raster (image) representing P. *oceanica* meadows. Changes between old and present sub-marine meadows were analyzed in terms of concordance (remaining areas), positive discordance (expanding areas) and negative discordance (lost areas). The proportion of declining shallow limits (in length) was estimated from the projection on the coastline of the negative discordant areas. Only meadows drawn with a level 1 of certainty were considered for these calculations. According to the working scale used (1:15 000, see above), we estimate that we were able to detect a 5 m minimum difference between old and present pictures/maps.

Where negative discordances (loss in areal extents) were observed, the comparison of old and present pictures also allowed to draw three types of coastal developments directly or indirectly impacting seagrass beds: artificialized coastlines (namely harbours, ports of refuge, landfills, artificial beaches, groynes and pontoons), submarine pipelines and aquatic farms. These types of developments were chosen according to MEDAM (2014). Their direct influence on the loss of P. *oceanica* meadow areas (level 1 of certainty) was acknowledged when former meadows have been physically replaced by these developments. Indirect impacts were assumed when losses were observed around these developments. We also considered the effects of anchoring and military activities when they were obvious (visible trails and bomb impacts on the meadows). As the responsible factors (boats, bombs…) were not observed on the pictures, no difference was made between direct and indirect impacts for those. The remaining losses, for which the origin could not have been determined, were classified as “undetermined origin”.

Impact distances

The impact magnitude of all different types of coastline settlements (harbours, ports of refuge, landfills, artificial beaches, groynes and pontoons) that caused losses (level 1 of certainty) were analyzed. The area (in m²) of every settlement of each type was calculated and the total (direct + indirect) area of meadow loss was estimated within a 200 m, 500 m, 1 000 m, 2 000 m, 5 000 m and 10 000 m radius from them. The magnitude of the impact (area of meadows destroyed for 1 m² built) was calculated as the ratio between the seagrass loss area and the settlement area within each radius. The maximal distance of impact was the distance from the impacting source where the increment of accumulated seagrass loss with increasing distance was equal to zero. Only settlements that were clearly identifiable owing to their remoteness were considered in order to avoid mixed effects. In the case of big principal buildings that necessitated other secondary constructions (harbour / landfills for example), the loss was assumed to be due to the principal building.

3.4 Results

In the 1920’s P. *oceanica* beds area along the coastline, between 0 and 15 m depth, was over 14 528.3 ha (considering the three levels of uncertainty). Around 36 % of this mapped area was dependent on the lab technician’s subjectiveness (level 3 of certainty) because of the bad quality of data. Present pictures revealed 13 111.8 ha of seagrass meadows (Table 1). Former meadows represented 14 528.3 ha among which 7 696.8 ha corresponded to level 1 of certainty; these 7696.8 ha were entirely covered by the present map. All the following results were based on data qualified by level 1 of certainty; it represented 53 % of the study site. The general map (with several zooms in order to make the visualization easier) used to obtain the quantitative data is presented in Figure 3. The high resolution entire map may be freely (with login) consulted online on www.medtrix.fr within the SURFSTAT project.

Table 1. Synthesis of seagrass beds mapped from present and old pictures. Three levels of certainty were defined in order to qualify our confidence in the interpretation of the old pictures. Level 1 of certainty qualified seagrass beds with distinct contours, growing on identified substrates. Level 2 qualified areas with a more difficult but reliable interpretation (water turbidity, swell, shadows) solved thanks to the lab technician's experience of the area and the help of external data. Level 3 concerned subjective interpretations and/or an absence of data. The comparison between old and present seagrass areal extents was based only on very reliable data (level 1 of certainty). Changes between old and present meadows were analyzed in terms of concordance (remaining areas), positive discordance (expanded areas) and negative discordance (lost areas). The assumed direct (physical replacement) or indirect (nearby coastal settlements) origins of negative discordances are indicated. All data are presented in terms of areas (ha) and percents (%).

		Area (ha)	Percent (%)
	Total - Present meadows	13 111.8	
	Total – Old meadows	14 528.3	
	Level 3 of certainty	5 216.6	35.9
	Level 2 of certainty	1 614.9	11.1
	Level 1 of certainty	7 696.8	53.0
Comparison between old and present meadows	Concordance	6 583.7	85.5
	Positive discordance	83.2	1.0
	Negative discordance	1 029.9	13.4
Origins of negative	Man-made coastline - direct	220.8	21.4
	Man-made coastline - indirect	281.2	27.3
	Submarine pipeline - direct	3.9	0.4
	Submarine pipeline- indirect	45.8	4.4
	Aquatic farms - direct	9.6	0.9
	Military activities	7.2	0.7
	Anchoring	2.7	0.3
	Undetermined	458.7	44.5

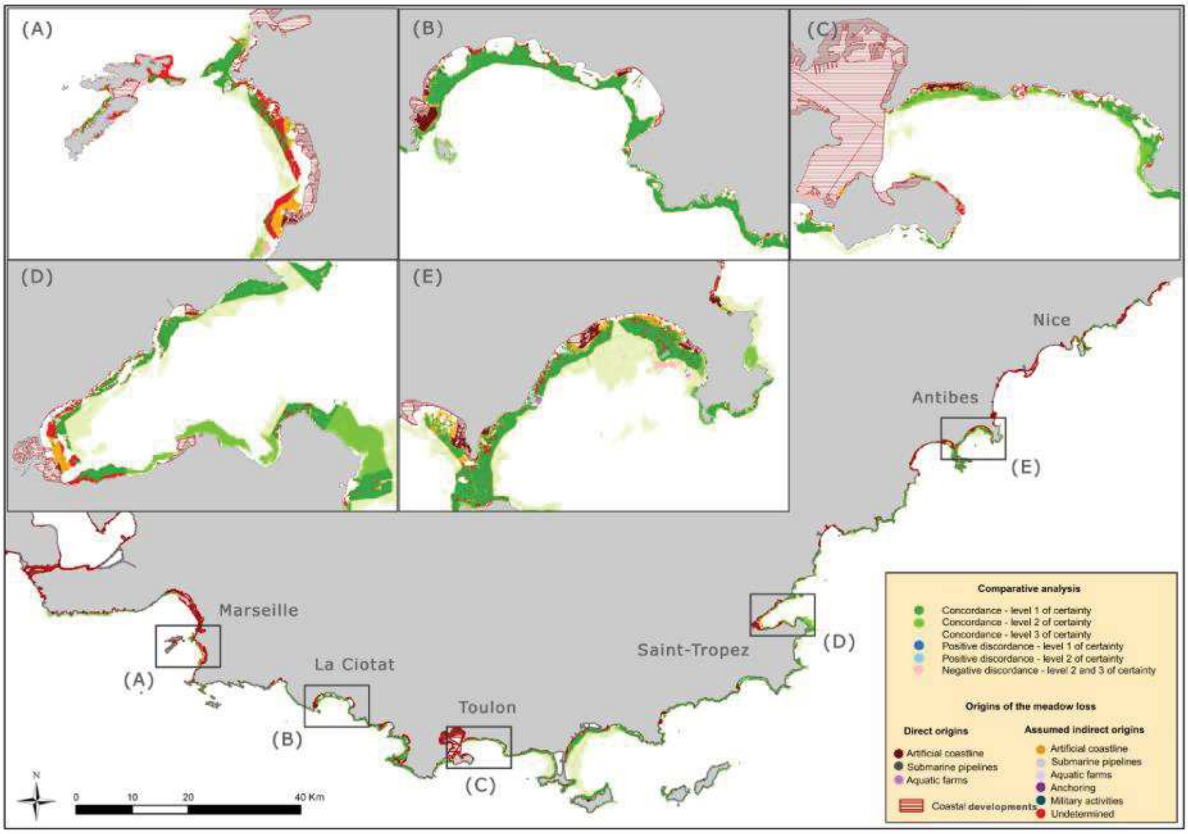

Figure 3. Overall comparative map showing changes (concordance, positive discordance or negative discordance) in *Posidonia oceanica* meadows distribution over 85 years of coastal development. Five sites (A, B, C, D, E) are taken as examples in order to better visualize local data. Three levels of certainty were defined in order to qualify our confidence in the interpretation of the old pictures. Level 1 of certainty qualified seagrass beds with distinct contours, growing on identified substrates. Level 2 qualified areas with a more difficult but reliable interpretation (water turbidity, swell, shadows) solved thanks to the technician's experience of the area and the help of external data. Level 3 concerned subjective interpretations and/or an absence of data. Only the most reliable maps (level 1 of certainty) were used to analyze direct and indirect origins of lost meadows (areas in negative discordance). Coordinate system: RGF Lambert 93 / IAG GRS 1980

The comparison between old and recent maps showed that 73 % of the shallow seagrass limits have declined. P. *oceanica* areal extents were essentially concordant: 6583.7 ha remained at the exact same place after 85 years, thus the 85.5 % of remaining areas (Table 1). The Var French department presented the highest part of remaining areas (92 %) compared to Alpes-Maritimes (73 %) and Bouches-du-Rhône (70 %) (Fig.4). Positive discordance was poor with only 83.2 ha (1%): this represents on average an expanding of 0.97 ha per year. This expanding occurred through small patches here and there, mostly in place of old bomb impacts or other past damages. On the contrary, negative discordance accounted for 1029.9 ha (13.4 %); this represents a loss of 12.1 ha per year or 332 m² per day (Table 1). The loss was the highest in the Bouches-du-Rhône French department (29 %) mainly around major cities. In the Alpes-Maritimes, the loss essentially occurred around Cannes, Cagnes-sur-mer and Nice. In the Var French department, the areal loss was concentrated around Toulon, Sainte Maxime and Fréjus (Fig. 4).

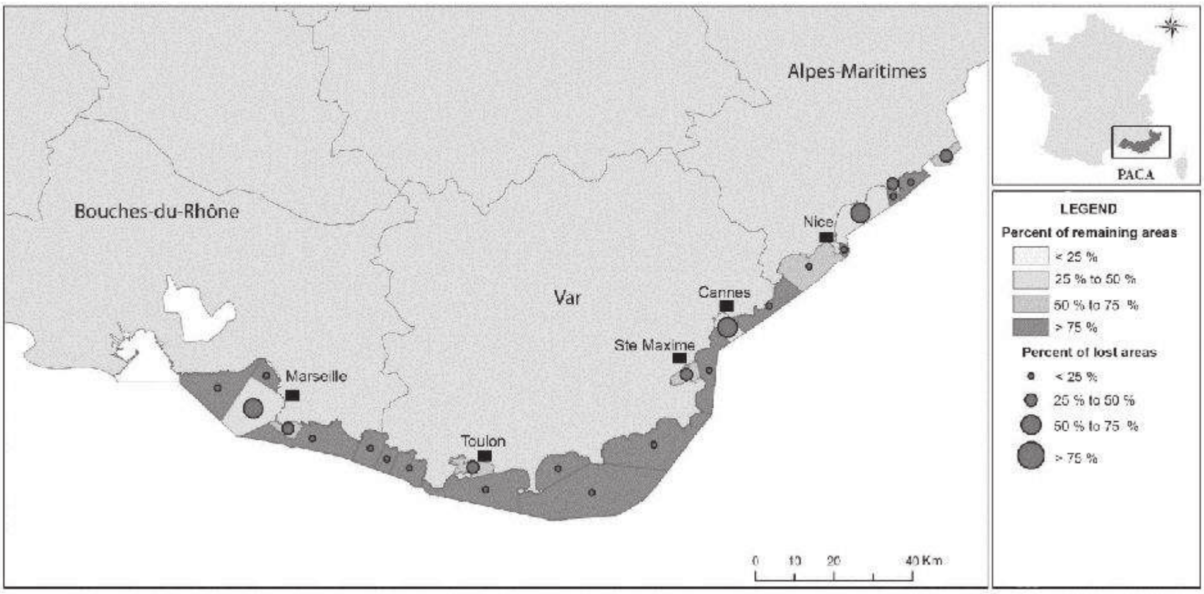

Figure 4. Percents of concordance and negative discordance between old and present mapped meadows, interpreted as remaining and lost areas of *Posidonia oceanica* meadows (between 0 and -15 m) observed per water body within the study site (PACA = Provence-Alpes-Côte-d'Azur region). Only maps with level 1 of certainty (=seagrass beds with distinct contours, growing on identified substrates) were used to define former meadows distribution. Coordinate system: RGF Lambert 93 / IAG GRS 1980.

The coastal facilities (physical holdings) under consideration were clearly linked to 55.5 % of the areal loss. The meadow loss mainly corresponded to artificialized (man-made) coastlines (48.7 %), with a weak disequilibrium in favor of indirect impacts (i.e. due to changes in water quality, turbidity or currents (hyper-sedimentation or erosion) during and/or after the installation) compared to direct ones (Table 1). The principal causes were harbours (83.8 %) and artificial beaches (10.3 %) (Fig.5). The second most important identified origin of loss were submarine pipelines (4.8 %). Other identified activities i.e. aquatic farms, military activities and anchoring represented less than 1 %, respectively 0.9, 0.7 and 0.3 %. After the analysis, 44.5 % of the losses remained undetermined (Table 1).

In total, 96 settlements were analyzed in the light of their impact distances on seagrass meadows: 5 groynes, 6 pontoons, 9 artificial beaches, 13 landfills, 21 ports of refuge and 42 harbours (Fig. 6). No meadow loss could be clearly linked to a given settlement beyond 5 km. The highest impact at the longest distance from its point of origin was caused by harbours: 2.9 m^2± 5.2 destroyed for 1 m^2 built at 5 km, the high variability being linked to the harbour size. Only artificial beaches presented such a distant impact (5 km) but with a lower strength (0.7 m^2 ± 0.7 destroyed for 1 m^2 built). Meadow losses caused by ports of refuge and landfills were visible on a shorter distance: respectively 2.2 m^2 destroyed at 1 km and 2.3 m^2 at 500 m. Groynes presented the shortest impact: 200 m (0.6 m^2). Pontoons were the least surrounded by areas of lost meadows: 0.3 m^2 at 200 m and 0.5 m^2 at 500 m, the maximal distance of impact.

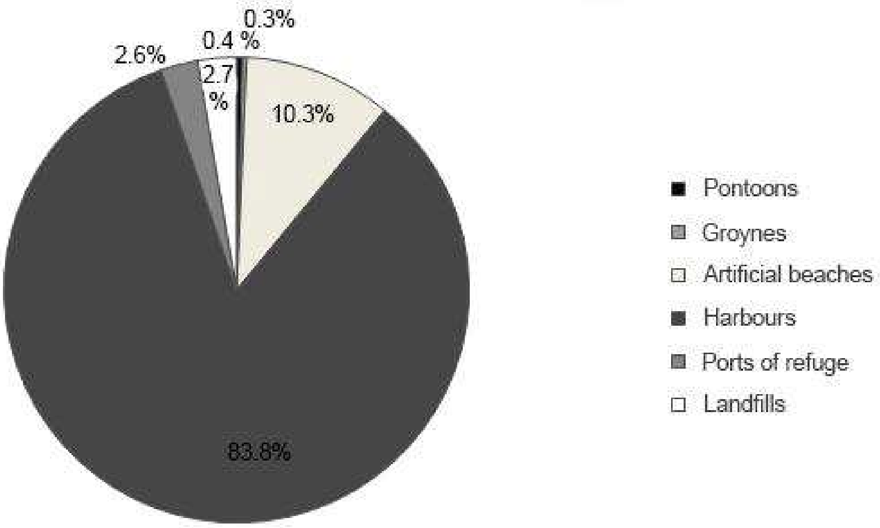

Figure 5. Relative importance (in percents) of the different types of man-made coastline developments responsible for the direct loss (in areal extents) of *Posidonia oceanica* meadows (between 0 and -15 m) within the study site. Direct losses assessment was obtained from the comparison between old meadows mapped with level 1 of certainty (=seagrass beds with distinct contours, growing on identified substrates) and present meadows.

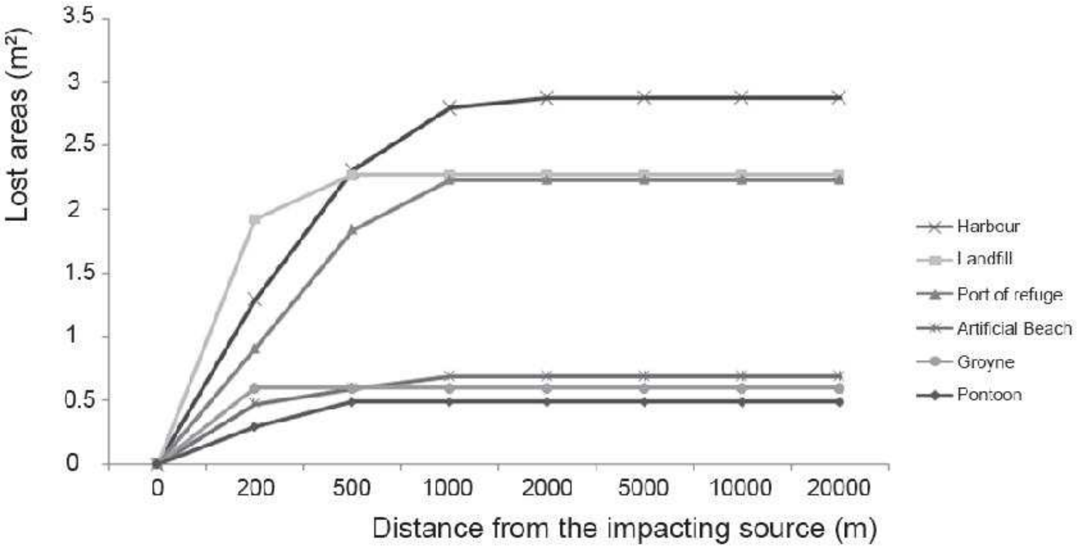

Figure 6. Lost *Posidonia oceanica* areas (in m²) according to the distance to different types of man-made coastline developments (96 facilities precisely) in meters. Results are presented for 1 m² of each type of development.

3.5 Discussion

P. oceanica meadows: declining limits and lost areas

As expected, P. *oceanica* seagrass beds have disappeared through a regression of their limits. Most of *P. oceanica* shallow limits (73 %) have declined over the last 85 years. With a loss of 13.4 % of the initial (1920's) meadow areas, this study confirms the overall loss (between 13 and 38 %) recently estimated by Marbà et al (2014b) and the 10 % generally assumed (see the introduction). Unfortunately, these values mostly concern the North Western Mediterranean while a lack of data regarding the Eastern and Southern Mediterranean makes it difficult to generalize to the entire basin.

Coastal settlements that we considered to explain this areal loss represent 55.5 %. At the same time, the cause of 44.3 % of lost meadow areas remains undetermined meaning that i) either no obvious single pressure (role of varied pressures) ii) or no visible pressure (invisible pressures or pressures considered to be not visible) could be observed next to these losses. i) Numerous regions locally concentrate varied pressures like coastal-based impacts, ocean-based pollution and maritime activities (Halpern et al 2008; Coll et al 2011). Marbà et al attributed 39 % of seagrass loss to more than one single pressure (Marbà et al 2014b). Undetermined losses could thus be due to a mix of close visible factors. ii) Invisible and factors that were not considered are: former settlements not visible on pictures anymore, hardly identified activities like anchorage, private swimming pool and rainwater discharges, but also changes in water characteristics (turbidity and sediment in deficit or in excess, salinity, temperature, chemical substances, pollution; see introduction) due to wastewater discharges, soil leaching, rivers... Marbà et al (2014) showed for example that 30 % of the meadows are impacted by water eutrophication. Almost 98 % of the contaminants found in the French part of the Mediterranean sea come from the rivers (the Rhône river alone is responsible for almost 75 % of them) (DIRM Méditerranée 2013) and 80% of urban sewage discharged into the Mediterranean is not treated. At global level, 80 % of the pollution of the marine environment comes from the land, the most important source being “non point-source pollution”, which occurs as a result of runoff (septic tanks, cars, trucks, boats, farms, ranches, and forest areas) (WWF 2014). The presence of exotic invasive macrophytes may also be counted as invisible factors (not visible on the maps). Exotic invasive species (in particular Caulerpa spp) are a priori not able to eliminate a healthy P. *oceanica* meadow (but see Lophocladia Lallemandii impact on healthy meadows (Marbà et al 2014a) and the meadows impacted by biological invasions (=2.4 %, Marbà et al 2014b)) but they can amplify the decline of stressed and degraded seagrass meadows that offer a favourable environment for their development (Boudouresque et al 2009). Finally, the observed loss of seagrass may also be due to global warming (higher water temperatures and rise in sea level). Indeed, P. *oceanica* is sensitive to high sea surface temperatures in summer (Mayot et al 2005; Celebi et al 2006; Marbà and Duarte 2010; Pergent et al 2014). Shallow water (0-80 m depth) warming in particular at -20 m (+ 1.4°C for the Spanish Catalan coast for example) was demonstrated for over last 30 years along the NW Mediterranean basin coasts (Prieur 2002; Salat and Pascual 2002; Vargas-Yáñez et al 2008; Boudouresque et al 2009; Pergent et al 2014) and especially after 2000 (Marbà and Duarte 2010; Pergent et al 2014).

Relative influence of the different coastal engineering on seagrass meadows

Coastal engineering is involved in half of the seagrass losses. The most important loss occurred around the largest coastal cities, especially in line with man-made coastlines (mainly harbors and to a lesser extent artificial beaches). This means that without considering wastewaters (counted with the pipelines) major cities play an important part in P. *oceanica*'s loss, mainly because of commercial, leisure and touristic activities. Man-made coastlines destroy areas a little more indirectly than directly according to previous local studies (Astier 1984; Boudouresque et al 2012). P. *oceanica* meadows were thus either buried by some coastal development or the related construction work, or died later because of the new conditions created close to a harbour (hydrodynamism, nutrient-epiphytes, grazers, siltation, pollution) (Ruiz and Romero 2003). The ones that did not die generally present a reduced productivity and abundance (Ruiz and Romero 2003). However, harbours are often old facilities thus means of action are limited. Focus can be made on containing the potential extension of their indirect impacts, by monitoring the frequentation, the water quality and the quality of the products used or inventing new ways of transportation (clean energy) and new ways of doing in ports. Even small settlements may have great impacts as for example pipelines directly involved in 4.8 % of the losses, mostly indirectly. On the contrary, the impact of anchoring is weak (0.3 %) but might be underestimated for three reasons: i) it is hard to identify the impacts of anchoring on the meadows using only aerial pictures, ii) the method consisted in matching the meadow loss around harbours first, implying artificialized coastlines leaving little possibilities to anchor and iii) the impact of anchoring is far greater beyond 10 m depth where the matte is less compact, the meadow more sensitive and the anchor chain longer (Andromède Océanologie 2014).

This thorough analysis of artificialized coastlines helps to assess the relative impact of the different settlements on the meadow loss. Although no impact was detected beyond 5 km, but most probably because of a doubtful detection (mixed effects) than an absence of impact, the present study shows that harbours are the most damaging man made coastal developments (2.9 ± 5.2 m² destroyed for 1 m² built over 5 km). Only beaches present such a distant impact (5 km) but with a lower strength (0.7 m² ± 0.7 destroyed for 1 m² built). Compared to these values, the "500 m safety distance" generally used for seagrass meadows seems ridiculous (Pergent-Martini et al 2006; Cabaço et al 2008; Tuya et al 2014). These data will be very useful for the modeling of anthropogenic pressure impacts and the prediction of the possible ecosystem services loss after construction works. For this protected plant which loss can hardly be compensated, the sequences "avoid" and "reduce" must be seriously taken into account. It is all the more important because estimating the real cost of these losses is hard. A recent work has identified 25 ecosystem services provided by P. *oceanica* meadows, among which eleven have been evaluated for their seven goods and benefits (Campagne et al, in press). The total value ranged between 283 and 513 € ha-1/ yr-1 which equates to 25.3 to 45.9 million € per year for the species. Under these conditions, a decline of 13 % generalized to the entire Mediterranean (3.5 million ha for now (Laffoley and Grimsditch 2009)) would represent a minimal loss ranging between 128.7 and 233.4 € per year in the contribution to human beings and their well-being. In addition to this annual economic loss, the destruction of P. *oceanica* represents a long-term decline in some ecosystem services usually provided, like the release of carbon, heavy metals and sediment sequestered until destruction in the matte.

Large remaining seagrass areal extents

Although most of the shallow limits have declined, remaining *P. oceanica* meadows areas are predominant (85.5 %), confirming there by former long-term studies led at smaller spatial scale (Pasqualini et al 2001; Bonacorsi et al 2013). Similarly, compiled published data analyzed by Marbà and her colleagues estimated the overall remaining areas ranging between 62 and 87 % since 1960 within the Mediterranean (Marbà et al 2014b). The restriction of the present work to shallow limits (0 to 15 m depth) generally presenting slower losses explains the highest proximity found with the maximal remaining value. Actually, while the shallowest depth limits generally go deeper according to an absolute rate of 0.04±0.1 m yr-1, the deepest limits decrease more than 10 times faster (0.61±0.29 m yr-1) (Marbà et al 2014b). However, most of *P. oceanica* shallow limits have declined (73 %) and this relative impression of stability considering the areal extents must also be adjusted with three biases of the study. i) the methodology does not detect any change inferior to 5 m (work scale = 1:5 000), ii) only the shallow part of the meadow that is expected to decline slower than the deepest (see above) is considered, iii) the value “85.5 % of remaining areas” only considers extents in areas and does not take into account the shoot density within this area while the average loss in shoot density was recently estimated to 27.51 shoots m-^{2} yr-1(Marbà et al 2014b).

The largest remaining areas were observed where the coastline is the least man-made and where changes in the coastlines had occurred previously to the beginning of the study and thus impacted the meadows a long time ago without any recolonization afterwards. This points out the fact that *P. oceanica* meadows are little resistant and not resilient. Actually, ecosystem stability is generally defined by two dynamics: resistance, as the ability to with stand disturbance, and resilience, as the ability to recover from disturbance (Pimm 1984). Seagrass meadows do not generally face important declines in sectors characterized by null or poor anthropogenic impacts (Boudouresque et al 2009) and its relatively quick response to disturbance has been extensively demonstrated (Ruiz and Romero 2003; Leoni et al 2006), hence its use as a bio-indicator (see introduction). Besides, resilience of *P. oceanica* is largely recognized as almost null (Boudouresque et al 2009; Boudouresque et al 2012; Pergent et al 2014).

The quasi-null resilience of *P. oceanica* meadows

Over 85 years, a very small expanded area has been observed: 0.9 ha yr-1representing only 1 %. This result was obtained from a large continuous area but a relatively small area compared to the estimated potential areal extent of P.*oceanica* in the Mediterranean (0.15 % for 76.97 km² analyzed from 50 000 km² of covered coastal seafloor estimated in the past, (Bethoux and Copin-Montégut 1986)). However, it confirmed the value (0.69 ha yr-1 = 1.31 %) obtained from a pool of 519 small studies covering in the end an area only a little larger (1 %) at the scale of the Mediterranean (Marbà et al 2014b). This very weak progression and resilience is a characteristic of climax ecosystems. *P. oceanica* meadows are a climax ecosystem found on most Mediterranean subtidal bottoms (Boudouresque et al 2012). Its clonal spread mode has allowed *P. oceanica* to maintain highly competitive clones over more than 100 000 years (it is the oldest living organism (Arnaud-Haond et al 2012)), and to develop extensive monospecific meadows protected from native competitors and major predators (Hemminga and Duarte 2000). However, 70 years old traces of bombs are still visible even in the middle of healthy growing meadows. The colonization of new areas and the recolonization of lost areas, via seeds, vegetative fragments or marginal spread of the meadow are extremely slow (horizontal growth is on average 1 to 6 cm / year (Marbà et al 1996; Marbà and Duarte 1998; Pergent-Martini and Pasqualini 2000; Boudouresque et al 2012)). Each loss being almost irreversible, this highlights the importance of combining all available means to prevent damage to the protected *P. oceanica* meadows.

3.6 Conclusion

This work is the first on *P. oceanica* led at such a large spatial and temporal scale in Mediterranean Sea; the results obtained are thus important for stakeholders, managers and environmentalists. Not with standing large remaining areas, most shallow limits have declined and shallow *P. oceanica* meadows have lost 13.4 % of their areal extent, i.e. 332 m² every day over the last 85 years. Decline is note worthy because once disappeared, recolonization is almost impossible. The influence of anthropogenic pressures is obvious especially of man-made coastlines. The quantitative impact of different types of costal settlements was highlighted but a large part of the loss remained undetermined. From now, it is a question of statistically testing the relative influence of each anthropogenic factor (including physical impacts and water eutrophication) but also of the environment of the undetermined losses. This will be done at a larger scale by considering the entire *P. oceanica* and dead matte distribution along the French coast. However, it is now also time to analyze human-driven impacts at a finer scale than the usual, namely a scale that would really allow designing management measures for marine key ecosystems. Indeed, an efficient conservation program relies on understanding the relationships between major threats and the ecological status of those ecosystems (Coll et al 2011).

Acknowledgments

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4 Article 3 : La cartographie de la biodiversité en trois dimensions met au les stratégies de conservation marine : l'exemple des assemblages coralligènes dans le nord-ouest de la mer Méditerranée.

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Mapping biodiversity in three-dimensions challenges marine conservation strategies: The example of coralligenous assemblages in North-Western Mediterranean Sea

Aggeliki Doxa[a,*], Florian Holon[a,b], Julie Deter[a,b], Sébastien Villéger[c], Pierre Boissery[d], Nicolas Mouquet[a]

[a] Université de Montpellier/Institut des Sciences de l'Evolution (ISEM), UMR 5554 CNRS IRD, Campus Triolet de l'Université de Montpellier, 34095 Montpellier Cedex 5, France
[b] Andromède Océanologie, 7 place Cassan, 34280 Carnon, France
[c] CNRS, Laboratoire Biodiversité marine et ses usages (UMR 5119 MARBEC), Université de Montpellier, Place Eugène Bataillon, 34095 Montpellier, France
[d] Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France

4.1 Abstract

Multi-facet diversity indices have been increasingly used in conservation ecology but congruence analyses both on horizontal and vertical axes have not yet been explored. We investigated the vertical and horizontal distributions of α and β taxonomic (TD), functional (FD) and phylogenetic diversity (PD) in a three-dimensional structured ecosystem. We focused on the Mediterranean coralligenous assemblages which form complex structures both vertically and horizontally, and are considered the most diverse and threatened communities of the Mediterranean Sea. Although comparable to tropical reef assemblages in terms of richness, biomass and production, coralligenous assemblages are less known and studied, especially because of their presence in deep waters. Our study covers the entire distribution of coralligenous habitats in the Mediterranean France, representing the most complete database for this important ecosystem up to now. To our knowledge, this is the first analysis of spatial diversity patterns of marine biodiversity on both horizontal and vertical scales.

We revealed that taxonomic diversity differed from functional and phylogenetic diversity patterns at the station level, the latter two being strongly structured by depth, with shallower stations generally richer than deeper ones. Phylogenetic diversity was less congruent to taxonomic diversity but more congruent to functional diversity than randomly expected. Low FD and PD values at both α and β diversity indicated functional and phylogenetic clustering. Communities' dissimilarities (β-diversity) increased over depth especially in central and eastern part of continental France and in north Corsica, indicating coralligenous vertical structure within these regions. Overall horizontal β-diversity was higher from 50 to 70m depth belts.

We conclude that taxonomic diversity alone is inadequate to set conservation goals for this ecosystem and additional information, at least on phylogenetic diversity, is needed to preserve ecosystem functioning and coralligenous evolutionary history. Our results highlight the necessity of considering different depth belts at regional scale conservation efforts. Current conservation approaches like marine protected areas are insufficient in preserving coralligenous habitats. The use of multi-facet indices should be considered, focusing on preserving local diversities and compositional dissimilarities both vertically and horizontally.

Key-words: alpha diversity, beta diversity, communities' dissimilarities, coralligenous outcrops, marine conservation, multi-facet diversities, vertical diversity

4.2 Introduction

A wide range of animal and plant species exhibit a vertically stratified distribution in terrestrial ecosystems, with most documented examples treating variations in species distributions along the vertical gradient from canopy to understory (ex. small mammals (Pardini et al., 2005), butterflies (Walla et al., 2004; Molleman et al., 2006) ants (Vasconcelos & Vilhena, 2006) and saproxylic species (Wermelinger et al., 2007)). Similar vertical stratification is documented along the water column in marine ecosystems, for fish larvae and zooplankton groups (Pilar Olivar et al., 2010), as well as in bacterial and archaeal communities (Ghiglione et al., 2008; Ye et al., 2009; Zinger et al., 2011). In marine ecosystems, most of the key abiotic factors (light, water movement, nutrient availability, sedimentation and temperature) vary strongly along bathymetric (vertical) as well as horizontal gradients, thus structuring species communities' composition (Bonecker et al., 2014). Whether vertical or/and horizontal gradients structure more species communities may significantly modify conservation efficiency but such aspects still remain poorly documented.

Except from being a proxy of environmental gradients and structuring species ecological niche, depth is also related to human pressure. In the Mediterranean basin, most human activities that may have an impact on marine ecosystems are depth related; recreational fishing, navigation and/or diving activities, for instance, take place in rather shallow waters (less than 50 meters) setting thus some depth belts more prominent to perturbations than others (Meinesz & Blanfuné, 2015). However, other sources of perturbations, like professional fishing, mechanistic destruction (ex. submarine cable placement) and pollution coming from terrestrial or marine sources can act as a disturbance to even deeper zones (especially in > 50 meters depth belts) (Grall & Hall-Spencer, 2003; Meinesz & Blanfuné, 2015). Finally, as recently reviewed over a 30-year period analysis, depth remains an important factor for marine conservation efforts in the Mediterranean basin, with more recent focus being given to deeper than 50 meters ecosystems (Meinesz & Blanfuné, 2015). However, the network of marine protected areas (MPAs) lacks of a specific planning and of specific ecological criteria for the selection of the target depth belts. Indeed, most Mediterranean MPAs were established with limited ecological, social, and economic data (Claudet & Pelletier, 2004; Claudet et al., 2006).

While some taxonomic groups, like fish and seagrass meadow communities, have been studied extensively within the Mediterranean basin (Ruíz et al., 2009; Mouillot et al., 2011) and used as indicators for European environmental policies (Devlin et al., 2007; Gobert et al., 2009; Personnic et al., 2014), deeper marine ecosystems remain poorly known (Cartes et al., 2004). Among the most vulnerable and diverse ecosystems within the Mediterranean basin, coralligenous outcrops are comparable in species richness and abundance to tropical reef assemblages (Bianchi & Morri, 2000; Ballesteros, 2006). Coralligenous reefs are found between 20 to 120 meters depth and are composed of a hard substrate of concretions of biogenic origin, produced mainly by the accumulation of encrusting algae growing at low light levels (Garrabou and Ballesteros 2000). Little was known about this highly diverse ecosystem until recently (Ballesteros, 2006), and it is only during the last few years that technical diving improvements have enabled their systematic survey over extended spatial zones (Deter et al., 2012a, 2012b). As they have little to no resilience against perturbations, due to their particularly low increase rate, of only 0.006 to 0.83 mm/yr in the western Mediterranean Sea (Littler, 1991; Sartoretto et al., 1996), the risk of extinction may be high given current and future global changes. However, no regional diversity analysis has been conducted to date on this important Mediterranean habitat.

Although taxonomic indices have been traditionally, widely used in conservation, recent research shows that other biodiversity facets, such as species ecological traits (functional diversity) and species evolutionary history (phylogenetic diversity), are important to ecosystem functioning (Hooper et al., 2005; Mouquet et al., 2012; Srivastava et al., 2012). More importantly, the spatial patterns of these two biodiversity facets are often not congruent with taxonomic diversity patterns (Forest et al., 2007; Devictor et al., 2010a), since the three facets result from different processes in community assembly, and in turn provide relevant inputs for the identification of conservation targets (Devictor et al., 2010b; Mouillot et al., 2011). It has thus been increasingly acknowledged that conservation planning should focus on preserving ecosystem processes and functions and more effort should be made to identify areas that preserve functional traits and evolutionary legacies at various spatial scales (Abdulla et al., 2009; Coll et al., 2012). However, diversity's multidimensionality is not yet reflected in existing conservation planning. MPAs, for instance, used as one of the main conservation tools worldwide, are designated on the basis of a few charismatic taxa (e.g. fish, mammals), not via an integrated ecosystemic approach (Fraschetti et al., 2005).

Exploring the vertical vs. horizontal distribution for all three diversity facets is a completely unexplored field. We chose to address these questions on the Mediterranean coralligenous assemblages, because of their complex structures both vertically and horizontally, and their high level of richness and vulnerability. In order to understand and preserve the three-dimensional nature of these high diversity assemblages, it is urgent to examine compositional dissimilarity levels along both vertical and horizontal gradients, especially since all previous and existing conservation efforts have focused on rather shallow coastal areas (< 40 m), disregarding deeper ecosystems (Abdulla et al., 2009). To answer this conservation issue, a preliminary step is to combine measures of diversity distribution (community diversity = α diversity and between-community dissimilarities = β diversity) (Legendre et al., 2009; Veech & Crist, 2010) in the three dimensions. To our knowledge, this has to date never been done using the three facets of diversity for any studied terrestrial or marine ecosystem.

We assessed taxonomic, functional and phylogenetic diversities of North-Western Mediterranean coralligenous assemblages along both horizontal (1 700 km) and vertical (73 m) gradients, which represents the entire distribution of coralligenous habitats in the Mediterranean France. Comparing to previous studies (Balata et al., 2006; Piazzi et al., 2009; Piazzi & Balata, 2011), which considered a small fraction of the possible depths (30 to 40m) at which one can encounter the coralligenous concretions, our database represents the most complete distribution range ever used for the studied organisms. We aim to answer the following questions: 1) How are the three facets of diversity spatially distributed and what is their degree of congruence? 2) How does spatial distribution of biodiversity vary from the horizontal to the vertical axis? 3) To what extent does the existing network of marine protected areas encapsulate the multidimensionality of coralligenous biodiversity (vertical/horizontal, taxonomic/functional/phylogenetic)? Our results have important implications not only for the conservation biology of coralligenous assemblages but also for any kind of ecosystem organised along both horizontal and vertical dimensions.

4.3 Materials and methods

Study area and coralligenous assemblages monitoring

The coralligenous monitoring program started in 2010 and continues until now. Three regions, composing the French coralligenous distribution area, are surveyed: (i) Eastern Provence-Alpes-Côte d'Azur (PACA), (ii) Corsica and (iii) Languedoc-Roussillon and Western PACA regions. Given the complexity of the survey, monitoring is conducted every year during June in one of these regions. Three consecutive years are thus needed to obtained comparable data along the entire monitored area. We used data from 2010 to 2012 which corresponded to one year of survey per region. During this period, a total of 120 stations were surveyed along the French Mediterranean coast (Fig. 1). In these ecosystems, temporal changes from one year to another are rare compared to spatial changes (Garrabou et al., 2002; Deter et al., 2012a), except in cases of extreme stochastic perturbations (Garrabou et al., 2009; Teixidó et al., 2013). No such extreme event occurred along the French coast since 2006, so no time effect was considered in our analysis.

Monitored stations were chosen as follow. First, all areas, where coralligenous assemblages could potentially occur, were identified based on a bathymetric cartographic analysis (http://observatoire-mer.fr/en/surfstat.html). Then, 69 sites (Fig. 1) were chosen to represent different environmental conditions, according to physico-chemical characteristics of coastal water bodies. Following the European Water Framework Directive, these water bodies were defined by the French water agency, dividing coastal waters into homogenous water bodies based on catchment areas (WFD, Directive 2000/60/EC). At each site, divers recorded the upper and lower limit of coralligenous concretions and conducted a survey stop every 10 meters within the local bathymetric extension. Each site thus contained one or several stations (Fig. 1).

A survey was performed at each station using a non-invasive method consisting of taking 30 photographic quadrates (50 x 50 cm) along a 40-meter long horizontal transect as described in Deter et al. (2012b). Photographic methods are commonly used for the analysis of coralligenous assemblages because they are non-destructive, save time underwater, are repeatable, and permit data archiving (Ferdeghini et al., 2000; Virgilio et al., 2006; Baldacconi & Corriero, 2009; Kipson et al., 2011; Deter et al., 2012a, 2012b). All photographs at a given station were taken from the same orientation and depth, using a compass and a depth meter attached to the quadrate. Station depth varied between 17 and 90 m, but more than 90% of all surveyed stations were between 30 and 70 meters deep. Station's distribution over depth is presented in Appendix S1. Stations were grouped into four depth categories for β-diversity analyses: 30-40, 40-50, 50-60 and 60-70m deep. Grouping stations in depth categories requires having a sufficient number of stations in each category. We thus excluded very shallow and very deep stations (i.e. four stations between 17-24 m and five stations between75-90 m), keeping 111 stations for further analysis.

Coralligenous assemblages were identified for each station based on the photographs. Each picture was analysed by a single person using 64 random points via the coralligenous assemblages version of CPCe 4.1 (Kohler & Gill, 2006). This methodology was validated using part of the analysed dataset (Deter et al., 2012b). All points that indicated mobile organisms such as fish and crustaceans, as well as unidentifiable organisms, or abiotic substrate (rock, sand, rubble) were excluded from further analysis. This left more than 90% of the total points in each station. Finally, we identified coralligenous species according to the taxonomic nomenclatures of Guiry and Guiry (2013) and Rodriquez-Prieto et al. (2013). In some cases, identification was not possible at species level, but only at genus level. Hydrozoa and encrusting Bryozoa were not identified further and were classified as “Hydrozoa” and “Encrusting Bryozoa”, each being considered a “species” for further analysis.

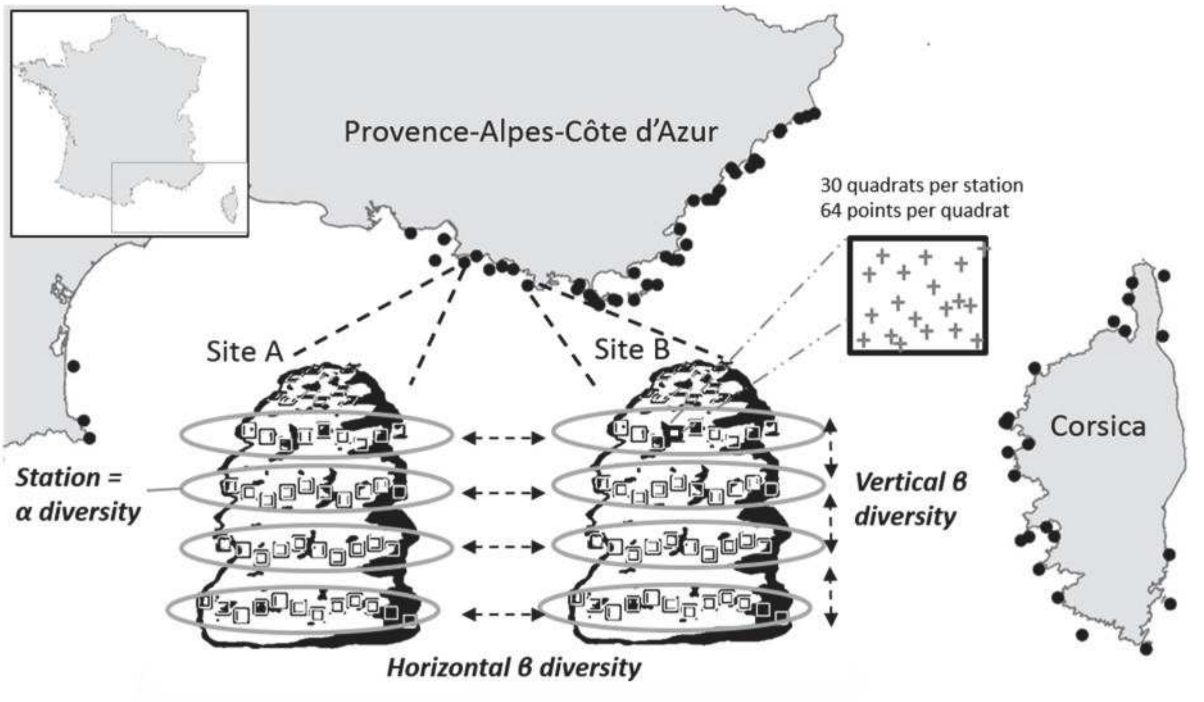


Figure 1. Map of study zone. Coralligenous assemblage sites are shown by black dots. Within each site, several stations at different depths were surveyed (grey ellipsis). Within each station, 30 photographs were taken along a 40m long horizontal transect. α diversity facets are estimated at station level. β-diversity was computed horizontally i.e. among stations within the same depth category and vertically i.e. among stations within the same region.

Assessing facets of biodiversity

We measured the three facets of coralligenous assemblages’ biodiversity (i.e. taxonomic, functional and phylogenetic) at two spatial scales: (i) station level i.e. α- diversity and (ii) between-stations level i.e. β-diversity.

Assessing functional and phylogenetic diversity of assemblages requires accounting for the functional or phylogenetic distances between species present in the regional pool. Given that no a priori information exists on the functional role of coralligenous species and thus on the traits that are related to these functions, we considered a set of complementary traits describing the diverse facets of the coralligenous species niche (Appendix S2): morphology, reproduction mode, feeding strategy, defense and distribution traits. This functional database was built using information compiled from the existing bibliography (Hofrichter, 2004, 2005; Rodriquez-Prieto et al., 2013; http://doris.ffessm.fr/, Copyright 2012), with in some cases the addition of experts' judgement (Ballesteros, pers comm). We considered a large set of traits to avoid the risk of over-estimating functional redundancy (Calba et al., 2014).

To avoid analysing inter-correlated functional traits, we computed the weighted Gower's distance between taxa pairs (Gower, 1971). We then performed a principal coordinate analysis (PCoA), which is the appropriated method for building functional space when functional traits are not continuous (Villéger et al., 2008). The PCoA was conducted using the functional distance matrix to build an 8-dimensions functional space. The Euclidean distances between species in this functional space accurately reflect the initial Gower's distances based on traits values. The number of dimensions was selected according to the best indicator available to date. (Appendix S3).

As for most studied species no sequence data was available, we estimated phylogenetic distances between species based on species cladistics (Appendix S4). Phylogenetic distances between species were then estimated as the distance between a set of tips of the phylogenetic tree, calculated as the number of nodes separating each pair of species encountered in each community (Webb, 2000). Given that our measures of phylogenetic similarity are based only on the identified taxa within our communities, phylogenetic diversity does not reflect absolute species dissimilarities, as no complete dated phylogeny exists for all coralligenous species, but only relative ones, which does allow us to comparatively distinct communities within our study zone according to species relatedness, from similar to less similar ones (Webb, 2000).

Taxonomic, functional and phylogenetic α-diversity were computed using the Rao's quadratic entropy (Q) index. A range of different indicators exists to assess functional and phylogenetic diversity (see Cadotte et al., 2010; Mouillot et al., 2013). The Rao quadratic entropy index has the advantage to account for species abundances and to be adaptable for all three diversity facets (i.e. TD, FD, PD) which is essential to conduct diversity facets comparisons and explains its use in many similar to ours studies (e.g. Devictor et al., 2010b; Mouillot et al., 2011). The Rao quadratic entropy index combines species dissimilarity and species relative abundance (Pavoine et al., 2004; Ricotta, 2005; De Bello et al., 2010): $Q = \sum_{i=1}^{S} \sum_{j=1}^{S} d_{i} \; p_i p_j$. In this case, p stands for the estimate of relative coverage of each taxa at the station level, estimated as the number of points per identified species per station. The distance between species pairs (d) for computing functional diversity (FD) was the distance in the 8-dimensions functional space after scaling values by dividing by their maximum. Similarly, phylogenetic diversity (PD) was computed considering dij equal to the number of nodes among all pairs of species as a proxy for the between them phylogenetic distance and then divided by their maximum. Both functional and phylogenetic Rao's quadratic entropy was computed based on ultrametric distance matrices. Taxonomic diversity (TD) was computed with dij=1 for all species pairs, which yields the Simpson diversity index (Ricotta, 2005).

Raw values of TD, FD and PD were transformed into equivalent number of species (Ricotta & Szeidl, 2009) by applying the following transformation: Qeqnbsp=1/(1-Q). Such transformation allows comparison of values among communities for the three facets. Low FD or PD values indicate that the dominant species within a community are functionally or phylogenetically close, whereas high FD or PD values indicate communities dominated by species that have distinct functional traits or from different lineages.

We assessed two types of β-diversity, horizontal and vertical. Our aim was to investigate how station dissimilarities vary according to their vertical or horizontal distances. A schematic representation of both vertical and horizontal β-diversities is given in Fig. 1. Vertical distances were estimated by subtracting station depths. Horizontal distances between each pair of stations were estimated as Euclidean distances using the latitude and longitude coordinates of each station. However, as this method makes no distinction over land or sea areas, we also considered horizontal distances by estimating the shortest distance from sea among all pairs of stations, using thus a mask for land areas. Analyses were further conducted using both distances (see Appendix S5 for further details and results).

As horizontal structure can vary among depths, we considered four categories of sampled depths each spanning 10 m (also referred to as depth belts), ranging from 30 to 70 meters. We measured pairwise horizontal β-diversity as follow. We considered all stations within a specific depth belt, i.e 36 stations within 40-50m and 25 stations within each of the other three depth categories (30-40m, 50-60m and 60-70m). We considered all pairs of stations within a given depth belt. We then estimated the mean α diversity for each pair of stations, based on Rao's index expressed as equivalent number of species (Ricotta & Szeidl, 2009; Villeger et al., 2012). γ-diversity for each pair of stations was computed using the same index based on average taxa abundance in the two stations. Using the multiplicative framework built on the equivalent number of species, β-diversities were then estimated as the ratio between γ and α diversities (Villeger et al., 2012). In addition to pairwise β-diversity we also calculated overall β-diversity in each depth category as the ratio of overall γ diversity of all stations situated within each depth category over the mean of their α diversity values.

For vertical β-diversity we considered 7 geographical regions: 4 along continental France [western PACA with 11 stations, central PACA (16 stations), central-eastern PACA (18 stations) and eastern PACA (13 stations)] and 3 regions around Corsica island [northern Corsica with 18 stations, central Corsica (16 stations) and southern Corsica (15 stations)]. The aim of identifying geographical regions and depth belts is to focus on coralligenous structure at a spatial scale that might correspond to the species biology and may be adapted for conservation purposes. A better approach would be to define regions based on the actual connectivity level among assemblages, in order to more accurately reflect species dispersal capacities, but no adequate data exist so far for coralligenous species. Vertical β-diversity was not computed in the four stations located in the Languedoc Roussillon region because in this area coralligenous concretions occur only at low depth. We then estimated pairwise vertical β-diversity following the same procedure as with the horizontal β-diversity. We additionally calculated overall β-diversity per region as the ratio of overall γ-diversity of the region over mean α-diversity of all stations situated within the same region.

Maps representing spatial distribution of biodiversity facets were produced using kriging models which automatically fit a variogram to the data on which they are applied (Hiemstra et al., 2010). Kriging models were only considered to produce maps and no further statistical analysis was conducted on them. All statistical tests concerning spatial structure and congruence were thus based only on observed data coming from the surveyed stations. Statistical analyses, plots and maps were produced with R (3.1.0) (R Development Core Team, 2014) using automap, ecodist, gdata, mgcv and vegan packages.

Congruence among diversity facets

To test congruence levels among the diversity facets, we used partial Mantel tests. These tests calculate the spatial correlations between two dissimilarity matrices (e.g. TD and FD distance matrices), conditioned by a third matrix corresponding to the geographical distances between stations (Legendre & Legendre, 2012). Congruence was estimated for the three pairwise combinations between diversity facets, i.e. TD, FD and PD. In order to investigate whether and how congruence levels vary over depth, we considered congruence levels in three different cases, i.e. taking all stations (n = 111 stations) and dividing stations into two depth range: stations up to 50 meters deep (n = 59 stations) versus deeper stations (n = 52 stations).

Two supplementary analyses were used to test for significance and confirm that the observed levels of congruence among diversity facets were not driven by the dominance of a few taxa. We first considered null models where species were randomly sampled from the study pool i.e. the pool of all species encountered in our study, according to their relative abundances (Calba et al., 2014). We repeated this random sampling procedure 1000 times, estimated the corresponding simulated TD, FD and PD and their congruence levels. We finally compared the observed levels of congruence with the distribution of the simulated data. This null modelling approach allows the validation of an ecological hypothesis, i.e. whether the observed congruence is different from what we should expect given the regional species pool, and is preferable than using the permutations procedure incorporated in the Mantel tests, which can in some cases be biased (Guillot & Rousset, 2013). In addition to that, to investigate relationships between FD and PD independently of the TD effect, we estimated net FD as the standardised residuals of the generalized additive model (gam) predicting FD from TD (Devictor et al., 2010b). The net PD was respectively computed from the gam (PD~TD). We finally estimated the congruence between net FD and net PD and compared it to the corresponding simulated data of these net indices. All plots of the relationships among diversity facets and the congruence levels between net FD and net PD are shown in Appendix S6. As both approaches revealed similar results, we further estimate robustness of observed congruence levels using the null modelling approach.

Diversity structure over space

We investigated how α- and β-diversity were structured over space. We used gam models with depth, longitude and latitude as the explanatory variables for α-diversity (Wood & Augustin, 2002). A spline effect was considered for all factors with three degrees of freedom (df) and the Fisher test was used to estimate the significance of explanatory variable trends. We considered separately horizontal and vertical β-diversities. We used generalised linear models (glm) to test for significant trends in β-diversity over vertical or horizontal distances for each of the regions or depth belts considered. For this specific analysis, our goal is not to identify the curve (linear or non-linear relationship) that best fits the data, in order to minimize residuals (as we did through gam for the alpha diversity), but to compare trends and identify zones where beta diversity increases or decreases over distance (in terms of depth or geographical terms) in higher rates than elsewhere. We additionally used overall β-diversities to compare dissimilarity levels among depth categories and regions.

Assessing the efficacy of Marine Protected Areas (MPAs) for coralligenous habitats

In a recent review, Meinesz & Blanfuné (2015) thoroughly documented how to properly define protected zones in the Mediterranean France in order to evaluate their development over the last 30 years. Following the European Community Habitat Directive, supplementary protection networks, such as the Natura 2000 sites, were set up since 2009, and are commonly considered in France as part of the marine protected areas network (Meinesz & Blanfuné, 2015). However, management and regulation specificities among sites led the authors to consider as MPAs only those areas where specific restrictions on recreational and/or commercial fishing are imposed. Thus only areas including specific no take zones i.e. complete restriction of fishing, are considered. This is also consistent with the definition of MPAs in other European countries. We thus considered only MPA sites and disregarded Natura 2000 sites from our analysis to be consistent with the definition of MPAs in Europe.

We estimated the proportion of the three facets of biodiversity found within the marine protected areas according to Mouillot et al. (2011). First, we ranked coralligenous stations from the most to the least diverse for each facet of biodiversity. Second, we calculated the cumulative percentages of protected stations along these decreasing diversity gradients. Finally, we estimated the overall proportion of protected stations among all the stations.

We used the overall proportion of protected stations as a reference level to assess which facet of diversity is under- or over-represented in the protected areas. If MPAs are effective in protecting coralligenous habitats, we expected an over-representation of highly diverse stations located in protected areas. On the contrary, an under-representation of highly diverse stations i.e. less than the overall proportion within protected areas, would indicate low MPA efficacy (Mouillot et al., 2011).

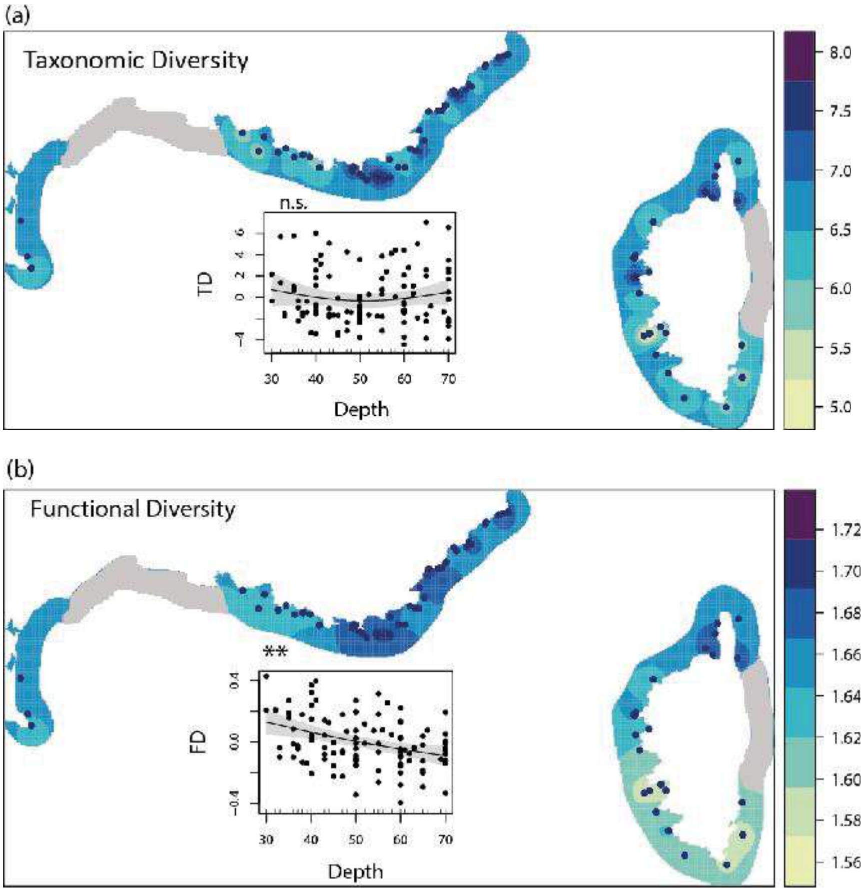
Three distinct cases were considered based on (i) all stations, (ii) on stations up to 50 meters deep and (iii) on more than 50-m deep stations. For all three cases, we assessed the cumulative proportion of taxonomic, functional or phylogenetic diversity present in MPAs.

4.4 Results

Vertical and horizontal distribution of diversity facets

Taxonomically richer communities were found in the central and eastern parts of the French Mediterranean coastline and in the northern part of Corsica (Fig. 2(a)). Functionally richer communities were observed principally along the eastern French coastline, whereas phylogenetically richer communities were mainly observed in Corsica and along the central continental coastline (Fig. 2(b) & 2(c)). Mean α-diversities ranged for TD from 1.9 to 13.9, for FD from 1.2 to 2.1 and for PD from 1.3 to 5.7 in equivalent species numbers. Thus, FD was considerably lower than PD and TD at the station level. The concept of number of equivalent species implies that FD and PD could not be higher than TD. FD can only be equal to TD, when all species have the same abundance and are maximally dissimilar. TD was thus more variable than FD and PD, since coralligenous species in each assemblage tend to have uneven abundances and be phylogenetically and functionally not highly dissimilar.

We further focused on each facet's variation to analyze their respective spatial pattern. Analysis of α-diversity structure over space revealed that TD did not significantly vary over depth ($p = 0.4$; Fig. 2(a)). However, both α-FD ($F = 7.8$, $p < 0.01$; Fig. 2(b)) and α-PD ($F = 5.2$, $p = 0.02$; Fig. 2(c)) significantly decreased with depth (Fig. 2). α-FD and α-PD increased with longitude ($F = 5.0$, $p = 0.03$ and $F = 7.1$, < 0.01 respectively), with higher FD and PD in eastern regions. All three facets increased with latitude (TD: $F = 4.3$, $p = 0.02$; FD: $F = 7.5$, $p < 0.001$ and PD: $F = 3.5$, $p = 0.03$).



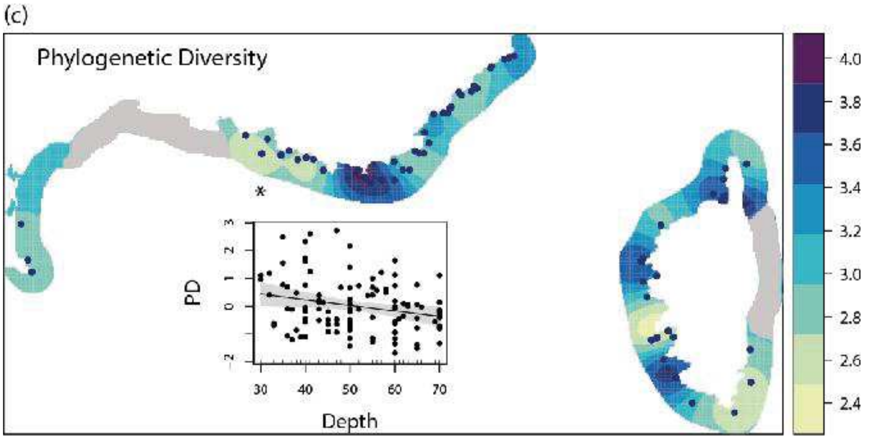


Figure 2. Spatial distribution of the three α diversity facets: (a) TD, (b) FD and (c) PD. Low diversity levels are indicated in light yellow and high diversity levels in blue. Blue dots indicate the exact position of stations. Areas where no coralligenous assemblages are present are indicated in grey. α diversity structure over depth is shown in plots. n.s. indicate non-significant trends, * indicates significant ($p < 0.05$) and ** highly significant trends ($p < 0.01$).

