



**Universitat de les  
Illes Balears**

**DOCTORAL THESIS**

**2015**

**INTEGRATION OF MARINE HABITAT INFORMATION  
INTO THE STUDY OF FISH ECOLOGY: NEW APPROACHES  
FOR ECOSYSTEM BASED FISHERIES MANAGEMENT**

**Diego Álvarez Berastegui**





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Diego Álvarez Berastegui

Director: José Manuel Hidalgo Roldan

Director: Lorenzo Ciannelli

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Tutor: Antoni Martínez Taberner

Doctor by *Universitat de les Illes Balears*

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## List of Manuscripts

### Articles published in international peer review scientific journals.

- Diego Álvarez-Berastegui, José Amengual, Josep Coll, Olga Reñones, Juan Moreno-Navas, Tundi Agardy, 2013. Multidisciplinary rapid assessment of coastal areas as a tool for the design and management of marine protected areas. *Journal for Nature Conservation*. DOI: <http://dx.doi.org/10.1016/j.jnc.2013.07.003>

- Diego Álvarez-Berastegui, Lorenzo Ciannelli, Alberto Aparicio-González, Patricia Reglero, Manuel Hidalgo, José Luis López-Jurado, Joaquín Tintoré, Francisco Alemany, 2014. Spatial Scale, Means and Gradients of Hydrographic Variables Define Pelagic Seascapes of Bluefin and Bullet Tuna Spawning Distribution. *PLoS ONE* 9(10): e109338. doi:10.1371/journal.pone.0109338

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Álvarez, I., J. M. Rodríguez, I. A. Catalán, M. Hidalgo, D. Álvarez-Berastegui, R. Balbín, A. Aparicio-González and F. Alemany. 2015. Larval fish assemblage structure in the surface layer of the northwestern Mediterranean under contrasting oceanographic scenarios. *Journal of Plankton Research* 37:834-850.

Carbonell, A., A. Tor, D. Álvarez-Berastegui, P. Vélez-Belchi, A. dos Santos, R. Balbín and F. Alemany. 2014. Environmental driving forces determining the epipelagic decapod larval community distribution in the Balearic Sea (western Mediterranean). *Crustaceana* 87:686-714.

Coll, J., A. Garcia-Rubies, G. Morey, O. Reñones, D. Álvarez-Berastegui, O. Navarro and A. M. Grau. 2013. Using no-take marine reserves as a tool for evaluating rocky-reef fish resources in the western Mediterranean. *ICES Journal of Marine Science: Journal du Conseil* 70:578-590.

Edelvang, K., H. Kaas, A. C. Erichsen, D. Álvarez-Berastegui, K. Bundgaard and P. V. Jorgensen. 2005. Numerical modelling of phytoplankton biomass in coastal waters. *Journal of Marine Systems* 57:13-29.

Goñi, R., S. Adlerstein, D. Álvarez-Berastegui, A. Forcada, O. Reñones, G. Criquet, S. Polti, G. Cadiou, C. Valle and P. Lenfant. 2008. Spillover from six western Mediterranean marine protected areas: evidence from artisanal fisheries. *Marine Ecology Progress Series* 366:159-174.

Hidalgo, M., P. Reglero, D. Álvarez-Berastegui, A. P. Torres, I. Álvarez, J. M. Rodriguez, A. Carbonell, R. Balbín and F. Alemany. 2015. Hidden persistence of salinity and productivity gradients shaping pelagic diversity in highly dynamic marine ecosystems. *Marine environmental research* 104:47-50.

Hidalgo, M., P. Reglero, D. Álvarez-Berastegui, A. P. Torres, I. Álvarez, J. M. Rodriguez, A. Carbonell, N. Zaragoza, A. Tor and R. Goñi. 2014. Hydrographic and biological components of the seascape structure the meroplankton community in a frontal system. *Marine Ecology Progress Series* 505:65-80.

Muhling, B. A., P. Reglero, L. Ciannelli, D. Álvarez-Berastegui, F. Alemany, J. T. Lamkin and M. A. Roffer. 2013. Comparison between environmental characteristics of larval bluefin tuna *Thunnus thynnus* habitat in the Gulf of Mexico and western Mediterranean Sea. *Marine Ecology Progress Series* 486: 257–276.

Peña, M., A. Carbonell, A. Tor, D. Álvarez-Berastegui, R. Balbín, A. dos Santos and F. Alemany. 2015. Nonlinear ecological processes driving the distribution of marine decapod larvae. *Deep Sea Research Part I: Oceanographic Research Papers* 97:92-106.

Puerta, P., M. E. Hunsicker, A. Quetglas, D. Álvarez-Berastegui, A. Esteban, M. a. González and M. Hidalgo. 2015. Spatially explicit modeling reveals cephalopod distributions match contrasting trophic pathways in the western mediterranean sea. *PloS one* 10:e0133439.

- Reglero, P., L. Ciannelli, D. Álvarez-Berastegui, R. Balbín, J. L. López-Jurado and F. Alemany. 2012. Geographically and environmentally driven spawning distributions of tuna species in the western Mediterranean Sea. *MEPS* 463:273-284.
- Reglero, P., D. P. Tittensor, D. Álvarez-Berastegui, A. Aparicio-González and B. Worm. 2014. Worldwide distributions of tuna larvae: revisiting hypotheses on environmental requirements for spawning habitats. *Marine Ecology Progress Series* 501:207-224.
- Rodríguez, J. M., I. Álvarez, J. López-Jurado, A. García, R. Balbín, D. Álvarez-Berastegui, A. P. Torres and F. Alemany. 2013. Environmental forcing and the larval fish community associated to the Atlantic bluefin tuna spawning habitat of the Balearic region (Western Mediterranean), in early summer 2005. *Deep Sea Research Part I: Oceanographic Research Papers* 77:11-22.
- Stobart, B., D. Álvarez-Berastegui and R. Goñi. 2012. Effect of habitat patchiness on the catch rates of a Mediterranean coastal bottom long-line fishery. *Fisheries Research* 129:110-118.
- Tintoré, J, Vizoso, G, Casas, B, Heslop, E, Pascual, A, Orfila, A, Ruiz, S, Martínez-Ledesma, M, Torner, M, Cusí, S, Diedrich, A, Balaguer, P, Gómez-Pujol, L, Álvarez-Ellacuria, A, Gómara, S, Sebastian, Kr, Lora, Sebastián, Beltrán, J, Renault, L, Juza, M, Álvarez, D., March, D, Garau, B, Castilla, C, Cañellas, T, Roque, D, Lizarran I., Pitarch, S, Carrasco, M.A., Lana, A, Mason, E, Escudier, R, Conti, D, Sayol, J.M., Barceló, B, Alemany, F, Reglero, P, Massutí, E, Vélez-Belchí, P, Ruiz, J, Oguz, T, G. Marta, Álvarez, E, Ansorena, L, Manríquez, M. 2013. SOCIB: The Balearic Islands coastal ocean observing and forecasting system responding to science, technology and society needs. *Marine Technology Society Journal* 47:101-117.
- Vázquez-Luis, M., D. March, E. Álvarez, D. Álvarez-Berastegui and S. Deudero. 2014. Spatial distribution modelling of the endangered bivalve *Pinna nobilis* in a Marine Protected Area. *Mediterranean Marine Science* 15:626-634.
- Zaragoza, N., A. Quetglas, M. Hidalgo, D. Álvarez-Berastegui, R. Balbín and F. Alemany. 2015. Effects of contrasting oceanographic conditions on the spatiotemporal distribution of Mediterranean cephalopod paralarvae. *Hydrobiologia* 749:1-14.

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# Summary

The overexploitation of marine living resources challenges the scientific community for developing new analytical approaches providing effective tools for marine management, ensuring long-term conservation of the harvested and threatened species. Currently, the scientific efforts are mainly focused on the development of techniques and concepts to improve the assessment and management of these populations from a holistic point of view within the framework of the Ecosystem Based Management (EBM). While the principles and objectives of EBM have been accepted by the scientific community and those responsible for the management of the fisheries and conservation, there is not a consensus about how it should be implemented. One of the decisive reasons hindering its implementation is the complexity related to the modeling of complex socio-ecological systems, which covers from environmentally driven effects to social aspects in the management. Focusing on key processes of ecosystems such as the relationships between species ecological processes and essential habitats offers a path to advance towards the implementation of EBM without having to reach the development of excessively complex end-to-end models of an ecosystem.

The research developed along this PhD has two main objectives. 1) the application of new concepts and techniques to improve the characterization of essential habitats of two top predator species, the dusky grouper (*Epinephelus marginatus*, Lowe 1834) and the Atlantic bluefin tuna (*Thunnus thynnus*, Linnaeus 1758). 2) To propose new methodologies based on habitat information to improve current assessment and management approaches of those species.

Conservation of dusky grouper and Atlantic bluefin tuna exploited populations is tackled today from different technical approaches due to differences in their ecological characteristics. Dusky grouper is a highly resident species that inhabits rocky bottoms in coastal Mediterranean ecosystems, where conservation of exploited populations through the establishment of marine protected areas has provided positive results. Atlantic bluefin tuna is a highly migratory pelagic species with a wide geographical distribution along the Atlantic waters, and performs long migrations during spring to reach the spawning areas, among of which the Balearic Sea is one of the main ones. Management of Atlantic Bluefin tuna is approached mainly by technical measures such as minimum weight regulation and limitations in the total allowable catches, established as a function of the status of the adult stock populations calculated from virtual population analysis.

In recent years the application of traditional *landscape ecology* techniques to characterize habitat in the coastal environment has promoted the beginning of the *seascape ecology* discipline. These techniques have been applied here to improve the definition of dusky grouper essential habitats and to identify changes in habitat use along ontogenetic development. The definition of dusky grouper habitats at different developmental stages provided insights about the species ecology and provided criteria for designing more efficient Marine Protected Areas (MPAs). Findings from the study of dusky grouper essential habitat and the improvement on habitat definition by using seascape metrics provide the basis for developing new methods for MPA design within the framework of Rapid Assessment Programs. Therefore, in this PhD a method is proposed for rapid multidisciplinary environmental assessment of coastal areas for the design and management of MPAs. This method provide tools for the selection, design and management of coastal MPAs when time, budget or potential human pressures, either alone or in combination, create an urgent need for prioritization.

The conceptual scheme applied to link littoral species with essential habitats and the transference to management has been adapted to the pelagic environment. Transferring ideas and techniques of seascape ecology to the pelagic realm was not straightforward. New pelagic seascape metrics have been proposed and tested to study the Atlantic Bluefin tuna spawning habitats around the Balearic Sea, advancing in the knowledge of species ecology. The developed pelagic seascape metrics have been applied to the development of a spawning habitat forecasting model to assist managers. This methodology is entirely based on oceanographic data obtained from operational data sources. Finally, monitoring and modeling Atlantic bluefin tuna pelagic essential habitats at larval stages allowed developing new larval indices, providing information on Atlantic bluefin tuna adult eastern stock population.

# Resum

L'estat actual dels recursos vius i dels ecosistemes marins suposa un desafiament constant per a la comunitat científica, que obliga a un progrés continu que asseguri, a llarg termini, la seva explotació sostenible i la seva conservació. Avui en dia els esforços en la investigació se centren, en gran mesura, en el desenvolupament de tècniques i conceptes per millorar l'avaluació i la gestió d'aquestes poblacions des d'un punt de vista holístic en el marc de la gestió basada en l'ecosistema (EBM per les seves sigles en anglès). Mentre que els principis i objectius de l'EBM han estat acceptats per la comunitat científica i pels responsables de la gestió de la pesca i de la conservació, no existeix un consens sobre com s'ha d'aplicar. Un dels motius determinants que obstaculitzen la seva implantació és la complexitat associada a la modelització de sistemes socio-ecològics complexos, que abasta des dels efectes ambientals fins a aspectes socials en la gestió dels recursos. La identificació de processos clau en un ecosistema, com puguin ser les relacions entre l'ecologia de determinades espècies amb els seus hàbitats essencials, ofereix una possibilitat per avançar cap a l'aplicació de la EBM sense haver d'assolir el desenvolupament de models super-complexos que abordin tots els processos que ocorren en un ecosistema.

Els diferents estudis desenvolupats al llarg d'aquesta tesi doctoral tenen dos objectius principals. En primer terme s'aborda l'aplicació de noves tècniques i conceptes per tal de millorar la caracterització dels hàbitats essencials de dues espècies marines localitzades en els estatges superiors de la cadena tròfica, com són l'anfós (*Epinephelus marginatus*, Lowe 1834) i la tonyina vermella (*Thunnus thynnus*, Linnaeus 1758). En segon terme, el desenvolupament de metodologies basades en la informació obtinguda sobre els seus hàbitats essencials, la qual cosa permetrà millorar l'avaluació i gestió de les poblacions d'aquestes espècies.

Avui dia, la conservació de les poblacions explotades de l'anfós i la tonyina vermella s'aborden des d'enfocaments ben diferenciats, sobre la base de les seves característiques ecològiques. L'anfós és una espècie altament resident que habita fons rocosos dels ecosistemes costaners de l'oceà Atlàntic, l'oceà Índic i el mar Mediterrani. La conservació de les seves poblacions explotades es basa principalment en l'establiment d'àrees marines protegides. Per la seva banda, la tonyina vermella és una espècie pelàgica amb una àmplia distribució geogràfica al llarg de l'Atlàntic, que realitza llargues migracions durant la primavera per arribar a les àrees de reproducció, entre les quals s'hi troba el Mar Balear. La gestió

d'aquesta espècie es basa, principalment, en l'establiment de quotes de pesca a partir de l'avaluació de l'estat de les poblacions mitjançant l'aplicació de models monoespecífics.

En els darrers anys l'aplicació de tècniques procedents de l'ecologia del paisatge terrestre (*landscape ecology*) en estudis d'ecologia del medi costaner ha donat lloc a l'inici de la "ecologia del paisatge marí" (*seascape ecology*). En els estudis desenvolupats en el marc d'aquest doctorat s'han aplicat aquestes tècniques per millorar la definició dels hàbitats essencials de l'anfós i per identificar canvis en l'ús de l'hàbitat al llarg del seu desenvolupament ontogènic. La millora en la identificació dels hàbitats essencials de l'anfós a diferents etapes del seu desenvolupament ha proporcionat informació rellevant sobre la seva ecologia i criteris per al disseny d'àrees marines protegides més eficients quant a la conservació d'aquesta espècie. Els resultats obtinguts durant l'estudi dels hàbitats essencials de l'anfós i la millora en la caracterització de l'estructura dels hàbitats a través de l'aplicació de mesures de paisatge submarí han proporcionat la base per al desenvolupament d'una metodologia de disseny d'àrees marines protegides en el marc dels programes d'avaluació ràpida (RAPs per les seves sigles en anglès). En aquesta tesi doctoral es proposa un mètode per a l'avaluació ràpida d'àrees costaneres des d'un enfocament multidisciplinari. Aquest mètode proporciona eines per a la selecció, disseny i gestió d'àrees marines protegides costaneres quan el factor temps, el pressupost o l'acció humana, ja sigui individualment o combinats, crea una necessitat urgent de prioritització.

L'esquema conceptual aplicat en l'anàlisi de les relacions entre l'ecologia de l'anfós i els seus hàbitats essencials i la transferència d'aquesta informació a la millora en l'avaluació i gestió de l'espècie s'ha adaptat a l'ambient pelàgic. La transferència d'idees i tècniques de l'ecologia del paisatge marí a l'ambient pelàgic ha suposat un repte al llarg d'aquest doctorat. S'han proposat noves mètriques de paisatge marí pelàgic, que s'han aplicat en l'estudi dels hàbitats de reproducció de la tonyina vermella en el Mar Balear, la qual cosa ha permès avançar en el coneixement de l'ecologia d'aquesta espècie i d'altres de túnids. Aquests avanços han permès el desenvolupament d'un model de predicció de la localització de les zones de reproducció de la tonyina a les Balears, basats completament en l'aplicació de dades procedents de l'oceanografia operacional (teledetecció i models hidrodinàmics). Finalment, la monitorització dels estadis larvaris de la tonyina vermella i l'anàlisi dels seus hàbitats essencials han permès desenvolupar índexs d'abundància larvària, i demostrar que la informació sobre l'hàbitat millora significativament l'avaluació d'aquest índexs. Aquests índexs larvaris estan permetent analitzar l'evolució de la fracció adulta de la població oriental de tonyina vermella de l'Atlàntic, que es reproduïx al Mediterrani.



# Resumen

El estado actual de los recursos vivos y de los ecosistemas marinos supone un desafío constante para la comunidad científica, obligando a un progreso continuo que asegure, a largo plazo, su explotación sostenible y su conservación. Hoy en día los esfuerzos en la investigación se centran, en gran medida, en el desarrollo de técnicas y conceptos para mejorar la evaluación y gestión de estas poblaciones desde un punto de vista holístico en el marco de la gestión basada en el ecosistema (EBM por sus siglas en inglés). Mientras que los principios y objetivos de la EBM han sido aceptados por la comunidad científica y los responsables de la gestión de la pesca y la conservación, no existe un consenso sobre cómo debe aplicarse. Una de las razones determinantes que obstaculizan su implementación es la complejidad asociada a la modelización de sistemas socio-ecológicos complejos, que abarca desde los efectos ambientales hasta aspectos sociales en la gestión de los recursos. La identificación de procesos clave en un ecosistema, tales como las relaciones entre la ecología de determinadas especies con sus hábitats esenciales, ofrece una posibilidad para avanzar hacia la aplicación de la EBM sin tener que alcanzar el desarrollo de modelos super-complejos que aborden todos los procesos que ocurren en un ecosistema.

Los diferentes estudios desarrollados a lo largo de esta tesis doctoral tienen dos objetivos principales. En primer lugar, la aplicación de nuevas técnicas y conceptos para la mejora de la caracterización de los hábitats esenciales de dos especies marinas localizadas en los estados superiores de la cadena trófica, el mero (*Epinephelus marginatus*, Lowe 1834) y el atún rojo del Atlántico (*Thunnus thynnus*, Linnaeus 1758). En segundo lugar, el desarrollo de nuevas metodologías, basadas en la información obtenida sobre hábitats esenciales, que permitan mejorar la evaluación y la gestión de las poblaciones de estas especies.

Hoy en día, la conservación de las poblaciones explotadas del mero y el atún rojo se abordan desde enfoques técnicos bien diferenciados, en base a sus características ecológicas. El mero es una especie altamente residente que habita en fondos rocosos de los ecosistemas costeros del Mediterráneo. La conservación de sus poblaciones explotadas mediante el establecimiento de reservas marinas ha dado buen resultado. El atún rojo del Atlántico es una especie pelágica con una amplia distribución geográfica a lo largo de las aguas del Atlántico, que realiza largas migraciones durante la primavera para llegar a las áreas de reproducción, entre las que se encuentra el Mar Balear. La gestión de esta especie se basa, principalmente,

en el establecimiento de tallas mínimas de captura y de cuotas de pesca a partir de la evaluación del estado de las poblaciones mediante la aplicación de modelos uni-específicos.

En los últimos años la aplicación de técnicas procedentes de la ecología del paisaje terrestre (landscape ecology), en estudios de ecología en el medio costero ha dado lugar al inicio de la “ecología del paisaje marino”(seascape ecology). En los estudios desarrollados en el marco de este doctorado se han aplicado estas técnicas para mejorar la definición de los hábitats esenciales del mero y para identificar cambios en el uso del hábitat a lo largo de su desarrollo ontogénico. La mejora en la identificación de los hábitats esenciales del mero en diferentes etapas de desarrollo ha proporcionado información relevante sobre la ecología de esta especie y criterios para el diseño de áreas marinas protegidas más eficientes en cuanto a su conservación. Los resultados obtenidos durante el estudio de los hábitats esencial de mero, y la mejora en la caracterización de la estructura de los hábitats mediante la aplicación de medidas de paisaje marino, han proporcionado la base para el desarrollo de una metodología de diseño de áreas marinas protegidas en el marco de los programas de evaluación rápida (RAPs por sus siglas en ingles). En esta tesis doctoral se propone un método para la evaluación rápida de aéreas costeras desde un enfoque multidisciplinar. Este método proporciona herramientas para la selección, diseño y gestión de áreas marinas protegidas costeras cuando el factor tiempo, el presupuesto o la acción humana, ya sea solos o en combinación, crea una necesidad urgente de priorización.

El esquema conceptual aplicado en el análisis de las relaciones entre la ecología del mero y sus hábitats esenciales y la transferencia de esta información a la mejora en la evaluación y gestión de esta especie, se ha adaptado al ambiente pelágico. La transferencia de ideas y técnicas de la ecología de paisaje marino al ambiente pelágico ha supuesto un reto a lo largo de este doctorado. Se han propuesto nuevas métricas de paisaje marino pelágico, que se han aplicado en el estudio de los habitats de reproducción del Atún rojo en aguas del Mar Balear, lo que ha permitido avanzar en el conocimiento de la ecología de esta especie y otras especies de túnidos. Estos avances han permitido el desarrollo de un modelo de predicción de la localización de las zonas de reproducción del atún rojo en Baleares, basados enteramente en la aplicación de datos procedentes de la oceanografía operacional (teledetección y modelos hidrodinámicos). Finalmente, la monitorización de los estadios larvarios del atún rojo y el análisis de sus hábitats esenciales han permitido desarrollar índices de abundancia larvaria, y demostrar que la información sobre hábitat mejora significativamente la evaluación de estos índices. Estos índices larvarios están permitiendo analizar la evolución de la población adulta de la población oriental del atún rojo del Atlántico, que se reproduce en el Mediterráneo.

## Acronyms and glossary terms

The following table contains a list of acronyms most commonly used along the manuscript of this Doctoral thesis. Other acronyms, related to environmental variable names used in specific analyses are detailed along the manuscript.

Acronyms	
AGRRA	<b>A</b> tlantic and <b>G</b> ulf <b>R</b> apid <b>R</b> ef <b>A</b> ssessment. An international collaboration project for the assessment of the regional condition of reefs in the Western Atlantic and Gulf of Mexico.
AIC	<b>A</b> kaik <b>e</b> <b>I</b> nformation <b>C</b> riterion. A measure of the quality of various statistical models relative to each other. It is used for model selection.
AUC	<b>A</b> rea <b>U</b> nder the <b>C</b> urve. Performance metric of a logistic regression. It is a commonly used evaluation metric for binary classification problems like predicting presence-absence individuals of the species.
B60	<b>B</b> ongo <b>60</b> . A plankton sampling gear with a 60 cm radius double mouth.
B90	<b>B</b> ongo <b>90</b> . A plankton sampling gear with a 90 cm radius double mouth
BFT	Atlantic <b>B</b> luefin <b>t</b> una ( <i>Thunnus thynnus</i> , Linnaeus, 1758)
BOE	<b>B</b> oletin <b>O</b> ficial del <b>E</b> stado. Spanish government official gazette.
CANP	<b>C</b> abrera <b>A</b> rchipelago <b>N</b> ational <b>P</b> ark. A terrestrial-maritime national park located in the Balearic Islands
CPUE	<b>C</b> apture <b>p</b> er <b>U</b> nit <b>E</b> ffort. Relation between the catch (e.g. weight, number of individuals) and the effort associated to the catch. It is commonly used as index of abundance
EBM	<b>E</b> cosystem <b>B</b> ased <b>M</b> anagement. A management approach integrating the various elements and interactions composing an ecosystem, including human interactions.
EXA	<b>E</b> xposure estimates for fragmented <b>A</b> rchipelagos. A technical approach to measure wave energy on coastal areas.
GAM	<b>G</b> eneral <b>A</b> dditive <b>M</b> odel. A nonparametric statistical modeling approach where the response variable depends on explanatory variables through nonlinear link functions
GCV	<b>G</b> eneralized <b>C</b> ross- <b>V</b> alidation. Parameter informing about the quality of the statistical model. It is used for model selection within the same model family.
GOM	<b>G</b> ulf of <b>M</b> exico. It is one of the main spawning areas of the western stock of Atlantic bluefin tuna.
ICCAT	<b>I</b> nternational <b>C</b> ommission for the <b>C</b> onservation of <b>A</b> tlantic <b>T</b> una. An

	international organization responsible of the Atlantic bluefin tuna assessment that sets the total allowable catch.
ICZM	<b>I</b> ntegrated <b>C</b> oastal <b>Z</b> one <b>M</b> anagement. A management approach aiming at provide tools for planning, management and monitoring of coastal areas by integrating multidisciplinary information and involving all stakeholders
IUCN	<b>I</b> nternational <b>U</b> ion for <b>C</b> onservation of <b>N</b> ature
MPA	<b>M</b> arine <b>P</b> rotected <b>A</b> rea. A geographic area where human activities are restricted and regulated.
MSFD	<b>M</b> arine <b>S</b> trategy <b>F</b> ramework <b>D</b> irective. A European legislative framework aiming at achieving or maintaining a good status of the European Marine environment by 2020.
NOAA	<b>N</b> ational <b>O</b> ceanic and <b>A</b> tmospheric <b>A</b> dministration. The EEUU federal agency focused on understanding and predicting changes in climate, weather, oceans, and coasts as well as conserving and managing coastal and marine ecosystems and resources
ODP	<b>O</b> perational <b>D</b> ata <b>P</b> roducts. Specific data products (e.g. sea surface temperature maps or salinity profiles for example) provided by the application of operational oceanography techniques
SITIBSA	<b>S</b> erveis d' <b>I</b> nformacio <b>T</b> erritorial <b>d</b> e <b>l</b> es <b>I</b> lles <b>B</b> alears. The Balearic islands territorial information service
SSB	<b>S</b> pawning <b>S</b> tack <b>B</b> iomass. Biomass (weight) of individuals of a particular fish species that are sexually mature
TAC	<b>T</b> otal <b>A</b> llowable <b>C</b> atch. A catch limit set for a particular fishery during a fishing season (definition by the Organization for Economic Co-operation and Development)
UVC	<b>U</b> nderwater <b>V</b> isual <b>C</b> ensus. A visual based technique for evaluating populations of nectobenthic species performing diving transects.
VPA	<b>V</b> irtual <b>P</b> opulation <b>A</b> nalysis. A modeling approach based on cohort structure and evolution used in fisheries management for analyzing historical fish abundances.
WWF	<b>W</b> orld <b>W</b> ildlife <b>F</b> und.

<b>Glossary terms</b>	
End to end model	Modeling approach aiming at simulate ecosystem processes considering the dynamic effects of both the physical environment and the human activities, including all trophy levels
Full ecosystem	A group of techniques aiming to model the entire ecosystem by

approach modeling	disentangling all the interrelationships among predator and preys. Opposite to minimum-realistic approach.
Landscape ecology	A scientific discipline focusing on the characterization of the landscape pattern structure, how it changes over time and how it affects ecological processes
Landscape ecology metric	A quantitative measure of landscape structure pattern (e.g. fragmentation, core area, connectivity)
Minimum-realistic approach modeling	Analytical ecosystem approach aiming to identify key aspects of an ecosystem that are affordable to be describe by realistic numerical modeling.
Operational fisheries oceanography	A scientific discipline that focuses on improving fisheries management by taking advantage of operational oceanography tools and data
Operational oceanography	Systematic and long-term routine measurements of the seas and oceans and atmosphere, and their rapid interpretation and dissemination (definition of EuroGOOS)
Pelagic seascape ecology.	A scientific discipline aiming at improving the understanding on how the structure patterns of pelagic habitats affects species ecology.

# **CHAPTER 1**

## **General introduction, objectives and thesis structure**

## 1.1 Motivation of the PhD thesis

The conservation of threatened marine species and the long-lasting sustainable exploitation of living resources pose continuous challenges in the scientific community endeavors. Current efforts are framed in the Ecosystem Based Management (EBM), in which scientists mainly focus in the development of techniques and concepts to improve the assessment and management of populations from a holistic perspective. Eventhough scientific community agrees in the general principles of EBM, there is not a clear consensus yet on how to implement it. One of the most important reasons is the high complexity of the socio-ecological systems, and its high degree of variation within a regional scale. This complexity covers from environmental drivers to social aspects of the management. This evidences the crucial importance of focusing in understanding and modeling key processes of marine ecosystems as the most effective avenue to ensure the implementation of EBM, rather than developing highly complex ecosystem models (Cury *et al.* 2008). In this sense, the relationships between key ecological processes of species and essential habitats are, without doubts, a main pillar for the future development and implantation of EBM.

Recent advances in data acquisition of benthic and pelagic marine habitats offer today the possibility of analyzing habitat-ecology relationships that traditionally have been limited due to the lack of environmental information. The main motivation throughout this PhD thesis has been **developing and progressing in techniques and concepts necessary to address the study of relationships between species ecology and essential habitats, and the application of that knowledge to the improvement of the assessment and management of marine exploited and threatened species**. To do that, I framed the thesis on the development of the emerging “seascape ecology” discipline by transferring techniques and concepts from the “landscape ecology”, widely developed in the Earth's environmental science, to the marine environment.

These issues have been approached for two emblematic species in the Balearic archipelago, Dusky grouper (*Epinephelus marginatus*, Lowe 1834), species characteristic of the coastal ecosystem and whose management is addressed mainly through marine protected areas (MPAs) and Atlantic bluefin tuna (*Thunnus thynnus*, Linnaeus 1758 ) a highly migratory pelagic species that has one of the most important spawning areas around the Balearic Islands. Developing specific products from science directly applicable in the improvement of species assessment and management was the original motivation of this PhD thesis.

## 1.2 Background

### 1.2.1 General state of marine resources and ecosystem

The steady increase over recent decades in the number of endangered marine species (Baillie *et al.* 2004), the evidence of overexploitation of fishery resources (Pauly *et al.* 1998; Jackson *et al.* 2001; Schiermeier 2002; Swan & Gréboval 2005) and the marine biodiversity loss (Sala & Knowlton 2006; Worm *et al.* 2006) are examples of the persistent deterioration of marine ecosystems and the services they provide for the society (Tegner & Dayton 1999; Steele & Schumacher 2000; Jackson *et al.* 2001; Worm *et al.* 2006). The human impacts on marine ecosystems affects the populations of key top predators, such as sharks (Baum *et al.* 2003), mammals (Schipper *et al.* 2008), large migratory fish (Myers & Worm 2003; Allen 2010; MacKenzie & Mariani 2012) and other species even when they are not direct objective of fishing (Lewison *et al.* 2004). An overview on the temporal evolution of the status of fishery resources shows that along the last four decades the proportion of over-exploited fisheries has increased continuously to reach a level of 30%, being 60% of the fisheries in the limit of sustainability, whereas only 10% of fisheries are considered in good condition (FAO 2014). The situation may be worse if we consider that this information refers only to fisheries under some degree of control. Several studies suggest that much of the fishing effort is out of the statistics and has not been considered in these analyses, so the pressure over exploited species is most likely greater (Watson & Pauly 2001; Belhabib *et al.* 2014; Cressey 2015).

### 1.2.2 Developments in the approaches applied to the assessment and management of marine resources

Within the current context of overexploitation, continuous improvement in the strategies for the conservation of the species and marine ecosystems is a must that has promoted actions in the field of policy, assessment, management and science (Rice 2011). Most efforts have focused on understanding the functioning of marine ecosystems, monitoring, and transferring the acquired knowledge to the processes of assessment and management, moving from a single-species to more holistic approaches (Caddy & Cochrane 2001). Single-species based approaches are still dominant in the assessment and management of many relevant fisheries worldwide (Pitcher *et al.* 2009). However, while single-species approach may be valid for many fisheries (Cowan Jr *et al.* 2012), current and forefront research in fisheries ecology is fostering the need of evaluating and managing marine resources from a broader perspective in



which the relationships of the species with their environment are considered. Ecosystem Based Fisheries Management (EBFM), developed from the application of EBM to fisheries (Link 2002; García 2003; Pikitch *et al.* 2004; McLeod & Leslie 2009; Essington & Punt 2011), has been widely assimilated by countries and international agencies managing relevant fisheries (Rice 2011; Hilborn 2011). Nowadays, even broader perspectives of management are proposed, aiming to encompass all aspects involved in a socio-ecological system (Ostrom 2009), and including adaptive management (Allen *et al.* 2011). The main objective of these approaches is to maintain marine ecosystems in a healthy, productive condition and optimum resilience, so that they can provide the necessary services for the society (Liquete *et al.* 2013).

### **1.2.3 Lines of development within the ecosystem based management framework**

The assessment and management approaches of marine resources towards more holistic conceptual frameworks have evolved through different paths of technical development and implementation. One of these paths has been the development of procedures for the assessment of the status of populations through multi-species and ecosystem models (Fulton *et al.* 2003). These models consider the effect of many environmental and biological factors over a particular exploited resource, finally integrating the ecosystem effects in a wider modeling framework considering also the economic and sociological aspects of management (García & Cochrane 2005). This approach (see figure 1.1), which is becoming the focus of new advances within relevant agencies, such as the International Council for the Exploration of the Seas (ICES), is highly complex and requires an enormous amount of information relating to the entire socio-ecological system. Practical implementation is still in progress with great effort (Fulton *et al.* 2003; Fulton *et al.* 2011; Dickey-Collas 2014). Although the need for the application of these concepts to management is accepted, there is still not a clear consensus about how to progress towards the objectives proposed by the EBM (Essington & Punt 2011), and implementation at present is still limited (Pitcher *et al.* 2009). Therefore, the application of end to end ecosystem models for assessing fisheries management may found reluctance among responsible fisheries due the over complexity associated with these techniques, causing political interferences in management of marine living resources (Cowan Jr *et al.* 2012). The increase of uncertainty associated with this type of models has resulted in the assertion that it is preferable to deal with the analyses of the impacts on the marine environment and species through the combination of several simple but realistic models ("minimum-realistic" approach) rather than using models trying to include all the possible

interrelations of all elements within an ecosystem ("full ecosystem" approach) (Fulton *et al.* 2003; Cury *et al.* 2008).

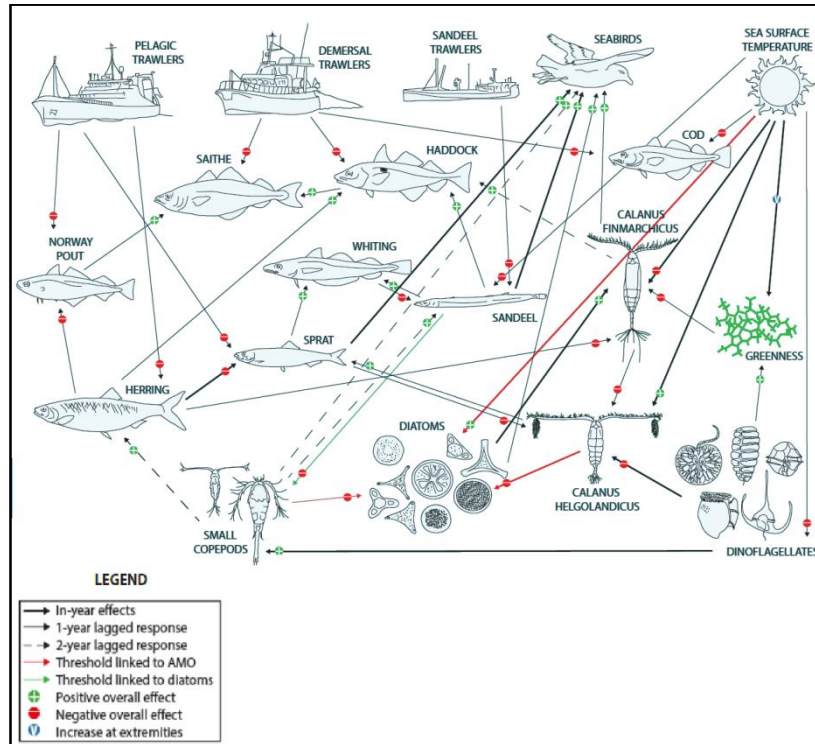


Figure 1.1: The North Sea ecosystem model. Relations between functional groups used to evaluate key food-web links, climate change and fishing pressure in the framework of the “ICES WG Integrated Assessments of the North Sea”. Adapted from SCICOM 2014.

Another different approach for implementing EBM for fisheries and conservation in marine environments has been the development of Marine Protected Areas (MPAs) (Lauck *et al.* 1998; Roberts *et al.* 2001; Pauly *et al.* 2002; Fulton *et al.* 2003). The application of MPAs implies a significant conceptual change in the way that marine resources are managed, involving conservation of the diverse components of an ecosystem, the habitats and species and in consequence their interactions, as a whole, instead of focusing on discrete elements as a single species. MPAs can reverse important impacts on habitats and exploited species (García-Charton *et al.* 2008) and could serve as economic boost of local coastal villages (Sala *et al.* 2013). MPAs are considered appropriate management tools for coastal fisheries, with multiple examples in the Western Mediterranean (Pérez-Ruzafa *et al.* 2008). Nowadays, MPAs are being proposed for managing living resource in the pelagic environment as well (Game *et al.* 2009). Dynamic pelagic closure areas, where the geographic limits of the fisheries restricted domain is modified as a function of the local oceanography, have already applied been

successfully in eastern Australia (Hobday *et al.* 2010), maybe the only current example available of a pelagic dynamic MPA.

#### **1.2.4 Essential habitats as a key element for the improvement of assessment and management approaches**

The ecology of exploited and endangered species is strongly linked with their essential habitats (Valavanis 2009). Therefore, for the successful implementation of the two previously raised assessment and management approaches, ecosystem modeling and MPAs, consideration of the relationships between the species of interest and their essential habitats is paramount (Rosenberg *et al.* 2000).

In the case of MPA, the success of protection measures will depend on many factors related to the MPA design (Claudet 2011), especially those linked to the adequacy of habitats protected (Roberts *et al.* 2003b). In the Mediterranean coastal MPAs, the carrying capacity of the fisheries targeted species depends on the essential habitats protected (Coll *et al.* 2012). In the case of coastal artisanal fisheries assessed by trends of capture per unit effort (CPUE), strong uncertainties in fish population may appear if the total area of essential habitats sampled is not considered in the calculations (Stobart *et al.* 2012). In the pelagic environment, hydrographic and biogeochemical conditions associated to particular oceanographic processes, such as frontal zones, define essential habitats of many species (Shillinger *et al.* 2008; Hobday *et al.* 2010; Scales *et al.* 2014), and variability of the local or regional oceanography may also determine relevant ecological processes such as recruitment (Cury *et al.* 2008; Ruiz *et al.* 2013). However, despite the information provided from the analysis of essential habitats could improve the assessment and management of many species, its direct application is still a pending issue.