Mantel correlation analysis revealed that α-FD and α-TD were not more congruent than expected from the random model (Fig. 3(a)), and similar results were obtained when considering two categories i.e. up to 50 meters stations or deeper stations (Fig 3(b)-3(c)). Interestingly, while α-PD was less congruent to TD than expected at random (Fig. 3(d)), congruence between FD and PD was higher than expected at random (Fig. 3(g)). Similar results were observed for deeper than 50 meters stations (Fig. 3(f)-3(i)), with FD being even more congruent to PD than previously observed i.e. 0.84 *vs* 0.69 for all stations (Fig. 3(g) *vs* 3(i)). For shallower than 50 meters stations congruence was not significantly different than random distributions for any facet considered (Fig. 3(b), 3(e) & 3(h)). Congruence levels between FD and PD remained significantly higher than random even when retrieving the TD effect for FD and PD i.e. net FD and net PD (see Appendix S6).

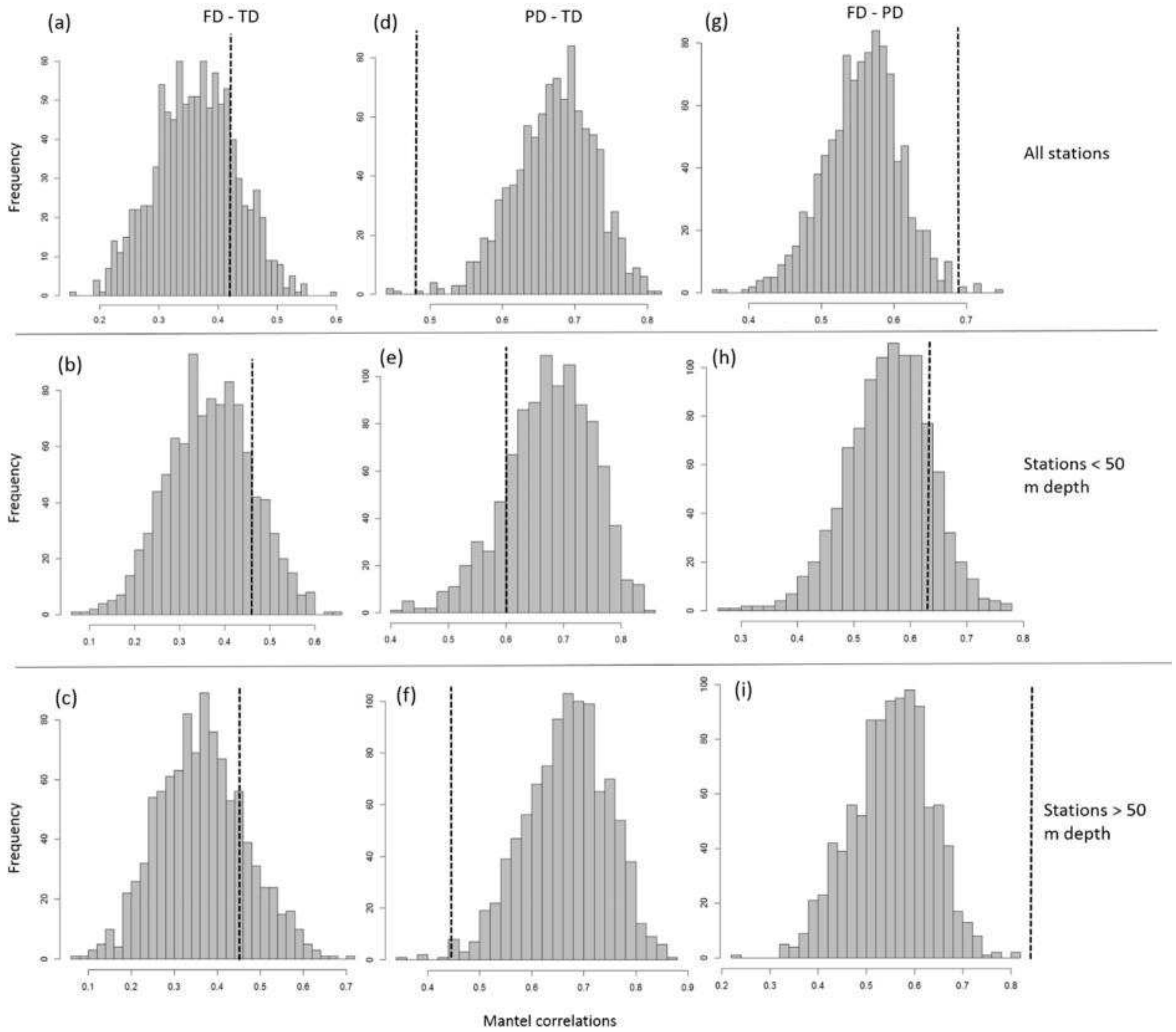
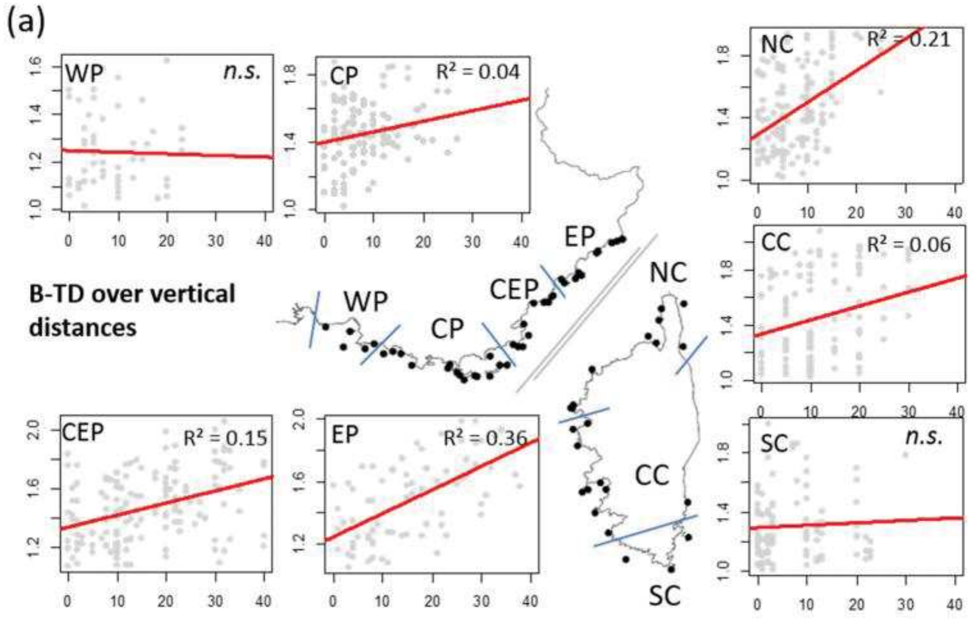
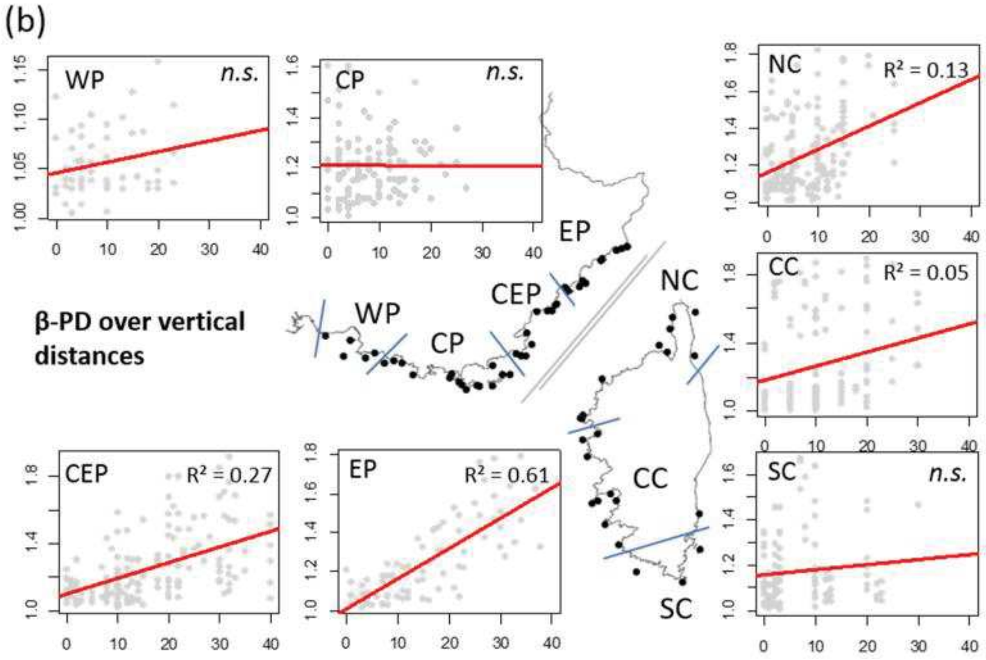


Figure 3. Congruence levels among diversity facets: (a)-(c) FD-TD, (d)-(f) PD-TD and (g)-(i) FD-PD for all stations considered and according to two depth ranges i.e. < and > 50m deep stations. The robustness of the observed congruence levels (indicated with the dotted lines) is tested using the distribution of the simulated data coming from the null models (shown as grey bars).

Vertical β-diversity mainly increased with depth for all three diversity facets (Fig. 4). Vertical taxonomic structure varied among regions, with the eastern PACA (t = 6.71, p<<0.001) and northern Corsica (t = 6.43, p<<0.001) being the most vertically structured regions i.e. where dissimilarities among stations highly increase over depth (Fig. 4(a): EP & NC). In central Corsica and central east PACA (Fig. 4(a): CC & CEP), taxonomic β-diversity increased with depth (t = 3.03, p<0.01 and t = 5.20, p<0.001 respectively), but less than in eastern PACA. No trends in taxonomic β-diversity were observed over depth within western PACA (WP: p = 0.8) and southern Corsica (SC: p = 0.6). Vertical functional and phylogenetic β-diversities showed similar patterns in all regions (Fig. 4(b) for PD trends and Appendix S7 for FD trends). Functional and phylogenetic β-diversities mostly increased with depth in eastern PACA (t = 9.35, p<0.001 and t = 10.94, p<0.001 respectively), in central eastern PACA (t = 6.65, p<0.001 and t = 7.51, p<0.001 for FD and PD respectively) and in north Corsica (t = 4.19, p<0.001 and t = 4.83, p<0.001 respectively). Non-significant trends were obtained in south Corsica, west and central PACA regions for both functional β-diversity (p = 0.9, p = 1, p = 0.2 respectively) and phylogenetic β-diversity (p = 0.3, p = 0.1, p = 1).

Figure 4. Vertical β diversity structure per region. Plots show β taxonomic (a) and phylogenetic diversities (b) over depth (vertical distances in meters). Note that actual distance between Corsica and Provence-Alpes-Côte d'Azur (PACA) regions is not represented for visibility reasons. Eastern (EP) and central-eastern parts of PACA region (CEP) are the most vertically structured regions for all three diversity facets. The lowest levels of β over depth are observed in the western PACA region (WP). Non-significant trends are indicated as *n.s.*





Overall vertical β-diversities ranged between 1.51 and 3.09 for taxonomic facet, from 1.05 to 1.20 for functional facet and between 1.10 and 1.78 for phylogenetic facet, with the highest overall β-diversities being observed in central eastern PACA and the lowest in western PACA, for all three facets of diversity (Fig. 5(a)). Functional β-diversity was lower than phylogenetic β-diversity and even lower than taxonomic β-diversity for both vertical and horizontal scales (Fig. 5(a)). Overall horizontal β-diversity ranged for TD from 2.31 to 2.53, for FD from 1.13 to 1.16 and for PD from 1.39 to 1.63, with higher overall β-diversity being observed in low depth belts (50-70 m) and lower in more shallow depth belts (30-50m), for all three facets of biodiversity (Fig. 5(b)).

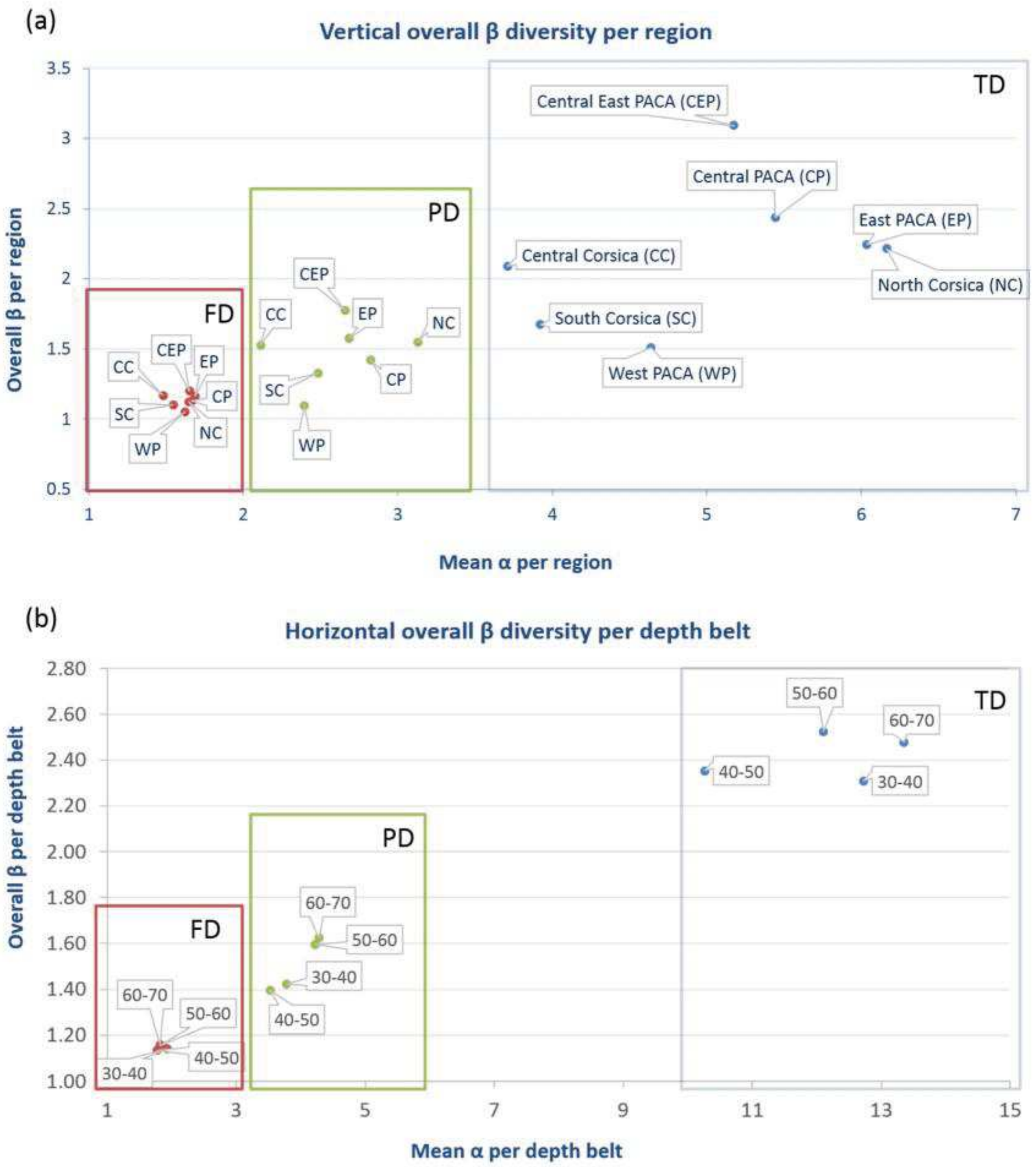


Figure 5. Overall vertical and horizontal β diversities. Dissimilarities per region (a) and per depth belt (b) are plotted over the corresponding mean α diversities.

Pairwise horizontal taxonomic, functional and phylogenetic β-diversity increased with distance for all four depth belts ($p<0.001$; Appendix S5). However, horizontal distance among stations explained a small part of the β-diversity of the three facets i.e. β-TD, β-FD and β-PD, i.e. R^2 of β-diversity over horizontal distances varied from 0.01 to 0.16, when considering the Euclidean distances between stations. Similar results were obtained, when considering sea distances among stations. All trends of β-TD, β-FD and β-PD over horizontal distances, using both approaches to estimate geographic distances can be seen in Appendix S5.

Conservation efficacy

All three facets of diversity are globally under-represented within MPAs which include specific no take zones (Fig. 6). Stations with the highest diversity levels were generally not in protected areas. Separating stations into two depth categories revealed that MPAs seem to be better at protecting stations up to 50 meters deep, but do much worse in deeper stations (from 50 to 70m) (Fig. 6(b) *vs* 6(c)). The overall proportion of protected diversity for all stations was around 13%, rising to 17% for stations up to 50m deep. Conservation efficacy was particularly low for depths over 50 meters, as almost one third of the stations of this depth category – corresponding to the 15-17 richest stations according to the diversity facet considered - are not located inside MPAs, and only 8% of the stations of this depth category were located within protected areas (Fig. 6(c)).

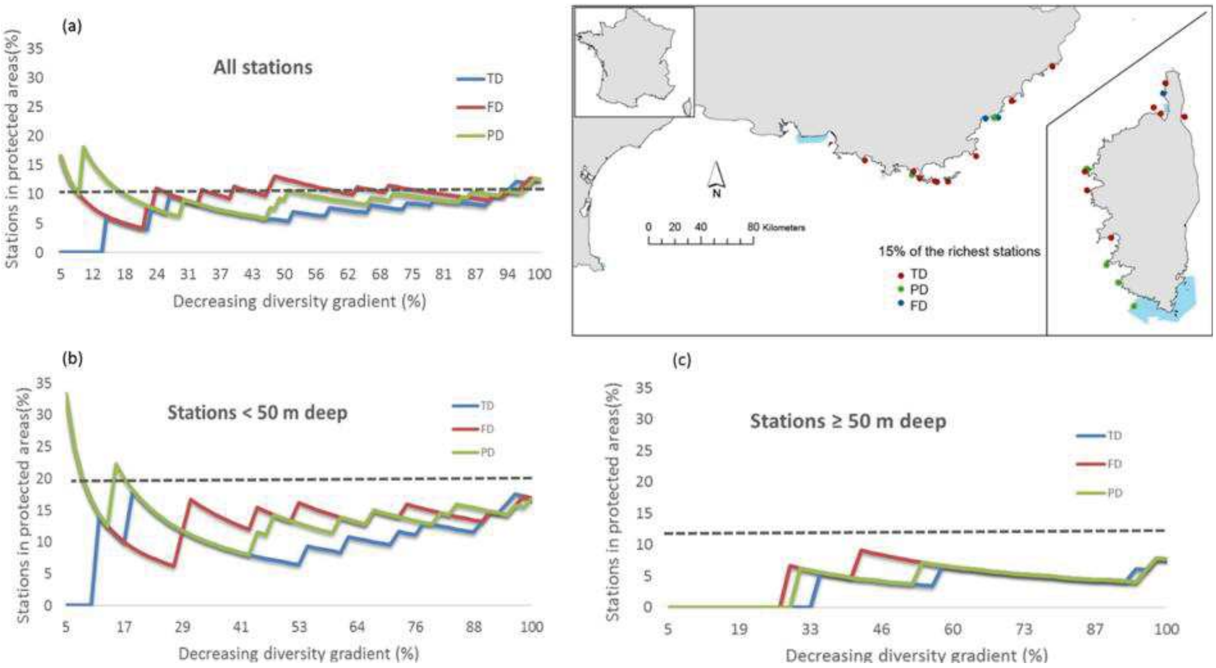


Figure 6. Proportion of diversity facets included in Marine Protected Areas (MPAs) with no take zones. Cumulative proportion of taxonomic, functional and phylogenetic diversities for (a) all stations, (b) for stations up to 50m deep and (c) for over 50m deep stations. Dashed lines indicate the overall proportions of protected stations: (a) 12.6% of all considered stations (N = 111), (b) 16.9% of stations up to 50m deep (N = 59) and (c) 7.7% of stations over 50m deep (N = 52). The map shows the location of the 15% of the richest stations for all three facets and the considered MPAs (indicated by light blue zones).

4.5 Discussion

Multi-facet diversity of coralligenous assemblages

We revealed that functional diversity was generally lower than phylogenetic diversity and even lower than taxonomic diversity, indicating the presence of a few, abundant, functionally similar, species within most stations. Moreover, TD was not significantly congruent to FD but it was significantly less congruent to PD than expected at random. This indicates that TD and PD patterns provide complementary information about coralligenous assemblages. More interestingly, FD and PD were highly congruent to each other, with similar spatial distribution patterns over vertical and horizontal axes. Finally, congruence between functional and phylogenetic diversities increased for stations deeper than 50 meters, where taxonomic diversity was not congruent to either functional or phylogenetic diversities.

The use of phylogenetic diversity as a proxy for functional diversity (Cadotte et al., 2008) has been debated and several recent studies bring evidence of significant lack of congruence between the two facets (Devictor et al., 2010b; Mouillot et al., 2011; Safi et al., 2011). In our case, the high congruence between functional and phylogenetic diversity might be driven by niche conservatism. In old lineages, environmental tolerances are restricted, leading to a phylogenetic clustering (Swenson et al., 2006; Swenson, 2011). On the other hand, if species divergence is more recent, we may expect traits overdispersion (Ndiribe et al., 2013). However, when old and more recent lineages are encountered within communities, as typically occurs in coralligenous assemblages, traits' phylogenetic signal may be masked by lineage-specific differences in trait evolution (Smith & Donoghue, 2008; Ndiribe et al., 2013). Given our results i.e. low phylogenetic and functional diversities and high congruence between them, we assume that functional and phylogenetic clustering occurs in coralligenous assemblages, yet making them even more vulnerable to climate change or human perturbations. Indeed, species sharing similar adaptations and restricted environmental tolerance, are expected to respond similarly to increasing perturbation risks, and thus even small future changes can have detrimental consequences in ecosystem functioning (Mouquet et al., 2012; Srivastava et al., 2012).

However, the exact functional role of coralligenous habitats in the Mediterranean basin has not yet been clarified (Georgiadis et al., 2009). Previous studies indicated that coralligenous species strongly structure their environment by capturing particulate organic matter sedimentation, creating niche and improving the food availability for a number of species, like queen scallops, soft clams, sea urchins, starfish and gadoids (Kamenos et al., 2004; Lloret et al., 2007). Yet, which specific traits relate to the ecosystem functioning are not yet been clearly identified for the coralligenous concretions, making thus the estimation of these ecosystem engineers' functional diversity particularly challenging. Here, we estimated functional diversity based on a set of complementary species characteristics describing their morphology, reproduction and ecology. This set is similar to the one recently used to characterize functional diversity of fossil and present benthic assemblages (Villéger et al., 2011). Future studies on coralligenous ecosystems could permit to better distinguish particular aspects of ecosystem functioning like resistance to invasion or traits linked to the flow of matter and energy within the ecosystem to enable more interpretable estimations of functional diversity.

While incredibly diverse, the coralligenous assemblages remain largely inaccessible to systematic sampling. We preferred to use a non-invasive technique, based on photos, which is recommended for underwater ecological systems (UNEP-MAP-RAC/SPA, 2011) and commonly used (Balata et al., 2005; Baldacconi & Corriero, 2009; Kipson et al., 2011; Deter et al., 2012a, 2012b). However, this approach does not allow identifying all individuals at the species level (particularly for cryptic species). In our analysis, we adopted a conservative approach by keeping identifications at the genus level or superior groups, instead of risking to over-estimate richness in cases of impossible identification at the species level. The number of species is thus underestimated in all stations for both alpha and beta diversity components but this bias is common to all stations, allowing thus comparisons among stations, which is the main aim of the present work.

In the absence of adequate molecular information for coralligenous species, we used cladistics to calculate phylogenetic diversity. Other methods, based on dated phylogenies, have also been developed and used to estimate phylogenetic diversity when studying community structure and ecosystem stability (Cadotte et al., 2010, 2012; Flynn et al., 2011). Molecular approaches are more accurate and should be preferred when needing to estimate phylogenetic distances (Pavoine & Bonsall, 2011), but demand that every single species should be adequately sequenced, which is until now infeasible for coralligenous communities. Recent evidence showed that plants phylogenetic diversity estimated by undated taxonomic hierarchies, such as the one used here, is highly related to the diversity obtained by dated phylogenies (Ricotta et al., 2012). This should be further verified in the future for other organisms. Given the ecological importance of coralligenous habitats and despite all the above limitations related to PD and FD estimations, our study provides the first multi-facet distribution analysis of coralligenous assemblages worldwide. Further research on molecular sequencing and studies on particular functions of coralligenous species can help refine the phylogenetic and functional diversity of this rich ecosystem.

Vertical and horizontal structure of coralligenous assemblages

At local scale (α- diversity), depth was the main axis structuring functional and phylogenetic diversities, with shallow stations being on average richer than deep stations. No significant variation was observed over depth for taxonomic diversity. A slight horizontal structure was observed for all facets of diversity, with richer communities being located in the north-eastern part of our study area i.e. eastern PACA and north Corsica. In marine ecosystems, species' functional roles and evolutionary history are driven by spatially distributed environmental conditions, such as light and nutrient availability (Thingstad et al., 2005; Elser et al., 2007). Typically, communities sharing high water clarity and light availability conditions will be characterized, in shallower waters, by higher covers of erect Udoteaceae (Flabellia petiolata) and Mesophyllum alternans (Balata et al., 2006; Ballesteros, 2006; Piazzi & Balata, 2011). These taxa are replaced in deeper waters by encrusting and laminar Rhodophyta (Ballesteros, 2006), or Porifera at greater depths (Deter et al., 2012a). Such changes in assemblage composition, resulting from the replacement of one species (or group of species) by another, may often not be perceived in terms of taxonomic diversity (Devictor et al., 2010a). Functional and phylogenetic information on community composition better captures this turnover, and is therefore key to establishing the spatial variations of biological diversity (Mouquet et al., 2012).

Taxonomic, functional and phylogenetic dissimilarities among assemblages (β-diversity) increased with depth mainly in the central and eastern Provence-Alpes-Côte d'Azur (PACA) region and in north Corsica, where the vertical gradient was high. We observed no vertical structure in other regions, in south Corsica or west PACA for example. Strong vertical structure among coralligenous assemblages may signify that spatial heterogeneity is pronounced along the water column. The absence of verticality on the other hand may signify environmentally unfavourable habitats for coralligenous concretions, where for example turbidity and human-induced disturbances contribute to vertically homogeneous environments, thus leading to biotic homogenization. In cases where water turbidity and sedimentation levels are high, the shift between shallow, more light-requiring to deep sciaphilic macroalgal communities may occur to a shallower than usual depth (Balata et al., 2006). Moreover, in human impacted zones, species that are sensitive to disturbance may decrease, while more generalists species become more dominant (Balata et al., 2007a, 2007b). These effects can result in low dissimilarities among communities. The western part of PACA region may be such an example, where the β diversity of all three facets remained very low and invariable over different depths.

We observed little evidence of the coralligenous structure over the horizontal axis as horizontal distances among stations explained a small part of the between them dissimilarities in all three β-diversity facets i.e. TD, FD and PD. Comparatively higher horizontal β-diversity was observed for intermediate and deep stations, i.e. 50-70 meters deep. This depth range represents the vast majority of the analysed stations, regrouping areas from the PACA and the Corsica regions. Water's characteristics and transparency are expected to vary among regions especially between the west and central PACA and the Corsica stations. Terrestrial inputs in deltaic areas, such as the Rhone's river, should contribute to regions differences. Rhone's delta is an important source of particulate organic matter sedimentation with particular peaks at intermediate depths (30-50m) and least contribution at 70–100 m depth (Darnaude et al., 2004). Such sedimentation inputs have an impact on marine macrobenthos activity (Darnaude et al., 2004) which may be reflected in several of the PACA stations coralligenous composition.

In our approach, we used geographic and depth distances as a proxy for the environmental conditions that vary over space. However, as previously mentioned, water's physico-chemical characteristics and among species interactions can modify vertical distribution of the coralligenous species resulting to different community compositions at similar depths (Airoldi, 2003; Dauer et al., 2008; Piazzi & Balata, 2011). Thus a more specific analysis of spatial patterns in β-diversity related to key environmental and human pressure variables, such as temperature, turbidity and pollution, and taking into account various vertical and horizontal distances, should shed light on the mechanisms that drive diversity patterns in coralligenous habitats. This should result in more effective conservation measures to preserve biodiversity at both local and regional scales. Nevertheless, and regardless the limitations of using depth as a proxy, our analysis provided evidence of coralligenous α- and β-diversity vertical structure, which should be further considered for setting future conservation goals.

Conservation implications

New options for marine conservation may result from multifaceted three-dimensional analyses. Vertical and horizontal variability can provide useful input for conservation planning, indicating whether the most effective approach to conservation is local and/or regional (Baselga, 2010). If conservation effort focuses on preserving local diversities, restrictions should be imposed with increasing intensity from shallow to deeper water levels. However, this local approach could fail to protect species characteristic of deeper habitats (such as sponge species for example). Contrary to the more classical approach of marine reserves that focus on protecting specific sites vertically, horizontal β-diversity could be favoured by implementing conservation measures that take into account depth range. In some areas, more effective preservation of biotic dissimilarities could be achieved by regulating human activities at certain depths, thus adding a horizontal type of reserves to the existing vertical ones.

Both conservation efforts and disturbance coming from human activities are depth dependent. Indeed, while reviewing MPAs development over the last 30 years in the Mediterranean France, one realises that while conservation planning initially focused on rather shallow waters habitats, more recent efforts are contributing to the protection of deeper than 50 m ecosystems (Meinesz & Blanfuné, 2015). However, the distribution of the protected areas by depth range remains partial and inappropriate (Meinesz & Blanfuné, 2015) and seems insufficient to protect coralligenous habitats. Additional to the MPAs initiatives, Natura 2000 sites and European Union regulations, specifically include coralligenous formations among the list of habitats to be protected. However, human activities and in particular commercial fishing interests are in conflict with conservation goals. Since coralligenous communities compose the habitat of several species of fish, mechanistic destruction due to fishing remains as one of the main threats for coralligenous habitats in the Mediterranean region (Georgiadis et al., 2009). The efficacy of future conservation measures will thus largely depend on the selection of regions and depth belts to preserve.

Zones deeper than 50 meters may continue to be less impacted by human disturbances than shallow waters, deeper coralligenous concretions can act as refuge areas at least for some species. It would be interesting to test whether the deep reef refuge hypothesis (Bongaerts et al., 2011; Serrano et al., 2014) - according to which shallow communities can profit from deeper ones following disturbance, by using them as local recruitment sources - can be applied to the coralligenous habitats. Future research on larvae vertical and horizontal migration would thus be needed to clarify such aspects. This will additionally shed light on connectivity issues that might explain the observed diversity patterns of coralligenous concretions and become critical to future connectivity of marine protected areas (Andrello et al., 2015).

Given the important ecological role of coralligenous habitats, providing habitat and filter functions for several microorganisms and fish species, and their low resilience to disturbance (Georgiadis et al., 2009). Mediterranean marine conservation needs to be designed and implemented based on spatially adaptive methods able to monitor the three-dimensional spatial variability of diversity. We suggest that using taxonomic diversity alone may be misleading, and that additional information, at least on phylogenetic diversity, is required to properly set future conservation goals and enable conservationists to capture the spatial variations of communities composition (Mouquet et al., 2012). There should, moreover, be a particular focus on ecosystem verticality, which may help preserve dissimilarities among coralligenous concretions. Local conservation measures should be embedded in a larger scale strategy able to identify areas and depth belts that hold higher diversity levels. Although three-dimensional analyses are complex and constitute a challenge for conservationists and biogeographers, they should be further considered in order to encapsulate the different facets of biotic diversity and propose adequate ways of preserving biodiversity at different scales.

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4.7 Supporting information

Additional Supporting Information:

Appendix S1 Stations’ distribution over depth.

Appendix S2 List of traits used to calculate coralligenous functional diversity.

Appendix S3 Principal Coordinate Analysis for functional distances.

Appendix S4 Cladistics of species considered in the study.

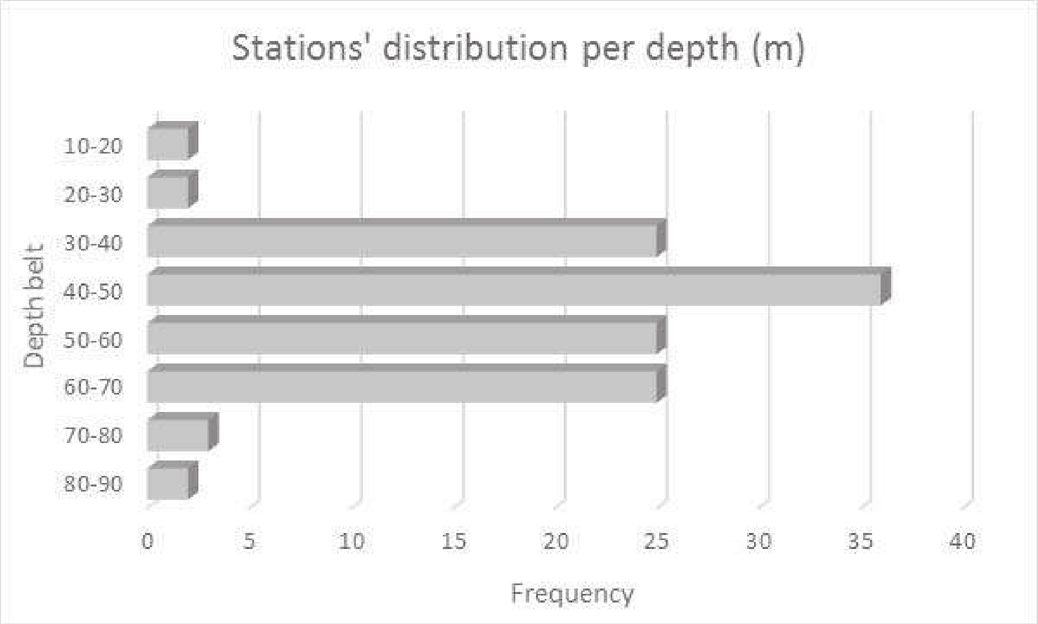
Appendix S5 Horizontal distances among stations over horizontal β-diversity.

Appendix S6 Relationships between α taxonomic, functional, phylogenetic, net functional and net phylogenetic diversity anomalies.

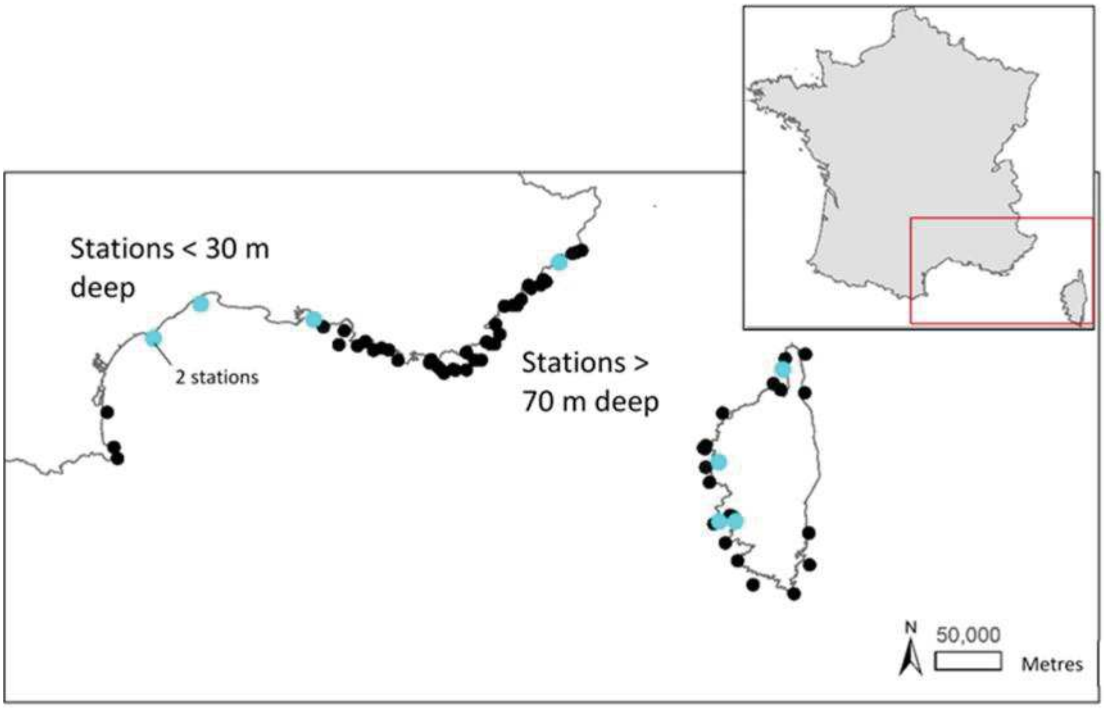
Appendix S7 Vertical β functional diversity structure per region.

Supporting Information

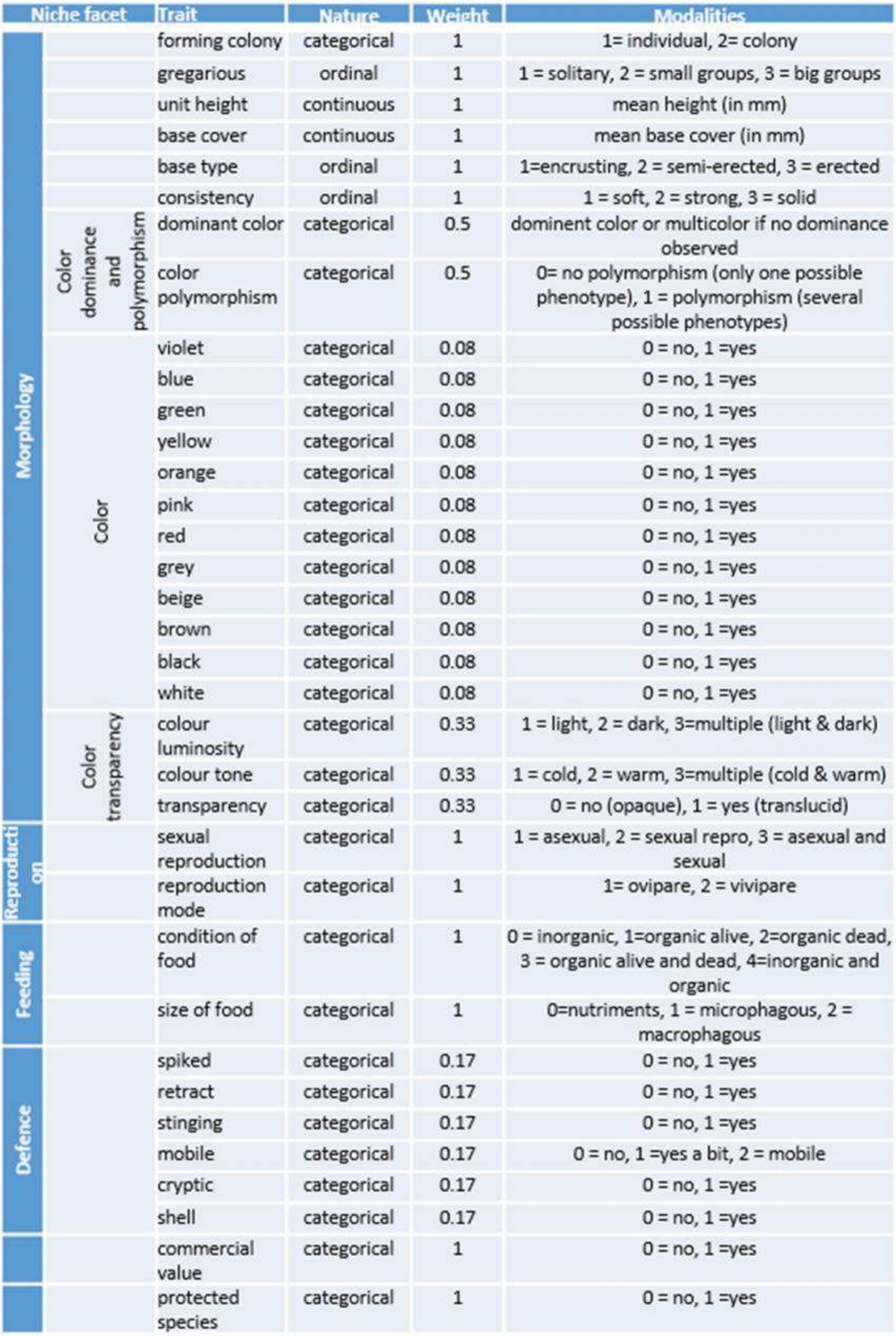
Appendix S1. Distribution of the 120 stations over depth.



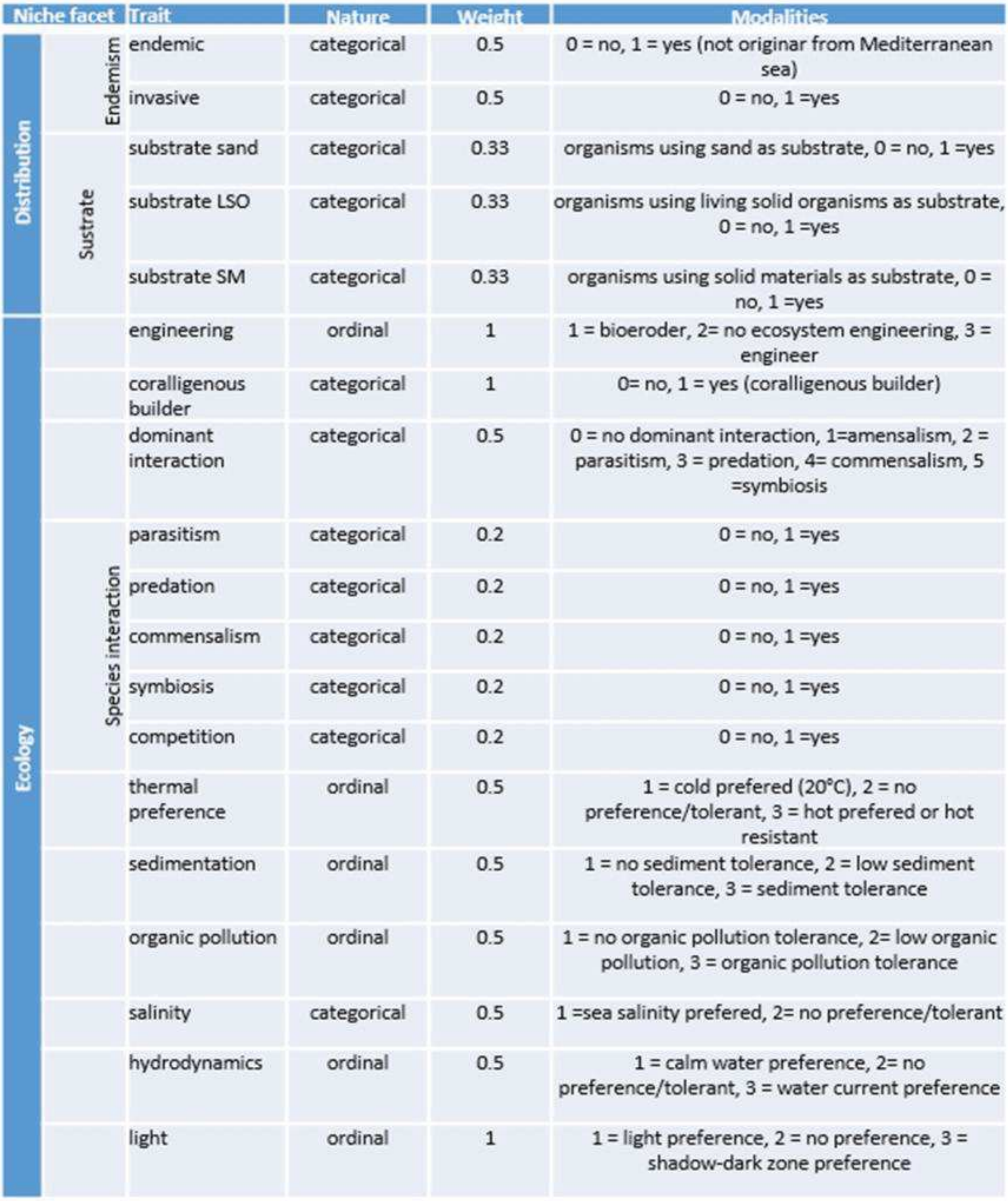
We excluded nine stations that were situated either at very shallow or very deep levels (i.e. four stations at 17-24 m and five stations at 75-90 m). They are shown on the map by light blue dots. All four shallow stations were situated in the Languedoc-Roussillon region and the five stations deeper than 70m were situated in Corsica and eastern PACA.



Appendix S2. List of traits used to calculate coralligenous functional diversity. The modalities of the traits, the nature of the data and the weights used for the dissimilarity matrix calculation are also given.



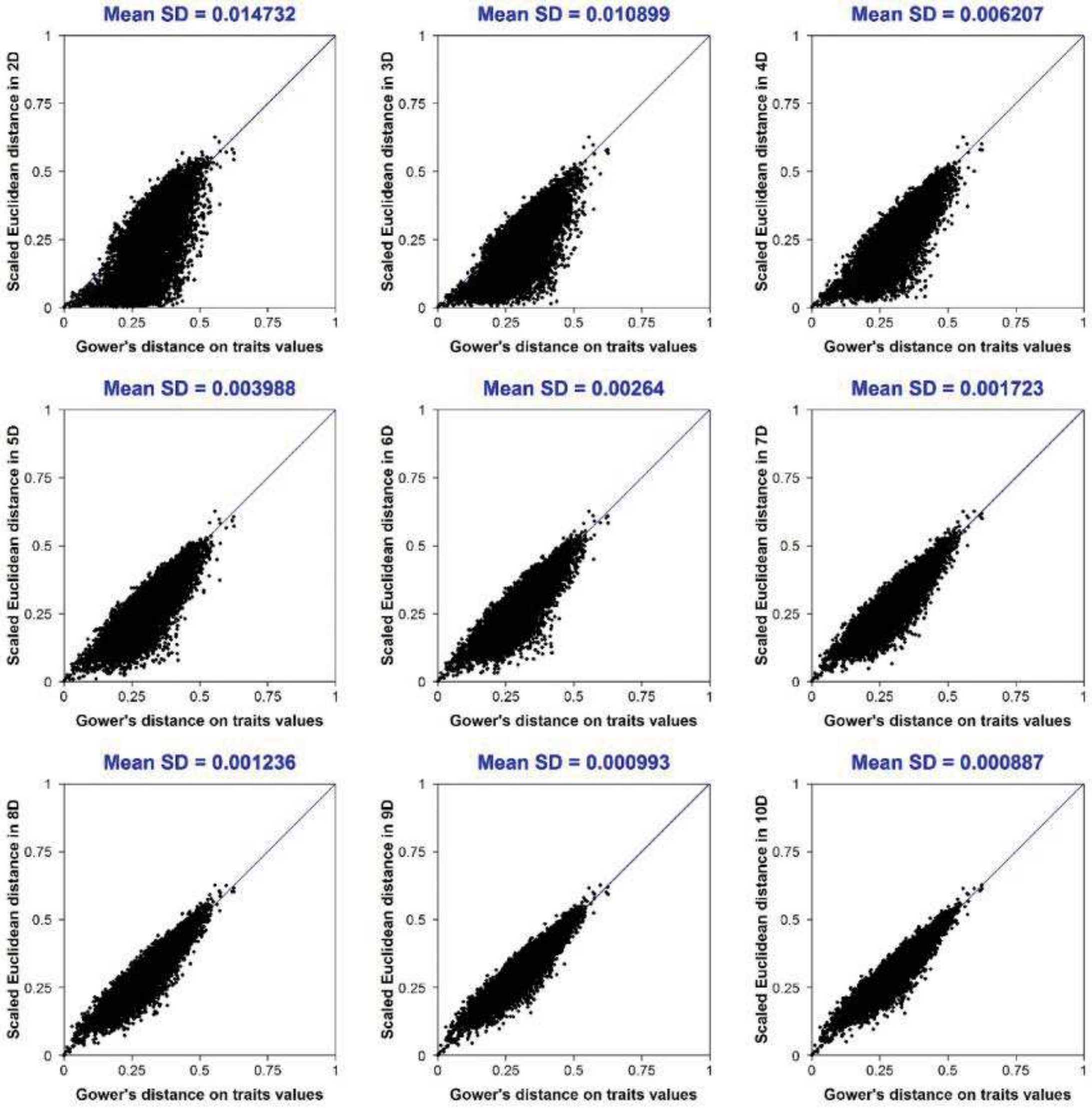
Niche facet		Trait	Nature	Weight	Modalities
Morphology		forming colony	categorical	1	1= individual, 2= colony
		gregarious	ordinal	1	1 = solitary, 2 = small groups, 3 = big groups
		unit height	continuous	1	mean height (in mm)
		base cover	continuous	1	mean base cover (in mm)
		base type	ordinal	1	1=encrusting, 2 = semi-erected, 3 = erected
		consistency	ordinal	1	1 = soft, 2 = strong, 3 = solid
	Color dominance and polymorphism	dominant color	categorical	0.5	dominent color or multicolor if no dominance observed
		color polymorphism	categorical	0.5	0= no polymorphism (only one possible phenotype), 1 = polymorphism (several possible phenotypes)
	Color	violet	categorical	0.08	0 = no, 1 =yes
		blue	categorical	0.08	0 = no, 1 =yes
		green	categorical	0.08	0 = no, 1 =yes
		yellow	categorical	0.08	0 = no, 1 =yes
		orange	categorical	0.08	0 = no, 1 =yes
		pink	categorical	0.08	0 = no, 1 =yes
		red	categorical	0.08	0 = no, 1 =yes
		grey	categorical	0.08	0 = no, 1 =yes
		beige	categorical	0.08	0 = no, 1 =yes
		brown	categorical	0.08	0 = no, 1 =yes
		black	categorical	0.08	0 = no, 1 =yes
		white	categorical	0.08	0 = no, 1 =yes
	Color transparency	colour luminosity	categorical	0.33	1 = light, 2 = dark, 3=multiple (light & dark)
		colour tone	categorical	0.33	1 = cold, 2 = warm, 3=multiple (cold & warm)
		transparency	categorical	0.33	0 = no (opaque), 1 = yes (translucid)
Reproduction		sexual reproduction	categorical	1	1 = asexual, 2 = sexual repro, 3 = asexual and sexual
		reproduction mode	categorical	1	1= ovipare, 2 = vivipare
Feeding		condition of food	categorical	1	0 = inorganic, 1=organic alive, 2=organic dead, 3 = organic alive and dead, 4=inorganic and organic
		size of food	categorical	1	0=nutriments, 1 = microphagous, 2 = macrophagous
Defence		spiked	categorical	0.17	0 = no, 1 =yes
		retract	categorical	0.17	0 = no, 1 =yes
		stinging	categorical	0.17	0 = no, 1 =yes
		mobile	categorical	0.17	0 = no, 1 =yes a bit, 2 = mobile
		cryptic	categorical	0.17	0 = no, 1 =yes
		shell	categorical	0.17	0 = no, 1 =yes
		commercial value	categorical	1	0 = no, 1 =yes
		protected species	categorical	1	0 = no, 1 =yes



Niche facet		Trait	Nature	Weight	Modalities
Distribution	Endemism	endemic	categorical	0.5	0 = no, 1 = yes (not originar from Mediterranean sea)
		invasive	categorical	0.5	0 = no, 1 =yes
	Sustrate	substrate sand	categorical	0.33	organisms using sand as substrate, 0 = no, 1 =yes
		substrate LSO	categorical	0.33	organisms using living solid organisms as substrate, 0 = no, 1 =yes
		substrate SM	categorical	0.33	organisms using solid materials as substrate, 0 = no, 1 =yes
Ecology		engineering	ordinal	1	1 = bioeroder, 2= no ecosystem engineering, 3 = engineer
		coralligenous builder	categorical	1	0= no, 1 = yes (coralligenous builder)
		dominant interaction	categorical	0.5	0 = no dominant interaction, 1=amensalism, 2 = parasitism, 3 = predation, 4= commensalism, 5 =symbiosis
	Species interaction	parasitism	categorical	0.2	0 = no, 1 =yes
		predation	categorical	0.2	0 = no, 1 =yes
		commensalism	categorical	0.2	0 = no, 1 =yes
		symbiosis	categorical	0.2	0 = no, 1 =yes
		competition	categorical	0.2	0 = no, 1 =yes
		thermal preference	ordinal	0.5	1 = cold prefered (20°C), 2 = no preference/tolerant, 3 = hot prefered or hot resistant
		sedimentation	ordinal	0.5	1 = no sediment tolerance, 2 = low sediment tolerance, 3 = sediment tolerance
		organic pollution	ordinal	0.5	1 = no organic pollution tolerance, 2= low organic pollution, 3 = organic pollution tolerance
		salinity	categorical	0.5	1 =sea salinity prefered, 2= no preference/tolerant
		hydrodynamics	ordinal	0.5	1 = calm water preference, 2= no preference/tolerant, 3 = water current preference
		light	ordinal	1	1 = light preference, 2 = no preference, 3 = shadow-dark zone preference

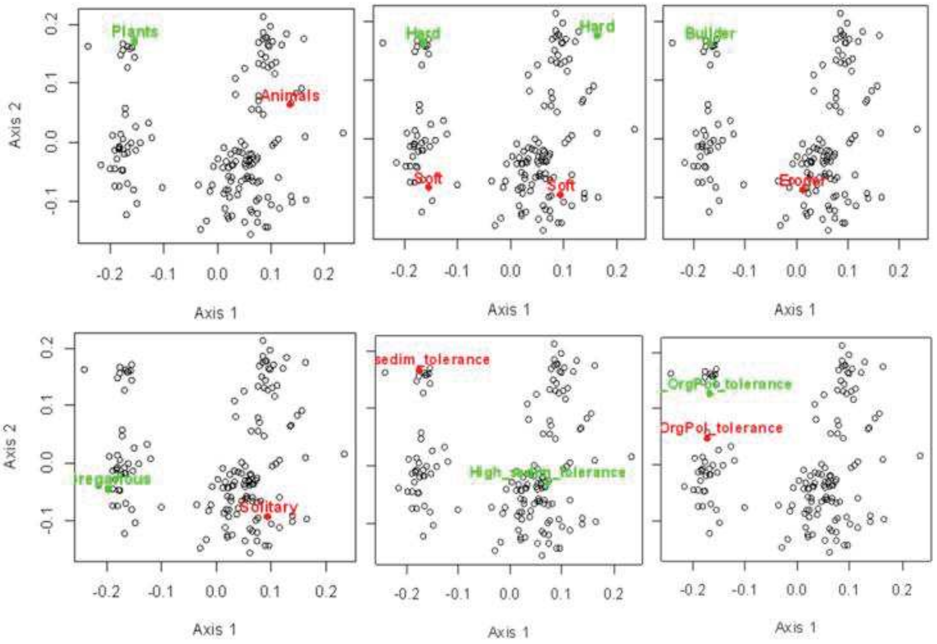
Appendix S3. Principal Coordinate Analysis for functional distances.

The quality of 9 functional spaces (from 2 to 10 dimensions) computed for the coralligenous taxa is illustrated as the correlation between pairwise distance computed on taxa traits (Gower's distance) and standardized Euclidean distance in each of the multidimensional space obtained with a Principal Coordinates Analysis. The mean squared deviation (Mean SD) between these two functional distances is provided at the top of each panel, the lower it is the higher the quality of the functional space.



For 8 dimensions considered, the Mean SD was considerably low (~ 0.001) and adding further dimensions did not considerably improved this quality i.e. difference of 0.0002-0.0003 in Mean SD (see panels for 9D and 10D). To respect parsimony, we thus retained eight axes to estimate functional distances between species.

Typical examples of some of the traits are given in green and red for axes 1&2. No specific axis interpretation should be applied, however, as they represent the combined information from all traits used.



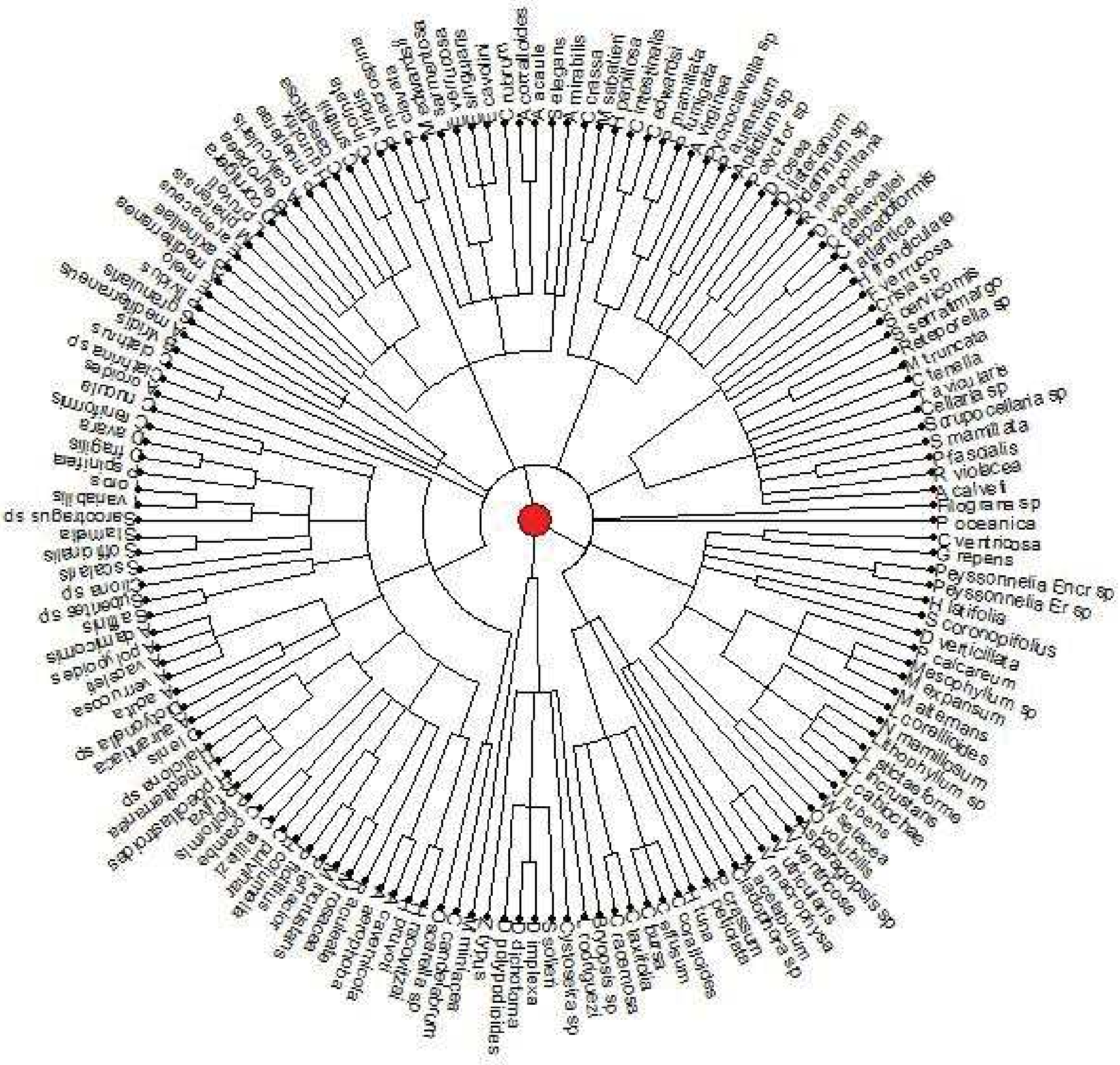
Appendix S4. Cladistics of species considered in the study. Most dominant species are indicated in bold. We considered as dominant species, those that had more than 1000 occurrences in the analysed database, which corresponds approximately to one third of the photos taken in all retained stations (i.e. 10 photos x 111 stations).

Species	Genus	Family	Order	Class	Phylum	Kingdom
Acanthella acuta	Acanthella	Dictyonellidae	Halichondrida	Demospongiae	Porifera	Animalia
Acetabularia acetabulum	Acetabularia	Polyphysaceae	Dasycladales	Ulvophyceae	Chlorophyta	Plantae
Adeonella calveti	Adeonella	Adeonellidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Agelas oroides	Agelas	Agelasidae	Agelasida	Demospongiae	Porifera	Animalia
Alcyonium acaule	Alcyonium	Alcyoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Alcyonium corralloides	Alcyonium	Alcyoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Alicia mirabilis	Alicia	Aliciidae	Actinaria	Anthozoa	Cnidaria	Animalia
Antedon mediterranea	Antedon	Antedonidae	Comatulida	Crinoidea	Echinodermata	Animalia
Aplidium sp	Aplidium	Polyclinidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Aplysina aerophoba	Aplysina	Aplysinidae	Verongida	Demospongiae	Porifera	Animalia
Aplysina cavernicola	Aplysina	Aplysinidae	Verongida	Demospongiae	Porifera	Animalia
Ascidia virginea	Ascidia	Ascidiidae	Phlebobranchia	Ascidiacea	Chordata	Animalia
Asparagopsis sp	Asparagopsis	Bonnemaisoniaceae	Bonnemaisoniales	Florideophyceae	Rhodophyta	Plantae
Astroides calycularis	Astroides	Dendrophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Astropartus mediterraneus	Astrospartus	Gorgonocephalidae	Euryalida	Ophiuroidea	Echinodermata	Animalia
Axinella damicornis	Axinella	Axinellidae	Halichondrida	Demospongiae	Porifera	Animalia
Axinella polypoides	Axinella	Axinellidae	Halichondrida	Demospongiae	Porifera	Animalia
Axinella vaceleti	Axinella	Axinellidae	Halichondrida	Demospongiae	Porifera	Animalia
Axinella verrucosa	Axinella	Axinellidae	Halichondrida	Demospongiae	Porifera	Animalia
Axinyssa aurantiaca	Axinyssa	Halichondriidae	Halichondrida	Demospongiae	Porifera	Animalia
Balanophyllia europaea	Balanophyllia	Dendrophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Bonellia viridis	Bonellia	Bonelliidae	Bonelliida	Echiuroidea	Echiura	Animalia
Bryopsis sp	Bryopsis	Bryopsidaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Caryophyllia inornata	Caryophyllia	Caryophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Caryophyllia smithii	Caryophyllia	Caryophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Caulerpa racemosa	Caulerpa	Caulerpaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Caulerpa taxifolia	Caulerpa	Caulerpaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Cellaria sp	Cellaria	Cellariidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Chartella tenella	Chartella	Flustridae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Chondrilla nucula	Chondrilla	Chondrillidae	Chondrosida	Demospongiae	Porifera	Animalia
Chondrosia reniformis	Chondrosia	Chondrillidae	Chondrosida	Demospongiae	Porifera	Animalia
Chrysymenia ventricosa	Chrysymenia	Rhodymeniaceae	Rhodymeniales	Florideophyceae	Rhodophyta	Plantae
Ciona edwardsi	Ciona	Cionidae	Phlebobranchia	Ascidiacea	Chordata	Animalia
Ciona intestinalis	Ciona	Cionidae	Phlebobranchia	Ascidiacea	Chordata	Animalia
Cladocora caespitosa	Cladocora	Caryophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Cladophora sp	Cladophora	Cladophoraceae	Cladophorales	Ulvophyceae	Chlorophyta	Plantae
Clathrina clathrus	Clathrina	Clathrinidae	Clathrinida	Calcarea	Porifera	Animalia
Clathrina sp	Clathrina	Clathrinidae	Clathrinida	Calcarea	Porifera	Animalia
Clavelina lepadoformis	Clavelina	Clavelinidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Clavelina dellavallei	Clavelina	Clavelinidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Cliona sp	Cliona	Clionaidae	Hadromerida	Demospongiae	Porifera	Animalia
Codium bursa	Codium	Codiaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Codium effusum	Codium	Codiaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Codium coralloides	Codium	Codiaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Corallium rubrum	Corallium	Coralliidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Corticium candelabrum	Corticium	Plakinidae	Homosclerophorida	Homoscleromorpha	Porifera	Animalia
Corynactis viridis	Corynactis	Corallimorphidae	Corallimorpharia	Anthozoa	Cnidaria	Animalia
Crambe crambe	Crambe	Crambeidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Crambe tailliezi	Crambe	Crambeidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Crella pulvinar	Crella	Crellidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Cribrinopsis crassa	Cribrinopsis	Actiniidae	Actinaria	Anthozoa	Cnidaria	Animalia
Crisia sp	Crisia	Crisiidae	Cyclostomatida	Stenolaemata	Bryozoa	Animalia
Cystoseira sp	Cystoseira	Sargassaceae	Fucales	Phaeophyceae	Ochrophyta	Chromista
Dendrophyllia cornigera	Dendrophyllia	Dendrophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia

Species	Genus	Family	Order	Class	Phylum	Kingdom
Dendroxea lenis	Dendroxea	Chalinidae	Haplosclerida	Demospongiae	Porifera	Animalia
Diazona violacea	Diazona	Diazonidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Dictyonella sp	Dictyonella	Dictyonellidae	Halichondrida	Demospongiae	Porifera	Animalia
Dictyopteris polypodioides	Dictyopteris	Dictyotaceae	Dictyotales	Phaeophyceae	Ochrophyta	Chromista
Dictyota dichotoma	Dictyota	Dictyotaceae	Dictyotales	Phaeophyceae	Ochrophyta	Chromista
Dictyota implexa	Dictyota	Dictyotaceae	Dictyotales	Phaeophyceae	Ochrophyta	Chromista
Didemnum sp	Didemnum	Didemnidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Diplosoma listerianum	Diplosoma	Didemnidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Distaplia rosea	Distaplia	Holozoidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Dudresnaya verticillata	Dudresnaya	Dumontiaceae	Gigartinales	Florideophyceae	Rhodophyta	Plantae
Dysidea avara	Dysidea	Dysideidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Dysidea fragilis	Dysidea	Dysideidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Echinus melo	Echinus	Echinidae	Camarodonta	Echinoidea	Echinodermata	Animalia
Epizoanthus arenaceus	Epizoanthus	Epizoanthidae	Zoanthidea	Anthozoa	Cnidaria	Animalia
Eunicella cavolini	Eunicella	Gorgoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Eunicella singularis	Eunicella	Gorgoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Eunicella verrucosa	Eunicella	Gorgoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Filograna sp	Filograna	Serpulidae	Sabellida	Polychaeta	Annelida	Animalia
Flabellia petiolata	Flabellia	Udoteaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Frondipora verrucosa	Frondipora	Frondiporidae	Cyclostomatida	Stenolaemata	Bryozoa	Animalia
Gloiocladia repens	Gloiocladia	Faucheaceae	Rhodymeniales	Florideophyceae	Rhodophyta	Plantae
Haliclona sp	Haliclona	Chalinidae	Haplosclerida	Demospongiae	Porifera	Animalia
Haliclona mediterranea	Haliclona	Chalinidae	Haplosclerida	Demospongiae	Porifera	Animalia
Haliclona poecillastroides	Haliclona	Chalinidae	Haplosclerida	Demospongiae	Porifera	Animalia
Reniera fulva	Haliclona	Chalinidae	Haplosclerida	Demospongiae	Porifera	Animalia
Halimeda tuna	Halimeda	Halimedaceae	Bryopsidales	Bryopsidophyceae	Chlorophyta	Plantae
Halocynthia papillosa	Halocynthia	Pyuridae	Stolidobranchia	Ascidiacea	Chordata	Animalia
Halymenia latifolia	Halymenia	Halymeniaceae	Halymeniales	Florideophyceae	Rhodophyta	Plantae
Hemimycale columella	Hemimycale	Hymedesmiidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Hexadella pruvoti	Hexadella	Ianthellidae	Verongida	Demospongiae	Porifera	Animalia
Hexadella racovitzai	Hexadella	Ianthellidae	Verongida	Demospongiae	Porifera	Animalia
Hoplangia durotrix	Hoplangia	Caryophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Hornera frondiculata	Hornera	Horneridae	Cyclostomatida	Stenolaemata	Bryozoa	Animalia
Idmidronea atlantica	Idmidronea	Tubuliporidae	Cyclostomatida	Stenolaemata	Bryozoa	Animalia
Ircinia oros	Ircinia	Irciniidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Ircinia variabilis	Ircinia	Irciniidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Jania rubens	Jania	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Laminaria rodriguezi	Laminaria	Laminariaceae	Laminariales	Phaeophyceae	Ochrophyta	Chromista
Leptogorgia sarmentosa	Leptogorgia	Gorgoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Leptopsammia pruvoti	Leptopsammia	Dendrophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Lithophyllum cabiochae	Lithophyllum	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Lithophyllum incrustans	Lithophyllum	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Lithophyllum stictaeforme	Lithophyllum	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Lithophyllum sp	Lithophyllum	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Lithothamnion corallioides	Lithothamnion	Hapalidiaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Maasella edwardsii	Maasella	Paralcyoniidae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Madracis pharensis	Madracis	Pocilloporidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Mesophyllum alternans	Mesophyllum	Hapalidiaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Mesophyllum expansum	Mesophyllum	Hapalidiaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Mesophyllum sp	Mesophyllum	Hapalidiaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Microcosmus sabatieri	Microcosmus	Pyuridae	Stolidobranchia	Ascidiacea	Chordata	Animalia
Miniacina miniacea	Miniacina	Homotrematidae	Rotaliida	Polythalamea	Foraminifera	Chromista
Myriapora truncata	Myriapora	Myriaporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Myxilla incrustans	Myxilla	Myxillidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Myxilla rosacea	Myxilla	Myxillidae	Poecilosclerida	Demospongiae	Porifera	Animalia

Species	Genus	Family	Order	Class	Phylum	Kingdom
Neogoniolithon mamillosum	Neogoniolithon	Corallinaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Oscarella sp	Oscarella	Oscarellidae	Homosclerophorida	Homoscleromorpha	Porifera	Animalia
Osmundaria volubilis	Osmundaria	Rhodomelaceae	Ceramiales	Florideophyceae	Rhodophyta	Plantae
Palmophyllum crassum	Palmophyllum	Palmophyllaceae	Palmophyllales	Chlorophyta_incertae_sedis	Chlorophyta	Plantae
Paracentrotus lividus	Paracentrotus	Parechinidae	Camarodonta	Echinoidea	Echinodermata	Animalia
Paramuricea clavata	Paramuricea	Plexauridae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Paramuricea macrospina	Paramuricea	Plexauridae	Alcyonacea	Anthozoa	Cnidaria	Animalia
Parazoanthus axinellae	Parazoanthus	Parazoanthidae	Zoanthidea	Anthozoa	Cnidaria	Animalia
Pentapora fascialis	Pentapora	Bitectiporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Petrosia ficiformis	Petrosia	Petrosiidae	Haplosclerida	Demospongiae	Porifera	Animalia
Peyssonnelia sp	Peyssonnelia	Peyssonneliaceae	Peyssonneliales	Florideophyceae	Rhodophyta	Plantae
Phallusia fumigata	Phallusia	Ascidiidae	Phlebobranchia	Ascidiacea	Chordata	Animalia
Phallusia mamillata	Phallusia	Ascidiidae	Phlebobranchia	Ascidiacea	Chordata	Animalia
Phorbas fictitius	Phorbas	Hymedesmiidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Phorbas tenacior	Phorbas	Hymedesmiidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Phymatolithon calcareum	Phymatolithon	Hapalidiaceae	Corallinales	Florideophyceae	Rhodophyta	Plantae
Pleraplysilla spinifera	Pleraplysilla	Dysideidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Polycitor sp	Polycitor	Polycitoridae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Polyclinum aurantium	Polyclinum	Polyclinidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Polycyathus muellerae	Polycyathus	Caryophylliidae	Scleractinia	Anthozoa	Cnidaria	Animalia
Posidonia oceanica	Posidonia	Posidoniaceae	Alismatales	Monocots	Tracheophyta	Plantae
Pycnoclavella sp	Pycnoclavella	Pycnoclavellidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Raspaciona aculeate	Raspaciona	Raspailiidae	Poecilosclerida	Demospongiae	Porifera	Animalia
Reptadeonella violacea	Reptadeonella	Adeonidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Reteporella sp	Reteporella	Phidoloporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Rhopalaea neapolitana	Rhopalaea	Diazonidae	Aplousobranchia	Ascidiacea	Chordata	Animalia
Sagartia elegans	Sagartia	Sagartiidae	Actinaria	Anthozoa	Cnidaria	Animalia
Sarcotragus sp	Sarcotragus	Irciniidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Scalarispongia scalaris	Scalarispongia	Thorectidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Schizomavella mamillata	Schizomavella	Bitectiporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Schizotheca serratimargo	Schizotheca	Phidoloporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Scrupocellaria sp	Scrupocellaria	Candidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Smittina cervicornis	Smittina	Smittinidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Spatoglossum solieri	Spatoglossum	Dictyotaceae	Dictyotales	Phaeophyceae	Ochrophyta	Chromista
Sphaerechinus granularis	Sphaerechinus	Toxopneustidae	Camarodonta	Echinoidea	Echinodermata	Animalia
Sphaerococcus coronopifolius	Sphaerococcus	Sphaerococcaceae	Gigartinales	Florideophyceae	Rhodophyta	Plantae
Spongia lamella	Spongia	Spongiidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Spongia officinalis	Spongia	Spongiidae	Dictyoceratida	Demospongiae	Porifera	Animalia
Suberites sp	Suberites	Suberitidae	Hadromerida	Demospongiae	Porifera	Animalia
Suberites affinis	Suberites	Suberitidae	Hadromerida	Demospongiae	Porifera	Animalia
Turbicellepora avicularis	Turbicellepora	Celleporidae	Cheilostomatida	Gymnolaemata	Bryozoa	Animalia
Valonia macrophysa	Valonia	Valoniaceae	Siphonocladales	Ulvophyceae	Chlorophyta	Plantae
Valonia utricularis	Valonia	Valoniaceae	Siphonocladales	Ulvophyceae	Chlorophyta	Plantae
Valonia ventricosa	Valonia	Valoniaceae	Siphonocladales	Ulvophyceae	Chlorophyta	Plantae
Womersleyella setacea	Womersleyella	Rhodomelaceae	Ceramiales	Florideophyceae	Rhodophyta	Plantae
Zanardinia typus	Zanardinia	Cutleriaceae	Cutleriales	Phaeophyceae	Ochrophyta	Chromista

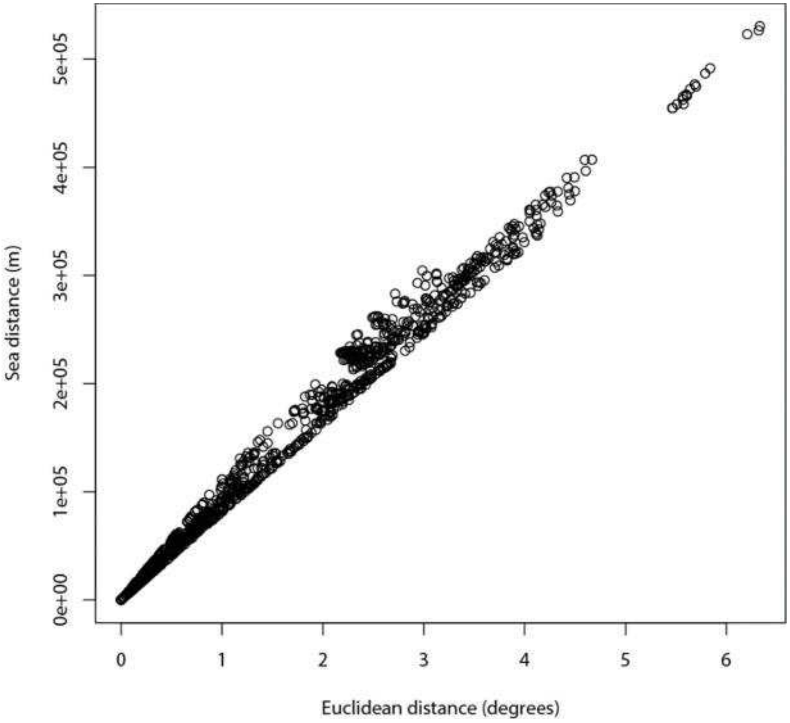
The phylogenetic tree is presented below. Note that this is not an ultrametric tree, as no sequence data were available for the studied ecosystem. We thus use the number of nodes as a proxy of distances between all pairs of species found within studied communities.



Appendix S5. Horizontal distances among stations over horizontal β-diversity.

We considered two different ways to estimate horizontal distances among stations: i) using Euclidean distances based on their geographic coordinates and ii) considering the shortest distance from sea (by using a mask over land areas). The two ways of estimating distances gave similar results.
The correlation plot between the two measures of horizontal distances is shown below.

Figure S5.1.



We further considered each one of these horizontal distances to estimate whether they explain an important part of the variance observed in horizontal β-TD, β-FD and β-PD. In both cases, horizontal distances explained a very small part of the observed variance, i.e. the r^2 varied from 0.01 to 0.18 for sea distances and from 0.01 to 0.16 for Euclidean distances. The r^2 per diversity facet and per depth category are shown below in Figures S5.2 and S5.3.

Figure S5.2. Horizontal distances using minimum sea distances.

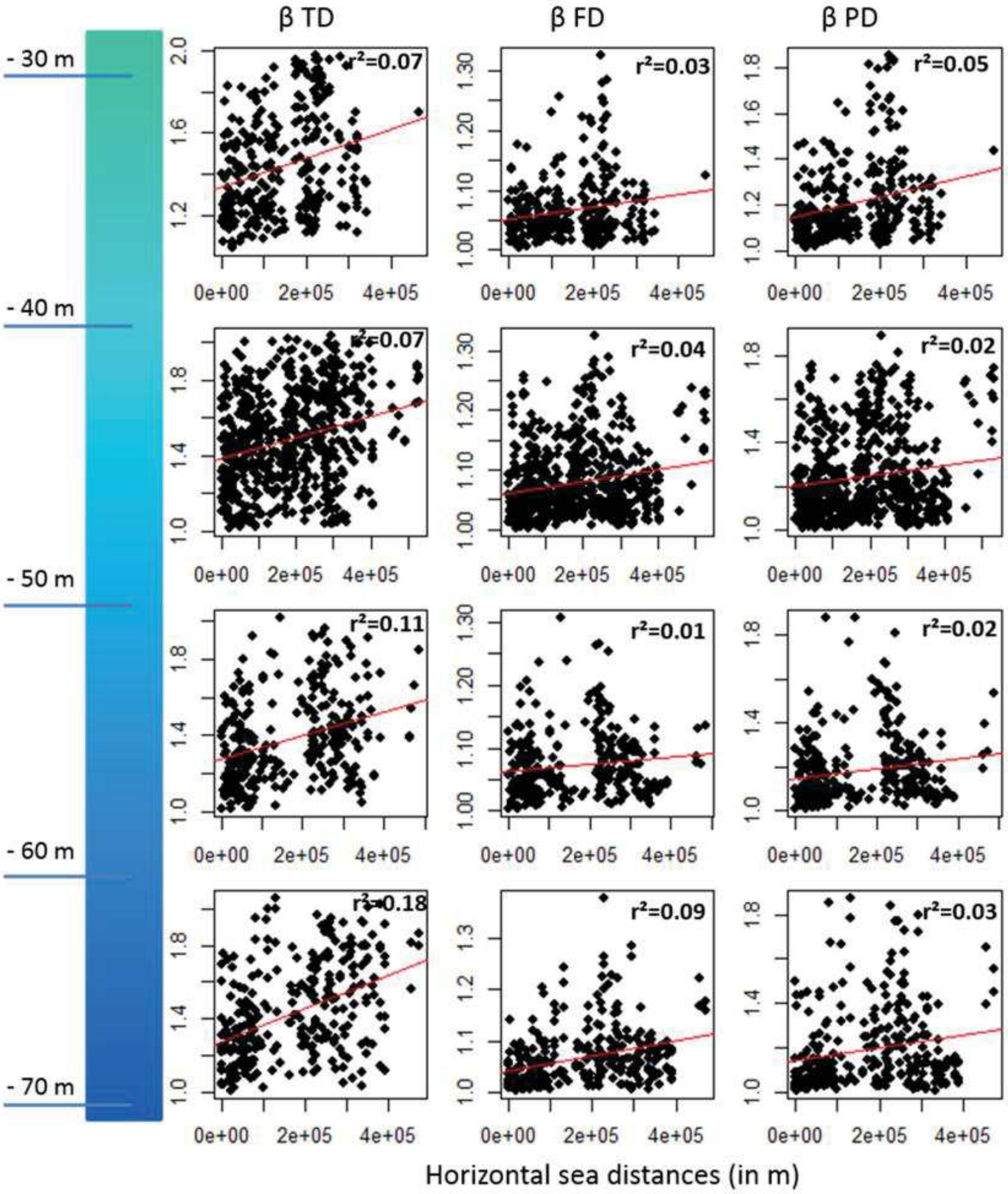
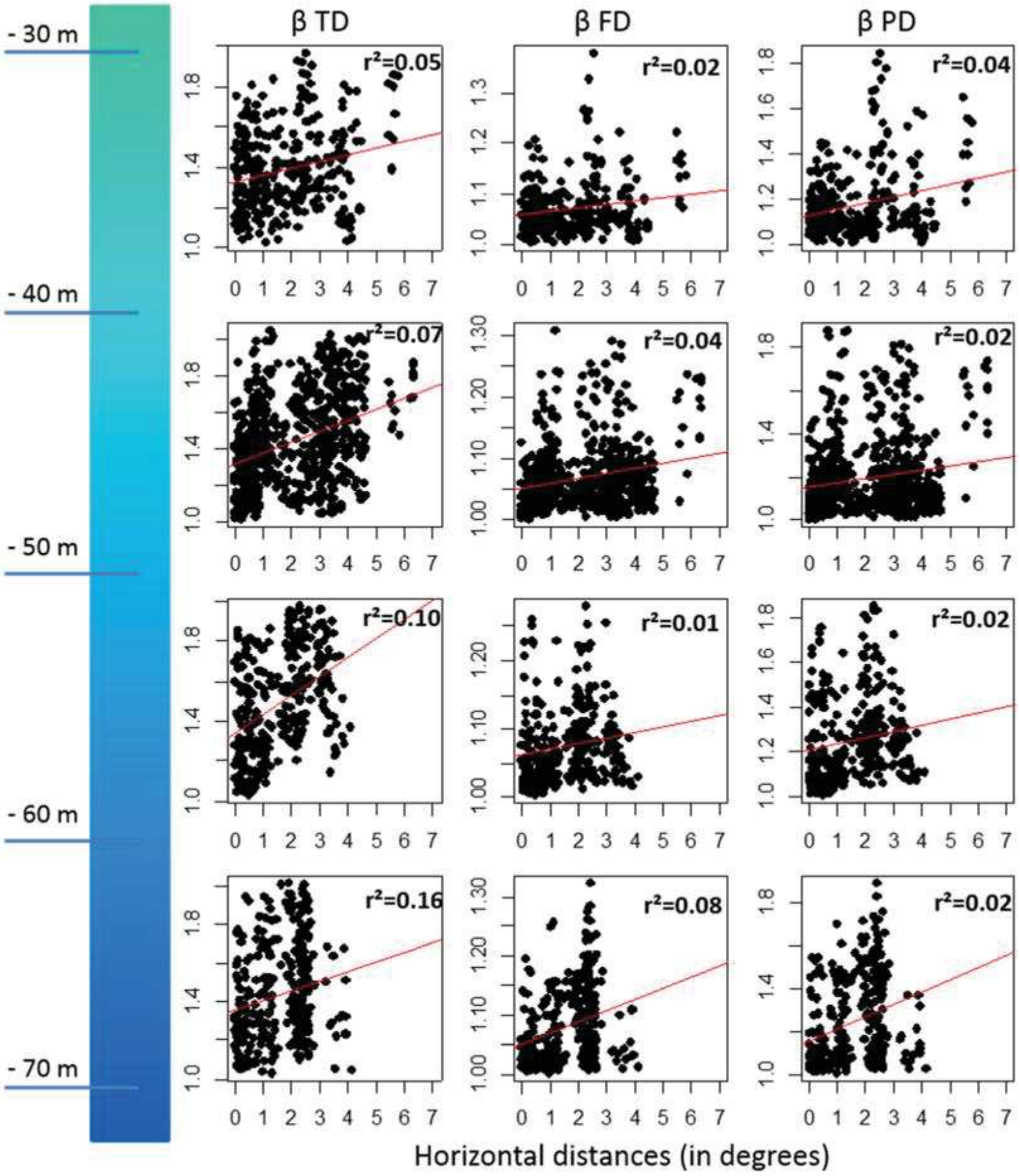


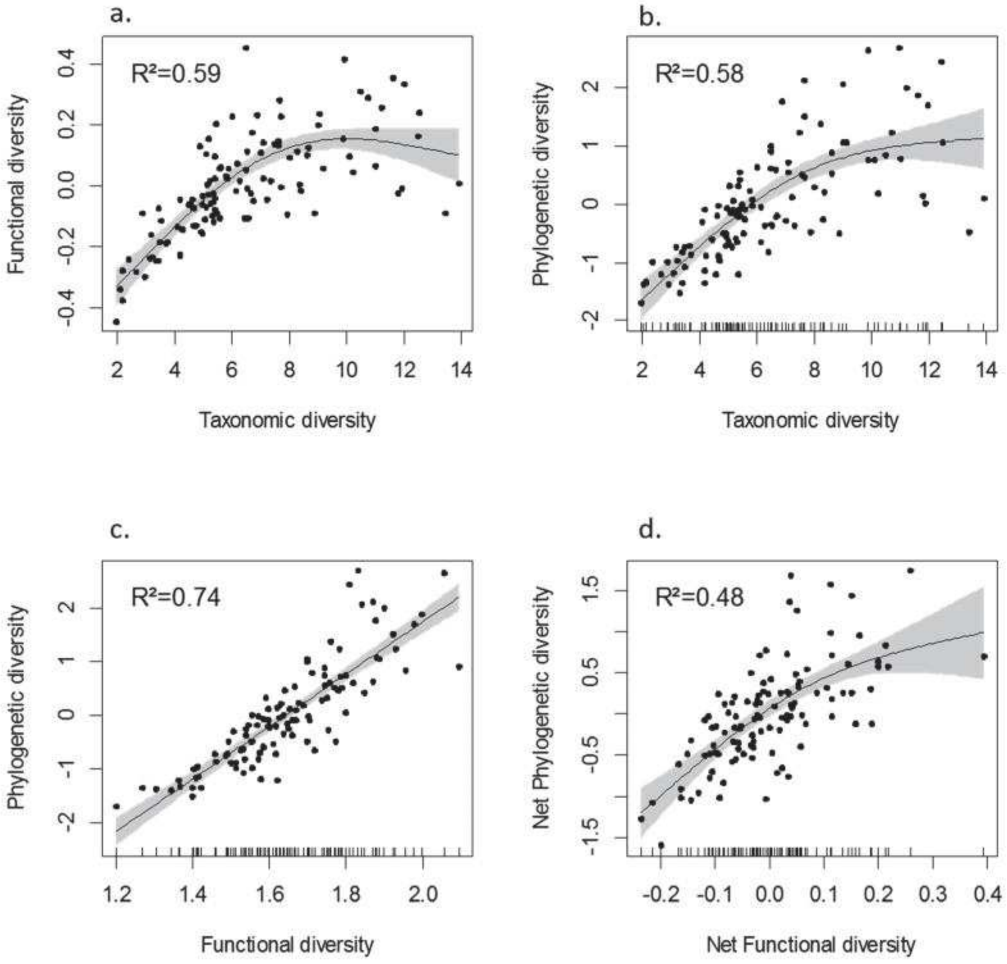
Figure S5.3. Horizontal Euclidean distances.



Appendix S6. Relationships between alpha taxonomic, functional, phylogenetic, net functional and net phylogenetic diversity anomalies.

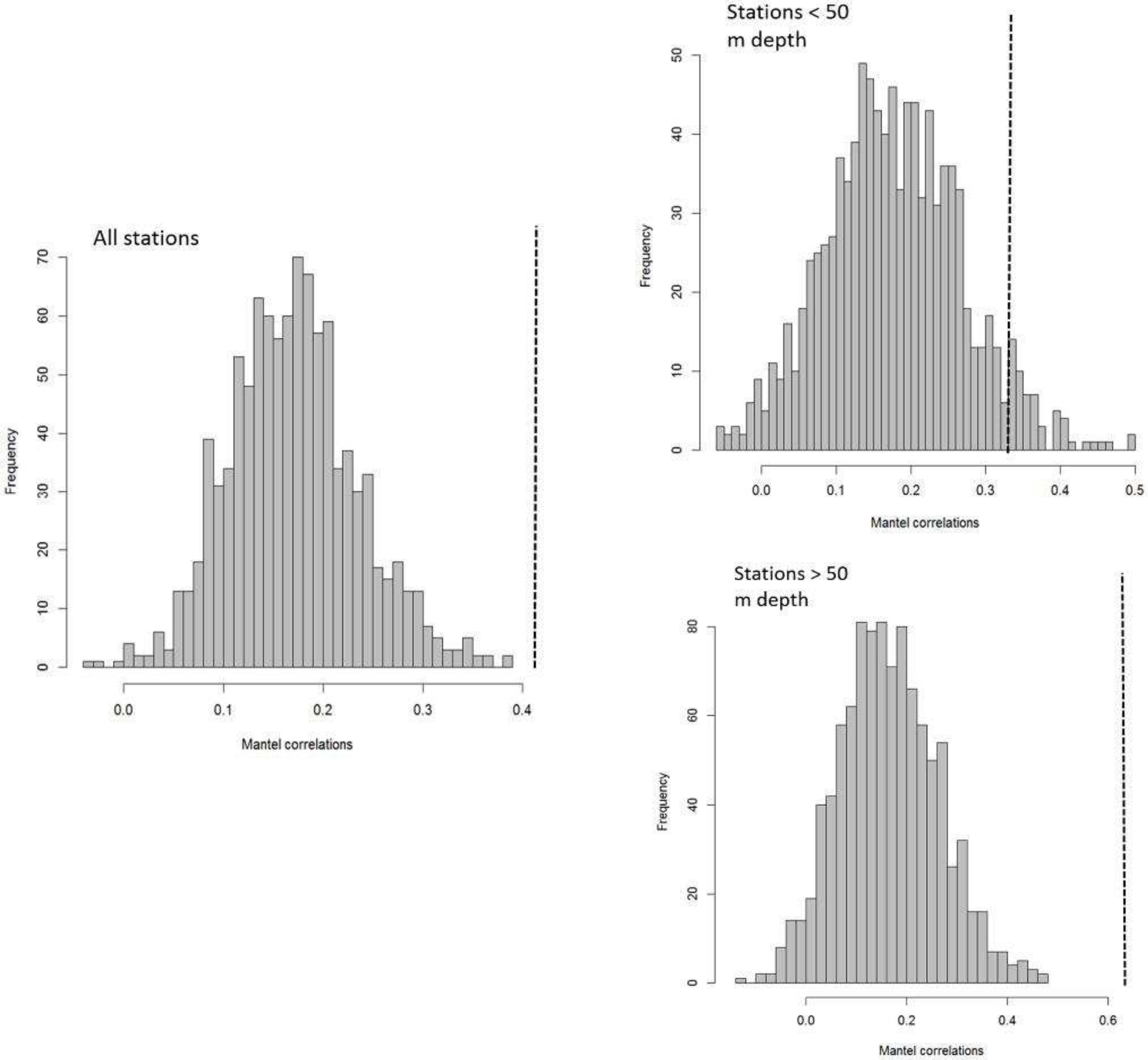
We tested for global linear or non-linear relationships between the three facets of biodiversity, using generalized additive models (GAM) (Wood & Augustin, 2002). A spline effect was considered for all factors with three degrees of freedom (df). Linearity between TD and FD was observed for stations that were relatively poor taxonomically and functionally, whereas functional redundancy was observed for taxonomically richer stations (Fig. a). The same pattern was observed in the relationship between PD and TD (Fig. b). PD and FD were linearly correlated to each other, with no saturation observed (Fig. c). Similar results were obtained regarding net PD and net FD (i.e. residuals of the models PD~TD or FD~TD), even though linearity was less pronounced (c vs d).

Figure S6.1.



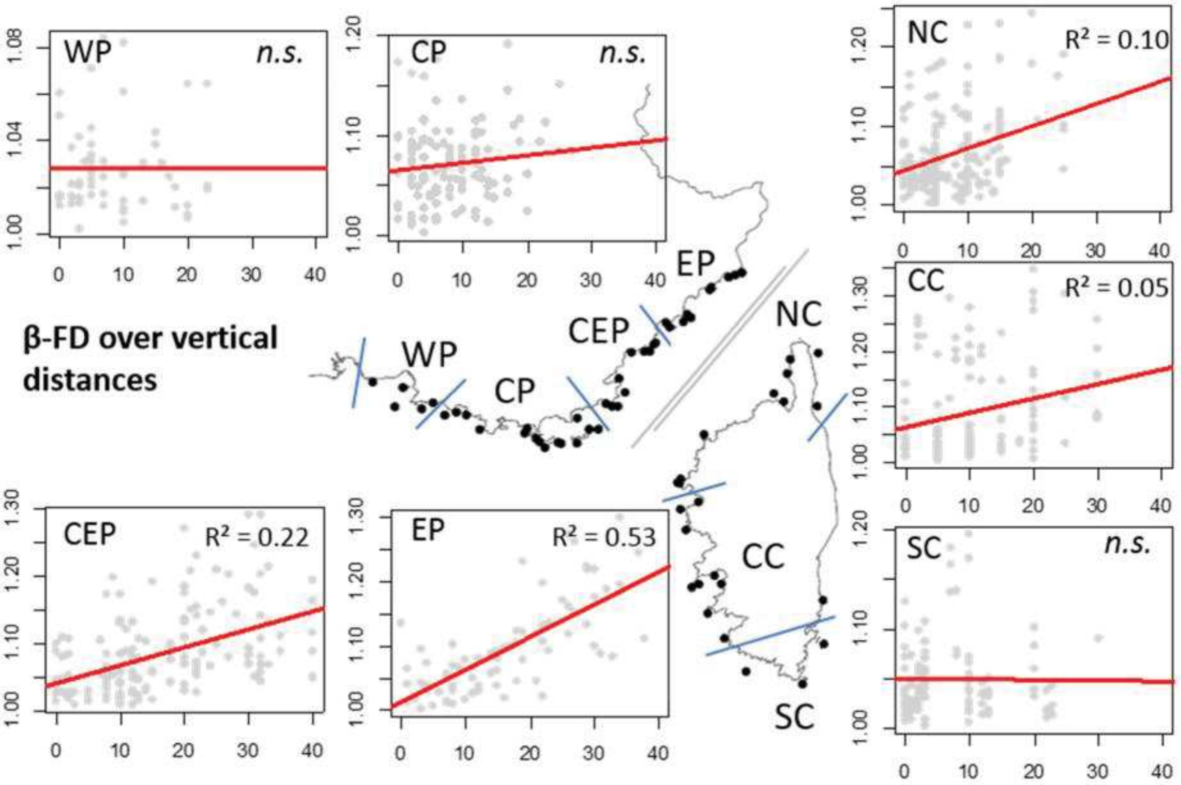
We further verified that the congruence levels between FD and PD continued to be significant after retrieving the TD effect (net FD and net PD). We first estimated observed congruence between net FD and net PD and then compared with the distribution of simulated data. Simulations were obtained from the study species pool repeating a random resampling procedure 1000 times. This congruence analysis revealed similar results than those obtained from the raw data i.e. net FD and net PD were significantly higher that random for all stations considered. Congruence continued to be higher than randomly expected for deeper than 50 meters stations but was not significantly different than random for shallower than 50 metres stations (Figure S6.2.). These results are similar to those obtained for raw data i.e. congruence between FD and PD indices (see Fig. 2 in the main text).

Figure S6.2.



Appendix S7. Vertical beta diversity structure per region. Plots show beta functional diversity over depth (vertical distances in metres). Note that distance between Corsica and Provence-Alpes-Côte d'Azur (PACA) regions is not respected for reasons of visibility. Non-significant trends are indicated as *n.s.*

Functional dissimilarities over depth



Chapitre 2 : Analyse des pressions anthropiques côtières et de leurs intéractions avec les écosystèmes sensibles

Ce chapitre s'articule autour de :

- Un article publié en accès libre dans PlosOne en août 2015 sous le titre : « Fine-scale cartography of human impacts along French Mediterranean coasts: a relevant map for the management of marine ecosystems »
- Un Article à soumettre dans la revue « Biological conservation » sous le titre : « Explaining *Posidonia oceanica* ecological status using coastal anthropogenic pressures along the French coast: implications for management »
- Un article en préparation sous le titre : « Medtrix: a cartographic database concerning marine ecology and anthropogenic pressures along the Mediterranean coast »

1 Articulation entre les articles

1.1 Modélisation des pressions anthropiques côtières

Pour la mise au point d'indicateurs de qualité écologique, il est nécessaire de quantifier les pressions anthropiques subies par les sites suivis. Plusieurs auteurs (Lopez y Royo and Pergent, 2008, Gobert et al., 2009, Selleslagh et al., 2012; Deter et al., 2012a) ont utilisé un indice de pression anthropique pour relier les variables biologiques mesurées localement avec ces dites pressions. L'approche est toujours basée sur la mesure de la distance entre le point d'échantillonnage et la source de la pression en y ajoutant un facteur de pondération suivant la pression considérée. Notre approche dans ce chapitre a été de cartographier les sources de différentes pressions anthropiques, de découper la mer Méditerranée française en pixels de 20 mètres de côté et d'obtenir pour chacun de ces pixels l'intensité de chaque pression en pondérant cette valeur suivant la distance à la source de la pression, la bathymétrie et la courantologie.

La première étape a été de localiser les sources des pressions anthropiques, puis de trouver dans la bibliographie les distances d'impact sur les habitats marins. Une classification de ces pressions anthropiques a ensuite été réalisée en fonction de leur valeur (par exemple, un émissaire urbain classifié en fonction de son débit et de son équivalent habitant). Suite à la modélisation de l'intensité de chaque pression, un impact cumulé a été estimé en couplant la sensibilité des habitats marins à chaque pression selon la méthodologie décrite par Halpern et al. (2007).

L'article publié dans PlosOne sous le titre "*Fine scale cartography of human impacts along the french mediterranean coasts, a relevant map for the management of marine ecosystems*" décrit à la fois la cartographie des fonds marins et les pressions anthropiques sur ces habitats. En corrélant l'intensité des pressions avec le statut écologique des écosystèmes littoraux, nous pourrons ensuite définir des indicateurs de l'état écologique des écosystèmes. Ce travail réalisé à une échelle fine (pixel de 20 mètres de côté) devrait contribuer à hiérarchiser les besoins environnementaux et aider à mieux appliquer et coordonner les mesures de gestion à une échelle pertinente pour la conservation de la biodiversité.

1.2 Liens entre déclin d'herbiers et pressions anthropiques.

La vulnérabilité des habitats marins a été abordée par l'article publié dans PlosOne (Holon et al., 2015) cité précédemment. Celui-ci a notamment montré que les herbiers à posidonie étaient les habitats les plus soumis aux pressions anthropiques côtières et donc les plus vulnérables.

J'ai souhaité dans cette nouvelle partie aborder leur sensibilité : Quelles pressions anthropiques causent la régression des herbiers à posidonie ? Peut-on prédire les régressions futures sur la base des régressions passées ? Comment interagissent ces pressions ? Quelle est la résistance de l'herbier à posidonie face à ces pressions ? Y-a-t-il un seuil dans cette résistance ? Et enfin, quelles informations peut-on en tirer pour aider à une meilleure gestion de cet écosystème ?
J'ai utilisé, pour aborder ces questions, une modélisation basée sur la construction d'arbres et de forêts statistiques. Celles-ci testent aléatoirement, un grand nombre de fois, un échantillon de variables explicatives (ici les pressions anthropiques et la profondeur) sur la régression des herbiers à posidonie. Elle est estimée selon l'indice de conservation IC = surface de matte morte / (surfaces herbier + matte morte) *100 (Moreno et al., 2001). Cet indice est nommé « ecological decline » dans l'article en opposition à « ecological status ». Les surfaces d'herbier et de matte morte proviennent de la cartographie continue des habitats ainsi que de l'analyse des pertes surfaciques d'herbiers (voir chapitre 1).

La précision du modèle a été particulièrement bonne pour une grille de 50 x 50 m couvrant l'ensemble du littoral français méditerranéen : les pressions anthropiques et la profondeur expliquant 77,5 % de la variance de l'indice de conservation. De même, ce modèle appliqué à la prédiction des régressions des herbiers montre un coefficient de corrélation de Pearson entre les valeurs prédites et observées égal à 0,86 permettant d'envisager d'appliquer le modèle créé aux régions voisines d'Italie et d'Espagne.

La découverte de seuils (changepoints) dans l'influence des pressions anthropiques sur le déclin des herbiers permet aussi d'aborder nos résultats sous l'angle de la résistance des écosystèmes. L'ensemble du littoral a ainsi été classé en quatre catégories afin de faciliter les prises de décision concernant la gestion côtière: zone à préserver, zone vulnérable, zone en cours de dégradation et zone déteriorée.

1.3 Une plateforme cartographique en libre accès pour la diffusion des données : Medtrix.fr

La création de cette plateforme s'est imposée d'elle-même pour répondre au besoin de diffusion des données sur la surveillance du milieu marin. Elle a été mise en place initialement pour intégrer les données de Recor (réseau Coralligène) et de Tempo (réseau de suivi des herbiers). Complétée par d'autres réseaux et d'autres organismes, cette plateforme permet aujourd'hui une visualisation globale des données disponibles sur le milieu marin essentiellement en méditerranée française.

Ces données sont mises à disposition gratuitement pour les professionnels de la mer: scientifiques, gestionnaires, élus et intervenants. La plate-forme propose différentes fonctionnalités comme l'édition de cartes ou la comparaison de sites. De nouveaux développements sont en cours par exemple pour permettre le suivi temporel des cartographies.

Un article rédigé sous la forme d'un « data papper » a été soumis à la revue Journal of Marine Biology sous le titre : « Medtrix: a cartographic database concerning marine ecology and anthropogenic pressures along the Mediterranean coast ».

2 Article 4 : Cartographie à fine échelle des pressions anthropiques le long de la côte méditerranéenne française : une carte pertinente pour la gestion des écosystèmes marins.

PLOS | ONE

RESEARCH ARTICLE

Fine-Scale Cartography of Human Impacts along French Mediterranean Coasts: A Relevant Map for the Management of Marine Ecosystems

Florian Holon[1,2]*, Nicolas Mouquet[2], Pierre Boissery[3], Marc Bouchoucha[4], Gwenaelle Delaruelle[1], Anne-Sophie Tribot[1], Julie Deter[1,2]

1 Andromède Océanologie, 7 place Cassan, 34280 Carnon, France, **2** Institut des Sciences de l'Evolution (ISEM)—UMR 5554 CNRS—IRD—UM, Campus de l'Université de Montpellier, 34095 Montpellier cedex 5, France, **3** Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France, **4** Laboratoire Ifremer Environnement Ressources Provence-Azur-Corse, Centre Méditerranée—Zone Portuaire de Brégaillon—CS20 330–83507 La Seyne-sur-Mer Cedex, France

2.1 Abstract

Ecosystem services provided by oceans and seas support most human needs but are threatened by human activities. Despite existing maps illustrating human impacts on marine ecosystems, information remains either large-scale but rough and insufficient for stakeholders (1 km² grid, lack of data along the coast) or fine-scale but fragmentary and heterogeneous in methodology. The objectives of this study are to map and quantify the main pressures exerted on near-coast marine ecosystems, at a large spatial scale though in fine and relevant resolution for managers (one pixel = 20 x 20 m). It focuses on the French Mediterranean coast (1700 km of coastline including Corsica) at a depth of 0 to 80 m. After completing and homogenizing data presently available under GIS on the bathymetry and anthropogenic pressures but also on the seabed nature and ecosystem vulnerability, we provide a fine modeling of the extent and impacts of 10 anthropogenic pressures on marine habitats. The considered pressures are: man-made coastline, boat anchoring, aquaculture, urban effluents, industrial effluents, urbanization, agriculture, coastline erosion, coastal population and fishing. A 1:10 000 continuous habitat map is provided considering 11 habitat classes. The marine bottom is mostly covered by three habitats: infralittoral soft bottom, *Posidonia oceanica* meadows and circalittoral soft bottom. Around two thirds of the bottoms are found within medium and medium high cumulative impact categories. Seagrass meadows are the most impacted habitats. The most important pressures (in area and intensity) are urbanization, coastal population, coastal erosion and man-made coastline. We also identified areas in need of a special management interest. This work should contribute to prioritize environmental needs, as well as enhance the development of indicators for the assessment of the ecological status of coastal systems. It could also help better apply and coordinate management measures at a relevant scale for biodiversity conservation.

2.2 Introduction

Oceans and seas are very important for human well-being; their ecosystems provide among the most important ecosystem services: provision of food, natural shoreline protection against storms and floods, water quality maintenance, support of tourism and other cultural benefits, and maintenance of basic global life support systems [1]. The challenge lies in keeping these resources in a sustainable state of use, which is the main objective of the European Union's Marine Strategy Framework Directive (MSFD, 2008/56/EC) by achieving Good Environmental Status (GES) of EU's marine waters. Yet marine ecosystems and marine resources are under severe anthropogenic threats: population growth, land use change and habitat loss, overfishing and destructive fishing methods, illegal fishing, invasive species, climate change, pollution, increased demand for food and a shift in food preference [2]. The human impact is so great that no region can be considered virgin territory [3–5]. Protecting marine biodiversity and the essential ecosystem services it supports is considered a top priority by different authorities: the scientific community, resource managers, national and international policy agreements, including the MSFD and the Convention on Biological Diversity [6]

In this context, it is essential to analyze species and habitat distribution, environmental variables and human threats but also their correlations. Spatial distribution of anthropogenic pressures is particularly important because it is the basis of numerous other studies: ecological indicators development, species distribution analysis, design of marine reserves and of conservation plans. In this context, large-scale (continental, worldwide) studies are now commonly conducted while local studies (regional) are lacking [7–9]. Naturally, generalization often leads to an extrapolation of the spatial and temporal scales at which reliable predictions can be made because by definition large-scale models are not able to fully account for fine-grained complexity [10,11]. Moreover, large-scale predictions and their limitations may be particularly hard to understand and to use for regional managers and local policy makers focusing on specific interests (i.e. < $1km^2$ grid cells). There is thus a paradox between the international scale of political will and the local scale of biodiversity conservation, but also a gap between global analyses and what can really be done in the field [12]. Consequently, there is a need to provide managers and stakeholders with local fine-scale information.

In order to fill this gap, fine-scale mapping efforts are multiplying in Europe especially in France [13–15], Spain [16–18], Italy [12,19–21] or Greece [22,23] or along the Baltic sea [24]. Because of the high costs to acquire such data, these fine-scale maps are generally funded in order to respond to specific and local objectives (the study of protected areas, a specific habitat [25] or particular features [26], environmental impact assessment [27]). Consequently, they mostly remain local (often a bay) and thus fragmentary (in space but also for the considered habitats and/or pressures) and heterogeneous in their methodology [12]. Moreover they are often available with difficulty: grey literature [15] or communications during conferences [28] instead of publications (but see [29]). All of this can be an obstacle to understand the impact of pressures on coastal marine habitats and thus to make decision at a local and regional scale. Fine-scale (15 x 15 m grid cells) spatial models have been recently developed in order to link multiple pressures with various coastal ecosystem status within a marine protected area [30]. The implementation into geographical information systems (GIS) allows a predictive approach of the consequences of different management alternatives [30–32]. This represents an important decision support tool for choosing efficient management solutions in the face of complex interactions and high uncertainty. Information concerning pressure distribution (presence/absence of relevant human activities, weighted distance of these activities) have been here successfully used in order to map potential impacts [33–36]. These data are so useful for local managers that they should exist all along the coast. A map of the diverse coastal marine habitats, of coastal pressures and impacts on these habitats extended to the entire coast would have an interest for the local managers and stakeholders but also for regional and national authorities. It would permit to feed the overall think on the coastal use but also highlight conservation and management priorities, compare different sites and management ways and assess the water body quality.

The objective of this work is to map and quantify, at a large spatial scale though in fine and relevant resolution for managers, the main drivers and pressures triggering changes within coastal marine ecosystems. In order to reach these objectives, localization of the different pressures exerted and their impacts is needed as much as maps of marine ecosystems. Among numerous seas listed on Earth, Mediterranean presents the particularity of being a biodiversity hotspot facing numerous and strong threats [37–40]. While maps of cumulative human impacts on marine ecosystems exist at the scale of the Mediterranean and Black sea and even at worldwide scale [5,40–42], this information could be completed along the coast. For instance, the resolution used by Micheli et al. [41] within the Mediterranean and Black sea is 1 km² pixels and no data is associated with the first pixels close to the coast, where most anthropogenic pressures are concentrated.

Interested in data that could be of real use to local managers and stakeholders, we sampled data from a homogeneous environmental policy context and thus focused on a unique country: France with its 1700 km of Mediterranean coastline, including Corsica. Our goals were to (1) provide the first complete marine coastal habitat map of the French Mediterranean coast (including Corsica), and (2) to quantify and map cumulative impacts to provide the data needed (one pixel = 20 x 20 m) to help the development of an effective marine policy. On these bases, we identified the most and least impacted areas (water bodies), the top threats affecting coastal waters, and the areas representing top priorities for ecosystem-based management and conservation efforts. The cumulative impact map obtained will be useful for local decision makers and thus complementary to large-scale previous works [41].

2.3 Materials and Methods

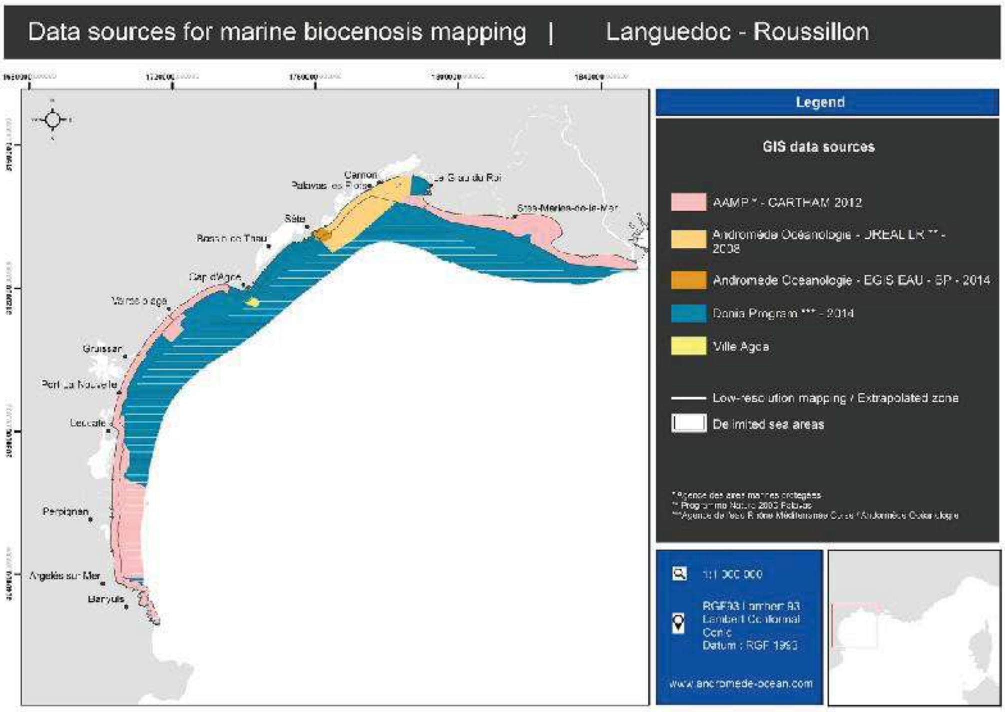
Marine habitats

The study considers the entire French Mediterranean coastline (including Corsica) included within the 46 water bodies of homogeneous water according to the Water Framework Directive (WFD,2000/60/EC) [43]. Interested in costal-based impacts, we particularly focus on the shallow part: between 0 and -80 m. After a bibliographic synthesis, we gathered and homogenized data on habitat maps; these data were collected by Andromède Océanologie, Agence de l'Eau RMC;Conservatoire du Littoral, DREAL PACA; EGIS EAU, ERAMM, GIS POSIDONIE, IFREMER, Institut océanographique Paul Ricard, Nice Côte d'Azur, TPM, Programme CARTHAM - Agence des Aires Marines Protégées, ASCONIT Consultants, COMEX-SA, EVEMAR, IN VIVO, Sintinelle, Stareso, Programme MEDBENTH, Université de Corse (EQEL), Ville de St Cyr-sur-mer, Ville de Cannes, Ville de Marseille, Ville de St Raphaël, Ville de St Tropez (S1-S3 Figs.).

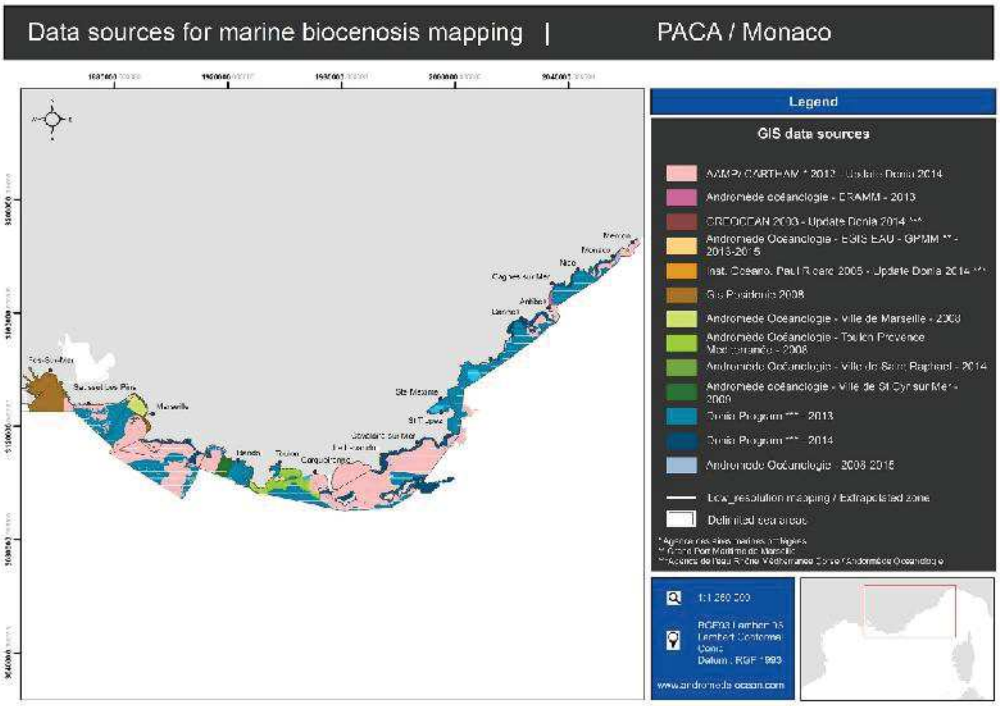
Gaps were completed with the program DONIA [44] with a fine scale (1:10 000 map) between 0 and -80 m and a lower resolution (1:25 000) beyond (S1-S3 Figs). Campaigns were led between 2010 and 2014 using first aerial or satellite photography (in order to measure the spatial extent of habitats in shallow waters) and a multi-beam echo-sounder GeoSwath Plus (Kongsberg Geoacoustics LTD) survey (to draw the bathymetry). Then, a side-scan sonar survey (used in more turbid and deeper (< -15 m) waters) was led. By ensonifying a swath of seabed and measuring the amplitude of the backscattered return signals, an image of the seabed was built up with information on the morphology and substrate content. We used a Klein System 3900 with a frequency comprised between 445 and 900 kHz. After that, sonar information was post-treated to determine the potential presence and coverage of underwater habitat representation. All of these data allow achieving a preliminary cartography of benthic habitats.

Numerous uncertainties still remained after this preliminary cartography work. Direct observations (“ground-truth points”) were thus needed through diving sessions (around 1600 dives between 0 and -80 m all along the coastline between 2010 and 2014). They included classic dives and “towed dives” that allowed the sampling of 20 920 ground-truth points. During “towed dives”, the diver was actively able to maneuver a “towboard” to maintain a relative constant elevation above the seabed. The towboard was equipped with an underwater GPS transducer providing the accurate position and exact depth of the diver in real-time to the surface operator. The diver equipped with an integrated communication system transmitted a large quantity of information on benthic habitats (community of organisms which lived on, in, or near the seabed, state of the habitat, occurrence of impacts on the habitat). Occasional exploring dives aimed, by means of in situ observation, to clarify data. These dives allowed to recognize the nature of the seabed and to characterize benthic populations. Field work was organized in cooperation with the French water agency (public authority) which gave permission to conduct the work. Field work was also declared to the authorities responsible of the concerned marine parks. The field studies did not involve endangered or protected species.

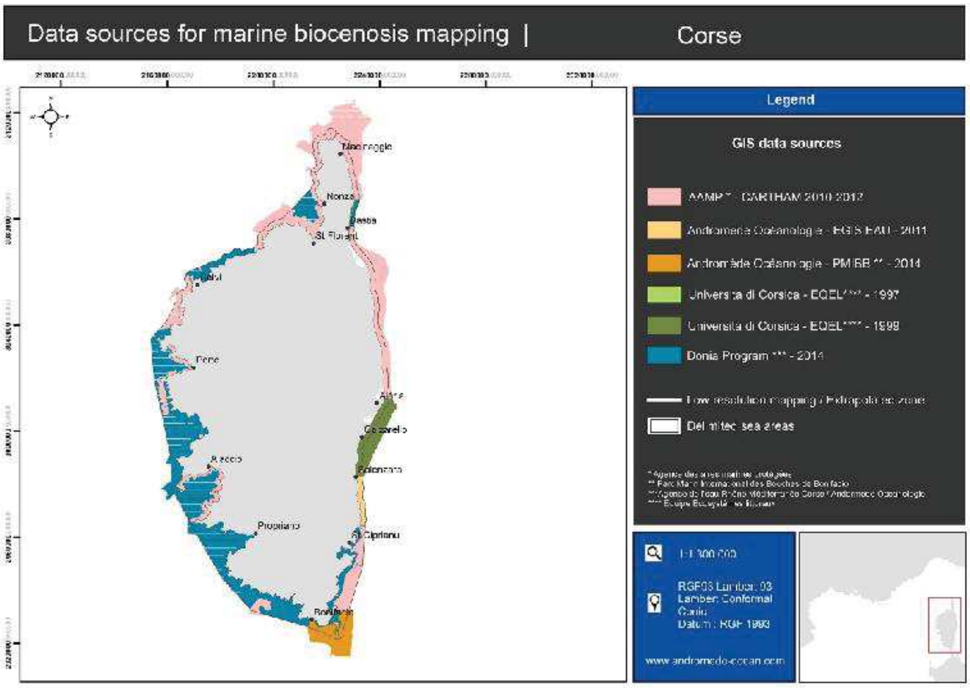
A final continuous habitat map (scale = 1:10 000 between 0 and – 80 m and 1:25 000 beyond in the case of deeper water bodies) was realized comprising eleven habitat classes: *Cymodocea nodosa* seagrass, *Zostera marina* and *noltii*. seagrass*, Posidonia oceanica* seagrass, dead matte association, infralittoral shingle association, infralittoral soft bottoms, photophilous algae association, coralligenous assemblages, circalittoral soft bottoms, artificial habitats, offshore rocks. Ecosystem data were finally converted into presence/absence 20 x 20 m pixel layers (in order to be adapted to the pixel size related to the anthropogenic pressures, see below); the habitat corresponding to each pixel was defined by the major habitat observed within the grid (percent cover > 50 %).



S1 Fig. Origin of the data used for marine habitat mapping on the Western part of the French Mediterranean coast (Languedoc-Roussillon French region). The study considers the coastline included within the water bodies. We particularly focus on the shallow part: between 0 and -80 m. After a bibliographic synthesis, we gathered and homogenized data on habitat maps. Gaps were completed with the program DONIA with a fine scale (1:10 000 map) between 0 and -80 m and a lower resolution (1:25 000) beyond.



S2 Fig. Origin of the data used for marine habitat mapping on the Eastern part of the French Mediterranean coast (Provence Alpes Côte d'Azur French region). The study considers the coastline included within the water bodies. We particularly focus on the shallow part: between 0 and -80 m. After a bibliographic synthesis, we gathered and homogenized data on habitat maps. Gaps were completed with the program DONIA with a fine scale (1:10 000 map) between 0 and -80 m and a lower resolution (1:25 000) beyond.



S3 Fig. Origin of the data used for marine habitat mapping on the Eastern part of the French Mediterranean coast (Corsica French region). The study considers the coastline included within the water bodies. We particularly focus on the shallow part: between 0 and -80 m. After a bibliographic synthesis, we gathered and homogenized data on habitat maps. Gaps were completed with the program DONIA with a fine scale (1:10 000 map) between 0 and -80 m and a lower resolution (1:25 000) beyond.

Anthropogenic pressures

Drivers and pressures are here defined according to the DPSIR framework (drivers-pressures-states-impacts-responses) [45] with drivers such as the main socio-economic and socio-cultural forces increasing or mitigating pressures on the environment (rapid population expansion for example). Pressures are defined as stresses that human activities induce on the environment (e.g. wastewater), states being the condition of the environment (e.g. water quality or species richness). Impacts are defined as the effects of these pressures on the environment (e.g. biodiversity loss) and responses are what society does in order to improve the environmental situation (e.g. better wastewater treatment or regulation). Impacts may differ according to the ecosystems considered because of their variable vulnerability: all ecosystems are not threatened in the same way (functional impact, scale, frequency) and are not equally sensitive (resistance and recovery time) [46]. Here we modeled the spatial extent of anthropogenic pressures on the marine environment. We only focused on pressures that can be controlled by local stakeholders. Thus we did not take climate change issues and industrial fishing into account contrary to Micheli et al. [21] because their control appeals for high-level decisions. In addition to this, climate drivers are not considered among the MSFD's good environmental status descriptors [47]. Ten different pressures (based on quantitative data) were considered: (1) man-made coastline (big harbours / harbours / artificial beaches, ports of refuge / pontoons, groynes, landfills and seawalls areas), (2) boat anchoring (number and size of boats observed during summer), (3) aquaculture (total area of the farms), (4) urban effluents (capacity, output), (5) industrial effluents (chemical oxygen demand), (6) urbanization (land cover), (7) agriculture (land cover), (8) coastline erosion (land cover), (9) coastal population (size and density considering the inhabitants-residents) and (10) fishing (traditional and recreational fishing areas) [see S4 text for details]. Even if continuous pressures (e.g. wastewater) are generally distinguished from discrete pressures (e.g a groyne building), low resilience of marine ecosystems (especially *Posidonia oceanica* beds and coralligenous reefs; [48,49]) allow the combination of both pressures within the same methodology.