Identifying how essential habitats drive key ecological processes of species of interest will provide a way to integrate that information in the assessment and management, moving towards to EBM while maintaining the principle of "minimum-realistic" approach. For example, the identification of how essential habitats shape the spatial distribution of a species may be the main issue for zoning uses and activities within a MPA. The challenge is similar in the pelagic environment, in which improving the management through the identification of how oceanography drives key processes has been also proposed within the framework of the "ecosystems oceanography" (Cury *et al.* 2008).

### 1.2.5 Main scientific and technical challenges

A first step to advance towards the development of new assessment and management approaches based on the knowledge on species-habitat relationships is the identification and evaluation of these relationships. Recently, several authors propose advancing in the study of essential habitats in marine species within the framework of the developing discipline of "seascape ecology" (Pittman *et al.* 2011). The development of concepts and techniques for the study of how essential habitats drive the ecology of the species on the terrestrial environments has been one of the main objectives of the "landscape ecology", a scientific discipline with already more than seven decades of history (Turner *et al.* 2001). Within landscape ecology, landscapes are defined as a geographic area in which the descriptor variables are heterogeneously distributed in space. For example, in a forest, the elements that configure the landscape may be defined by the mosaic of different tree species, rivers or meadows. Within a political landscape on a continental scale, these elements may be defined by the countries ranked by their growth domestic product (Ercan 2013). Landscape ecology researchers have developed tools for the analysis of landscapes by considering these elements as a mosaic of patches or polygons (figure 1.2). From here, multiple metrics were developed to quantify dispersion, aggregation, extension or level of interrelation (Gustafson 1998). Adapt techniques of landscape ecology to their application to the marine environment is the main challenge of the recent "seascape ecology" (Hinchey *et al.* 2008b; Pittman *et al.* 2011).

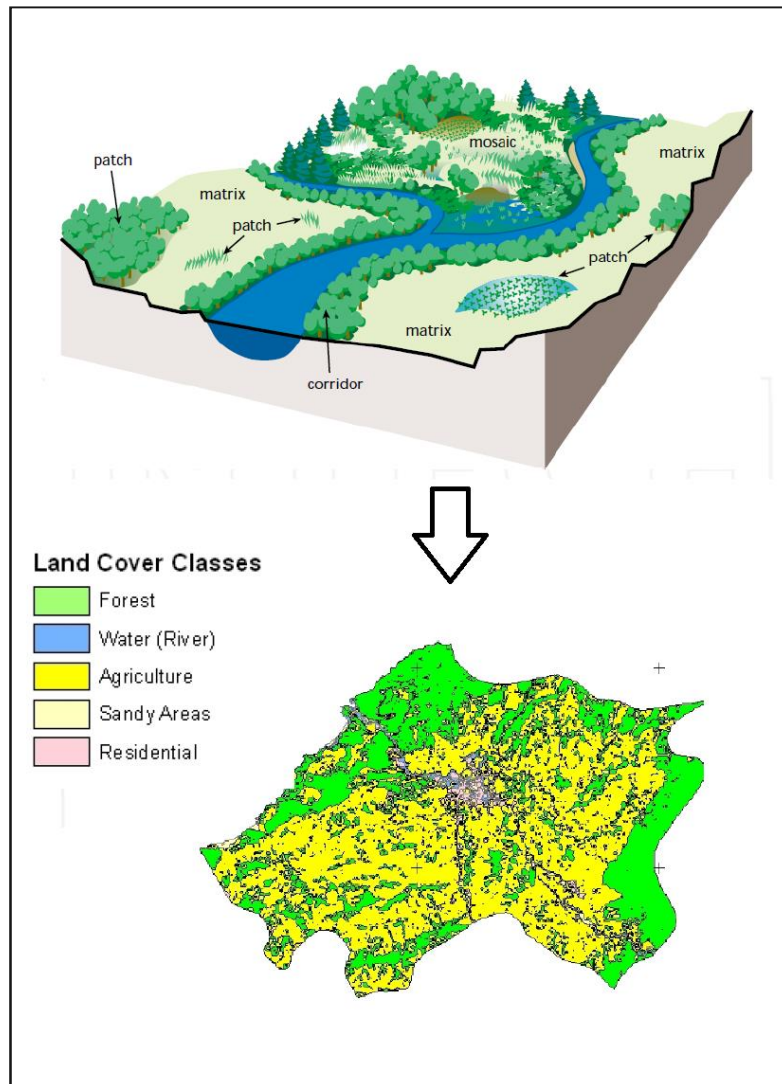


Figure 1.2: ·Example of landscape categorization using the patch mosaic concept of the landscape structure. Land cover classes are used to evaluate changes of landscape structure over time (adapted from Gökyer 2013).

Over the last decade, seascape ecology has focused on the application of landscape ecology concepts to the study of benthic habitats in coastal areas, where detailed habitat maps are available (Bostrom *et al.* 2011). In coastal areas, rocky bottoms of different structural complexity, seagrass meadows or detritic bottoms can define a mosaic of different habitat patches. Therefore concepts and techniques of landscape ecology, like the mosaic based landscape metrics used to quantify habitat patterns (Turner 1991), are directly applicable. However, the transfer of concepts and techniques of the ecology of the landscape to the pelagic environment is not straightforward (Pittman *et al.* 2011). In the pelagic environment, the different elements that configure a particular essential habitat do not present clear

delimitations and most of them are ephemeral and highly dynamic (figure 1.3). Therefore, the traditional analysis for the parameterization of the landscape tools cannot be applied. Finding new approaches to analyze the pelagic seascapes in the way that concepts and questions from landscape ecology can be tackled, is today a challenge of the emerging “pelagic seascape ecology”.

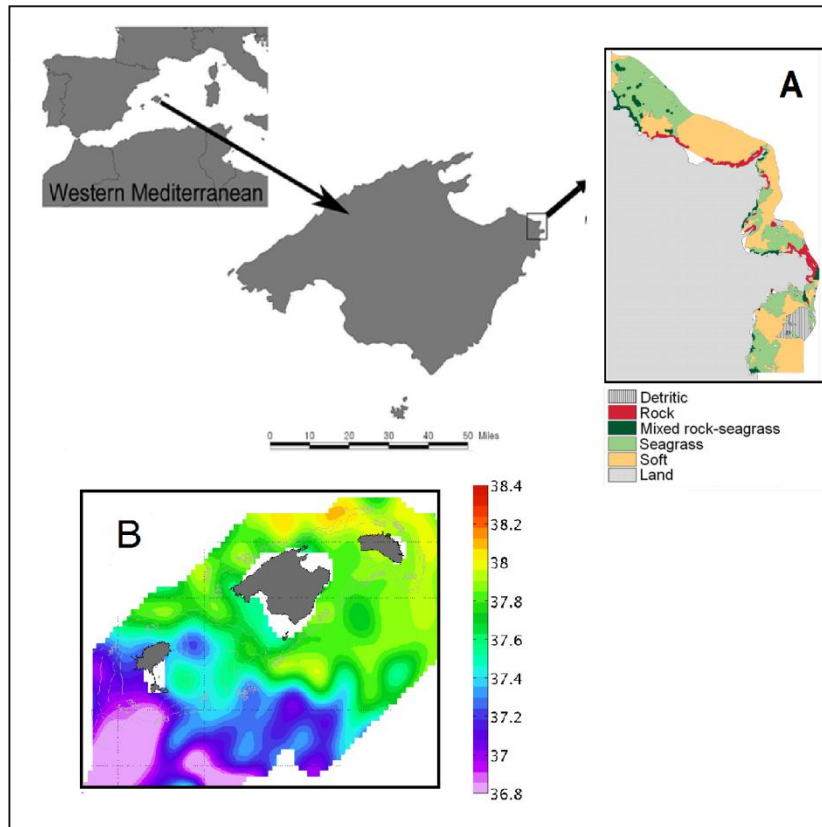


Figure 1.3: **A**-Patch mosaic benthic seascape of a coastal area at the North-East of Mallorca. Various bottom types are categorized and delimited to study habitat structure patterns using traditional landscape ecology metrics. **B**- Spatial variability of sea surface salinity around the Balearic archipelago.

One of the paramount questions of analysis in landscape ecology is how the spatial scale of the observations affects our ability to identify specific habitat patterns to properly characterize a key ecological process. The concept of spatial scale has great relevance also in seascape ecology. Multiscale hierarchical analysis show that spatial scale is a key factor for understanding how environmental forgings affects species abundances in coastal areas (Pittman *et al.* 2004) and pelagic communities assemblages (Álvarez *et al.* 2015). The analysis of seascapes at multi-scale levels have improved also the definition of essential fish habitat in benthic ecosystems (Kendall *et al.* 2011; Pittman & Brown 2011). The possibility of applying

multiscale seascape approaches may have also great relevance for the parameterization of pelagic habitats when trying to identify how oceanographic phenomena (e.g., fronts, eddies, filaments, turbulence) affects species ecology. If a particular ecological process is driven by oceanographic phenomena occurring at large spatial scales, such as the migration of leatherback (*Dermochelys coriacea*) in the Northwest Atlantic Northwest (Dodge *et al.* 2014), or the distribution of large migratory fish (Scales *et al.* 2014), the relation between the species-ecology and the oceanographic variables (e.g. distribution of chlorophyll, surface temperature, frontal processes) will emerge only if the observation occurs at large spatial scales. Also, if the distribution of a species is determined by processes occurring at smaller scales, these relations not will emerge in the analysis if the scale of observation is too wide.

In this PhD thesis these concepts of seascape ecology and spatial scale are applied to define the essential habitats at particular developmental stages of two species, the dusky grouper (*Epinephelus marginatus*), and the Atlantic bluefin tuna (*Thunnus thynnus*). The two species, relevant from the ecological and economical point of view, are top predators of the coastal and pelagic ecosystems respectively, as well as target species of important fisheries (Reñones *et al.* 1999; Fromentin & Powers 2005; Hinchey *et al.* 2008b). Knowledge gained on habitat-species relationships of these two species is applied to improve current assessment methods applied nowadays within two management approaches: MPAs-related framework and population dynamics modeling.

### **1.2.6 Geographic framework, species and assessment approaches analyzed in the thesis**

The Balearic Island provides a unique scenario to make advances on the study of species and essential habitat relationships, presenting in addition the potential for implementing this knowledge on assessment and management of fisheries. In this geographical area different management frameworks exist for different key species, habitat maps are available for existing MPAs (Posidonia-LIFE-Project 2001), and progress on the field of operational oceanography provides *in situ*, satellite and hydrodynamic data (Tintoré *et al.* 2013; Aparicio-González *et al.* 2015).

Within the Balearic Islands MPAs, the dusky grouper is a coastal species with a positive response to the protection (Reñones *et al.* 1999). The definition of dusky grouper essential habitats provides key information to evaluate the design of existing MPAs to protect this species and to assess new ones. Besides, novel approaches based on essential habitats

information allows assessing the status of exploited populations of this species in coastal areas (Coll *et al.* 2013b).

Bluefin tuna (*Thunnus thynnus*) is a highly migratory top predator that sustain relevant fisheries (Fromentin & Powers 2005). During winter season this species is distributed along feeding areas in the Atlantic. During the spring, adult individuals migrate to reproductive areas in the Gulf of Mexico and the Mediterranean. Within the Mediterranean, the Balearic Sea is one of the most relevant spawning areas (Fromentin & Powers 2005; García *et al.* 2005a). In this area, reproductive ecology of this species is strongly affected by mesoscale oceanography (Reglero *et al.* 2012; Muhling *et al.* 2013), and therefore spatial location of spawning grounds may vary among years as a function of the inter-annual differences in the oceanographic scenario. Identifying how local mesoscale oceanography affects spawning habitats provides key information for improving adult stock evaluation from larval indexes. Larval index relate adult population with total larval abundance, and it is the unique method routinely available for assessing bluefin tuna stock status from fishery-independent data (Ingram *et al.* 2010). Besides, spawning habitats mapping would provide the possibility of for applying new management concepts as the dynamic pelagic MPAs for tuna species conservation (Hobday *et al.* 2010).

### **1.3 Objectives and structure of the PhD Thesis**

This PhD thesis has two general objectives, firstly, to develop concepts and analytical tools to advance in the study of species ecology and the essential habitat relationships, and secondly, to propose specific ways to apply that knowledge in the evaluation and management of two species in the littoral and pelagic ecosystem. These species, dusky grouper and Atlantic bluefin tuna, are the target of important fishing activities and are key elements in marine ecosystems.

To reach these general objectives, the essential habitats of dusky grouper and Atlantic bluefin tuna are investigated transferring concepts and techniques from the discipline of seascape ecology. In the two cases, the effect that spatial scale has on our capability for defining these habitats has been investigated. Knowledge gained in these studies was applied for the development of techniques to improve current assessment and management of these species.



### 1.3.1 PhD Thesis structure

The PhD thesis document has been structured in eight chapters to achieve the proposed objectives. Chapter one presents a general overview of the objectives and challenges that will be addressed along the following chapters. Chapters two and three focus on the study of dusky groupers and marine protected areas, as fisheries and conservation management tool, in coastal ecosystems. Chapters four, five and six are focused on the study of Atlantic bluefin tuna pelagic habitats and assessment. Chapter seven presents a general discussion for the PhD, and chapter eight provides the general conclusions. All studies were developed in the geographic framework of the Balearic Islands as study area.

More specifically, **chapter two** investigates how rocky habitat structure and topography observed at different spatial scales provides information on the habitat definition for dusky grouper in two ontogenetic states, juvenile and adult. This information, combined with fishing pressure and local hydrodynamics, is applied to get a habitat model from a multiscale seascape approach. Results are discussed in terms of species ecology and design and management of MPAs.

In **Chapter three**, concepts of habitat, scale and stakeholder participation are applied to the development of a methodology for rapid assessment of coastal areas that facilitates the design of MPAs from a multidisciplinary point of view. This study analyzes the process of MPA design from an ecosystem approach framework, which combines both biological (e. g. spatial distribution of threatened species) and social aspects (e.g. the spatial distribution of uses and activities within the area). This methodology leads to the generation of specific end products that facilitate the processes of participation and decision-making during the design of a coastal MPA.

**Chapter four** presents a methodological approach for the parameterization of the pelagic seascape. This approach allows addressing the issue of the scale of observation on the identification of bluefin tuna and bullet tuna (*Auxis rochei rochei*, Risso 1810) essential habitats, as it has been addressed previously for the coastal ecosystem. New pelagic seascape metrics are proposed and tested for the improvement on the identification of spawning areas for the two species. The proposed seascape approach allowed identifying differences in the reproductive ecological strategies among highly migratory bluefin tuna and the smaller and more resident bullet tuna.

**Chapter five** presents a methodology for predicting the location of bluefin tuna spawning areas in the Balearic Sea, based exclusively on environmental data from operational

oceanography (hydrodynamic models and remote sensing data). These models are intended to design dynamics marine protected areas in the pelagic environment and to improve the larval indices, a method for the assessment of spawning stock abundance of species from larval abundances.

**Chapter six** presents the development of a new larval index for the Eastern stock of Atlantic bluefin tuna that reproduces in the Mediterranean. The new larval index accounts for errors in traditional larval index methods derived from changes in the spatial distribution of spawning habitats, applying the knowledge gained in the Chapters four and five. This new larval index improves current methodologies applied by the International Commission for the Conservation of Atlantic Tunas (ICCAT) to assess the trends of bluefin tuna adult stock abundances.

Detailed discussions for the results obtained in each study are included in each chapter. **Chapter seven** summarizes the main findings and present how the previous chapters link together, analyzing the role that the habitat and environmental descriptors play in shaping dusky grouper and bluefin tuna essential habitats, and how the this information has been applied to improve assessment and management by developing new technical approaches.

## **CHAPTER 2**

**Essential habitats of juvenile and adult dusky grouper (*Epinephelus marginatus*, Lowe 1834) in Western Mediterranean, a multiscale seascape analysis**

## 2.1 Introduction

Littoral ecosystems have been deeply transformed over centuries (Pauly 1995; Jackson 1997; Jackson 2001). Extinction or depletion of macrofauna makes top elements of the trophic chain to be just a glance of what they were in pristine ecosystems. In this framework, macro-carnivores fish like groupers become the main top predators structuring littoral food webs (Sala 2004). Groupers' populations have shown positive responds to protection in coral and rocky reefs environments in tropical (Polunin & Roberts 1993; Sluka *et al.* 1997; Dahlgren 2014) and temperate ecosystems (Reñones *et al.* 1999; García-Rubies *et al.* 2013). These results confirm the potential of marine protected areas for recovering overfished populations, and to provide the appropriate sites for investigating the essential habitats of these species; a knowledge that otherwise would be sharply skewed due to effects of fishing pressure (Reñones *et al.* 1999; Reñones *et al.* 2007, Hereu *et al.* 2006).

Among the different species of groupers that stand out within Mediterranean rocky fish assemblages it is the dusky grouper, *Epinaphelus marginatus*, the more frequent and abundant species. Analyses of dusky grouper essential habitats in well established MPAs provide key information for understanding species ecology and improving the design of MPAs. Most studies on dusky groupers have been conducted using underwater visual census (UVC), covering relative small spatial scales, from 2m<sup>2</sup> (La Mesa *et al.* 2002) to 250 m<sup>2</sup> (Louisy *et al.* 2007; Vacchi *et al.* 2007). These studies found a relationship between the abundance of groupers and the habitat descriptors measured at small spatial scales (within the UVC transects), such as rocky habitat complexity and depth. Some of these studies found that essential habitats change along ontogeny (i.e. Harmelin & Harmelin-Vivien 1999). Juveniles are mainly located at shallow depths above 15 m associated to rocky bottoms with shelters generated by medium size rocky blocks, while adults occupy deeper depths than juveniles with a preference to rocky bottoms with steep slopes with large cryptic shelters. These environmental characteristics define requirements of habitat at different developmental stages stablishing thresholds of habitat variables defined at small spatial scales.

Nevertheless, the habitat structure could be better defined when observed at multi-scale levels. The issue of spatial scale is paramount in ecology and it is one of core elements in "landscape ecology" research, a well established scientific discipline that is focused on how habitat patterns drive species ecological process (Turner *et al.* 1989; Dungan *et al.* 2002; Rietkerk *et al.* 2002). If a specific ecological process is driven by an environmental feature characterized at specific spatial scale, it will not be possible to identify the species-habitat

relationships if the analyses are developed at a different spatial scale (Turner *et al.* 1989). Furthermore, the issue of scale is even more complex, as an ecological phenomena may show variability in a range of spatial scales (Levin 1992). Concepts and techniques developed in the framework of landscape ecology, like the application of habitat structure metrics, or the effect of scale on habitat identification, have started to be applied in marine environments initiating a new “seascape ecology” discipline (Hinchey *et al.* 2008a; Huntington *et al.* 2010; Pittman & Brown 2011; Pittman *et al.* 2011). Application of a multiscale seascape ecology approach to analyze the essential habitats of the dusky groupers at different developmental stages would provide new insights on dusky grouper ecology providing new key information for MPA design, and would serve as case study to advance in seascape ecology.

In this study, we analyze the dusky grouper essential habitats at two different developmental stages, juvenile and adults, within a marine protected area. We compare how rocky habitat structure and topographic features measured at different spatial scales provide different information on the dusky grouper habitat requirements. We also identify the spatial scales at which essential habitat are best defined for the two developmental stages. Finally, we approach a multiscale seascape definition of the essential habitats for adult and juvenile individuals, combining the habitat descriptors at the different spatial scales providing complementary information with local hydrodynamics and fishing effort spatial distribution, factors that also affect spatial distribution of this species (Zabala *et al.* 1997; Reñones *et al.* 1999; Álvarez Berastegui *et al.* 2010). This analytical framework allows identifying which environmental variables and which spatial scales better define the dusky grouper essential habitats. We discuss the results in terms of species ecology and MPA design and management.

## 2.2 Methods

### 2.2.1 Study area.

The Cabrera Archipelago National Park (CANP) is located at the South East of Mallorca Island (Balearic Islands, Spain), in the Western Mediterranean (Figure 2.1). The archipelago was declared in 1991 (ley 14/1991, de 29 de abril de 1991, de creación del Parque nacional marítimo-terrestre del archipiélago de Cabrera, BOE, nº. 103, de 30 de abril de 1991). Activities within the park are regulated through various legislative documents (Plan de Ordenación de los Recursos Naturales, Real Decreto 1431/1992, de 27 de noviembre; Plan Rector de Uso y Gestión Decreto 58/2006 de 1 de julio). The archipelago is composed by a group of islands

surrounded by a number of small islets. The park embraces a total area of 10.021 ha, from which 8.705 are marine, with a high variety of coastal morphologies, as high and low cliffs, coves and bays. Coverage of the archipelago from the North-West by the Mallorca Island and openness in other directions, produce a strong gradient of wave exposure along the coast (Álvarez Berastegui *et al.* 2010). The archipelago integrates diverse coastal benthic habitats, with rocky and sandy bottoms, seagrasses, underwater caves and detritic sea beds. A full cartography of marine bottoms can be found at <http://lifeposidonia.caib.es/user/carto/PDFs/CABRERA.pdf>. The archipelago also shows areas with different topographic characteristics, from the gentle bathymetric gradients in the North-East to the steep bathymetry in the Southern area (see bathymetric isolines in Figure 2.1). The CANP was declared National Park, the highest protection level in Spain, in 1991. The marine zoning divides the park waters into *integral reserves*, *restricted use* and *moderate use* areas. Integral reserves and restricted use zones are no take areas while some professional artisanal fisheries occur in moderate use areas. Among them, bottom longlines are the main gear used to capture dusky grouper. Fishing is forbidden at depths above -20 meters and, at the time of this study there was a minimum length regulation from which individuals of dusky grouper below 45 cm cannot be retained on the fishing vessels.

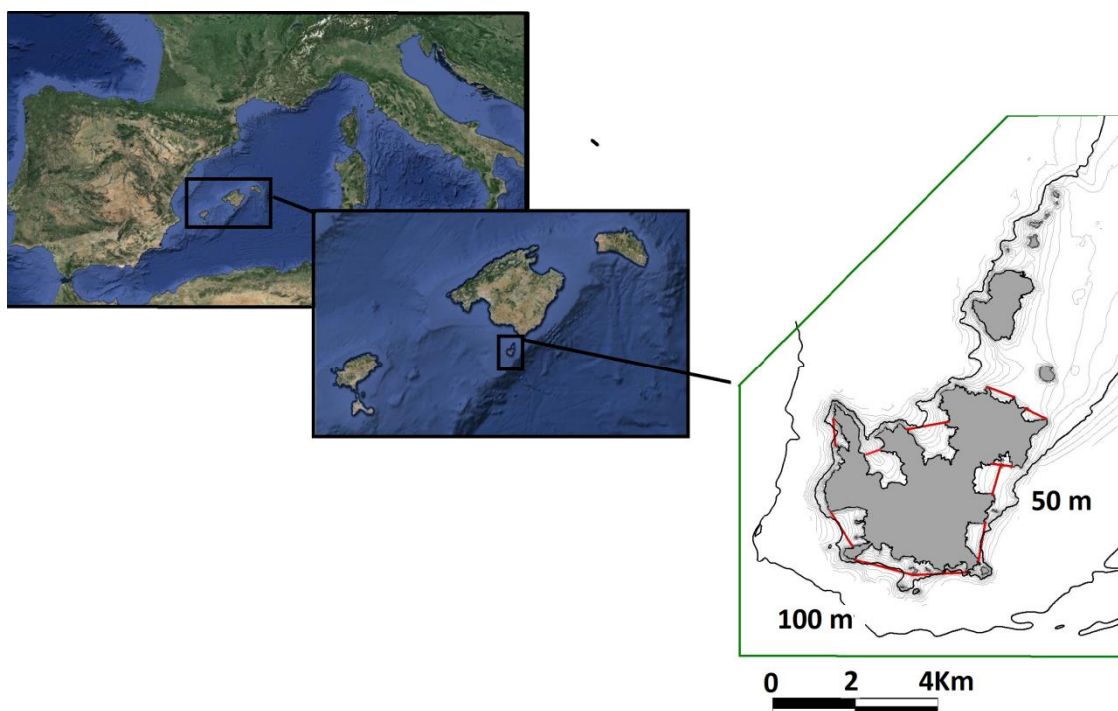


Figure 2.1 Location of CANP in Western Mediterranean. Red lines show limits of the *integral reserves* and *restricted use* areas where fishing is forbidden. Grey lines show a detailed bathymetry near the coast.

### 2.2.2 Sampling design, population and habitat data collection

A total of 301 Underwater Visual Census transects (UVC), covering a surface of 150 m<sup>2</sup> (5 m x 30 m), were conducted within the CANP in July 2008 over rocky bottoms from the surface to 40 m depth, covering the different coastal configuration and hydrodynamic regime conditions (Figure 2.2).

The location of every UVC transect was georeferenced by setting small displayable buoys at the beginning of each sampling transect. The coordinates of these buoys were recorded with a GPS. The UVC were carried out by two divers, one annotated the length of dusky grouper individuals while other took information on habitat characteristics within the transect. The length distribution of dusky grouper was split into two categories, juvenile and adults. The threshold length for the splitting was 50 cm, length at which female reach sexual maturity in the CANP (Reñones *et al.* 2012).

The habitat variables collected within the area covered during the underwater visual census (150 m<sup>2</sup>) provided information about rocky habitat structure and depth, and were denoted as micro-scale habitat variables as they hold information at the highest spatial definition in this study. The micro-scale habitat variables selected for this study were mean depth, percentages of flat rocky area and blocks, percentages of blocks at three different size categories (small, medium and big), percentage of rocky substrate (sum of flat rocky areas and boulders) and the roughness index of the rocky substrate (Ordines *et al.* 2005). Acronyms, definition and value ranges of all variables used in this study are provided in table 2.1.

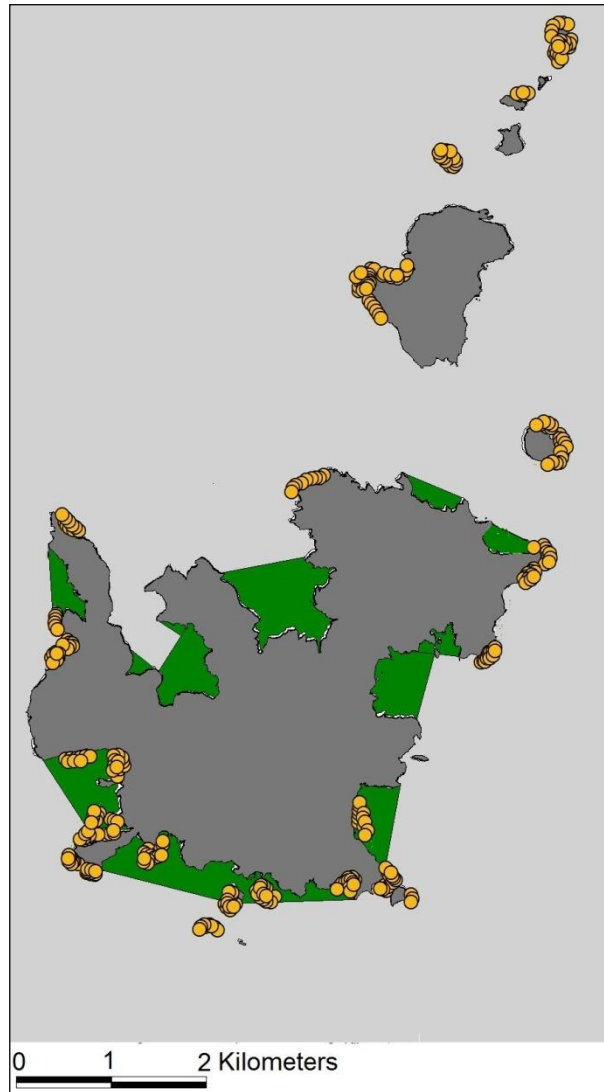


Figure 2.2. Distribution of underwater visual census samples around the Cabrera archipelago shown in yellow. No take areas where fishing is forbidden are shown in green.

Habitat characteristics at spatial scales wider than the micro-scale transect level were calculated applying traditional landscape ecology metrics, using marine habitat and bathymetric maps (Wedding *et al.* 2011). These variables (see table 2.1), denoted as zone seascape variables were calculated in an area around the location of underwater visual census. Calculation was processed at various spatial scales (see Figure 2.3).



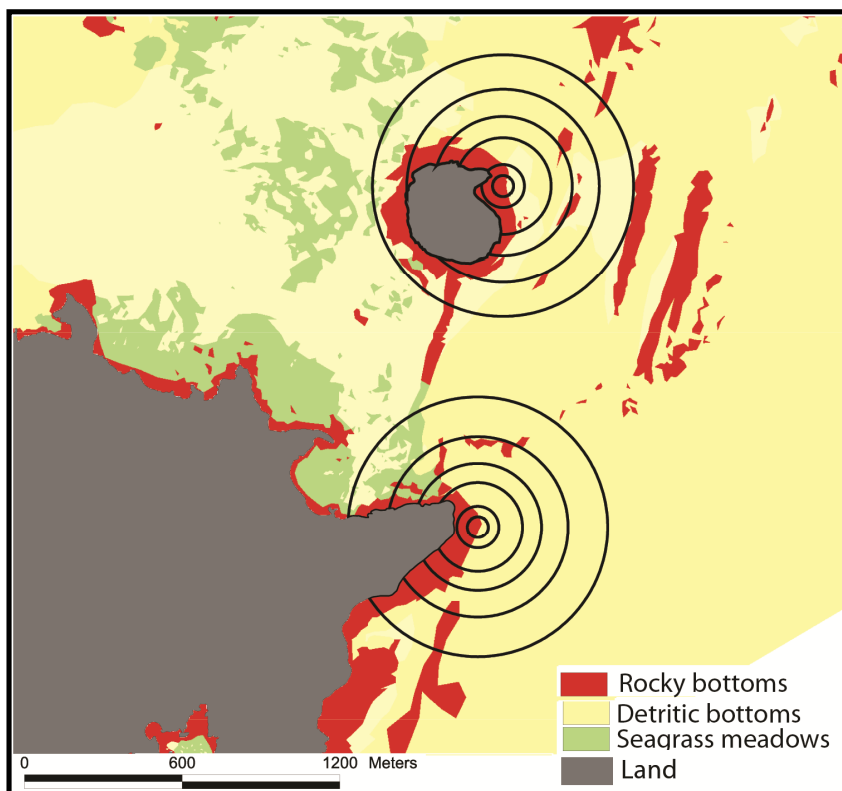


Figure 2.3: Buffers used for habitat zone seascape descriptors at various scales. At each scale areas within the buffer are the same at both locations. Examples centered at two different locations of underwater visual census sampling.

Around each UVC six buffers (denominated A,B,C,D,E,F) were calculated ranging from scale A=0.5 ha to scale F= 64 ha. These scales were calculated as different ratios of an area of 80 m radius, relative to the mean home range proposed for this species in previous studies developed in the Mediterranean (Lembo *et al.* 1999).

Mean depth (zDepth) and slope (zSlope) informed about the topography in the area. Number of patches (zNumP), total surface (zSrock) and mean roughness (ZrOUG), all of them referred to rocky bottoms, informed about the structure of habitat in the area. These metrics were processed with the Patch Analyst-grid 3.0 (Rempel *et al.* 2012). Original cartographies were provided by the local Government (<http://lifeposidonia.caib.es/user/carto/PDFs/CABRERA.pdf>).

Improvement of the cartographic topology in areas near the coast line were approached by adjusting the limits of the rocky bottoms and bathymetric isolines observed in the habitat maps to selected reference points identified *in situ* during the UVC. The adjustment of the original maps for matching these points was approached by nonlinear deformation of the

vector layers in specific areas using the rubbersheeting tool of the ArcGIS software (Environmental Systems Research Institute, Inc.) (see Figure 2.4).

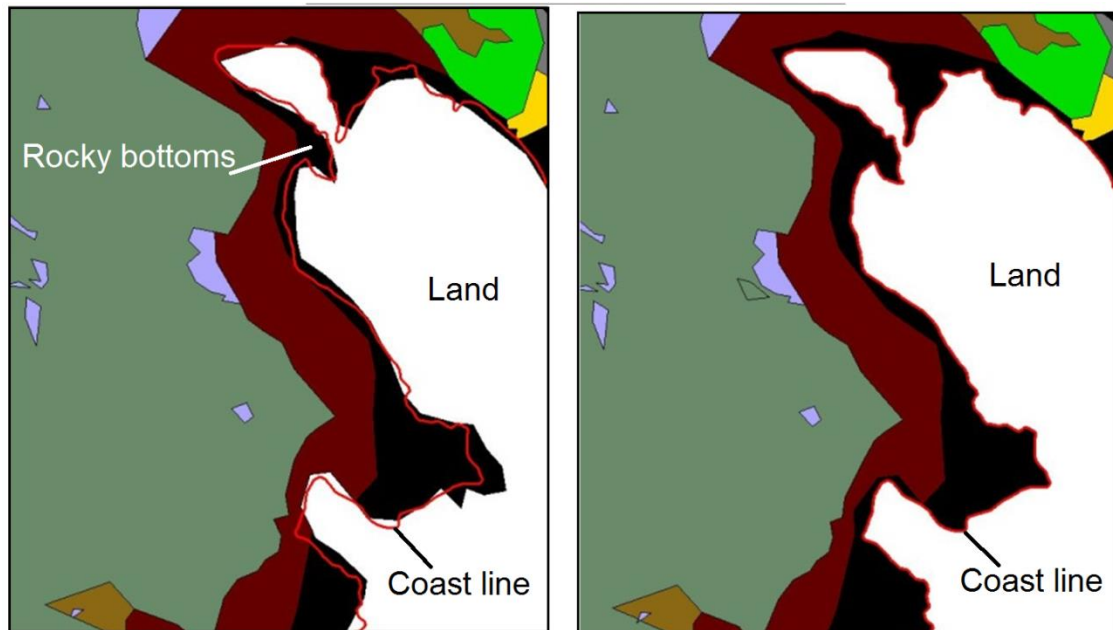


Figure 2.4: Improvement of map georeference after the nonlinear deformation of habitat maps. The red line indicates the reference coast line.

Standard GIS methods are prepared to create buffers of constant radius around the UVC, what derive in buffers of different area for one particular radius due to the proximity of land. Applying these types of algorithms strongly affects landscape metric calculations. To solve this limitation we applied a specific algorithm ensuring that buffers around all UVC at one particular scale had the same area, by adjusting the radius for each buffer. This algorithm to process buffers with equal area accounting for barriers (coastal lines) was provided by the ESRI software resources in avenue scripting language for Arcview 3.2. (<http://forums.esri.com/> last access 2015/05/05).

### 2.2.3 Coastal wave exposure characterization of the study area.

Coastal wind derived exposure is a relevant environmental factor affecting geomorphology and littoral benthic species ecology (Ekebom *et al.* 2003; Burrows *et al.* 2008), also in the CANP (Álvarez Berastegui *et al.* 2010). For evaluating if this parameter affects dusky grouper distribution, we considered the coastal exposure index. This parameter was calculated

following the EXA method (EXposure estimates for fragmented Archipelagos) (Ekebom *et al.* 2003), that informs about the average wave power in one particular location of the coastal area, and is mainly based on the mean fetch over a time period. Wave exposures were standardized from 0 to 100 within the archipelago.

Table 2.1: List of habitat descriptors at micro-scale (A) and at zone level (b), hydrodynamic descriptor (c) and fishing effort descriptor (D).

<b>A-Microscale variables, collected during UVC.</b>		
<b>Variable name</b>	<b>Definition</b>	<b>Variable range, units</b>
mDepth	Mean depth in the transect	-3: -38 (meters)
pmSrock	Percentage of rock substrate	35:100 (%)
pmflatrock	Percentage of flat rocky bottoms	0:100 (%)
pmAllblocks	Percentage of rock blocks (accounting for all sizes)	0:100 (%)
pmBblocks	Percentage of big size blocks (>2)	0:90 (%)
pmMblocks	Percentage of medium size blocks (>1,<2)	0:60 (%)
pmSblocks	Percentage of small size blocks (<1m)	0:90 (%)
mRoug	Roughness index	1:3 (non dimensional)
<b>B- Zone seascape variables, measured from habitat maps at 6 different scales</b>		
zDepth	Mean depth in the area	-2:-40 (meters)
zSlope	Mean slope in the area	4:40 (degrees)
zNumP	Number of rocky patches in the area	6:46 (n patches)
zSrock	Surface of rocky bottom in the area	0.52-8.1 (ha)
zRoug	Mean roughness of rocky bottoms in the area	15:40 (non dimensional)
<b>C- Hydrodynamic variables</b>		
Exp	Coastal exposure index	0:100 (%)
<b>D- Fishing effort</b>		
Feffort	Spatial distribution of bottom longline effort	0:2736 (n hooks/cell)

## 2.2.4 Spatial distribution of fishing effort within the MPA

The fishing effort of bottom long lines intended for groupers was estimated accounting for the number of hooks deployed within cells of 250 meters side, along two years of fisheries sampling (2003-2004). The fishing effort data set (see Figure 2.5) was developed in the

framework of the BIOMEX project (QLRT-2001-0891) in the study area (See Goñi *et al.* 2008 and Planes 2005 for details).