Data concerning the origin and intensity of these pressures are available in published databases: MEDAM [50], CORINE land cover [51], INSEE [52], MEDOBS data [53] but were also provided by Agence de l'Eau RMC and Ifremer completed with an analysis of satellite-aerial pictures and unpublished data (Andromède Océanologie). Models of the spatial extent of the pressures were built using ArcGIS 10 (ESRI) with a 20-m distance matrix. We applied a pressure curve (type y=ae-bx) considering the distance to the source with a negative exponential shape ranging between 100 % (origin) and 0 % (no more impact) to each type of pressure. We included the bathymetry to model the spread of each pressure based on literature synthesis and our expert knowledge. Details and parameters of each modeled pressure are given in the supplementary file S4 text.

Cumulative human impacts

We used a cumulative impact model following Halpern et al. [5,25] and Micheli et al. [21]. First, we assembled spatial datasets for n= 10 anthropogenic pressures (value Di) (see S4 text) and m= 11 habitats (value Ej). Secondly, all pressure layers were then log[X+1]-transformed and rescaled between 0–1 to allow direct comparisons. The sum of the different pressures per pixel was calculated. Then, cumulative impacts scores (IC) for each 20 x 20 m pixel were calculated according to Micheli et al. [21] and Halpern et al. [5]:

$$I_C = \sum_{i=1}^{n} \sum_{j=1}^{m} D_i * E_j * \mu_{i,j}$$

Where Di is the value of an anthropogenic pressure at location i, Ej is the presence or absence of habitat j and µi,j is the impact weight of anthropogenic pressure I and habitat j [5]. Like Micheli et al. [21], values of impact weights were deduced from Halpern et al. [25].

Cumulative impact to individual ecosystems (I_E) was calculated as follows:

$$I_E = \sum_{i=1}^{n} D_i * E_j * \mu_{i,j}$$

and impact of individual pressures across all ecosystem types (I_D) was calculated as follows:

$$I_D = \sum_{i=1}^{m} D_i * E_j * \mu_{i,j}$$

To simplify visualization, impacts were classified in six categories depending on I_C values: very high (Ic>10); high (8<Ic<10); medium high (2.1<Ic<8); medium (0.6<Ic<2.1); low (0.1<Ic<0.6); and very low impact (Ic<0.1). We calculated and mapped I_c along the entire French Mediterranean coasts, for each marine habitat and for each water body (water bodies are here used for their interest in marine policy and as spatial references). Pixels free from any pressure (all pressures equal to null values) were not further considered.

2.4 Results

Marine habitats

The final continuous map of marine habitat (1:10 000 between 0 and -80 m and 1:25 000 beyond) consists of 5 785 972 pixels and covers 373 206 ha (Table 1): *Cymodocea nodosa* seagrass (506 ha), *Zostera marina* and *noltii* seagrass (572 ha), *Posidonia oceanica* seagrass (70 641 ha), dead matte association (5 693 ha), infralittoral shingle association (211 ha), infralittoral soft bottoms (102 451 ha), photophilous algae association (12 617 ha), coralligenous assemblages (2 661 ha), circalittoral soft bottoms (177 483 ha), artificial habitats (233 ha), offshore rocks (138 ha). Maps (one pixel = 10 x 10 m) are freely available on www.medtrix.fr in DONIA® expert (see box 1 and an example of map concerning the golfe of St Tropez in S5 file). A total of 231 606 ha is considered in this study after removing 141 600 ha not concerned by any of the pressures taken into account; the removed areas are located along the deepest limits (deeper than – 100 m) of the water bodies. Most (92 %) of the mapped marine bottom is covered with three habitats: infralittoral soft bottoms (38 %), *P. oceanica* meadows (28 %) and circalittoral soft bottom (25 %).

Table 1. Analysis of the cumulative impact scores per marine habitat. Average, standard deviation (SD) and sum of the cumulative impact scores (I_C) obtained by each 20 x 20 m cell composing each marine habitat (j). Areas of the habitats are indicated in ha.

Habitat	Area	Average (SD)	Sum
Cymodocea nodosa	446	4.85 (3.10)	54 048.88
Zostera marina and *noltii*	571	5.43 (1.50)	77 440.71
Posidonia oceanica	65 817	2.79 (2.51)	4 588 523.53
Dead matte	5 173	3.57 (2.35)	462 194.07
Infralittoral shingle association	169	5.48 (2.56)	23 153.43
Infralittoral soft bottoms	88 716	2.88 (2.35)	6 396 440.24
Photophilous algae	10 605	3.75 (3.02)	993 520.08
Coralligenous habitat	1 762	1.83 (1.83)	80 630.67
Circalittoral soft bottoms	58 049	0.73 (0.82)	1 056 582.46
Offshore rocks	21	0.85 (1.12)	435.96
Artificial habitats	109	2.56 (1.04)	6 975.16

Anthropogenic pressures

Maps concerning each of the ten pressures are available on www.medtrix.fr in IMPACT project (see box 1 and examples of maps concerning the golfe of St Tropez in S5 file). Five pressures concern more than 40 % of the considered area: urbanization (70 %), coastal population (54 %), coastal erosion (47 %), man-made coastline (43 %) and agriculture (41 %). Pressures showing the highest cumulated value are urbanization, coastal population and man-made coastline (Fig. 1). Urbanization is the most important pressure exerted on all habitats except on coralligenous assemblages, circalittoral soft bottoms and offshore rocks where fishing prevails and on artificial habitats where man-made coastline predominates (Fig. 2). All pressures affect every habitat except for *Zostera marina* and *noltii* meadows which are not impacted by aquaculture, urban effluents, agriculture and fishing, and offshore rocks which are not affected by anchoring, aquaculture and industrial effluents (Fig. 2).

Figure 1. Total cumulated value (sum of all 20 x 20 m cell values concerning a pressure) of each individual pressure (no unit).

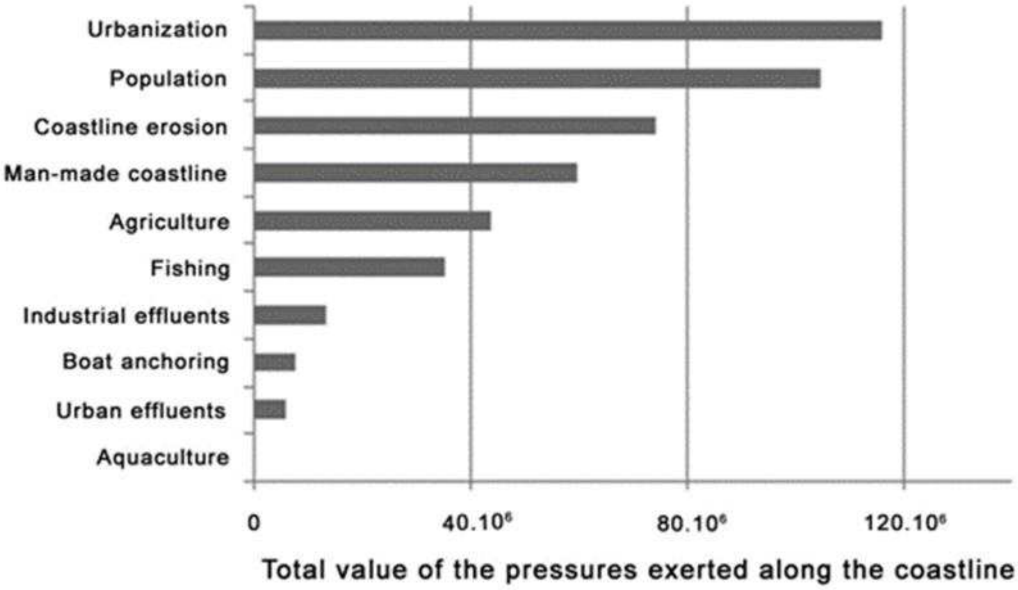
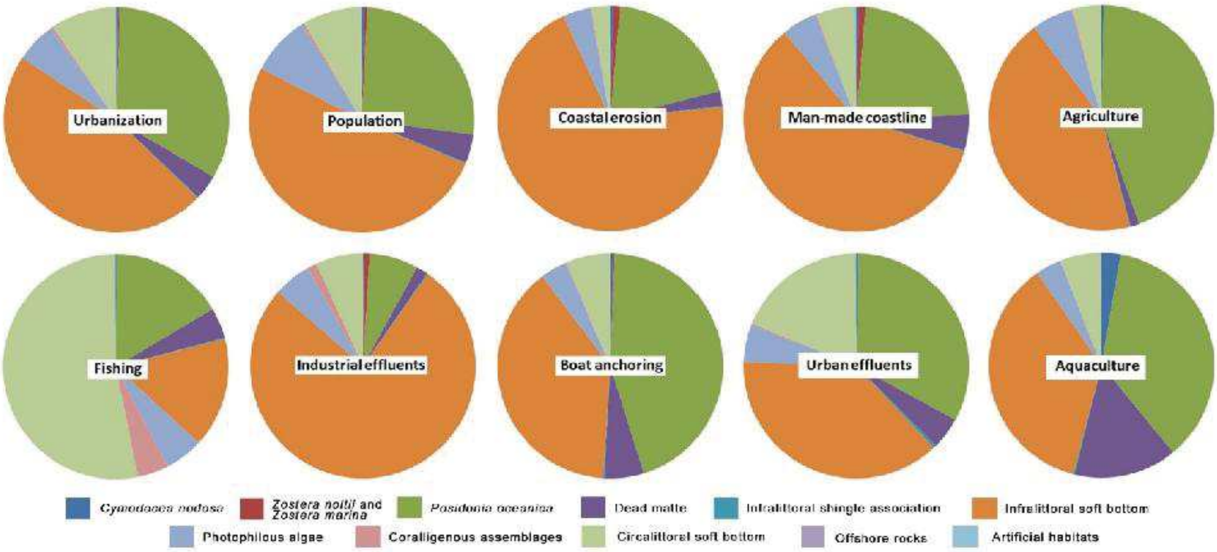


Figure 2. Repartition of the total cumulated value of each pressure in function of the habitats. Each pie chart indicates the repartition (in percent) of the total cumulated value (sum of all 20 x 20 m cell values concerning a pressure) of each individual pressure (name indicated in the white rectangles) in function of the habitats.



Cumulative human impacts

Analysis per habitat

Cumulated impact scores range between 0 and 15 (Fig. 3 and S5 file). The highest sums of cumulated impact scores (I_c) are observed on infralittoral soft bottoms and *P. oceanica* meadows. The strongest mean I_c range between 4.85 and 5.48 (medium-high impact): they are observed on infralittoral shingle association, *Zostera marina* and *noltii* and *Cymodocea nodosa* meadows. I_c presents the highest variance on *Cymodocea nodosa* meadows and photophilous algae (Table 1).

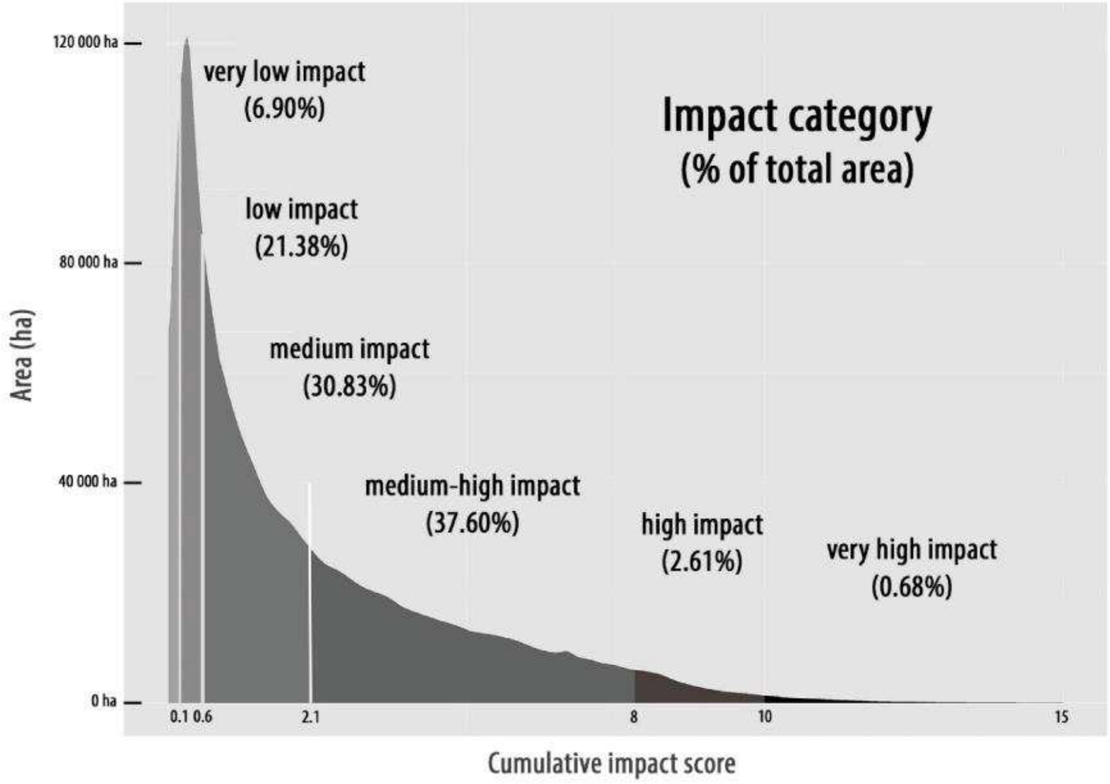


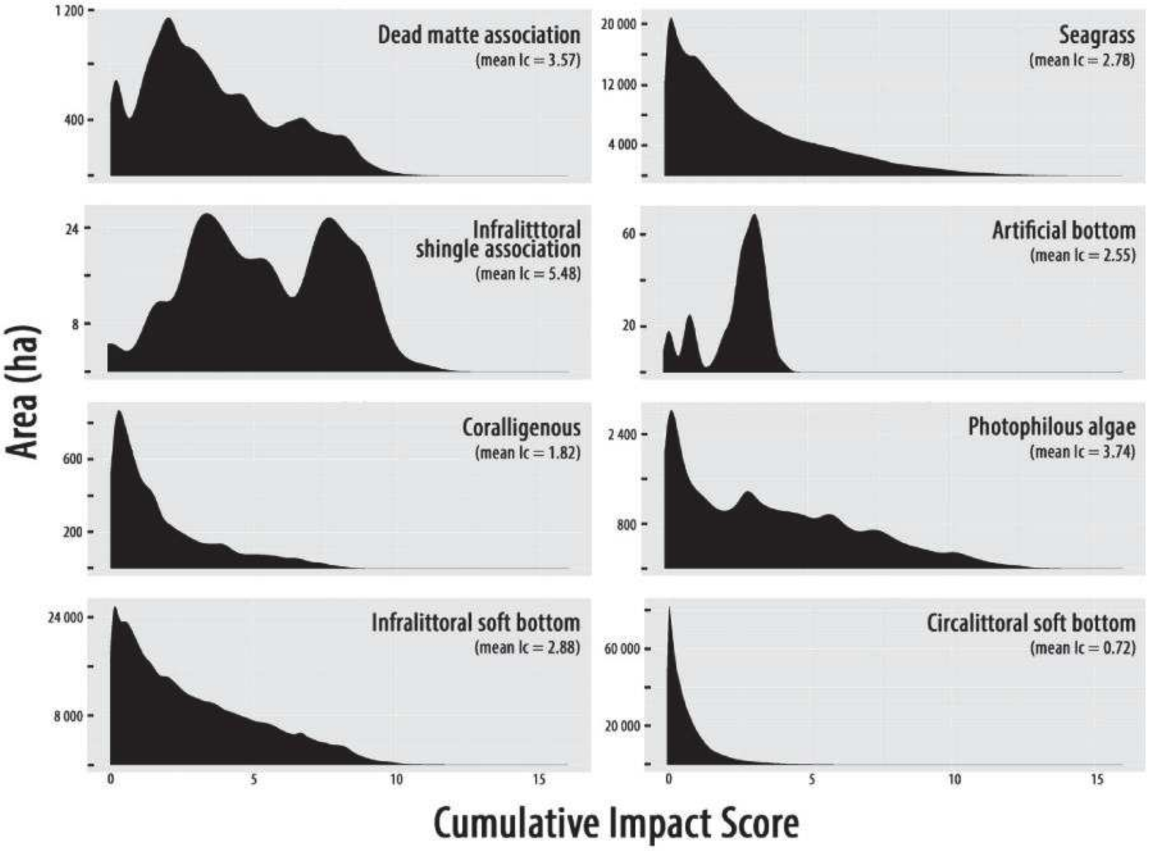
Figure 3. Cumulative impact scores (I_C) depicting the area (in ha) and the percent of total area (in parentheses) that falls within each impact category.

Table 2. Analysis of the cumulative impact categories per marine habitat. Percent of each marine habitat area affected by the different cumulative impact categories: very high impact (Ic>10); high impact (8-10); medium-high impact (2.1-8); medium impact (0.6-2.1); low impact (0.1-0.6); and very low impact (<0.1). Areas are indicated in ha.

Habitat	Area	Percent of area affected by the different cumulative impact categories					
		Very low	Low	Medium	Med-high	High	Very high
Cymodocea nodosa	446	4.18	5.62	8.36	61.37	12.42	8.05
Zostera marina and *noltii*	571	0.00	0.00	0.03	96.01	3.97	0.00
Posidonia oceanica	65 817	3.78	15.03	32.62	43.67	3.43	1.47
Dead matte	5 173	2.52	5.96	23.41	62.90	5.02	0.19
Infralittoral shingle association	169	1.97	1.02	7.72	68.65	19.35	1.30
Infralittoral soft bottoms	88 716	4.20	13.62	29.31	49.67	3.02	0.18
Photophilous algae	10 605	5.26	13.55	18.31	52.37	6.79	3.72
Coralligenous habitat	1 762	6.43	25.04	37.06	31.18	0.28	0.00
Circalittoral soft bottoms	58 049	15.25	43.58	34.50	6.67	0.00	0.00
Offshore rocks	21	9.94	46.39	33.92	9.75	0.00	0.00
Artificial habitats	109	0.22	7.12	16.03	76.63	0.00	0.00

All marine habitats are mostly subjected to medium high impacts except for coralligenous assemblages subjected to medium impacts, and circalittoral soft bottoms as well as offshore rocks concerned by low impacts (Table 2, Fig. 4). Dead matte, infralittoral shingle association and artificial habitats are less subject to an I_c inferior to medium-high (Fig. 4). Around 3.3 % of habitats undergo high or very high I_c (especially *P. oceanica* meadows). On the contrary, 28.3 % of habitats are associated with low or very low *Ic* especially circalittoral soft bottoms, infralittoral soft bottoms, then *P. oceanica* meadows (Table 2). The mean I_c is the highest between 0 and-15 m depth for almost all habitats (except circalittoral soft bottoms and offshore rocks absent beyond-15 m depth).

Fig. 4. Distribution of cumulative impact scores (I_C) for each habitat

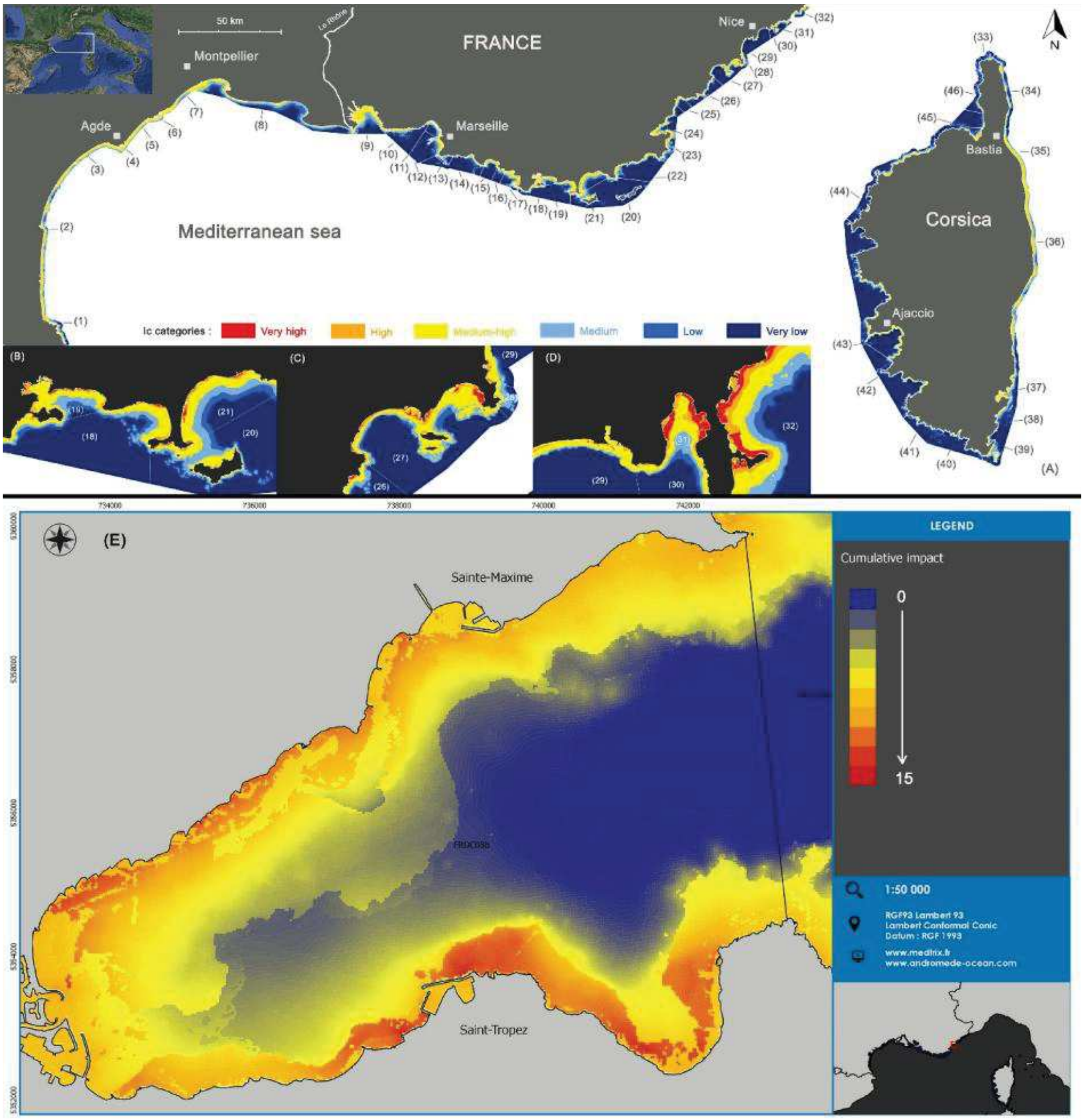


Spatial analysis per water body

Concerning water bodies (localization presented in Fig. 5), the highest sums of cumulated impact scores (sum of the score of each 20 x 20 m cell) are observed within water bodies 36 and 2, two of the biggest water bodies of the area (Table 3). The highest mean I_c correspond to medium-high impact values; they are observed among the smallest water bodies: average = 7.14 within water body 31 (also presenting the highest variance) and average = 5.82 within water body 40 (Table 3). All water bodies are mostly subjected to medium-high impacts (average ranging between 2.1 and 8, Fig. 3 and Table 3). All water bodies (except two: 13 and 33 located around the Calanques of Marseille and the Northern cape of Corsica) contain areas with high or very high I_c (Table 3). On the contrary, all water bodies contain areas associated with low or very low I_c especially water bodies 8, 14, 15, 33, 34, 41, 42, 44 and 46 located in Corsica, within the Calanques of Marseille and within the Western part of the Rhône (Fig. 5 and Table 3).

Regarding the anthropogenic pressures, urbanization is the major pressure affecting all water bodies except for 5, 7, 41, 12, 13, 14, 27, 32, 38 (for which population is the major pressure); 39, 40, 33, 34 (for which agriculture is the major pressure); 9, 11 (for which man-made coastline is the major pressure); 8

(for which coastal erosion is the major pressure) (Fig. 6). The relative influence of agriculture is higher for water bodies 33 to 46 *(*Corsica), while that of industrial effluents is higher within water bodies 9 and 14 (Fos-sur-mer and Eastern part of Marseille), and urban effluents are relatively more important between the Eastern part of Marseille and Nice (Fig. 6).



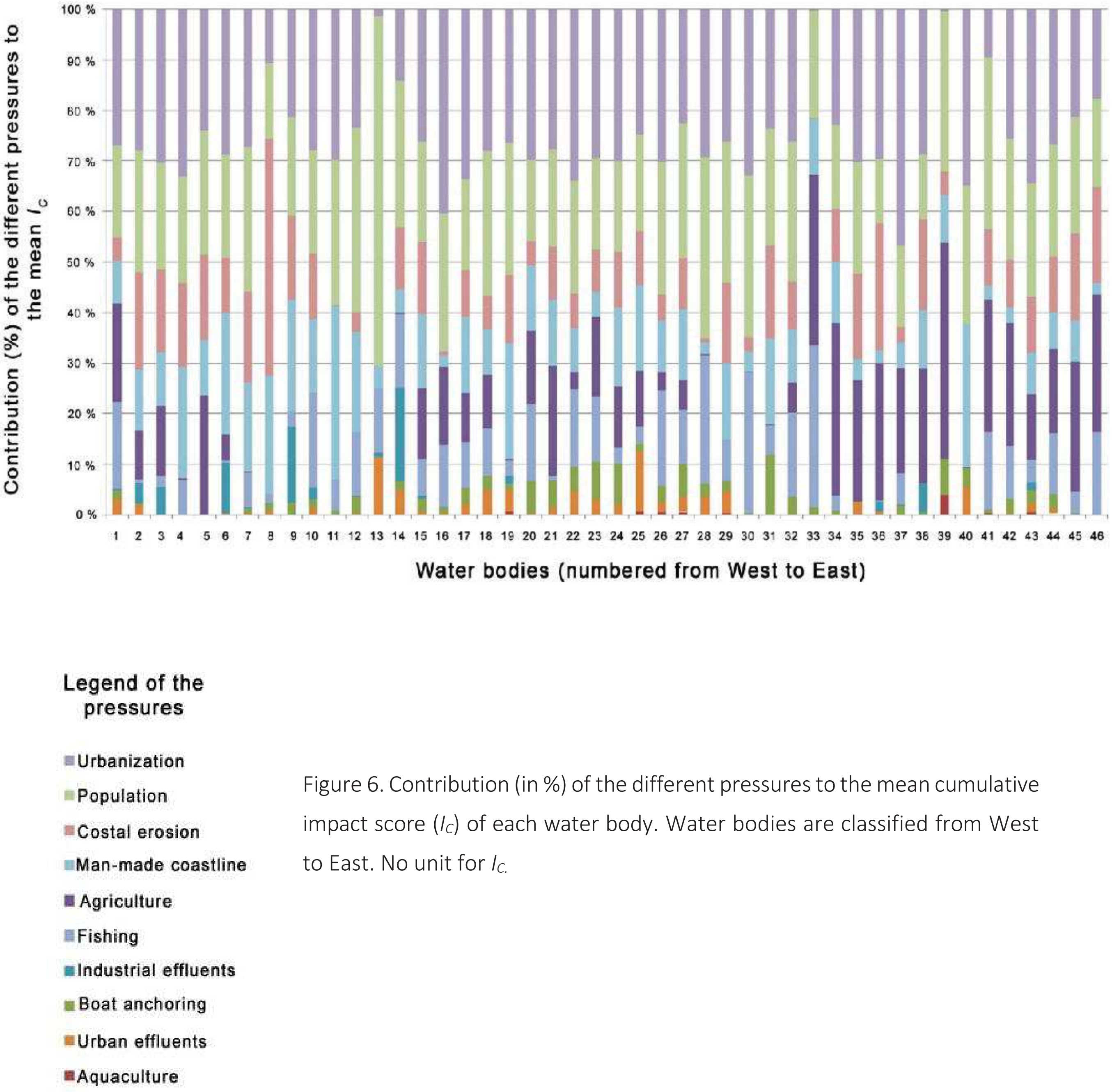


Figure 6. Contribution (in %) of the different pressures to the mean cumulative impact score (I_C) of each water body. Water bodies are classified from West to East. No unit for I_C.

Table 3. Analysis of the pressures per water body. For each water body, the table describes its total area, its area covered by each cumulative impact score (I_C) category (very low, low, medium, medium-high, high, very high impact), the average and standard deviation (SD) of I_C and the sum of the I_C obtained by each 20 x 20 m cell composing the water body. Coastal water bodies are numbered from West to East. Areas are indicated in ha.

Water body	Total area	Area covered by each cumulative impact category						Average I_C (SD)	Sum of the I_C
		Very low	Low	Medium	Medium-high	High	Very high		
1	2 910	142	634	897	1 115	90	31	2.56 (2.53)	185 865.11
2	15 474	149	1 712	5 544	7 651	316	102	2.83 (2.22)	1 094 139.14
3	4 196	1	79	1 502	2 456	151	8	3.25 (2.10)	341 164.18
4	1 626	0	25	600	954	43	4	3.29 (2.14)	133 758.01
5	2 766	0	31	856	1 847	31	0	3.37 (1.88)	232 941.63
6	1 869	0	86	568	994	216	4	3.87 (2.71)	180 724.37
7	12 738	1 033	1 781	3 428	6 349	146	0	2.57 (2.11)	817 864.25
8	17 004	2 444	6 008	5 819	2 728	4	0	1.07 (1.23)	456 055.90
9	11 815	371	1 538	2 877	6 377	579	74	3.34 (2.63)	987 026.81
10	4 314	272	1 055	1 420	1 231	252	85	2.61 (2.91)	281 823.67
11	2 145	203	338	623	963	18	0	2.76 (2.52)	147 828.64
12	4 149	412	1 154	1 233	1 316	32	1	1.94 (2.10)	201 065.91
13	1 503	141	408	671	282	0	0	1.20 (1.07)	44 930.12
14	3 239	372	1 106	971	713	55	22	1.62 (2.15)	131 416.28
15	1 860	85	277	432	839	157	71	3.88 (3.22)	180 445.61
16	1 087	92	302	332	360	2	0	1.88 (1.88)	51 201.63
17	4 059	177	678	1 070	1 797	244	94	3.24 (2.90)	328 886.93
18	5 574	367	1 115	1 399	2 456	197	40	2.85 (2.64)	397 702.71
19	3 299	87	322	678	1 704	445	62	4.51 (3.06)	372 188.49
20	7 196	662	1 985	2 540	1 882	109	19	1.70 (1.95)	305 718.59
21	5 937	345	1 160	1 569	2 412	335	116	2.98 (2.86)	441 896.51
22	6 632	263	1 480	2 281	2 430	147	30	2.30 (2.28)	381 960.28
23	4 700	172	869	1 722	1 763	143	31	2.56 (2.46)	300 740.66
24	2 498	83	259	563	1 216	241	135	4.13 (3.24)	258 084.35
25	1 105	55	211	253	453	122	10	3.37 (3.04)	93 168.68
26	2 709	119	533	882	991	115	69	2.88 (2.90)	195 065.22
27	4 600	110	473	1 054	2 295	463	206	4.09 (3.18)	470 680.08
28	512	22	58	283	141	8	0	2.20 (2.10)	28 182.38
29	1 948	113	181	313	1 031	259	51	4.52 (3.13)	220 359.62
30	173	15	34	44	72	7	0	2.62 (2.60)	11 318.62
31	175	0	0	20	73	30	50	7.14 (3.65)	31 145.63
32	1 659	76	201	537	617	140	88	3.66 (3.30)	151 729.88
33	3 302	447	1 419	1 103	332	0	0	0.84 (1.00)	69 107.80

34	5 726	664	1 974	1 463	1 545	65	15	1.63 (1.99)	234 051.52
35	4 869	108	226	1 591	2 871	67	7	3.00 (1.95)	365 370.69
36	16 148	248	1 809	6 179	7 796	101	15	2.52 (1.87)	1 017 238.49
37	12 178	955	3 268	4 514	3 411	27	3	1.58 (1.52)	479 838.34
38	11 72	0	0	1	1 083	83	5	5.03 (1.78)	147 395.07
39	325	8	57	92	168	0	0	2.35 (1.75)	19 156.30
40	30	0	0	0	29	1	0	5.84 (1.42)	4 367.32
41	6 788	1 009	2 468	1 922	1 374	14	1	1.19 (1.45)	201 776.52
42	18 421	1 981	6 108	5 623	4 539	136	34	1.54 (1.85)	708 704.95
43	4 892	225	775	1 189	2 366	267	71	3.31 (2.80)	405 107.02
44	12 238	1 614	4 121	3 619	2 796	80	7	1.42 (1.76)	432 929.48
45	2 325	143	561	625	915	78	4	2.46 (2.46)	142 885.88
46	1 645	195	625	480	324	22	0	1.34 (1.73)	54 935.89

2.5 Discussion

Fine-scale mapping of coastal habitats and pressures

This study presents the first large-scale (1700 km of coastline) continuous map of coastal Mediterranean marine habitats. It confirms the importance of areas covered by seagrass [13,54,55] and completes the coralligenous habitat distribution recently mapped [56]. Marine habitat maps are important for marine ecology and essential for managing organizations [57–59]. More importantly, the fine scale (one pixel = 20 x 20 m and even 10 m x 10 m on Medtrix) and the large area (231 606 ha) covered are available at a relevant scale for the implementation of management and conservation measures in natura. The initial stage of a management plan is the description of the natural components of the environment and of potential pressures and threats they are faced with. This is essential for both the identification of management priorities and the design of action plans [57,60], but also for helping managers develop dialogue with other stakeholders. Three major types of spatial information are lacking in the Mediterranean, compared to other regions such as Australia and the USA: bathymetry, habitats, and species biodiversity distribution [61]. Our results are thus important and should be completed for the rest of the Mediterranean especially for the Eastern part where information is particularly lacking.

A heterogeneous spatial distribution of pressures

The map of cumulative human impacts highlights the widespread but heterogeneous distribution of pressures and their resulting impacts along the Mediterranean French coast. Around two thirds (68.5 %) of the areas are found within medium and medium high categories, an order of magnitude similar to the one found within the French national territory waters (52.9 %; [41]) and within the entire Mediterranean and Black sea (65.9 % subject to medium cumulative impact [41]). Further direct comparisons with previous findings [42,62] are difficult because they have considered a larger offshore waters area and have not presented data on coastal waters. However, at several sites, visual comparison between maps produced by both studies suggests that near-coastal and offshore situations seem to be concordant and complement one another. For example, comparing to Micheli et al's Ic [41] or Coll et al's coastal-based impacts [39,42,62], the Eastern part of the Rhône shows large-scale problems with strong Ic in shallow (present results) and offshore waters [41], or well-preserved Northern Corsica and Camargue (Western part of the Rhône) are subjected to weak Ic values whatever the study and thus distance from the coast. High and very high priority areas highlighted by Giakoumi et al [42] are large and roughly include several points West of Montpellier, extend from Marseille to Nice, a small western part of the Rhône and areas at along the North Western and the South-Western coast of Corsica. Interest areas with low or very low Ic as defined by this present study are covered by these priority areas except two: West of Montpellier where average Ic is medium-high and the Northern Corsican cape where average Ic are the smallest of Corsica. Despite the small local differences observed (due to the finer scale and/or the greater diversity of habitats considered in the present study) the high concordance observed between the findings make the results stronger and suggest that coastal impacts keep on spreading offshore. Concerning shallow near-coastal areas that were not included in previous studies [41,63], we confirm that pressures are mostly concentrated between 0 and -15 m where the most sensitive marine habitats are also developing; seagrass meadows for example show important regressions at this bathymetric level, especially because of artificial coastlines [64,65].

The most important pressures (considering both area and intensity) are urbanization, coastal population, coastal erosion and man-made coastline, which are directly related to coastal developments and territorial planning. Our results might help stakeholders prioritize their policy actions. Two regions should particularly draw attention: the bay of Villefranche-sur-mer (close to Nice) and Bonifacio (Southern cape of Corsica), respectively water bodies 31 and 40 presenting the highest mean I_c and variance. Urbanization is broadly the major pressure exerted but its impact depends on the sites because of the variety of the surrounding seabed nature (and thus vulnerability). For instance, the weight of the impact is lower around a largely urbanized site surrounded by soft bottoms than when by more vulnerable seagrass meadows. Where urbanization is not the major pressure, town-planner's attention should be drawn to the impact of man-made coastline that is particularly important, such as around the harbour of Marseille and its neighbour Fos-sur-mer (two major industrial and commercial harbour areas), but also on coastal erosion which occurs around the largest river, the Rhône. Similarly, coastal population is the major pressure affecting several areas known for their touristic attractiveness, although the seasonal population was not taken into account (the offshore bar between Agde and Sète, Palavas-les-Flots, Marseille, Frioul island, Cassis, Cannes, Porto Vecchio) and where urbanization remains paradoxically quite contained. These areas should be particularly kept under surveillance because urbanization is likely to increase there. Finally, agriculture mostly affects Corsican coasts, the last region where farmlands still remain in coastal areas. Corsica is actually a well-preserved island with the lowest population density of Metropolitan France (36.3 inhabitants/km² - INSEE data) and a wild coast appreciated by tourists. Nevertheless, Corsica had the highest French demographic growth since 2006 (1.3 % / year compared to 0.6 %/year at national level – INSEE data) especially around the two coastal cities of Ajaccio and Bastia. Corsica's coastal population pressure should thus particularly be monitored.

Our results show that two areas are particularly preserved from the analyzed anthropogenic pressures and should thus deserve particular attention and protection in the future: the Calanques of Marseille and the Northern cape of Corsica. Moreover, these regions present medium to high levels of biodiversities [39,40,66] and aesthetics [67]. The strong interest of these sites is indeed taken into account as a national marine park was created in the Calanques of Marseille in 2012 [68] and a project is under consideration within the Northern cape of Corsica since summer 2014 [69].

Threatened habitats

Because of the methodology, high I_c are associated with numerous pressures. Almost all habitats are affected by all pressures except the closest or the furthest from the coast (*Zostera marina* and *noltii* meadows, offshore rocks). This highlights the importance of coordination for action plans focusing on pressures threatening coastal marine habitats.

Principally derived from other damaged habitats, a relatively weak proportion of dead matte and artificial habitats are logically subjected to low cumulative impacts in comparison with other habitats. Indeed, dead matte is the biological remains of dead *P. oceanica* meadows and artificial habitat is a man-made habitat replacing natural ones. However, these habitats should not be abandoned because they are interesting substrates for restoration measures. For example, seedlings of *P. oceanica* transplanted on dead matte show a higher survival rate than on sand or shingle substrate [70,71]. Artificial structures may also be directly (artificial reefs, green infrastructures), or indirectly (i.e. colonization of pipelines) used for biodiversity management. For instance, several ongoing projects aiming at using artificial habitats (harbours, seawalls, groynes) to boost biodiversity might be more successful if local pressures are not too high [72,73].

Infralittoral shingle association is also largely submitted to medium and high cumulative impacts but for other reasons: it covers a relatively small area and it is localized at shallow depths (shingles carried by rivers are localized near the coastline) and where pressures are strong (major cities are located along the rocky coast). Finally, the most important cumulative pressures occur within the largest habitat, which is also the most sensitive: seagrass meadows. Seagrass meadows have to be a priority for management and conservation plans, being among the most efficient ecosystems considering the ecosystem services provided per surface unit [74]. Despite different protection measures *P. oceanica* meadows already beneficiate from (European Habitats Directive, Barcelona Convention, Bern Convention), they are still strongly damaged [64,75,76].

The deepest habitats (coralligenous habitat, circalittoral soft bottoms and offshore rocks) are relatively less subjected to pressures (among the ones considered for this study) than the other habitats because they are generally more distant from the coast. Although many pressures are known to impact these habitats [27,77], we show that they are mostly threatened by fishing in the present study. Even non-industrial fishing practices (traditional, recreational, spearfishing) can cause rapid and substantial negative effects as well as represent an important part of the total fish catches (30 % for example on the French Atlantic coasts) [78,79]. Despite the importance of this pressure, particularly on coralligenous habitats, very little actions have been taken to limit this recognized threat (medium Ic) [49] while outputs could be controlled and size or catch of fish limited.

Utility for management despite several limitations

According to Giakoumi et al [42], “a prerequisite to quantification of threats and effective implementation of conservation actions is the acquirement of fine scale spatial data”. Maps of marine habitats and of pressures that can impact these habitats are the basic knowledge necessary for management work (see the first paragraph of discussion). Developing and measuring indicators of water quality or of ecological status of habitats, also needs to locate and assess the pressures acting locally; these are generally roughly estimated [80–82] and our maps of pressures and cumulative impacts might help to refine these works. Knowing precisely where are sensitive habitats and how they are impacted is essential before the deployment of adequate mitigation measures [42]. Our work will also make easier the measurement of management action efficiency; for example once anchoring is targeted as the major impacting pressure in an area, managers can choose how and where they want to contend with: mooring prohibition, mooring buoys, access to maps for boaters (i.e. Donia application [44]). Our work may help stakeholder to prioritize their means: protect areas where cumulative impacts are low or very low or try to act on “controllable” pressures where cumulative impacts are medium to high. Finally, these maps are like photos of the state of the coast at a t time and might be done again in five to six years in order to see the eventual changes, compare similar sites and test the efficiency of different management choices.

Lastly, when interpreting the results, it is important to consider the data limitations and uncertainties inherent in this work. First, we assumed a linear relationship between pressure intensity and impact on habitat and ignored thus the existence of thresholds that certainly do exist. Like previous studies [41,42,62], we skipped the thresholds because there is a lack of information about them. Similarly, for the same reasons our analyses did not include eventual synergy or antagonism between pressures acting at the same place. Secondly, maps represent what we know in 2014; it is thus possible that some pressure or habitat is invisible on the maps because the information was unavailable at this time. Moreover, available data were collected during an extended period of time (four years) so they represent an average situation (besides without any seasonal variation) even if local managers easily communicate us their feed-back on the maps now available on line. Then detailed information are hard to access and thus data could be refined if they became available: use a finer grid size (presently 20 x 20 m) for pressures and avoid to degrade the presently available information for habitats (1:10 000 map), use more precise denominations for habitats (i. e. levels within habitat types, plant densities for *P. oceanica*, data on species assemblages). Similarly, in perspective, numerous new pressures may be added if the data (raw data or model outputs) are available: climate change, alien species, industrial fishing (including trawling), diving activities or marine traffic for example.

2.6 Conclusion

Our study provides the first maps of habitat and cumulative pressures distribution on the French Mediterranean coastline. These maps are now urgently needed for marine systems which are deteriorating faster than other ecosystems [2]. They will be very relevant to biodiversity conservation, to help communicate, prioritize environmental issues, make political choices, better understand the links between anthropogenic pressures and ecological status of coastal systems, and to apply and coordinate management measures. Combined with maps of species distribution they may be viewed as maps of potential biodiversity loss and will help evaluate the objectives of the European directives (MSFD, 2008/56/EC and WFD, 2000/60/EC). The maps we have produced here represent a single snap shot of marine habitats and pressures but they could easily be updated and used to fuel models to predict future impacts if appropriate scenarios are available [30]. On the long term, mixing these kind of extensive spatial mapping of habitats and pressures with ecological modeling will prove particularly useful because “long-term and large-area ecological processes are particularly poorly understood; and yet, in a number of areas, issues and well-defined policies have not been sufficiently developed” [2]. In particular, field data concerning ecosystem responses to pressures and thus the relationship between cumulative impact scores and ecosystem condition should now be considered a top priority [41].

Acknowledgments

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2.8 Supporting information

S4 text. Details and parameters of each anthropogenic pressure.

Methods

General Approach

We used a 4-step process cumulative impact model, an approach proposed by [1] and used then by [2,3]. First, spatial data were pooled for each anthropogenic pressure (Di) and each habitat (Ej). Second, a single and unitless scale was used for the pressures by log[X+1]-transforming and rescaling them between 0-1, and habitat data were converted into 20 x 20 m presence/absence layers. Third, pressure-by-habitat combinations (pressure layer multiplied with each habitat layer) were created for each 20 x 20 m marine cell and then multiplied by the appropriate vulnerability weighting variable (uij) (= relative impact of a pressure on an habitat) calculated on the basis of an expert survey [4]. The sum of these weighted pressure-by-habitat combinations represents the relative cumulative impact of human activities on all habitats in a particular 20 x 20 m cell. Finally, to make easier the visual display of results, according to [3,5], impacts undergone by marine habitats were classified in six ecologically meaningful categories for the Mediterranean sea and inspired by[1] on the basis of IC values: very high (Ic>10); high (8-10); medium high (2.1-8); medium (0.6-2.1); low (0.1-0.6); and very low impact (<0.1).

This study focuses on Mediterranean French coastal waters. A total of 10 pressures (Table S1) and 11 marine habitats spatial datasets were pooled and used. These pressures are assumed to capture different impacts endured by the coastal habitats. Maps can be viewed at www.medtrix.fr ("DONIA® expert" project for the habitats and "IMPACT" project for the pressures and Ic). We describe below the data layers used in the analyses. The pressures considered and their data source and resolution are summarized in the table S1.

Table S1. List and characteristics of spatial data layers representing pressures influencing marine ecosystems used to assess and map cumulative human impacts along the Mediterranean coastline.

Pressures	Resolution	Source
Coastal engineering	20 m	Meinesz, Chancollon, Garcia, & Markovic, 2010 MEDAM on www.medam.org
Boat anchoring	20 m	MEDOBS on www.medtrix.fr Satellite / aerial pictures
Aquaculture	20 m	2013 data base provided by French Water Agency and Ifremer Satellite / aerial pictures
Urban effluents	20 m	2013 data base provided by French Water Agency and Ifremer Outputs and capacity of discards on http://assainissement.developpement-durable.gouv.fr/
Industrial effluents	20 m	2013 data base provided by French Water Agency and Ifremer
Urbanization (land cover)	20 m	CORINE Land Cover
Agriculture (land cover)	20 m	CORINE Land Cover
Coastline erosion	20 m	CORINE Land Cover
Costal population (size and density)	20 m	INSEE
Fishing (traditional and recreational)	20 m	MEDOBS on www.medtrix.fr + unpublished field data between 2008 and 2014

Coastal engineering: harbours, artificial beaches and other reclamations from the sea

Coastal engineering cause modifications in initial conditions and disrupt local hydrodynamics and siltation [6]. Actually, costal engineering are reclamations from the sea (harbours, groins, man-made beaches,…) causing irreversible destructions of sublittoral seabed by covering and leading important changes in the hydrodynamics around these layouts [6]. The inventory of the reclamations from the sea along the French Mediterranean coast [7] showed 11.10 % of man-made coastline including 3.03 % irreversibly destroyed shallow (0-20 m) bottoms (http://www.medam.org). The impact distance is hard to estimate but studies concerning *Posidonia oceanica* seagrass beds, the most sensitive marine ecosystem in shallow Mediterranean waters, may help [8,9]. The ecological status of this ecosystem declines as the driver source comes closer, with an impact that may be visible along several kilometers [10]. A recent large-scale (800 km) study estimates impact distances from the spatial changes occurred between historical (1922) and present aerial pictures considering coastal development (layouts identified by the MEDAM database) regrouped in four categories. Our models use these mean maximal impact distances: (1) 15 km for very large harbours and installations (Marseille and Toulon) (2) 10 km for harbours (3) 3 km for ports of refuge, artificial beaches, (4) 1 km for pontoons, groins and landfills. Driver sources come from the MEDAM GIS layer (http://www.medam.org). A 20-m distance matrix is created. A negative exponential pressure curve is applied starting from the sources and decreasing between 100 and 0 % with the following equations: (1) $y=99.175e-0.304$ for big harbours; (2) $y=99.175e-0.456x$ for smaller harbours; (3) $y=99.175e-0.912x$ for artificial beaches and ports of refuge and (4)

$y=99.175e-0.912x$ for pontoons, groins and landfills. Bathymetry is taken into account with a theoretical loss of 10 % per 10-m depth group.

Anchoring

Boat anchoring (free mooring) may cause important damages to shallow ecosystems especially seagrass meadows [11] on which the impacts are more visible and studied. Anchors and their chains mechanically destroy the meadows in particular at their deeper limits where meadows are less dense and more sensitive [12]. *Posidonia oceanica*, the most common seagrass in Mediterranean sea is also the species that presents the lowest growth rate (1-10 cm per year) in the world [13,14]: it is thus particularly little resilient, making any damage irreversible at a human scale [8].

Anchorage GIS layer was created on the basis of MEDOBS data available at www.medtrix.fr (anchoring boats counted from a plane during summers, see more details on http://www.observatoire-mer.fr/en). Anchorage areas and the mean number of boats (with their size < or > 20 m long) observed during summer are completed with satellite and aerial [15] http://www.bing.com/maps/) pictures, and www.sea-seek.com. Each anchorage area is classified among one of three categories depending on the mean number of boats: 1) <100 boats, 2) > 100 and < 200 boats and 3) > 200 boats. An increase to the upper category is applied in areas counting big vessels (> 20 m long). A 20-m distance matrix is created. A one-kilometer buffer zone is created around the origins of sources with an exponentially decreasing impact depending on the categories and the following equations: 1) $y=41.084e-3.776x$, 2) $y=69.235e-4.079x$, 3) $y=99.175e-4.56x$. Bathymetry is taken into account by applying an increasing impact ($y=-0.004x3 + 0.3624x2 – 6.5008x + 22.681$) between 0 and – 5 m, a maximal impact between – 5 and -15 m (no buffer) and a decreasing impact between -15 and – 45 m with a theoretical loss of 10 % per 10-m depth group.

Aquaculture

Aquatic farms are used for the production of fish, mollusks, shellfish and aquatic plants. Food supply, high density of organisms and materials used may cause damages to the marine environment. [16] classifies the local potential negative impacts within five domains: introduced marine species, organic matters in the effluents, pathogen transfer, therapeutics and other products, genetic degradation of wild populations. The environmental impact of aquatic farms is the subject of numerous studies [17,18]. They show that an excess in organic matters and nutrients decreases the available luminosity by eutrophication. The degradation of these components leads then to the formation of sulphides causing a sedimentary hypoxia with a toxic effect on seagrass [19]. This negative effect is maintained even after the cessation of activity because of the destabilization and the accumulation of organic matters within the sediment [20]. The enrichment in organic matters is the most important just under the farms and may be detectable until 50 [17] or 84 m around [21]. An impact on seagrass meadows (especially on the vertical growth) is observed until 800 m around [22], and some authors recommend a minimal distance of 400 m between farms and meadows [19]. Nevertheless, an anormal increase in epiphytes is observed on seagrass even at 3 km of fish farms, but without any significant alteration of the meadows [23]. As for the benthic macrofauna, an impact is noted until a distance of 907 m but this distance strongly varies according to the sediment nature (rough or sludgy) with a tendency to be less detectable on rough soft bottoms preferred by seagrass [24].

Aquaculture GIS layer is created on the basis of a database provided in 2013 by the French Water Agency and Ifremer and completed with a research of new companies on the directory and of new sites on aerial pictures [15]. Each aquatic farm is classified within a category in function of its cover (< or > 3977 m^2) after the drawing of polygons on the map. The limit area is established using a Jenks natural breaks classification method. A 20-m distance matrix is created. An exponentially decreasing driver is modeled until 500 m for the smallest farms with the following equation $y=99.175e-9.119x$ and until 1000 m for bigger farms with the following equation: $y=99.175e-4.56x$. The influence of the bathymetry is considered from 0 to – 100 m according to a theoretical loss of 10 % per 10-m depth group.

Urban effluents

Urban effluents are generally given off via a pipe within a marine area characterized by a strong hydrodynamics able to guarantee their dilution. It results in a highly variable spatial influence of the effluents depending on the dilution and spread of contaminants by waves and currents [25]. These effluents have clear negative impacts on marine habitats particularly on *Posidonia oceanica* meadows [26]. The influence of effluents on the benthic macrofauna and on habitats was studied at different distances from the discard pipe et for different types of discards in the framework of annual surveys [27–31] and the modeling of plumes is sometimes realized [28,32,33]. A work comparing the areas covered by *P. oceanica* meadows before and after 90 years (1920-2010) of coastal development along a French region evidenced the importance and the localization of regressions in link with the building and the discarding of different pipes [34]. Another study has showed that seagrass beds are contaminated by heavy metals until at least 5 km from the driver source like at Canari in Corsica (France) [35]. Concerning the soft bottoms, a discard might influence the benthic assemblages until 8 km according to a negative exponential curve starting at the driver origin [36].

The GIS layer regrouping all waste water discard points is built on the basis of a database provided in 2013 by the French Water Agency and Ifremer. This is completed and corrected in function of different data sources: the pipes localized on 1:25 000 IGN maps, discards localized on the platform eau France (www.eaufrance.fr) and our field knowledge. The outputs (in m3/day) and the capacity (population equivalent) of each discard are obtained from http://assainissement.developpement-durable.gouv.fr/. Discards are classified into five categories according to their output: 0) ≤ 1 500, 1)]1 500; 5 000], 2)]5 000 ; 10 000], 3)]10 000; 50 000], 4) > 50 000 m3/day; and their capacity in population equivalent: 0) < 4 000, 1)]4 000; 10 000], 2)]10 000; 40 000], 3)]40 000; 100 000] and 4) < 100 000 population equivalent. Categories were chosen on the basis of the output and capacity distribution. A maximal impact distance is then given according to the mean note mixing the notes concerning capacity and output: 1 km for a note of 0 or 1 (a), 3 km for a note of 2 (b), 5 km for a note of 3 (c) and 10 km for a note of 4 (d). A 20-m distance matrix is created. An exponentially decreasing driver is modeled according to the following equations in function of the mean note obtain: $y=99.175e-4.56x$ (note=a), $y=99.175e-1.52x$ (note=b), $y=99.175e-0.912x$ (note=c), $y=99.175e-0.456x$ (note=d). Hydrodynamics is considered by subtracting the force vector of the current per pixel (obtained from the 2010-2011-2012 Previmer model (www.previmer.org)). Bathymetry (from SHOM) is considered with a theoretical loss of 10 % per 10-m depth group.

Industrial effluents

Industrial growth is often accompanied by an increase of contaminant discharge within the natural environment [37]. These may strongly impact entire ecosystems [38]. There is no information concerning contaminant fluxes to the sea at a large homogeneous scale within Mediterranean. In this absence, one of the most commonly used indicators of water quality is the COD (Chemical oxygen demand), an indirect measure of the amount of organic compounds in water. The GIS layer regrouping all the industrial discards within the sea and their COD value is built on the basis of a database made available by the French Water Agency and Ifremer in 2013. A 20-m distance matrix is created. Industrial effluents are assumed to exponentially decrease from their origin until 5 km for COD < 100 mg/l according to $y=99.175e-0.912x$, until 10 km according to $y=99.175e-0.456x$ for COD ranging between 100 and 1000 mg/l, and until 20 km according to $y=99.175e-0.228x$ (for COD > 1000 mg/l. Bathymetry (from SHOM) is considered in each model with a theoretical loss of 10 % per 10-m depth group.

Land cover: urbanization, agriculture and coastline erosion

Tourism and coastal urbanization are two intimately linked activities [39]. Urbanization represents the replacement of natural or semi-natural (farmlands) lands by buildings, roads, industries, etc. It avoids the provision of many ecosystem services [40]. Tourism plays an important role in the development of coastal areas; it necessitates important layouts like parking areas and roads located at less than 1 km from the coastline. Tourism developed along the coast causes two types of impact: 1) mass tourism and associated transports and 2) leisure transports. Tourism and urbanization know an exponential increase since 1950 [41]. Similarly, agriculture represents an important part of the potential impacts on the natural and in particular marine environment. [42–44]. Continuously increasing needs in food and energy push agriculture to use fertilizers and pesticides as artificial processes of optimization. Fertilizers brought during inadequate meteorological conditions and/or in quantities superior to what the plants are able to absorb, are taken away by the water and wind to the sea [45]. Eutrophication may then occur and leads to an over-production of phytoplankton and algae. Beyond the visual pollution due to an accumulation of algae on the beaches, it decreases the available oxygen for other aquatic organisms [46]. Similarly, pesticides are found in the marine environment with potential negative impacts on marine fauna and flora but also human health [47,48].

Urbanization and agriculture refer to land use. Different indices measuring anthropogenic pressures undergone by the marine environment use land cover data. Urbanization, agriculture and natural areas or coastal erosion are the most largely used [49] [3]. Data layers concerning urban and agricultural lands within coastal municipalities and on the erosion or aggradation of the coastline were directly extracted under GIS (polylines) from CORINE Land Cover database [50]. 20-m distance matrices are created. The influence on marine waters of urban areas on one part and agricultural lands on another part is assumed to exponentially decrease according to the following equation: $y=99.175e-0.456x$ until 10 km. Concerning the coastline erosion, it is assumed to exponentially decrease according to the following equations: $y=99.175e-0.912x$ until 5 km for an aggradations and $y=99.175e-1.52x$ until 3 km for the erosion. Bathymetry is considered in each model with a theoretical loss of 10 % per 10-m depth group.

Coastal population density

Population along the coast involves the presence of consumers, so an increased need in resources (water, energy, raw material) and in natural areas for recreational activities and the emission of varied discards in waters, soils and air [44]. [51] interested in identifying the impacts linked to the nautical frequentation, highlighted the important role of boats (anti-fooling, sacrificial anode, fumes) and the pollutions and nuisance caused by amateur yachtsmen (organic and bacterial pollution via direct discards, soaps and detergents, macrowaste, fishing, scuba-diving and spearfishing).

The GIS layer is composed of the different municipalities located within a 5-km radius from the sea. For each municipality, different data are added from the INSEE database (www.insee.fr): the population size and density in 2009, the number of hotel rooms in 2013, the number of campsites in 2013, number of secondary residences in 2010 and the percent of secondary residences in 2013. A note is attributed to each criterion in function of the following categories:

- Population size in 2009: (0) ≤ 500, (1)]500;2000], (2) >2000 inhabitants
- Population density in 2009: (0) ≤ 10 , (1)]10;30], (2)]30;80], (3)]80;300], (4)]300;2000], (5) >2000 inhabitants /km²

A 20-m distance matrix is created. An attenuation curve is assumed with a negative exponential decrease from the origin until 1 km for a global note (mean of the notes concerning population density and population size) of 0, 1 or 2 according to $y=99.175e-4.56x$, until 3 km according to $y=99.175e-1.52x$ for a note of 3, until 5 km according to $y=99.175e-0.912x$ for a note of 4 and until 20 km according to $y=99.175e-0.228x$ for a note of 5. Bathymetry is considered in each model with a theoretical loss of 10 % per 10-m depth group.

Fishing

Fishing causes direct and indirect effects on the marine environment. The European Union divides the directs effects into four categories [52]: 1) the impact on the target species and their size, 2) the impact on the non-target species (bycatch + ghost fishing + species depending on fished species), 3) impacts on the habitats and 4) impacts on mammals, birds and other vulnerable species. Indicators have been developed in order to measure the impact of fishing on fish communities.

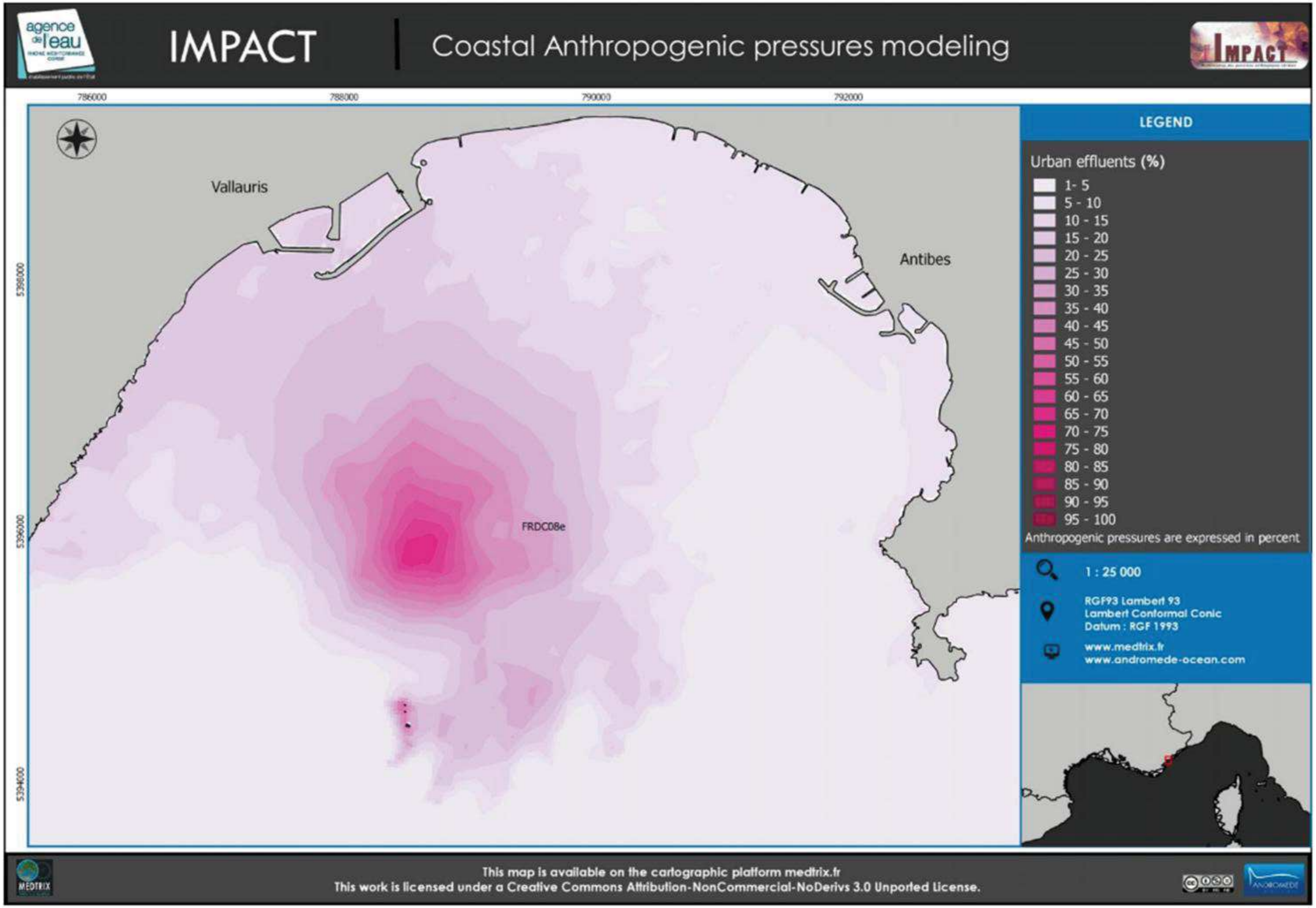
[53] showed that the most relevant indicators for management are those linked to the population level (exploitation rate, average catch size, for example), but that several indicators are necessary. The impact of fishing actually depends on the engine type used, the spatio-temporal intensity of the activity,

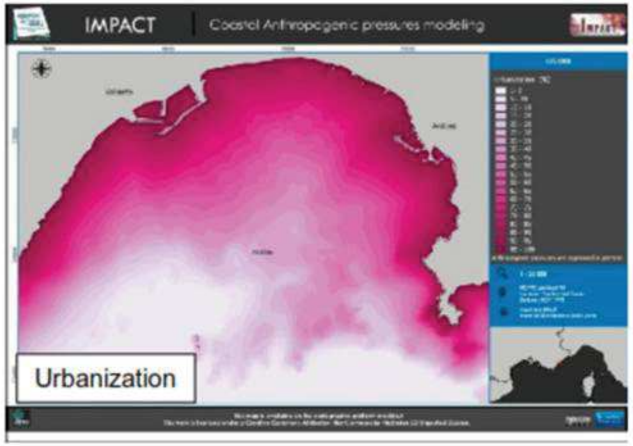
synergistic effects of others activities or the initial complexity level of the physical and biological environment [54]. For example trawling on *P. oceanica* seagrass meadows is recognized as major degradation source especially between -15 and -30 m [55]. Fishing engines may be subdivided into three categories: towed engines (trawling, dredging), passive engines (nets and drift nets, longline and drift longline, handline), and mobile engines (seine, downrigger tracking). Each one focuses on different type of fish and affects the environment by a different way [54,56,57].

The fishing GIS layer contains the localization of traditional and recreational fishing areas after cross referencing based on field data (Andromède océanologie, unpublished data) noting the position of buoy net along the coast and pontoons or groyns concentrating fishing activities, MEDOBS data (aerial pictures, http://www.observatoire-mer.fr/en/autres_pressions.html) localizing buoy nets and fishermen on the sea (unpublished data) and inquiries (data obtained from marine protected areas and managed marine areas). Field data were opportunistically acquired on the occasion of boat campaigns along the coast between 2008 and 2014 (around 100 days per year, 1/3 of the French coast per year). MEDOBS data were acquired between 2011 and 2014 with the help of a georeferenced sophisticated numeric video camera taken on a plane. The data acquisition was led between april and october at a rate of once per week during the summer and once every two weeks in spring and autumn.

A 20-m distance matrix is created. A negative exponential attenuation curve ($y=6E\text{-}05x3+0{,}0034x^2\text{-}0{,}0583x+3{,}5699$) is assumed with a decrease from –50 m to –120 m deep.

S5 File. Examples of detailed maps obtained during the study for the different anthropogenic pressures, marine habitats, cumulative impact scores (I_C) and cumulative impact categories. The Golfe of St Tropez is taken as an example. All the detailed maps are available: www.medtrix.


agence de l'eau
IMPACT
Coastal Anthropogenic pressures modeling
786000
788000
790000
792000
5398000
5396000
5394000
Vallauris
Antibes
FRDC08e
LEGEND
Urban effluents (%)
1- 5
5 - 10
10 - 15
15 - 20
20 - 25
25 - 30
30 - 35
35 - 40
40 - 45
45 - 50
50 - 55
55 - 60
60 - 65
65 - 70
70 - 75
75 - 80
80 - 85
85 - 90
90 - 95
95 - 100
Anthropogenic pressures are expressed in percent
1 : 25 000
RGF93 Lambert 93
Lambert Conformal Conic
Datum : RGF 1993
www.medtrix.fr
www.andromede-ocean.com
MEDTRIX
This map is available on the cartographic platform medtrix.fr
This work is licensed under a Creative Commons Attribution-NonCommercial-NoDerivs 3.0 Unported License.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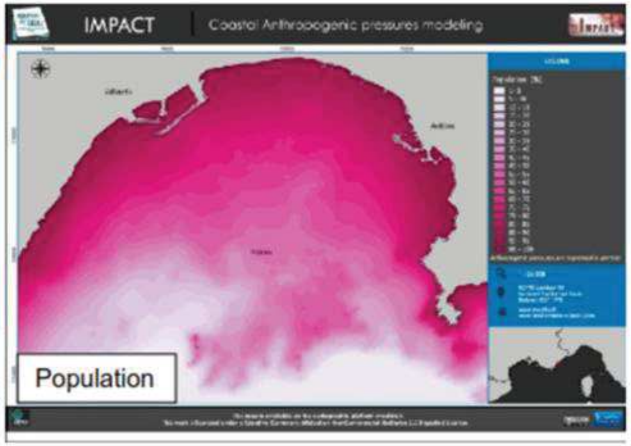
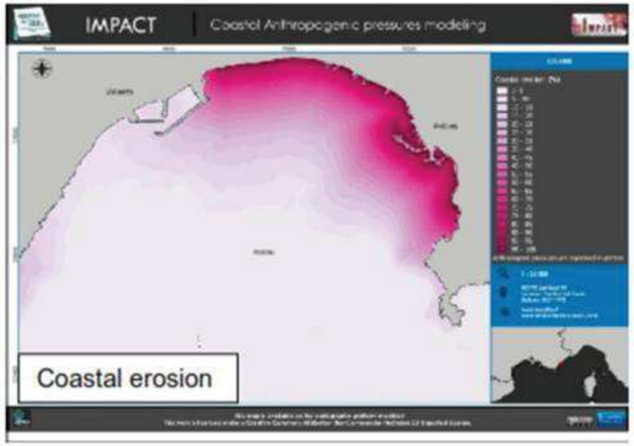
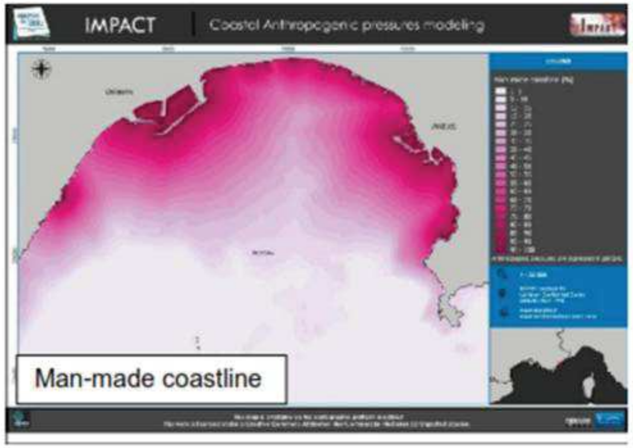
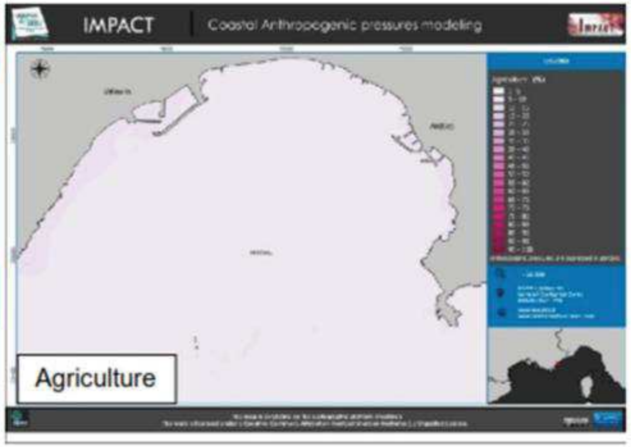
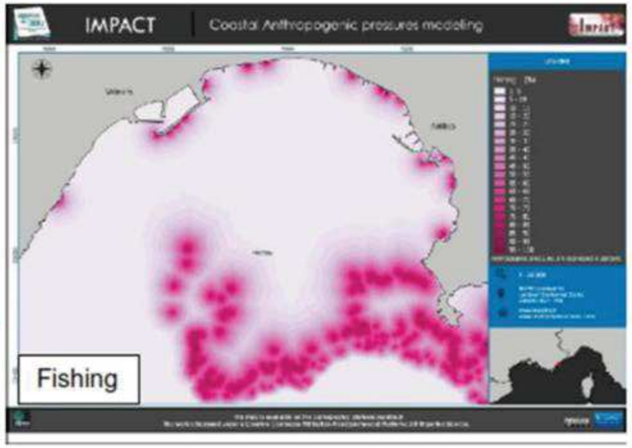
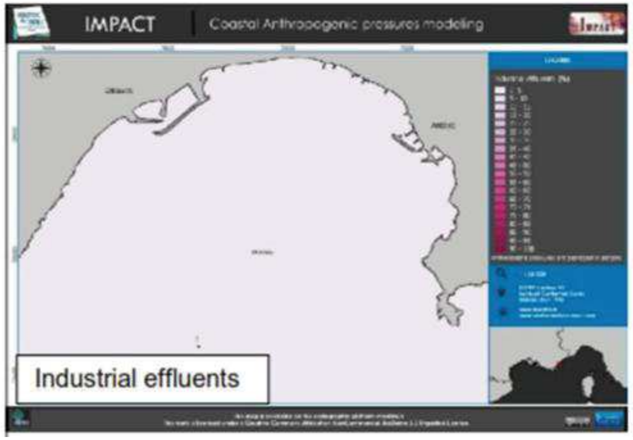

Population

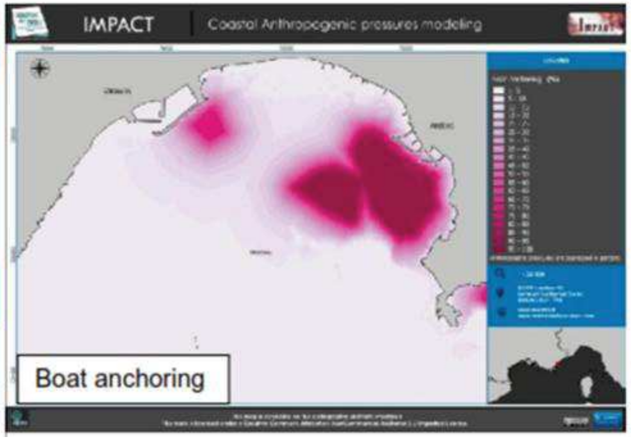

Coastal erosion

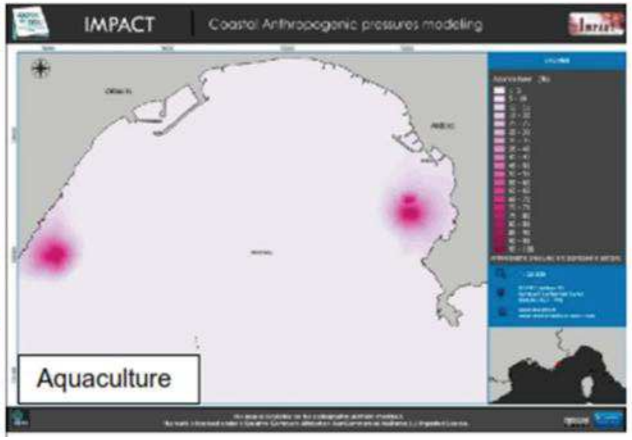

Man-made coastline

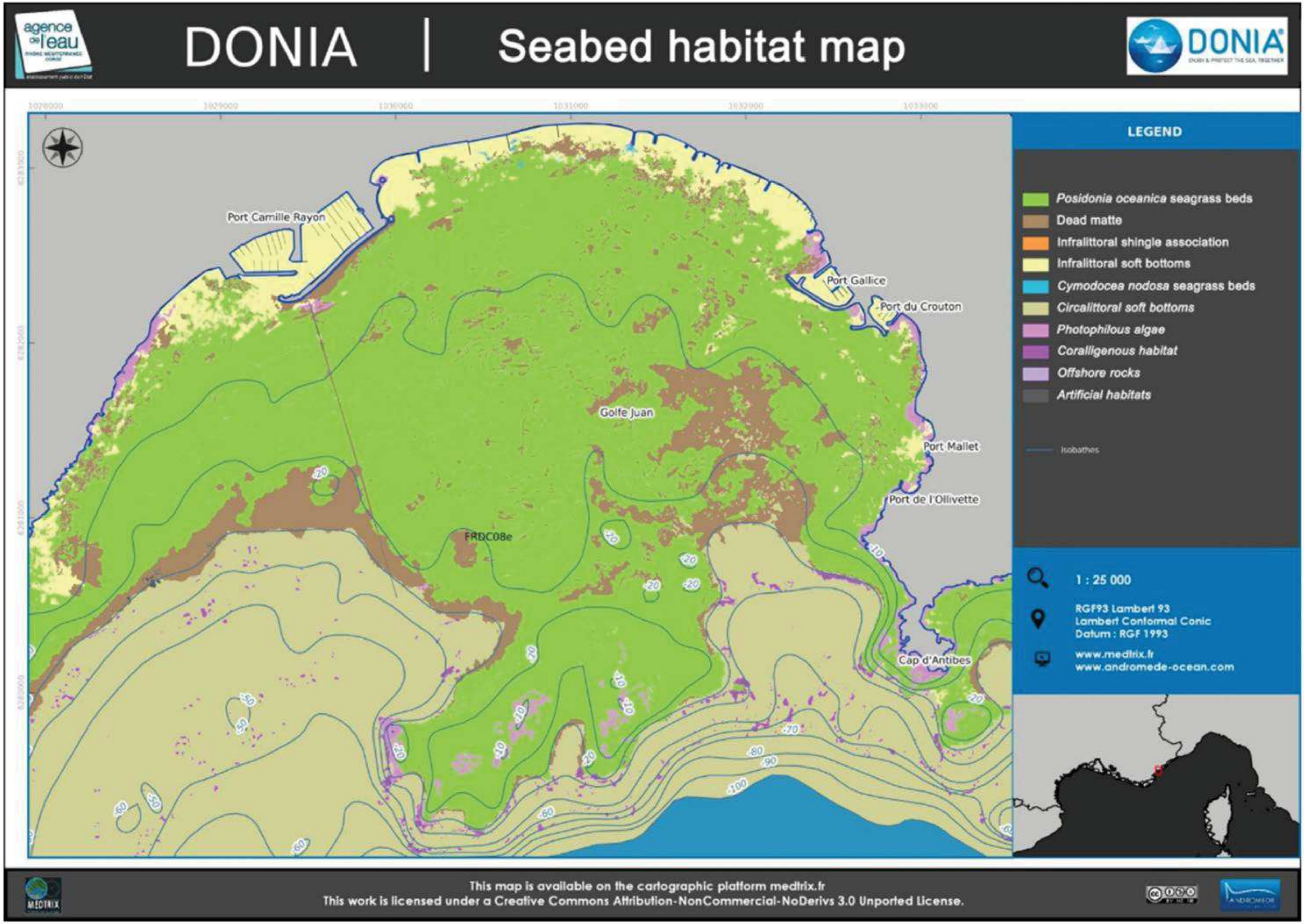

Agriculture


Fishing


Industrial effluents


Boat anchoring


Aquaculture



DONIA | Seabed habitat map

LEGEND

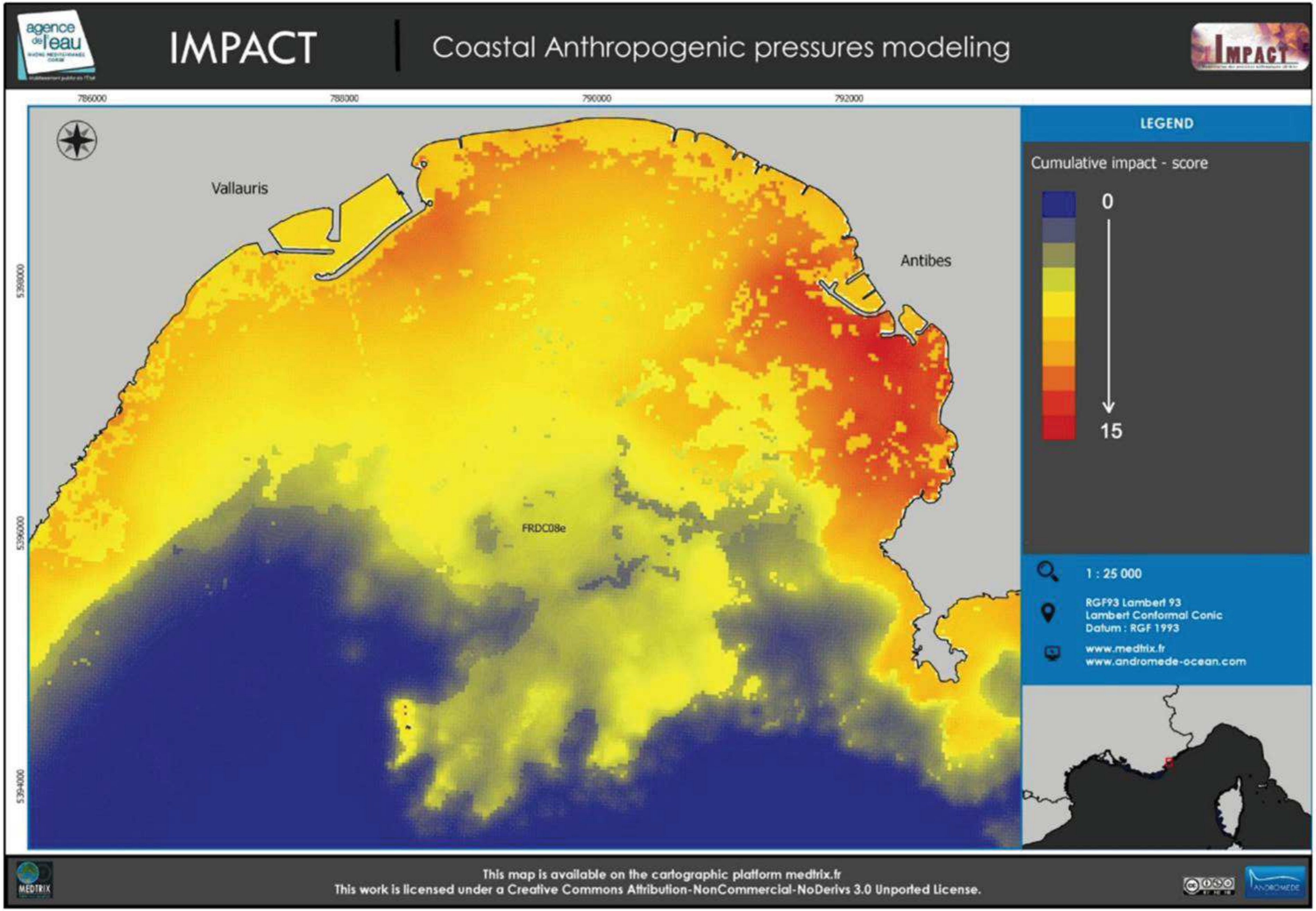
- 🟩 *Posidonia oceanica* seagrass beds
- 🟫 Dead matte
- 🟧 Infralittoral shingle association
- 🟨 Infralittoral soft bottoms
- 🟦 *Cymodocea nodosa* seagrass beds
- 🟨 *Circalittoral soft bottoms*
- 🟪 *Photophilous algae*
- 🟪 *Coralligenous habitat*
- 🟪 *Offshore rocks*
- ⬛ *Artificial habitats*

1 : 25 000

RGF93 Lambert 93
Lambert Conformal Conic
Datum : RGF 1993

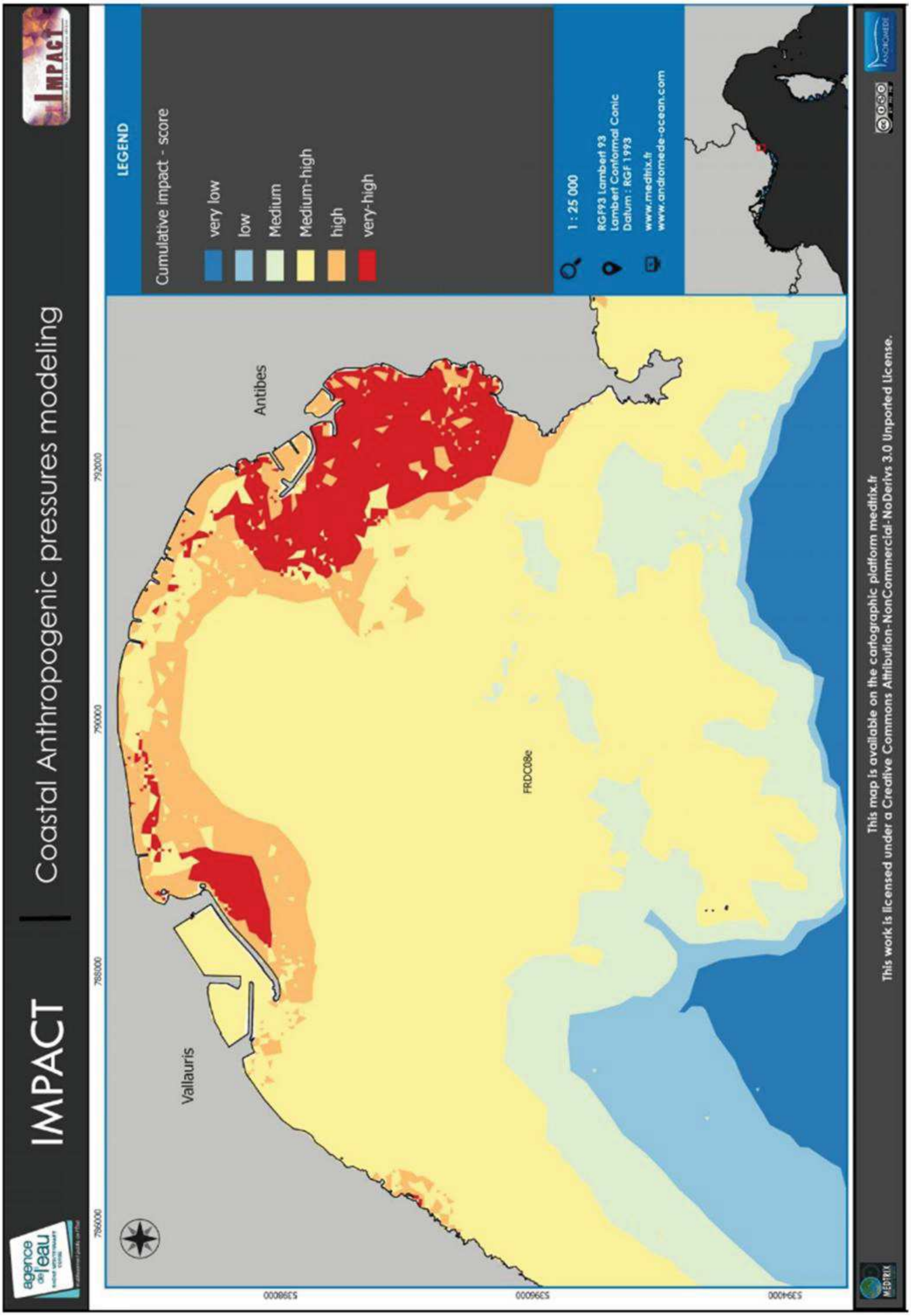
www.medtrix.fr
www.andromede-ocean.com

This map is available on the cartographic platform medtrix.fr



IMPACT | Coastal Anthropogenic pressures modeling

This map is available on the cartographic platform medtrix.fr


IMPACT
Coastal Anthropogenic pressures modeling
agence de l'eau
LEGEND
Cumulative impact - score
very low
low
Medium
Medium-high
high
very-high
1 : 25 000
RGF93 Lambert 93
Lambert Conformal Conic
Datum : RGF 1993
www.medtrix.fr
www.andromede-ocean.com
Vallauris
Antibes
FRDC08e
786000
788000
790000
792000
5398000
5396000
5394000
This map is available on the cartographic platform medtrix.fr
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MEDTRIX
ANDROMEDE

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3 Article 5 : Les pressions anthropiques côtières et les changements de phase expliquent l'état écologique d'un habitat marin: implications pour la gestion et la conservation

Coastal anthropogenic pressures and phase shift explain the ecological decline of a marine habitat : implications for management and conservation

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Authors: Holon Florian(a,b), Parravicini Valeriano(c), Mouquet Nicolas(b), Bockel Thomas(a), Boissery Pierre(d), Pierre Descamp(a), Tribot Anne-Sophie(a,b) and Deter Julie(a,b)
(a)Andromède Océanologie, 7 place Cassan, 34280 Carnon, France. Emails: florian.holon@andromede-ocean.com; gwenaelle.delaruelle@andromede-ocean.com; anneso.tribot@free.fr; julie.deter@andromede-ocean.com
(b)Institut des Sciences de l'Evolution (ISEM) - UMR 5554 CNRS – IRD – UM, Campus de l'Université de Montpellier, 34095 Montpellier cedex 5, France. Emails: nmouquet@univ-montp2.fr
(c)3CRIOBE, USR 3278 EPHE-CNRS-UPVD, LABEX Corail, University of Perpignan, Perpignan, France
(d)Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France. E-mail: pierre.boissery@eaurmc.fr

Abstract:

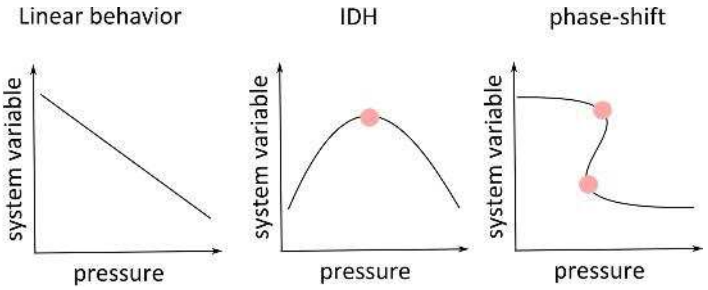
During the last half of the century, the development of coastal anthropogenic activities is at the origin of increasing pressures on marine coastal ecosystems. The management of those multiple and simultaneous threats requires reliable and precise data on the distribution of the pressures and of the most sensitive ecosystems. *Posidonia oceanica* seagrass beds (Tracheophyta) are protected but declining habitats encountered between the sea surface and a depth of 30-40 meters. We used Random Forest modeling to explain *P. oceanica* ecological decline (dead matte area / (seagrass beds + dead matte area) x 100) according to depth and ten anthropogenic pressures along the Mediterranean French coast (1700 km of coastline including Corsica). Three datasets with different work scales were tested: 20 x 20 m, 50 x 50 m and 100 x 100 m grid cells. Model accuracy was particularly good for the 50 x 50 m cells datasets (percent of explained variance = 77.5, Pearson correlation between predicted and observed values = 0.86). Depth, agriculture, man-made coastline, coastal population and urbanization were the predictor variables influencing the most the *P. oceanica* ecological decline. Thresholds (changepoints) were detected for all except two predictor variables (agriculture and coastal population). Using these thresholds values, the coastal seabed covered by seagrass and dead matte was classified into four categories: preserved area, vulnerable area, damaged area, deteriorated area. The two first categories covered the largest part of the study area (essentially in Corsica) while most of the South Eastern France was classified as “deteriorated area”. These maps bring important information to help managers preserving this essential ecosystem.

3.1 Introduction

Ecosystems are globally threatened by anthropogenic and climatic pressures (Halpern et al., 2008; Hoekstra et al., 2005; Jackson et al., 2001; Stachowitsch, 2003; Vitousek et al., 1997). Future socio-economical projections predict that a significant demographic growth (from 7 to 12 billion people by 2100 according to the worst Shared Socioeconomic Pathways scenario) will add to climatic perturbations (such as temperature increase) and ocean acidification (Ebi et al., 2014). The increasing impact of humans on ecosystems is thus expected to be accompanied by an increasing demand of ecological services (e.g. production of edible biomass or nutrient cycling). In such context, concerns are globally emerging about our capacity to balance natural ecosystem management between human impact, ecosystem status and the provision of services. These concerns are particularly justified for coastal ecosystems which concentrate high marine biodiversity, but also the vast majority of human population on Earth (Halpern et al., 2008). The development of new predictive tools and concepts to support decision makers in maintaining healthy coastal ecosystem status and high services provision despite increasing pressures are therefore urgently needed.

Since the formulation of the intermediate disturbance hypothesis, the relationship between pressures intensity and the status of ecosystem is largely acknowledged (Wilkinson, 1999). Other well-known examples include the 'phase shift' (or regime shift) which implies a dramatic change from a healthy to a degraded ecosystem status after a tipping point is reached (Hughes, 1994). The existence of a non-linearity in ecosystem response to disturbance certainly adds complexity and challenges for the development of predictive statistical tools, however, on the other hand non-linearity opens new possibilities for management if discontinuities and tipping points are identified (Folke et al., 2004). Indeed, the same variation in pressure intensity may have negligible or dramatic effect on ecosystems according to the nature of the system-pressure relationship and to the position of the ecosystem status relative to the discontinuity points (Figure 1). The development of tools able to quantify the nature of the pressure-system relationship and the relative distance to discontinuity points is therefore essential to inform managers about the risk and the implications of their decisions (Graham et al., 2015).

Figure 1. Different ecosystems responses to disturbance (pressure) according to a linear behavior, the intermediate disturbance hypothesis and a phase-shift.



In this study, we developed a spatially explicit statistical approach to characterize the shape of the system-pressure relationship for multiple pressures, identify discontinuities and quantify and map the relative position of ecosystems to these discontinuities. The Mediterranean seagrass meadow (*Posidonia oceanica*) is chosen as a model system because of its relevance in terms of the services provided to humans and its inherent vulnerability to coastal activities (Campagne et al., 2015). *Posidonia oceanica* is a protected plant (Pergent et al., 2010) which forms extensive meadows from the surface to 30 - 40 m depth (depending on water transparency and temperature). This ecosystem plays important ecological (nursery, spawning, feeding, oxygenation) and economic roles (coastal protection and sediment trapping) (Borum et al., 2004; Boudouresque et al., 2012), but it is threatened by many human activities such as man-made coastline, anchoring, or wastewater releases (Boudouresque et al., 2012; Marbà et al., 2014). The global decline (loss of area) generally accepted for *P. oceanica* over the last 100 years has been evaluated to 10% (Boudouresque et al., 2012). A recent study led along the French South-Eastern coast specified that 73% of the shallow limits had declined during the last 85 years while lost areas accounted for 13% (Holon et al., 2015b). Lost areas were mainly found along man-made coastlines such as harbours (Holon et al., 2015a). Coastal infrastructures were also recently recognized as a major threat for *P. oceanica* food web (Giakoumi et al., 2015).

Considering the extensive field data collected at high resolution, this study aims at proposing a framework to quantify the role of multiple anthropogenic pressures in a coastal ecosystem decline (bad ecological status) and to use such information to highlight management priority areas according to their relative position to potential discontinuities. We used a fine resolution (1:10000 map) dataset covering the entire French Mediterranean coastline, combining the distribution of *P. oceanica* and 10 anthropogenic pressures in a statistical modelling framework. Our approach comprises four main steps: (1) the GIS (geographical information system) mapping of human pressures and their intensities at three scales, (2) the GIS mapping of marine ecosystem decline, (3) the modeling of the relationships between human pressures distribution and marine ecosystem decline, and the choice of the better scale, (4) the use of the best model to build maps of management priority according to pressures thresholds detected in (3).

3.2 Materials and Methods

3.2.1 Study area and seagrass beds maps

The study considers the entire French Mediterranean coastline (1700 km including Corsica) at a 0-40 m depth, the bathymetric range of *Posidonia oceanica* in France (Boudouresque et al., 2012). Two ecosystem states were considered: living *P. oceanica* seagrass beds and dead matte covering respectively 70 641 ha and 5693 ha of seabed (Holon et al., 2015). The 1:10 000 map of these marine habitats and bathymetric data were obtained during a previous work and are freely available on www.medtrix.fr (DONIA expert project, see Holon et al. (2015a) and Holon et al. (2015b) for details concerning the data and the map building). The map was rasterized using three different grid cell sizes in order to find the scale which permits to obtain the best model: 20 x 20 m, 50 x 50 m and 100 x 100 m. For each cell size dataset, the ecological decline of *P. oceanica* meadows was calculated as the rate (in %) of dead matte cover (interpreted as a decline rate) per grid cell, using the following formula for each cell: D = dead matte area / (seagrass beds area + dead matte area) x 100 inspired from (Moreno et al., 2001). Only cells where the ecological decline value was superior to zero within each grid cell were taken into account for the analyzes.

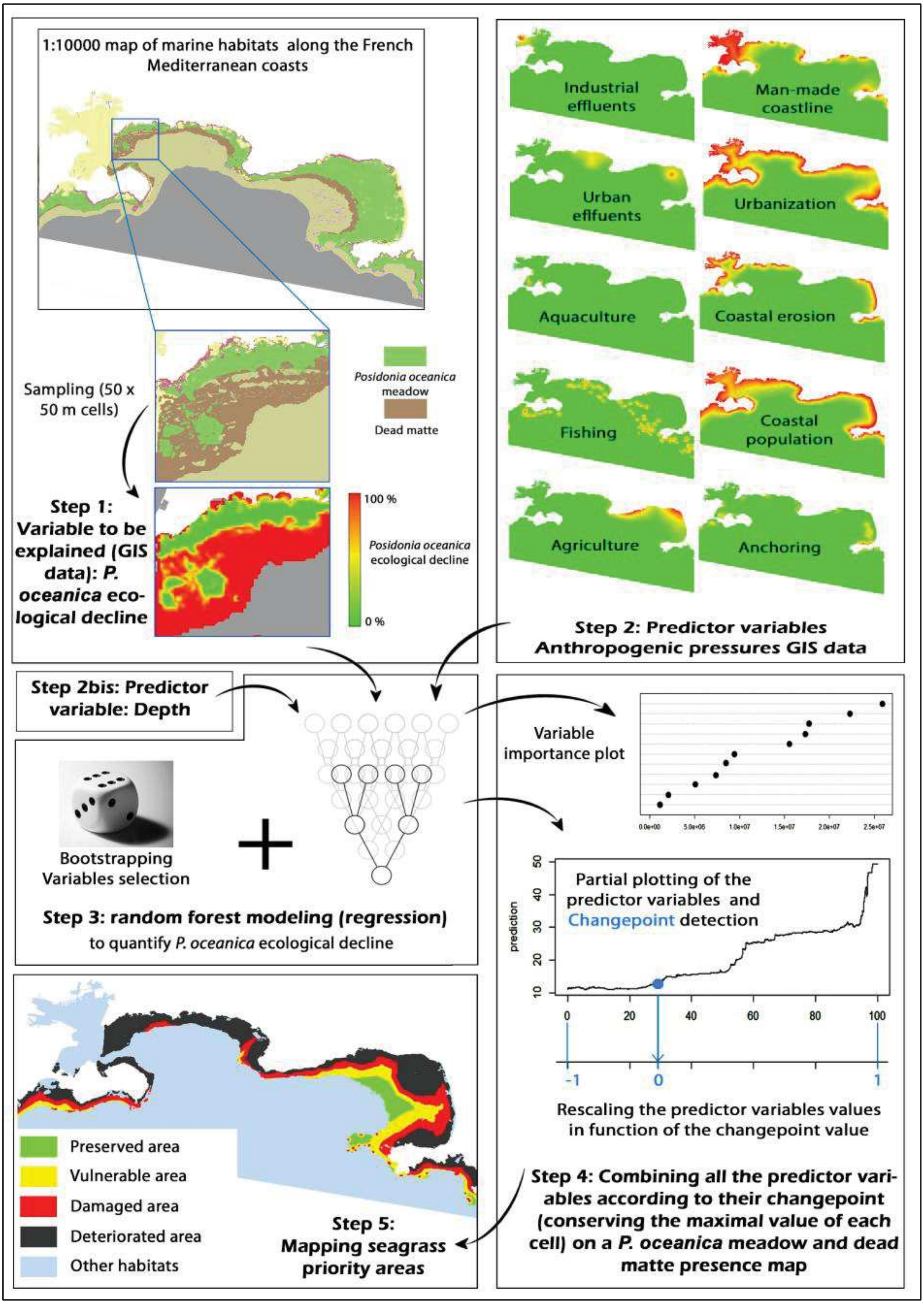
3.2.2 Anthropogenic pressures

After a literature review, we considered ten relevant pressures for the study of seagrass beds. The ten pressures taken into account were: (1) agriculture (land cover), (2) aquaculture (total area of the farms), (3) coastal erosion (land cover), (4) industrial effluents (chemical oxygen demand), (5) man-made coastline (big harbours / harbours / artificial beaches, ports of refuge / pontoons, groynes, landfills and seawalls areas), (6) boat anchoring (number and size of boats observed during summer), (7) fishing (traditional and recreational fishing areas), (8) coastal population (size and density considering the inhabitants-residents), (9) urban effluents (capacity, output), (10) urbanization (land cover). For each 20 x 20 m grid cell, the pressure values ranged from 0 (no pressure) to 100 % (spatial origin of the pressure). Details concerning the data and the map building are described in Holon et al.(2015b). The map is freely available from www.medtrix.fr (IMPACT project). Two additive layers with larger scales (50 x 50 m and 100 x 100 m grid cells) were built on the basis of this map.

3.2.3 Link between *Posidonia oceanica* ecological decline and predictor variables

The quantitative data concerning *P. oceanica* ecological decline, depth, and anthropogenic pressures were sampled from the different layers for each cell (Fig 2). Depth was considered as a variable buffering the pressure impacts. Depth also acts on *P. oceanica* presence and vitality through its impact on the penetration of light into the water column, water temperature, water column mixing and sedimentation process (Boudouresque et al., 2012, 2009a). All statistical analyzes were performed with R 3.0.2. (R Development Core Team, 2014).

Figure 2. Overview of the processing steps followed in this study. Explanations are given throughout the text in section 2.



P. oceanica ecological decline was modeled as a function of depth and anthropogenic pressures using Random Forests (RF, R package “randomForest” by Prasad et al., (2006) and Liaw and Wiener, (2002)) (Breiman, 2001; D. Richard Cutler et al., 2007). RF is a machine learning method that builds a set of classification or regression trees. Trees are built using a bootstrap sample of the observed data and a random set of predictor variables in order to decide the best split at each tree node. Trees are grown to maximum size without pruning and aggregation of trees is performed by averaging (Breiman, 2001; D. Richard Cutler et al., 2007). The estimation of response values is performed using the withheld out-of-bag observations (Richard Cutler et al., 2007; Prasad et al., 2006). The model variance explained is assessed based on the accuracy of the prediction of ‘out-of-bag data’. Random forests have been found to be ideally suited to ecological data as they do not require linear relationships, they effectively model variable interactions, can handle missing data and correlated variables, are more stable than traditional regression trees to minor changes in input data, and have high predictive power (Breiman, 2001; Breiman et al., 2013; Catherine et al., 2010; D Richard Cutler et al., 2007; Parravicini et al., 2012; Prasad et al., 2006). The choice of RF building parameters was optimized using the R “caret” package (Kuhn, 2008): RF were built using 1000 trees to allow stabilization of out-of-bag error and testing randomly seven potential splitting variables at each node.

3.2.4 Choice of the scale

For each dataset (20 x 20 m, 50 x 50 m and 100 x 100 m grid cell sizes) a model was built and the prediction capacity of the three models were compared. For each model, the percentage of explained variance and the Pearson correlation between predicted (prediction forest built on 80 % of the dataset) and observed (20 % remaining data) values were calculated. The dataset for which the model presented the best tradeoff between explained variance and correlation was used thereafter.

3.2.5 Estimation of the relative influence of the predictor variables on *P. oceanica* ecological decline

In RF, the importance of a predictor variable is quantified by comparing the accuracy of the model predictions using the original variable with the accuracy of the same model using a randomly permuted variable (Siroky, 2009). Two output metrics are used. The first, called IncMSE is a normalized comparison of the mean square error of model predictions with predictions generated using randomly permuted predictor values from the out-of-bag data (D. Richard Cutler et al., 2007). The second, called IncNodePurity, is the average total decrease in node impurity attributed to splitting on each measured variable using the residual sum of squares; it provides an indication of node prediction accuracy attributed to each variable. The relative importance of each quantitative predictor variable (depth and 10 anthropogenic pressures) on *P. oceanica* ecological decline was assessed through both of these metrics.

3.2.6 Thresholds detection

To characterize the shape of the system-pressure relationship, *P. oceanica* ecological decline was then plotted as a function of each predictor variable. Discontinuities = single changepoints (i. e. the point at which the statistical properties of a sequence of observations change) were detected using the "Changepoint" R package (Killick and Eckley, 2013) with the default method ("Amoc"). To make these changepoints (measured in percent) more concrete, the values were then translated into field data units (see paragraph 2.2) by applying the respective inverse equation used by Holon et al. (2015a) for each pressure (i.e. a pressure curve (type y = ae-bx) considering the distance to the source with a negative exponential shape ranging between 100% (origin) and 0% (no more impact)).

3.2.7 Maps building

Each variable (in percentage) was then scaled between -1 and 1 according to its changepoint value (changepoint value equal to 0) when a changepoint was detected and then projected on a map of *P. oceanica* meadow and dead matte presence. An additional map combining all the predictor variables according to their changepoint was also built (raster mosaic conserving the maximal value of each cell). For a better visualization and to make management decisions easier, the results were presented according to four categories: [-1 to -0.5] = preserved area,]-0.5 to 0] = vulnerable area,]0 to 0.5] = damaged area,]0.5 to 1] = deteriorated area.

3.3 Results

3.3.1 Workscale

While the models obtained with the two finest datasets showed similar percentages of explained variance (respectively 77.5 % and 76.6 % for the 20 x 20 and 50 x 50 m datasets, Table 1), the Pearson correlation value was higher with the 50 x 50 m dataset (Table 1). The model obtained with the 100 x 100 m dataset clearly showed a lower good prediction capacity (55.9 %, Table 1). The comparison between the prediction capacities of the models obtained with each of the three datasets led us to perform the rest of the analyzes using the 50 x 50 m grid cells dataset.

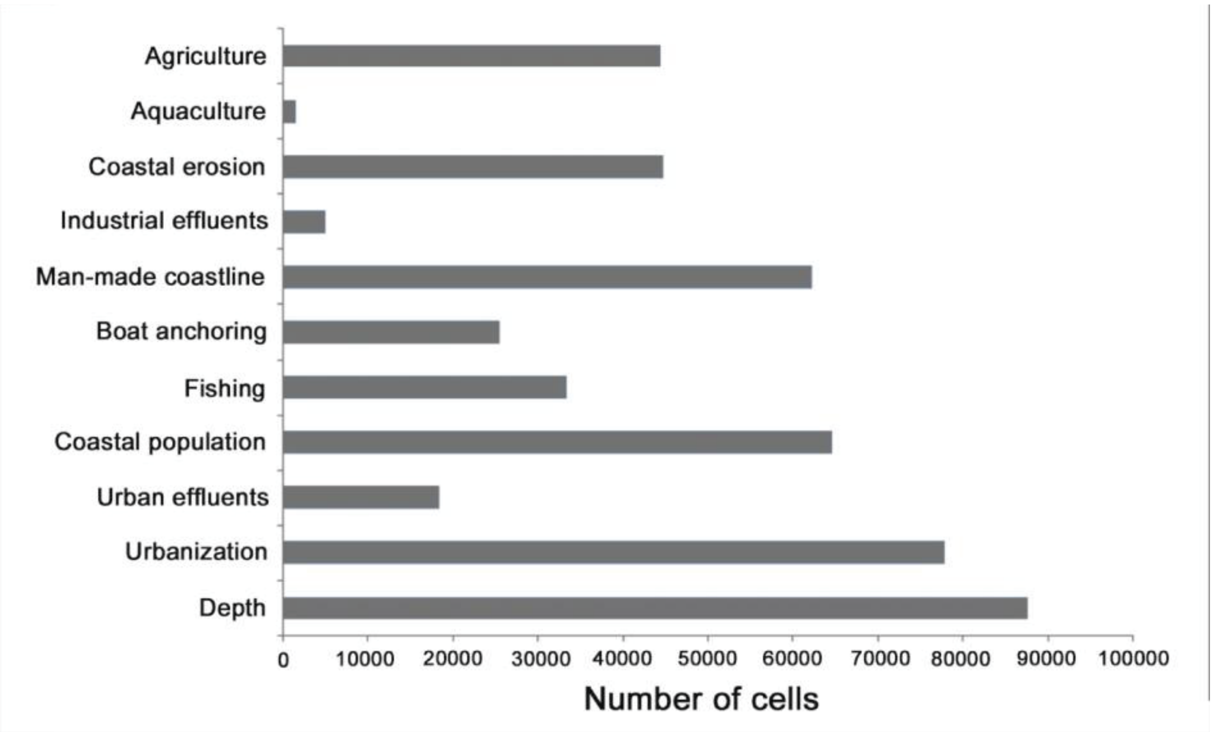
Table 1. Comparison of the models obtained with each dataset (20 x 20 m, 50 x 50 m and 100 x 100 m grid cell sizes) to explain *Posidonia oceanica* ecological decline with depth and the 10 anthropogenic pressures using Random Forests (RF, R package "randomForest") models (1000 trees, 7 splitting variables at each node). The Pearson correlation was calculated between predicted (prediction forest built on 80 % of the dataset) and observed (20 % remaining data) values. The numbers of considered cells (where *P. oceanica* ecological decline > 0) are indicated.

Dataset	Number of cells	Mean of squared residuals	Percent of explained variance	Pearson correlation
20 x 20 m grid cells	575800	335.20	77.50	0.81
50 x 50 m grid cells	89 767	370.32	76.65	0.86
100 x 100 m grid cells	18 421	740.71	55.94	0.74

3.3.2 Relative influence of the predictor variables on *Posidonia oceanica* ecological decline

Analyzing the 50 x 50 m grid cells map, three anthropogenic pressures occupied the highest numbers of cells: urbanization (87 % of the cells presenting a *P. oceanica* ecological decline superior to zero), coastal population (72 % of the cells) and man-made coastline (69 % of the cells) (Fig 3). On the contrary, aquaculture (2 % of the cells) and industrial effluents (6 % of the cells) only occupied a few cells (Fig 3).

Figure 3. Number of 50 x 50 m cells occupied by the different variables predictor used to model the *Posidonia oceanica* ecological decline. Only cells where the ecological decline value was superior to zero were taken into account. Total number of cells = 89 767.



The analysis of the average total decrease in node impurity and of the mean square error showed the relative importance of the variables to predict *P. oceanica* ecological decline. Depth, agriculture, man-made coastline, coastal population and urbanization were influencing *P. oceanica* ecological decline the most (Fig 4). Boat anchoring, industrial effluents and aquaculture were the least influencing variables (Fig 4). Results obtained directly with the mean square error were similar despite several small order inversions (e.g. depth with agriculture or man-made coastline with coastal population) (Fig 4). Individual partial plots showed how the predicted *P. oceanica* ecological decline varied as a function of the different predictor variables (Fig 5). Globally, all the predictor variables influenced positively the ecological decline of *P. oceanica* (the higher is the pressure, the higher is the dead matte cover). A small negative influence of agriculture, coastal population and fishing low values (between 0 and 18 %) was however observed (Fig 5B, 5D and 5F).

Figure 4. Importance of each of the 11 predictor variables ((1) agriculture (land cover), (2) aquaculture (total area of the farms), (3) coastal erosion (land cover), (4) industrial effluents (chemical oxygen demand), (5) man-made coastline (big harbours / harbours / artificial beaches, ports of refuge / pontoons, groynes, landfills and seawalls areas), (6) boat anchoring (number and size of boats observed during summer), (7) fishing (traditional and recreational fishing areas), (8) coastal population (size and density considering the inhabitants-residents), (9) urban effluents (capacity, output), (10) urbanization (land cover), (11) depth) in the model, in terms of mean square errors (IncMSE) and node prediction accuracy (IncNodePurity). The higher are IncMSE and IncNodePurity and the greater is the importance of the variable for the *P. oceanica* ecological decline.

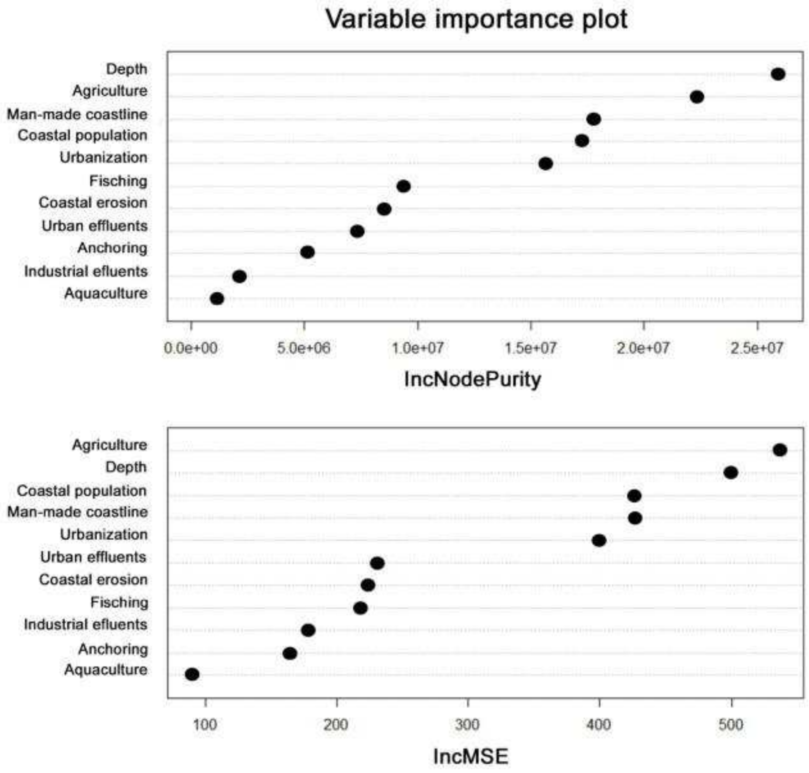
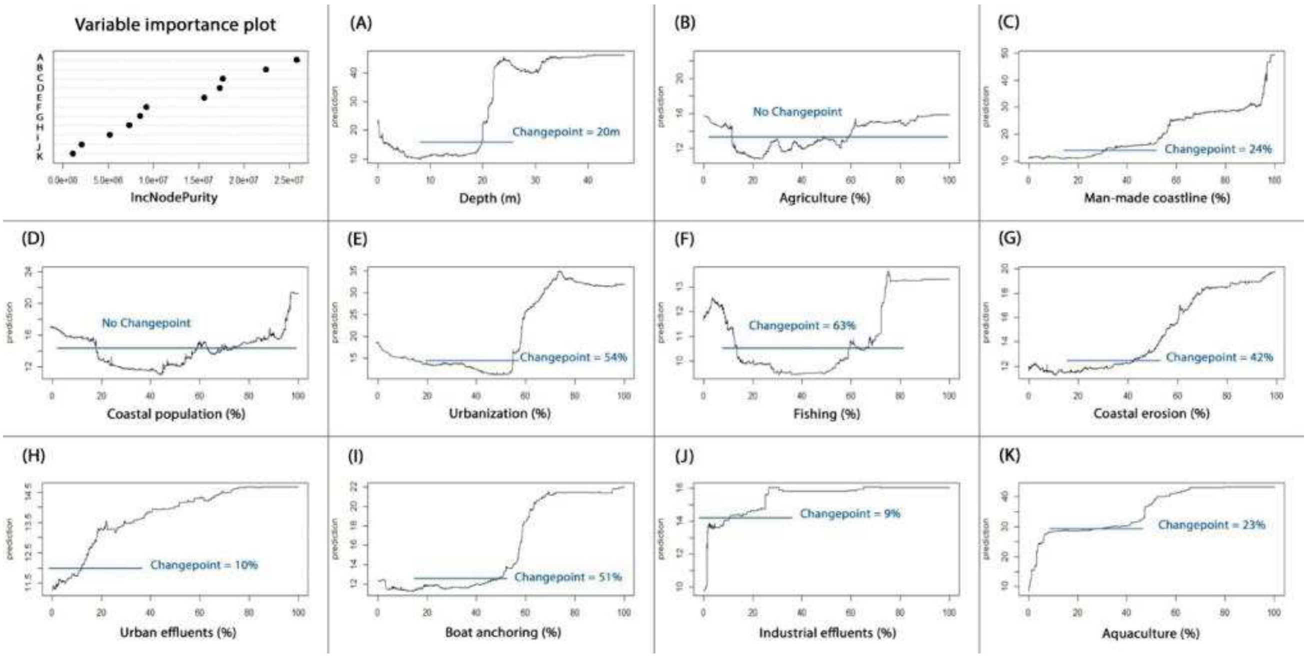


Figure 5. Partial dependence plots of the predicted *Posidonia oceanica* ecological decline (in %) as a function of each predictor variable (in %) through the Random Forest model. Note that for a better visualization, the y axis scale was adapted to each case. Single changepoints (i. e. the point at which the mean ecological declines value changes) are detected for each plot. For a better understanding, this changepoint value is changed into the initial data measurement unit (data used for the map building). The eleven predictor variables are: (A) depth (in m), (B) agriculture (land cover), (C) man-made coastline (big harbours / harbours / artificial beaches, ports of refuge / pontoons, groynes, landfills and seawalls areas), (D) coastal population (size and density considering the inhabitants-residents), (E) urbanization (land cover), (F) fishing (traditional and recreational fishing areas), (G) coastal erosion (land cover), (H) urban effluents (capacity, output), (I) boat anchoring (number and size of boats observed during summer), (J) industrial effluents (chemical oxygen demand), (K) aquaculture (total area of the farms).



3.3.3 Thresholds

Changepoints were detected for all variables except for agriculture and coastal population that showed only a positive linear relationship with *P. oceanica* ecological decline (respectively $y = 0.036 x + 12.77$ with $R^2 = 0.34$ and $y = 0.035 x + 13.27$ with $R^2 = 0.19$, Fig 5). *P. oceanica* ecological decline values clearly increased with depth beyond – 20 m (Fig 5A). The lowest threshold values (< 10 %) were observed for the effluents (urban and industrial) but remained associated to low ecological decline values (12 and 14 %, Fig 5H and 5J). Fishing was also associated to low ecological decline values (18 % max) but with a higher changepoint value (63 %, Fig 5F). Changepoints detected for man-made coastline and urbanization were particularly interesting as the ecological decline value respectively varied from 15 to 50% and from 15 to 35 % beyond the thresholds 24% and 54 % (Fig 5C and 5E). To a lesser extent, coastal erosion and boat anchoring presented quiet similar patterns with ecological decline values increasing from 10 to 20 % beyond threshold of respectively 42 and 54 % (Fig 5G and 5I). Aquaculture showed a strong positive influence (ecological decline increased from 10 to 30 % and then to 43 %) especially for low aquaculture values (< 23 %, Fig 5K).

To make the results more concrete, the correspondence between changepoints expressed above in percentage and "field values" was calculated. For example, a changepoint of 51 % estimated for boat anchoring corresponded to 100 boats per summer, meaning that the *P. oceanica* ecological decline started to increase beyond 100 boats per summer (Table 2). Similarly, man-made coastline, urban effluents and industrial effluents influenced the *P. oceanica* ecological decline to respectively 1.8 km (for a harbour), 1.7 km (for a 40 000-100 000 population equivalent discard) and 2.3 km (for effluents presenting a chemical oxygen demand ranging between 100 and 10 000 mg/l) (Table 2). Lower threshold distance were observed for urbanization (800 m), coastal erosion (450 m), aquaculture (90 m) and fishing (55 m) (Table 2).

Table 2. Correspondance between the anthropogenic pressures thresholds (changepoints detected from partial plots, in percent) and field data measurements at – 15 m (data used for the map building). The influence Changepoints were detected for partial plots modeled by Random Forests between *Posidonia oceanica* ecological decline in percent (=dead matte area / (seagrass beds + dead matte area) x 100) and anthropogenic pressures in percent (ranging from 0 (no pressure) to 100 % (spatial origin of the pressure)). No changepoint was detected for agriculture and coastal population. See (Holon et al., 2015a)for details concerning the data and the map building.

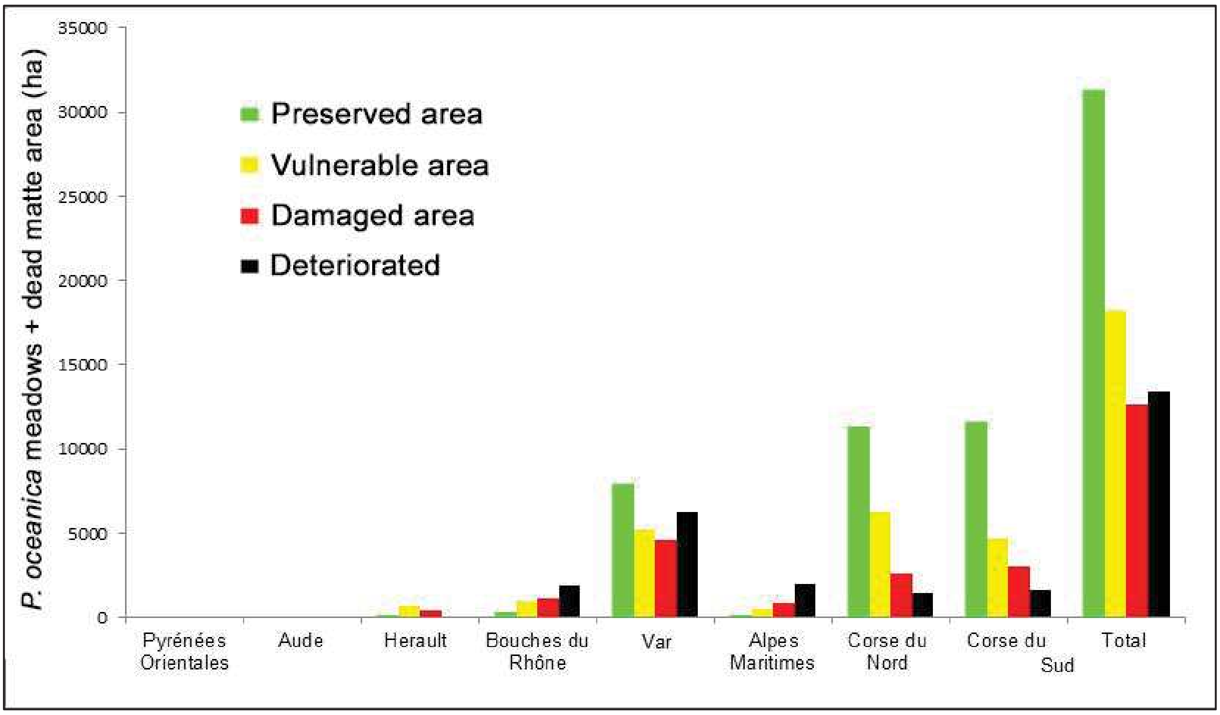
Anthropogenic pressure	Changepoint value in percent	Changepoint value in field data units
Man-made coastline	24	Distance of 1.8 km from a harbour
Urbanization	54	Distance of 800 m from a cell totally covered by urbanization
Fishing	63	Distance of 55 m from a traditional and recreational fishing area
Coastal erosion	42	Distance of 450 m from a cell totally covered by erosion
Urban effluents	10	Distance of 1.7 km from a discard whose capacity was 40 000 – 100 000 population equivalent
Boat anchoring	51	100 boats during summer
Industrial effluents	9	Distance of 2.3 km from effluents presenting a chemical oxygen demand ranging between 100 and 10 000 mg/l
Aquaculture	23	Distance of 90 m from an aquatic farm covering less than 3977 m²

3.3.4 Maps

General description

Combining all the pressures, most of the French *P. oceanica* meadows and dead matte were in a relative good state (49 580 ha = 31 348 ha to be preserved + 18 232 ha of vulnerable areas) (Fig 6). Only Var (French department) and Corsica areas presented to be more widely preserved than areas in other categories (Fig 6). Note that these zones in general presented the largest areas classified as deteriorated areas (respectively 6 289 ha and 10 159 ha). The largest vulnerable areas (10 983 ha) were also located in Corsica (Fig 6).