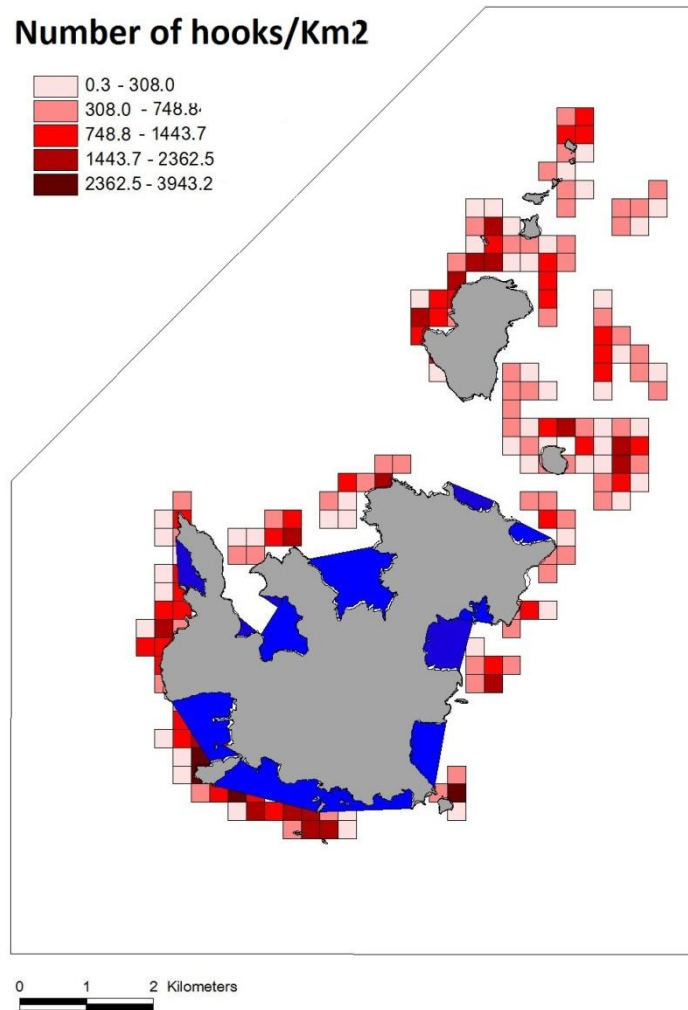


Figure 2.5: Spatial distribution of fishing effort of bottom long lines directed to dusky grouper in the Cabrera archipelago during 2003-2004.

### 2.2.5 Identification of essential habitats from micro-scale habitat variables.

The identification of the relevant micro-scale habitat variables shaping the essential habitats of dusky grouper was assessed using general additive models (Wood 2006). This modeling approach has the advantage of allowing the exploration of non linear relationships between the response variable and the explanatory variable. To avoid over fitting, the number of knots associated to each covariate was limited to 3. Adult and juvenile densities (number of

individuals per 150 m<sup>2</sup>) within the underwater visual census transects were used as response variable in separate analyses for the juvenile and adult individuals. Multicollinearity of the explanatory variables was assessed using the variance inflation factor, VIF (Zuur *et al.* 2010). The VIF values were calculated with all variables at micro-scale level and the variable with the highest VIF value was excluded from the data set. Then the VIF was recalculated. This process was run till all variables showed a VIF value below 3. Then, correlation among pairs of remaining variables was tested. When two variables showed correlation values above or equal to 0.60 one of them was removed, attending to their ecological meaning. With the remaining variables a backward selection process based on AIC values was applied for retaining only significant variables ( $p < 0.05$ ).

### **2.2.6 Identification of essential habitats from zone seascape variables.**

We identified which of the six scales used for calculation of the zone seascape variables provided more information for defining essential habitats of juvenile and adults. The seascape metrics provided information about topographic characteristics (zSlope and zDepth), and the habitat structure (NumP, zSrock, ZRoug) in the buffer area. The scales that provide the best information may be different for these two different set of variables. Therefore we computed two groups of GAMs at each scale, one with the topographic metrics and the other with the habitat structure metrics. The effect of the spatial scale on the model performance was assessed with scalograms showing how AIC values of computed models (used as model performance indicator) vary along scales. The best model at zone level was configured with the combination of the topographic and habitat structure variables at the spatial scales that minimized model AIC in the scalograms. The model configuration and variable selection process was similar to that applied in the previous section. Interaction terms were considered for the number of patches (NumP) and total area of rocky bottoms (zSrock) variables. The interaction of these two variables provides additional information about the structure of the rocky habitat fragmentation at zone level. AICs were used as criteria for including these variables as interactive or additive terms.

### **2.2.7 Exploration of essential habitats at multiscale level.**

The effect of habitat at multiscale level was approached by combining the significant micro-scale habitat variables and zone seascape variables, together with coastal exposure and fishing effort variables. Variable selection and modeling approach followed similar process

than in previous sections. Variance inflation factors were used for multi collinearity exploration and correlation among pairs were tested. Effects on population were assessed with GAMs. This analysis allowed obtaining a multiscale seascape model for juveniles and adults separately providing information on how different environmental variables define dusky grouper essential habitat.

## 2.3 Results.

The results obtained from the analyses performed at the different spatial scales are presented in table 2.2, showing the habitat variables excluded due to VIF values over 3 or correlations over 0.6, the variables excluded during the backward selection approach, the variables selected for the GAMs and the explained deviances for each model. Correlations among all variables considered in each of the different analyses are provided as supporting information (see Tables S2.1-S2.5 at the end of this chapter)

### 2.3.1 Analyses of the habitat at the micro-scale level.

Five micro-scale habitat variables were included in the model for juveniles, mean depth, total rocky area surface, roughness index and percentages of medium and small size blocks. The percentage of total rocky blocks, flat rocky areas and big size blocks were excluded in the VIF and correlation analysis. Response plots showed a negative effect of depth (mDepth) on the density of juveniles (Figure 2.6) and positive effect of small and medium size blocks as well as the roughness index (pmSblocks, pmMblocks, mRoug respectively).

Only depth and roughness index (mDepth and mRoug) were retained in the model for adults. Depth had a positive effect on densities which significantly increased till 25 meters depth (Figure 2.7). From 25 meters down to 40 meters, adult densities kept constant and over the mean. The Roughness index also showed a positive effect, but only substrata with high roughness (value 3) presented densities over the mean.

Table 2.2: Environmental covariates selection process (VIF analyses and non significant variables identification), descriptors included in the models and model performance.(N=301).

Scale	Response variable	Excluded, 1: VIF>3 or 2: R <sup>2</sup> >=0.60	Non significant	included in the model (p<=0.05)	% deviance	AIC
<b>Micro-scale</b>	Number of juveniles	PmAllblocks(1) Pmflatrock(1) pmBblocks(2)	-	mDepth ; pmSrock pmSblocks ; pmMblocks; mRoug	21.0	930.7
	Number of adult	PmAllblocks(1) Pmflatrock(1) pmBblocks(2)	pmSrock pmSblock pmMblock	mDepth; mRoug	22.2	731.8
<b>zone</b>	Number of juveniles	-	zSlope.A zRUG.C	zDepth.A (zNUMP.C-zSrock.C)*	16.7	948.8
	Number of adult	-	zRUG.F	zDepth.A; zSlope.A (zNUMP.F-zSrock.F)*	20.3	752.9
<b>Multi scale</b>	Number of juvenile	zDepth.A(1)	zSrock.D Feffort	mRoug; mDepth ; Exp zNUMP.D; pmSrock; pmSblocks ; pmMblocks;	24.0	921.7
	Number of adult	zDepth.A(1) ;Exp(1)	Feffort	mDepth mRoug; zSlope.A (zNUMP.F-zSrock.F)*	29.5	711.8

\*Habitat structure variables included in the model as interaction terms.

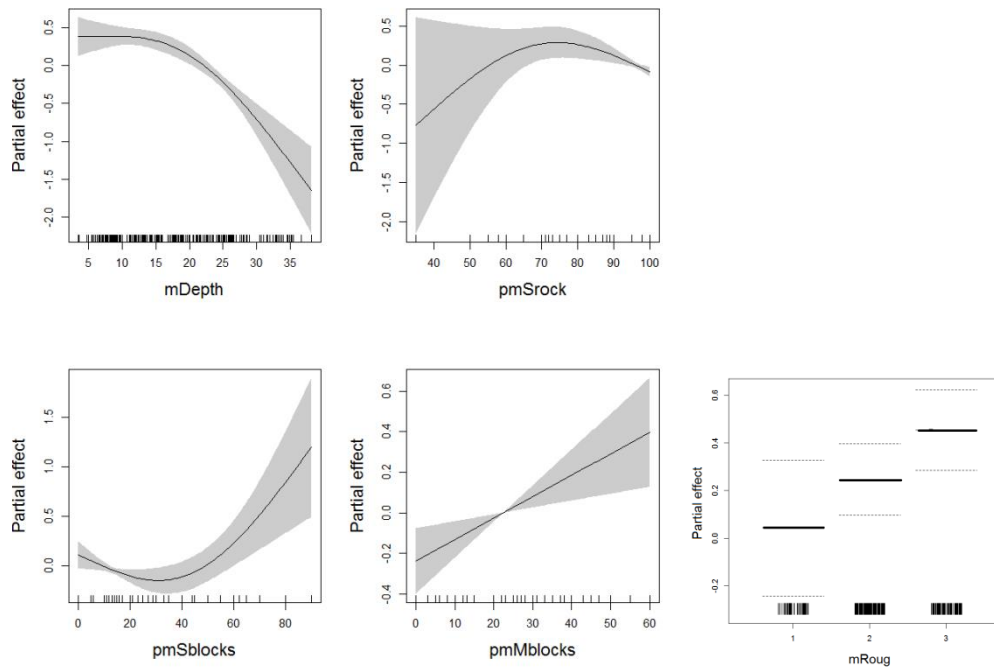


Figure 2.6: Response plots of the micro-scale habitat descriptors for juvenile individuals.

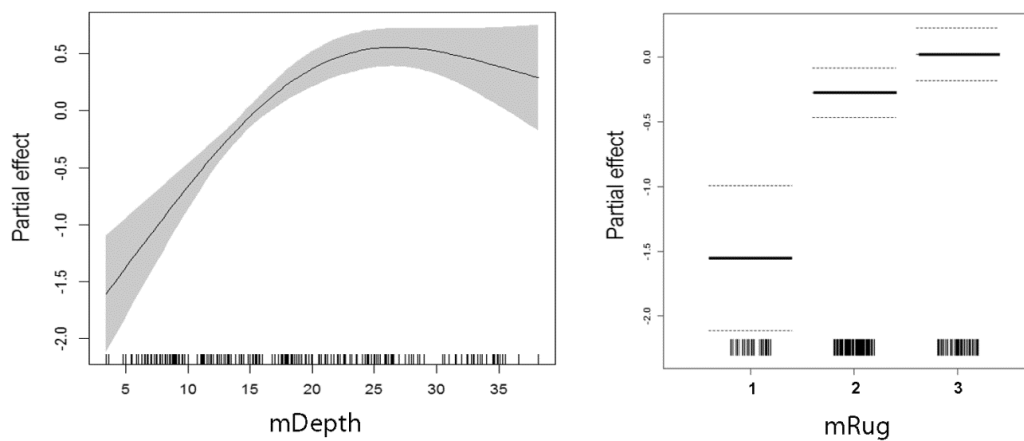


Figure 2.7: Response plots of the micro-scale habitat descriptors for adult individuals.



### 2.3.2 Analyses of the habitat at the zone seascape level.

Spatial scale at which zone seascape metrics were calculated affects to the performance of models. The scalograms processed with the topographic and habitat structure variables are presented in Figure 2.8. Trends of AIC values (the lower the better) for the topographic variables (zDepth and zSlope) showed a decreasing model performance when the scales of observation becomes higher, for both adult (Figure 2.8.A) and juvenile individuals (Figure 2.8.B). The highest spatial definition the better modeling result. The scalogram showing the results of the models with the habitat structure variables (NUMP, zSrock and zRUG) showed an opposite pattern. Model performance improved when scales increases, reaching lower AICs at scale D for juvenile and scale E-F for the adults (Figure 2.8.C and Figure 2.8.D respectively).

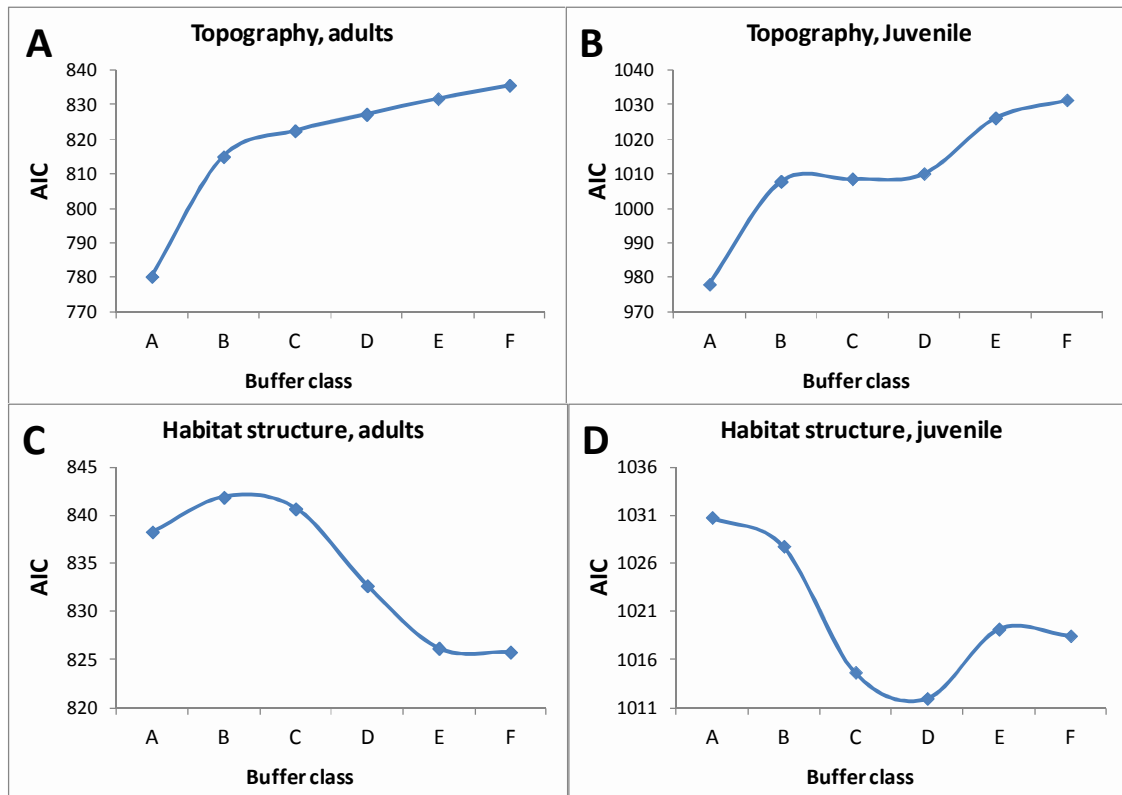


Figure 2.8: Density models performance along zone scales, habitat structure and topographic variables

The best model resulting from the combination of the topographic and habitat structure zone seascape variables (see table 2.2) retained the following variables: mean depth in the buffer A (zDepth.A), number of patches (zNUMP.D) and total rocky area in buffer D (zSrock.D). The model considering the interaction term of these variables (zNumP.D~ zSrock.D) presented lower AIC (better model performance) than the model considering the additive terms. The response plots of the model show a decrease of juveniles with depth, being constant in surface

waters above the 15 m (Figure 2.9.A). The plot of the interaction between NumP.D and zSrock.D shows lower juvenile densities in areas characterized by low total rocky surface and with small number of rocky patches (Figure 2.9.B, blue area). Higher densities appear when surface of rocky areas or rocky patchiness is high

When analyzing the adult population the selected variables were mean depth and mean slope in the buffer A (zDepth.A and zSlope.A), and the interaction term between number of patches and total rock surface at scale F (zNUMP.F~zSrock.F). Response plots of zDepth.A (Figure 2.10.A) show a positive effect of bathymetry and densities increase from surface down to 20 meters depth, from which densities remain constant. The effect of slope of the bottom substrata around the transect (zSlope.A) is positive and linear along all the range (Figure 2.10.B). The structure of rocky habitats measured in the biggest buffer (buffer F) show higher densities in areas where the total rocky bottoms is lower than 13ha, and where number of patches is at intermediate values.

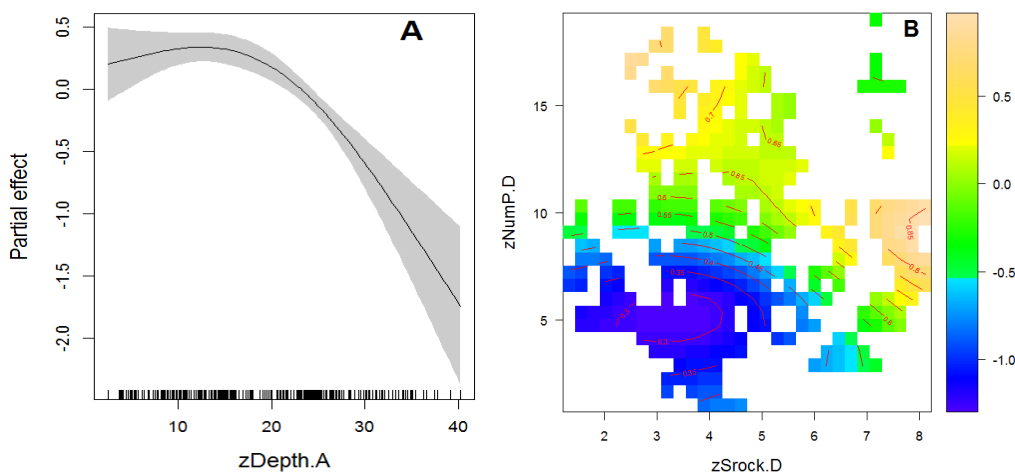


Figure 2.9: response plots of zone seascape descriptors for juvenile individuals. A: Partial effect of depth at zonal level in the buffer “A”; B: Effect of the interaction term

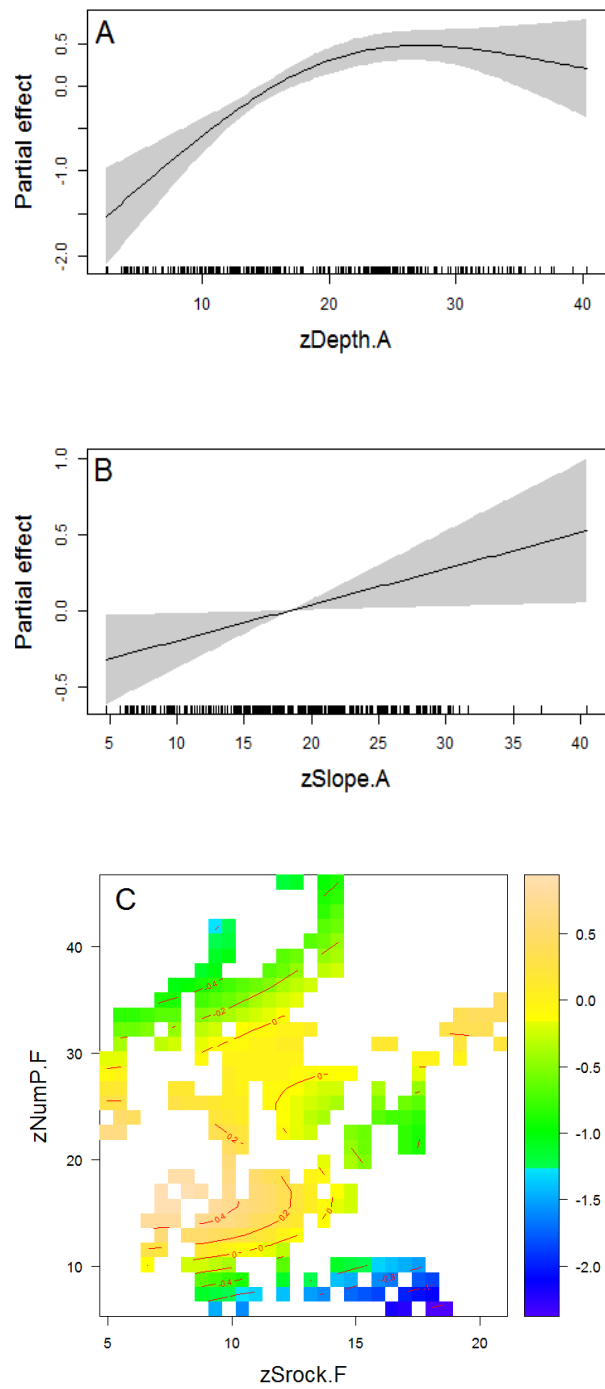


Figure 2.10: response plots of zone seascape descriptors for adult individuals

### 2.3.3 Multiscale seascape analysis of the habitat

The multi scale analyses for juveniles showed that the best model retained seven variables in total, five variables at micro-scale scale, one at zone level (number of patches at scale D, zNump.D) and the coastal exposure index (Exp)(see Table 2.2). The model considering interaction terms presented higher AICs (lower model performance) and therefore the number of patches was included as additive variable. Response plots (see Figure 2.11) confirm variable effects found previously during the analysis at micro-scale level: a negative effect of depth and a positive effect of rocky area in the transect and percentages of small and medium size blocks, and show the importance of rocky habitat fragmentation around the transect (zNumP.D). Wave exposure showed a negative effect and fishing effort was showed no effect on juvenile density distribution.

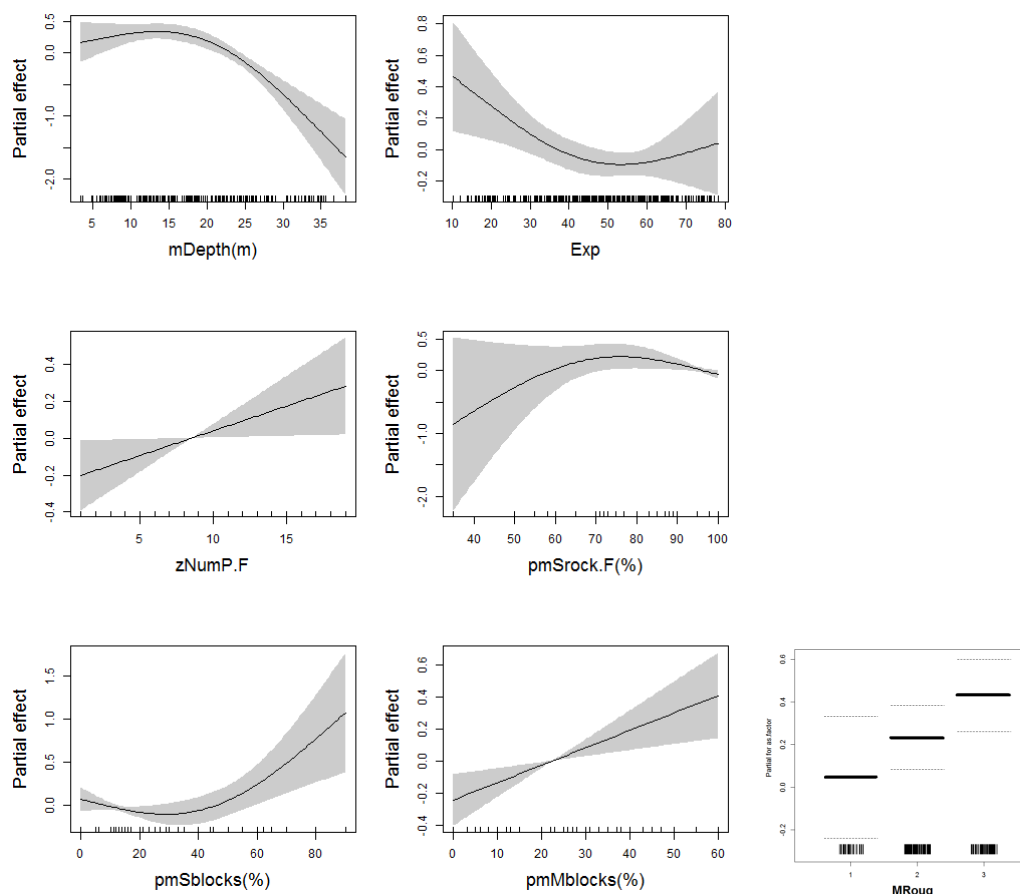


Figure 2.11: response plots of multiscale analysis for juvenile individuals

The GAM fitted for the adults retained five variables, three related to seascape habitat structure at two spatial scales (zSlope.A,zNumpF,zSrock.F), and two related to the habitat at

micro-scale level (mDepth,mRoug). The coastal exposure index was excluded in the VIF analysis and fishing effort was excluded due to low significance ( $p>0.05$ ). Response plots (Figure 2.12) showed similar responses than the model developed previously at micro-scale and zone levels, indicating the capability of habitat descriptors at different scales for providing complementary sources of information.

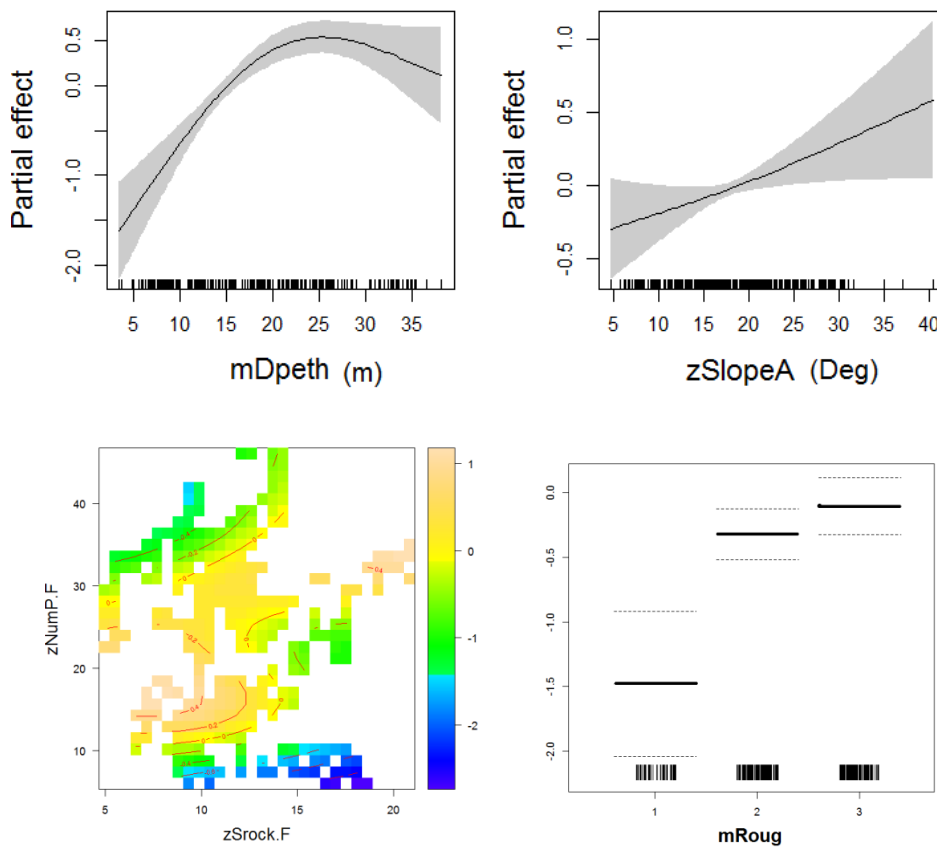


Figure 2.12: response plots of multiscale analysis for adult individuals

## 2.4 Discussion

Multiscale seascape analyses improved the definition and delineation of dusky grouper essential habitats compared to the analysis performed at single scales, and allowed the identification of the main differences in habitat requirements between juveniles and adults. The strong effect of depth in shaping the spatial distribution of both stages was identified when analyzing habitats at both the micro-scale and the zonal level, but all other habitat descriptors were only identified at specific scales, demonstrating the need of combining information at different scales to define appropriately the dusky grouper essential habitats. The results show that the spatial scales that best define the essential habitats for each developmental stage are different. Essential habitats for juveniles were driven mainly by variables collected at the micro-scale level. Coastal exposure and variables at zone level provided additional information on juvenile essential habitat requirements improving the deviance explained and AIC of multiscale habitat models against single scale models. For adults, both micro-scale and zonal models performed similarly and provided complementary information to the multiscale analyses that show a relevant improvement when compared to the single scale analysis. Differences on spatial scales that best defined adult and juvenile essential habitats were also identified in the scalogram analyses; the best buffer size to characterize adult rocky habitat structures was 4 times bigger than the buffer for juveniles.

The strong effect of depth and rock complexity at the scale of visual census samplings on abundance and biomass of dusky grouper is well known from previous studies (Harmelin & Harmelin-Vivien 1999; La Mesa *et al.* 2002; Machado *et al.* 2003; Vacchi *et al.* 2007). Here we found juvenile and adult individuals all along the depth range analyzed (from 3 to 38 m), but general additive models allowed identifying non linear responses of the density distribution with depth for both developmental stages, showing a counter distribution along bathymetry. Juvenile individuals were homogeneously distributed from the surface til 15 meters depth from where densities drop linearly to 40 meters depth. The abundance of adults increased linearly from the surface to 22 m depth, remaining constant between 22-40 m depth.

Beyond the effect of depth, the multiscale analyses showed how the rocky structure at the small scale strongly affects distribution of juvenile individuals, that are associated to areas with high level of substrate roughness generated by high percentages of small and medium block sizes. The combination of this information together with the effect of coastal exposure, affecting negatively to juvenile densities, and the positive effect of the number of rocky patches around an area of 15 ha, offer a wide picture of dusky grouper juvenile habitat

requirements. The ecological traits behind these requirements may be influenced by a number of factors both from top-down and bottom-up control processes. In general, metabolic rates of juveniles fishes are higher than adults in order to maximize growth, therefore the higher temperatures in the shallow depths, where juveniles are found, compared to the bottom depths may be energetically adequate. Besides, in the Cabrera archipelago small to medium shelters associated to small rocky blocks, are more frequent in shallow areas right near the coast line than in deeper waters, ensuring refuge from other predators. On the other hand, feeding strategies of young individuals of *E.marginatus* differ from that in adults in the size of preys (Reñones 2002, Linde 2004), and depth strongly shapes size distributions of preys. Higher number of rocky patches at zone level may ensure a secure area for hunting while avoiding predation. Therefore, metabolic needs, predator avoidance and differences on feeding strategies between adults and juveniles may explain the responses found in the analysis.

The preference of adults for deep areas with cliffs and complex rocky substrates has been found in other studies (Zabala *et al.* 1997; Harmelin & Harmelin-Vivien 1999). As for juveniles, our non linear analyses allowed identifying the best habitat variables to describe the dependency with depth and exploring specific responses. Densities of adults increased linearly down to 20 m depth, from where densities remain constant, just the opposite to the observed trend in juveniles. While depth and rocky habitat complexity variables were better defined at micro-scale level for adults, preference for intermediate slopes was only identified from analysis at zone level (at scale of 0.5 ha). The analysis at the zone spatial scale (66 ha) showed that the adults prefer locations surrounded by a fragmented rocky habitat of patches at intermediate size (considering the patch size distribution in Cabrera archipelago). In contrast, low densities were associated to areas surrounded by a high number of small size patches or extensive continuous areas of rock with low number of patches. In the Cabrera archipelago, the combination of the habitat characteristics positively influencing the distribution of adults are mainly associated to the bottoms around the main capes and small islets. An interesting question emerge from this association since wave exposure are important in these areas (Álvarez Berastegui *et al.* 2010). However, the coastal exposure index was not retained in the multiscale model for adults in our study as it was excluded in the VIF analysis due to colinearity with depth, slope and number of patches.

Studies in other MPAs in the Balearic Islands found that steep slope rocky bottoms with high coastal exposures are hotspots for rocky fish biomass (Coll *et al.* 2013b). In their work, these authors proposed ecological mechanisms driving biomass concentration around these

areas associated to the effect of the hydrodynamics and the presence of deep waters, generating higher levels of primary, secondary production and food availability for top predators. Besides, the effect of the coastal hydrodynamics on the spatial distribution of adults may ensure larval dispersal during reproduction, a relevant mechanism that has been proposed as driving the location of adults (Zabala *et al.* 1997). Therefore, coastal exposure index very likely have a positive effect on densities of adult dusky groupers. This relation could not be identified with the analysis here performed due to correlation among environmental covariates. This question could be of interest when designing MPAs or when redefining new zoning in existing ones. Further analysis from the data set here applied, blocking effects of correlated covariates could provide insights of the coastal exposure and coastal hydrodynamics on the ecology of dusky grouper.

When analyzing the areas of the buffers that best explained the habitat characteristics at zone level, one may tend to associate these buffer areas with the species home range. Studies from acoustic telemetry have shown that dusky grouper is a highly resident species (Lembo *et al.* 1999), but mean home ranges around 1.3 km (=530 ha) (Afonso *et al.* 2011), is over eight times the area of the buffer “F” (66ha) used to characterize adult dusky groupers habitat. Besides, individuals may perform long displacements associated to various causes, as colonization of better vacant habitats derived from the effect fishing or natural mortality (Chauvet & Francour 1989), changes on habitat requirements along ontogeny (Afonso *et al.* 2011) or small displacements during spawning seasons (Zabala *et al.* 1997; Koeck *et al.* 2014). These movements occur at higher spatial scales than the 66 ha of the buffer. The buffer size here identified for adults and juveniles inform about the minimum area necessary to identify rocky habitat structure patterns defining essential habitats at both stages for this species, but not home ranges. Nevertheless, a relation between how individuals explore surrounding habitats at different developmental stages may affect both at home ranges and buffer sizes characterizing essential habitats. For example, the lower effect of predation on larger individuals may allow them exploiting more diverse habitats than smaller individuals, and therefore habitat structure would be only identified at bigger buffer sizes, independently that juvenile may perform longer displacements associated to ontogenetic changes.

The results obtained in this study in relation to the buffer sizes that better defined habitat structure are specific for the Cabrera archipelago. This marine protected area show a particular coastal rocky bottoms habitat patterns, where capes and bays alternate providing alternated essential habitats for juvenile and adult stages. Other coastal geomorphologies may present other rocky habitat structure patterns that could be better defined from buffers at different



spatial scales. So this “characteristic buffer size” that better define habitat structure could be species-stage and site-specific. Evaluating how characteristic buffer sizes changes among MPAs where populations of dusky groupers are well developed could provide a relevant step forward in seascape ecology. Especially interesting for this purpose would be to assess those questions in MPAs with integral reserves where rocky bottoms would be more extensive and more continuous as they are in the Cabrera archipelago, and where core areas of highly complex rocky patches would be bigger. We believe that even “characteristic buffer size” may be different at different geographies; habitat patterns defining essential habitats may remain similar as they are in concordance with ecological requirements of the species at different stages.

Dusky grouper has been classified as endangered species by International Union for Conservation of Nature (IUCN) (<http://www.iucnredlist.org/details/7859/0>) and definition of dusky grouper essential habitats is a key information for multiple management purposes. In the Mediterranean Sea, this species responds well to protection (Reñones *et al.* 1999; Guidetti & Sala 2007; Harmelin *et al.* 2007) and MPAs are probably the best management option. Nevertheless marine zoning (spatial distribution of uses and activities within an MPA) is a key issue. The particular design of the no take zones in relation to the habitats they protect will determine whether an MPA will succeed or not (Roberts *et al.* 2003b). The results obtained in this study show a spatial segregation of this species due to differences in habitat requirements. Managers should not consider only the necessity of protecting rocky habitats in coastal MPAs, but also consider the structure of those habitats and their topographic characteristics. Considering that a well designed MPA should ensure protection of key ecosystem species at all stages and enclose an area bigger than the species home range, a proper design of an MPA in the Mediterranean should be designed to protect i) rocky habitats from surface down to 50 meter, ii) extensive areas of shallow rocky bottoms above 20 m, ensuring representing depths of maximum juvenile density, with low-intermediate coastal exposures and with high complexity derived from the accumulation of small and medium rocky blocks, iii) Areas of cliffs down to 30 m with intermediate slopes, ensuring depth of maximum adult densities, with important rocky falls providing a high level of habitat complexity from big size blocks, surrounded by sea beds of fragmented habitats and with relevant coastal hydrodynamics. The differences found in the essential habitat definition for juveniles and adults are derived from changes in the ecological traits along the dusky grouper development. Bottom up ecological traits, such as food requirements may be related to the preferences for cliffs with intermediate slopes with high hydrodynamic activities on adults, while top down ecological traits such as

predation on smaller individuals, could drive requirements of small and medium block presence for juvenile stages. Depth also shapes the population distribution and individuals colonize deeper areas as they grow. Due to the gradual evolution on habitat requirements along the dusky grouper ontogenetic development, an appropriate MPA design would require good connectivity among the different areas specified before. An extensive no take area covering essential habitats for both stages would be the ideal design. These findings may be of interest to set the scientific basis for MPA design.

Information of essential habitats in littoral ecosystems at a proper scale also allows predicting ecosystem carrying capacity and the time necessary to recover fish populations (Coll *et al.* 2013a; García-Rubies *et al.* 2013). Such predictions could be a key information for coastal resources assessment serving for reference base lines for the Good Environmental Status evaluations within the Marine Strategy Framework Directive (MSFD), (EU 2008). Therefore, an appropriate definition of dusky grouper essential habitat could provide the basis to evaluate carrying capacity of littoral ecosystems for this particular species.

Finally, the study of MPA specific responses to protection, and the differences among areas of one single MPA, have been partially explained by differences in the habitat features within areas of 125 m<sup>2</sup> to 250 m<sup>2</sup> (the area of one sampling unit collected in visual census techniques), while a high percentage of deviance remain unexplained (García-Charton *et al.* 2004; Coll *et al.* 2012; Sala *et al.* 2012). The results obtained here shows the relevance of describing the habitat at wider spatial scales than the micro-scale. Including seascape metrics at appropriate spatial scales in studies directed to understand which factors affects protection success could provide new insights into the MPA functioning.

## 2.5 Acknowledgements.

Main founding for this study was provided by the Spanish Ministry of Science (project EPIMHAR, grant number 012/2007), part of the work was supported by the Agriculture and Fisheries Department of the Balearic Islands Government (grant number: BOIB Num. 128-Resolució Num. 16679). I would like to thank the people that participate in the EPIMHAR project during the underwater visual census and the geographic data processing as well as the supervisors of my research grant: Olga Reñones, Josep Coll, Lucia Rueda, Gabriel Morey, Oliver Navarro, Ben Stobart, Alberto Aparicio, Antoni Grau and Raquel Goñi.

**Supporting information, tables S2.1-S2.5:** Correlations among environmental variables. In bold, correlations equal or higher to (+,-) 0.60

Table S2.1: Pearson correlations among selected variables for analysis of juvenile and adult individuals at micro-scale level

	mDepth	mRoug	Pmflatrock	pmAllblocks	PmSblocks	PmMblocks	PmBblocks	PmSrock
mDepth	1,00							
mRoug	0,01	1,00						
Pmflatrock	0,08	-0,58	1,00					
pmAllblocks	-0,10	<b>0,67</b>	-0,9	1,00				
PmSblocks	-0,02	-0,01	-0,30	0,32	1,00			
PmMblocks	-0,13	0,35	-0,54	<b>0,60</b>	0,05	1,00		
PmBblocks	-0,02	<b>0,60</b>	-0,51	0,60	-0,41	0,04	1,00	
PmSrock	-0,10	0,27	0,12	0,33	0,07	0,17	0,22	1,00

Table S2.2: Pearson correlations among selected variables for analysis of juvenile individuals at zone level

	zDepth.A	zSlope.A	zNUMP.D	zSrock.D	zRug.D
zDepth.A	1,00				
zSlope.A	0,36	1,00			
zNUMP.D	-0,14	0,02	1,00		
zSrock.D	-0,36	-0,29	0,02	1,00	
zRug.D	-0,11	-0,24	0,03	-0,23	1,00

Table S2.3: Pearson correlations among selected variables for analysis of adult individuals at zone level

	zDepth.a	zSlope.a	zNUMP.h	zSrock.h	zRug.h
zDepth.A	1,00				
zSlope.A	0,36	1,00			
zNUMP.F	-0,07	0,14	1,00		
zSrock.F	-0,21	-0,29	-0,01	1,00	
Rug.F	-0,04	-0,24	-0,23	0,06	1,00

Table S2.4: Pearson correlations among selected variables for analysis of juvenile individuals at multiscale level

	zDepth.A	zNUMP.E	mDepth	mRoug	PmSrock	PmSblocks	PmMblocks	PmBblocks	Exp	Feffort
<b>zDepth.A</b>	1,00									
<b>zNUMP.E</b>	-0,14	1,00								
<b>mDepth</b>	<b>0,93</b>	-0,12	1,00							
<b>mRoug</b>	0,04	-0,02	0,01	1,00						
<b>PmSrock</b>	-0,09	0,02	-0,10	0,27	1,00					
<b>PmSblocks</b>	-0,03	-0,02	-0,02	-0,01	0,07	1,00				
<b>PmMblocks</b>	-0,14	-0,04	-0,13	0,35	0,17	0,05	1,00			
<b>PmBblocks</b>	0,01	-0,03	-0,02	<b>0,60</b>	0,22	-0,41	0,04	1,00		
<b>Exp</b>	0,56	-0,14	0,54	0,04	0,14	0,07	-0,06	0,00	1,00	
<b>Feffort</b>	0,19	-0,11	0,18	-0,12	0,19	0,08	-0,17	-0,08	0,52	1,00

Table S2.5: Pearson correlations among selected variables for analysis of adult individuals at multiscale level

	zDepth.A	zSlope.A	zNUMP.F	zSrock.F	mDepth	mRoug	Exp	Feffort
<b>zDepth.A</b>	1,00							
<b>zSlope.A</b>	0,36	1,00						
<b>zNUMP.F</b>	-0,07	0,14	1,00					
<b>zSrock.F</b>	-0,21	-0,29	-0,01	1,00				
<b>mDepth</b>	<b>0,93</b>	0,26	-0,10	-0,23	1,00			
<b>mRoug</b>	0,04	0,16	-0,06	-0,05	0,01	1,00		
<b>Exp</b>	0,56	0,31	-0,31	-0,55	0,54	0,04	1,00	
<b>Feffort</b>	0,19	0,34	0,18	-0,43	0,18	-0,12	0,52	1,00

# **CHAPTER 3**

**Multidisciplinary rapid assessment of coastal areas as a tool for the design and management of marine protected areas**

## 3.1 Introduction

The management of coastal areas must integrate information from a multi-disciplinary perspective, including governance, human uses and activities, and ecological quality or status of habitats and their spatial distribution (Agardy, 2010; Barragán Muñoz, 1997; Pomeroy et al., 2004; Post et al., 1996). Related cartographic products are main tools used to support ecosystem based coastal zone management (Agardy, 2010; Nobre and Ferreira, 2009). Getting these maps can be challenging, however. The data collection for mapping parameters related to benthic habitats (Kenny et al., 2003; Sheehan et al., 2010), spatial distribution of human activities (Balaguer et al., 2011; Halpern et al., 2009), water quality or bio-physical indicators (See Pomeroy et al., 2004, for a review of indicators), may need specific, sometime costly, equipment and time-demanding post processing procedures. Additionally, field work in coastal areas is highly dependent on the sea conditions. These constraints must be considered when planning time and budget for data collection prior to developing a management plan for a coastal area.

The collection of information in the field when time and budget is limited is what led to the emergence of Rapid Assessment Programs or RAPs (Alonso et al., 2011). These programs, originally designed for the conservation of the natural environment, formally came into being at the end of the 1980s with two different objectives in mind: 1) to quickly provide the minimum biological information needed to bring forward conservation proposals and actions; and 2) to improve biodiversity conservation in these areas through management measures implemented as a result of the assessments.

So far, most RAPs programs in the marine environment focus on tropical environments (DeVantier et al., 1998; Dutra et al., 2006; Kramer, 2003; McKenna, 2011; Wilkinson et al., 2006). There are very few examples of applications in the Mediterranean, mainly related to surveys of the biodiversity of Greek coastal lagoons (Arvanitidis et al., 2005), the monitoring of the Albanian coast for the loggerhead turtle and the monk seal (White et al., 2005), and the evaluation of oil spill effects in the sea during the 2006 Israeli-Lebanese conflict (Steiner, 2006). In this geographical area, considered a biodiversity hot spot at a global level (Myers et al., 2000), Marine Protected Areas (MPAs) are an important management approach for conservation (Garcia-Charton et al., 2008).

The percentage of Mediterranean littoral zone that has been scored from a comprehensive ecological perspective with conservation in mind is rather low, further complicating proper MPA design and implementation. Most of what has been already

surveyed is along the northern coast, particularly the sector located in the most developed countries, and in particular the European countries with a coastline affected by the obligations of Habitats Directive 49/92 in relation with the Natura 2000 network. Although rather complete in ecological terms, the selection of N2000 sites is based exclusively on habitats and gaps in knowledge about uses and impacts in the marine arena are still pervasive. Total MPA coverage is still minute in the Mediterranean Basin (De Juan et al., 2012) and basic knowledge of huge stretches of littoral from North Africa and the Eastern Mediterranean is still rather poor. Considering the present day situation in the context of global environmental initiatives, Objective 11 of the Nagoya conference –which encourages the Parties of the Convention on Biological Diversity to take actions in order to declare at least 10% of their territorial waters as MPAs, is far from being reached.

Developing a RAP framework for Mediterranean coastal areas would be of interest for coastal planners and management agencies aiming to design MPAs, especially in areas where the existing information is poor and/or the resources or time available may restrict more expensive and time demanding sampling programs. Even in places where scientific information exists, efforts must be made in the development of comprehensive tools for connecting science and decision makers (Gregg and Chan, 2011), especially in complex Social Ecological Systems (SES) such as those that exist in most sites where MPAs are or will be designated (Pollnac et al., 2010).

Although MPAs are best designed by thoroughly assessing community perceptions, socio-economics, politics, and institutional weaknesses and strengths, a cost-effective RAP can provide the foundation for selecting target sites around which to begin such participatory planning processes. The information obtained from multidisciplinary RAP can also help harness the ecological science for the purposes of management, by highlighting which areas within the overall target area are most critical to protect, which human activities are sources of potential conflicts and which stakeholders will be affected by new regulations.

Thus, there is an interest in developing specific assessment methodologies for coastal conservation purposes that adopt a systemic approach going far beyond the simple management of fishing resources or the protection of a particular habitat or species. Multidisciplinary coastal RAPs will facilitate ecosystem based marine management (Agardy, 2010) serving as basis for implementing actions in zoning, management and regulation of coastal areas requiring protection.

In this work we propose a rapid assessment methodology for coastal marine areas, integrating information of human activities and bio-physical aspects of the area under study, selecting a set of indicators and integrating them in final products to support decision making process. These products are designed to be discussed openly, prior to MPA declaration, in order to grant the local stakeholders a relevant role in the decision taking procedures.

To be rapid and cost effective, the method has been based on obtaining and merging existing information for the area, knowledge of local stakeholders, and an optimized design of *in situ* sampling. The main outputs consist of three cartographic products, giving information on: a) human pressure, b) marine environment quality and c) coastal environmental quality, as well as a matrix of conflicts among different human activities.

The proposed methodology was tested in a coastal area of Mallorca Island (Balearic archipelago, NW Mediterranean) that comprises 26 linear km of coast, covering a total of 2013 ha (1445 ha of marine surface and 568 ha of land) and including a main fishing and leisure boat harbor, Cala Ratjada (Figure 3.1). This study area was selected because it presents multiple human activities characteristic of the western Mediterranean coast, both on the terrestrial and the marine side as well as different levels of anthropogenic impacts.

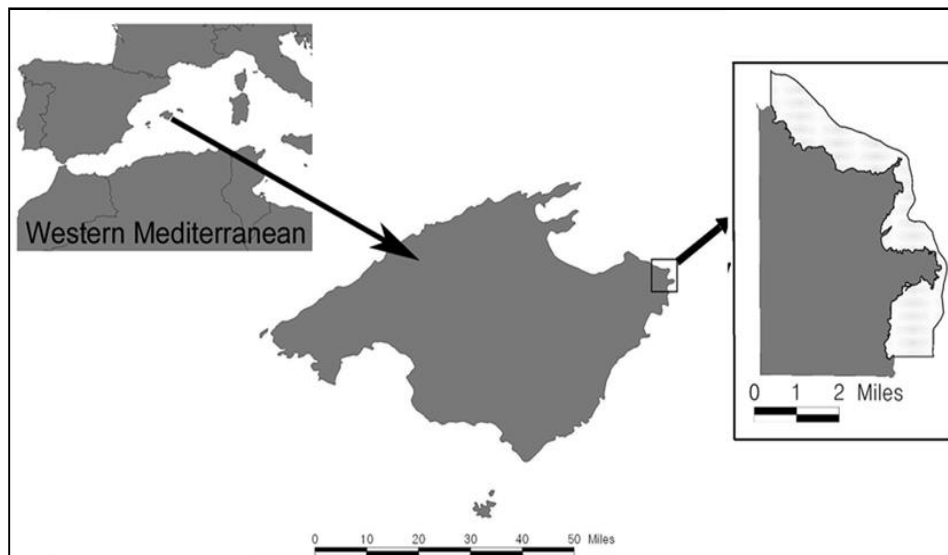


Figure 3.1: Mallorca Island, Balearic, Spain. Location of study site



## 3.2 Methods

The information needed to address the multidisciplinary approach was identified from studies oriented to the selection of candidate sites for MPAs (Roberts et al., 2003), MPA evaluation and zonation (Kelleher, 1999; Pomeroy et al., 2004; Villa et al., 2002), general Integrated Coastal Zone Management (ICZM) (Barragán Muñoz, 1997), integration of information obtained from local stakeholders (Vella et al., 2009), and integrated management of Mediterranean MPAs (Ojeda-Martinez et al., 2009). The general framework to rank indicators and develop cartographies has been adapted from already proposed methodologies developed for terrestrial environment (Machado, 2004) and for coastal areas (Ballesteros et al., 2007).

Based on these studies we adapted a number of available environmental and socio-economic indicators. When different indicators provided information about a common subject they were grouped into indices facilitating data management and interpretation (Ebert & Welsch, 2004; Hargrave, 2002).

The selection of indicators and their corresponding weighting factors have been obtained by a “nominal group technique of experts” (Stewart et al., 2007) that allows the capture of most of the knowledge flow among experts. For the application of this technique, experts on coastal conservation, marine protected areas monitoring and marine biology make an initial proposal of design of each index (identification of indicators and weighting factors), including a rationale of the proposal. Information is then shared among experts. After similar proposals were grouped or eliminated, an open discussion forum ends with a second proposal and ranking of solutions.

The selection of indicators for the RAP was guided by the following criteria: a) easy to measure; b) data gathering without the need to install oceanographic equipment; c) adjustable to simple scales of measurement, and d) minimizing the need for data processing after sampling. Selected indicators were classified attending to their spatial scale and to the origin of the information. Indicators at macroscale (>1Km), mesoscale (100-500 m) and microscale (<100 m) levels were identified. Indicators were divided into two kinds: Class 1, derived from existing information or interviews to local stakeholders, and Class 2, based on *in situ* sampling. The complete list of indicators, the scales to which they belong and the variable about which it gives information about are presented in Table 3.1.

Human activities pressure indices (id. 2, table 3.1), Fisheries diversity index (id. 3, table 3.1), Habitat mesoscale index (id. 13, table 3.1) and Habitat microscale index (id. 14, table 3.1) are related to human activities, fisheries diversity and habitats of interest found in the study area. Therefore the methodology has to be easily adapted to the local characteristics. Here we propose the generic methodologies for data collection, indicator calculation and ranking.

Table 3.1: List of indicators (\*=information collected from questionnaires).

Class	Scale	Name	Variable	Id
1	Macro	Degree of exposure index	Habitat quality for rocky fish communities	1
		Human activity pressure indices (activity specific)	One pressure index for each human activity identified	2
		Fisheries diversity index	Species and habitat richness	3
		Urban pressure index	Human pressure in the coast	4
		Level of administrative protection	Inclusion or non inclusion of Natura 2000 sites, MPAs, etc	5
	Meso	Quality diving site index	Habitat quality state	6
		Intensity of recreational diving	Pressure by recreational scuba diving	7
		Biodiversity hotspots	Habitat and fish community richness	8
		Habitat of interest	Presence of habitats of community interest (Dir 43/92), threatened and/or legally protected	9
		2		Naturalness index
Water quality from coastal macro algae	Coastal water quality			11
Seabird index	Degree of habitat conservation			12
Habitat mesoscale index	Quality of habitat			13
Micro	Habitat micro-scale index		Quality of habitat	14
	Invasive algae		Degree of wild state	15
	Singular species		Degree of wild state	16

Macroscale indicators were always Class 1 indicators as they inform of geographically wide areas. The method is designed to avoid having to complete *in situ* sampling along the entire study area, taking instead existing information and knowledge from local stakeholders and combining those with limited sampling. Mesoscale indicators were Class 1- when they were calculated from interviews or Class 2- when they were referred to habitat characteristics. Microscale indicators, giving information about environmental quality of particular habitats were always Class 2 indicators.

Categorization of indicators is an advantage when different types of information have to be merged in “easy-to-use” products for managers (Hargrave, 2002). Following this approach local values of indicators presented in table 3.1, used for calculation of final products were standardized to simple scales 1 (very low), 2 (low), 3 (medium), 4 (high) and 5 (very high), interpretation of values are provided for each indicator along the methodology.

A scheme of the three methodological phases and specific activities is presented in Table 3.2.

Table 3.2: Methodological phases and tasks.

<b>Phase</b>	<b>Task</b>	<b>Activities</b>
<b>Phase 1</b>	1.1-Compilation of existing information And GIS development	Compilation of cartography, previous studies, main human activities in the area, identification of local experts. Setting up a GIS
	1.2-Structured interviews	Development of questionnaires for the structured interviews. Carrying interviews to relevant stakeholder to collect information about: 1) Fisheries, 2) Diving sites and biodiversity hot-spots, 3) Human activities and conflicts.
	1.3-GIS project & Drawing up descriptors base maps	-Setting a GIS and the grid map for data integration. -Develop cartography of study area descriptors: Geomorphologic units, Homogeneous slope areas and main habitats. -Identification of target habitats for evaluation
	1.4- Calculation of class 1 indicators.	Evaluate and standardize the class 1 indicators (table 3.1).
<b>Phase 2</b>	2.1- <i>In situ</i> sampling indicators that do not require underwater techniques	I- Naturalness index for the terrestrial side of the coast (indicator 10 in Table 3.1). II- Water quality and the seabird index for the marine side of the coast (indicators 11 and 12 in Table 3.1).
	2.2-Identification of mesoscale units and calculation of habitat mesoscale indices.	For each habitat under evaluation: I-Identify relevant mesoscale habitat specific indicators. II-Delimitation of areas where mesoscale habitat specific indicators present homogeneous values. III-Sampling the mesoscale habitat specific indicators at each unit of analysis. IV-Process mesoscale index.
	2.3-Identification of microscale sampling locations and calculation of habitat microscale index	For each habitat under evaluation : Delimitate areas of homogeneous mesoscale habitat specific indicators values. Sample microscale habitat specific indicators at each area delimited.
<b>Phase 3</b>	3.1-Process final products	I- Thematic map of marine environment quality. II- Thematic Map of terrestrial environmental quality. III- Human pressure thematic map. IV- Activities conflict matrix.

### 3.2.1 Phase 1.

#### **Task 1.1. Compilation of existing information.**

Several sources of spatial data for the study area, both for the terrestrial and marine environments, were explored among the different administrations at local, national and European levels as well as online public remote sensing data sources. Previous research studies on the marine environment, especially those related to local fisheries in the area, were also compiled. Key administrative bodies and other local stakeholders were identified.

#### **Task 1.2. Structured interviews**

The local stakeholders were guided through structured interviews (Taylor and Bryan, 2002) to complete predefined questionnaires. Three types of questionnaires were designed for the study area for collecting information proposed as relevant for evaluating management effectiveness of MPAs from local stakeholders (Vella *et al.* 2009):

Questionnaire 1. Fisheries related information: Structured interviews to fishermen and fisheries technicians provided the identification of fishing tactics (gears) of professional and recreational fisheries, target species, fished habitats and other habitats of interest in the area, and the spatial distribution of fishing effort of each tactic. Spatial distribution of fishing effort was evaluated by stakeholders from 1 -areas supporting minimum fishing effort- to 5 -areas supporting maximum fishing effort.

Questionnaire 2. Diving related information: Structured interviews to diving centers/clubs, responsible of marine monitoring and marine researchers provided information about: Scuba diving locations and relative diving pressure at site, their relative quality in terms of seascape and the locations of biodiversity hot spots and identification of habitats of interest in the area.

Questionnaire 3. Human pressure related information: Structured interviews to stakeholders allowed identifying important human activities in the area, the relative intensity of its spatial distribution and the degree of conflicts among all activities.

Parameters in questionnaires 2 and 3 were evaluated by stakeholders from 1 to 5 five following similar ranks that in questionnaire 1.

#### **Task 1.3. Setting up a GIS and development of seabed cartography.**

A GIS (ArcView 3.2, ESRI Inc.) was set up to manage all cartographic information from tasks 1.1 and 1.2. This information was used for delimitating marine habitats of interest and

designing a grid. Each cell of the grid acquired a value for each indicator (Figure 3.2). The grid cell size is chosen upon spatial information accuracy of maps obtained in task 1.1. When one of the habitat mesoscale or micro-scale index presented different values in one cell due to the presence of different habitats of interest, the maximum value was given to the cell.

Cells located over an area affected by the presence of a harbor were flagged as harbor affected area and were excluded from the evaluation of any other indicator, ensuring that cells are easily identifiable in all final products.

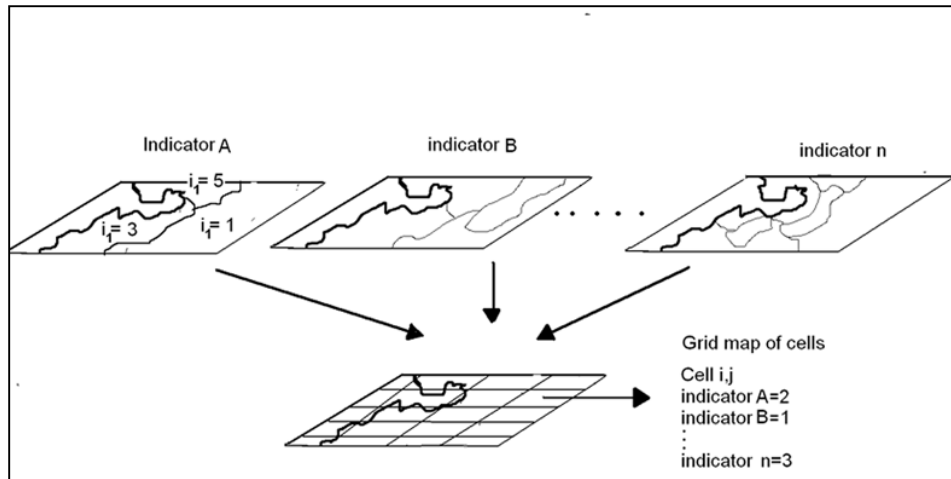


Figure 3.2: GIS scheme for integration of indicators and indices.

The cartographic information integrated in the GIS was used to generate three base maps (herein referred as study area descriptors) used for the later calculation of environmental indices and identification of *in situ* sampling sites. The three base maps are:

-Map of geomorphologic units: Delimitation of areas with similar geomorphologic structures. Different geomorphologic classes were adapted locally from information collected in task 1.1.

-Homogeneous slope areas: Delimitation of areas with homogeneous values of bathymetric slopes calculated from bathymetric data obtained from task 1.1. The grid cells are ranked from 1 (very low slopes) to five (very high slopes)

Map of marine habitats: Delimitation of main habitats identifiable in the areas from cartographic products collected in previous tasks.

Habitats in the study area that will be evaluated during the assessment have to be selected with the information coming from tasks 1.1 and 1.2. Specific information to evaluate

the quality status of those habitats is based on a set of meso and micro-scale indicators, which are either selected or adapted from scientific literature.

**Task 1.4. Calculation of Class 1 Indicators.**

Degree of exposure index: Areas of the same habitat with similar slopes are delimited. Ranks for each area are proposed as follows, based on the effect of slopes in the species richness of fish assemblage (Hamner *et al.* 1988; Harmelin 1988; Mcgehee 1994; Kiflawi & Genin 1997): (1) gently sloping coves and bays that connects without discontinuity with respect to adjacent shoreline, (2) straight, gently, sloping shoreline, (3) small caves, closed and shallow with bathymetric and habitat discontinuity with respect to the exterior, (4) islets or capes with even depths and habitat similar to those on the surrounding coast, straight shorelines with cliffs and sharp bathymetric drops, (5) islets or capes with sharp bathymetric drops that contrast with the surrounding coast.

Human pressure indicators: Adapted from previous studies (Villa *et al.* 2002; Pomeroy *et al.* 2004; Vella *et al.* 2009). Questionnaires obtained in task 1.2 provided grid maps of activity pressure in the area. The final value of pressure index for each activity was calculated by means of the spatial distribution values obtained from the questionnaires.

The fisheries diversity index informs about the species diversity and type of habitats. Values associated to fishing grounds were locally ranked from one- low diversity of target species being fished on low interest habitat-, to five- high diversity of species being fished over habitats of interest-. Local ranks were obtained through a nominal group technique who evaluated the fisheries operating at each fishing ground in the study area. The evaluation of fisheries data to obtain fishing ground indicators has been adapted from Mouillot *et al.* (2002).

The urban pressure index: Values were assigned as a function of buildings and roads density based on aerial photos and maps. Areas of similar density of urban development were grouped and values were assigned to the area from one (very low pressure: absence of buildings and/or roads in the cell, to five: very high dense urban development).

The level of protection: Ranked from 1, areas not protected, to 5, maximum protection level for biodiversity conservation found inside the study area. Equivalent to a marine or terrestrial reserve, or a marine no entry/no take zone (Pomeroy *et al.* 2004).

Mesoscale class 1 indicators: Quality diving site index, Biodiversity hotspots (id. 7, table 3.1) and Habitat of interest, allows mapping locations of diving site highly prized both for divers and clubs, and already known by local divers. Pressure on these areas is measured with the Intensity of recreational diving value. The indicators were proposed, with slight

modifications from Vella *et al.* (2009). They are ranked from 1 to 5, from standardized means of the spatial distribution values obtained from the questionnaires in task 1.2.

### **3.2.2 Phase 2.**

#### ***Task 2.1. Sampling of indicators that do not require underwater techniques***

Aerial pictures from task 1.1 and landscape photographs taken by the working team assessed the main habitats and state of conservation for the evaluation of the naturalness index (Machado 2004).

The macroalgae indicator species was evaluated visually in the surf zone during surveys along the coast for the calculation of the water quality index. Surveys were conducted in spring - due to the seasonal growth of the algae considered for the calculation of this index - with an inflatable boat that allowed nearshore sampling (Ballesteros *et al.* 2007). Seabird values were obtained from the location of the seabird colonies of threatened species observed during these surveys.

#### ***Task 2.2. Identification of units of analysis for the selection of mesoscale underwater sampling sites and calculation of habitat mesoscale index***

The set up of the site sampling selection procedure begins with the identification of homogeneous areas which have to be sampled at meso and micro-scale levels, herein referred as units of analysis, which are habitat specific. The identification of these areas starts with the selection of the main environmental variables that drive and control the development of that particular habitat. These environmental variables are expressed as indicators calculated from previous obtained information and ranked from 1 to 5. Areas with homogeneous values are delimited defining the units of analysis of that particular habitat. Each unit of analysis has to be sampled to obtain representative values of the mesoscale habitat specific indicators. The number of samples per unit of analysis can be adjusted to the time and budget available. This approach for sampling stratification allows maximizing the resources to ensure the collection of a spatially representative sample of mesoscale indicators.

Once the selected sites have been sampled, mesoscale indicator values must be transferred to the cells in the grid map of the GIS. Other non sampled cells where that habitat is present take the value for each indicator from the closest sampled site in the same unit of analysis.



For each habitat, mesoscale indicators were grouped into the habitat specific mesoscale index designed by a nominal group technique.

***Task 2.3. Identification of micro-scale sampling locations and calculation of habitat micro-scale index.***

Samples of micro-scale habitat specific indicators have to be taken at representative sites for the mesoscale index homogeneous areas inside each unit of analysis of that habitat, and eventually at hot spots identified in Phase 1. If budget limitations affect micro-scale sampling regimes, sites must be selected to ensure data from units of analysis with the highest values of mesoscale index.

Micro-scale habitat specific indicators were grouped into the habitat micro-scale index, formulated specifically for each type of habitat following a nominal group technique.

Additionally two more indicators were always evaluated:

Invasive algae abundance index: Invasive algae present in the study area were identified in phase 1, percentage of coverage of the bottom was visually evaluated and final values were ranked from 1 to 5 (minimum and maximum coverage detected in the area).

Singular species with indicator value: Species with value in terms of conservation, and potentially present in the study area identified in phase 1, were selected for evaluation. The presence of each individual was annotated. On each transect a final value of abundance for each species identified was obtained. Abundances in the transect were ranked from 1 to 5, corresponding to the minimum and maximum values in the area.

### **3.2.3 Phase 3.**

This phase consists of data analysis and generation of the final outputs, integrating the information provided by the indicators from three different perspectives: (1) an ecosystemic evaluation reflected in the thematic map of marine environmental quality index and the thematic map of coastal environmental quality index, (2) the spatial distribution of the human pressure, reflected in the thematic map of human pressure index and (3) the synergies of different uses and activities reflected in the activities conflict matrix.

**Product 1:** *Thematic map of Marine Environmental Quality Index (MEI)*. The grid values are calculated from the sum of selected indicator classified in three levels according to the assigned weighing factors: indicators of the highest importance were assigned a weight factor

of 2; indicators of high importance were assigned a weight factor of 1.5 and finally indicators of medium importance were assigned a weight factor of 1.

Final values obtained from each equation are ranked linearly from 1 (minimum value of the index in the study area) to 5 (maximum value of the index in the study one area).

Equation 1:

$MEI = 2 (\text{mesoscale index} + \text{degree of exposure index}) + 1.5 (\text{fisheries diversity index} + \text{the biodiversity hot-spots} + \text{singular species} + \text{micro-scale index}) + (\text{quality diving sites index} - \text{invasive algae} + \text{quality index derived from coastal macro algae})$

**Product 2:** *Thematic map of Coastal Environmental Quality Index (CEQI)*. Includes class I indicators for the terrestrial environment giving information on environmental quality.

Equation 2:

$CEQI = \text{seabird index} + \text{naturalness index} - \text{urban pressure index}$

**Product 3:** *Thematic map of Human Pressure Index (HPI)*. Each cell takes the sum value of the activity pressure intensity processed from information obtained by questionnaires in Phase 1.

Equation 3:

$HPI = \text{Anchoring Pressure index} + \text{Navigation pressure index} + \text{Professional fishing pressure index} + \text{Recreational fishing pressure index}$

**Product 4:** *Activities conflict matrix*. Assessing the present or potential conflicts among stakeholders is a key question to be considered for the management of an MPAs (Vella *et al.* 2009). Here the evaluation is based on the matrix of conflicts, where activities and interests on the marine environment can eventually collide with each other in pairs (Barragán Muñoz 1997). Conflicts among activities were evaluated by stakeholders during questionnaires used in Phase 1.

### 3.3 Results.

We field-tested the methodology by first identifying the relevant stakeholders, including: National Agency for Coastal Management, Balearic Government departments of fisheries and environment, local agency of cartographic data (SITIBSA), Administrative departments of Artá city, Spanish Oceanographic Institute and Mediterranean Institute for Advanced Studies, diving centers operating in the study area and local fishermen association.

The main sources of cartographic information gathered during Phase 1 included nautical charts, remote sensing images from SPOT satellite obtained from Google Earth and aerial photographs from the coast provided by local Government. The main source of information for identification of threatened species, indicator species, species of commercial interest and invasive species were found in inventories of fish communities in the Balearic archipelago (Riera *et al.* 1993), studies on invasive species (Boudouresque & Verlaque 2002), IUCN lists (Hilton-Taylor 2000) and local red lists (Mayol *et al.* 2011),

The main human activities identified during tasks 1.1 and 1.2 were professional and recreational fishing, swimming and tourism, navigation and anchoring, diving, harbouring and urban development. The questionnaires developed for the *structured interviews* (n=14) in task 1.2 allowed mapping the distribution of human pressure from these activities. This information also provided the spatial distribution of fishing tactics and identification of potential hotspots for biodiversity. The information related to local fisheries collected in task 1.1 and 1.2 allowed the setting up of the local ranks (Table 3.3) for the local fisheries diversity index.

Table 3.3: Fisheries diversity index local ranks.

Type of fisheries within each cell	Indicator value
Surface trolling line fishing area.	<b>1. Very low.</b>
Fishing areas with a concurrence of surface troll lines, trolling lines and hand lines.	<b>2. Low.</b>
Fishing for red mullet and cuttlefish (trammel nets) associated with seagrass meadows zones where hand lines fisheries converge.	<b>3. Medium.</b>
Ancient semi pelagic long line fishing grounds where surface and bottom troll lines is also practiced.	<b>4. High.</b>
Ancient fishing grounds for semi pelagic longline fishing and spear fishing for target species that are indicators of the reserve effect, where trolling is practiced and with fishing areas where trammel nets are used for red mullet.	<b>5. Very high.</b>

Identified substrates and key habitat codes (Aguilar *et al.* 2006) in the study area were: 1- Rocky reefs associated with submarine promontories, 2-Rocky reefs associated with limestone

islets, 3- *P.oceanica* meadows over rocky platforms or sandy bottoms, 4- Submarine caves and 5- sandy beds.

The *Geomorphologic units* base map descriptor were adapted from previous generic classification (Bird 2008) and included for the experimental site 1) cliffs; 2) flat rocky shore without scree; 3) flat rocky shore alternating with small caves, and 4) beaches.

The *main habitats* descriptor base map obtained in task 1.3 was compared with bionomic maps from high quality sampling techniques (side scan sonar and trawled video) to evaluate the level and quality of the information provided by free access data and local fisheries distribution (Figure 3.3).

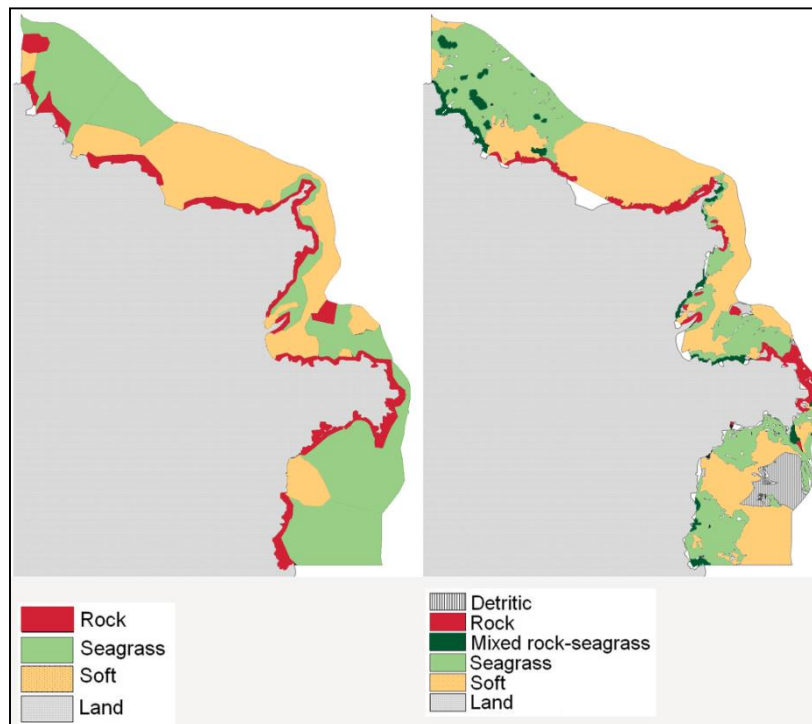


Figure 3.3: Bottom types identified in the study area. Data from Class 1 mesoscale indicators (left) and data from high resolution side scan sonar and trawled video.

Based on this cartography, a grid cell size of 300 m was chosen, which represents six times the minimum mapping unit of the developed habitat map. This relation showed the best equilibrium between spatial definition required and spatial quality of available information. Class 1 indicators were calculated inside this grid. Cartography of *fisheries diversity index*,

naturalness index and human activity pressure index, calculated for boat traffic navigation are presented as examples of cartographic results (Figure 3.4).

The habitats selected for evaluation were those associated to rocky bottoms and *Posidonia* meadows, while sandy bottoms were excluded. This selection was based on the higher abundance of fish species that can respond to protection in these types of bottoms. Submarine caves were evaluated as part of rocky bottoms and following the same techniques.

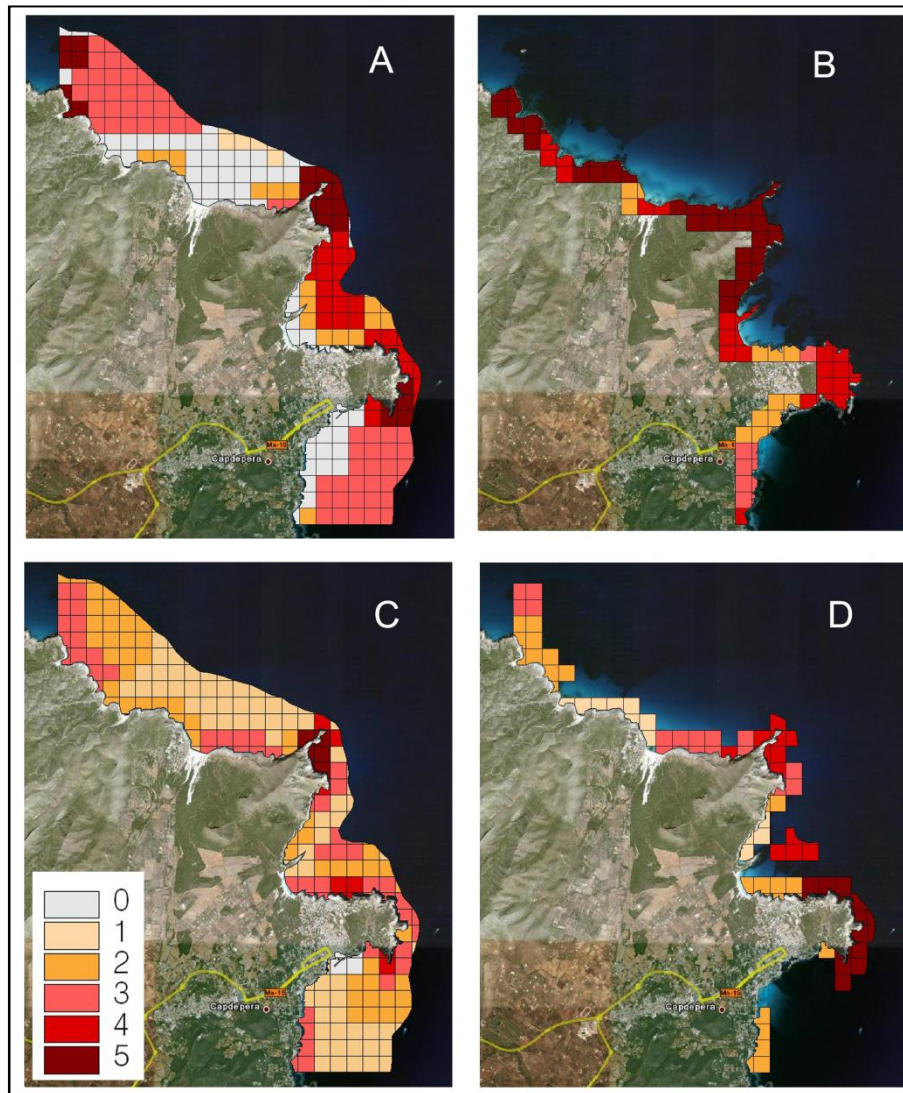


Figure 3.4: Fisheries diversity index. (A), Naturalness index (B), Mesoscale index, each cell present values related to main habitat inside: rocky bottom or *P. oceanica* meadows (C), Micro-scale index over rocky bottoms. (D).