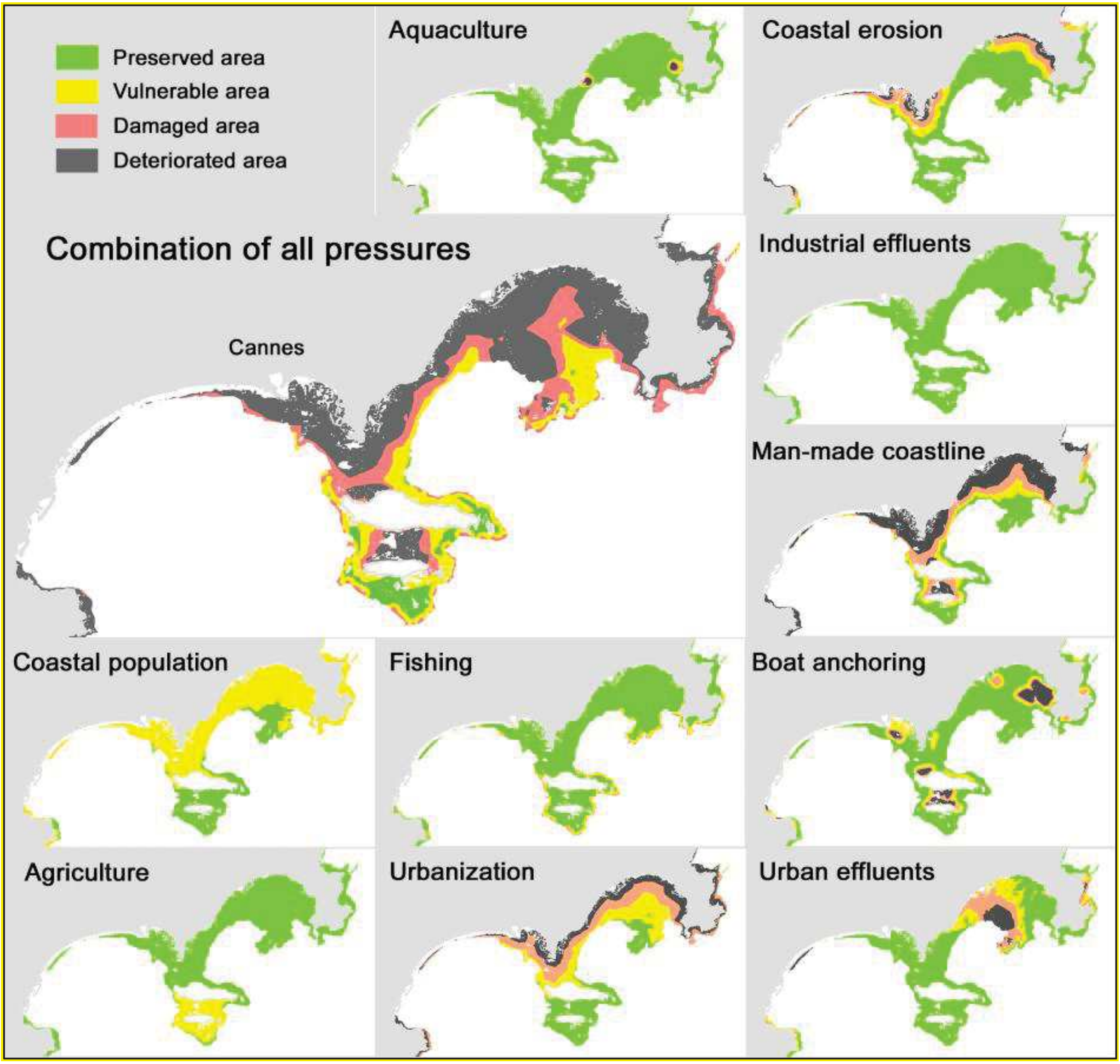
Figure 6. Classification of *Posidonia oceanica* meadows and dead matte areas (in ha) per French department (see Fig. 8 for their localization) and in total along the French Mediterranean coast as a function of the thresholds of the different pressures modeled (agriculture, man-made coastline, coastal population, urbanization, fishing, coastal erosion, urban effluents, boat anchoring, industrial effluents and aquaculture). Corsica is composed on Corse du Nord and Corse du Sud.



Threshold maps for individual pressures

Individual threshold maps for each pressure were constructed (detailed maps are available after free subscription on www.medtrix.fr, POP project, see Fig 7 for an example).

Figure 7. Examples of detailed maps classifying *Posidonia oceanica* meadows and dead matte depending on ten pressures (agriculture, man-made coastline, coastal population, urbanization, fishing, coastal erosion, urban effluents, boat anchoring, industrial effluents and aquaculture) according to their threshold (changepoint) values and their combination (raster mosaic conserving the maximal value of each cell). The Gulf of Cannes is taken as an example. All the detailed maps are available: www.medtrix.fr, POP project.

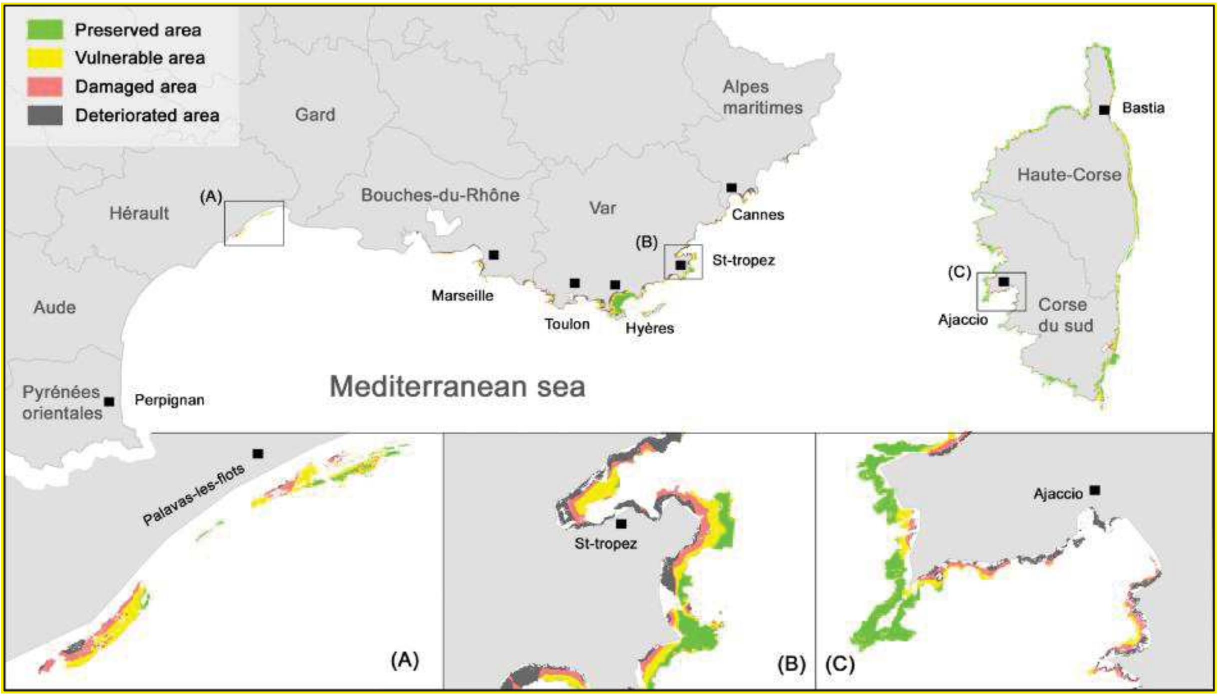


Meadows and dead matte subjected to maximal values of agriculture were observed in Corsica and along small islands while meadows and dead matte ranging between Marseille and St Tropez were submitted to maximal man-made coastline values. Along the mainland, all the meadows and dead matte located near the coast overtook the coastal population threshold except very small areas including islands. Many points corresponding to over-thresholds values of fishing were observed in the middle of meadows and dead matte along the Western part of the study area and along the Northern part of Corsica; elsewhere these points were located along the deepest limits of the ecosystem and were particularly numerous near Cannes and its islands. A few meadows and dead matte areas were associated to high coastal erosion values; most of them were on the Eastern part of the French mainland. Areas where urban effluents mostly influenced meadows and dead matte were located near the main Corsican cities and between Toulon and St Tropez while industrial effluents were particularly high on the Eastern part of Marseille. The most worrying areas for boat anchoring were near the islands off Hyères, St Tropez and Cannes. Finally, two areas presenting high decline values were observed for aquaculture off Cannes.

Threshold map combining all pressures

Combining all pressures, most of the meadows and dead matte located in close proximity of coastline corresponded to deteriorated areas (Fig 8 and detailed maps are available after free subscription on www.medtrix.fr, POP project). On the contrary, off areas were classified as “preserved areas” especially where meadows and dead matte occupied the largest areas (Fig 8). Rare are the preserved areas which entirely extended from the coastline to off waters: they were located around the islands off Hyères, South of St Tropez, along the Corsican Northern cape and at several places along the Corsican Western coast (Fig 8).

Figure 8. Map classifying *Posidonia oceanica* meadows and dead matte depending on a combination of ten pressures (agriculture, man-made coastline, coastal population, urbanization, fishing, coastal erosion, urban effluents, boat anchoring, industrial effluents and aquaculture) according to their threshold (changepoint) values (raster mosaic conserving the maximal value of each cell). French departments are indicated in grey and main cities in black. All the detailed maps are available at www.medtrix.fr, POP project. Three zooms are presented in three French departments: A) in Hérault, B) in Var, C) in Corse du Sud.



3.4 Discussion

3.4.1 An effective framework to detect and map threshold effects of multiple pressures

The objectives of this study were to quantify the effect of multiple anthropogenic pressures on the decline (bad ecological status) of a coastal ecosystem and to use such information to highlight management priority areas according to the identified thresholds. Accordingly we have mapped human pressures and their intensities but also *Posidonia oceanica* ecological decline estimated from maps of marine habitats. Fifty per fifty meters grid cells were enough to obtain a model with excellent performances. As expected, we showed that most pressures presented a non-linear negative effect on the studied ecosystem decline. Indeed, thresholds were detected and their values were highly variable depending on the pressure considered; this confirms that all the pressures do not have similar impacts and cannot be managed by the same way.

The map built on the basis of these results provides useful information for managers. Protecting marine biodiversity and the essential ecosystem services it supports is considered a top priority by different authorities including the Marine Strategy Framework Directive and the Convention on Biological Diversity (United Nations Environment Programme, 2014). Keeping marine resources in a sustainable state of use by achieving Good Environmental Status (GES) of EU's marine waters is precisely the main objective of the European Union's Marine Strategy Framework Directive (MSFD, 2008/56/EC). Our study proposes a simple methodology helping to achieve GES targets that could be easily reproduced for other ecosystems, given enough data are available. Note however that the combination of the thresholds should be adapted to which ecosystem is being considered. Here we have used the maximal values of each pressure because of the poor resilience of *P. oceanica* (Boudouresque et al., 2012; Marbà and Duarte, 1998; Marba et al., 1996; Pergent-Martini and Pasqualini, 2000). However, another combination like a sum moderated by the relative importance order of the predictor variables in the model would be less severe (it would decrease the "deteriorated areas") and could be more relevant for other, more resilient, ecosystems (eg soft bottoms).

3.4.2 Variables acting on *Posidonia oceanica* ecological decline

Our work showed that the environment (ie depth) is the first factor explaining *P. oceanica* ecological status. Depth, was the most important variable because the intensity of pressures were modeled with a decreasing shape according to increasing depths (Holon et al., 2015a) but also for ecological reasons. Depth shapes other variables (see 2.3) acting on meadows like temperature, light and sedimentation (Boudouresque et al., 2012, 2009a). The optimal depth (- 20 m) observed with the partial dependence plots corresponds well with the optimal growth found in the literature (Bellan-Santini et al., 1994; Boudouresque et al., 2012).

Concerning the anthropogenic pressures, man-made coastline was very frequent (almost continuous (in presence) but at a varying intensity along the coastline) and was, with agriculture, the most influencing predictor. One-off pressures like aquaculture or boat anchoring referred to local conditions, and were less frequent; their relative influences were weak in the model but may be very important locally to predict the decline (their presence are associated to high local decline values). These findings confirm previous results obtained from modeling in the region (Micheli et al., 2013) or elsewhere (Andersen et al., 2015; Ban et al., 2010) but contrasts with expert knowledge compiled for the entire ecosystem (Giakoumi et al., 2015). Pressures considered as major threats for Mediterranean seagrass (Giakoumi et al., 2015; Grech et al., 2012) were globally in agreement with the predictor variables influencing the most *P. oceanica* ecological decline: agriculture, man-made coastline, coastal population and urbanization. However, in the absence of empirical data, experts attributed very low certainty to the impacts of agriculture on *P. oceanica* (Giakoumi et al., 2015). Man-made coastline influenced the *P. oceanica* ecological decline to respectively 1.8 km (for a harbour) and urbanization to 800 m while the relation between coastal population and ecological decline tended to be linear. Harbours are the most damaging man-made coastal infrastructures with destroyed meadows found until 5 km (2.9 ± 5.2 m^2 destroyed for 1 m^2; built over 5 km) and a strong increasing impact during the first kilometer (Holon et al., 2015b). A '500 m' safety distance' is generally assumed for seagrass meadows (Cabaço et al., 2008; Pergent-Martini et al., 2006; Tuya et al., 2013) and all the threshold distances found here were over 500 m except for coastal erosion (450 m), aquaculture (90 m) and fishing (55 m). Indeed, even though trawling was not considered (Giakoumi et al., 2015; Grech et al., 2012), fishing occupied a relatively high importance in the model: it was associated to low ecological decline values (18 % max) but with a high threshold value (55 m).

Our results can be of important relevance to managers; the thresholds can be translated into concrete measurements and bring crucial issues like for example to which minimal distance waste water effluents or an aquatic farm must be built from a *P. oceanica* meadow in order not to impact it or which number of anchoring boats not to exceed in order to avoid irreversible damages within a bay. Associated to, still incomplete, ecosystem services data (Campagne et al., 2015b), these data will allow building or refining the bioeconomic (combing environmental and economic aspects) analyzes (eg the work of Rabassó and Hernández (2015) for fish farming) that are necessary to evaluate the cost/benefit of new infrastructures building and choose the least impacting strategy. Actually, most of the models interested in environmental impact assessment address the problem from an ecological perspective, concentrating on calculating emissions, their impact on the seabed or on the water column and their interactions with marine flora and fauna while economic aspects are the first arguments understood by the customers.

3.4.3 Building a decision support tool

At the interface between economic development and biological conservation, stakeholders need to know where to act in priority on a territory scale and on what (which target) to focus. Our results are in the continuity of previous spatial models recently developed in order to link multiple pressures with various coastal ecosystem status; they allow a predictive approach of the consequences of different management alternatives (Bianchi et al., 2012; Parravicini et al., 2012; Stelzenmüller et al., 2010; Vacchi et al., 2014). This represents an important decision support tool helping to choose efficient management solutions in the face of complex interactions and high uncertainty. Associated to large datasets, this approach would help to free ourselves from site-specific effects and make easier the extrapolation of general rules (Crain et al., 2008).

Zonation facilitates decision making on monitoring and sampling aspects. For example, developing and measuring indicators of water quality or habitat conservation, requires knowing the ecological status of surveyed ecosystems but also locating and assessing the pressures acting locally (Berg et al., 2015; Gobert et al., 2009; Hayes et al., 2015; Lopez y Royo and Pergent, 2008). Our map with the classification in four categories will also help to decide the actions to be taken depending on the category and site. For instance, we showed that preserved areas occupying the entire depth gradient (shallow to deeper parts) of the ecosystem are scarce along the mainland coastline: the protected islands off Hyères, the North of Corsica and the South of St Tropez. This continuity is yet important for numerous ecological process within marine seagrass beds (Boudouresque et al., 2012; Irlandi et al., 1995) and beyond (Hilty et al., 2006; Pendoley et al., 2014). Except their natural remoteness, no legal protection does exist for these areas except for the island off Hyères (Port-Cros and Porquerolles national park). Moreover, our maps by combining pressures with large and small spatial impacts could be used to better counter the weakness of present marine protected areas recognized as an absence of network, the choice of areas according to opportunistic reasons instead of ecological ones, the lack of law enforcement officials making sure that areas are really protected (Montefalcone et al., 2009; Roberts et al., 2003).

On the other hand, our map will help to decide the best mitigation measures to be taken locally in vulnerable areas in order to avoid exceeding thresholds. Damaged areas may also be targeted for mitigation measures to avoid tipping into the last category and try to come back under the threshold. In the same time they may also be preferred for restoration actions (or tests). *P. oceanica* meadow restoration is still very hard because the colonization of new areas and the recolonization of lost areas, via seeds, vegetative fragments or marginal spread of the meadow are extremely slow (horizontal growth is on average 1-6 cm/year (Boudouresque et al., 2012; Marbà and Duarte, 1998; Marba et al., 1996; Pergent-Martini and Pasqualini, 2000). Seedlings of *P. oceanica* transplanted on dead matte show a higher survival rate than on sand or shingle substrate (Balestri et al., 1998; Molenaar and Meinesz, 1995) and the advantages of rocky substrate were recently highlighted (Alagna et al., 2015; Badalamenti et al., 2015). Thanks to its spatial cross between ecological decline and pressures data, our result could help to highlight the best places for restoration and select the best parent-plants or propagules to be used. An absence of individuals with low and medium levels of genetic variation, was found at impacted (commercial shipping, organic pollution (pesticides) and cumulative impact) locations suggesting these individuals were less resistant and resilient (Jahnke et al., 2015). Next hopes of restoration enhancement would thus come from the genetic selection of well resistant seedlings possibly thanks to studies like this one.

While the final map combining the different predictors is interesting for a regional analysis (define areas with priority conservation issues or restoration capacities), local stakeholders and managers could also take advantage of the individual maps (pressure by pressure). For instance to decide on which pressure they should act in priority and what threshold not to exceed. Some pressures are relatively easy to modulate (fishing, anchoring, aquaculture or effluents in a lesser extent) while other are quiet unalterable (coastal population). Our study shows that an anchoring pressure over 100 boats in 2 500 m² per summer need to be reduced at risk of provoking irreversible damages; action means are for example mooring prohibition, mooring buoys or access to maps for boaters (Montefalcone et al., 2006; Okudan et al., 2011). However, it is important to integrate the very low resilience of *P. oceanica* meadow: even if efforts are done once an over-threshold pressure is targeted in an area, it will be hard to inverse the tendency (but see very small existing recolonization areas in Holon et al. (2015b)). Empirical data are needed in order to know until which dead matte percent a patchy meadow is able to recover if pressures are under thresholds. Similarly, it would be interesting to know the pressure threshold it would be necessary to reach back in order to allow new colonization of old dead matte.

3.4.4 Biases and perspectives

Lastly, when interpreting our results, it is important to consider the data limitations and uncertainties inherent to this work. Some biases are inherent to the datasets: the number of pressures (climate change, invasive species, industrial fishing (including trawling), diving activities or marine traffic for example were not tested because of a lack of data), the absence of synergy or antagonism between the pressures and the way used to spatially model the pressures depending on the bathymetry and the distance to the origin (see also Holon et al. (2015a)). Some biases are related to our underlying hypothesis, for instance decline was calculated as the ratio between dead matte and alive *P. oceanica* cover assuming that dead matte corresponds to seagrass decline. However this estimation does not necessarily relates to recent declines as dead matte may persist for millennium. Part of the recorded declines (especially beyond the lower limit) could thus be linked to the natural continuous rise in the sea level since 19,000 BP (Clark et al., 2009; Laborel et al., 1994; Morhange et al., 1996) and not to anthropogenic pressures (Boudouresque et al., 2009b; Vacchi et al., 2010). In our model, this part is explained by the depth and is assumed to be weak as we have limited our study area to the *P. oceanica* bathymetric range (i.e. 0 to -40 m).

As a direct perspective, a model with a high accuracy as the one obtained here could be used to predict *P. oceanica* ecological decline at the Mediterranean scale. Subject to pressure data availability, it could be tested along close coastal areas like Italy or Spain. The predictive model could also be used to build scenarios depending on expected changes in pressure data (e.g. increasing coastal population). Using 50 x 50 m data cells, our model well explains regional *P. oceanica* ecological decline but its predictive capacities at a bay scale should be tested; very local pressures (< 50 x 50 m) may occur and present a higher impact. This refining step is necessary before using our model to predict at a fine-scale the impact of a new infrastructure (e.g. harbour expansion within a bay). Similarly, a predictive model including more environmental variables like the sea surface temperature could be combined to the IPCC (Intergovernmental Panel on Climate Change, https://www.ipcc.ch/index.htm) climate changes projections and use to predict their impacts on *P. oceanica* meadows. Some of these impacts are already observed and are expected to increase in the future (Boudouresque et al., 2009a; Duarte, 2002; Jordà et al., 2012; Pergent et al., 2015, 2014).

3.5 Conclusion

This study develops a statistical model of the effect of the main human coastal pressures along the French Mediterranean coast and the ecological decline of the most important ecosystem in the region. Based on maps of marine habitats and maps of different pressures with 50 x 50 m grid cells, our model shows good performances to predict the ecological decline of *P. oceanica* meadows. Moreover, our study brings stakeholders and managers useful findings like pressure thresholds and prioritization maps; they will facilitate decision making concerning impact assessment, actions addressing specific threats, conservation but also monitoring and sampling aspects.

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4 Article 6 : Medtrix: une base de données cartographiques sur l'écologie marine et les pressions anthropiques le long des côtes méditerranéennes

Medtrix: a cartographic database concerning marine ecology and anthropogenic pressures along the Mediterranean coast

Authors: Holon Florian(a,b), Delaruelle Gwenaelle(a), Boissery Pierre(c), Pierre Descamp(a), and Deter Julie(a,b)
(a)Andromède Océanologie, 7 place Cassan, 34280 Carnon, France. Emails:
florian.holon@andromede-ocean.com; gwenaelle.delaruelle@andromede-ocean.com;
anneso.tribot@free.fr; julie.deter@andromede-ocean.com
(b)Institut des Sciences de l'Evolution (ISEM) - UMR 5554 CNRS – IRD – UM, Campus de l'Université de Montpellier, 34095 Montpellier cedex 5, France. Emails: nmouquet@univ-montp2.fr
(c)Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France. E-mail: pierre.boissery@eaurmc.fr

4.1 Abstract

An efficient management of marine resources necessitates large databases provided at a fine spatial resolution. Here we describe Medtrix (http://www.medtrix.fr), a cartographic platform that regroups different databases made available for marine professionals: scientists, managers and stakeholders. This meta-database is the first one available at such a good resolution (20 m for the anthropogenic pressures) all along the French Mediterranean coast and along some other countries (Tunisia, Italia). These data concern for now anthropogenic pressures (man-made coastline, boat anchoring, wastewater, coastal population density, aquaculture, land use), 1:5000 seabed habitat maps, marine mammals observations but also the monitoring of the two most important ecosystems in Mediterranean sea: *Posidonia oceanica* seagrass (presence/absence, vitality) and coralligenous habitat (presence/absence and diversity data). The platform proposes different functionalities like editing maps or directly comparing sites. One of these databases is already the basis of a management tool (DONIA® application) which helps yachtsmen to anchor in a safe (environment and security) way.

Key words: coastal water quality; ecological quality; seascape ecology; urbanization consequence; marine habitats; human-driven impacts modeling

4.2 Introduction

Conservation biology interests in addressing the biology of species, communities and ecosystems that are directly or indirectly perturbed by human activities or other agents in order to preserve biodiversity. A multidisciplinary science in essence, it appeals to population biology, environmental monitoring, biogeography but also management or public policy for example (Soulé 1985). Thus, it comes up against the difficulties to have available, at the same time, data from different fields, a spatial resolution relevant for management considerations and the appropriate analysis tools. This is even more complicate for marine ecosystems still more unknown than their terrestrial counterparts despite the widespread impacts of human activities on the oceans (Selig et al. 2014). Large-scale data are more and more available concerning biodiversity data (Kaschner et al. 2013) or human threats (shipping traffic and port data, pollutions (urban run-off, pesticides) (Halpern et al. 2008), invasive species (Lasram and Mouillot 2009), aquaculture (Coll et al. 2011) or cumulative impacts (Micheli et al. 2013) but the resolution remains low (mostly 0.5 ° =75 km or sometimes 0.1° =15 km). The combination of modeled spatial distributions of 12,500 species with spatial information on cumulative human impacts has recently revealed that Mediterranean sea should be a priority for marine conservation (Selig et al. 2014). Indeed, representing 0.8 % of the oceans but harboring 7 to 8 % of the known marine species (around 17000 species), Mediterranean sea is a biodiversity hot spot (PNUE/PAM-Plan Bleu 2009; Coll et al. 2010). At the scale of the basin, biodiversity would globally decrease from northwestern to southeastern regions and from coastal areas to deeper ones (Coll et al. 2010). Changes in habitat (loss and degradation), followed by fishing impacts, pollution, climate change, eutrophication, and alien species are the most important threats and affect the greatest number of taxonomic groups. All these impacts are expected to be more and more important in the future, especially climate change and habitat degradation (Coll et al. 2010).

Two ecosystems contribute to the most part of Mediterranean marine biodiversity, productivity and ecosystem services: *Posidonia oceanica* (L.) Delile seagrass beds and coralligenous reefs (PNUE/PAM-Plan Bleu 2009). P. *oceanica* is an endemic marine phanerogam from Mediterranean sea growing between 0 and 40 m deep. It constitutes sub-marine meadows with essential ecological roles: primary production, oxygen production, water transparency by trapping particules in suspension, base of numerous food chains, spawning areas, nursery, shelter against predators or permanent habitat for thousands of animal and plant species (Boudouresque et al. 2012). P. *oceanica* is a biological element used to measure the ecological status of coastal water bodies according to the Water Framework Directive (WFD, 2000/60/EC). Coralligenous reefs are found between -12/-50 and -40/-120 m deep depending on the water transparency (Ballesteros 2006). They consist of biogenic blocks built by calcareous algae and animal builders such as bryozoans, serpulids, cnidarians, mollusks, sponges, crustaceans and foraminifera and secondarily colonized by many other animals and plants (Ballesteros 2006). Threatened by numerous human activities (PNUE/PAM-Plan Bleu 2009), P. *oceanica* and coralligenous ecosystems are recognized as natural habitats of communitarian interest, whose conservation requires the designation of Zones of Special Conservation at European level (92/43/CE Habitat Directive, habitat codes 1120 : Posidonia beds and 1170-14: Reefs, coralligenous assemblage).

If everyone agrees that some progress has been made in the past 30 years in marine ecology conservation, greater effort will be required to reach the same level of practical benefits to coastal management than for terrestrial conservation (Pittman et al. 2011). Here, we fill in a gap by making available seabed habitat maps and ecological data concerning the two most important Mediterranean marine ecosystems (*Posidonia oceanica* seagrass and coralligenous habitat), but also anthropogenic pressures (man-made coastline, wastewater, population density, aquaculture, boat anchoring, land use, macrowaste, hydrocarbon pollution localization and/or impact). Here we describe Medtrix, a cartographic platform that regroups different databases made available for marine ecologists. This meta-database is the first one available at such a good resolution (20 m for the anthropogenic pressures) all along the French Mediterranean coast and along some other countries (Tunisia, Italia). In this paper, we will first describe the functionalities of this cartographic platform before briefly presenting each database and their potential uses.

4.3 Overview

Medtrix is a cartographic platform that regroups different databases made available for professionals: marine ecologists, stakeholders and marine site managers or administrators. These databases (called "projects" on the platform) concern: water quality monitoring networks (RECOR, TEMPO), pressure monitoring networks (IMPACT, MEDOBS) and a seabed habitat description (DONIA® expert with its so-called application). Moreover, other databases are expected in the future. The methodologies used for each project are detailed on the website http://www.observatoire-mer.fr/en/. Thus we describe briefly thereafter the functionalities of the cartographic platform and then the procedures used for the acquisition and mapping of data for each database.

The data hosted by Medtrix are only disabled for logged-in people trough the "connection" index at http://www.medtrix.fr/. Interested people need thus previously to freely create a count at the homepage by clicking on « register ». For the moment (July 2015), 538 professionals are registered on Medtrix.fr and we count two new users, 55 connections and 40 prints of maps every day. Data made available on the platform Medtrix are provided according to the terms of the Creative Commons Attribution-Non Commercial-NoDerivs 3.0 Unported License.

4.4 Medtrix operations

Medtrix uses the open source application LizMap, a complete solution for the publication of QGIS maps on the web. The basic operations of Medtrix consist of making available different databases (projects) related to the Mediterranean coastal water quality and proposing dynamic and cartographic results. Once the user is logged-in, he can choose the project he wants to open. For each project, the user can choose between get to the map showing the results or read a brief description providing an abstract, the data sources, the map projection and the geographic extent.

Basic Data mapping

Each map displays localization options by layer (region, site, water body, …), a legend, dynamic functions (zoom, scale, localization) and pictograms corresponding to different menus and tools (Fig.1a). The choice of the variables that can be displayed on the map depends on the project (see 4). In the legend, the user can choose the variables he wants to display on the map (Fig.2a); the color correspondence appears after a click on the arrow and the definition of each variable can be displayed in a reactive window. The different functions of the menu (Fig. 1b) permit to access to the homepage, display the map without legend and localization, display or close the localization index, display information concerning the project, compare stations, print a pdf of the map at the chosen scale (Fig.2b), measure areas, perimeters or lengths and geolocalized your position via the web navigator. Moreover, pop-up windows linked to each sampling point inform also the user on the characteristics of the sites: water body, pressure sources, date of sampling, local picture, a link to see the photographic quadrats, etc.

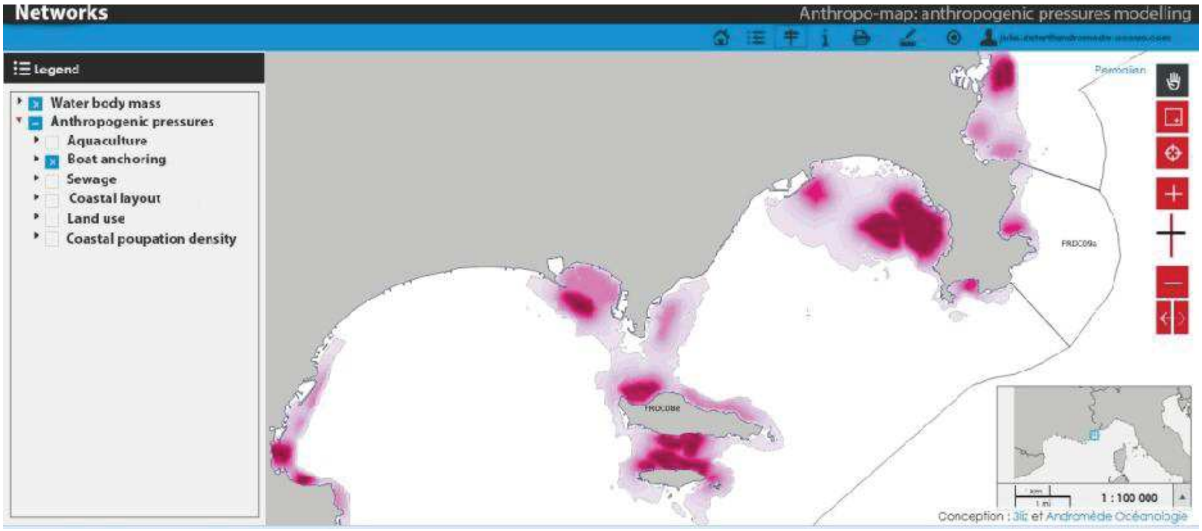
Figure 1. Example of map presented within the project RECOR with legend, tools and menus pictograms (a). For each menu pictogram, a pop-up window displays its mean (b).

Compare stations

Stations sampled respectively for TEMPO and RECOR may be compared through the function "compare" of the menu. The user can choose the scale (Mediterranean, region, water body) used for the comparison and the stations (visible on the map) he wants to compare within a project. A click on "generate" creates a table in a new window (Fig.3). Each parameter (30 for RECOR, 4 for TEMPO) presented within the table, is associated with a graphic cursor repositioning the observed value within the chosen spatial scale (minimum, maximum and average values observed in Mediterranean, the region or the water body). These tables can be exported as pdf.

Figure 2. Examples of maps realized on the cartographic platform medtrix.fr a) on line for the impact of boat anchoring pressure, one of the six variables available with anthropo-map (darker red indicates a higher impact) and b) after a pdf export for the sludge cover percentage, one of the eight variables available with RECOR (darker color and bigger circles indicate a higher value).

a)



b)

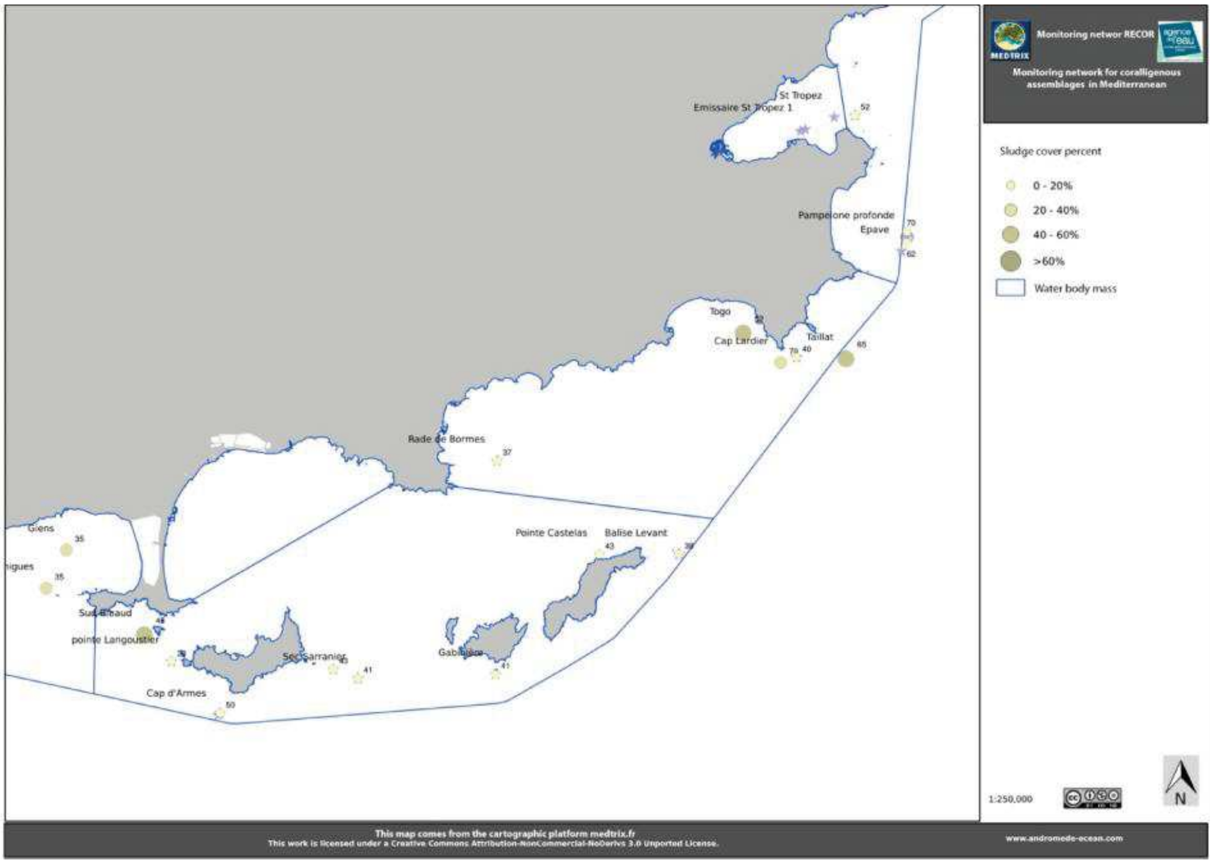


Figure 3. Examples of a) six RECOR parameters (among 29) compared for one site in Corsica in 2011 (Site Tabernacle within the golfe of Ajaccio), sampled at different depths between -40 m and -90 m. b) Examples of all TEMPO parameters compared for Cap Roux, Juan, Agay Ouest and Cap Nice Est in 2008 and/or 2013 in PACA region. The value presented at the extreme left, middle and extreme right of the cursor respectively correspond to the minimal, mean and maximal values observed at the chosen spatial scale.

a)

Region	Corsica	Corsica	Corsica	Corsica	Corsica	Comparison scale Corsica Region Legend
Water body	FREC04b	FREC04b	FREC04b	FREC04b	FREC04b	
Site name	Tabernacle	Tabernacle	Tabernacle	Tabernacle	Tabernacle	
Depth (m)	90	70	60	50	40	
Year	2011	2011	2011	2011	2011	
Living %	35.19	67.58	68.99	68.34	81.17	35.19 60.48 81.17
Non-living %	64.81	32.42	31.01	31.66	18.83	18.83 39.52 64.81
Sludge %	36.36	22.99	11.49	10.82	4.11	2.87 19.15 42.18
Biological remains %	3.88	1.02	4.79	5.22	0	0 1.16 5.22
Crevice %	13.29	8.2	11.97	15.3	14.61	7.11 16.62 34.21
Necrosis rate	15.95	26.8	31.91	26.65	0	0 51.46 216.22

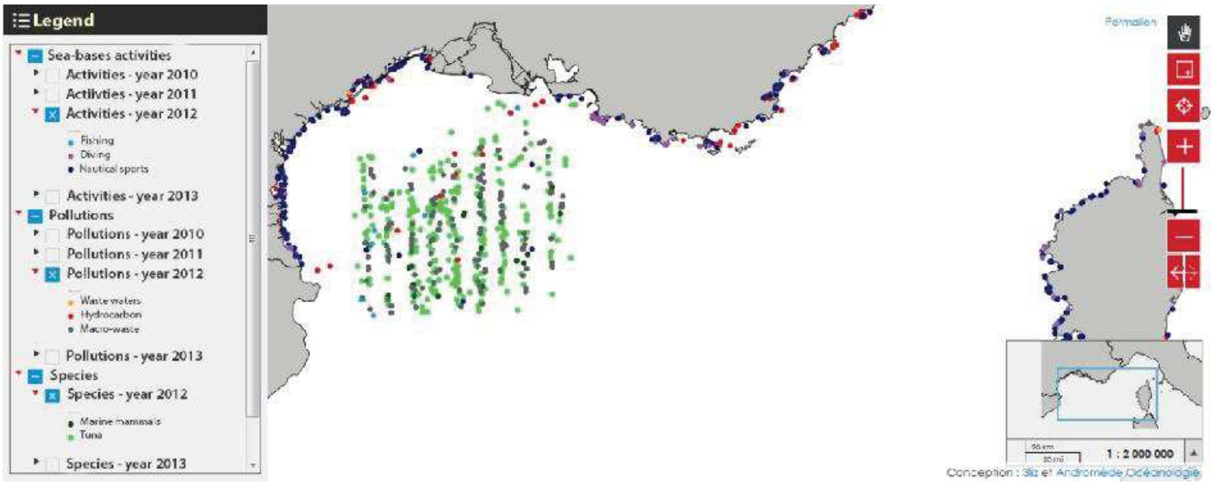
b)

Region	PACA	PACA	PACA	PACA	PACA	PACA	Comparison scale Mediterranean Legend
Body mass	FRDC08d	FRDC08d	FRDC08e	FRDC08e	FRDC08d	FRDC09c	
Site name	Cap Roux	Cap Roux	Juan	Juan	Agay Ouest	Cap Nice Est	
Year	2013	2008	2013	2008	2013	2013	
EQR TEMPO site	0.58	0.64	0.39	0.39	0.56	0.54	0.3 0.51 0.8
EQR' Typology	0.66	0.66	0.21	0.21	0.66	0.66	0.21 0.51 0.89
EQR' Depth	0.64	0.64	0.66	0.66	0.51	0.51	0.16 0.67 1
EQR' Density	0.43	0.62	0.29	0.29	0.5	0.44	0.15 0.36 0.83

4.5 Projects

MEDOBS. This network identifies and positions pressures (activities and pollution) undergone by the Mediterranean sea since 2010 with the help of a sophisticated numeric video camera taken on a plane. Activities are divided in fishing, diving and nautical sports practice while the different researched pollution categories are dirty waters, macro-waste and hydrocarbon pollution (Fig4). Since 2012, data concerning marine mammals and tuna are also referenced. MEDOBS' objectives are not to follow the uses with a fine and exhaustive way but to appreciate pressure occurrence on the territory in order to better identify stake areas (biocenose alteration, social manners conflicts ...) and organize the manners. The monitoring was initially realized along the French Mediterranean coast (3 nautical miles) but extends offshore (Pelagos sanctuary and canyon heads) since 2014. Data acquisition is annual, with a low rate of flights during winter and a higher rate in summer, high frequentation period of the Mediterranean water plan. With Medtrix, the user can visualize the position of identified pressures with date and hour of observation (1400 points in July 2015).

Figure 4. Example of map built with MEDOBS data (activities, pollutions, species) collected in 2012.

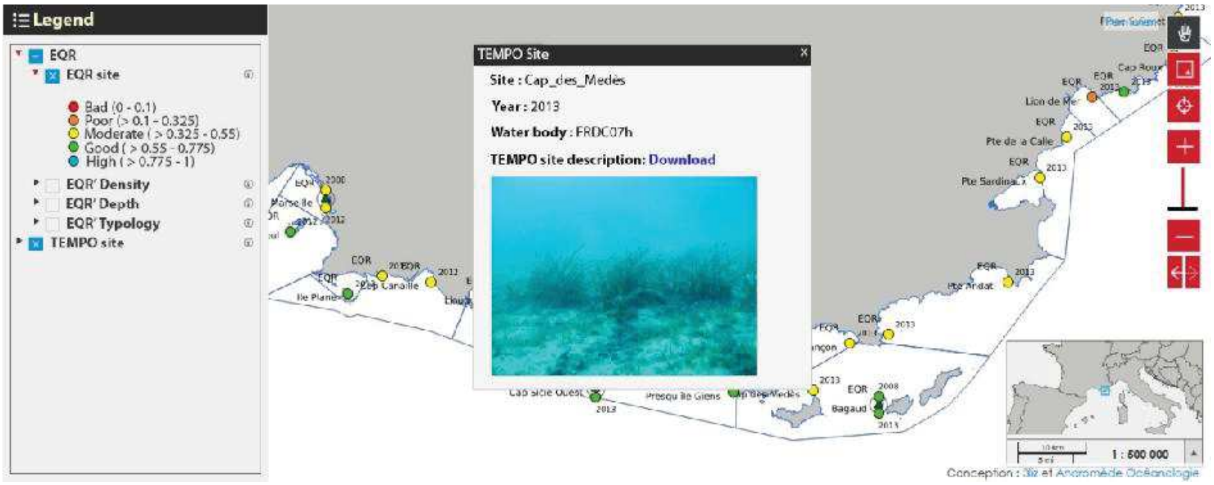


IMPACT. This project aims for the development of spatial models for a better understanding of the links between anthropogenic pressures and the ecological status of coastal ecosystems. It consists in modeling at a large-scale (French Mediterranean) the spatial extent of anthropogenic pressures impacts on the marine environment. Six different pressures (bases on quantitative data) are visualized for their spatial extent: aquaculture (total area of the farms), boat anchoring (number and size of boat observed during summer), sewage (capacity, output), man-made coastline (big harbours / harbours / artificial beaches, ports of refuge / pontoons, groins, landfills and seawalls areas), land use (proportion of natural areas, urbanized lands, farmlands) and costal population density (Fig.2a). Data concerning the origin and intensity of these pressures come from a database supplied by Agence de l'eau RMC and Ifremer completed by MEDOBS data and an analysis of Google Earth maps but also MEDAM (MEDAM 2014), CORINE land cover (http://www.statistiques.developpement-durable.gouv.fr/donnees-ligne/li/1825.html) and INSEE (http://www.insee.fr/fr/). Models are built using ArcGIS 10 (ESRI) with a 20-m distance matrix, and we apply a pressure curve with a negative exponential shape ranging between 100 % (origin) and 0 % (no impact anymore) to each type of pressure. The spread modalities of the pressures take the bathymetry (data from SHOM and Andromède Océanologie) into account and are defined after a literature synthesis and expert judgment. Details concerning these models and their use for the prediction of seagrass meadow distribution are described in a paper under progress.

RECOR. RECOR is a monitoring network for coralligenous assemblages using a non-destructive methodology (Andromède océanologie 2013a). RECOR's objectives are to collect data describing the state and functioning of the coralligenous habitat and follow their changes in time and space. It covers the entire French Mediterranean coast with at least one site per water body. RECOR includes thus 157 stations regrouped in 86 sites (several stations = sampling depths per site) localized within the coralligenous habitat distribution known from DONIA (see thereafter). Fifty-four sites (93 stations) are located in the French Provence-Alpes-Côte-d'Azur (PACA) region, five in Languedoc-Roussillon (L-R, 7 stations) and 27 in Corsica (57 stations). These stations are distributed between 17 and 90 meters deep and are monitored every three years at the end of spring (June). The stations were chosen in order to represent different environmental conditions with a minimum of one site per water body (body of water having relatively homogeneous physical-chemical characteristics following the Water Framework Directive (WFD, Directive 2008/56/EC)).At each site, GPS coordinates are collected from the boat. They correspond to the location of a steel spike (50 cm long) set within the coralligenous reef and marking the beginning of the deepest transect. Coralligenous assemblages (sessile organisms) are described from photographic quadrats taken at each station by a CCUBA (Closed Circuit Underwater Breathing Apparatus) diver. A compass and a depth meter are attached to the quadrat in order to keep homogenous the orientation (North at 12h) and depth for the photos. Each station is sampled using 30 photographic quadrats (50 x 50 cm) along a 40 m-long transect as described in (Deter et al. 2012b). Each picture is analyzed by a single person using 64 random points via CPCe 4.1 "coralligenous assemblages version" (Kohler and Gill 2006) for the estimation of the percentage of the total area covered by each taxa. Necrose are specified for each point when it was relevant (for encrusting red algae, gorgonian and erect bryozoans). Non-living parameters like sludge, biological debris and cavity percent covers are estimated by the same way. All living organisms are finally expressed in relative abundances corresponding to their frequency considering sessile living organisms only (extraction of non-living parameters and mobile organisms like fish and crustaceans). Stations, sites, water bodies are visible on the map. In correspondence with each station, points colored in function of their value are available for the coralligenous assemblage index (CAI, see Deter et al. 2012a), the percent cover of sludge, the relative abundance of major builders (encrusting red algae, Peyssonnelia sp, encrusting bryozoan, Scleractinian, Leptopsammia pruvoti, Miniacina miniacea, Myriapora truncata, Pentapora fascialis) and the relative abundance of disturbing species (filamentous algae, Codium bursa, Asparagopsis sp, Womersleyella setacea, Caulerpa taxifolia and Caulerpa racemosa). With the help of a graphic indicator, the user can also compare 30 variables per stations (Fig.3a): Simpson and Shannon indices, CAI, cover percentage of living organisms, cover percentage of non-living organisms, sludge, biological remains, crevice, necrosis but also the relative abundance of major builders, disturbing species, erect bryozoans, encrusting bryozoans, Mesophyllum sp, Lithophyllum sp, erect Peyssonnelia sp, encrusting Peyssonnelia sp, Axinella sp, massive sponges, encrusting sponges, echinids, gorgonians, filamentous algae, Cliona sp, Cystoseira sp, Corallium rubrum, Womersleyella setacea, Caulerpa taxifolia, Caulerpa racemosa and Filograna sp. These variables regroup important taxa in terms of abundance, functioning, sensitivity to pressures and/or water quality (Andromède océanologie 2013a).

TEMPO. TEMPO is a monitoring network that collects descriptive data concerning *Posidonia oceanica* beds dynamics and studies their evolution in time and space. Since 2011, 60 sites (100 m² in average) between 5 and 40 m depth are monitored all along the Mediterranean French coast at a rate of one third of the coast per year. Each TEMPO site is monitored at the end of spring (June). The non-destructive method chosen for the monitoring is detailed in Andromède océanologie (2013b). Three types of data are sampled at the lower limit: a general description of the site (research of disturbances, (Blouet et al. 2011; Lepareur 2011), a centimeter accuracy map using acoustic telemetry (Descamp et al. 2011) and stationary measurements of vitality (limit type (Meinesz and Laurent 1978; Montefalcone 2009), density per m² and dislodging (Boudouresque et al. 2012)). The depth, type and shoot density at the lower limit are then used to calculate Ecological Quality Rates (EQR) modified after Lopez y Royo et al. (2010). From June 2015, the *Posidonia oceanica* Rapid Index, PREI (Gobert et al. 2009) is also calculated at several sites. Medtrix allows the user to directly visualize EQR on the dynamic maps while the precise maps done with acoustic telemetry for the lower limits can be downloaded after a click on the corresponding TEMPO site (Fig.5).

Figure 5. Example of map obtained with TEMPO data. A pop-up window appears after a click on the chosen TEMPO site. More information concerning the site and a precise map done with acoustic telemetry for the lower limits can be downloaded after a click on the link.



TEMPO Site

Site : Cap_des_Medès

Year : 2013

Water body : FRDC07h

TEMPO site description: Download

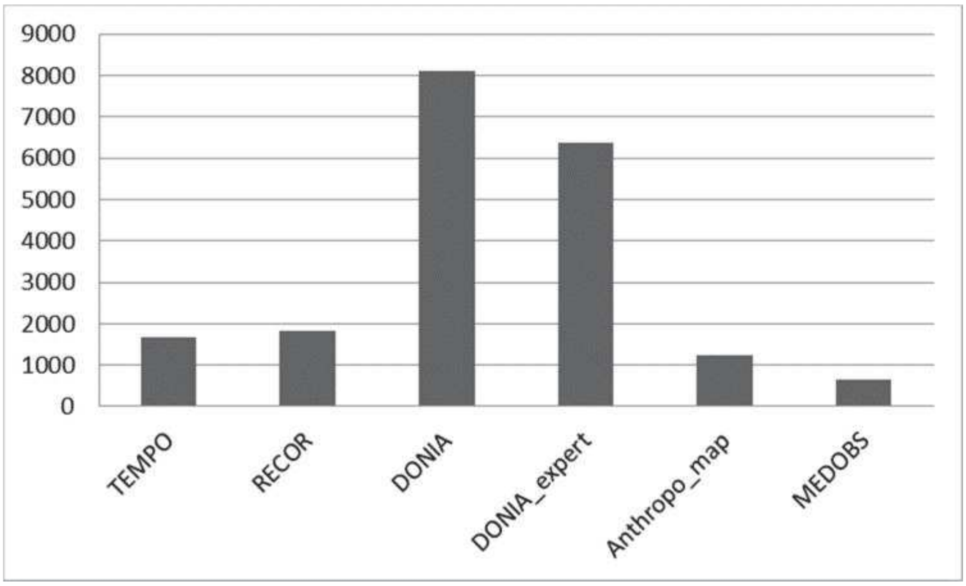
DONIA®expert. It makes available a 1:5000 map for seabed habitats using ten classes: Cymodocea seagrass, Posidonia seagrass, dead matte association, infralittoral shingle association, infralittoral soft bottoms, infralittoral algae, coralligenous habitat, offshore rocks, circalittoral soft bottoms, artificial habitats (Fig.6). The user can also display isobaths (every ten meters from 0 to -50 m) and pictures illustrating the different marine habitats and species. Besides the complete French Mediterranean coast (L-R, PACA, Corsica), several Mediterranean islands are concerned in DONIA®expert: Galite archipelago in Tunisia, Zembra island in Tunisia and Tavolara - Punta Coda Cavallo in Sardinia (Italy). This work pools and homogenizes the maps built in the framework of numerous programs: DONIA (Andromède Océanologie), CARTHAM - AAMP (GIS Posidonie, Sintinelle, Stareso, Evemar, Andromède océanologie), PIM (Conservatoire du littoral, Agence de l'eau RMC), MEDBENTH (IFREMER). Campaigns were led between 2010 and 2015 using Aerial photography (in order to see the spatial extent of habitats in shallow waters), multi-beam echo-sounder surveys (to draw the bathymetry), side-scan sonar surveys (used in more turbid and deeper waters), sonar information post-treatment (in order to determine potential presence and covering of underwater habitat representation), production of a preliminary cartography which is then validated by diving sessions 'ground truth' (field work) (Andromède Océanologie 2014). A simplified database called DONIA® intended for general public is also available on Medtrix without any login. The same regions are concerned but habitats were simplified into four classes (seagrass, dead matte, sand, rock) instead of ten. This simplified dynamic map is freely accessible to all though DONIA® application (App Store and Google play), helping boat to anchor safely outside of sensitive habitats.

Figure 6. Example of map obtained with DONIA®expert data.

4.6 Medtrix: uses and perspectives of uses

Since its opening in September 2013 with only two databases at the beginning, the cartographic platform Medtrix has largely expanded with six projects in August 2014. Its utility is demonstrated by its 338 professional users with varied profiles: stakeholders, managers, researchers, engineers and other ecologists. In spite of its opening in the first place, RECOR and TEMPO (respectively 1837 and 1659 views) are largely less visited than DONIA® and DONIA®-expert (8113 and 6375 views) opened three months later (Fig. 7). But the perspectives remain huge and we briefly expound them thereafter.

Figure 7. Number of views registered per project on Medtrix in July 2015. Projects on the left were opened earlier (September 2013) than projects on the middle (December 2013) or on the right (January 2015)



The success of DONIA ® is easily explained by its general public target without any log-in and the communication it beneficiated thanks to two awards received: "enterprise and biodiversity" from the French Ministry of environment and "bateau bleu" from the French nautical industrials. This success should largely increase in summer 2015 with the launch of the complete application and the communication plan it comes with it, to yachtsmen and harbors. Linked to this simplified database and proposing more general data, DONIA®-expert is thus naturally more consulted than RECOR and TEMPO. This project provides 1:5000 maps of seabed habitats between 0 and 80 m depth. Such information is so easy to obtain for terrestrial lands (Google map, CORINE land cover, etc) that we almost forget its absence under the sea (but see Andromède Océanologie (2014) for a paper or non-interactive pdf version). However it is essential in order to understand the presence and abundance of sessile and vagile species or simply plan inventory or sampling field campaigns. Coupled with environmental data, potentialities are great: management plans, niche modeling... Moreover, note that these maps are now available in two dimensions but the passage to 3D is in progress and will permit to calculate real areas covered by habitats in reliefs (Hoechstetter 2008; Andromède Océanologie 2014). This will be the aim of the next project available soon on Medtrix: SURFSTAT (Andromède océanologie 2013c). For example, 3D consideration shows that coralligenous reefs surfaces are generally increased of 11.11 % (Pavy 2013) with certain vertical coralligenous rims underestimated of 74 % in 2D (Holon et al. 2013). Thanks to the biocenose maps performed for DONIA® we finally have access to the seabed heterogeneity in composition (relative area of different habitats) and configuration (closeness of contacts between different habitats). These are the basic data needed by an emerging disciplinary: marine landscape ecology or seascape ecology. It aims at determining the mechanisms by which spatial patterns influence key processes in ecology. It should focus in the future on identifying and mapping connections and corridors; identifying structures driving biotic assemblages and the distribution of biodiversity; or understanding the impacts of global change on seascape patterns (spatial shifts, fragmentation, loss of habitats) (Wedding et al. 2011; Boström et al. 2011; Pittman et al. 2011).

RECOR, TEMPO (and to a lesser extent MEDOBS) provide abundance and localization data for diverse species among which several ones are protected (Corallium rubrum, Pinna nobilis, P. *oceanica*, …). Moreover, sampling at a community scale (RECOR) permits to consider conservation questions on the base of an integrated ecosystem approach more than on some charismatic species (Fraschetti et al. 2002; Fraschetti et al. 2005). Present conservation efforts badly preserve biodiversity at an ecosystem level (Coll et al. 2011; Mouillot et al. 2011). In addition to the species unit (taxonomy) classically used, we know now that other components of biodiversity such as functional diversity and evolutionary history (phylogeny) are important to ecosystem functioning (Hooper et al. 2005; Mouquet et al. 2012; Srivastava et al. 2012) and not necessarily spatially congruent with taxonomic diversity. For example, RECOR data revealed differences between taxonomic versus functional and phylogenetic diversities according to their spatial distribution analysis at the station level revealed. In particular, no significant variation was observed over depth (30-70 m) for taxonomic diversity, whereas functional and phylogenetic diversities significantly decreased vertically (Doxa et al, submitted).This reveals different processes in community structure and provides complementary information to identify conservation priority areas (Devictor et al. 2010; Mouillot et al. 2011). Joint analyses between abundance/diversity data brought by RECOR and environmental data classically used in marine ecology (sea surface temperature, salinity, turbidity, chlorophyll a, etc) help to understand the mechanisms involved in the heterogeneity of coralligenous outcrops (Holon et al. 2014). RECOR and TEMPO provide also less classical environmental data sampled on seabed like sand, biological remains, crevice or sludge percent covers but also the abundance of macro-waste, so many new variables describing the local state of deep stations. Finally, RECOR, TEMPO and MEDOBS meet their first objectives as they inform on the coastal water quality and provide data that help for the development of enhanced or new ecological quality indices (Deter et al. 2012b; Deter et al. 2012a).

Anthropo-map is particularly interesting for all the scientists working on the Mediterranean sea as it regroups spatial quantitative data concerning the most important pressures undergone by the basin. Some data already do exist but they are either very localized and restricted to one or some pressures (Kress et al. 2004; Forchino et al. 2011) or very extended with a higher resolution (one degree = 25 km; Halpern et al. 2008; Micheli et al. 2013). While localized studies help to define the parameters (impact distance) used for our model (Holon et al submitted), large resolution works support coordinated and comprehensive actions (at a large scale) ensuring a good ecological status of waters, and considering all impacts across the basin (Micheli et al. 2013). The fine resolution of Anthropo-map (20 m) allows at last apprehending the spatial scale managers are interested in. Such maps are particularly important because these are the bases of numerous other studies as i) an help for decision by targeting areas with strong stakes: such mapping might improve and rationalize spatial management of human activities (Crowder et al. 2006). This will be done by quantifying the ecological tradeoffs associated with different human uses of marine ecosystems and identifying locations and strategies to minimize ecological impact and maintain sustainable use (Micheli et al. 2013). ii) A basis for ecological indicator development which necessitates linking pressures with ecological data and measuring the impacts of human on ecosystems (Gobert et al. 2009; Deter et al. 2012a). iii) A predictive tool for the management of coastal areas through sensitivity mapping (Parravicini et al. 2012) and/or reserve design (Montefalcone et al. 2009; Mouillot et al. 2011).

Finally, Medtrix will be continuously updated by adding new data to the present projects and by attracting new projects. Two new projects are already planned for Sptember 2015 on Medtrix: SURFSTAT (see above) and RESPIRE. RESPIRE is a project in ecological restoration hold by the company ECOCEAN. It aims for the monitoring of fish and marine flora developing on artificial habitats (biohut®) equipping 20 harbours. The objective is to evaluate the capacity of harbours to let larval fish stay.

4.7 Summary

The cartographic platform Medtrix is a prodigious source of information for a large community of scientists with varied questions. Though five different projects, it provides i) 1:5000 maps describing seabed habitats along the French Mediterranean sea et several foreign islands (DONIA® expert), ii) ecological data concerning the two most important marine ecosystems of Mediterranean: coralligenous assemblages (RECOR) and P. *oceanica* seagrass beds (TEMPO), iii) spatial referencing of three different pressures and marine mammals (MEDOBS) and iv) high resolution (20 m) maps modeling the origin and intensity of six different anthropogenic pressures: aquaculture, boat anchoring, sewage, costal layout, land use and costal population density (Anthropo-map). Different functionalities allow to display information concerning the project, see pictures, print a pdf of the map at the chosen scale, measure areas, perimeters or lengths and geolocalized your position via the web navigator. Moreover, stations may be easily and directly compared between each other thanks to the generation of tables and graphic cursors.

By making available all of these data, the cartographic platform Medtrix helps to evaluate the ecological consequences of spatial patterns and structural changes in the submerged landscapes of coastal ecosystems. For the first time, we have at our disposal marine ecological information at spatial scales that are operationally relevant to management.

Medtrix is continually updated and suggestions-collaborations from other researchers are welcome. Medtrix is freely available for general public via DONIA® and for all logged-in professionals (researchers, stakeholders…) via the other data bases. The databases hosted by Medtrix are disabled at http://www.medtrix.fr/ and information concerning these databases can be found at http://www.observatoire-mer.fr/en/. Data made available on the platform medtrix.fr are provided according to the terms of the Creative Commons Attribution-Non Commercial-NoDerivs 3.0 Unported License.