Habitat specific indicators at mesoscale and micro-scale level for each habitat selected for evaluation in the study area are presented in table 3.4. Rationale for selection, techniques for sampling and ranks for these indicators are provided as supporting information (appendix A).

Habitat macroscale and micro-scale indices obtained through the nominal group technique to group that information are provided as supporting information (appendix B).

Table 3.4: List of mesoscale and microscale habitat specific indicators

Habitat	Scale	Name	Variable	
Rocky bottoms	Meso	Habitat heterogeneity	Number of different habitat	1
		Habitat connectivity	Spatial structure and patchiness of different habitat	2
		Substrate roughness	Number and structure of holes and cavities	3
		Underwater landscape value	Habitat and fish community richness	4
	Micro	Specific richness of vulnerable species	Degree of conservation	5
		Specific richness of threatened species	Degree of conservation	6
		Fish species biomass index	State of fish communities	7
		Invasive algae	Degree of wild state	8
		Singular species	Degree of wild state	9
Posidonia meadows	Meso	Habitat heterogeneity	Number of different habitat	10
		Habitat connectivity	Spatial structure and patchiness of different habitat	11
		Substrate roughness	Number and structure of holes and cavities	12
		Underwater landscape value	Habitat and fish community richness	13
		<i>P.oceanica</i> meadow cover	Percent of the coverage in sampled area	14
		<i>P.oceanica</i> meadow landscape	Habitat and fish community richness	15
	Micro	Density of shoots and rhizomes	Quality state of the meadow	16
		Demographic balance	Quality state of the meadow	17
		<i>P.oceanica</i> coverture	Quality state of the meadow	18
		Maximum depth of <i>P.oceanica</i> meadow	Water transparency	19
		Invasive algae	Degree of wild state	20
Singular species	Degree of wild state	21		

Macroscale indicators (from table 3.1) and area descriptors selected for delimitation of the units of analysis (task 1.5) for rocky bottoms were: geomorphologic units, degree of exposure index and depth. These variables were selected according to its effect on the richness of associated fish communities (Harmelin 1988; García-Rubies 1997; Reñones *et al.* 1997; García-Charton *et al.* 2000; Massuti & Reñones 2005), the dynamics of benthic communities (Wheeler 1980; Ballesteros 1989; Ballesteros 1991) and the richness of particular fish species (Hamner *et al.* 1988; Harmelin 1988; Mcgehee 1994; Kiflawi & Genin 1997).

*Seagrass* meadows macroscale indicators considered for the delimitation of units of analysis were: The degree of urban pressure index (id. 4, table 3.1) and water quality derived from coastal macro algae. Both indicators are related to the introduction of organic elements and nutrients into the marine environment, which directly affects the state of health of meadows, particularly when these grow on sediments very rich in carbonates (Calleja *et al.* 2007; Diaz-Almela *et al.* 2008).

The sampling sites for application of mesoscale indicators were selected randomly in each unit of analysis. Data were gathered at sites of special interest, and additional mesoscale samplings were conducted at seabeds with high diversity associated with scree areas along with edges, promontories, islands and biodiversity hot spots identify during task 1.2. Nineteen sites were sampled in total.

The spatial distribution of the mesoscale index inside each unit of analysis allowed the establishment of several criteria for micro-scale site selection:

- i- Areas of special interest (bottoms with high diversity associated with scree areas along with ledges, promontories, islands and submarine caves);
- ii- For each unit of analysis: representative rocky bottoms located at depths below 15 meters with high roughness and high heterogeneity, representative rocky bottoms at depths greater than 15 meters with low roughness and low heterogeneity, and *Posidonia* meadows.

*Pinna nobilis* (Linnaeus, 1758), was selected as a threatened species, although invasive algae species were not found during the sampling at the study site.

With all the information collected final products were produced: Marine Environmental Quality Index (MEI, equation 1), Coastal Environmental Quality Index (CEQI, equation 2), Human Pressure Index (HPI equation 3) and the activities conflict matrix (table 3.5). Values of stress among human activities from this table allowed the identification of potential conflicts

in the area, which were: 1) professional fishing with diving and recreational fishing, 2) diving with anchoring and boating, and 3) protection of certain areas with anchoring and urban development.

Maps of MEI and CEQI (Figure 3.5 B and D) showed four hot spots in the marine side (areas 1,2,3,4, in Figure 3.5.A) and two hot spots in the terrestrial area (areas 8 and 9, in Figure 3.5.A). Area number 4 is located near a terrestrial area with the low-medium CEQI, while areas 1 and 2 present the best connectivity between marine and terrestrial well conserved sites, as CEQI in nearby areas is high to very high. The area number 1 also present low to medium HPI while areas 2, 3 and 4 presents high to very high HPI values. Area number 1 presents the best combination of the three indices with the highest environmental global value.

Cartographies also allow identifying areas, 5 and 6 associated to sandy bottoms, with very low MEI. The area number 6 is located near the harbor. Finally it is also possible to identify areas (as number 7) with high HPI but low-medium values of MEI.

Table 3.5: Activities conflict matrix (1: Low; 5: Very high).

	Recreational fishing	Professional fishing	Bathing	Diving	Boating	Anchoring	Urban development	Ports	Natural heritage protection	Cultural heritage protection
Recreational fishing		4	0	0	0	0	0	0	2	0
Professional fishing			0	5	4	2	0	0	2	0
Bathing				0	5	4	3	0	2	0
Diving					4	5	1	0	3	0
Boating						0	0	0	1	0
Anchoring							0	0	4	1
Urban development								0	5	0
Ports									2	0



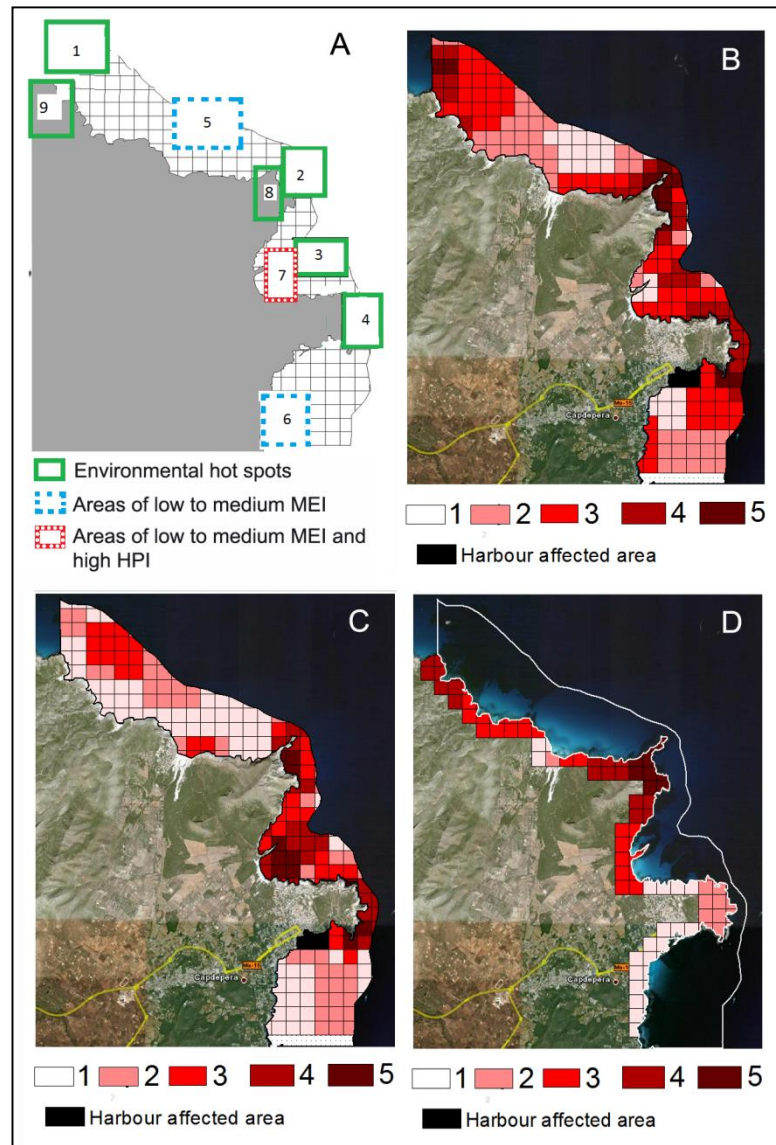


Figure 3.5: Marine natural quality index (A), marine human pressure index (B), Terrestrial natural quality index (C).

## 3.4 Discussion

Our work provides a step by step methodology to integrate indicators containing information related to different uses and activities as well as the quality of the coastal ecosystems, from a multidisciplinary approach. Final products have shown their capability to discriminate areas depending on the environmental quality of the marine and terrestrial coastline and the human pressure exerted on them.

The identification of marine environmental hot spots connected with areas of high terrestrial environmental quality makes them candidates for no take zones or integral reserves in an eventual zoning of a MPA linked to the site. This information, combined with the human pressure index, suggest the best option among the candidate sites for maximum protection level, areas of public use, diving sites or other specific activities, thus making the final products a useful tool for the development of MPA management plan and its zoning. This type of information is relevant policy making criteria that precedes decision making procedures conducted by managers. The use of indices facilitates the discussion of these aspects with final users and even with stakeholders having limited experience in coastal zoning and conservation.

The higher weighting factors assigned to bottom substrate roughness and habitat diversity for the rocky habitat mesoscale index (See Appendix B, equation B.1 ) reflects the fact that experts consulted during the nominal groups gave higher priority to the potential of the habitat to promote high biomass and diversity of Mediterranean littoral fish communities. Additionally the higher weighting factor of mesoscale index and degree of exposure index in the MEI final product against the micro-scale one (equation 1), also indicates that experts considered more relevant the capabilities of habitats to recover than the actual status of the habitat, shown by the micro-scale indicators like species assemblages or seagrass meadow growing capacity.

The process to develop the methodology considered two premises: its adaptability in relation to the resources available for the survey and in relation to the different ecological characteristics within the study area. This explains the relevance given to Class 1 indicators, the interaction with local stakeholders to take advantage of all the existing information and the choice of techniques of nominal groups for index design instead of other multivariate analysis (DeVantier *et al.* 1998).

In order to be applicable in other areas the method has been designed to be flexible in relation to: 1) measuring different type of human activities, 2) including other type of habitats not considered during the study case and 3) applying other indicators for water quality. In this work we proposed a generic approach for obtaining information from stakeholders about human activity pressures and conflicts through the *structured interviews* and to obtain indicators for different type of habitats following a generic scheme for the identification of *units of analysis* and design mesoscale and micro-scale habitat specific indicators. This implies the identification of macroscale environmental variables that drive important differences in the structure and state of that type of habitat, the mesoscale variables expressing real differences in the state of conservation/quality of that habitat, and the key indicators for micro-scale samples. Indicators for mapping coastal water quality have been based on methods based on Mediterranean algae species with indicator values and therefore the application to other geographical areas would require adapting these methods or follow guidelines to design new indicators of water quality (Beliaeff & Pelletier 2011).

Researchers and managers responsible for assessment in other areas may have other interests or objectives for zoning an MPA that might require other set of indicators. New indicators could be included in the final products by assigning correspondent weighing factors. This flexibility also allows the incorporation of other type of ecosystem indices as PREI (Gobert *et al.* 2009) or POMI (Romero *et al.* 2007), proposed for *P. oceanica* in the Water Framework Directive of the European Union's coastal areas evaluation.

Limitations in use of expensive and time-consuming oceanographic equipment and the time required for post processing data analysis influences the selection of indicators and techniques. Once the *in situ* sampling procedure has started, data collected in each task drives the decisions for data collection at the following task. Increasing the time needed between tasks due to oceanographic equipment handling and data processing would increase the time and cost needed for the development of the final products.

In the study case presented here diving based techniques have been selected to collect data at mesoscale and micro-scale level to provided information about habitat potential for recovering (habitat specific mesoscale indicators, Supporting information: appendix A) and habitat quality status (micro-scale habitat specific indicators, Supporting information: appendix A). Even though these techniques require a certain level of infrastructure and expertise, underwater visual census and other diving based approaches are some of the most used techniques in marine rapid assessment for the evaluation of ecosystem quality status (McKenna *et al.* 2002; McKenna 2011). They offer a good balance between quality data and

time required for post processing in comparison to other techniques giving information about similar variables (Stobart *et al.* 2007).

If time and budget are not limitations the method can be enriched from the application of several other techniques, such as acoustic (Kenny *et al.* 2003) or combined acoustic/video for habitat mapping and classification (RODPer & Zimmermann 2007), trawled video for marine species evaluation (Sheehan *et al.* 2010); (Assis *et al.* 2007) or baited underwater video (Stobart *et al.* 2007; Colton & Swearer 2010). These techniques would improve the quality of final products as they could provide better spatial definition and they would also allow to scale up the area evaluated, limited in this study to shallow waters due to the dependence on diving sampling.

The availability of existing data in the study area from oceanographic devices deployed for monitoring or operational oceanography could also improve significantly the quality of final products. For example the existence of previous cartographies of habitat from side scan sonar would improve the spatial definition and quality of final maps, or the access to meteorological buoys could provide better maps of coastal exposition (Ekebom *et al.* 2003).

An important issue to consider for the application of existing oceanographic deployments to the RAP framework is the level and formats of the data provided. Products obtained from deployments must be available in an “end-user friendly format”; on the contrary, time for data formatting or post processing would prevent its application in the scope of the RAP.

The methodology proposed here has been designed to take the most of the resources available, but running this method with very low number of *in situ* sampling may reduce the accuracy of marine quality index. We have recommended at least one sampling site at each identified hot spot and at each unit of analysis, but increasing the sampling effort will for sure improve the final products as spatial definition will be higher.

The techniques used to rank the environmental and human pressure indices from 1 to 5 allowed the separation of minimum and maximum values, thus increasing the spatial differences inside the study area to facilitate the marine zoning, but the use of the same relative indices in other geographic areas may be not feasible.

In many cases the lack of information or resources available for carrying extensive surveillance restricts the possibility of developing comprehensive management plans. In other cases, even when this information exists there are limitations in the knowledge transfer from scientists to managers. RAPs provide cost effective tools to face data collection, but techniques have been developed and examples of their use have been published in relation with tropical

areas and particular studies, as is the ecological state of particular species or ecosystems, but not on the selection and design of MPAs so far. The methodology proposed here, and the final products obtained try to overlap these limitations and constrains, allowing the integration of anthropic activity indicators as well as the global environmental quality of the coastal ecosystems from a multidisciplinary approach.

## **3.5 Conclusions**

The method described herein provides a way to collect and integrate information from multiple disciplines related to management of coastal areas as social ecological systems. The final results are designed to be easy to use tools for managers responsible for zoning and managing marine protected areas. The capacity to incorporate information from local stakeholders and previous studies, along with the feasibility for use in other geographical areas, makes the method a reasonable choice in a time of global budgetary constraints. This tool, with its reduction in costs and time needed to survey, the coordination of already existing but dispersed information, the extensive use of public and accessible resources, the applicability in different biogeographic regions with minor modifications and the maximization of results, can be useful when economic or time constraints run against the effective protection, even at a basic level, of the coastline under threat.

## **3.6 Acknowledgements**

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## 3.7 Supporting information.

### **Appendix A: Mesoscale and microscale habitat specific indicators used in the case study area Artà. Rationale for selection, sampling techniques and ranking.**

#### **3.7.1 Rocky bottoms specific indicators.**

##### **3.7.1.1 Mesoscale rocky habitat indicators (indicators 1 to 4, table 3.4):**

These indicators are measured visually during underwater surveys. Large areas can be covered in as far as divers can be towed by a boat or use submarine scooters. The lengths surveyed can vary from 100 to 500 m depending on the size of the area to be sampled. Indicators are specific for each type of habitat. Values of four mesoscale indicators - habitat heterogeneity, habitat connectivity, substrate roughness and underwater landscape value- were collected in the selected *in situ* sampling sites of rocky bottom zones for each unit of analysis:

The habitat heterogeneity (Indicator 1, table 3.4) expresses the number and the relative extension of sub-habitats (rocky boulders, lifts, caves and flat areas) present in the area assessed. The rank values are higher when the number of sub-habitats and evenness is higher following the same concept of Shannon-Weaver (1948) diversity index. The values are scaled as (1) a single type of sub-habitat, or more than one type providing that 80% of the surface area corresponds to a single type, (2) two types of sub-habitat balanced in size, (3) more than 2 types of sub-habitats that are not balanced in size, and (4) three or more types of sub-habitats that are balanced in size.

The habitat connectivity (Indicator 2, table 3.4) measures the continuity of a type of habitat both in depth and in extensiveness, scaling from (1) habitats in the sampling site that are separated by marked ecotones, (2) boundaries between habitats are patchy or interlocked, and (3) very interconnected habitats without boundaries. These values measures the influence of habitat continuity on the fish species biomass exportation/retention (Gillanders et al., 2003; Palumbi, 2004; Forcada et al., 2009) giving higher rank to habitat structure promoting species movement.

The *substrate roughness* (Indicator 3, table 3.4) is a semi-quantitative measurement of the vertical relief and of the presence of dips. This index is derived from the semi-quantitative application and on a larger scale of the roughness concept (Luckhurst & Luckhurst, 1978) which has demonstrated important effects on Mediterranean littoral fish assemblages (Ordines *et al.*, 2005; Mellin *et al.*, 2007). Classification values are: (1) flat rock; (2) small boulders and/or scattered crevices or dips occupying less than 25% of the subzone's surface area, and/or small scale cavities; (3) boulders of various sizes and/or crevices or dips occupying more than 50% of the surface area and (4) areas mostly occupied with scree and/or cavities of a certain size.

*Underwater landscape* (Indicator 4, Table 3.4). Visual landscapes have shown to provide valuable information for the evaluation of landscape functions in terrestrial environments (Tveit *et al.*, 2006). In this methodology indicator values for rocky habitat have been obtained by a nominal group of experts in MPA monitoring: (1) homogeneity of the seabed in terms of habitat type and bathymetry, (2) heterogeneous seabed with slight bathymetric variation, (3) heterogeneous seabed with a steep bathymetric gradient and (4) significant topographic and bathymetric heterogeneity, diversity of benthic communities and outstanding elements.

### **3.7.1.2 *Microscale rocky habitat indicators (indicators 5 to 7, table 3.4):***

In rocky habitats, microscale indicators were collected in transects of 5 minute-long track where 2.5 meters at each side of the center of the track were sampled following a *visual fast count method* (Kimmel 1985). These are carried out at low speed, swimming at a distance of about 1 m above the bottom. .

Indicators are based on the evaluation of biomass and diversity of fish species as well as of the presence or absence of indicator species (algae, molluscs, crustaceans). Similar indicators are standard in RAP methods used in marine tropical environments (McKenna *et al.*, 2002; Atlantic and Gulf Rapid Reef Assessment AGRRA <http://coral.aoml.noaa.gov/agra/>)

In each transect, the fish species vulnerable to professional or recreational fishing (angling and spear fishing) belonging to spatial categories 3, 5 and 6 according to (Harmelin, 1987) were counted and size estimated in 2 cm classes. For species from category 1 only presence/absence data were recorded. The indicators to be calculated for each transect are:

*Specific richness of vulnerable species (S)* (Indicator 5, table 3.4). The species to consider as vulnerable are those littoral species over rocky substrates that are included in the spatial categories 1, 3, 5 and 6 from the list of the target species in local fisheries (Harmelin, 1987). The ranking is performed on data obtained in field work by splitting its distribution in four

classes considering the maximum values obtained in marine protected areas in the Balearic Islands (Coll et al., 2012).

*Specific richness of threatened species (Sa)* (Indicator 6, table 3.4). The values are calculated as in previous indicator. The list of threatened species must be obtained for the area of study from local Red Lists, previous research or interviews.

*Fish species biomass index (Bv)* (Indicator 7, table 3.4). It is Calculated from biometry, correlating fish size with weight (Morey et al., 2003) the values are ranked from 1 to 5 as in S. Minimum and maximum values where adjusted from marine protected areas in the western Mediterranean (Coll et al., 2012; García-Charton et al., 2004).

### **3.7.2 *Posidonia* meadows specific indicators.**

#### **3.7.2.1 *Mesoscale Posidonia meadow indicators (indicators 10 to 15, table 3.4):***

For *seagrass* meadows four mesoscale indicators were selected, the habitat heterogeneity and connectivity, the meadow cover and the meadow landscape. The habitat heterogeneity and connectivity (indicators 10 to 11, table 3.4) following the same techniques as for rocky bottoms. During the transects two more indicators were collected

*Meadow cover* (Indicator 14, Table 3.4). It expresses the percentage of the sampled area covered by the *P.oceanica*, in contrast to the percentage covered by other types of habitats (sandy bottoms, rocky without seagrass). The scale of values was: 1= less than 20% of the sampled area covered with seagrass meadows; 2= from 20 to 40%; 3= from 40 to 60% o; 4= from 60 to 80%; 5= more than 80% covered with *seagrass* meadows.

*Meadow landscape* (Indicator 15, Table 3.4). Proposed following same methods than for indicator 23. This indicator enables a first comparison of the different meadows and is useful when it comes to determining the location of microscale sampling. Classification values are 1 to 3 (low, medium, high) and evaluate the general aspect of the area in terms of density of the *P.oceanica*, species associated and presence of small and rich patchy habitats. Low: *P.oceanica* meadows patchy distributed in sand or flat rocky areas with a poor fish communities associated. Medium: continuous *P.oceanica* meadow with richer fish and indicator species communities associated. High: very wide *P.oceanica* meadows with high canopy and shoot density alternated with patches of rough rocky areas and presence of rich associated fish and invertebrate communities.



**3.7.2.2 Microscale Posidonia meadow indicators (indicators 10 to 15, table 3.4):**

For selected sites of *P. oceanica* meadows, the sampling consisted on duplicated 20 meter transects marked with a tape, measured and fixed to the bottom with stakes. The indicators *P.oceanica* cover, shoot density and demographic balance (Ids 29-32, Table 3.1) were measured. Annual net population growth rate and demographic balance ( $\mu$ ) were calculated following methods of (Duarte et al., 1994) to provide information on the biological state of the meadows. The maximum depth of the meadow was calculated as it determines the quantity of light available at the bottom and therefore the lower limit to which the meadow can extend (Duarte et al. 1994) and the structure of the meadow, particularly the shoot density (Marba et al. 2002).

**Appendix B: Habitat macroscale and microscale indices obtained through the nominal group technique for habitats selected in the study area.**

The rocky bottom mesoscale index (RBmeso). (Equation B.1)

RBmeso = 2xsubstrate roughness + 2xhabitat heterogeneity + 1.5xunderwater landscape value+ habitat connectivity.

Rocky bottom microscale index (RBmicro). (Equation B.2)

RBmicro= 1.5xBiomass of threatened species value+ Specific richness of threatened species value

*P. oceanica* mesoscale index (Pmeso). (Equation B.3)

Pmeso= (Habitat heterogeneity value + habitat connectivity value) + meadow cover value+ 2xmeadow landscape value.

*P. oceanica* meadow microscale index (Pmicro). (Equation B.4)

$$P_{micro} = \frac{RB_{micro} + P_{meso}}{2}$$

Estimated values for Pmicro were ranked from 1 to 5 considering 1 as very low (significantly receding), 2 as low (in recession), 3 as normal or stable, 4 as high (in expansion) and 5 as very high (significantly expanding).



## **CHAPTER 4**

**Spatial scale, means and gradients of hydrographic variables define pelagic seascapes of bluefin (*Thunnus thynnus*, Linnaeus 1758) and bullet tuna (*Auxis rochei rochei*, Risso 1810) spawning distribution**

## 4.1 Introduction

Seascape ecology represents an emerging field in the study of how the habitat structure shapes the spatial distribution of marine species and influences key ecological processes (Hinchey *et al.* 2008a; Pittman *et al.* 2011). This discipline initiated applying techniques and metrics from the traditional landscape ecology to characterize and quantify spatial structure of benthic habitats, observed as a mosaic of patches of different habitat classes (Hinchey *et al.* 2008a; Pittman *et al.* 2004; Boström *et al.* 2011). Nevertheless, there is still a gap in the development of concepts and techniques providing metrics to characterize the spatial structure of the seascape in the pelagic environments, where there are no clear boundaries delimitating different habitats (Pittman *et al.* 2011; Wedding *et al.* 2011). In the framework of landscape ecology spatial gradients have been recently proposed as more appropriated metric than traditional categorical patch mosaic based metrics to characterize continuous habitats (Cushman *et al.* 2010). Accordingly, a location in a pelagic seascape would be better characterized by the combination of the value of a particular hydrographic variable and its spatial gradient.

Several studies have applied gradients of hydrographic parameters to characterize the spatial distribution of marine species during various life history stages, as nursery, foraging or spawning (Mannocci *et al.* 2013; Worm *et al.* 2005; Druon *et al.* 2011; Louzao *et al.* 2011; Hidalgo *et al.* 2012). It is likely that the scale at which an individual perceive a change in the environment (i.e., a gradient) varies according to life history, ontogeny and to the hydrographic variable in exam. For instance, while large-scale gradients associated with the North Pacific transition zone drive the location of many pelagic predators including albacore tuna (*Thunnus alalunga*) during their feeding migratory stages (Block *et al.* 2011), once off the west coast of the US, albacore tuna distribution is associated with smaller scale features linked to upwelling fronts (Phillips *et al.* 2014). In the Mediterranean Sea during spawning, bluefin tuna distribution is regulated by oceanographic variables that can change at relatively small scales (Alemany *et al.* 2010; Reglero *et al.* 2012). In spite of the expected importance of gradient scales, to our knowledge there are no studies that have evaluated the effect of changing the spatial scales at which environmental gradients are calculated to model the spatial distribution of fish. The goal of our study is to examine the distribution of large pelagic predators during spawning by explicitly considering mean, gradients and scale of gradients of hydrographic variables. Atlantic bluefin tuna (*Thunnus thynnus*) and bullet tuna (*Auxis rochei rochei*) are two species of pelagic predators showing different spawning strategies. We target

these species in the Balearic Sea, known as a recurrent spawning area for large pelagic species in the Western Mediterranean (Torres et al. 2011). Bluefin tuna is a highly migratory species; the Eastern population enters in the Mediterranean Sea from the North Atlantic at the end of spring and early summer (Block et al. 2005; Rooker et al. 2008). Their spawning activity at the Balearic Sea is linked to the regional oceanography with spawning grounds located in the vicinity of frontal structures formed when the recent Atlantic water mass encounters the more saline resident surface Atlantic waters (Alemany et al. 2010). The area is characterized by highly dynamic processes that trigger a seascape shaped by filaments, fronts and eddies whose location varies between years (La Violette et al. 1990; Balbín et al. 2014). Bullet tuna, by contrast, is smaller and more frequent in near coastal areas (Sabatés & Recasens 2001). The spawning of bullet tuna is associated with the geography. Young larvae are found mainly in coastal areas and with little influence of the local oceanography in comparison with bluefin (Reglero et al. 2012).

We expect that, when selecting spawning locations, a large-bodied and long-distance migratory pelagic fish, such bluefin tuna explore their environment at larger spatial scales than bullet tuna, a small-bodied and non-migratory pelagic fish. Therefore, we expect that pelagic seascape metrics based on the combination of hydrographic parameter values and their gradients calculated at appropriate spatial scales provide relevant information for bluefin tuna but not for bullet, where we expect a greater reliance on geographic and hydrographic parameters, calculated at comparatively small-scale.

In this work we analyze the influence of the pelagic environment by depicting the spatial scales at which gradients of hydrographic variables are linked to the spawning ecology of these tuna species. We investigate the two most relevant hydrographic variables describing their spawning spatial distribution: salinity and geostrophic currents velocity (Reglero et al. 2012), already determined in previous studies. Our analytical approach has two steps. Firstly, we identify the scale at which each hydrographic variable (see Figure 1) maximizes the performance of a model fitted to larval distribution. Second, we evaluate whether components of the seascape (i.e., mean and gradients of oceanographic variables) are interactively affecting the spatial distribution of tuna larvae. By performing this analysis on two species we evaluate how fish with contrasting life history strategies perceive their environments when deciding for spawning locations.

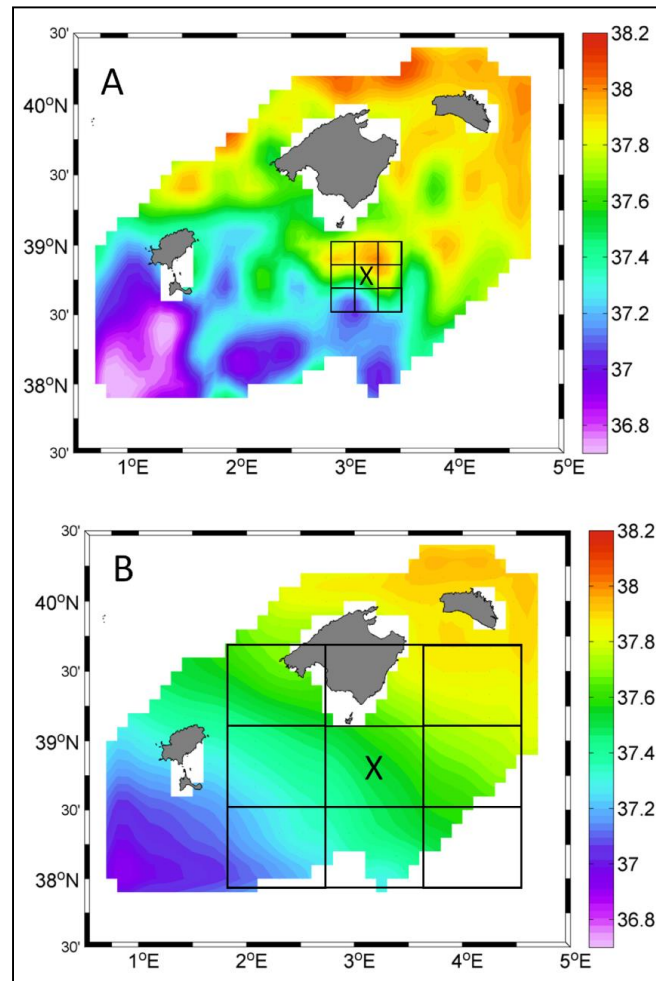


Figure 4.1: Sea surface salinity field in 2033. Spatial means of sea surface salinity processed at two different scales A) 0.15 degrees and B) 0.75 degrees. Spatial means were interpolated following an objective analysis onto a regular grid by using minimum error variance methods (Bretherton, 1976). The squares in each figure show the polygons used for the calculation of the spatial gradients at the two scales at station X.

## 4.2 Materials and Methods

### 4.2.1 Data acquisition

Bluefin and bullet tuna larvae were collected during ichthyoplankton surveys using Bongo nets from 2001 to 2005. The surveys were conducted by the Instituto Español de Oceanografía ([www.ieo.es](http://www.ieo.es)), an Spanish Government marine research institution, using oceanographic vessels belonging to the Spanish Government. The sampling scheme was communicated and approved by the Spanish Directorate of Fisheries before the sampling was conducted. No

specific ethical approval was required and the survey of biological data was conducted using Bongo nets ,which are accepted humane standard techniques for this type of surveys, used worldwide for the collection of plankton samples, including billfish and tuna larvae (Alemany *et al.* 2010; Torres *et al.* 2011; Muhling *et al.* 2010; Rooker *et al.* 2012). The nets were towed at low speeds, around 2 knots, during 8-10 minutes, and plankton samples were immediately fixed with 4% formalin buffered with borax onboard. These surveys are needed to provide scientific knowledge about tuna species and contribute to the understanding of interactions among species and the processes involved in their recruitment, therefore, conservation and survival.

Around 200 stations were sampled every year, in a regular sampling grid of 10x10 nm located between 37.85°- 40.35° N and 0.77° -4.91° E, covering a total area of 101360 km<sup>2</sup> (=280x362km) around the Balearic archipelago (Figure 4.2). The sampling was conducted during June-July coinciding with the spawning period of bullet and bluefin tuna (see (Alemany *et al.* 2010) for details of the sampling procedures). Tuna larvae were identified to the species level and measured in standard length. All larvae identified as yolk sac and preflexion stages (<4.5 mm) were classified as “stage 1”. This stage has been defined to get a proxy of spawning locations (Reglero *et al.* 2011).

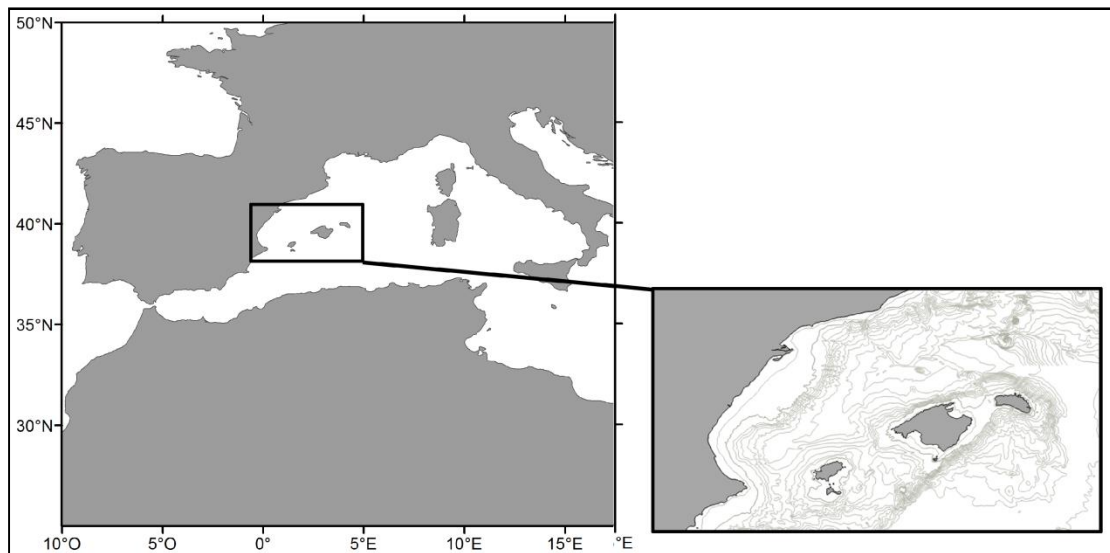


Figure 4.2: Location of the Balearic Islands, Western Mediterranean.

Vertical profiles of conductivity, temperature and pressure data were recorded at all stations, by means of Sbe911 CTD. Sea surface salinity at each station (SAL) was calculated as the mean salinity over the mixed layer depth. Geostrophic velocities (GVEL) were calculated at sea surface from the first-derivative of the sea surface height between adjacent points, which

was obtained by vertical integration of the specific volume, using 600 m as the level of no motion (Balbín *et al.* 2014).

These two variables (SAL and GVEL) were selected, since they have been demonstrated to be the two most relevant environmental variables describing the spawning spatial distribution of tuna (Reglero *et al.* 2012). Sea surface temperature was also included in the models since in this area it is a secondary but relevant variable mainly related to the phenology of the spawning process (Blank *et al.* 2004). However, the spatial gradient was not explored because sea surface temperature during the summer changes relatively fast due to solar irradiance (Balbín *et al.* 2014).

## 4.2.2 Processing of spatial gradients along continuous spatial scales

Spatial gradients from the sea surface salinity field (gSAL) and geostrophic velocity field (gGVEL) within the sampled region were calculated at six spatial scales, from 0.15° to 0.90° with a spatial increment of 0.15°. The minimum (0.15°) and the maximum scale (0.90°) were in the range of the smallest (from 0.13° to 0.27°) and largest (up to 0.92°) mesoscale oceanographic structures in the area (Balbín *et al.* 2014). For the computation of the gradients, nine square polygons at every scale were delimited around each sampling position (see examples for scale 0.15° and 0.75° in Figure 4.1A-B respectively). The gradient was then computed as the maximum absolute difference between the mean hydrographic variable at the center polygon and each of the eight surrounding polygons standardized to distance (Worm *et al.* 2005). The software for the spatial processing was developed in R language (RDevelopment 2011).

## 4.2.3 Identification of spatial scales

Comparison of how models perform along scales allowed identifying the spatial scales at which information provided by gradients is maximized. The effect of gGVEL and gSAL at each scale on the abundance of bullet and bluefin tuna larvae was assessed using nonparametric regression statistical models (generalized additive models, GAMs, (Wood 2006). A base model was formulated to describe inter-annual variability (variable YEAR), sampling location (latitude and longitude variables), and the hour of the day on the catch of tuna larvae. Over-dispersed Poisson distribution family and a natural-log link were selected to model larval data. The volume of water filtered was included as an offset (after natural log transformation), to



account for the effort expended in catching the sample (Equation 1). The effects of these variables on the base model have been already analyzed in previous studies (Reglero *et al.* 2012). Here, the base model represented the null hypothesis of no gradient effect on tuna larvae distribution, against which all other more complex formulations will be compared.

Equation 1: Base model

$$\text{Larvae abundance} = \text{offset}(\log(\text{vol})) + \text{factor}(\text{year}) + s_1(\text{long, lat}) + s_2(\text{hour})$$

vol= volume filtered by the bongo nets (m<sup>3</sup>); long=longitude; lat=latitude; hour=hour of the day expressed from 0 to 1,  $s_1$  and  $s_2$  the smoothing functions

At each spatial scale a GAM model was processed including the gradient of one hydrographic variable (gSAL, gVEL) as a new additive term ( $s_3$ ) in the standardization model. The number of knots for the new smoother was always set to a maximum of three (i.e. two degrees of freedom) in order to avoid over fitting in the responses.

The identification of characteristic spatial scales (cgSAL, cgVEL) was assessed with scalogram where the scale of the covariate is plotted against a measure of the model goodness of fit, which in our case were represented by the adjusted R-squared (Rs<sub>q</sub>, the higher the better), and the Generalized Cross Validation (GCV, the lower the better) (Wood 2006). We selected the scale that maximize Rs<sub>q</sub> and minimize the GCV. Results of the base model (when a seascape covariate was not included) were presented in the same graphics. Note that due to the greater complexity of the gradient model higher Rs<sub>q</sub> values do not necessarily imply an improvement in relation to the base model, while they do represent a better performance when compared to other gradient models.

Significant differences of Rs<sub>q</sub> values between models, or GCVs, were obtained from t-test of these parameters obtained from 500 iterations where 10% of the data was excluded. For all cases, alternative hypothesis (difference in means is not equal to 0) was accepted only if the P value were lower than 0.001, with a confidence level of 0.99. When one variable presented similar Rs<sub>q</sub> and GCV values at various scales, selection was assessed by inspection of the plot showing the response of the abundance in relation to the gradient processed at those scales.

Once the characteristic scale of the gradients was identified, we tackled the questions of whether the information provided by the gradients is different and complementary to the information provided by the hydrographic variables from which they were calculated, and in that case, how the information from these two variables (spatial mean and gradient) should be combined to maximize the goodness of fit of the models and the ecological information they

provide. To assess these questions we analyzed the performance of models with different complexity:

i) The base model from equation 1.

ii) Hydrographic models combining the sea surface salinity, geostrophic velocity and sea surface temperature at the sampling station (stSAL, stGVEL, stSST).

iii) Seascape models combining the gradients at characteristic scales (cgSAL, cgGVEL) and the hydrographic variable at the sampling location (stSAL, stGVEL, stSST). Different seascape models were constructed including the two components of the seascape (values at stations and gradients) as additive and interactive terms. An interaction may be ecologically meaningful when a species is selecting its spawning habitat on a specific side of a frontal region, for example. In such case, it is the combination of both the gradient and the mean that provide the suitable conditions for spawning. The performance of different model configurations for each species was assessed by the delta AIC ( $\Delta AIC$ ), calculated as the difference between model AICs and the base model AIC. The AIC in this case is best suited for model comparisons because each model had different number of variables (Burnham & Anderson 2002).  $R_{sq}$ , GCV and explained deviances were used to compare how models perform between the two species, as AIC values among models with different dependent variables are not comparable.

## 4.3 Results

### 4.3.1 Identification of characteristic spatial scales

In all years considered, the recent Atlantic water masses encountered the more saline resident water masses forming an oceanic frontal zone inside the study area. The size of such frontal zone was bigger than other oceanography phenomena as small eddies and meanders derived from the instabilities along the haline front and the effect of strong bathymetric changes (See Figures S4.3-S4.4 showing the sea surface salinities and geostrophic currents in the area during the five years analyzed).

The scalogram of gGVEL for bluefin showed that  $R_{sq}$  values gradually improve as the spatial scales increased to a maximum at  $0.6^\circ$  ( $R_{sq}= 0.44$ , Figure 4.3A), which was chosen as the geostrophic velocity gradient characteristic scale for bluefin tuna. Values of GCV showed a similar pattern of model improvement, being significantly better than the base model at  $0.6$  degrees (Figure 4.3B). At this characteristic scale the response of the larvae abundance is positively related to the gradient of geostrophic velocity (Figure 4.3C).

The scalograms of gSAL for bluefin tuna showed also an increment of Rsq with higher values at 0.6° and 0.75° that also coincide with lower values of GCV (Figure 4.3D-4.3E). Differences of R-sq between these two scales (0.6° and 0.75°) were not significant. The characteristic scale for gSAL was chosen at 0.6° as the model response at this scale presented a less ambiguous effect on larval abundance (Figure 4.3F). The gSAL at 0.75° spatial gradients displays a dome-shaped response with a less clear ecological interpretation (Figure S4.1). At this scale GCV was lower than the base model.

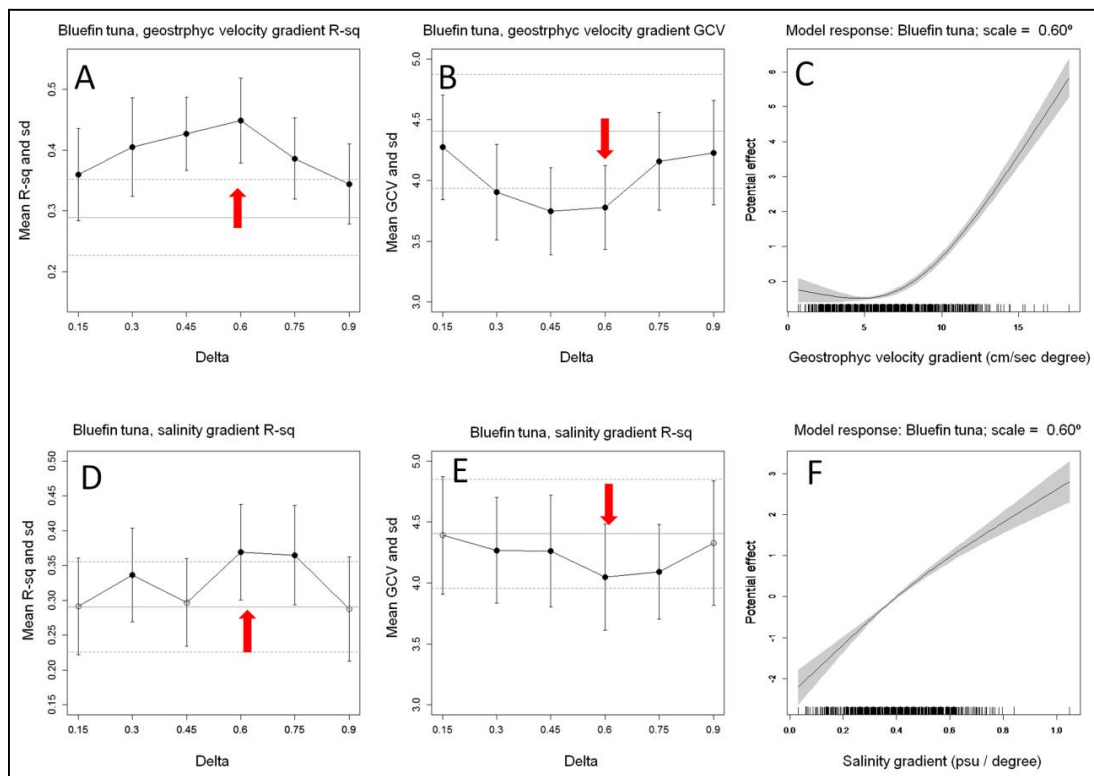


Figure 4.3: Rsq and GCV Scalograms of bluefin tuna larva abundance models along spatial scales, standard deviations. Horizontal grey lines indicate statistics from the base model (Straight line=mean, dashed line= Sd). Black dots show scales at which values are significantly different from the standardization model. Red arrows indicate the selected characteristic scale.

In contrast to the results obtained for salinity, the gradients of geostrophic velocities (gGVEL) did not show any single scale that maximize R-square and minimize GCV (Figure 4.4A-4.4B). The Rsq scalogram showed a flat trend with the highest value at 0.15 degrees. The Rsq value at this scale (= 0.18) showed similar values than other scales (values between 0.170 and 0.173) or when compared to the base model (=0.166). On the contrary the GCV scalogram showed significant lower values than the base model at 0.45 and 0.6 degrees, scales at which

Rsq were not even significantly higher than the base model. Therefore, the contradictory response of the model performance indicators, the flat trend of Rsq scalogram and their very low values (despite the higher complexity of the gradient models in relation to the base model), may indicate that the spatial gradient of geostrophic velocity is not a valid seascape metric for the spawning locations of bullet tuna. Consequentially, gGVEL was excluded from further analysis in relation to this species.

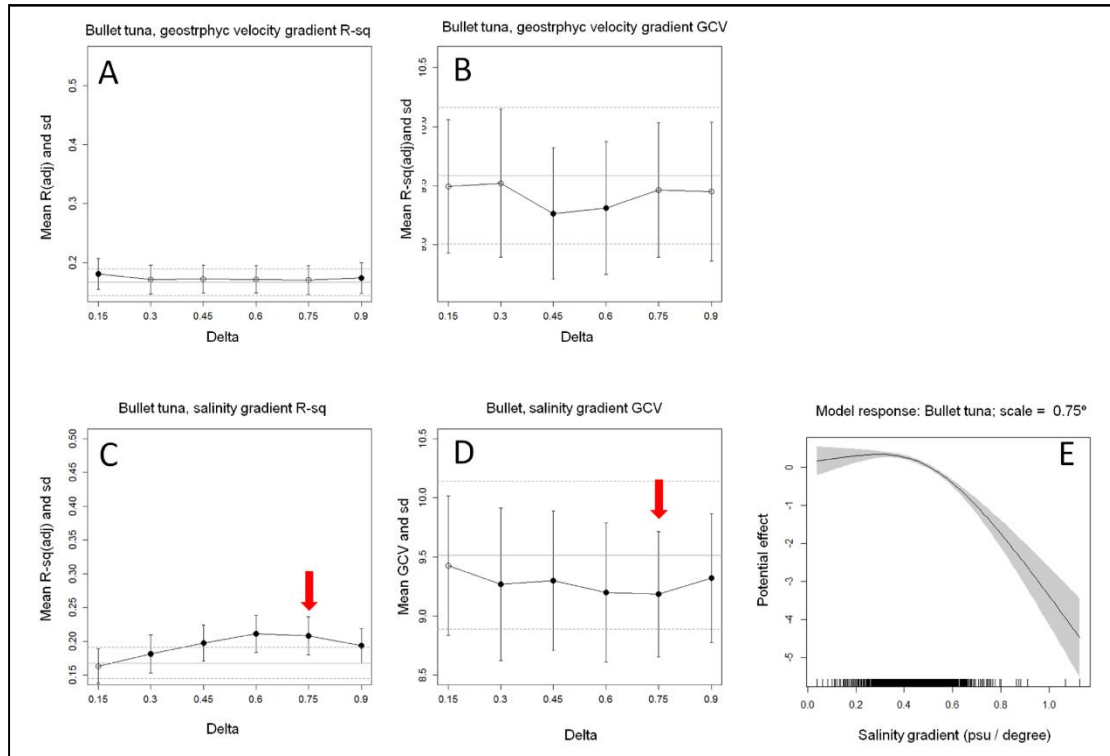


Figure 4.4: Rsq and GCV scalograms of bullet tuna larva abundance models along spatial scales, standard deviations. Horizontal grey lines indicate statistics from the standardization model (Strait line=mean, dashed line= Sd). Black dots show scales at which values are significantly different from the standardization model. Red arrows indicate selected characteristic scale.

The gSAL scalogram of bullet tuna showed a moderate effect of the scale at which gradients were calculated. Shape of the scalogram did not show a peak scale at which model performs better (Figure 4.4C). Scales from 0.45°, 0.6° and 0.75° for gSAL seemed to maximize Rsq, but not being different of each other. In this case GCV-scalograms showed the lowest at 0.75° (Figure 4.4D), being significantly lower than the base model, which was selected as characteristic scale. The Rsq at this scale was higher (=0.21) than that of the base model (Rsqu=0.16). At this scale, gradients displayed a negative effect on the bullet tuna larvae

abundance (Figure 4.4E) showing that bullet tuna spawning locations are found with more probability in areas where salinity is spatially homogeneous— a result that contrasted to that obtained for bluefin tuna.

### 4.3.2 Species-specific seascape characterization

The best model for each species included a gradient and a mean term (Tables 4.1 and 4.2). Note however that the hydrographic model already represents an improvement respect to the standardization model (Tables 1 and 2). For bluefin tuna the best seascape model had an improvement of 186% of the  $R_{sq}$  when compared to the base model ( $R_{sq}$  base mode=0.23;  $R_{sq}$  best seascape model=0.66, Table 4.1). The improvement for bullet was 68%, considerably lower compared to bluefin ( $R_{sq}$  base mode=0.16;  $R_{sq}$  best seascape model=0.27, Table 4.2).

Table 4.1: Summary of gam models of larvae abundance for Atlantic bluefin tuna (*Thunnus thynnus*). Interaction terms included in parenthesis.

Model group	Model variables	R2	Dev(%)	GCV	AIC	delta AIC
Base model	(latitude, longitude) + filtered volume+ hour	0.232	40.8	4,596	3985,54	0
One additive variable models	base model + stGVEL0.15	0,271	43,4	4,412	3827,79	157,75
	base model + gGVEL0.6	0,39	49,4	3,947	3465,03	520,50
	base model + stSAL0.15	0,222	41,8	4,534	3924,70	60,84
	base model + gSAL0.6	0,301	45,1	4,275	3723,53	262,01
Hydrographic model	base model + stGVEL + stSAL + stTEMP	0,472	51,6	3,814	3338,62	646,92
GVEL seascape models	Hydrographic model + stGVEL + gGVEL0.6	0,53	55,8	3,500	3087,14	898,40
	Hydrographic model + (stGVEL,gGVEL0.6)	0,666	59,3	3,251	2881,29	1104,25
SAL seascape models	Hydrographic model + stSAL + gSAL0.6	0,506	54,8	3,571	3145,80	839,74
	Hydrographic model +( stSAL,gSAL0.6)	0,533	57,1	3,417	3009,58	975,96

Pearson correlation coefficients between hydrographical variables and their gradients at characteristic scales were in all cases below 0.50 and pair plots showed no clear tendencies on the correlations (Figure S4.2), indicating that selected gradients provided complementary information to that of the hydrographical variable. Models showed a better performance (i.e. lower GCV and higher  $\Delta$ AIC; Tables 1 and 2) in all the cases when the gradient and the hydrographic value were considered as an interaction term, suggesting dependence in their effect on larvae abundance rather than an additive response. However, larvae abundance of each species responded differently to the interaction of seascape components (Figure 4.5A,B,C)

Table 4.2: Summary of gam models of larvae abundance for bullet tuna (*Auxis rochei rochei*). Interaction terms included in parenthesis.

Model group	Model variables	R2	Dev(%)	GCV	AIC	delta AIC
Base model	(latitude, longitude) + filtered volume+ hour	0,158	32,8	9,615	8857,17	0
One additive variable models	base model + GVEL st0.15	0,177	35,3	9,281	8572,91	284,26
	base model + stSAL0.15	0,16	36,5	9,130	8440,06	417,11
	base model + gSAL0.75	0,207	39,6	8,741	8098,28	758,89
Hydrographic model	base model + stGVEL + stSAL + stTEMP	0,192	38,8	8,847	8183,39	673,78
SAL seascape models	Hydrographic model + stSAL + gSAL0.75	0,215	40,6	8,617	7984,18	872,99
	Hydrographic model + (stSAL,gSAL0.75)	0,255	43,1	8,327	7708,82	1148,35

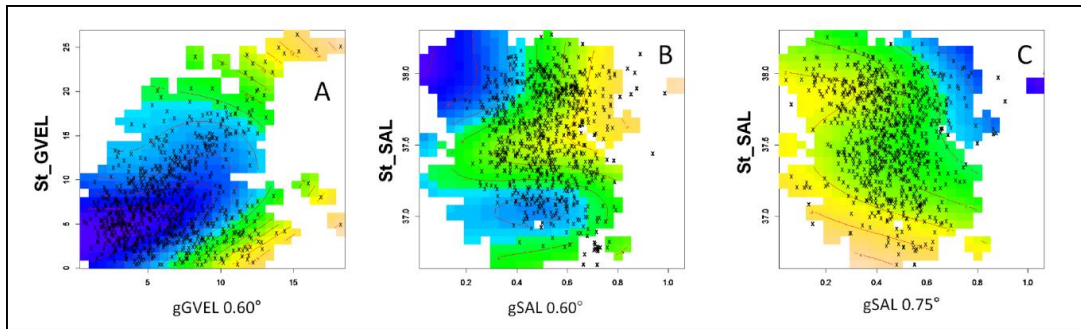


Figure 4.5: The effect of the interactions of the seascape components on the larval abundance as estimated from the seascape generalized additive model. The effects are shown for bluefin tuna (A-B) and bullet tuna (C). For bluefin tuna: A) the effect of the gGVEL and st\_GVEL interaction. B) the effect of the gSAL and st\_SAL interaction. For bullet tuna C) the effect of the gSAL and st\_SAL interaction. Isolines indicate larval abundances predicted by the model. Peak of abundances are indicated in pink-yellow. Low and very low abundances are indicated in green and blue, respectively.

For bluefin tuna, higher probability of spawning was associated to higher values of geostrophic velocity gradients, but where velocities at station may present either high or low values (Figure 4.5A). Considering that a gradient is characterized by an area with high current speed near an area of low current speed, this result indicates that spawning locations were not associated to a particular side of the gradient, but in an area around the location where maximum velocities occurs. The extension of this area would be around a circle of 0.6 degrees of radius (aprox. 65 km in the study area), the characteristic scale at which the gradients were more relevant. In contrast, the interaction of the salinity seascape variables showed high larvae abundances in areas with high salinity gradients and intermediate-high salinity levels (Figure 4.5B), indicating an effect of the location of the main haline front and a preference for spawning at the high salinity values of that front. The characteristic correlation scale of local oceanographic structures in the area is around 18 nmi (0.15 nm) (Balbín *et al.* 2014) and therefore smaller spatial scales surface oceanographic structures are ephemeral. At higher spatial scales as those found relevant for bluefin tuna, structures are more permanent and related to processes as the Med-Atlantic salinity front.

The functional form of the interaction terms was different for bullet tuna. The relation between the bullet tuna larvae abundance and the interaction of spatial distribution of sea surface salinity and its gradients is presented in Figure 4.5. Spawning locations were associated to areas where the salinity at the station were lower and gradients presented intermediate values, but the interaction plot revealed that spawning also appears in areas of higher

salinities associated to very low gradients. Areas defined by this twofold combination were located at both sides of the front avoiding more mixed waters. This spatial distribution was more evident in years 2001, 2003 and 2005 (Figures S4.9 and S4.10)

## 4.4 Discussion

We have found that the combination of sea surface current velocities, salinities and their gradients calculated at characteristic spatial scales are relevant for the parameterization of the pelagic seascape affecting a key ecological process of bluefin tuna. For bullet tuna only salinity and their gradient provided a valid seascape metric not being relevant the gradients of sea surface current velocities. In agreement with our expectations, the importance of these metrics was much higher for bluefin, a large-bodied, long distance migratory and more dependent on local oceanography than were for bullet a smaller coastal species with shorter migration distance.

Previous studies have documented the links between the frontal activity and the spawning of bluefin tuna (Alemany *et al.* 2010; Reglero *et al.* 2012; Muhling *et al.* 2013). In this study we add to these results by examining the effect of gradients and their interactions with hydrographic mean. These metrics improved our understanding of the conditions for bluefin and bullet tuna spawning when compared to models using just the hydrographical values but not the gradients. Furthermore the identification of characteristic scales of gradients provided a new source of information for the interpretation of how local oceanography determines the selection of the site to spawn.

For bluefin tuna larvae, the characteristics spatial scale of both salinity and geostrophic velocity gradients was at 0.6 degrees. The higher abundance of bluefin tuna larvae in areas with intermediate to high salinities and with high gradients of velocity is consistent among years. Higher abundance occurs around the location of the main frontal area, at the side of higher salinity of the front and where current speed present high values. Higher salinity water likely has higher resident time near the islands than the less saline water, which may run along the front towards east getting farther from the archipelago. Spawning at the higher salinity side may favor spatial overlap with other larval species that are also located in this water mass (Torres *et al.* 2011; Rodriguez *et al.* 2013).

Results for bullet tuna showed that pelagic seascape metrics are not as relevant to explain the spawning distribution as they are for bluefin. In the western Mediterranean, spawning of bullet tuna have been associated to near coastal areas (Sabatés & Recasens 2001),



being less influenced by the local oceanography than bluefin tuna (Reglero *et al.* 2012), which is consistent with our results.

Despite the lower importance of bullet tuna the seascape metrics, the inclusion of salinity gradients provided additional information for the identification of spawning sites. The analysis indicated that bullet tuna spawning areas are mostly found in areas where salinity gradients are low. Bullet tuna was found at both sides of the front but avoiding more mixed waters, located closer to the front. This was verified when observing the spatial distribution of larvae in relation to the salinity seascapes among the different years. For instance, in 2001, 2003 and 2005 high larvae abundances were observed North of the archipelago (high salinity waters with very low gradients), but intermediate abundances, indicating spawning, also occurs in Southern areas (low salinities and intermediate gradients). In 2002 and 2004 higher abundances were linked to low salinity and intermediate gradients shown in the south of the archipelago (Fig S.9 and S.10). These results reinforce the theory of bullet tuna spawning occurs in widespread geographic areas, and not only close to the coast and suggest that the location of the main haline front negatively affects the spawning of this species.

Overall results related to bullet tuna point to the fact that besides the avoidance of area near strong surface haline gradients other factors not considered in this study may also be relevant for the site selection for the spawning of this species. It is also relevant that the spatial pattern in relation to the salinity is the opposite to that shown by bluefin tuna, located in areas near the front, suggesting possible avoidance of predators by bullet spawners (Bakun 2013).

The application of seascape metrics derived from salinity and geostrophic currents to characterize the spawning habitat provides new descriptors for environmental variables that improve model quality and predictions. This improvement allows a more precise identification of the relationships between the spatial location of the spawning grounds and the local oceanographic processes. Moreover, our study demonstrates that seascapes must be characterized at specific spatial scales to provide useful information as proposed in previous studies (Steele 1989) and supporting results on terrestrial landscapes (Wiens 1989; Wu & Li 2006) and bottom seascapes (Bostrom *et al.* 2011). Therefore, the relations between the location of spawning sites and the mesoscale oceanographic processes may show to be non significant if seascape metrics are not processed at the right spatial scales.

Seascape ecology is an emerging field generally being applied for the analysis of how benthic habitats pattern in coastal areas drives different aspects of marine species ecology

(Bostrom *et al.* 2011). Techniques are applied following categorical approach where the seascape is composed by a number of patches of different type of habitats (Forman 1995; Cushman *et al.* 2010). However, very little attention had been placed on the techniques and concepts to investigate pelagic seascape ecology due to the complex spatiotemporal dynamics of this system (Pittman *et al.* 2011). Thus, the work here presented sheds new light to modeling spatial distribution and investigating key ecological processes of species highly dependent on the variability of the pelagic environment, as spawning ecology of many of the big tuna species are (Reglero *et al.* 2014). In areas as the Balearic Sea, for which new operational oceanography platforms provides near real time data of hydrography (Tintore *et al.* 2013) and also in combination with remote sensing data (e.g. altimetry, Pascual *et al.* 2013) and modeling (Juza *et al.* 2013) these metrics will improve the species spatial distribution forecast that yet has demonstrated to be effective information for management (Hobday & Hartmann 2006).

In contrast to seascapes, landscape metrics have a long history in terrestrial ecology, and over time have improved. For instance, the effect caused on the habitat analysis derived from the spatial definition of the input habitat maps or the extent of the study area are common studied topics ,(Cushman *et al.* 2008),(Wu 2013). Likewise, calculation of seascape metrics and the final results from their application in ecological studies may be affected by different issues, like the different ways of computing the hydrographic variables and their gradients, or the origin of the input data source as from *in situ* measurements, remote sensing or hydrodynamic models, each with different sources of uncertainty. A relevant question is how seascapes can provide information for other type of species and ecological processes. Addressing all these challenges and developing comparative studies between different data sources, processing methods, species and ecological processes will allow advancing towards the understanding of how seascape metrics can provide information about how ecological processes and oceanography are linked together.

In summary, pelagic seascapes based on gradients and characteristic scales allows improving spatial distribution models and the identification of essential fish habitat of pelagic species. They also provide a tool for analyzing the links between particular ecological processes and local oceanography going far beyond than stochastic models based on just hydrographic parameters as salinity, temperature or geostrophic velocities. As a consequence these metrics will provide an improvement in all the management approaches and tools pending on the capability of models to identify essential habitats as near real-time spatial management based on habitat predictions (Hobday & Hartmann 2006),(Druon *et al.* 2011), pelagic species

distribution from deterministic models (Lehodey *et al.* 2008) or the standardization of larvae indices to assess adult stock (Ingram *et al.* 2008),(Muhling *et al.* 2011).

## 4.5 Acknowledgements

We thank the scientists and crew participating in the TUNIBAL and BLUFIN TUNA cruises. This study was funded by SOCIB and the Instituto Español de Oceanografía under the framework of the BLUEFIN TUNA project and by BALEARES project (CTM2009-07944 MAR). Partial supported from PERSEUS (FP7-287600) EU funded project is also acknowledged. I would like to thank the people involved in this research: , Lorenzo Ciannelli, Alberto Aparicio-Gonzalez, Patricia Reglero, Manuel Hidalgo, Jose Luis López-Jurado, Joaquín Tintoré, Rosa Balbín and Francisco Alemany

## 4.6 Supporting Information

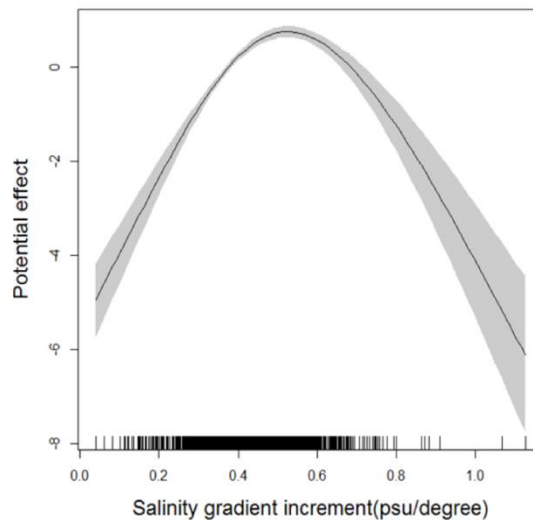


Figure S4.1. Model response of bluefin tuna in relation to salinity gradient processed at 0.75 degrees. Fitted line (solid line) and 95% confidence intervals (grey shaded areas) are shown. Whiskers on the x-axis show the locations of measurements.

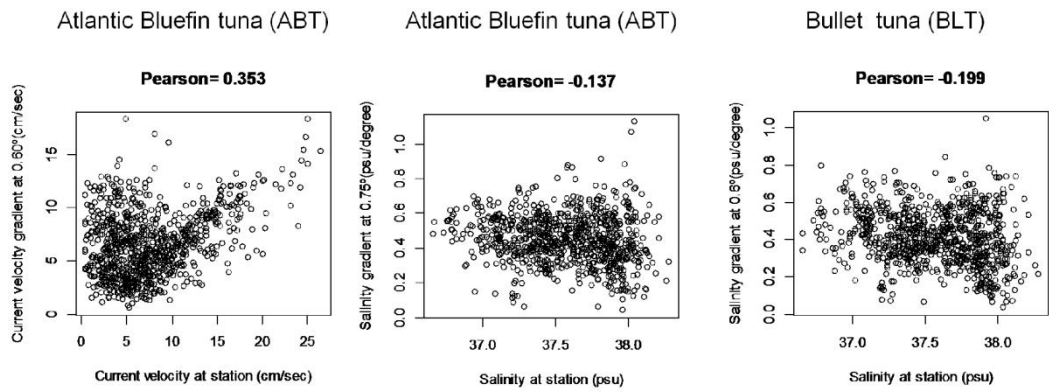


Figure S4.2. Correlation between the gradients at the characteristic scales and the hydrographical variables at the sampled station. A) Current velocity and B) salinity for Atlantic bluefin tuna. C) Salinity for Bullet tuna.

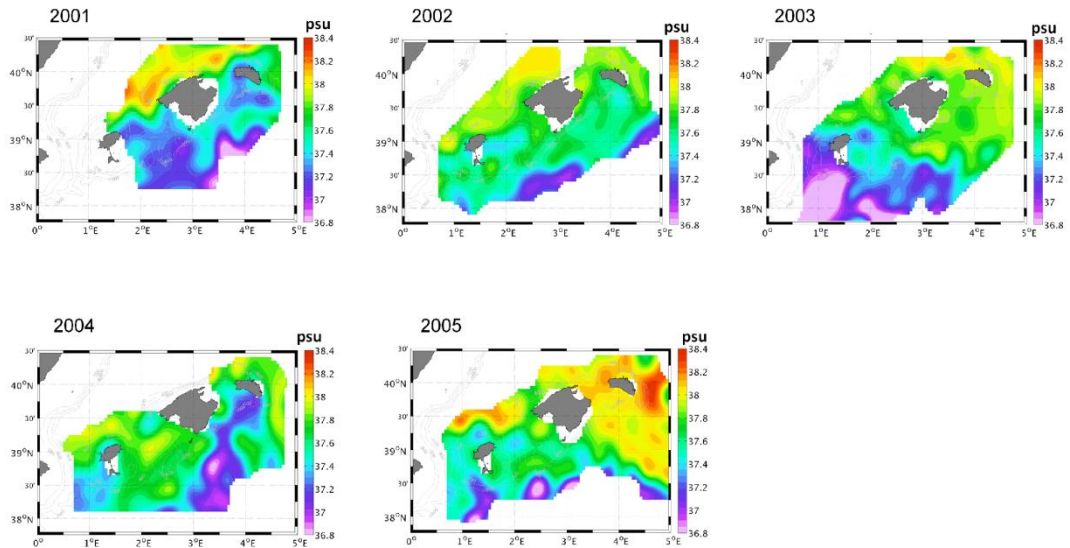


Figure S4.3. Sea surface salinity fields in 2001 to 2005

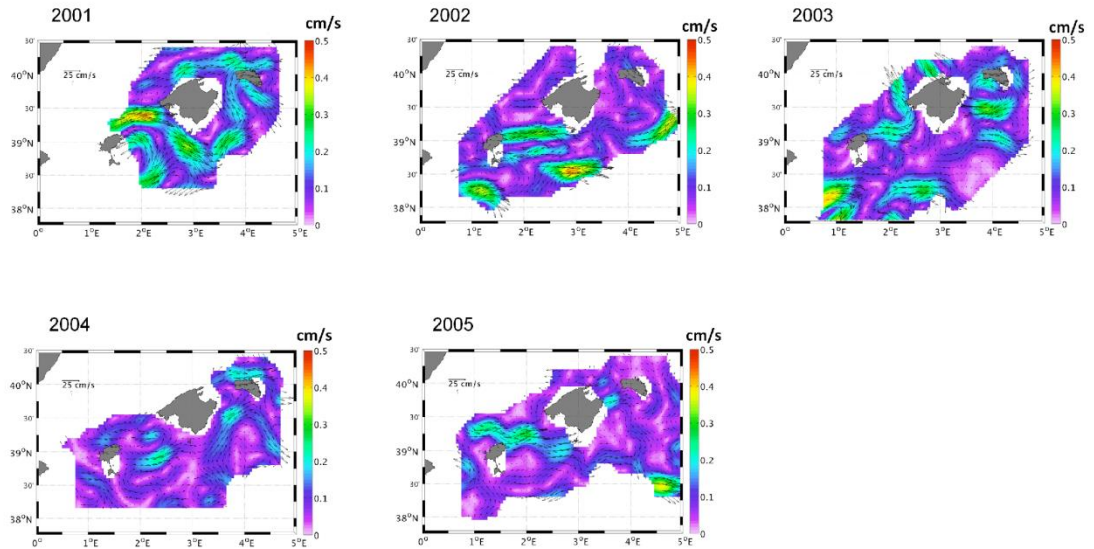


Figure S4.4. Geostrophic velocity at surface in 2001-2005

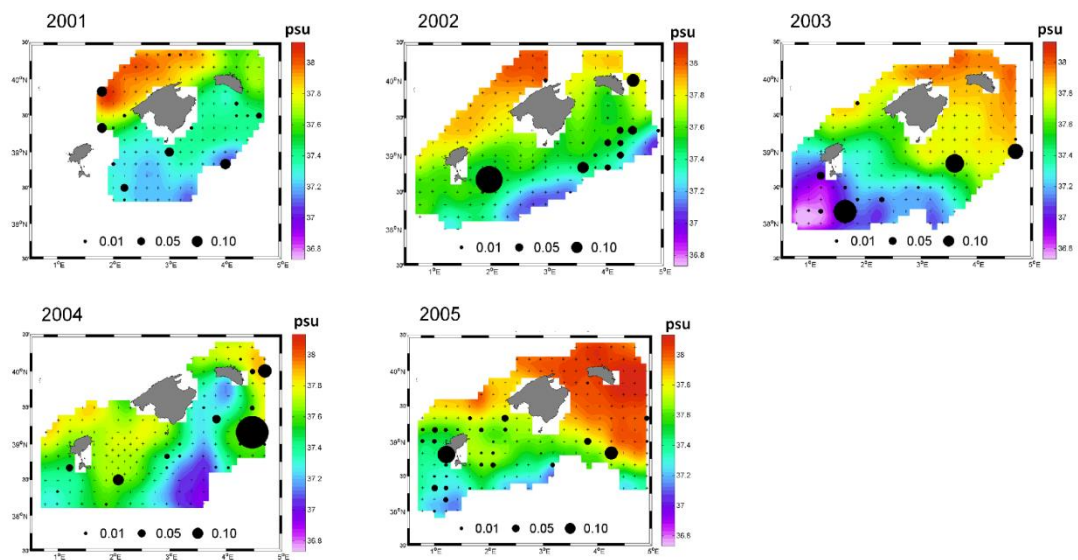


Figure S4.5. Spatial distribution of bluefin tuna (*Thunnus thynnus*) larvae in relation to the salinity mean calculated at its characteristic scale (0.6 degrees). Relative stage-1 larval abundances are shown in the maps as dots.

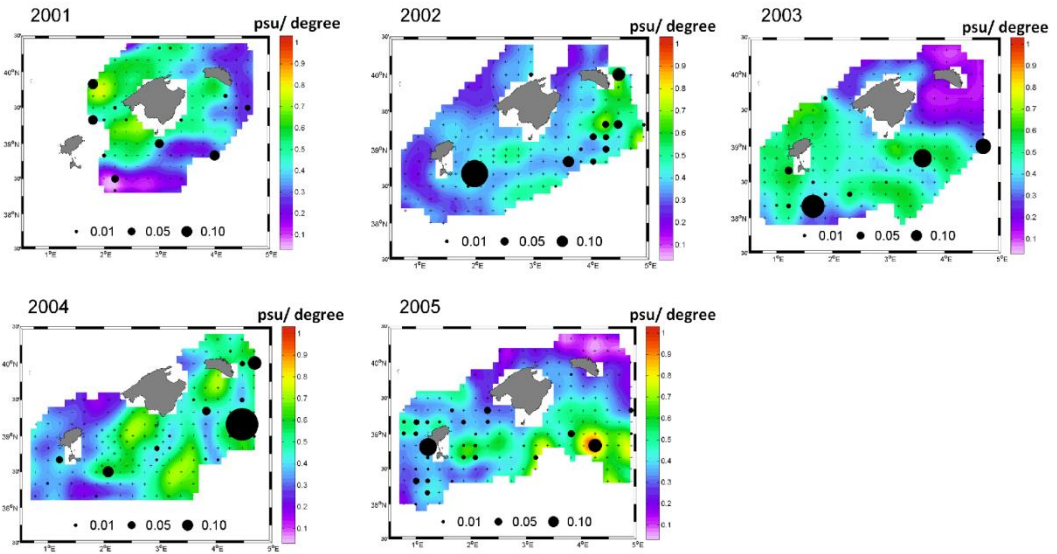


Figure S4.6. Spatial distribution of bluefin tuna (*Thunnus thynnus*) larvae in relation to the salinity gradient calculated at 0.6 degrees. Relative stage-1 larval abundances are shown in the maps as dots.

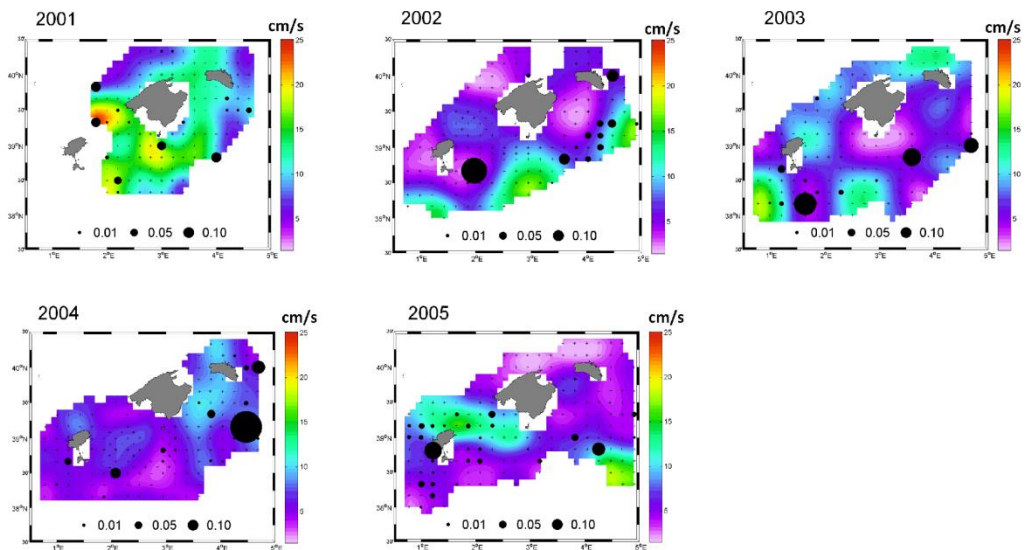


Figure S4.7. Spatial distribution of bluefin tuna (*Thunnus thynnus*) larvae in relation to the geostrophic velocity mean calculated at its characteristic scale (0.6 degrees). Relative stage-1 larval abundances are shown in the maps as dots.