Acknowledgments

This study and all data samplings beneficiated of a financing from the French Water Agency (Agence de l'eau Rhône-Méditerranée-Corse) in the framework of the different monitoring networks it supports. MEDOBS data are collected by Observatoire MEDOBS. TEMPO, RECOR data are collected by Andromède océanologie. Anthropo-map maps were created on the basis of data collected by Andromède Océanologie, IFREMER, INSEE, MEDAM, MEDOBS, MEDAM and SHOM. Maps available with DONIA® were created from data collected for DONIA and SURFSTAT by Andromède océanologie but also the Programm CARTHAM - AAMP (GIS Posidonie, Sintinelle, Stareso, Evemar, Andromède océanologie), and Conservatoire du littoral, IFREMER, Programm MEDBENTH. TEMPO and RECOR are part of OSU-OREME within the observation system "littoral et trait de côte". Florian Holon received a PhD grant (2013-2016) funded by LabEx CeMEB (Laboratoire d'excellence "Centre méditerranéen de l'environnement et de la biodiversité") and Andromède Océanologie. We thank all the people who participate to field work and/or map making in the framework of these monitoring networks. This manuscript was recorded within ISEM data set.

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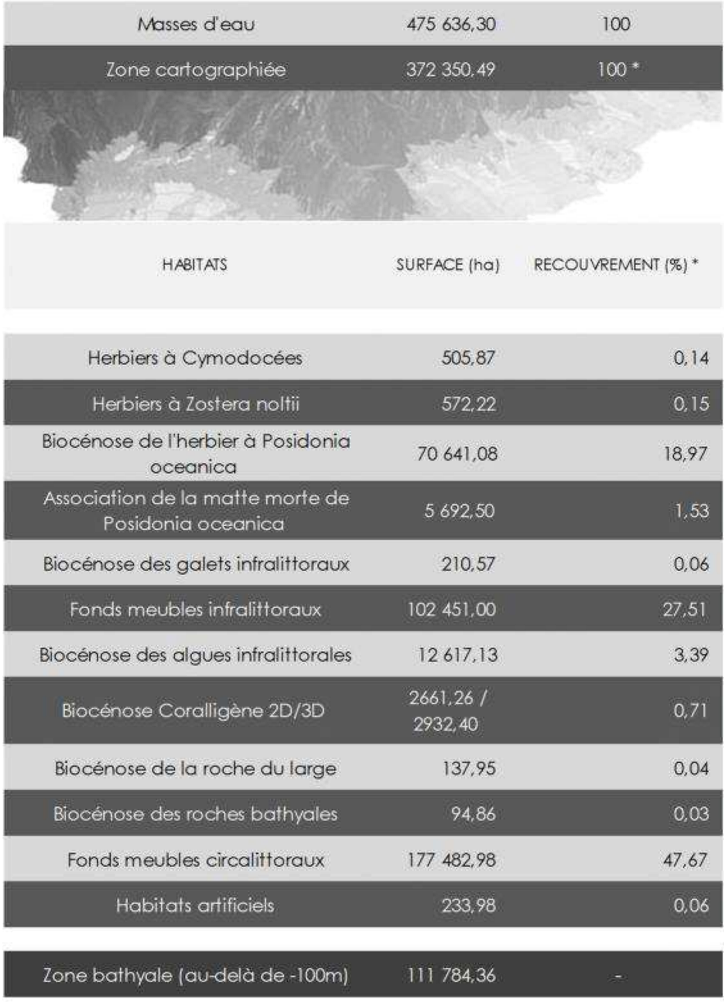
Synthèse et perspectives

Les objectifs de ce projet doctoral étaient de caractériser la vulnérabilité et d'aborder la sensibilité de deux écosystèmes marins : les herbiers de posidonie et les assemblages des communautés à coralligène. Dans cette conclusion, je fais le point sur les principaux résultats obtenus durant ma thèse et les perspectives qu'ils ouvrent, à savoir la quantification et la localisation de ces écosystèmes en France, une estimation de leur état écologique et la quantification et localisation des pressions anthropiques auxquelles ils sont soumis. Je ferai ensuite le bilan des relations mises en évidence entre ces écosystèmes et les pressions anthropiques et les apports de cette thèse pour la DCSMM. Enfin, je finirai par les efforts que j'ai mis en œuvre afin de valoriser et faire connaitre ces résultats auprès des professionnels de la biodiversité marine.

1 Cartographie et quantification des surfaces couvertes par les herbiers à posidonie et les assemblages à coralligène.

Nous présentons ici une étape importante dans la connaissance des fonds marins en méditerranée française sur plus de 1700 km de côte: la première carte continue des habitats marins. Cette connaissance fine (pixel 20*20m) de la distribution des habitats de chaque région, et en particulier des herbiers de Posidonie et du coralligène, est une étape essentielle pour une gestion appropriée et efficace à long terme des ressources marines (Wedding et al., 2011).

Ce travail cartographique a permis de connaitre, pour la première fois, la surface couverte par les différents habitats marins côtiers (Figure 19). Par exemple, nous savons désormais que les herbiers à *Posidonia oceanica* occupent 70 641 ha et le coralligène 2 661 ha. Mais également que les régressions des herbiers peuvent être estimées, suivant la



Masses d'eau	475 636,30	100
Zone cartographiée	372 350,49	100 *

HABITATS	SURFACE (ha)	RECOUVREMENT (%) *
Herbiers à Cymodocées	505,87	0,14
Herbiers à Zostera noltii	572,22	0,15
Biocénose de l'herbier à Posidonia oceanica	70 641,08	18,97
Association de la matte morte de Posidonia oceanica	5 692,50	1,53
Biocénose des galets infralittoraux	210,57	0,06
Fonds meubles infralittoraux	102 451,00	27,51
Biocénose des algues infralittorales	12 617,13	3,39
Biocénose Coralligène 2D/3D	2661,26 / 2932,40	0,71
Biocénose de la roche du large	137,95	0,04
Biocénose des roches bathyales	94,86	0,03
Fonds meubles circalittoraux	177 482,98	47,67
Habitats artificiels	233,98	0,06
Zone bathyale (au-delà de -100m)	111 784,36	-

Figure 19 Surface des habitats marins cartographiés en méditerranée française. (Holon et al., 2014)

surface des mattes mortes à au moins 5692 ha (entre 1920 et 2013), contrairement aux 2753 ha estimés précédemment par Telesca et al. (2015). Même si ce chiffre peut être légèrement relativisé en raison de l'existence de mattes « naturelles » liées à la dynamique naturelle de certains herbiers (Boudouresque et al., 2009), il me semble fiable car ces zones représente une surface restreinte par rapport à l'ensemble des mattes mortes observées .
Le contraste cartographique est important avec le milieu terrestre et témoigne à la fois de la difficulté de la tâche et de l'intérêt des services publics : si nous connaissons les surfaces couvertes par les forêts, prairies ou mares depuis le milieu du XIX^{ème} siècle, il aura fallu attendre 2014 pour connaitre les surfaces de nos habitats patrimoniaux sous-marins. Or, comment protéger ce que nous ne connaissons pas ?

Cette cartographie comme la plupart des cartographies est en deux dimensions. Cependant, plusieurs auteurs ont mis en évidence la nécessité d'intégrer les aspects de la troisième dimension dans l'analyse du paysage à grande échelle (Hoechstetter et al., 2008; Wedding et al., 2011). Par exemple un écosystème se développant sur des surfaces verticales comme les tombants de coralligène est sous-représenté puisque seule sa surface « aplatie » en 2D est comptabilisée. La non-prise en compte de la structure 3D peut donc entraîner des mesures erronées de paramètres comme le périmètre, l'aire, ou encore la distance entre les éléments (Hoechstetter et al., 2008).

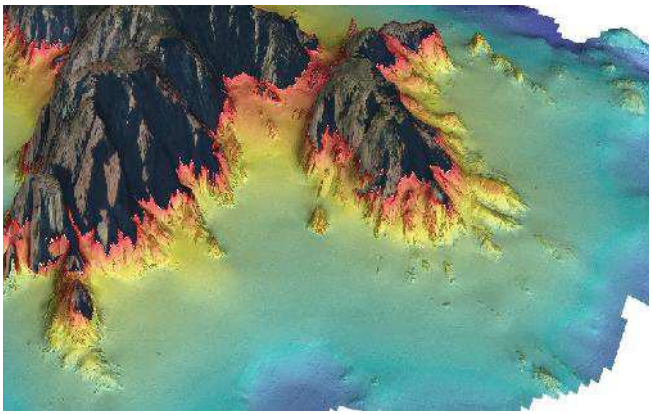


Figure 20 Modèle Numérique de Terrain d'une bathymétrie fine pour la cartographie des assemblages à coralligène. (en gris la zone terrestre, en rouge la zone -10 m et en bleu foncé la zone à -60 m)

Afin de prendre en compte le relief dans le décompte des surfaces des habitats marins, il est nécessaire d'avoir une bathymétrie fine de ces zones (Figure 20). Dans le cadre du programme Surfstat (Agence de l'eau RMC / Andromède Océanologie) des campagnes bathymétriques ont été menées en 2013/2014/2015 sur l'ensemble des tombants de coralligène en méditerranée française. Nous avons réalisé un modèle numérique altimétrique qui a permis d'estimer la surface totale 3D du coralligène (pour un pixel de 2 m) à 2 932 ha soit une augmentation de 10,1 % par rapport à la surface 2D. Ce chiffre reste relativement faible alors qu'il prend désormais en compte les caractéristiques propres à cet habitat à savoir l'inclinaison et le relief global des récifs. Toutefois, il ne considère toujours pas le volume véritablement représenté par cet habitat de par les différentes strates (encroûtante à arbustive) d'organismes qu'il fixe et l'endofaune qu'il abrite de par sa structure intra-récif complexe (réseau de galeries (Figure 21) (Ballesteros, 2006)). Il reste donc du travail à faire avant d'estimer plus précisément la surface écologiquement « utile » de cet habitat et la biodiversité abritée par cet écosystème (Ballesteros, 2006) mais aussi le volume de calcaire (stockage de CO2) qu'il représente et risque d’arrêter de stocker voire de relarguer en cas d’acidification de l’eau de mer dans les prochaines décennies (Martin and Gattuso, 2009; Noisette et al., 2013).

Figure 21 : Coupe schématique d'un récif coralligène, montrant la forte hétérogénéité de l'environnement à petite échelle et les différents micro-habitats. (Dessin de J. Corbera issu de Ballesteros (2006))

Ces données cartographiques vont s’avérer très utiles pour tout travail en lien avec la trame bleue marine proposée par le livre bleu des engagements du grenelle de la mer (Ministère chargé de l’écologie et de la mer, 2009), sorte de prolongation maritime de la trame verte et bleue du grenelle de l’environnement. Elles serviront également à de futurs projets en écologie du paysage permettant (Boström et al., 2011; Xiuzhen Li and Mander, 2009) de relier composition et configuration des fonds sous-marins côtiers avec des données de connectivité entre populations, d’impacts de pressions, de qualité de l’eau ou d’état écologique des différents habitats (Abadie et al., 2015; Tett et al., 2013).

D’ores et déjà, cette nouvelle connaissance cartographique a permis le développement (auquel j’ai participé) d’un outil innovant pour la préservation des herbiers sous-marins : l’application mobile Donia (http://www.donia.fr/). Il est désormais possible, gratuitement, à tout plaisancier en méditerranée française, de jeter son ancre dans une zone de moindre impact comme du sable et de préserver ainsi les herbiers alentours. Un objectif parallèle, via cette application, est que les gestionnaires des aires marines protégées puissent contrôler et limiter les impacts du tourisme nautique dans leur région, en particulier pour la navigation de grande plaisance. Une interface interactive permet une surveillance en temps réel du flux de bateaux sur une zone. L'envoi et la réception d'informations entre les gestionnaires et les utilisateurs renseignent sur les zones d’ancrage, la pollution marine, ou encore sur la présence d'espèces protégées.

Cette application ouvre de nouvelles perspectives en termes de gestion des aires marines protégées en facilitant la communication entre les gestionnaires de sites et les plaisanciers. Elle permet de sensibiliser et d’appeler à des pratiques nautiques plus durables et contribue également à l’émergence des observatoires citoyens.

2 Cartographie de l'état écologique des herbiers à posidonie

Le mode de propagation clonale des herbiers à posidonie leur permet de vivre plus de 100 000 ans battant ainsi tous les records de longévité (Arnaud-Haond et al., 2012). Cependant, les herbiers ont une très faible résilience comme le montre notre étude sur l'analyse comparative des photographies aériennes permettant de voir les traces des bombes d'il y a 70 ans toujours visibles actuellement. Chaque perte est quasi irréversible, ce qui souligne l'importance de combiner tous les moyens disponibles pour éviter d'endommager les prairies à posidonie.

Nos études confirment la régression des limites des herbiers à posidonie, et en précisent l'ampleur dans la tranche infralittorale 0 à -15 m avec 73 % du linéaire de la limite supérieure en déclin en région PACA et une perte surfacique de 13,4 % (face à 85 % en stabilité) au cours des 85 dernières années. Si ce chiffre peut sembler faible, cela représente au moins (seules les surfaces qui ont pu être comparées avec une certitude maximale sont considérées), 1 029,9 ha d'herbier disparus en 85 ans, soit 12,1 ha par an ou 332 m² par jour depuis 1920 sur une seule région française.

A l'échelle de la méditerranée française, en considérant nos chiffres (70 641 ha d'herbier à *Posidonia oceanica* et 5 692 ha de matte morte) nous estimons à au moins 7,45 % la perte surfacique des herbiers à *Posidonia oceanica*. L'ensemble de ces chiffres confirme la perte globale (entre 13 et 38 %) récemment estimée par Marbà et al. (2014a) et les 10 % généralement admis (Boudouresque et al., 2012). Ces valeurs concernent essentiellement la Méditerranée Occidentale du Nord tandis qu'un manque de données relatives à la Méditerranée méridionale et orientale est observé, il est ainsi aujourd'hui difficile de généraliser à l'ensemble du bassin.

Pour cette plante protégée dont les pertes peuvent difficilement être compensées ou restaurées, les séquences "éviter" et "réduire", lors d'un aménagement, doivent être sérieusement prises en compte. Ceci est d'autant plus important que l'estimation du coût réel de ces pertes est difficile. Un travail récent a identifié 25 services fournis par les herbiers à *Posidonia oceanica* (Campagne et al., 2015). Sur la base de l'évaluation économique de sept biens et bénéfices issus de onze de ces services, la valeur minimale totale des herbiers a été estimée entre 284 et 514 € l'hectare par an, ce qui équivaut en France (pour 70 641 ha) à un chiffre entre 20,1 et 36,3 millions € par an. Le déclin de 7,45 % observé en 85 ans engendre une perte de bénéfices pour notre société, en termes de services rendus, entre 1,62 millions et 2,92 millions d'euros par an. Ces chiffres sont un minimum, l'ensemble des services écosystémiques liés à l'herbier n'ayant pu être évalué (11 sur 25 !) de par le manque de données disponibles.

A l'échelle plus globale, une régression de 13 % généralisée à l'ensemble de la Méditerranée (3,5 millions d'hectares estimés (Laffoley and Grimsditch, 2009) représenterait une perte minimale comprise entre 129,22 et 233,87 millions d'euros par an.

3 Estimation et analyse de la diversité du Coralligène

3.1 La récolte des données

Suite aux menaces humaines actuelles sur les écosystèmes naturels, le suivi de la biodiversité est devenu un objectif central en écologie. À cette fin, des programmes de surveillance ambitieux sont maintenant utilisés pour fournir des cartes de la biodiversité à large échelle afin de concevoir des stratégies de conservation appropriées pour les écosystèmes terrestres et marins (Schmeller et al., 2015; Tittensor et al., 2014). Bien que ces programmes sont maintenant disponibles pour de nombreux taxons tels que les plantes terrestres (par ex (Thuiller et al., 2005)) ou les vertébrés (par ex (Zupan et al., 2014)), ils sont moins courants pour d’autres qui ne sont pas facilement accessibles en raison de leur taille ou de leur emplacement. Les communautés benthiques marines, en particulier les communautés coralligènes sont difficiles à étudier car elles se trouvent à des profondeurs qui ne permettent pas une observation facile. De plus leur hétérogénéité nécessite de grandes zones d'échantillonnage ainsi que des temps de plongée longs en raison de leur profondeur élevée (Kipson et al., 2011).

Ainsi, lors de la mise en place du programme RECOR en 2010, nous avons préféré utiliser une technique non invasive, basée sur un échantillonnage photographique, ce qui est recommandé pour les écosystèmes marins (PNUE and CAR/ASP - Centre d’Activités Régionales pour les Aires Spécialement Protégés, 2011) et couramment utilisé (Baldacconi and Corriero, 2009; Kipson et al., 2011; Piazzi et al., 2005). Cependant, cette approche ne permet pas d'identifier tous les individus au niveau de l'espèce (notamment pour les espèces cryptiques). Dans notre analyse, nous avons adopté une approche prudente en gardant certaines identifications au niveau du genre ou de groupes supérieurs. Le nombre d'espèces est ainsi sous-estimé pour nos stations mais ce biais est commun à toutes, permettant ainsi des comparaisons, ce qui était l'objectif principal de ce travail. En correspondance avec cette liste d’espèces, nous avons construit la liste de leurs traits fonctionnels sur la base d’une synthèse bibliographique. Les traits spécifiques en rapport direct avec le fonctionnement de l'écosystème n’étant pas encore clairement identifiés pour les concrétions à coralligène, rendent l'estimation de la diversité fonctionnelle de ces écosystèmes particulièrement difficile. Ici, nous avons estimé la diversité fonctionnelle d’un ensemble d'espèces caractéristiques de cet écosystème sur la base de leur morphologie, reproduction et de leur écologie. Les futures études sur les écosystèmes coralligènes pourraient permettre de mieux distinguer des aspects particuliers du fonctionnement de cet écosystème comme la résistance aux espèces envahissantes, aux agents de maladie ou des traits liés aux flux de matière et d’énergie afin de permettre des estimations plus justes de la diversité fonctionnelle.

3.2 Influence de la profondeur et de la distance géographique sur la biodiversité… conséquences pour la conservation

L’utilisation de la base de données Recor a permis, par krigeage la réalisation des premières cartographies à grande échelle des trois facettes de la diversité : taxonomique, fonctionnelle et phylogénétique (Figure 22).

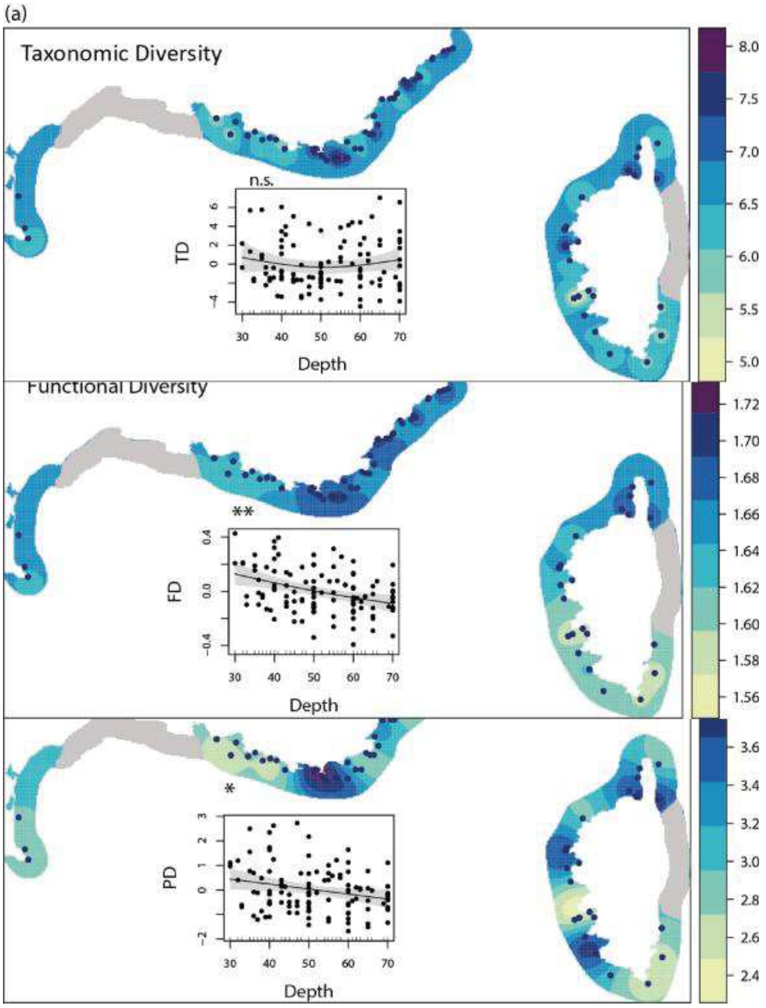


Figure 22 : Distribution Spatiale des trois facettes de la diversité α: Taxonomique (a), Fonctionnelle (b) et Phylogénétique (c).

Nos résultats montrent une relative redondance des fonctions au sein de la plupart des stations. En outre, nos résultats montrent que la diversité taxonomique n’est pas spatialement congruente avec la diversité fonctionnelle et phylogénétique. Cela indique que ces diversités fournissent des informations complémentaires pour la compréhension des assemblages à coralligène. Plus intéressant encore, les diversités fonctionnelles et phylogénétiques sont hautement congruentes, avec des modèles de distribution spatiaux similaires sur les axes verticaux (la profondeur) et horizontaux (distance géographique). Nous supposons ainsi un regroupement fonctionnel et phylogénétique dans les peuplements coralligènes. Ceci les rend encore plus vulnérables au changement climatique ou aux perturbations anthropiques. En effet, les espèces partageant des adaptations similaires et une tolérance à l'environnement restreinte, répondent de façon similaire aux perturbations ; ainsi même de petits changements futurs pourraient avoir des conséquences néfastes sur le fonctionnement de cet écosystème (Mouquet et al., 2012; Srivastava et al., 2012).

A l'échelle locale, intra-site (diversité α), c’est la profondeur qui structure diversités fonctionnelles et phylogénétiques, avec des stations peu profondes étant en moyenne plus riches que les stations profondes. Aucun lien significatif n'a été observé entre profondeur et diversité taxonomique mais une légère structure horizontale a été observée pour toutes les facettes de la diversité, avec les communautés les plus riches étant situés dans la partie Nord-Est de notre zone d'étude (PACA Est et le Nord de la Corse).

Ces dissemblances taxonomiques, fonctionnelles et phylogénétiques entre les assemblages (diversité β) augmentent avec la profondeur principalement dans la région centrale et orientale de PACA et dans le nord de la Corse, où le gradient vertical est élevé. Nous n’avons observé aucune structuration verticale dans d'autres régions. L’absence de verticalité peut signifier une connectivité verticale plus forte qu’horizontalement en raison de caractéristiques propres aux espèces et/ou aux conditions de l'environnement (par exemple la turbidité et les perturbations anthropiques pourraient contribuer à des environnements homogènes verticalement), conduisant ainsi à une homogénéisation biotique.

Cette analyse multifacette de la diversité en trois dimensions est une nouvelle approche pour la conservation marine. La variabilité verticale et horizontale peut apporter une contribution utile pour la planification de la conservation, indiquant si l'approche la plus efficace pour la conservation est locale (verticale) et / ou régionale (horizontale) (Baselga, 2010) d’autant plus pour les récifs coralligènes qui montrent une forte fragmentation (voir le chapitre sur la cartographie). L’approche de conservation locale utilisée habituellement longe directement les côtes et ne protège bien souvent pas les espèces caractéristiques des habitats plus profonds ; c’est ainsi que je me suis par exemple rendu compte en 2014 que le parc national de Port Cros-Porquerolles n’englobait aucune colonies de corail rouge toutes situées en périphérie et donc à l’extérieur de la réserve (Holon et al, en préparation pour les Cahiers de Port-Cros) ou que la réserve naturelle de Scandola délimitée de Cap à Cap ne tient pas compte de la présence des concrétions de coralligène à sa proximité. Contrairement à l'approche plus classique de réserves marines qui se concentrent sur la protection de sites à la verticale, la diversité horizontale pourrait être favorisée par une multiplication des réserves et/ou la mise en œuvre de mesures de conservation qui prennent en compte des tranches de profondeur à plus grande échelle. Ainsi dans certaines régions, une préservation plus efficace des différences biotiques pourrait être atteinte en réglementant les activités humaines à certaines profondeurs, ajoutant ainsi un aspect horizontal aux réserves marines. Ces données en appui sur des données de connectivité (encore très rares pour cet écosystème mais voir Guizien et al., 2014; Ledoux et al., 2010; Martínez-Quintana et al., 2014; Shanks et al., 2003) pourraient aider à améliorer la connexion au sein du réseau des aires marines protégées (Andrello et al., 2015).

L'amélioration de la conservation des assemblages à coralligène nécessite ainsi de tenir compte de la particularité de cet écosystème vertical, afin de préserver les dissemblances entre les concrétions coralligènes liées à la profondeur. Les mesures de conservation locale devraient être intégrées dans une stratégie plus large échelle afin d'identifier les zones et les ceintures de profondeur qui maintiennent les niveaux de diversité plus élevés. Bien que les analyses en trois dimensions soient complexes et constituent un défi pour les écologues et biogéographes, elles devraient être davantage prises en compte afin d'aborder les différentes facettes de la diversité biotique et proposer des moyens adéquats de conservation de la biodiversité à différentes échelles.

3.3 L'analyse de la biodiversité par l'analyse directe d'images

La méthode Recor que nous avons développée et appliquée (Deter et al., 2012a), est basée sur un échantillonnage photographique. Elle nécessite une détermination visuelle longue et parfois l'intervention de différents experts en taxonomie. Une alternative à l'identification visuelle pourrait provenir du traitement de l'image par analyse spectrale. Initialement développée pour mesurer la complexité physique de l'environnement (Proulx and Parrott, 2008), cette méthode a récemment été utilisée pour analyser la biodiversité benthique (Lambert et al., 2013; Mellin et al., 2012; Tanner et al., 2015). Fondamentalement, le procédé décompose des images en bandes spectrales et combine ces informations dans un index de complexité structurelle multivariable. Étant donné que pour les communautés benthiques coralligènes, la complexité de l'habitat est le résultat de constructions biogènes et l'accumulation des organismes fixés, il est probable que cet indice de l'hétérogénéité physique pourrait être liée à la biodiversité réelle. Par exemple Tanner et al. (2015) ont analysé les invertébrés benthiques et la biodiversité des algues dans le golfe St Vincent en Australie du Sud, et ont constaté que deux mesures de la complexité structurelle basées sur le MIG de Shannon (Mean information Gain) qui mesure l'hétérogénéité spatiale et le MMI (Mean Mutual Information) qui mesure la corrélation spatiale expliquaient jusqu'à 35 % de la variation de la richesse taxonomique, 25 % de l'uniformité (« evenness »), et 50 % de la dissemblance des taxons. Bien qu'elles soient inférieures à ce qui pourrait être obtenu par échantillonnage direct, ces valeurs sont comparables à d'autres mesures de substitution de la biodiversité benthique (Huang et al., 2012; Przeslawski et al., 2011).

L'utilisation de l'analyse spectrale pour déduire la biodiversité marine n'est encore qu'à ses débuts et fait l'objet de discussions dans la communauté scientifique (Lambert et al., 2013; Tanner et al., 2015), mais elle est une méthode prometteuse que je souhaite poursuivre afin de continuer le travail initié lors du stage de Guillaume Cornette que j'ai encadré au début de ma thèse.

Lors du stage de Guillaume Cornette, nous avons trouvé que les différents indices calculés à partir des images sont expliqués par les variables environnementales et biologiques estimées sur l'ensemble des photographies d'une station. Ainsi environ 50 % de la variance des différents indices est expliquée par les pourcentages de recouvrement d'espèces bioconstructrices. L'abondance totale et le pourcentage de recouvrement par du vivant sont également significativement impliqués dans la complexité de l'image. Les modèles mis en place pour approximer la diversité au sein des stations expliquent 10 à 17 % de la variance des indices de Shannon et de Simpson. Environ 20 % de la variance de l'abondance totale a également été expliquée par un modèle combinant le Marginal et le MIG. Ces scores sont plus faibles que ceux obtenus par Mellin et al. (2012) pour des écosystèmes coralliens de la Grande Barrière de Corail. Notre analyse se base sur l'étude des microstructures au sein d'un habitat très complexe mais dont la texture est relativement similaire d'une station à l'autre. Des améliorations notamment dans la recherche des modèles les plus significatifs, ou dans le nombre de classes prises en compte dans l'analyse, pourraient probablement permettre de meilleurs scores. Elle pourrait être utile en fournissant une évaluation rapide de la biodiversité benthique marine à de grandes échelles spatiales et temporelles où les observations directes sont impossibles.

4 Cartographie et quantification des pressions anthropiques côtières

Notre carte des impacts cumulés des pressions anthropiques sur les habitats marins permet de visualiser à une échelle fine (pixel de 20m*20m) la distribution et l'hétérogénéité des pressions et de leurs impacts sur tout le littoral méditerranéen français.

La comparaison directe avec les résultats d'études précédentes (Coll et al., 2011; Giakoumi et al., 2013) est difficile ; celles-ci étant à une échelle plus large (pixel de l'ordre du Km), les données sur la zone côtière ne semblent pas fiables (en raison de l'échelle plus fine et / ou de la plus grande diversité d'habitats considérés). Cependant, la comparaison visuelle entre ces cartes et nos résultats suggèrent qu'à leurs points de jonction, elles semblent concordantes et se compléter mutuellement. Concernant les fonds peu profonds, à proximité du littoral, nos observations confirment que les pressions sont essentiellement concentrées entre 0 et -15 m où les habitats marins les plus sensibles sont également en développement ; exemple des herbiers en régressions importantes à cet étage bathymétrique, notamment en raison de l'artificialisation des côtes (Holon et al., 2015; Marbà et al., 2014) (voir le paragraphe « cartographie de l'état écologique des herbiers »).

Selon notre étude, à l'échelle de la méditerranée française, les pressions les plus importantes (en tenant compte à la fois de la surface et de l'intensité) sont l'urbanisation, la population côtière, l'érosion côtière et l'aménagement du littoral, qui sont directement liés à l'évolution côtière et à la planification territoriale. Presque tous les habitats sont affectés par toutes les pressions. Cela souligne l'importance des plans d'action axés sur les pressions en lien avec les habitats marins côtiers. Ainsi nos résultats pourraient aider les parties prenantes à localiser et hiérarchiser leurs actions politiques.

Savoir précisément où se trouvent les habitats sensibles et comment ils sont touchés est indispensable avant le déploiement des mesures d'atténuation adéquates (Giakoumi et al., 2013). Notre travail peut également aider à améliorer l'efficacité d'actions de gestion. Par exemple, une fois l'ancrage ciblé comme une pression majeure dans une région, les gestionnaires peuvent intervenir et mettre en place différentes actions comme l'interdiction d'ancrage, des bouées d'amarrage, sensibiliser les plaisanciers et leur faire connaitre la nature des fonds (via l'application Donia par exemple (Andromède Oceanologie and Agence de l'eau RMC, 2014)). Notre travail peut aider les parties prenantes à prioriser leurs moyens: protéger les zones où les impacts cumulatifs sont faibles ou très faibles et agir sur les pressions «contrôlables» où les impacts cumulatifs sont de moyens à élevés. Enfin, ces cartes sont comme des photos de l'état de la côte à un temps donné et pourrait être actualisées dans cinq à six ans afin de mesurer les changements éventuels, comparer les sites similaires et tester l'efficacité des choix de gestion différents.

Dans la perspective d'une actualisation, des améliorations pourraient être proposées. Par exemple, les données actuellement mises à disposition via la plateforme Medtrix pourraient être affinées par les utilisateurs (échantillonnage participatif) concernant les sources des pressions, nous pourrions également utiliser des dénominations plus précises pour les habitats (différents niveaux au sein des types d'habitats, les mesures de vitalité pour *P.oceanica*, les données sur les assemblages d'espèces), de même, d'autres pressions pourraient être ajoutées si les données devenaient disponibles à l'échelle utilisée comme le changement climatique, les espèces exotiques envahissantes, la pêche industrielle (y compris le chalutage), les activités de plongée ou le trafic maritime par exemple.

Ces cartes sont d'une grande utilité pour la conservation de la biodiversité (et principalement les habitats sensibles aux pressions). Elles aident à communiquer, à prioriser les questions d'environnement, faire des choix politiques, mieux comprendre les liens entre les pressions anthropiques et l'état écologique des systèmes côtiers, et à appliquer et à coordonner les mesures de gestion. Combinées avec des cartes de répartition des espèces, elles peuvent donner lieu à des cartes de perte de biodiversité potentielle. Pour toutes ces applications, ces cartes peuvent largement servir à évaluer l'atteinte des objectifs des directives européennes.

5 Liens entre état des écosystèmes et pressions anthropiques côtières

5.1 Impact des pressions sur les herbiers

Sur la base des cartes d'habitats et des pressions obtenues précédemment (pixels de 50 x 50 m), j'ai pu relier l'état d'un écosystème avec les pressions auxquelles il est soumis. J'ai choisi de développer un modèle prédictif sur les herbiers de posidonie, car c'est un habitat particulièrement sensible, proche des côtes et pour lequel l'état pouvait être estimé. J'ai ainsi obtenu un modèle performant montrant que la plupart des pressions présentait un effet négatif non linéaire sur le statut écologique de l'écosystème étudié. Des seuils ont été détectés avec des valeurs variant en fonction de la pression considérée ; cela confirme que toutes les pressions n'ont pas des impacts et surtout des seuils d'impact similaires et ne peuvent pas être gérées de la même façon. Nos résultats sont des données utiles aux gestionnaires ; les seuils de résistance des herbiers aux pressions traduits en mesures concrètes intéressent des questions cruciales comme par exemple la distance minimale à respecter pour la construction d'un effluent d'eaux usées ou une ferme aquatique afin de minimiser l'impact sur les herbiers ou le nombre d'ancrage maximum des bateaux à ne pas dépasser sans l'associer à une gestion plus adaptée. Associé à des données encore incomplètes sur les services écosystémiques (Campagne et al., 2015), ces données permettront de construire ou d'affiner les analyses bioéconomiques (par exemple (Rabassó and Hernández, 2015) pour l'élevage de poissons) nécessaires afin de faire la part entre le coût / bénéfice et de choisir la solution la moins délétère tout en restant économiquement viable.

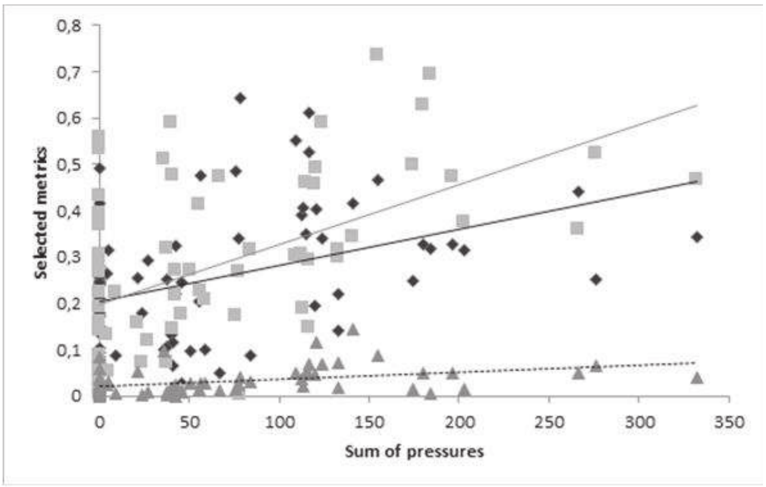
Nos résultats sont dans la continuité des précédents modèles spatiaux développés récemment afin de relier des pressions multiples avec le statut écologique de différents écosystèmes. Ces modèles permettent une approche prédictive des conséquences de différentes options de gestion (Bianchi et al., 2012; Parravicini et al., 2012a; Stelzenmüller et al., 2010; Vacchi et al., 2014). Celles-ci peuvent être ciblées, dans le cadre de notre étude, par des mesures d'atténuation afin d'éviter le basculement au-delà des seuils. Dans le même temps, elles peuvent également être associées à des actions de restauration en ce qui concerne les zones de dépassement de seuil. Alors que la carte finale combinant les différents facteurs prédictifs est intéressante pour une analyse régionale (définir des zones ayant une problématique de conservation ou des capacités de restauration), les acteurs et les gestionnaires locaux peuvent analyser les différentes cartes individuelles des pressions, afin de savoir sur quelle pression ils doivent agir en priorité et connaitre les seuils à ne pas dépasser localement. Certaines pressions sont relativement faciles à moduler (pêche, ancrage, aquaculture ou les effluents dans une moindre mesure) tandis que d'autres sont « imposées » (population côtière).

La méthodologie employée a pour objectif d'aider à maintenir les ressources marines en bon état et à définir les zones prioritaires d'actions par exemple concernant la gestion des pressions, la restauration écologique ou la conservation d'habitats. Atteindre le bon état écologique des eaux marines de l'UE est précisément l'objectif de la directive-cadre stratégie pour le milieu marin de l'Union européenne (DCSMM, 2008/56 / CE). Le modèle développé ici peut facilement être reproduit avec d'autres écosystèmes, à condition que les données soient disponibles, comme il peut servir à faire des prédictions en fonction de scénarios de développement socio-économiques. Il peut également être appliqué pour estimer et prédire l'impact du changement climatique (et ses conséquences sur l'eau de mer (température, acidification, salinité, turbidité) sur ces écosystèmes.

5.2 Le développement d'indicateurs

Les cartes de pressions cumulées peuvent également être utilisées pour le développement des indicateurs demandés par les directives cadres européennes à savoir des indicateurs d'intégrité des fonds marins pour la DCSMM ou de qualité écologique des eaux côtières pour la DCE. En effet, la mise au point de ces indicateurs nécessite de mettre en lien des variables biologiques avec les pressions concomitantes. Ainsi des données issues d'une même méthodologie disponibles tout le long de la côte sont une belle avancée pour l'amélioration des travaux existants. Pour les herbiers de posidonie, on peut citer le PREI (*Posidonia oceanica* Rapid Easy Index (Gobert et al., 2009)), le BiPo (Biotic index using *Posidonia oceanica* (Lopez y Royo et al., 2010)) et le POMI (*Posidonia oceanica* multivariate index (Romero et al., 2007)). Pour le coralligène, on peut citer l'ESCA (Ecological Status of Coralligenous Assemblages index (Cecchi et al., 2014)) et le CAI (coralligenous assemblages index (Deter et al., 2012a) basés sur des échantillonnages photographiques et le COARSE (COralligenous Assessment by ReefScape Estimate (Gatti et al., 2015)) basé sur des observations en plongée. Une analyse préliminaire des données RECOR sur 70 stations (une par site) montre par exemple que le recouvrement par le sédiment, l'abondance relative de bryozoaires encroûtants et d'algues rouges encroûtantes sont influencés et de façon positive par les pressions cumulées (respectivement, $R^2 = 0,17$; 0,18 et 0,19 ; tous les $P < 0,001$, Figure 23). Ces analyses restent à affiner afin de valider et/ou améliorer le CAI basé initialement sur le recouvrement de sédiment, de bryozoaires et d'espèces bio-constructrices (Deter et al., 2012a).

Figure 23 Trois métriques en lien avec les pressions cumulées : couverture de sédiments (diamants et ligne noire), l'abondance relative des algues rouges encroutantes (carrés et ligne grise) et l'abondance relative de bryozoaires encroutant (triangle et pointillés), en fonction de la somme des pressions sur les 70 sites étudiées. Les lignes représentent les courbes de tendance.



6 Diffuser le savoir avec la base de données cartographiques MEDTRIX

Depuis que je travaille en lien avec les réseaux de surveillance, je me suis rendu compte de la difficulté de connaître les différents réseaux existants et d'obtenir les résultats et données issues de ces réseaux. Les causes sont la multiplicité des acteurs, le mélange de différents fonds publics et privés, l'absence de budgets prévus pour la mise à disposition de ces données et la crainte de partager ces données. En partenariat avec l'Agence de l'eau RMC, j'ai participé durant ce travail de thèse à l'élaboration d'un outil permettant une communication facile et rapide des résultats sous format cartographique. Medtrix (http://www.medtrix.fr) est une plate-forme cartographique de regroupement de plusieurs bases de données disponibles pour les professionnels de la mer: scientifiques, gestionnaires et autres parties prenantes. L'ensemble des données produites dans le cadre de ce projet doctoral y compris les données sources y sont disponibles après une inscription gratuite. La plateforme offre différentes fonctionnalités comme l'édition de cartes, le téléchargement de données SIG ou la possibilité de comparer statistiquement les sites entre eux. Même si un tel outil n'offre aucune reconnaissance académique, j'espére qu'il rendra des services à ses utilisateurs et qu'il ouvrira la voie à de futures collaborations mixant les différentes approches.

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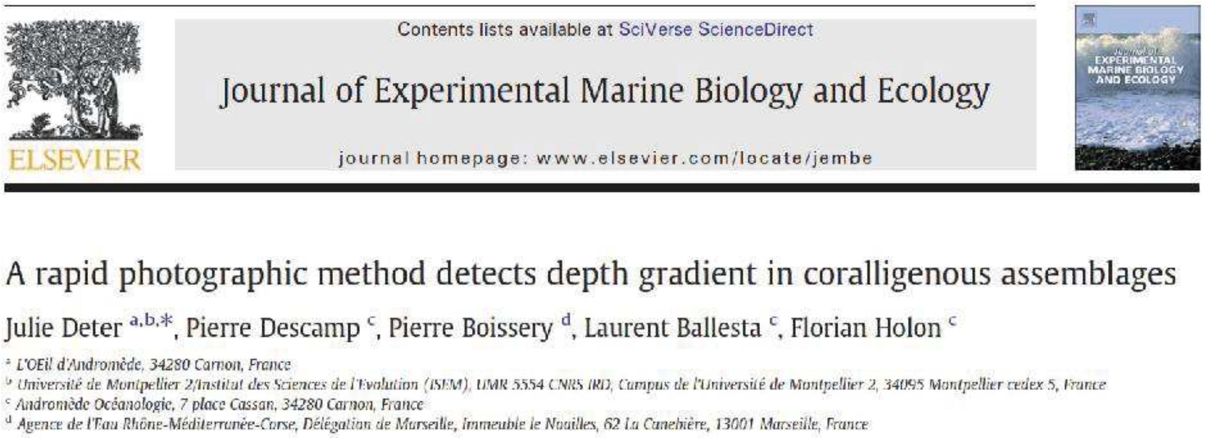
Annexe 1 : Une méthode photographique rapide détecte un gradient de profondeur dans les assemblages coralligènes



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Julie Deter [a,b,*], Pierre Descamp [c], Pierre Boissery [d], Laurent Ballesta [c], Florian Holon [c]

[a] L'OEil d'Andromède, 34280 Carnon, France
[b] Université de Montpellier 2/Institut des Sciences de l'Evolution (ISEM), UMR 5554 CNRS IRD, Campus de l'Université de Montpellier 2, 34095 Montpellier cedex 5, France
[c] Andromède Océanologie, 7 place Cassan, 34280 Carnon, France
[d] Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France

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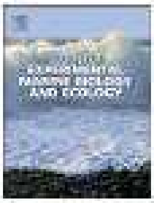

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Julie Deter [a,b,*], Pierre Descamp [c], Pierre Boissery [d], Laurent Ballesta [c], Florian Holon [c]

[a] L'OEil d'Andromède, 34280 Carnon, France
[b] Université de Montpellier 2/Institut des Sciences de l'Evolution (ISEM), UMR 5554 CNRS IRD, Campus de l'Université de Montpellier 2, 34095 Montpellier cedex 5, France
[c] Andromède Océanologie, 7 place Cassan, 34280 Carnon, France
[d] Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France

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ABSTRACT

Coralligenous assemblages represent the unique calcareous formations of biogenic origin in the Mediterranean Sea. Despite their importance in terms of biodiversity and biomass production, community analyses remain scarce. Actual sampling of these assemblages is complicated to carry out because their depth distribution (down to −120 m) necessitates complex diving logistics. We highlight a rapid, cost-effective, objective and accurate method for the sampling of coralligenous assemblages and tested its efficiency in delineating a depth gradient. We compared seven photographic methods for estimating the percentage cover of sessile organisms: visual estimates (VS) with the aid of a 25, 64 or 100 square-grid and random-point-quadrats (RQ) with 25, 64 or 100 random points or 64 stratified random points. Comparisons were made using two simulated quadrats for which percent cover values were known. RQ with 64 random points was the method that accumulated the highest number of advantages. Using this method, two field sites were sampled by divers at three depths (−50, −60 and −70 m) with increasing replication (10, 20, 30 and 40 photographic quadrats). The communities deduced from the 30 and 40 photos were similar. Community analyses showed an effect of depth nested in site on the assemblages observed. With increasing depth, encrusting algae get replaced by Porifera. Dissimilarity between −50 m and −60 m/−70 m was mainly due to *Crambe tailliezi* abundance. This methodology will be a useful tool for managers and administrators: it guarantees fast abundance estimation, non-destructive repeated sampling, the possibility of comparison among researchers and the permanent record of deep-sea communities.

1. Introduction

After *Posidonia oceanic* meadows, "coralligenous assemblages" are considered as the second biodiversity hot spot in the Mediterranean Sea (Boudouresque, 2004). Coralligenous assemblages colonize the deep littoral systems of the Mediterranean Sea from 12–50 m to 40–120 m depth depending on water transparency (Ballesteros, 2006). Coralligenous concretions are primarily produced by the accumulation of encrusting algae growing at low light levels and secondarily by bio-constructor animals such as polychaetes, bryozoans and gorgonians: they represent the unique calcareous formations of biogenic origin in the Mediterranean Sea (Ballesteros, 2006). The resulting complex structure allows the development of a patchwork of communities dominated by living algae, suspension feeders, borers or soft-bottom fauna (in the sediment within cavities). Two main morphologies can be distinguished (Ballesteros, 2006) for coralligenous frameworks: banks (built over more or less horizontal substrata) and rims (in the outer part of marine caves and on vertical cliffs). In terms of richness, biomass and production, coralligenous assemblage value is high and comparable to tropical reef assemblages (Bianchi, 2001). The engineering species composing these assemblages are fragile, present long life expectancy and low dynamics leading to a particular susceptibility of coralligenous assemblages (in their specific composition and structure) to anthropogenic disturbances (waste water, invasive species, fishing activities and divers) (Ballesteros, 2006). Identified under Community legislation (Habitats Directive 92/43/CEE; habitat code 1170-14: Reefs, coralligenous assemblage) as being a special habitat with biodiversity interest, coralligenous assemblages shall be monitored as a descriptor in the framework of Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC). MSFD aims to achieve good environmental status of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend. The final goal is to protect more effectively the marine environment across Europe but an important point is that each Member State "shall ensure that measures are cost-effective and technically feasible".

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* Corresponding author at: L'OEil d'Andromède, 7 place Cassan, 34280 Carnon, France. Tel.: +33 4 67 66 32 48; fax: +33 9 57 53 90 94.

E-mail addresses: julie.deter@andromede-ocean.com, julie.deter@univ-montp2.fr (J. Deter), pierre.descamp@andromede-ocean.com (P. Descamp), pierre.boissery@eaurmc.fr (P. Boissery), laurent.ballesta@andromede-ocean.com (L. Ballesta), florian.holon@andromede-ocean.com (F. Holon).

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Action plans for the conservation of the coralligenous and other calcareous bio-concretions highlighted a lack of standardized procedures for characterizing these habitats (UNEP-MAP-RAC/SPA, 2008, 2011). Moreover, Kipson et al. recently claimed that "methods are urgently needed to assess prevailing patterns, evaluate impacts to which they [coralligenous outcrops] are subjected and provide baseline data to explore future trajectories of these high diversity assemblages" (Kipson et al., 2011). For these reasons and because they cumulate study constraints such as various systems, slow dynamics, and more importantly presence at generally high depths limiting work time underwater, coralligenous assemblages are relatively poorly understood at the community level (Kipson et al., 2011). Saving time underwater is one of the advantages of photographic methods that are more and more frequently used (Balata et al., 2005; Baldacconi and Corriero, 2009; Deter et al., 2012; Ferdeghini et al., 2000; Kipson et al., 2011; Virgilio et al., 2006). They allowed counting the number of species and/or estimating species abundance using percent cover. Percent cover is used to quantify organisms that cover the substrate and have modular body organization, including most macroalgae, sponges and colonial animals such as corals, bryozoans and ascidians (Benedetti-Cecchi et al., 1996; Boudouresque, 1971). Feasibility and cost-efficiency but also rapidity, objectivity, repeatability, accuracy and sensitivity are all important criteria to be considered in the choice and/or development of a monitoring method.

Percent cover estimated from digitized images produced similar results (with equivalent quadrat size) (Parravicini et al., 2009) and were more repeatable and objective than direct estimates in the field (Meese and Tomich, 1992). This is especially true for the random-point method (described below) (Alquezar and Boyd, 2007). Despite the risk of misidentification, the main advantage recognized for digital methods is its rapidity (Macedo et al., 2006). However the necessity to spend time in the laboratory for the processing of pictures may discourage its employment for monitoring. Thus, the challenge lies in finding a good balance between time constraints (both underwater and in the laboratory), cost and efficiency by identifying a reliable technique for picture processing and assessing the replication effort needed.

Two methods are generally used for abundance estimation from photographic quadrats: visual methods (VS) and random-point methods (RQ). VS consists of evaluating percent cover with the aid of a grid projected on to the photographic quadrat. With RQ, a specified number of spatially random points are overlaid on the image and the features (species or substrates) lying beneath each point are user-identified. Percent cover is then calculated from the different number of points identified per species. With manual projections (grid or points on transparencies) both methods took comparable amounts of time and were similarly repeatable for abundant species (>10%) (Dethier et al., 1993). Thanks to CPCe (Coral Point Count with Excel extensions), a user-friendly software (Kohler and Gill, 2006), RQ is now easier to perform. This software reduces the time spent (image preparation and analysis) for RQ by grouping in a unique interface picture enhancement, point distribution and identification. Percent cover is then calculated and the results sent to the Excel spreadsheet automatically. First developed for coral reef studies, CPCe is easily adaptable to other assemblages by recoding the species to be identified (Kohler and Gill, 2006); it was consequently also applied to artificial habitats, rocky intertidal assemblage, and coralligenous assemblages sampling (Macedo et al., 2006; Màrquez I Canals, 2006; Pineda, 2007; Zintzen et al., 2008).

The present study aimed at testing and comparing the rapidity (time required), objectivity (variability among observers), repeatability (within-observer variance), accuracy (divergence of the results from the real values) and sensitivity (proportions of missed species occurrences, especially the rarest ones) between VS and RQ (with different number of points and type of projection) using simulated digitized photos. We tested the hypotheses that RQ was faster, more objective and more repeatable than VS for equivalent accuracy and sensitivity. Secondly, we evaluated the performance of the best method combination (method × number of points, selected from preceding tests) in the field to detect a depth gradient, and the number of pictures required (sampling effort needed).

2. Materials and methods

2.1. Methods tested with simulated quadrats

Two 0.25 m^2 quadrats (50×50 cm) were created using The GIMP (The GIMP Team, 2010. GNU Image Manipulation Program, www.gimp.org) 2.6 software to draw the distribution of *nine* and *eleven* "species" (one color per species, Table 1). The first simulated quadrat (quadrat A) mimicked highly interlocked distributions (Fig. 1A) while the second one (quadrat B) represented more patchy distributions (Fig. 1B). Three rearranged versions of each quadrat were created showing the same number of species and surface coverage but at different dispositions.

The two methods were tested on each simulated quadrat for the percent cover estimation (species cover in %): the visual method (VS) and the random-point-quadrat method (RQ). For VS, each photographic quadrat was divided into x small squares with the aid of a grid superimposed on the quadrat frame using The GIMP software. Species were identified and the percent cover estimated with the help of the grid; each small square totally filled by a species counted as $100/x$% cover, a square 3/4 filled count for $75/x$%, half filled for $50/x$% cover, etc. This mode eliminates the need for decision rules such as any square > half filled is counted as filled (Dethier et al., 1993). For RQ, CPCe 2.6 (Kohler and Gill, 2006) was used to project a number of points within the photographic quadrat frame. The distribution of these points was either totally random (no preferential spatial distribution of y points) or stratified random (projection of one point per cell within a grid of z cells dividing the quadrat frame). Species underlying the points were identified and percent covers automatically calculated.

Three observers analyzed the first version of quadrat A with VS (aid of $x=25$, 64 or 100 squares) and with RQ (aid of $y=25$, 64 or 100 random points or $z=64$ stratified random points). To avoid a possible bias due to user habit, all three observers were marine naturalists but non-trained to one or the other method. The number of 64 points was chosen after tests on three representative field quadrats

Table 1
Abundances (percent covers) and species represented on quadrats A and B.

Species	Percent cover
Quadrat A	
a	3.17
b	4.88
c	4.39
d	7.49
e	8.52
f	8.57
g	11.18
h	17.75
i	34.05
Quadrat B	
j	0.50
k	1.00
b	1.50
l	2.00
e	3.25
f	4.75
m	8.00
a	10.00
h	17.00
n	21.00
c	31.00

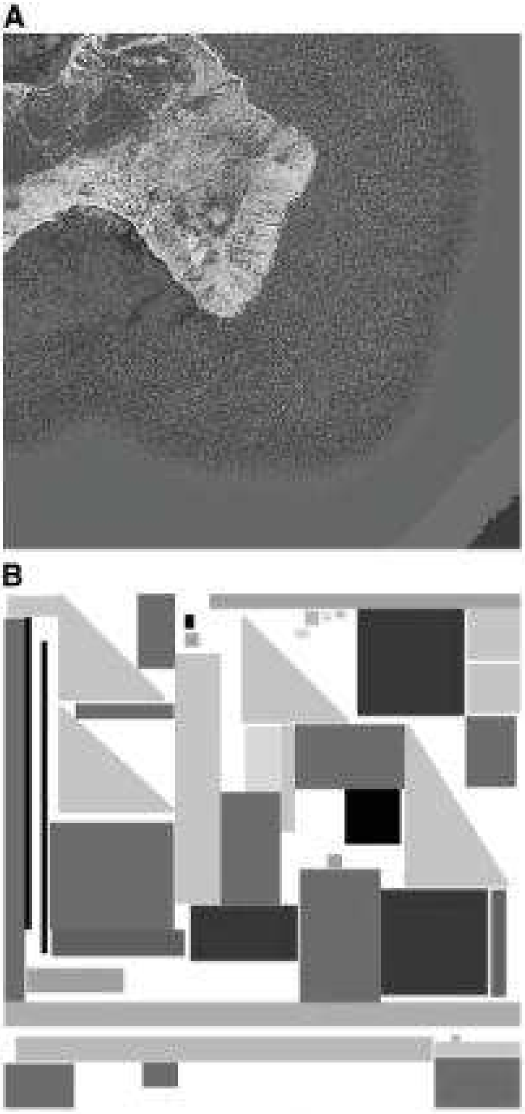

Fig. 1. First version of the simulated quadrats transformed in black and white for the publication. Quadrat A (A) mimicked a highly interlocked distributions of nine "species" when quadrat B (B) represented eleven "species" with patchy distributions (one color per species).

and varying numbers of points (9, 16, 25, 36, 49, 64, 81, 100, 144, 169, 196, 225 and 256 points per quadrat). Sixty-four points were needed to identify all the recognizable features of the pictures as more points did not add any more information. The numbers 25 and 100 were chosen to frame 64. Each combination (number of points × method) was performed three times per observer (three observers in total) once a week to avoid biases due to the observer remembering previous estimates. For the same reason, observers were not informed of the total number of simulated species. The time required for each quadrat sampling per method was noted. Rapidity (time required) and accuracy (divergence of the results from the real values = absolute value of the difference between observed and real values) were compared between groups of methods (two: VS and RQ) and observers (three) using analyses of variance (ANOVA) performed for repeated measures (three times). When a significant difference was found between groups of methods, post hoc tests (Kolmogorov–Smirnov tests or *t*-tests) were performed to highlight differences between combinations (number of points × method).

All dependent variables were normalized with adequate transformations. Differences in sensitivity (proportions of missed species occurrences) between the methods were tested with 2 × 2 contingency tables. Objectivity (variance between observers) was tested with non-parametric tests (*t*-tests).

The best combinations (method × number of points–squares) were tested by observer 1 with all versions of quadrats A and B added to the corresponding results obtained by observers 2 and 3 in order to test for the influence of the method and evaluate accuracy and sensitivity more precisely. Repeatability (variance for a same observer) was only tested with results obtained from observer 1 with an ANOVA for repeated measures. All of the statistical analyses were performed with Statistica 6.1 (Statsoft, Inc.).

2.2. Field data sampling

The aim of the field sampling was to test the ability of the method selected from tests with simulated quadrats in detecting a depth gradient and the number of pictures required (sampling effort needed).

Two sites localized in the Mediterranean Sea next to Fréjus (Var, France, Fig. 2) were sampled in June 2010 at three different depths (−50 m, −60 m and −70 m): Banc des vieilles (43°24′44″; 6°53′593″) and Chrétienne (43°26′371″; 6°55′443″). Both of these sites present similar coralligenous assemblages patterns (rim morphology according to Ballesteros, 2006) developed on vertical cliffs from −71 to −46 m at Banc des vieilles and from −70 to −50 m at Chrétienne. These sites were chosen because a depth gradient was clearly visible in assemblages according to the divers. The same diver took photos along a 20 meter-transect at each site. For practical and safety reasons, dives were performed using rebreathers INSPIRATION VISION (Ambiant Pressure Diving Ltd., United Kingdom). Gas recycling reduces the volume of breathing gas used, making longer dives possible with a lighter and more compact system than an open-circuit breathing set for the same duration (Bahuet et al., 2007).

Around 45 photographic quadrats per site were taken along the 40 m-transects at each depth (360 photos in total). One photo per 50 × 50 cm quadrat was taken using a digital camera (D2Xs Nikon at 12.4 megapixels with a 12–24 mm zoom-lens Nikon equipped with a housing, a dome and flashes SEACAM®). The camera was perpendicularly fixed 50 cm over the quadrat frame, thus minimizing possible parallax errors (Fig. 3). A compass and a depth meter were attached to the quadrat in order to keep homogenous the orientation (North at 12 h) and depth for the photos.

Forty photographic photos per depth within each site were randomly selected using the program Randomize (M. Rolland – Andromede

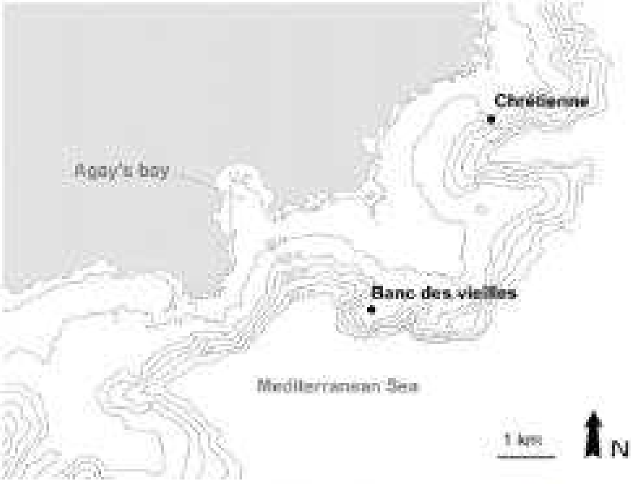

Fig. 2. Map presenting the field sites localized in Mediterranean Sea next to Fréjus (Var, France): Banc des vieilles (43°24′44″; 6°53′593″) and Chrétienne (43°26′371″; 6°55′443″). They were sampled in June 2010 at three different depths (−50 m, −60 m and −70 m).

Fig. 3. Photographic quadrat system used in the field. A digital camera was perpendicularly fixed 50 cm over the quadrat frame.

Oceanology) and enhanced for quality (contrast, luminosity, and sharpness) with The GIMP 2.6. The best method selected from simulated images was performed on the 40 photographic quadrats but also on 30, 20 and 10 of these images randomly chosen by the program Randomize.

Taxa (sessile organisms) were identified to the level of species or genus. Taxonomic nomenclature follows Appeltans et al. (2011) and Guiry and Guiry (2011). Where identification at the most detailed level of taxonomical resolution was not possible, animals were grouped in phyla. Hydrozoa and encrusting Bryozoa were not identified and classified as "Hydrozoa" and "Encrusting Bryozoa". Unidentifiable organisms were classified as "unknown" and were not considered in community analyses. Similarly, mobile organisms (fish, urchins) were not considered in community analyses. The matrix of species percent cover obtained on the base of the different number of photos was fourth-root transformed (because the data set was strongly dominated by certain variables) and analyzed with Primer 6.1.11 software (Primer-E), according to Clarke and Warwick (2001), Clarke and Gorley (2006) and Clarke et al. (2008). Similarity between samples was estimated using Bray–Curtis indices. A SIMPER breakdown was performed to determine the species that mostly contribute to the average similarity/dissimilarity within and between sites and depths. The existence of a depth gradient in species assemblages was tested using an ANOSIM: ANOSIM (two-way nested analysis of similarities) was performed on a similarity matrix and tested for a significant difference between sites and between depths within sites. The procedures SIMPROF and CLUSTER (hierarchical cluster analyses) were performed in order to test and visualize differences between the assemblages obtained from the different numbers of photos (10, 20, 30 or 40) and deduce the minimal number of pictures that did not change assemblage results (sampling effort needed). P-values were obtained with permutation tests (10,000 permutations).