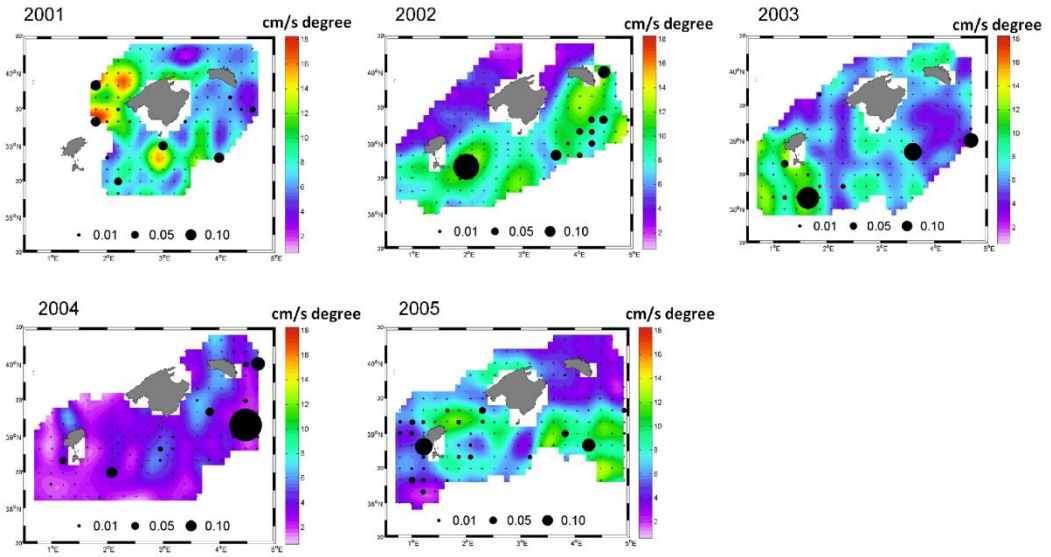


Figure S4.8. Spatial distribution of bluefin tuna (*Thunnus thynnus*) larvae in relation to the geostrophic velocity gradient calculated at the characteristic scale (0.6 degrees). Relative stage-1 larval abundances are shown in the maps as dots.

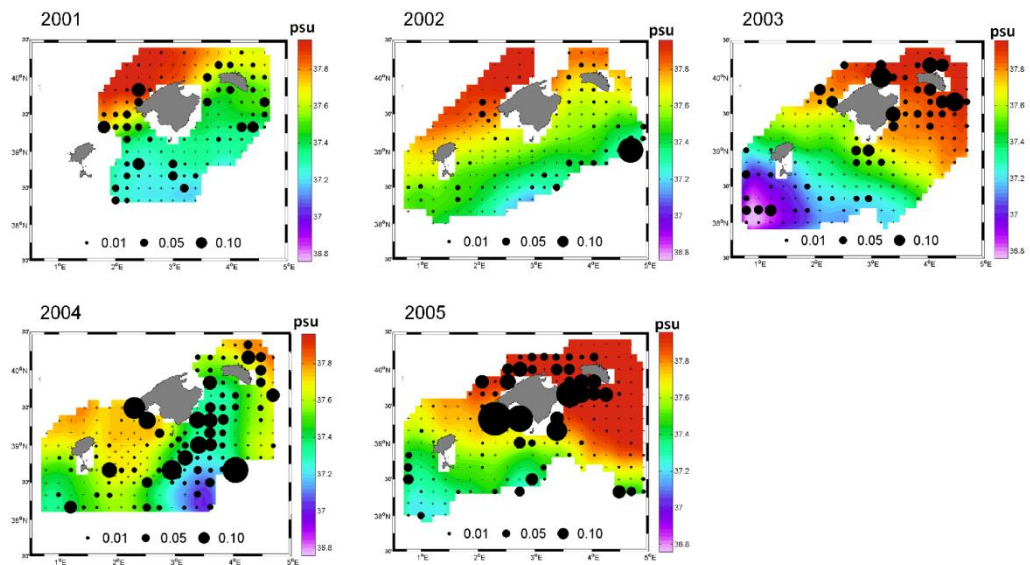


Figure S4.9. Spatial distribution of bullet tuna (*Auxis rochei rochei*) in relation to the salinity mean calculated at 0.75 degrees. Relative stage-1 larval abundances are shown in the maps as dots.

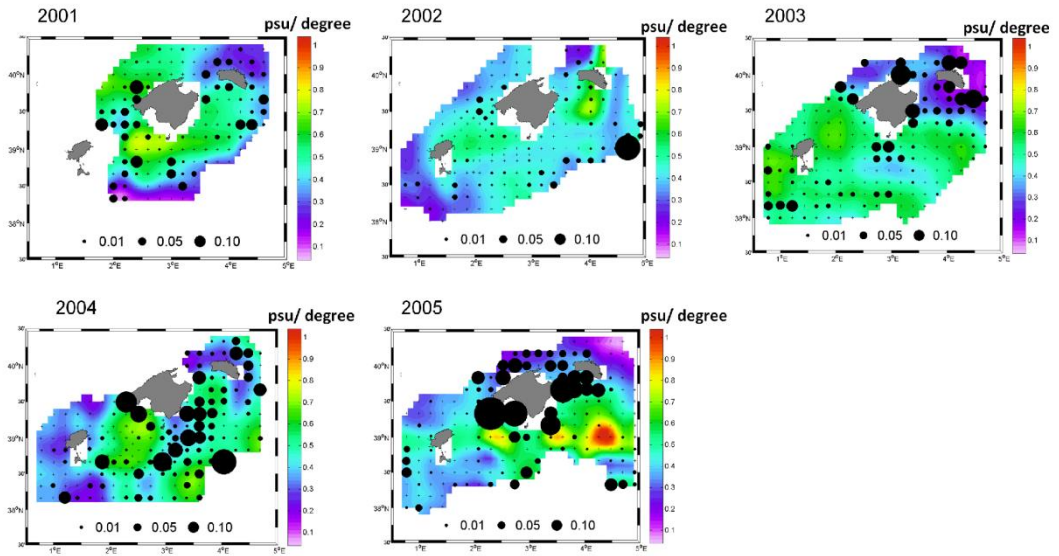


Figure S4.10. Spatial distribution of bullet tuna (*Auxis rochei rochei*) in relation to the salinity gradient calculated at 0.75 degrees. Relative stage-1 larval abundances are shown in the maps as dots.



## **CHAPTER 5**

**Pelagic seascape ecology for  
operational fisheries oceanography:  
modeling and predicting spawning  
distribution of Atlantic bluefin tuna  
(*Thunnus thynnus*, Linnaeus 1758) in  
Western Mediterranean**

## 5.1 Introduction

Essential habitats of pelagic species are strongly linked to dynamic oceanographic processes such as fronts and eddies which vary in space and time (Shillinger *et al.* 2008; Reglero *et al.* 2014; Scales *et al.* 2014). To define these habitats, it is necessary to design environmental descriptors specific to the dynamic component of the oceanographic scenario in areas where the species occur or where relevant processes such as feeding or spawning take place. For example, various studies have used gradients of hydrographic variables to identify the presence of frontal processes (Worm *et al.* 2005; Druon 2010; Louzao *et al.* 2011; Mannocci *et al.* 2013). Recently, Alvarez-Berastegui *et al.* (2014) proposed to define the pelagic seascapes as the combination of means and gradients of particular hydrographic parameters, evaluating how the spatial scale of observation affects our capability to capture important hydrodynamic processes influencing the spawning ecology of two different tuna species. These examples reveal the important role that pelagic seascape ecology plays to understand how the environment affects pelagic species ecology, providing new ways to characterize the spatio-temporal dynamics of pelagic environments.

Operational oceanography is a key tool for advancing towards species habitat modeling with applications to management (Manderson *et al.* 2011; Hobday and Hartog 2014). Modern ocean observing systems combining *in situ* observations, satellite and modeling data are able to provide realistic characterizations of physical oceanographic processes (Rayner 2010; Pascual *et al.* 2013; Tintoré *et al.* 2013a). In this sense, seascape metrics are of special interest when working with remote sensing and hydrodynamic models that offer continuous data at broad extent in space and time. Spatial and temporal resolution provided by these data sources occurs at adequate scales to describe the dynamics of fluid properties at which the marine top predators interact with their habitat. Environmental scenarios obtained from operational data sources can provide baseline information to produce long-term and near real-time forecast of pelagic essential habitats of paramount interest in fisheries management (Hobday and Hartmann 2006) and conservation (Game *et al.* 2009).

The monitoring of bluefin tuna spawning habitats provides a good opportunity to study how the application of operational oceanography and seascape metrics can be relevant to improve current assessment and management of pelagic species. Bluefin tuna is an iconic top predator with a relevant role in pelagic ecosystems (Mather *et al.* 1995; Fromentin and Powers 2005), and supports important fisheries in the Mediterranean and along the North East and West Atlantic coasts (Fromentin 2009). Mounting evidence shows that the spawning ecology

and habitat of this species are strongly linked to mesoscale oceanographic processes (Reglero *et al.* 2012; Muhling *et al.* 2013, Alemany *et al.* 2010, Alvarez-Berastegui *et al.* 2014), and therefore, operational oceanography provides a new potential tool to characterize and track these habitats. Spawning habitat models inferred from *in situ* data have improved the standardization of spawning biomass estimates of bluefin tuna based on larval abundance indices (Ingram *et al.* 2013). Thus, characterizing and monitoring spawning habitats from operational oceanography products would provide near real time information for larval sampling design and for larval abundance indices calculation. Besides, this type of information would facilitate the application of new management approaches based on spatial restrictions that could reduce bluefin tuna bycatch in the Mediterranean, such as those measures recently adopted in the Gulf of Mexico (US-DOC/NOAA/NMFS 2014). Moreover, dynamic habitat mapping could be used to manage pelagic marine protected areas within an adaptive framework, where spatial limits of closure areas may change (Hobday *et al.* 2010).

In this study, we applied a pelagic seascape approach based exclusively on operational oceanographic information to model the spawning habitats of Atlantic bluefin tuna in the Western Mediterranean Sea. Densities of early larval stages of bluefin tuna, collected in proximity to a principal spawning region in the Balearic Sea, were used as a proxy for spawning locations (Mather *et al.* 1995; García *et al.* 2005a). Seascape metrics, used as input in the modeling process, were selected and included on the basis of known dependencies of the bluefin tuna spawning ecology with local mesoscale oceanography from previous studies. The results will allow predicting the spatial location of bluefin tuna spawning areas and provide insights about the spawning ecology of this species.

We develop a specific cross-validation approach to assess the predictive capability of the habitat models. We also propose specific techniques to address possible biases in the predictions derived from displacements of the oceanographic features identified from hydrodynamic models or remote sensing. This work aims to develop and provide operational fisheries oceanography products that directly apply to current assessment and conservation of Atlantic bluefin tuna and other pelagic marine species of interest.

## 5.2 Methods

### 5.2.1 *In situ* data acquisition

Atlantic bluefin tuna (BFT) larvae were collected during five systematic oceanographic campaigns carried on in June-July of 2001-2005. A regular grid of 10 x 10 nautical mile were sampled each year covering the area between 37.85°- 40.35° N and 0.77° -4.91° E, (280x362 km). At each sampling location, BFT larvae were collected with a bongo net of 60 cm mouth diameter, towed till a depth of 70 m, or from 5 m above the bottom at coastal stations, to the surface maintaining the vessel speed at 2 knots. An average of 162 stations were sampled yearly around the Balearic archipelago (Figure 5.1), additional information about field campaigns is provided in supporting information (Table S5.1). The volume of water filtered was measured with flow-meters located at the center of the net. Plankton samples were preserved with 4% formalin buffered with borax. Further, tuna larvae were identified to the species level and measured in standard length.

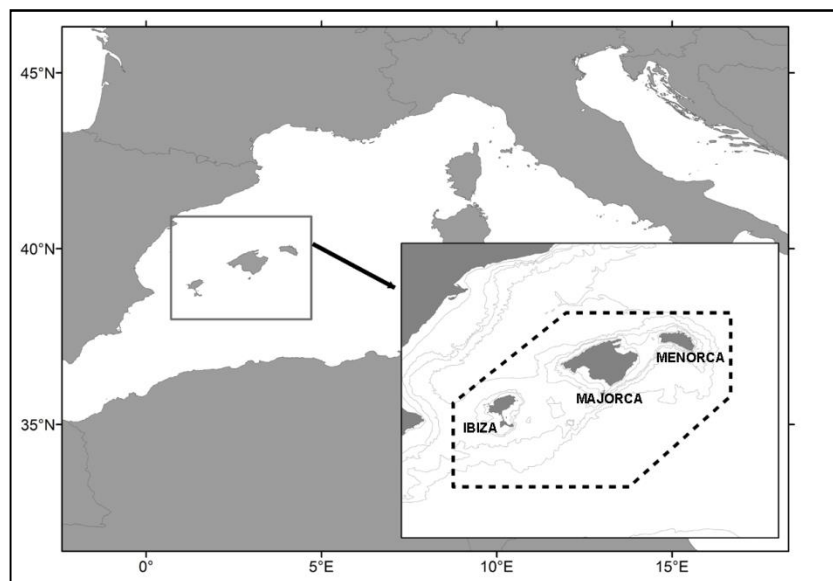


Figure 5.1. Geographic location of the Balearic Sea and location of sampling area (dashed line).

Number of bluefin tuna larvae belonging to the yolk sac and preflexion stages (< 4.5 mm) were calculated and used as proxy of spawning locations as in previous research in the study area (Reglero *et al.* 2012; Alvarez-Berastegui *et al.* 2014). The mean and maximum age of larvae below 4.5 mm are 6 and 11 days old respectively, calculated considering bluefin tuna larval growth rates (F.de la Gándara *et al.* 2013) and hatching times (Gordoa and Carreras 2014). Considering the mean age of 6 days, drift distances from the actual spawning location

are below 25 kilometers (around 1.4 times the sampling station distance), and for the maximum age, the drift distances are around 46 kilometers (around 2.6 times the sampling station distances). These values have been calculated following methods in Reglero *et al.* (2013).

## 5.2.2 Identification of operational oceanography data sources

Potential explanatory variables providing information on the location of BFT spawning habitats were identified from previous studies in the western Mediterranean. These studies found that sea surface temperature (SST), salinity (SSS) and surface current velocities are the main hydrographic variables driving location of spawning areas (Alemany *et al.* 2010; Reglero *et al.* 2012). In addition, BFT spawning mostly occur in areas with low chlorophyll a (CHL) (Muhling *et al.* 2011; Muhling *et al.* 2013). A more recent study shows that spatial gradients of hydrographic variables capture the location of oceanographic processes at mesoscale level, such as fronts, and significantly improves the identification of BFT spawning locations (Álvarez-Berastegui *et al.* 2014). Therefore, we selected potential operational oceanography data products (ODPs) providing information related to temperature, salinity, chlorophyll-a and currents, to compute pelagic seascape metrics (i.e. environmental covariates). Acronyms and additional information about selected ODPs datasets are provided in supporting information (Table S5.2). Additional considerations for the selection of operational products include:

- I. Being freely accessible.
- II. Having historical data for the sampling periods.
- III. Having near real-time data accessible online.
- IV. Having a similar or better spatial resolution than the *in situ* data set (18x18 km<sup>2</sup> for the BLT survey in the Balearic Sea).
- V. Having a minimum temporal resolution of a week, in relation to the temporal persistence of mesoscale structure in the area (Bouffard *et al.* 2014)
- VI. Availability of data product quality information for the study area.

## 5.2.3 Processing environmental explanatory variables from selected ODPs

A total of six seascape metrics were processed from the different ODPs datasets (see Table 5.1 for seascape metric names, groups, acronyms and spatial and temporal resolution).

All metrics were estimated at the sea surface layers, where spawning occurs (Aranda *et al.* 2013) and larvae are found (Torres *et al.* 2011). The SST from the Mediterranean Forecasting System hydrodynamic model dataset (ODP data set MFS-SST; (Tonani M. *et al.* 2014)) was selected rather than satellite data (ODP data set MODIS-SST) for the computation of the temperature descriptors. MFS-SST presents the advantage of being cloud-free compared to MODIS-SST. Moreover, MFS-SST provide very realistic estimates of the SST over the Western Mediterranean Sea (Juza *et al.* 2015). Two SST seascape metrics were computed from this model data set: i) the spatial averaged SST (SSTa, Table 5.1) providing information about the mean sea surface water temperature within an area of 0.5x0.5 degrees around the sampling location and, ii) the spatial averaged SST increment during the previous 15 days (iSSTa, Table 5.1). Kinetic energy (KE, Table 5.1) derived from sea surface height from satellite altimetry data (Pascual *et al.* 2007) was selected as proxy for sea surface currents (ODP data set AVISO-SSA). A Mediterranean specific mean dynamic topography (Rio *et al.* 2014) was used for the calculation of the absolute sea surface height from the satellite observed anomalies. A KE frontal index seascape (KEfi, table 5.1), providing information about the spatial variation of KE and the potential location of sea surface current front, was calculated as the rate of change (computed as the slope) of the KE at sampling locations using a spatial window of 0.6x0.6 degrees. This spatial scale filters was proposed in previous analyses in the area (Álvarez-Berastegui *et al.* 2014).

Table 5.1: Seascape metrics (environmental covariates) processed from operational data sources

Variable group	Variable name	Variable acronym	Spatial resolution (degree/km)	Temporal Resolution (days)
<b>SST related variables</b>	Spatial averaged sea surface temperature	SSTa	0.5 / 55	1
	Spatial averaged increment of sea surface in the 15 previous days	iSSTa	0.5 / 55	15
<b>KE related variables</b>	Kinetic energy	KE	0.125 / 12.5	1
	Kinetic energy frontal index	KEfi	0.5 / 55	1
<b>CHLa related variables</b>	Spatial averaged sea surface chlorophyll_a	CHLa	0.5 / 55	7
	Chlorophyll-a frontal index	CHLafiCHLafi	0.5 / 55	7

Daily images of chlorophyll-a data from multisatellite images (ODP data set RS-CHL) processed with a Mediterranean specific ocean color algorithms (Volpe *et al.* 2007) were used to compute temporal composites over 7- day periods to reduce the percentage of pixels with no data due to clouds. The spatial averaged sea surface chlorophyll-a seascape (CHLa, Table 5.1) was calculated as the averaged value of weekly-composite RS-CHL in an area of 0.5

degrees around each sampling station. Composites were also used to compute the frontal index at 0.5 degrees as a proxy of the location of well-established chlorophyll-*a* fronts (CHLafi, Table 5.1) as for KE. Note that SSS from the ODP data set MFS-SSS (Table S5.2) was excluded for analyses because recent research demonstrated that this parameter was not properly represented by the model in the study area (Juza *et al.* 2015).

Examples of the pelagic seascape metrics used as environmental descriptors processed from the operational data sources are provided in Figure 5.2.

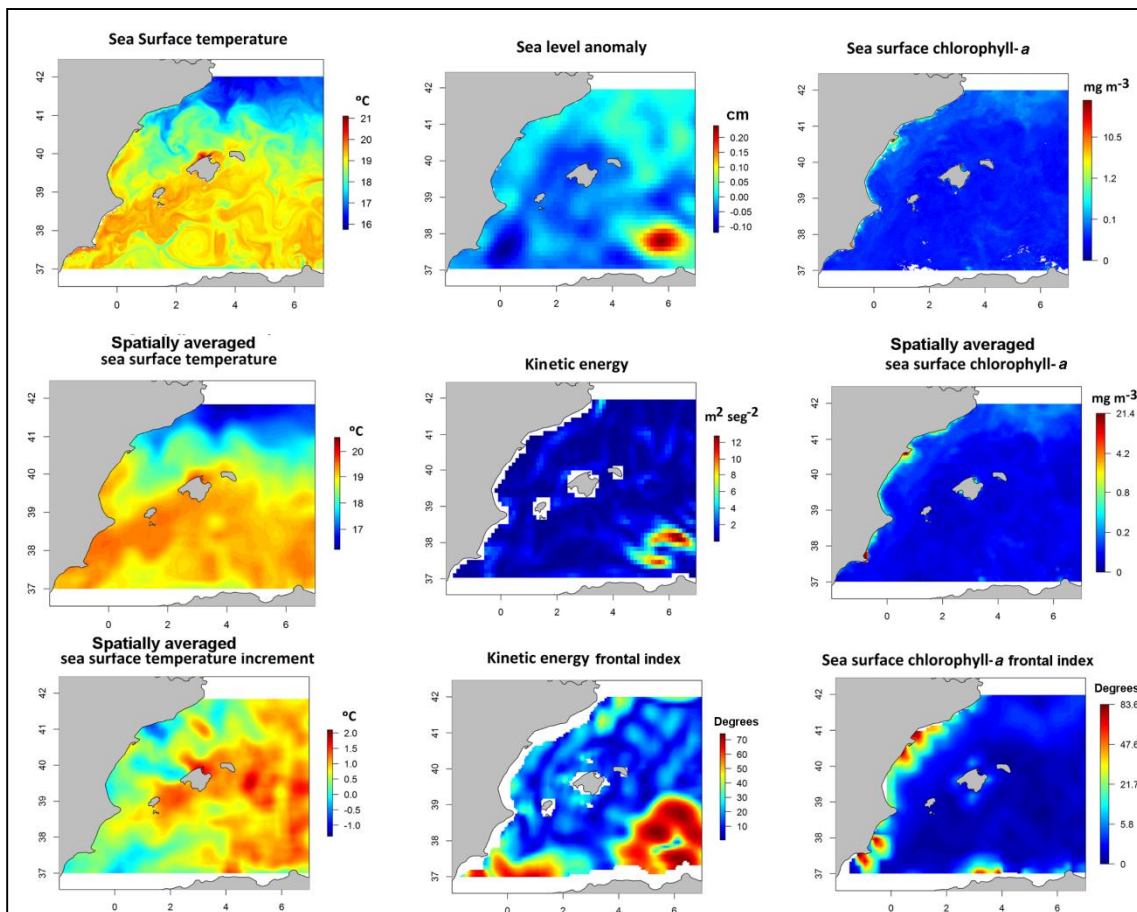


Figure 5.2. Examples of operational data products and seascape metrics for a specific date (2014/05/29) of original products (top row), spatially averaged derived seascapes (middle row) and heterogeneity estimate (bottom row). Sea surface temperature from hydrodynamic model (SST), derived zonal sea surface temperature (SSTa) and its increment in 15 days (iSSTa) (left column). Sea surface anomaly, derived Kinetic energy (KE) and kinetic energy frontal index (KEfi) (middle column). Sea surface chlorophyll-*a*, derived spatially averaged chlorophyll-*a* (CHLa) and the frontal index (CHLafi) (right column).

## 5.2.4 Modeling larvae abundances

The relationship between environmental covariates and larval abundances collected along the five years was modeled using nonparametric regression (generalized additive models, GAMs, Wood 2006). All GAMs included a bivariate smoother combining latitude and longitude in the formulation, the year factor accounting for the inter-annual effect in the abundances and the diurnal variation effect in the catchability. Four different model configurations were compared to test the larval abundance:

I) A model with no environmental covariates (equation 1), referred as *reference model*

Equation 1 :  $LA = \text{offset}(\log(m3)) + \text{factor}(\text{year}) + \text{sm}_1(\text{long, lat}) + \text{sm}_2(\text{hour});$

Where LA=larvae abundances below 4.5 mm; sm= Smoothing functions;

II) A model where all environmental variables were included as additive (equation 2).

Equation 2:  $LA = \text{offset}(\log(m3)) + \text{factor}(\text{year}) + \text{sm}_1(\text{long, lat}) + \text{sm}_2(\text{hour}) + \text{sm}_3(\text{SSTa}) + \text{sm}_4(\text{iSSTa}) + \text{sm}_5(\text{KE}) + \text{sm}_6(\text{KEfi}) + \text{sm}_7(\text{CHLa}) + \text{sm}_8(\text{CHLafi})$

III) A model with all variables from the same group included as interactive (see equation 4).

Equation 3:  $LA = \text{offset}(\log(m3)) + \text{factor}(\text{year}) + \text{sm}_1(\text{long, lat}) + \text{sm}_2(\text{hour}) + \text{sm}_3(\text{SSTa, iSSTa}) + \text{sm}_4(\text{KE, KEfi}) + \text{sm}_5(\text{CHLa, CHLafi})$

IV) A model resulting from the combination of ii) and iii) (one or two groups included as interactive while other groups are included as additive).

For all possible combinations (nine in total, see Table 5.2), a backward selection process was applied to remove variables with no significant effect ( $p > 0.05$ ). In order to restrict potential over fitting in the models the number of knots for each environmental covariate was limited up to 3, for univariate additive terms, and up to 9, for bivariate (interactive) terms.

Exploration of the model performance after non-significant variable removal was based on the maximization of the explained deviances and minimization of the Akaike Criterion Index (AIC, (Akaike 1981). The AIC parameter is a trade-off between the model goodness of fit and the model complexity. AIC values from different models were compared by calculation of the delta AIC ( $\Delta\text{AIC}$ ), which is the difference between model AIC and minimum AIC found among all models. A value of  $\Delta\text{AIC}$  equal to 2.5 was selected as threshold for no relevant model differences. Therefore, models with  $\Delta\text{AIC}$  lower than this threshold were considered to be of



equal quality (Hilbe 2011) and selected as potential candidates for calculation of the Spawning Habitat Quality.

### **5.2.5 Definition and calculation of the Spawning habitat quality (SHQ)**

The position of mesoscale oceanographic features, such as fronts, deduced from remote sensing images or modeling techniques is often found to be inaccurate when compared to the *in situ* observed positions (Bouffard *et al.* 2014; Bricheno *et al.* 2014). In these cases, the correlation coefficients between observed and predicted maps of larval distributions in relation to oceanographic features would be low, even if the models were able to identify spawning areas at coarse spatial resolution. To solve this limitation, each predicted map of larval abundances obtained from the GAMs was spatially smoothed at the spatial scale that optimized the correlation between observed and predicted values. The larval abundances at the optimal spatial scale were defined as the SHQ. The best spatial scale for the smoothing process was assessed with scalogram plots that showed the correlation between observed and predicted SHQ calculated at 9 different spatial scales (i.e. from 7 to 75 km radius area). The Spearman coefficients were computed between the maps of observed SHQ and the predicted SHQ smoothed at each of these 9 spatial scales. Predicted SHQ was computed with all different candidate GAM models. All computations were coded in R software (R Development Core Team 2008), the raster processing functions were computed with the raster package (Hijmans & van Etten 2012). A methodological graphical scheme showing the SHQ calculation is provided as supporting information (Figure S5.1).

### **5.2.6 Evaluating the SHQ predictions through an inter-annual cross-validation approach and model selection**

Maps of SHQ obtained from models fitted with all the five years available from all candidate models (thereafter 'fitted SHQ') were compared to the observed SHQ using Spearman correlations. Both fitted and observed SHQ were computed at the same spatial resolution depicted from the scalograms. This correlation provided information of the SHQ on a yearly basis.

To assess the capability of different models to predict the SHQ, we performed an inter-annual cross-validation by splitting the data into a validation and training datasets. The former

was composed by one-year data extracted from the five years available and the training dataset was composed by the four remaining years. Candidate GAMs were adjusted with the training dataset. SHQ were predicted with these models (thereafter 'predicted SHQ') and compared with observed SHQ values. This process runs for each year available and, therefore, five predictions were estimated for each candidate model.

## 5.3 Results

### 5.3.1 Performance of the different processed models

General additive models presented better performance indicators (explained deviances and AICs) when environmental covariates, either as a combination of additive or interactive terms, were included in the model design (see model outputs in Table 5.2). All these models showed similar values for the two performance indicators. The model including only additive variables (Madd in table 5.2) showed the lowest AIC. During the variable selection process this model retained iSSTa, KEfi and CHLa, while SSTa, KE and CHLafi were not significant. These results demonstrated the high relevance of environmental parameters related to the spatio-temporal variability of SST and KE. However, AIC of Madd did not differ from the other models and, thus, models with a  $\Delta$ AIC below 2.5 (models Madd, comb 1, comb 2, comb 4, comb 5, comb 6 in Table 5.2), including different interactions among covariates, were selected as potential model candidates for calculation of the SHQ. Therefore, scalograms were computed for each model in order to depict the spatial scale at which the cross-validation should be processed.

### 5.3.2 Scalogram processing and spatial scale selection for SHQ definition

The correlations between observed and fitted SHQ (adjusted from models including the five years available) show similar trends among the six models selected. That is, correlations increased with increasing spatial scales (Figure 5.3). The scale at which the correlation curve became asymptotic is related to the spatial lag between observed and modeled spawning areas. The lowest is the scale the closer the modeled and observed spawning locations are. Selecting high spatial scales improved the correlations between observed and modeled data but reduced the spatial resolution of the final products. After examination of the processed

scalograms, we selected 31.5 km as the best radius distance for the spatial smoothing defining the SHQ computation from now onwards. At this spatial scale the slope of the correlograms decreased for all years except 2003 and the curve became asymptotic for 2004 and 2005 (Figure 5.3).

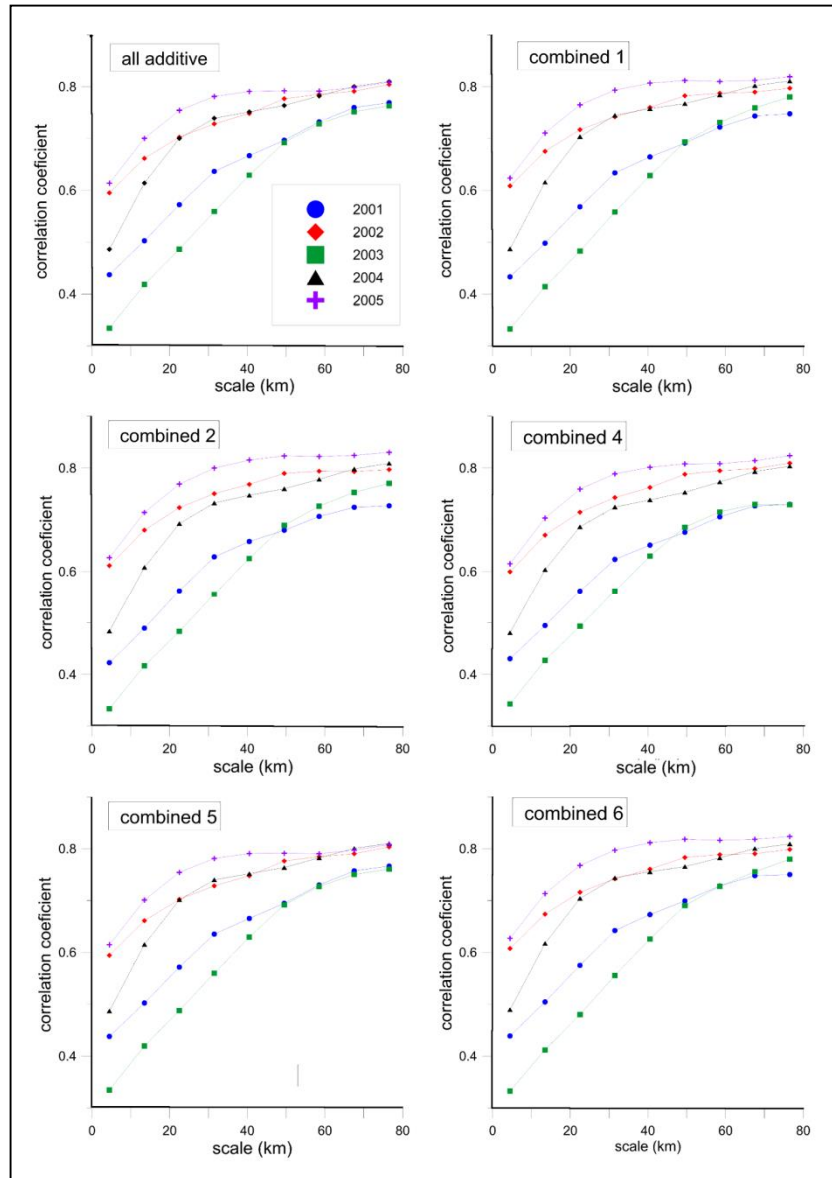


Figure 5.3. Scalograms computed for each selected model and year. See Table 5.2 for model name and structure definition.

Table 5.2: Processed models, environmental variables included and performance indicators.

<b>Model ID</b>	<b>Model characteristics</b>	<b>Model formulation N=805</b>	<b>Deviance explained(%)</b>	<b>AIC</b>	<b>ΔAIC</b>
<b>M1</b>	No environmental covariates included	(long, lat) + (hour) +year	40.7	1047.71	51.4
<b>Madd</b>	All environmental variables as additive	iSSTa+KEfi+CHLa	52.4	995.60	0
<b>Mint</b>	Environmental variables as interactive	(SSTa,iSSTa)+ (KE,KEfi)+ (CHLa,CHLafi)	52.6	998.88	3.2
<b>Comb 1</b>	Chla variables as additive, SST and KE as interactive	(SSTa,,iSSTa)+ (KE,KEfi)+ CHLa	52.6	997.59	1.9
<b>Comb 2</b>	KE variables as additive, SST and CHLa as interactive	(SSTa,,iSSTa)+ KEfi+ (CHLa,CHLafi)	52.7	997.04	1.4
<b>Comb 3</b>	SST variables as additive, KE and CHLa as interactive	iSSTa+ (KE,KEfi)+ (CHLa,CHLafi)	52.6	998.32	2.7
<b>Comb 4</b>	SST and KE variables as additive, CHLa as interactive	iSSTa+ KEfi+ (CHLa,CHLafi)	52.6	996.39	0.8
<b>Comb 5</b>	SST and Chla variables as additive, KE as interactive	iSSTa+ (KE,KEfi)+ CHLa	52.5	997.52	1.9
<b>Comb 6</b>	SST variables as interactive, KE and CHLa as additive	(SSTa,iSSTa)+ KEfi+ CHLa	52.7	995.73	0.1

### 5.3.3 Best operational model selection

The correlations between observed and fitted SHQ varied among years (see table 5.3) but they presented similar values among models, with correlation values above 0.5. For all models the year 2005 presented the best correlation and 2003 the worst (i.e. maximum correlation coefficient of 0.8 and minimum 0.55 respectively). Results from the evaluation of the real prediction capabilities through the cross-validation approach are presented in table 5.4. Correlation coefficients between observed and predicted SHQ from all selected models performed significantly better than the M1 model (model with no environmental covariates, Table 5.4). This result demonstrated the relevance of the proposed seascapes for predicting spawning habitats. Although all candidate models showed similar cross-correlation coefficients, the model containing interactive terms for SST and KE related variables and additive for CHLa showed best predictive capabilities with correlation values above 0.5 for four out of five years predicted (model combined-1 in Table 5.4). Therefore, the processing scheme that provided the best predictive performance was defined as the model “combined 1” and spatially smoothed in the radius distance of 31.5 km as previously depicted.

Table 5.3: Spearman correlation of observed and fitted SHQ.

Year	M1	All additive	Comb.1	Comb.2	Comb.4	Comb.5	Comb.6
2001	0.62	0.64	0.63	0.63	0.62	0.63	0.64
2002	0.57	0.73	0.74	0.75	0.74	0.73	0.74
2003	0.50	0.56	0.56	0.56	0.56	0.55	0.55
2004	0.51	0.74	0.75	0.73	0.72	0.74	0.74
2005	0.51	0.78	0.79	0.80	0.79	0.78	0.79

Table 5.4: Spearman correlation of observed and predicted SHQ.

Year	M1	All additive	Comb.1	Comb.2	Comb.4	Comb.5	Comb.6
2001	0.42	0.47	0.51	0.44	0.43	0.47	0.48
2002	0.25	0.62	0.62	0.63	0.63	0.62	0.63
2003	0.18	0.28	0.26	0.26	0.26	0.27	0.27
2004	0.40	0.70	0.69	0.69	0.68	0.69	0.70
2005	0.28	0.67	0.73	0.74	0.67	0.66	0.73

### 5.3.4 Relation between spawning habitat quality and environmental information

The effect of each covariate on the SHQ analyzed from the predictive model (model combined 1) is shown in Figure 5.4. The latitude-longitude map displays the mean spatial distribution along the five years with the lowest values found at the South-East of the study area (Figure 5.4.A). The main spawning habitats were located South-West Ibiza Island and West of the archipelago (Figure 5.4.A). The bivariate plot of the interactive effects of SSTa-iSSTa indicates a positive effect of both variables (Figure 5.4B).

For the response of the KE-KEfi interaction, the bivariate plot indicates a positive effect of KEfi and a low effect of KE (Figure 5.4.C). The partial effect for the diurnal variation shows changes in catchability along the day, with maximum values at midnight and noon (Figure 5.4.D). The effect associated to CHLa presents a negative trend, indicating higher probability of finding spawning events in areas of low CHLa values (Figure 5.4.E).

Responses obtained for the five models processed during the cross-validation process (Figure S5.2) showed that the patterns associated to the latitude-longitude, hour and CHLa are consistent throughout the years. The effect of the interaction SSTa-iSSTa was also consistent, with exception of the cross-validation on removing 2003. By contrast, the low effect of the KE covariate in the interaction term KE-KEfi may change when one particular year is removed from the training dataset. This suggests that the interaction terms may play a relevant role in explained deviances for years with particular conditions.

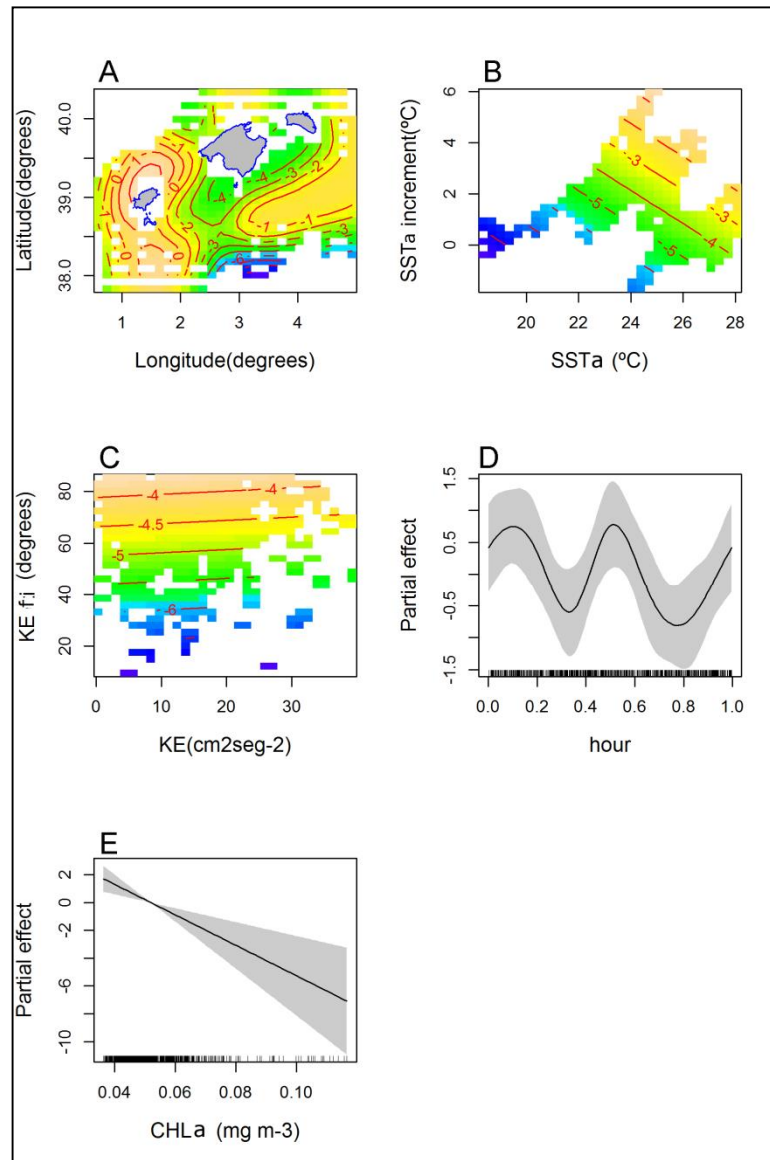


Figure 5.4. Environmental covariate response plots for the selected general additive model. Isolines indicate partial effect on the larval abundances. Pink-yellow colors indicate higher abundances and green-blue indicate lower abundances.