3. Results

3.1. Testing the methods with simulated quadrats

3.1.1. Rapidity

All number of points–squares confounded (25, 64 or 100), ANOVA with repeated measures ($N=21$ observations repeated three times) showed a significant effect of the method used ($F=12.677$, $P=0.001$) while no effect of the observer ($F=2.023$, $P=0.099$) or of the interaction observer × method ($F=1.244$, $P=0.317$) was significant. VS was the slowest type of method (Table 2). Taking the number of points into account, RQ with 25 points was the fastest (mean $=65\pm22$ s), followed by RQ with 64 points (no difference between totally random (190 ± 18 s) or stratified random (183 ± 52 s), Kolmogorov–Smirnov test, $P=0.100$) (Fig. 4A). RQ with 64 points was faster than VS with 25 squares (Kolmogorov–Smirnov test, all $P<0.010$).

Table 2
Mean results (standard deviation = SD in parenthesis) obtained for each combination tested (number of points–squares × method). RQ = random-point quadrat with 25, 64 or 100 S = stratified or R = random points; VS = visual method with the aid of 25, 64 or 100 squares. All data were pooled (quadrat A with three observers). Rapidity referred to the amount of time required in seconds, sensitivity to the number of missed species on the number of occurrence, accuracy to the divergence from real values and objectivity to the variance of estimated percent covers between the observers.

	Rapidity	Sensitivity	Accuracy	Objectivity
RQS64	182.78 (52.28)	2/81	7.73 (6.54)	4.38 (5.43)
RQR25	65.67 (22.33)	12/81	3.10 (2.46)	3.453 (3.22)
RQR64	190.33 (18.15)	3/81	3.45 (3.10)	1.78 (1.31)
RQR100	412.33 (194.65)	1/81	3.25 (3.48)	1.439 (1.31)
VS25	485.00 (116.00)	5/81	3.11 (3.04)	8.91 (9.98)
VS64	1147.87 (399.54)	5/81	2.94 (2.84)	12.26 (20.66)
VS100	1200.67 (750.09)	3/81	1.78 (0.83)	19.92 (23.58)

3.1.2. Sensitivity

No difference in sensitivity was found between the methods (18/324 for RQ with 2/81 for RQ with stratified random points and 16/243 with totally random points and 13/243 with VS, 2 × 2 contingency table, all $P>0.566$). When species were missed, real percent cover was <5%. RQ with 25 random points was the combination point-method that missed the highest proportion of species (12/81)

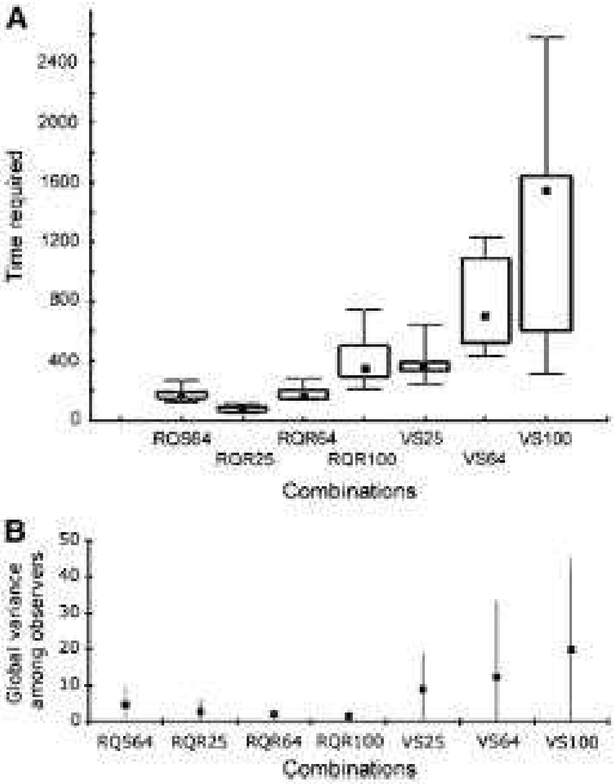

Fig. 4. Comparison of the results obtained from the first version of quadrat A analyzed three times per each observer for each combination (number of points–squares × method). A) Time required (in seconds) for the analysis. Black squares represented the median of the data, the box 25 and 75% of the data distribution and horizontal traits maximal and minimal values obtained on three observers. B) Global variance (for all species) among observers in percent cover estimation. Vertical traits represent standard deviation. RQ = random-point quadrat with 25, 64 or 100 S (= stratified) or R (= random) points; VS = visual method with the aid of 25, 64 or 100 squares. $N=189$ observations.

Table 3
List of taxa found at Banc des vieilles (B) and Chrétienne (C) at three different depths (−50, −60 and −70 m).

Taxa (alphabetical order)	Observed at ...
Annelida cl. Polychaeta	
Filograna implexa Berkeley, 1835 or *Salmacina* sp.	B60, C60
Other worms	All sites and depths
Bryozoa cl. Gymnolaemata	
Adeonella calveti Canu and Bassler, 1930	B70 and C50
Encrusting Bryozoa	C70
Myriapora truncata Pallas, 1766	B50, B60, B70, C50, C60
Non identified Bryozoa	All sites and depths
Pentapora fascialis Pallas, 1766	C50
Reteporella grimaldii Jullien, 1903	B60, B70, C50, C60, C70
Smittina cervicornis Pallas, 1766	B50, B60
Chlorophyta cl. Bryopsidophyceae	
Caulerpa racemosa f. *compressa* Weber-van Bosse, 1898	B50, C50, C70
Codium bursa (Olivi) C. Agardh, 1817	C50
Codium coralloides (Kützing) P.C. Silva, 1960	B50, C50, C60
Flabellia petiolata (Turra) Nizamuddin, 1987	B50, C50
Halimeda tuna (J. Ellis & Solander) J.V. Lamouroux, 1816	C50
Non identified macroalgae	All sites and depths
Chordata cl. Ascidiacea	
Ciona intestinalis Linnaeus, 1767	C60
Halocynthia papillosa Linnaeus, 1767	C50
Microcosmus sabatieri Roule, 1885	C50
Non identified Ascidiacea	All sites and depths
Phallusia fumigata Grube, 1864	C60
Polycitor sp.	C50 and C60
Cnidaria cl. Anthozoa	
Alcyonium coralloides Pallas, 1766	B50, B60, B70, C50
Alicia mirabilis Johnson, 1861	C60
Astroides calycularis Pallas, 1766	B70
Corallium rubrum Linnaeus, 1758	B60, B70, C60, C70
Eunicella cavolini Koch, 1887	All sites and depths
Eunicella singularis Esper, 1791	B50, B70, C60
Hoplangia durotrix Gosse, 1860	C60, C70
Leptogorgia sarmentosa Esper, 1789	B50, B60, B70, C50, C60
Leptopsammia pruvoti Lacaze-Duthiers, 1897	B60, B70
Non identified Alcyonacea	B60, C50, C60
Non identified Hydrozoanthidae	All sites and depths
Non identified Scleractinia	B60, B70, C70
Non identified Zoanthidea	B60, B70
Non identified Gorgonidae	B50, C70
Paramuricea clavata Risso, 1826	All sites and depths
Parazoanthus axinellae Schmidt, 1862	B50, B60, B70, C50
Foraminifera cl. Polythalamea	
Miniacina miniacea Pallas, 1766	All depths at C
Heterokontophyta cl. Phaeophyceae	
Cystoseira mediterranea Sauvageau, 1912	B50
Porifera cl. Calcarea	
Clathrina coriacea Montagu, 1818	B60, B70, C60
Porifera cl. Demospongiae	
Agelas oroides Schmidt, 1864	All sites and depths
Aplysilla sulfurea Schulze, 1878	B60, B70, C60, C70
Aplysina aerophoba Nardo, 1833	C60
Aplysina cavernicola Vacelet, 1959	B50, B60, B70, C60, C70
Axinella damicornis Esper, 1794	All sites and depths
Axinella polypoides Schmidt, 1862	B50, B60, C50, C70
Axinella vaceleti Pansini, 1984	B60, C60
Axinella verrucosa Esper, 1794	C60
Chondrosia reniformis Nardo, 1847	B50, B70, C70
Cliona schmidti Ridley, 1881	C50, C60
Cliona viridis Schmidt, 1862	All sites and depths
Crambe tailliezi Vacelet & Boury-Esnault, 1982	B60, B70, C60, C70
Crella (Crella) mollior Topsent, 1925	All depths at C
Diplastrella bistellata Schmidt, 1862	C60
Dysidea avara Schmidt, 1862	B50, B60, B70, C60, C70
Haliclona sp.	B50, B60, B70, C60, C70
Hemimycale columella Bowerbank, 1874	C60
Hexadella racovitzai Topsent, 1896	B50, B70, C50, C60, C70
Ircinia variabilis Schmidt, 1862	B50, B70
Myxilla (Myxilla) incrustans Johnston, 1842	C70
Non identified Porifera	All sites and depths
Phorbas tenacior Topsent, 1925	All sites and depths
Spirastrella cunctatrix Schmidt, 1868	All sites and depths
Spongia (Spongia) agaricina Pallas, 1766	C60
Porifera cl. Homoscleromorpha	
Oscarella lobularis Schmidt, 1862	B50, B60, B70, C60, C70
Rhodophyta cl. Florideophyceae	
Amphiroa sp.	C60
Halymenia floresia (Clemente) C. Agardh, 1817	C50
Lithophyllum sp. and *Mesophyllum* sp.	All sites and depths
Lithothamnium sp.	C70
Peyssonnelia squamaria (S. Gmelin) Decaisne, 1842	B50, B60, C50, C60

contrary to VS with 100 points and RQ with 100 points that detected all species (Table 2).

3.1.3. Accuracy

ANOVA with repeated measures on divergence from real values ($N=188$ observations repeated three times) did not show any significant effect of the group of methods (all points–squares confounded, $F=2.219$, $P=0.041$), the observer ($F=0.554$, $P=0.767$) or the interaction observer × method ($F=0.949$, $P=0.497$). Divergences in percentage of the real values ranged from 8% (species *i* with RQ and 100 random points) to 100% (species *g* with RQ and 64 stratified random points). A comparison with the real values revealed that percent cover was generally underestimated.

3.1.4. Objectivity

Percent cover showed variance among observers ranging between 0.018 (species *e*, RQ with 25 random points) and 78.801 (species *h*, VS with 100 squares) with the weakest global mean variance (mean of the variance estimated for each species) noted for RQ with 100 (1.439) and 64 (1.780) random points (Table 2, Fig. 4B). The highest variance was mostly observed for VS (regardless of the number of squares used in the grid) and especially for species covering more than 10%. VS was thus the least objective method. The grid of 25 squares produced the weakest variance within the VS method but was nevertheless more variable than RQ with 64 or 100 random points (*t*-test on paired data, $P=0.036$ and $P=0.022$ respectively).

3.1.5. Testing the best combinations

Finally, RQ with 64 totally random points was faster than with 100 points while maintaining the same objectivity. VS using 25 squares produced the least variable results with the shortest time compared to other VS combinations. Ranking the methods depending on the results obtained for each measured variable, RQ with 100 random points was the best method followed by RQ with 64 random points. Considering the time required, almost 7 min per quadrat was judged too long for RQ with 100 random points when RQ with 64 random points took around 3 min per quadrat. Following these results, the best trade-off was presented by RQ with totally random 64 points (called RQR64) and VS with the aid of 25 squares (VS25). These were thus tested by observer 1 with all versions of quadrats A and B.

VS25 was globally less accurate than RQR64 (4.187 > 2.911, *t*-test: $t=-2.649$, $P=0.009$). On the contrary, with 12 species missed on 234 occurrences, VS25 was more sensitive than RQR64 (26/234) (2×2 contingency tables, $\chi^2=5.61$, $P=0.178$). Real percent covers of missed species were 3.25% *maximum*. Concerning the repeatability, global variance of the results obtained from observer 1 was 9.787 ($SD=15.943$) with RQR64 and 12.920 ($SD=26.657$) with VS25 but the difference was not significant (ANOVA on $N=48$ observations repeated three times, $F=0.904$, $P=0.448$).

3.2. Field sampling and depth gradient detection

During the sampling, temperatures were 19 °C at the surface and 14 °C and 16 °C at −70 m respectively at Banc des vieilles and

Chrétienne. Both sites presented a thermocline at − 17 m and 10 m visibility. The current was South West at Banc des vieilles and East at Chrétienne. Following the results presented above, the RQR64 method (faster, better objectivity and accuracy) was performed with field data. The mean percentage of surface not covered by sessile organisms at both sites was 35%. A sludge percent was determined from the number of points (with 40 quadrats) projected on sludge; it was 28%, 24% and 42% at Banc des vieilles at − 50, − 60 and − 70 m and 34%, 34% and 30% at Chrétienne at − 50, − 60 and − 70 m. In total, 70 taxa were identified (Table 3). The most abundant species were the Porifera *Aplysina cavernicola* (Vacelet, 1959), the Rhodophyta *Mesophyllum* sp. and *Lithophyllum* sp., the Porifora *Crambe tailliezi* (Vacelet and Boury-Esnault, 1982),the Octocorallia *Paramuricea clavata* (Risso, 1826) and encrusting Bryozoa (Fig. 5). Weak differences were observed between results obtained from the different number of photographic quadrats (Fig. 5); CLUSTER with SIMPROF test showed that results obtained from 30 or 40 quadrats were always similar (Fig. 6) and sometimes not different from 20 quadrats (at Banc des vieilles − 60 m and Chrétienne − 50 m).

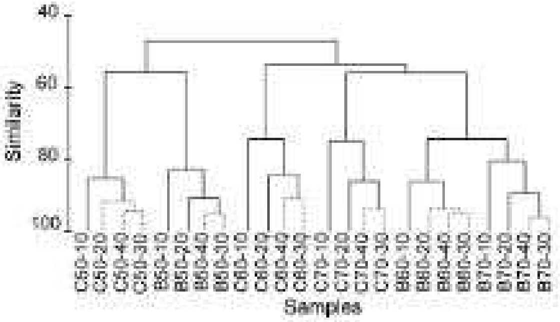

Fig. 6. Graphic representation of the CLUSTER results showing the samples grouped by similarity (Bray–Curtis Index). Continuous line linked dissimilar groups when dotted lines refer to non significant differences. The code used referred to site, depth and number of photos; C50-10 for example indicates a sample from Chrétienne at − 50 m on the base of 10 photographic quadrats.

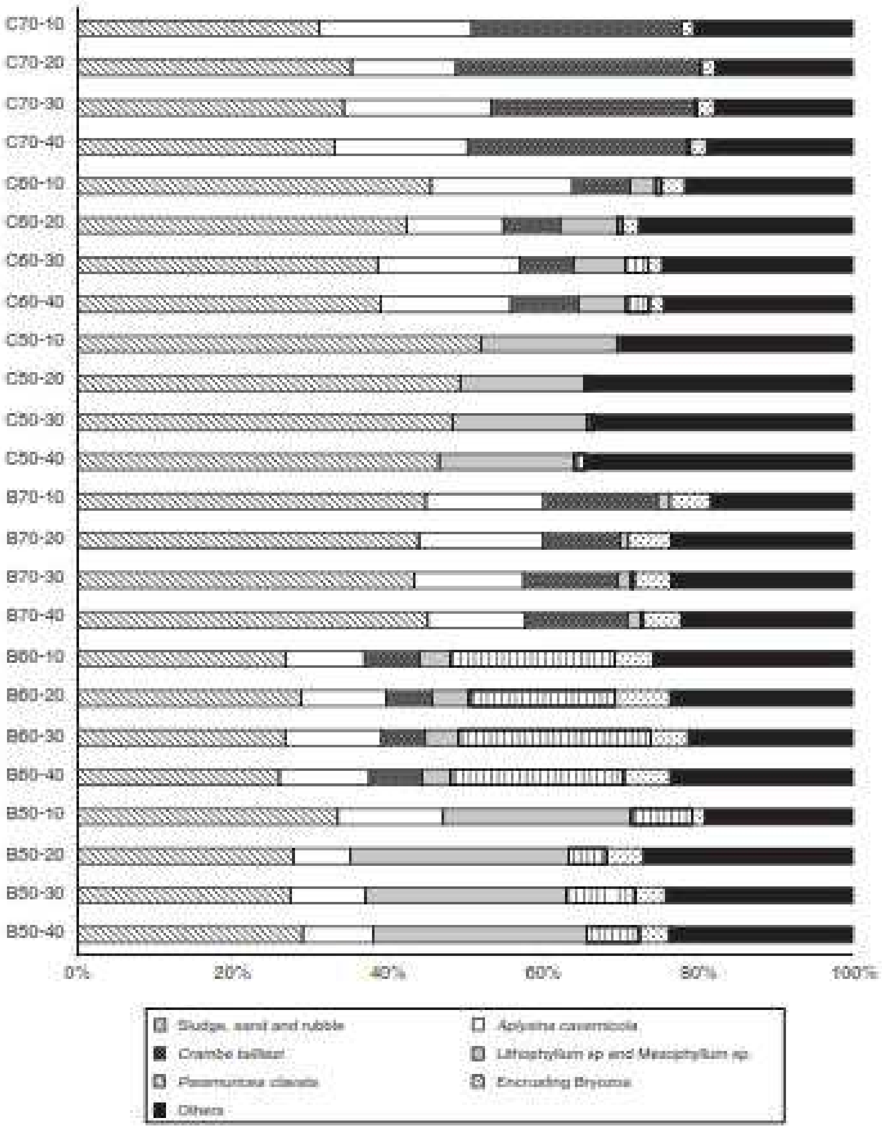

Fig. 5. Distribution of the most abundant species (in percent cover) observed at Banc des vieilles and Chrétienne at − 50, − 60 and − 70 m on the base of 40, 30, 20 and 10 photographic quadrats. The code used referred to site, depth and number of photos; C50-10 for example indicates a sample from Chrétienne at − 50 m on the base of 10 photographic quadrats.

Fig. 6 illustrates also how the deepest samples (−60 m and −70 m) were more similar than the shallowest ones. Dissimilarity between −50 m and −60 m/−70 m was mainly due to *C. tailliezi* abundance (SIMPER, contributed to 6.28% of the dissimilarity). Similarity within samples from −50 m was mainly produced by *Lithophyllum* sp. and *Mesophyllum* sp. (SIMPER, contribution of 10.81% to the similarity), from −60 m by *A. cavernicola* (contribution of 8.05%) and from −70 m by *C. tailliezi* (contribution of 9.63%). ANOSIM confirmed these observations: assemblages did not significantly differ between sites (ANOSIM, $P = 0.30$) whereas a highly significant difference was detected between depths within sites (ANOSIM, $P = 0.001$).

4. Discussion

4.1. *Comparison between results issued from simulated quadrats: the choice of random-point quadrat method with 64 points*

Whatever the method, no difference in sensitivity or accuracy was found. As hypothesized, VS was a far less objective method compared to RQ. In the field, Dethier et al. (1993) reported VS as the most difficult method to perform since "the sampler needs to concentrate on the distributions of various species rather than simply identifying and recording the species under each point". Although they did not test the influence of different grid sizes, they assumed "mental integration" would be easier with a larger relative number of small subdivisions. According to these assumptions, VS using 100 squares tended to be more accurate than VS25. Nevertheless, VS25 was the fastest and most objective VS combination. Whereas most studies only compared random vs. visual sampling, we also tested the use of stratified random points. Expected to avoid potential grouping of points, this combination was strangely the least accurate than the others probably because of a non relevant subdivision in regard to species distributions (non regular). Because of an absence of regularity in distribution patterns, Dethier et al. (1993) preferred to use random rather than systematic points.

Comparing all combinations within RQ, our results showed that a random-point photographic quadrat with 64 random points was among the fastest and most objective method tested and was equally accurate, sensitive and repeatable compared to the others. Even with a unique observer, results tended to be more repeatable with RQR64 although the difference was not significant. Only 3 min were necessary for one analysis with a simulated quadrat. Moreover, with RQR64 all species with percent covers superior to 4% were detected (missed species presented maximal percent covers of 3.17% with quadrat A and 3.25% with quadrat B). According to our criteria described in the Introduction, RQR64 could be a good method for the study of coralligenous assemblages.

4.2. *Random-point photographic quadrat with 64 random points applied at two field sites detects a depth gradient in species assemblages*

RQR64 was easily applied to field work with 70 taxa being identifiable. Our results highlighted another important point for further sampling campaigns: 30 ($=7.5\ \text{m}^2$) or 40 ($=10\ \text{m}^2$) photos per transect produced similar species assemblage results. Thirty photos instead of 40 represented a minimum of 32 min of saved time (considering the time counted with simulated quadrats) per transect with the computer (47 min counting preparation and analysis) and 7.5 min in mean underwater. For a 30-minute dive at −60 m, 7 min was saved underwater which reduces the time spent underwater by 25 min (including decompression stages according to MN90 tables with AIR dives) and thus considerably decreases accident risks for SCUBA divers. From these photos, species assemblages were analyzed and even with our small data set, we highlighted a clear depth pattern in coralligenous assemblages. In the Mediterranean Sea, vertical distributions of subtidal assemblages figures among the most widely documented pattern; related to light, water movement, sedimentation, temperature and nutrients, depth is the major factor influencing the distribution of marine organisms (Balata and Piazzi, 2008; Balata et al., 2005; Garrabou et al., 2002; Pérès and Picard, 1964; Piazzi and Balata, 2011; Virgilio et al., 2006). After algae, Porifera generally figured among the most abundant organisms above −50 m within coralligenous assemblages (Ballesteros, 2006; Ballesteros et al., 2009; Ferdeghini et al., 2000). When *Lithophyllum* sp. and *Mesophyllum* sp. were the most dominant species at −50 m, our study showed the deepest assemblages to be dominated by Porifera (*A. cavernicola* at −60 m and *C. tailliezi* at −70 m). As the depth increased, relative cover changed (more than the number of species) and coralline algae became less and less abundant, benefiting Porifera. *A. cavernicola* was already present at −50 m at Banc des vieilles while absent above −60 m in Chrétienne. On the contrary, *C. tailliezi* appeared at −60 m at both sites. Comparison with previous studies is difficult as deep-sea coralligenous studies are scarce. ROV observations (down to −160 m) off the Spanish Mediterranean coast showed that large coralligenous concretions were more common between 80 and 120 m depth than in deeper depths (Aguilar et al., 2009). No quantitative data were available but they observed *Lithophyllum* sp. at all four sites, *A. cavernicola* at all except one, *P. clavata* and *E. cavolinii* at one site and *C. tailliezi* was never noted.

5. Conclusion

Random-point photographic quadrat sampling, with 64 random points (RQR64) was the method that cumulated the highest number of advantages. It has a good trade-off between time required, sensitivity, objectivity and accuracy, the methodology proposed here guarantees fast abundance estimation, non-destructive repeated sampling, the possibility of comparison among researchers and the permanent record of deep-sea communities. Easy to carry out and user-friendly, anybody could apply this sufficiently powerful methodology to detect multivariate patterns like depth gradient. Moreover, a new version CPCe 4.1 "coralligenous assemblage version" adapted to species identifiable in Mediterranean coralligenous assemblages is now freely available. Photos can be taken equally by ROV or divers. However, it should be avoided for total species inventory or when species richness is the variable of interest because species covering less than 4% risked to be missed. In these conditions, only total scraping with hammer and chisel allows the collection of all organisms (see for example Balata and Piazzi, 2008). This method allows the conservation of samples for further analyses (if necessary) but is highly destructive thus its application is limited and should be restricted to specialists. Visual estimations directly performed underwater benefited from recent improvements reducing the time needed for estimations to be conducted in the field (Parravicini et al., 2010) and may be preferred when one person, the specialist, is sure to evaluate all quadrates.

Acknowledgments

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Annexe 2 : Une étude préliminaire vers un indice basé sur les assemblages coralligènes pour l’évaluation de l'état écologique des eaux côtières méditerranéennes françaises

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Julie Deter[a,b,*], Pierre Descamp[c], Laurent Ballesta[c], Pierre Boissery[d], Florian Holon[c]

[a] L'OEil d'Andromède, 7 place Cassan, 34280 Carnon, France
[b] Institut des Sciences de l'Evolution (ISEM) – UMR 5554, Université de Montpellier 2, 34095 Montpellier cedex 5, France
[c] Andromède Océanologie, 7 place Cassan, 34280 Carnon, France
[d] Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France

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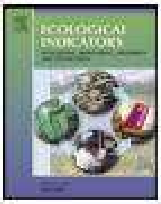

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Julie Deter[a,b,*], Pierre Descamp[c], Laurent Ballesta[c], Pierre Boissery[d], Florian Holon[c]

[a] L'OEil d'Andromède, 7 place Cassan, 34280 Carnon, France
[b] Institut des Sciences de l'Evolution (ISEM) – UMR 5554, Université de Montpellier 2, 34095 Montpellier cedex 5, France
[c] Andromède Océanologie, 7 place Cassan, 34280 Carnon, France
[d] Agence de l'Eau Rhône-Méditerranée-Corse, Délégation de Marseille, Immeuble le Noailles, 62 La Canebière, 13001 Marseille, France

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ABSTRACT

Despite the great contribution of coralligenous communities to Mediterranean biodiversity (second key-ecosystem after *Posidonia oceanica* meadows), they were never considered in the establishment of multimetric indices for ecological status assessment of marine environment. In this paper, we describe a method to evaluate the ecological status of coralligenous assemblages along Mediterranean French coasts. Several metrics were selected from literature for coralligenous assemblage description and include functional and structural information: percent cover of visible non-vagile species (using photographic quadrats along a transect) and gorgonian demography. Thirty eight field stations were sampled for these metrics in PACA (Provence-Alpes-Côte-d'Azur) region in June 2010 and considered for their morphology (bank, rim), geographical orientation and principal current direction (North, East, West, South) and depth (from −30 to −84 m). Metrics found to be linked to human pressures using ANCOVA and multiple correlation matrix were selected to be included in the index. The index (Coralligenous Assemblage Index, CAI) that we proposed was based on three selected metrics (Bryozoa percent cover, sludge percent cover, builder species percent cover) and considers depth; it was positively and significantly linked to anthropization (related to water quality). The 38 stations studied with theoretically good to bad environmental conditions were classified in levels of status in accordance with our field work knowledge. CAI variation was validated with three stations sampled with 30 other photos. This index could be an effective tool for the assessment of the ecological quality of coralligenous communities. It could be applied in the context of the Marine Strategy Framework Directive as well as in conservation and sustainable management of the marine environment.

1. Introduction

Coralligenous concretions are primarily produced by the accumulation of encrusting algae growing at low light levels and secondarily by bio-constructor animals as polychaetes, bryozoans and gorgonians; they represent the unique calcareous formations of biogenic origin in Mediterranean Sea (Ballesteros, 2006). The resulting complex structure allows the development of a patchwork of communities dominated by living algae, suspension feeders, borers or soft-bottom fauna (in the sediment within cavities). Two main morphologies can be distinguished (Ballesteros, 2006) for coralligenous frameworks: banks (built over more or less horizontal substrata) and rims (in the outer part of marine caves and on vertical cliffs). In terms of richness, biomass and production, coralligenous assemblage value is high and comparable to tropical reef assemblages (Bianchi, 2001). The other side of the coin is the important attract exerted over divers and fishermen and their damaging consequences (Ballesteros, 2006). Moreover, submarine outfalls (with their urban and industrial discharges) are widespread along the coast and rocky coasts are among the most vulnerable habitats to pollution, increased sediment loads and deposition (Airoldi, 2003; Ballesteros, 2006). Increasing anthropogenic pressures and their consequences on water quality decline have led the European Union to engage a new strategy to conserve and recover the ecological quality of the marine environment. With the Water Framework Directive (WFD, Directive 2008/56/EC), European commission aims to achieve (or maintain at least) a "good status" in all the European waters by 2015. WFD defines the ecological status as the quality of the structure and functioning of ecosystems associated with homogenous water bodies. The

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[*] Corresponding author at: L'œil d'andromède, 7, place Cassan, 34280 Carnon, France. Tel.: +33 4 67 66 32 48.

E-mail addresses: julie.deter@andromede-ocean.com (J. Deter), pierre.descamp@andromede-ocean.com (P. Descamp), laurent.ballesta@andromede-ocean.com (L. Ballesta), pierre.boissery@eaurmc.fr (P. Boissery), florian.holon@andromede-ocean.com (F. Holon).

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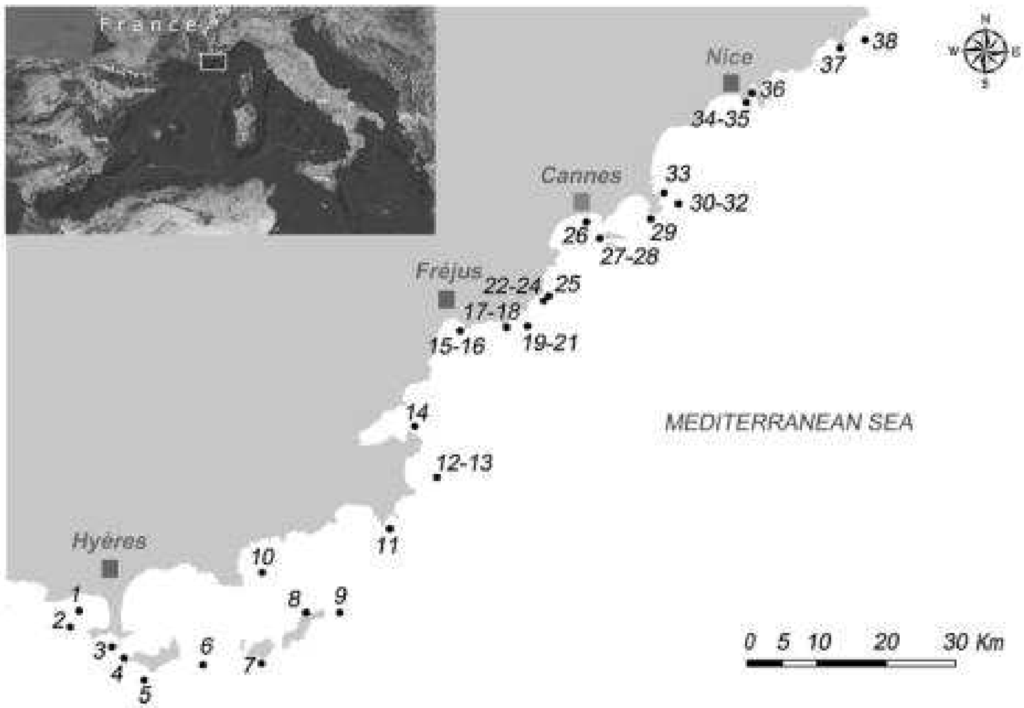

Fig. 1. Map presenting the 38 stations studied on the Mediterranean French coast (PACA region).

status of each water body is evaluated thanks to variables relative to organisms or groups of organisms sensitive to anthropogenic pressures and called biological quality elements (phytoplankton, macroalgae, angiosperms and benthic fauna) (Devlin et al., 2007). Despite the great contribution of coralligenous communities to Mediterranean biodiversity (Ballesteros, 2006) and its recognition as a natural habitat of communitarian interest, whose conservation requires the designation of Zones of Special Conservation at European level (92/43/CE Habitat Directive, habitat code 1170-14: Reefs, coralligenous assemblage), coralligenous assemblages were largely neglected for the water body quality assessment. Actually, most efforts in ecological status assessment of marine waters have been carried out in the implementation of angiosperms and soft bottom biotic indices (Borja et al., 2000; de-la-Ossa-Carretero et al., 2009; Gobert et al., 2009; Fitch and Crowe, 2010; Lopez y Royo et al., 2011). This relative lack of interest could be explained by the depth of this habitat (from 12–50 m to 40–120 m depth depending on water transparency (Ballesteros, 2006)) and a missing consensual methodology for its monitoring (UNEP-MAP-RAC/SPA, 2011). Even despite the lack of overall community analyses, several species (believed to be the most sensitive) found in coralligenous assemblages were nevertheless studied for their responses to common anthropogenic disturbances and highlighted for their potential quality indicator role. For example, gorgonians (very slow-growing threatened species), because of their particular sensitivity when faced with increasing disturbances, were proposed as potential indicators of the effects of climatic anomalies on the coralligenous community (Linares et al., 2008). Similarly, red coral was particularly well surveyed because it underwent specific harvesting as well as other anthopogenic pressures (Tsounis et al., 2006; Bruckner, 2010). Other large and/or erected species such as bryozoans (e.g. *Pentapora fascialis*) or ascidians (e.g. *Halocynthia papillosa*) are also influenced by diving frequency and waste water impacts (Sala et al., 1996; Pérez et al., 2002; Luna-Pérez et al., 2010).

Based on field data sampled around coralligenous assemblages, the aim of this study was to present a multimetric methodology for the environmental evaluation of water quality, in agreement with the principles of the WFD.

2. Materials and methods

2.1. Field work

In June 2010, 38 stations presenting coralligenous concretions were sampled (depth between −30 and −84 m) in 13 water bodies on the Eastern part of French Mediterranean coast (Fig. 1). Details concerning these stations are described in Table 1: they were chosen in order to represent different anthropogenic pressure conditions. Coralligenous assemblages, especially sessile species and species believed to be vulnerable, were described at each site. Two protocols were applied by CCUBA (Closed Circuit Underwater Breathing Apparatus) divers for the metrics measurements at each station.

Coralligenous assemblages (sessile organisms) description with photographic quadrats. Each station was sampled using 30 photographic quadrats (50 × 50 cm) along a 40 m-long transect. Pictures were taken using a digital camera (D2Xs Nikon at 12.4 megapixels with a 12–24 mm zoom-lens Nikon and used with a housing, a dome and SEACAM® flashes specially adapted to deep dives) perpendicularly fixed 50 cm over the quadrat frame, thus minimizing possible parallax errors. Pictures were analyzed using CPCe 3.6 (Kohler and Gill, 2006) for the estimation of the percentage of the total area covered by each species. This non-destructive method samples 64 random points per quadrat frame and is judged to be fast and efficient for coralligenous community analyse (Holon et al., 2010). Structure parameters like sludge and crevice percent covers were estimated by the same way. Taxa (sessile organisms) were identified to the level of species or genus. Where identification at the most detailed level of taxonomical resolution was not possible, animals were grouped in phyla. Hydrozoa and encrusting Bryozoa were not identified further and were classified as “Hydrozoa” and “Encrusting Bryozoa”. Unidentifiable organisms were classified as “unknown” and were not considered in community analyses.

Erected species demography (especially gorgonians). Gorgonian demographic structure was classically obtained from density and individual height measured underwater in 2 m^2 (eight 50 × 50 cm quadrats) (Sartoretto, 2003). Thirty 50 × 50 cm quadrats were used

Table 1
Description of the 38 stations sampled in June 2010.

Code	Name	Depth (in m)	Morphology	Principal current direction	Orientation (slope direction)
1	Sabran	35	Bank	N	SW
2	Off Fourmigues	35	Rim	NW	SW
3	South Ribaud	47	Bank	NW	SW
4	Langoustier edge	33	Rim	NW	SW
5	Cape Arme	50	Bank	W	S
6	Shallow Sarranier	41	Rim	W	SE
7	Gabinière	41	Rim	W	S
8	Castelas edge	43	Bank	SW	N
9	Levant Beacon	38	Bank	SW	SE
10	Baleine	36	Rim	S	SW
11	Ancres	39	Bank	SW	S
12	Pampelone-62	62	Bank	S	NE
13	Pampelone-70	70	Bank	S	NE
14	Rabiou beacon	52	Bank	S	E
15	Lion de mer-30	30	Rim	W	S
16	Lion de mer-39	39	Rim	W	S
17	Dramont-40	40	Rim	W	SW
18	Dramont-30	30	Rim	W	SW
19	banc de vieilles-50	50	Rim	W	SW
20	banc de vieilles-60	60	Rim	W	SW
21	banc de vieilles-70	70	Rim	W	SW
22	Chrétienne-50	50	Rim	SW	S
23	Chrétienne-60	60	Rim	SW	S
24	Chrétienne-70	70	Rim	SW	S
25	Off Cap roux	37	Bank	S	N
26	Midi	33	Rim	NW	SW
27	Dragon edge-63	63	Rim	N	W
28	Dragon edge-70	70	Rim	N	W
29	Shallow st pierre	40	Rim	S	SW
30	Raventurier-44	44	Bank	S	E
31	Raventurier-54	54	Bank	S	E
32	Raventurier-65	65	Rim	S	E
33	Bacon edge	38	Rim	S	E
34	American rim-63	63	Rim	W	S
35	American rim-84	84	Rim	W	S
36	Lido	36	Bank	N	S
37	Eastern Martin cape	55	Bank	SW	SE
38	Western Martin cape	48	Bank	W	S

N = Nord, S = South, E = East, W = West

for necrosis study: necrosis percent (Perez et al., 2000), distribution of necrosis (diffused or localized) and dating of necrosis colonization (with old, recent or mix colonization). Such metrics are routinely measured for gorgonian monitoring (Harmelin and Marinopoulos, 1994; Pérez et al., 2002; Sartoretto, 2003). Quadrats used for erected species demography were randomly chosen and were different from photographic quadrats. For time constraints, only one station was sampled when stations were located close-at-hand (Table 1). Consequently, gorgonian species demography was studied at 24 stations (codes 1, 3, 6-10, 11, 14, 17, 21, 23-26, 29-31, 33-35, 37, 38).

These protocols figured among the proposed standard methods for inventorying and monitoring coralligenous populations (UNEP-MAP-RAC/SPA, 2011) and moreover fitted with the methodological guide for the evaluation of the conservation state of Natura 2000 marine natural habitats (Lepareur, 2011).

2.2. Anthropogenic pressure estimation

We calculated an anthropogenic pressure index (API) for each field station considering three body water state descriptors (coastal artificialization percent, ecological and chemical state considering respective confidence index as communicated by the French Water Agency for each water body http://www.rhone-mediterranee.eaufrance.fr/gestion/dce/telechargements-sdage.php) and local descriptors. These last ones were principally based on an anthropization index defined by Gobert et al. (2009). It included a score (from 0 (no impact) to 5 (dramatic pressure)) for eight pressures: fish farming, population development, industrial development, agriculture, tourism, fishing, commercial ports and urbanization. Considering the possible depth and distance from the coast, an accessibility factor was added on the same way. API was defined as the sum of these twelve factors affecting the seawater quality and/or biotope quality. Scores for local descriptors were estimated from freely available information: trade association, INSEE (National Institute for Statistics and Economical Studies, http://www.insee.fr), French Ministry of Ecology, Industry, Sustainable Development and Sea (http://carmen.developpement-durable.gouv.fr/25/environnement.map), DREAL (Regional Direction for Environment, development and housing, http://www.paca.developpement-durable.gouv.fr), MEDAM (Coastal development on the French Mediterranean coast, http://sigcol.unice.fr/website/MEDAM/site_medam/index.php) and aerian pictures from Google Earth (http://www.google.fr/intl/fr/earth/index.html).

2.3. Selecting and testing candidate metrics

According to WFD (2000/60/EC), monitoring should concern quality elements which are indicative of the pressures to which each water body is subject. Drivers-pressures-state-impacts-responses (DPSIR) approach provides a good communication tool between researchers, stakeholders and decision makers, but also a global mechanism for assessment and management of environmental problems with regards to sustainable development. Different metrics measuring impacts may provide pertinent

Table 2
Drivers–pressures–state–impacts analysis.

Drivers	Pressures	State	Impacts	References
Urbanization, population	Runoff waters; sewage discharges; diving pressure	Toxic contamination; degradation of water/sediment quality; habitat loss; water temperature; eutrophication; habitat destruction	Diversity; dominance; abundance; age structure; massive death; selective mortality	Sala et al. (1996), Garrabou et al. (1998), Pérez et al. (2002), Fraschetti et al. (2006), Ballesteros (2006), Luna et al. (2009) and Luna-Pérez et al. (2010)
Agriculture and fish farming	Pesticides; effluent discharges; pathogens; individuals leak	Eutrophication; degradation of water/sediment quality; toxic contamination; habitat loss; biological pollution	Diversity; dominance; abundance; age structure; massive death; reproductive dysfunction; genetic variability	Hong (1983) and Claudet and Fraschettia (2010)
Industrial development	Industrial effluents discharges	Toxic contamination; degradation of water/sediment quality; habitat loss; eutrophication; water temperature	Diversity; dominance; biomass; abundance; age structure; massive death; reproductive dysfunction	Hong (1983); Terlizzi et al. (2002) and Aguilar et al. (2009)
Fishing	Destructive methods; selective catch, overfishing	Selective mortality; habitat destruction	Diversity; dominance; abundance; age structure	Tsounis et al. (2007), Ballesteros (2006) and Aguilar et al. (2009)
Port activity	Ballast waters; boats pressure	Toxic contamination; Degradation of water/sediment quality; biological pollution; noise perturbance	Diversity; dominance; abundance; age structure; massive death; exotic species; pathogens; genetic variability; reproductive dysfunction;	United Nations Environment Programme (2007) and Baldacconi and Corriero (2009)
Dredging activity	Contaminants and sediments suspension; sediments removing	Toxic contamination; degradation of water/sediment quality; biological pollution; habitat destruction; increased turbidity	Diversity, dominance; abundance; massive death; exotic species; genetic variability	Balata et al. (2005), United Nations Environment Programme (2007), UNEP-MAP-RAC/SPA (2008)

Drivers: main socio-economic driving human activities; pressures: consequences of the human activities in the environment; state: environment changes due to pressures force; Impacts: potential alterations in the coralligenous assemblages due to the increasing of the state.

information on coralligenous habitat vitality (at the individual and population level) for a wide spectrum of disturbance (water turbidity, nutrient concentrations, sedimentary dynamics, erosion) regularly described in the Mediterranean Sea (Table 2). In the development of the index described here, metrics we are interested in fitted with this DPSIR analysis and provided pertinent information on coralligenous habitat vitality. The candidate metrics obtained from the field work are based upon either presence/absence of potentially indicator species (protected species and/or vulnerable species) or big and easily detectible species, 'relative' abundance (number, density or percent cover) or number of present taxa (Table 3). These metrics were easy to acquire, present a good cost-efficiency ratio and require a low technological investment (no laboratory work). Once normalized, each metric (dependent variable) was tested for a link with API, depth (independent variables) and morphology (co-variable) using an ANCOVA. Depth (correlated to light irradiance) and morphology were included in the models because of their potential important influence on coralligenous assemblages (Ballesteros, 2006). Metrics significantly ($P < 0.05$) linked to API and that were not redundant (Pearson's $r < 0.8$ with a multiple correlation matrix) were selected for the index elaboration. Analyses were performed using Statistica 6.1 (Statsoft, Inc.).

2.4. *Combining metrics and building a Coralligenous Assemblage Index (CAI)*

The choice of the metrics was based on their ecological meaning, response to "state" of environmental change (interpretable and significant link with API) and the redundancy relation between them (excluding the redundant metrics). WFD states that the classification of ecological status shall be based on ecological quality ratios (EQR = observed values/reference values), i.e. the deviation of the status of the quality element from its potential status under pristine conditions (undisturbed, reference conditions). Given that no pristine conditions could be determined in the studied area, in a first approach, we postulated the reference condition as a "theoric optimal site", corresponding to the best values of each metric noted in the field. According to WFD, each EQR was expressed as a numerical value between zero and one, with high ecological status represented by values close to one and bad ecological status by values close to zero. The chosen metrics were added and averaged. The resulting value provided an overall coralligenous habitat classification index called Coralligenous Assemblage Index (CAI). Assignment of quality status for each station was established according to the five classes scale proposed by WFD and ranging from high (<0.75–1), good (>0.60–0.75), moderate (>0.40–0.60), poor (>0.25–0.40) to bad (0–0.25). Although the methodology used did not guarantee the reality of our reference limits, ecological results obtained for this study were comparable because they were obtained considering by a similar way. Finally, the robustness of our index obtained for each station was tested with a comparison to API using a linear regression. Stations classified in "high ecological status" were sampled a second time in June 2011. Selected metrics, CAI and classification were compared (variation relative to the values measured in the first survey and *t*-tests).

3. Results

3.1. *Candidate metrics measurement and selection*

All the 38 stations were analyzed for photographic quadrats. Unidentified species covered 4.84% (±3.30) of the quadrats. Crevice, sludge and rubble mean percents were 15.68% (±0.81), 39.94% (±2.84) and 2.35% (±0.47) [mean (±standard deviation)]. Mean percent covers were highly variable depending on groups considered (Table 3). Among these candidate variables, three were significantly linked to API and selected to be included in the CAI: sludge percent ($P < 0.001$), percent cover of builders ($P = 0.034$) and percent cover of bryozoans ($P = 0.016$) (results presented in Table 4). Sludge percent (Correlation coefficient $r = 0.471$) increased with API when percent cover of bryozoans ($r = -0.375$) and builders ($r = -0.283$) decreased with API (Fig. 2). Sludge percent ($r = 0.353$) and the percent cover of builders ($r = -0.600$) were also influenced by depth (Table 4).

Table 3
List of the candidate metrics tested to be included in the coralligenous index with mean values and standard errors calculated from the field stations sampled. Densities are expressed in number of individuals per m^2. Gorgonian colonies were counted as big when measuring more than 50 cm and small when measuring less than 15 cm. According to Ballesteros (2006), builder species were composed on Coralline species, bryozoans, scleractinians, *Miniacina miniacea* and *Leptopsammia pruvoti* and eroder species were composed on *Cliona* spp, *Echinus melo*, *Sphaerechinus granularis*, *Polydora* spp and *Lithophaga lithophaga*.

Fiel method	Metrics (mean value [standard error])
Photographic quadrats (38 stations)	Crevice percent cover (15.08% [±0.81])
	Sludge percent cover (39.94% [±2.84])
	Rubble percent cover (2.35% [±0.47])
	Percent cover of macroalgae (14.75% [±2.58])
	Percent cover of coralline species (9.11% [±1.24])
	Percent cover of ascidians (0.81% [±0.22])
	Percent cover of scleractinians (0.12% [±0.05])
	Percent cover of hydrozoans (2.15% [±0.38])
	Percent cover of alcyonarians (0.21% [±0.09])
	Percent cover of gorgonians (0.33% [±0.92])
	Percent cover of porifora (13.87% [±2.11])
	Percent cover of bryozoans (3.23% [±0.30])
	Percent cover of bryozoans with a height > 15 cm (2.08%[±0.50])
	Percent cover of builders (12.70% [±1.31])
	Percent cover of builders + *Peyssonnelia* sp (24.82% [±2.42])
	Percent cover of eroders (0.57% [±0.15])
	Percent cover of erected species (bryozoan + gorgonian + *Axinella polypoides*) (9.64% [0.85])
	Shannon weaver diversity index of gorgonians (0.53 [±0.36])
	Percent cover of *Corallium rubrum* Linnaeus, 1758 (0.84% [±0.57])
	Percent cover of *Paramuricea clavata* Risso, 1826 (3.41% [±0.56])
	Percent cover of *Eunicella cavolini* koch, 1887 (2.02% [±0.34])
	Percent cover of *Leptogorgia sarmentosa* Esper, 1789 (0.08% [0.27])
	Percent cover of *Halocynthia papillosa* Linnaeus, 1767 (0.08%[0.11])
	Percent cover of *Aplysina cavernicola* Vacelet, 1959 (2.29% [±0.63])
	Percent cover of *Axinella damicornis* Esper, 1794 (0.37% [±0.12])
	Percent cover of *Phorbas tenacior* Topsent, 1925 (0.67% [±0.19])
	Percent cover of *Adeonella calveti* Canu and Bassler, 1930 (0.37% [0.71])
	Percent cover of *Myriapora truncata* Pallas, 1766 (0.26% [0.27])
	Percent cover of *Parazoanthus axinellae* Schmidt, 1862 (0.81% [1.87])
	Percent cover of *Caulerpa racemosa* (Forsskål) Agardh, 1873 (0.25% [0.75]) Percent of *Filograna* sp or *Salmacina* sp (0.11% [0.14])
Gorgonian species demography (18 stations)	Density of *P. clavata* colonies (8.04 [±1.38])
	Number (or density) of *E. cavolini* colonies (4.90 [±0.85])
	Presence/absence of *L. sarmentosa* colonies
	Percent of gorgonians with necrosis > 10% (11.60% [±3.07])
	Percent of *P. clavata* with recent (0.86% [1.64])
	Percent of *E. cavolini* with recent or old necrosis (0% [0])
	Maximal class of necrosis for *P. clavata* or *E. cavolini*
	Density of big or small *P. clavata* individuals (0.50 [0.31] or 0.31 [0.35])
	Density of big or small *E. cavolinii* individuals (0.02 [0.05] or 0.39 [0.09]))
	Presence/absence of recruitment (colonies < 10 cm) in *P. clavata* (or *E. cavolini*)
	Maximal height of *P. clavata* or *E. cavolini* (115-120 cm or 65–70 cm)

These relationships were clearest for data samples at depth inferior to −45 m than superior to −45 m (graphs not showed). None of these three metrics was significantly influenced by morphology.

Among the 24 stations studied for gorgonian demography, six were not included in these data because of an insufficient number (<4 colonies) of gorgonians (stations 1, 3, 10, 24, 34, 35). Considering the 18 stations, mean density was 13.34 (±1.67) gorgonians per m^2 mostly due to red and yellow gorgonian (respective mean densities 8.04 (±1.38) and 4.90 (±0.85). Results are detailed in Table 3. The highest red and yellow gorgonian observed measured 115–120 (station 37) and 65–70 cm (station 38). No variable associated to gorgonian species demography was found to be significantly linked to API.

3.2. Coralligenous Assemblage Index (CAI) elaboration

CAI was based on the three metrics selected from statistical analyses (see above). This included sludge percent, percent cover of builders and percent cover of bryozoans. Reference values (best values = minimal or maximal values depending the sense of the correlation with API) considered the depth (more or less than −45 m) for all metrics except the last one (Table 5). All of the five levels of

Table 4
Results of ANCOVA testing for a link between anthropogenic pressure index (API), depth, morphology and candidate metrics ($N = 38$ stations). Only candidate metrics presenting a significant ($P < 0.05$) link with API are presented here.

Dependent variable (*Y*)	Explanatory factors (*X*)	Degree of freedom	Wald statistics	*P*
Sludge percent cover	API	1	13.836	<0.001
	Depth	1	8.800	<0.001
	Morphology	1	5.561	0.062
Percent cover of builders	API	1	4.317	0.038
	Depth	1	13.993	<0.001
	Morphology	1	0.203	0.652
Percent cover of bryozoans	API	1	5.492	0.019
	Depth	1	0.197	0.657
	Morphology	1	3.392	0.065

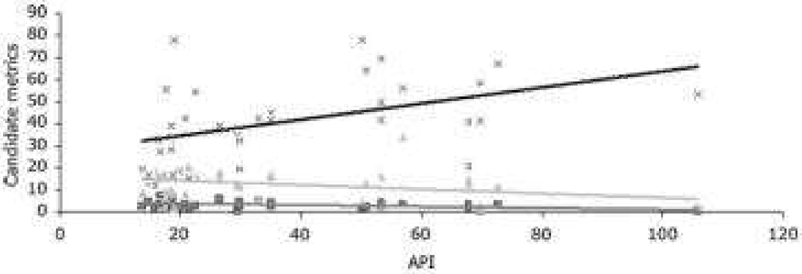

Fig. 2. Evolution of three selected metrics: sludge percent (cross and black line), percent cover of bryozoans (square and dark grey line) and percent cover of builders (triangle and light grey line) in function of API (anthropogenic pressure index) on the 38 stations studied.

Table 5
Reference conditions for each metrics.

	All depths	Stations shallower than −45 m	Stations deeper than −45 m
Sludge percent	–	12.093	26.920
Percent cover of builders	–	29.039	32.326
Percent cover of bryozoans	7.423	–	–

status were represented by the 38 stations sampled. The ecological statuses obtained in the application of the CAI are presented in Table 6. Most of the stations were classified between "moderate" and "good" status. Four stations were classified in "bad ecological status" (stations 13, 27, 36, 38), four in "poor ecological status" (stations 3, 29, 34, 35), and three in "high ecological status" (stations 5, 6 and 19). The quality of seawater expressed by API was negatively and significantly correlated with CAI (r = −0.518, Fig. 3). The highest API (105.96) was given to station 36, classified in "bad ecological status" using CAI (Table 6). This station presented high population (including tourism) and boat pressures but also an easy access and moderate fishing activities. Coastal artificialization percent was largely the highest (61%) and chemical state was judged bad by the French Water Agency. The lowest APIs were allocated to stations 7, 8 and 9 (14.67; 13.67; 15.67), classified in "moderate" and "good ecological status" according to CAI. These stations were located in moderately urbanized areas, with weak tourism and fishing activities; coastal artificialization percent was around 3%. The three stations classified in "high ecological status waters" (5, 6 and 19) presented only a little higher API values than stations 7, 8 and 9 (16.67, 18.67 and 18.47) mostly because of an easier accessibility and higher fishing activities.

A second sampling at these three stations classified in "high ecological status" did not show any significant inter-annual variation for selected metrics (*t*-test with paired samples, all $P > 0.05$) and all of the three stations conserved their classification as "high

Table 6
Coralligenous Assemblage Index (CAI) and anthropogenic pressure index (API) values calculated for the 38 stations studied on the French Mediterranean coast (PACA region).

Code	API	CAI	Ecological status level
1	29.82	0.501	Moderate
2	29.82	0.686	Good
3	17.67	0.340	Poor
4	19.67	0.565	Good
5	16.67	0.831	High
6	18.67	0.788	High
7	14.67	0.646	Good
8	13.67	0.525	Moderate
9	15.67	0.583	Good
10	29.47	0.406	Moderate
11	21.47	0.609	Good
12	20.91	0.509	Moderate
13	18.91	0.165	Bad
14	22.41	0.507	Moderate
15	67.88	0.574	Good
16	67.88	0.442	Moderate
17	26.47	0.632	Good
18	26.47	0.657	Good
19	18.47	0.919	High
20	18.47	0.714	Good
21	18.47	0.681	Good
22	17.47	0.528	Moderate
23	17.47	0.504	Moderate
24	17.47	0.452	Moderate
25	16.47	0.690	Good
26	56.9	0.653	Good
27	53.40	0.230	Bad
28	53.40	0.355	Poor
29	53.40	0.568	Good
30	35	0.511	Moderate
31	35	0.620	Good
32	33	0.554	Moderate
33	72.72	0.403	Moderate
34	69.79	0.282	Poor
35	69.79	0.306	Poor
36	105.96	0.238	Bad
37	50.29	0.211	Moderate
38	50.79	0.410	Bad

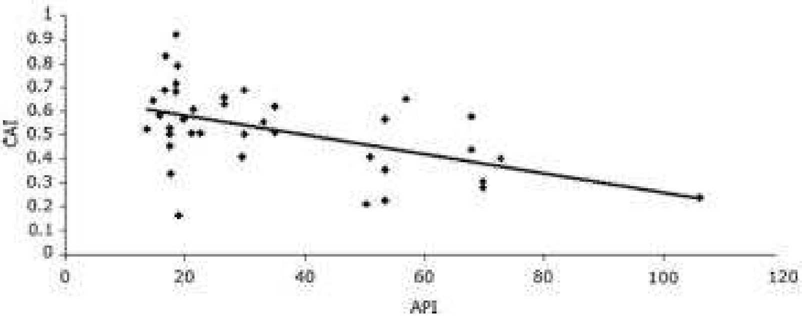

Fig. 3. Regression between the anthropogenic pressure index (API) and the classification obtained by the Coralligenous Assemblage Index (CAI). $N = 38$ sites, $R^2 = 0.268$, $P < 0.001$.

ecological status" (CAI = 0.829; 0.898; 0.790 for respectively stations 5, 6 and 19).

4. Discussion

Our study did not succeed in highlighting adequate metrics linked with anthropogenic pressures concerning gorgonian or target species variables. Because of their long life expectancy and slow dynamics, gorgonian populations are particularly vulnerable and show long-term consequences of disturbances (Garrabou and Harmelin, 2002; Linares et al., 2007). Climatic anomalies like 1999 mass mortality event were for example still visible on *P. clavata* populations (biomass and density) studied in 2003 (Linares et al., 2008). Because of this weak resilience, it is thus difficult, without population historic and long-term data, to link present observation with present anthropogenic pressures especially potentially recent enhancements; in other terms, metrics acquired during our study might be more explained by former events. Actually, punctual data are hardly exploited because life-history traits measured over large temporal scales are the most appropriate to long-lived marine species studies (Hughes and Connell, 1999). Concerning the percent cover of species targeted for their potential sensitivity, the absence of a significant link observed with API could be explained by the very small percents and weak variations recorded for these species during our study. It does not contradict previous studies (see the introduction) or reject the potential usefulness of these species as indicators of water quality but the method we used was inadequate to highlight such a role.

Three metrics were selected for the CAI design: percent cover of builders, percent cover of bryozoans and sludge percent cover. As expected, sludge percent cover increased with API when percent cover of bryozoans moderately decreased with API. Percent cover of builders weakly decreased with API mostly because of station 26. This station (the shallowest one) was strongly covered by builders although a moderate API perhaps because of its relative weak depth. Taken together, these metrics bring information concerning both of the two parameters defined for the evaluation of the conservation state of Natura 2000 marine natural habitats: structure - functionality (state of the bioconstructions, diversity of associated assemblages) with the percent cover of builders and bryozoans and threats - pressures (physical perturbations, biological perturbations and pollution) with sludge percent cover (Lepareur, 2011). Builder species intrinsically constitute the basis of coralligenous concretions and assemblages. They are influenced by various environmental factors such as light, nutrients and salinity potentially influenced in turn by human pressures (waste water for example) (Ballesteros, 2006). Similarly, bryozoans are sensitive to nutrients, sediment deposition and to mechanical pressures such as diving or fishing. Lastly, sludge, and thus, turbidity (re-suspended sediment) and sedimentation have terrestrial origins largely affected by human activities since the past few decades (land overuse, deforestation, increased erosion, alteration of sediment transport) (see Airoldi's review (2003)). Several studies showed the influence of sediment deposition on coralligenous assemblages through various suspected mechanisms (clogging of filtering apparatus, inhibition of recruitment, alteration of metabolic process, limitation of algal production, increase of burial and scouring, etc.) (Valiela, 1995; Airoldi, 2003; Balata et al., 2005). Finally, CAI, with its integrated response at three levels (physics with sludge, structure with builders and community with bryozoans) reflected the main anthropogenic factors that were found to occur in littoral zones. The quality of seawater expressed by API was significantly and negatively correlated with CAI and the resulting ecological classification using CAI was in accordance with our field knowledge.

All metrics included in the CAI were obtained from photographic quadrats. This cheap-cost method presents the advantages to provide objective and repeatable estimations thanks to a fast and non-destructive underwater work (Bianchi et al., 2004). Such a method is particularly well adapted to life cycle and temporal variation studies and/or to deep water station survey requiring scarce taxonomic precision (no method can be more accurate in taxonomy than direct collection) (Bianchi et al., 2004). Thanks to this method, this preliminary study benefited on a strong field sampling effort. All stations were sampled during a short period (three weeks), which avoided seasonality effects. Moreover, a second sampling at three stations did not show any inter-annual effect and classifications remained similar. The next field campaign planned for summer 2011 and 2012 should confirm the robustness of the CAI index with the acquisition of new data in Corsica and Languedoc-Roussillon. These new data will be analyzed using CPCe 4.1 "Coralligenous assemblages version" (CPCe 4.1, 2011).

Regarding WFD, coralligenous assemblages do not figure among the four biotic descriptors (biological quality elements) to be characterized and monitored by member states for coastal waters: phytoplankton, benthic invertebrate fauna, macroalgae and angiosperms. Pressure categories (use, discharges, alien species and development) recognized to be able to affect ecological status and the link pressure/state/impact/response is required in order to appreciate measurement program efficiency and corrective actions such as use management, cleanup, effluent discharges decrease etc. CAI will complete this obligatory dispositive. By providing a supplementary measurement but also data at higher depths than provided for required descriptors, CAI reinforces ecological status appreciation. Moreover, respecting the link state/pressure, this tool will help to appreciate corrective measures efficiency.The goal of the Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC) is in line with the objectives of WFD. It aims to achieve good environmental status of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend. The final goal is to protect more effectively the marine environment across Europe but an important point is that each Member State "shall ensure that measures are cost-effective and technically feasible". Identified under Community legislation (Habitats Directive 92/43/CEE) as being a special habitat with biodiversity interest, coralligenous assemblages shall be monitored as a descriptor in the framework of MSFD. Pragmatic and cheap, CAI brings first relevant elements to be considered for MSFD measurement program draw up.

5. Conclusions

To our knowledge, this study brought the first results concerning an assay with the determination of a multimetric index for coastal waters based on invertebrates from hard bottom community. The Coralligenous Assemblage Index (CAI) was based on three metrics acquired using photographic quadrat analyses: sludge percent cover, bryozoa and builder species percent cover. CAI was proved to be a useful tool for the assessment of the ecological quality of coralligenous communities and ecological status of Mediterranean French coastal waters in accordance with WFD requirements and United Nations Environment Programme, Mediterranean Action Plan and Regional Activity Centre for Specially Protected Areas proposal of standard methods (UNEP-MAP-RAC/SPA, 2011). This index was demonstrated to detect likely effects of anthropogenic pressures. However, further applications and validation are needed in order to better adjust the quality scores and properly evaluate the strengths and weaknesses. The selection of metrics that were cheap and easy to acquire and analyze will guaranty an effortlessly use and understanding by coastal managers and stakeholders.

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