### 5.3.5 Comparison of ‘observed’, ‘fitted’ and ‘predicted’ Spawning Habitat Quality (SHQ)

Spatial distribution observed, fitted and predicted SHQ for the five years of analyses are presented in Figure 5.5. The SHQ computed from observed data (observed SHQ) shows a patchy pattern. In general, observed spawning areas were mainly located at the East and South-West of the archipelago (Figures 5A, from 2001 to 2005). The most intense spawning habitats for each year, excluding 2003, occurred in the western part of the study area. The survey in 2005 shows the patchiest distributions, with spawning habitats distributed along the

southern area of the archipelago and no spawning habitats at the north (Figure 5.4.A for the year 2005). The spatial distribution of fitted and predicted SHQ are similar, and they both were able to identify the patchy distribution and the persistence of the spawning in the western part of the archipelago (Figures 5.B, 5.C years 2001-2005). Predictions obtained from the cross-validations located high values of SHQ within a distance range of 60 km from observed maximums for all years except for 2005 (Figure 5.5.C of 2005). For 2005, maximum values of SHQ were predicted at the South of Ibiza Island, that was an important spawning area for that year, but maximum observed SHQ values occurred at South-East. Graphical outputs and correlation coefficients from Tables 3 and 4 demonstrated that GAM fitted from operational data sources were able to model and predict spatial patterns of spawning areas.

## 5.4 Discussion

This study shows how pelagic seascapes from operational oceanography data sources can provide new insights on the environmental cues driving the spatial distribution of bluefin tuna spawning areas. Modeling and predicting the spatial distribution of spawning areas was possible by applying pelagic seascape descriptors that provided information on the spatio-temporal variability of sea surface temperature, geostrophic velocities and chlorophyll-a. Thus, our results provide new avenues of applied research, particularly in the emerging field of operational fisheries oceanography in which near-real time operational products from integrated ocean observing systems will serve as tools for the 21st century fisheries management (Berx et al. 2011; Manderson et al. 2011; Hobday et al. 2014).

Previous studies have already shown that mesoscale oceanography affects spawning of bluefin tuna around the Balearic Sea (Alemany et al. 2010; Reglero et al. 2012; Muhling et al. 2013). In this study, we found that the temporal evolution of sea surface temperature and the spatial variability of kinetic energy were more relevant to identify patterns in larval distribution than just the absolute values. These findings confirm previous studies investigating how the combination of means and gradients of a oceanographic variable improves our capability to investigate species-environment relationships (Alvarez-Berastegui et al. 2014) and highlight the importance of designing proper pelagic seascape metrics holding information about the dynamic behavior of the oceanographic processes. Sea surface temperature during the beginning of the spawning season is associated to the fast development of adult gonads, which are still undeveloped when entering in the Mediterranean (Medina et al. 2002). High means of sea surface temperature is also a requisite for eggs and larval growth and survival. These two



processes have driven evolutionary constraints for the location of spawning areas (Ciannelli et al. 2015). The combination of the mean and increment of sea surface temperature measured from operational data sources provides a good proxy for capturing complementary ecological and physiological processes affecting habitat preference of bluefin tuna.

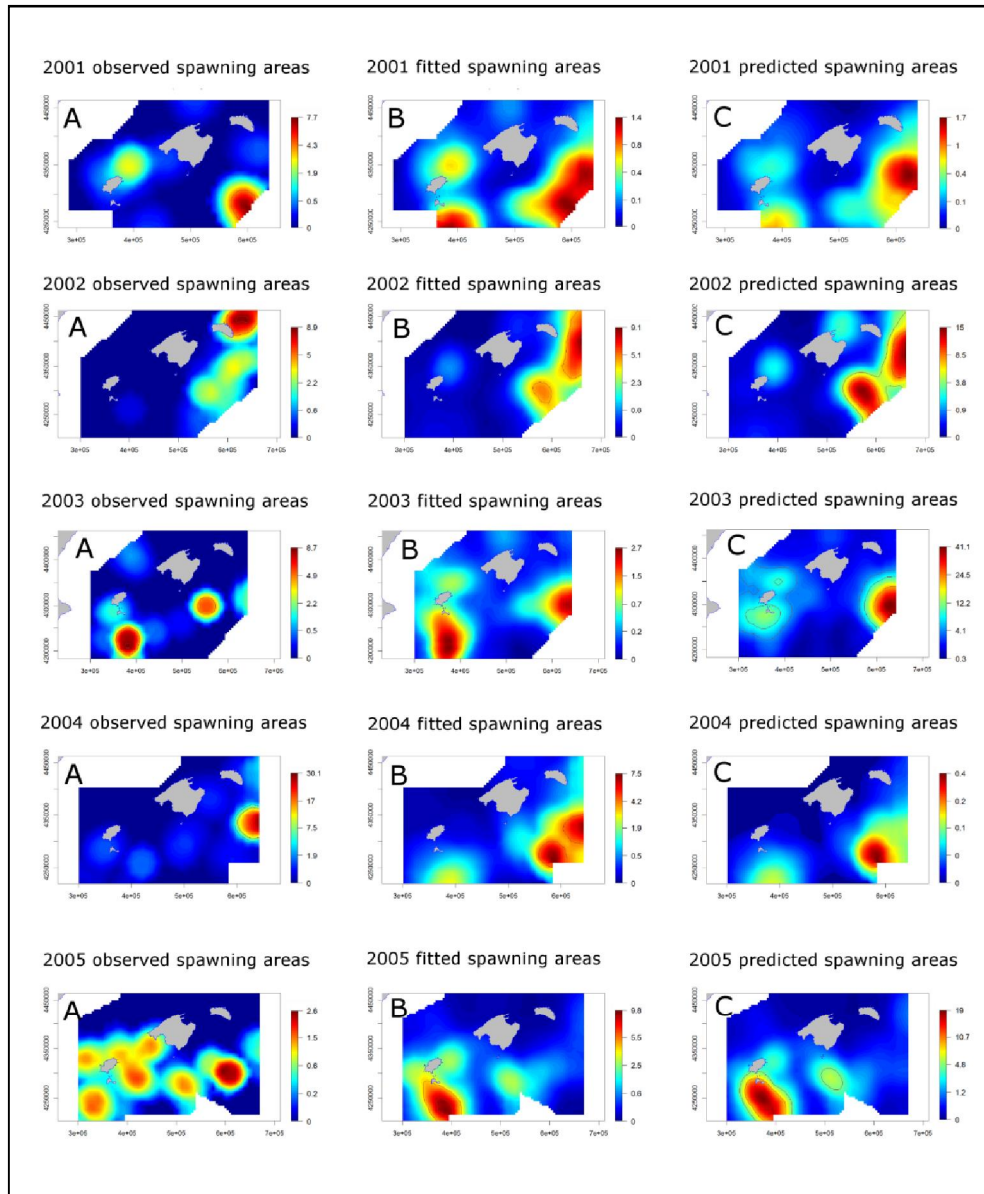


Figure 5.5. Yearly spatial distribution of observed (left), modeled (center) and cross-predicted (right) Spawning Habitat Quality (SHQ) from 2001 to 2005 (from first to fifth row)

The location of hot-spots of larval abundances (yolk sac and preflexion stages) used as proxy for spawning habitats, also depends on the interaction of kinetic energy and its spatial variability used as proxy of frontal processes. The main frontal area is formed when recent Atlantic waters enter in the Mediterranean through the strait of Gibraltar and reach the

Balearic Sea, mixing with saltier resident waters (Balbín et al. 2014). Previous studies show that the spawning occurs near this front (Alemany et al. 2010; Reglero et al. 2012). Nevertheless, the fact that higher larval abundances are associated with low-medium kinetic energy values near the front may indicate the retention role of the front rather than a spawning area. Recent studies analyzing the spatial distribution of eggs, identified from genetic analysis, also found a relation between the eggs abundances and the location of the main salinity front (Reglero P., personal communication). This result, along with the relative short drifts trajectories found for yolk sac and preflexion larval stages support the hypothesis that spawning is associated to mesoscale activity in the area.

The location of spawning areas has been also associated to salinity values associated to the frontal areas. Salinity is one of the most relevant environmental variable explaining the spatial distribution of bluefin tuna larvae in the Balearic Sea (Alemany et al. 2010; Reglero et al. 2012). This relation has been found also in other areas in the Mediterranean (Koched et al. 2013) and in the Gulf of Mexico (Muhling et al. 2010; Muhling et al. 2013). Nevertheless, whether bluefin tuna adults detect salinity gradients or whether they detect other processes associated to the front is not resolved. Improving hydrodynamic models to reach required quality of sea surface salinity projections will give the possibility of developing new relevant environmental descriptors for the spawning habitat forecast and to disentangle different causes associated to the dependency of the spawning ecology with the mesoscale activity. This is an important challenge for operational oceanography in the Mediterranean Sea (Juza et al. 2015).

The fact that chlorophyll-a from satellite plays an important role in the models developed in this study may be also associated to the lack of environmental variables providing information about the different water masses, such as salinity. Within the study area and for some of the studied years, lower values of chlorophyll-a were associated to the fresher recent Atlantic Water mass (Balbín et al. 2012). Thus, chlorophyll-a could be acting as a proxy of water masses. On the other hand, previous studies proposed that low chlorophyll-a values could affect spawning of bluefin tuna that select very low productivity areas in order to avoid predation during the very first developmental stages (Reglero et al. 2011). Whether the effect of chlorophyll-a on the spawning locations is direct or indirect associated to water masses needs further investigation.

### 5.4.1 Model selection and validation

One of the most relevant results from this study emerges from the cross-validation approach developed. Responses of spawning habitat quality to the different variables could be strongly driven by particular years (i.e. year-specific oceanographic scenarios). By processing the cross-validation approach, we tested whether the response variables were stable among different years, which is of paramount importance to provide projections of potential spatial distributions with appropriate certainty. All variables were consistent along the five years predicted, but small differences allowed identifying factors that had more relevance in some years than others.

Our models were able to reproduce the spatial distribution of yolk sac and preflexion stages of bluefin tuna larvae, used as proxy for spawning locations. However, absolute values of spawning habitat quality from observed and modeled data differ. This suggests that including additional variables of biological or oceanographic processes not considered in this study would improve the modeling results. For example, considering the inter-annual variability of the total abundance of the bluefin spawning stock or approaching the habitat quality definition by standardizing the spawning habitat quality among years are aspects to explore in the future.

### 5.4.2 Applications and further research

Predictive modeling of spawning habitats could serve as an operational tool for designing dynamic spatial management approaches (Hobday et al. 2010; Hobday et al. 2014), and for improving current larvae habitat models used in spawning stock biomass calculations (Ingram et al. 2013). Identification of relevant seascape metrics may also be applicable when predicting pelagic essential habitats under other methodological frameworks applied to larger spatial scales (Druon et al. 2011).

Nevertheless, in order to take the most of the 'pelagic seascape ecology' to advance towards the emerging field of 'operational fisheries oceanography' some analytical developments are required. Here, we number those we believe are the most relevant ones: i) the accessibility to long time series and near-real time predictions of key environmental variables in common data formats, ii) well validated environmental data products from remote sensing and hydrodynamic models (long term simulations and forecasts), iii) the appropriate knowledge about the oceanographic drivers on ecological processes of species, iv) the design of appropriated pelagic seascape metrics capturing the dynamic processes affecting the

species ecology, and v) identification of the specific needs in terms assessment and management.

In relation to these challenges, data from remote sensing and hydrodynamic models are available at global, regional and local scales from multiple operational oceanography data providers. However, standardization in data formats would be appreciated as well as software libraries for operational oceanography data handling in open source software packages such “R” (R Development Core Team 2008), what is also claimed by other researchers in the field (Hobday et al. 2014). Furthermore, software drivers for old data formats, as some hierarchical data formats (HDF), may not be available or easy to handle with new operating systems. In addition, specific calibration and validation of the operational oceanography products in specific areas of interest is a key issue for succeeding in the application and development of pelagic seascape ecology. This needs dedicated efforts at regional and local scales to develop data-assimilative high resolution hydrodynamic models combining data from multiple sampling platforms (Tintoré et al. 2013b).

Along this study we have referred to “pelagic seascape ecology” as a particular field within the emerging “seascape ecology”. Research in the field of seascape ecology differs when applied to the analysis of benthic habitats or to the pelagic habitats. The benthic is based on the analysis of habitats considered as a mixture of categorical habitat patches. In that case, techniques to quantify habitat patterns are mainly the same than those applied in the traditional landscape ecology (Wedding et al. 2011), which provides valuable information for the study of how benthic habitat patterns affects nectobenthic species (Pittman et al. 2011). However, pelagic seascapes are highly dynamic and do not present clear boundaries. Therefore, metrics based on the patch concept are not valid and different techniques and concepts are necessary to investigate species-habitat relationships. Pelagic seascape ecology is strongly linked to satellite remote sensing, hydrodynamic models and the development of algorithms for identification of specific oceanographic processes such frontal areas (i.e. Hobday and Hartog, 2014), while benthic seascape ecology is linked to categorical benthic habitat and topographic maps (i.e. Bostrom et al. 2011). Seascape ecology can be applied to study ecological processes regarding, for example, how benthic habitat fragmentation affects population distribution (techniques from benthic seascapes), or how oceanographic processes such as chlorophyll-a blooms in the area, affects their growth (techniques from pelagic seascapes). Adequate analysis for particular species may consider both pelagic and benthic seascape techniques incorporating different processes.

Operational fisheries oceanography must provide solutions to actual limitations in fisheries science, therefore the questions addressed in studies of seascape ecology must be focused on what fisheries assessment and management require. Here, we focused on the spawning habitat models, with direct applications to current assessment methods. However, assessments of fish stock status are implemented in various ways for different species and geographical areas. In addition, different management approaches are applied for conservation of exploited and endangered species. Therefore, this evidence the necessity to create multidisciplinary groups for assessment and management purposes that embrace expertise on seascape ecology, operational fisheries oceanography and classic assessment methods.

### **5.4.3 Conclusions.**

Previous studies evidence that the location of bluefin tuna spawning areas around the Balearic Sea is dependent on the oceanographic scenarios that may change among years. This dependency responds to specific ecological requirements of the bluefin tuna adults and larvae, and provides the scientific basis to model the distribution of spawning areas. The present study achieves that objective using exclusively operational oceanography products, such as remote sensing altimetry, chlorophyll-a and sea surface temperatures from hydrodynamic models. Further processing of these products, as the temporal evolution of the sea surface temperature or the spatial variability of kinetic energy, provided pelagic seascape metrics that had better capture the dynamic processes affecting bluefin tuna spawning ecology. These metrics allowed developing a predictive approach of the spatial distribution of spawning locations in a near real time, which opens a new generation of assessment and management tools. Improving prediction capabilities of the bluefin tuna spawning habitats in a broad variety of oceanographic scenarios has a direct effect in assessment and management. These capabilities will provide robust basis to design pelagic dynamic marine protected areas and to adjust larval indices used in the evaluation of the spawning stock biomass.

## **5.5 Acknowledgements**

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from hydrodynamic models (MFS) and multisensor chlorophyll-a data. Sea surface temperature data from satellite was provided by NASA (<http://oceancolor.gsfc.nasa.gov>). The altimeter products were produced by Ssalto/Duacs and distributed by AVISO, with support from CNES (<http://www.aviso.altimetry.fr/en/data.html>). The research was developed under the framework of the BLUEFIN TUNA Project driven by the Balearic Island Observing and Forecasting System ([www.socib.es](http://www.socib.es)) and the Spanish Institute of Oceanography ([www.ieo.es](http://www.ieo.es)). I would like to thank the people involved in this research: Manuel Hidalgo, Pilar Tugores, Alberto Aparicio, Lorenzo Ciannelli, Patricia Reglero, Melanie Juza, Baptiste Mourre, Ananda Pascual, Jose Luis Lopez-Jurado, Alberto García A., Rosa Balbín, José María Rodríguez, Joaquín Tintoré and Francisco Alemany.

## 5.6 Supporting information.

Table S5.1: Number of stations sampled, geographical coverage and dates of the sampling campaigns

Year	N stations sampled	Max lat	Min lat	Max long	Min long	Start and end day/month
2001	135	40.333	38.333	4.600	0.600	16/06 - 9/07
2002	173	40.336	37.997	4.911	0.777	07/06 - 30/06
2003	176	40.334	37.832	4.692	0.780	03/07 - 30/07
2004	160	40.333	38.158	4.698	0.773	18/06 - 10/07
2005	164	40.338	37.999	4.923	0.775	27/06 - 23/07

Table S5.2: Operational oceanography data sources.

Data set	source data	Environmental Parameter	Temporal resolution	Spatial resolution	Data provider
AVISO-SSA	Remote sensing Altimetry	Maps of sea level anomaly	1 day	12.5km	AVISO*
MODIS-SST	Remote sensing spectroradiometer	Maps of Sea surface temperature	8 day	4 km	NASA*
MFS-SST	Hydrodynamic modeling	Maps of Sea surface temperature	1 day	6.5 km	MY OCEAN*
MFS-SS	Hydrodynamic modeling	Maps of Sea surface salinity	1 day	6.5 km	MY OCEAN*
RS-CHL	Remote sensing spectroradiometer	Sea surface chlorophyll-a concentration	1 day	4 km	MY OCEAN*

\*AVISO: <http://www.aviso.altimetry.fr/en/services/partners.html>

\*NASA: <http://oceancolor.gsfc.nasa.gov/cms/>

\*MY OCEAN: <http://www.myocean.eu/>

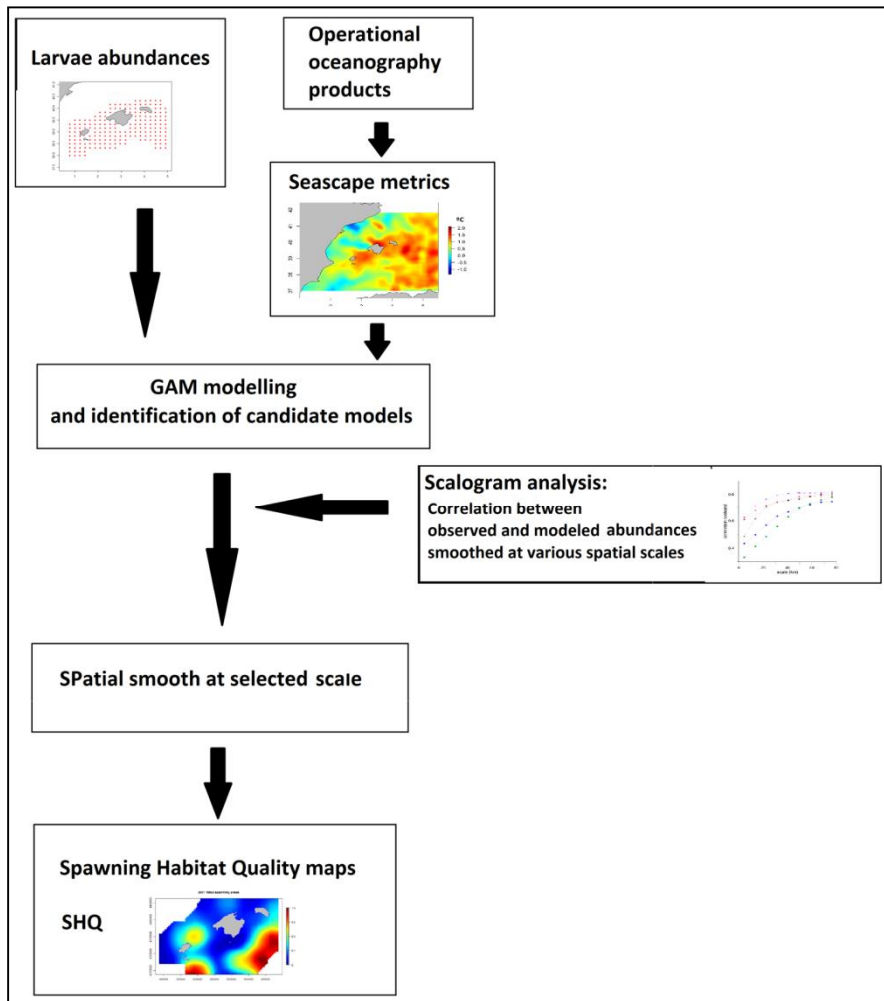


Figure S5.1. Methodological scheme of SHQ calculation from larval abundances and operational data products.



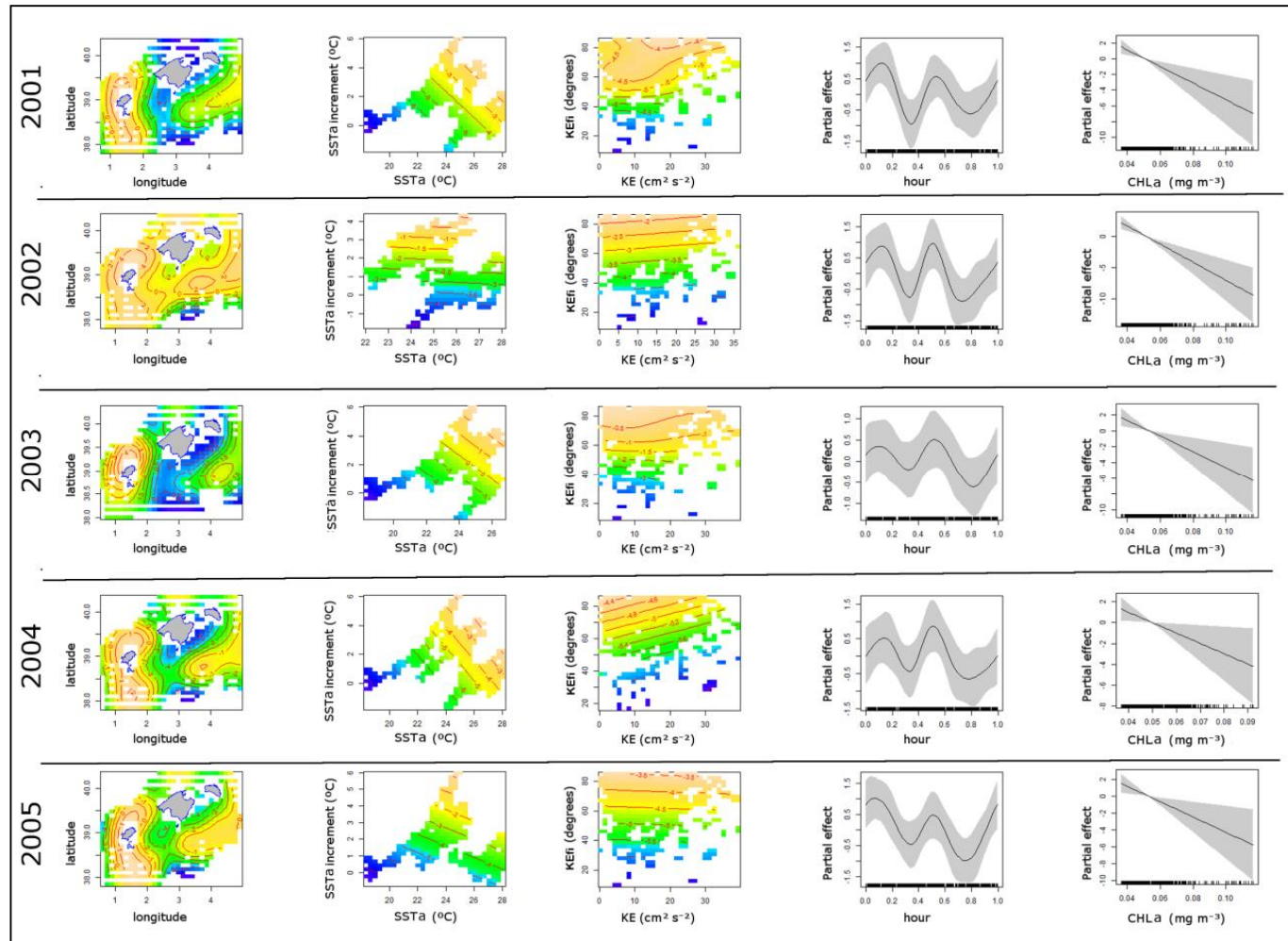


Figure S5.2. Partial effects of environmental covariate plots for the five models obtained from the cross-validation process.

## **CHAPTER 6**

**Incorporation of habitat information in  
the development of indices of larval  
bluefin tuna (*Thunnus thynnus*, Linnaeus,  
1758) in the western Mediterranean Sea  
(2001-2013)**

## 6.1 Introduction

Managers became concerned of the status of Atlantic bluefin tuna (*Thunnus thynnus*) stocks in the late 1960's. During recent years, international stock assessments of Atlantic bluefin tuna (BFT hereafter) have been conducted at least biannually by the ICCAT (International Commission for the Conservation of Atlantic Tunas). Management of BFT is accomplished using two differentiated stocks (the Eastern Stock, spawning in the Mediterranean and the Western Stock, spawning in the Gulf of Mexico), by establishing total allowable catches (TAC) and other complementary measurements, as temporal closures for specific fishing grounds or minimum size restrictions (Fromentin & Powers 2005). Virtual Population Analysis (VPA, eg. Butterworth & Rademeyer 2008) serves as technical basis for defining TAC under different fisheries scenarios (ICCAT 2013). Traditionally, results from virtual population analysis have been contrasted and calibrated with other complementary abundance indices, primarily based on fisheries-dependent data. For the Eastern stock, fisheries on juvenile BFT in the Gulf of Vizcaya provided information on recruitment rates; and Southern Spanish traps, targeting individuals during their reproductive migratory routes into the Mediterranean (Ortiz de Urbina *et al.* 2007), provided information on adult stock status. During the last decade the TAC of the Eastern stock has been reduced and minimum legal sizes for juvenile BFT changed, affecting these two fisheries and associated indices, which may negatively affect future index availability (ICCAT 2013), thus, possibly affecting the quality of final VPA results.

The possibility of developing indices of BFT stock status from fishery-independent data, based on the abundance of bluefin tuna larvae collected during dedicated ichthyoplankton surveys of NOAA Fisheries, was proposed in the beginning of the 1990s (Scott *et al.* 1993) for the Western stock. Recently, (Ingram *et al.* 2010) updated these indices using standardization via delta-lognormal models.

During recent decades, ichthyoplankton surveys targeting BFT larvae were conducted in several areas of the Mediterranean Sea. However, the surveys employed heterogeneous sampling strategies and methodologies, without any temporal continuity (Dicenta 1977; Dicenta & Piccinetti 1978; Piccinetti 1994; Piccinetti *et al.* 1997; Piccinetti *et al.* 1999; Oray & Karakulak 2005). In 2001 the IEO ([www.ieo.es](http://www.ieo.es)) started a series of standardized ichthyoplankton surveys, named TUNIBAL, around the Balearic Islands, recognized as one of the main spawning areas of BFT within the Mediterranean (García *et al.* 2005b; Alemany *et al.* 2010), with the aim of characterizing the spawning habitat of this species and deepen in the knowledge of its larval ecology, assessing the influence of environmental factors on larval distribution and abundance. These surveys followed an adaptive sampling strategy, combining intensive sampling of high density larval patches with quantitative sampling over a

systematic grid of stations. Similar surveys were carried out in 2012 and 2013, following the same sampling strategy, within the framework of a joint agreement between IEO and the Balearic Islands Coastal and Forecasting System (SOCIB, [www.socib.es](http://www.socib.es)) under the framework of the project BLUEFIN TUNA and the ATAME Spanish I+D National Plan competitive project .

The results from these surveys have shown that spatial location of spawning habitats of BFT are strongly influenced by mesoscale oceanographic processes in the Balearic sea (Alemany *et al.* 2010; Reglero *et al.* 2012; Muhling *et al.* 2013; Álvarez-Berastegui *et al.* 2014), which has been also demonstrated in the Gulf of Mexico (Lindo-Atichati *et al.* 2012; Muhling *et al.* 2013). Therefore, larval index values may be influenced by the type of habitat sampled among years. Improving the knowledge of how habitat information can increase the performance of larvae index models is of paramount importance to the advancement of stock evaluation methodologies independent from fisheries data. Previous larval index calculations (Ingram *et al.* 2013) have included salinity and temperature as environmental linear covariates, but other recent studies (Reglero *et al.* 2012) have demonstrated that their effect on the larval habitat characterization may not present a linear response.

The BFT larval abundance data gathered during these surveys are useful for developing an index of abundance, which would represent the second fishery-independent index of abundance of BFT in the world, and currently the only fishery-independent index concerning the eastern Atlantic stock. Therefore, the objective of this study is to present abundance indices of BFT larvae collected around the Balearic Islands based on delta-lognormal models and evaluate its adequacy as indicator of the temporal trend of spawning stock biomass. We also evaluated if considering environmental spatial variability into the larvae indices, accounting for differences in the percentage of larval habitat sampled each year, improves the quality of the larvae indices calculations. To reach these objectives different larval index models are compared between each other and also with adult spawning stock abundances obtained from the BFT stock assessment of ICCAT (ICCAT 2013).

## 6.2 Methods

### 6.2.1 Field sampling.

The sampling methodologies for the period 2001-2005 are described in detail in Alemany *et al.* (2010). BFT larvae were collected by oblique tows performed down to 70 meters in the open sea or down to 5 m above the sea floor in shallower stations, using a 333  $\mu\text{m}$  mesh fitted to 60 cm mouth opening Bongo nets. In addition, subsurface tows between 5 m deep and surface were carried out at

the same stations in 2004 and 2005 by means of a Bongo 90 net equipped with a 500  $\mu\text{m}$  mesh. Also, in 2012 and 2013, BFT larvae were collected by oblique tows performed down to the thermocline ( $\sim 30$  m), using a 500  $\mu\text{m}$  mesh fitted to a Bongo 90. In each of those years around 200 stations, located over the nodes of a regular grid of 10 x 10 nautical miles, covering most of the known BFT spawning areas in this region (from  $37.85^\circ$  to  $40.35^\circ$  N and from  $0.77^\circ$  to  $4.91^\circ$  E), were sampled during June and July, the spawning peak of the species in the Western Mediterranean. The exact number of sampled stations per gear and the dates of the surveys are shown in Table 6.1. Stations where adaptive sampling was performed, or otherwise were not strictly part of the survey grid, were not included in the dataset for analysis (Figure 6.1). In all haul-types, flowmeters were fitted to the net mouths for determination of the volume of water filtered. Plankton samples were fixed on board with 4% formaldehyde in seawater. In the laboratory, all fish larvae were sorted under a stereoscopic microscope. Tuna larvae were then identified to species level and BFT larvae standard length were measured by means on an Image Analysis System. In addition, at each station, a vertical profile of temperature, salinity, oxygen, turbidity, fluorescence and pressure was obtained using a CTD probe SBE911.

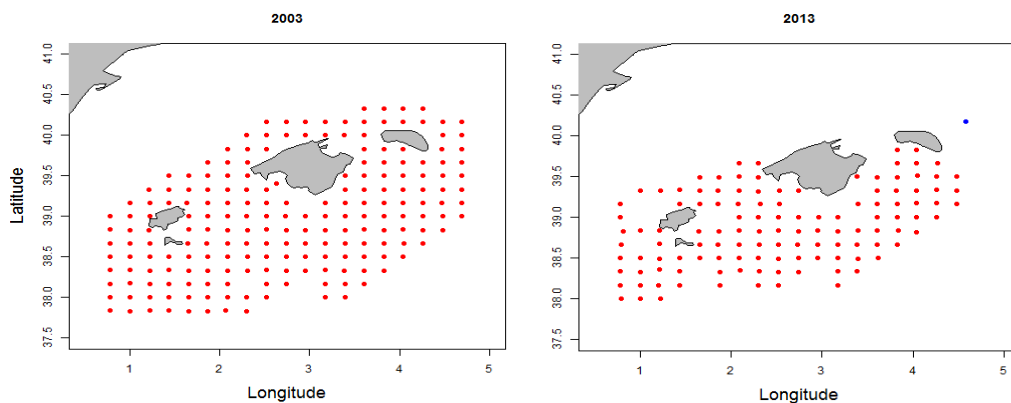


Figure 6.1. Distribution of survey sampling effort.

Table 6.1. Survey years, dates and number of stations of larval index dataset.

<b>Gear</b>	<b>Haul Type</b>	<b>Survey Year</b>	<b>Number of Stations</b>	<b>Start Date</b>	<b>End Date</b>
B60	deep oblique	2001	162	16-jun-01	07-jul-01
B60	deep oblique	2002	171	07-jun-02	28-jun-02
B60	deep oblique	2003	198	03-jul-03	29-jul-03
B60	deep oblique	2004	166	18-jun-04	08-jul-04
B60	deep oblique	2005	186	27-jun-05	23-jul-05
B90	subsurface	2004	166	18-jun-04	08-jul-04
B90	subsurface	2005	186	27-jun-05	23-jul-05
B90	mixed layer oblique	2012	153	21-jun-12	08-jul-12
B90	mixed layer oblique	2013	124	20-jun-13	10-jul-13

### **6.2.2 Calculation of larval abundances, response variable of larval index models.**

Relative Larval abundances at 2mm was used as response variable in larval index models. To calculate this parameter the numbers of specimens collected at a station were adjusted to the number of 2-mm larvae, using the decay in numbers at size, derived from a length-based catch curve for each gear-type (Figure 6.2). Due to the decreased selectivity in both gears for 2-mm larvae a coefficient was also used for adjustment: 1.022 for Bongo 60 and 2.199 for Bongo 90. For years 2004 and 2005, the Bongo 90 larval catches were not measured. Therefore, in order to adjust these numbers as the others, the length distribution of the 2004-2005, Bongo 90 was assumed to be that summarized from 2012 and 2013 surveys Bongo 90 length data. Finally, larval density was calculated by dividing the adjusted catch numbers by the volume filtered by the gear. Larval abundance was calculated by multiplying the density by the tow depth.

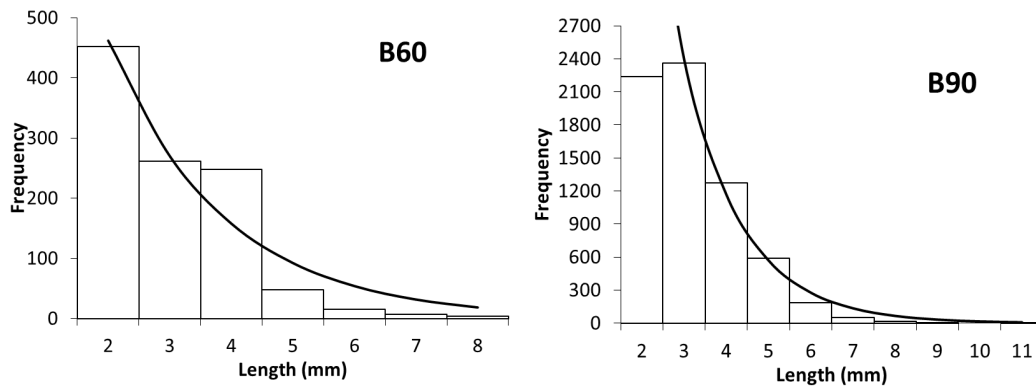


Figure 6.2. Decay curves used to back-calculate the number of 2-mm larvae. The equation for the bongo-60 (B60) curve is  $N = 1352.74 e^{-0.5372(\text{length})}$ , where  $N$  is numbers of larvae and length is in mm, and the equation for the bongo-90 (B90) curve is  $N = 20763.24e^{-0.7194(\text{length})}$ .

### 6.2.3 Habitat adjusted larval index models.

Larval index models provide the relative abundances of larvae at each station. From this models, inter annual variability on relative abundances are calculated and scaled to a mean of one to provide the “scaled larval index”, that is used as indicator of annual trends of spawning stock biomass.

Three different larval indexes were computed in order to evaluate how accounting for environmental information improves the performance of the models. The first model, denoted as “basic larval index” (BLI), included no environmental variables in its formation. The second model, denoted as “standard larval index” (SLI), included salinity and sea surface temperature to evaluate if there were any linear effects of these environmental variable, following previous versions of the larval index in the Balearic Sea (Ingram *et al.* 2013). The third model, denoted as “habitat corrected larval index” (HLI) included a “potential larval habitat” covariable. This new covariable provides, for each station, a rank value (from 0 to 1) of the habitat quality as a function of environmental information. Details of modeling approaches are provided in following sections.

### 6.2.4 Model configuration of the “Basic Larval Index” (BLI)

The BLI predicted relative larval abundances using a delta-lognormal model. Delta-lognormal models are a specific approach applied to deal with zero-inflated distributions by combining two “submodels”, the predictions probabilities of a presence/absence model, and the prediction probabilities of only presence data after logarithmic transformation. Application of delta-lognormal index for fisheries was proposed by Lo (1992). Similar covariates were tested for inclusion for both submodels to develop the BLI: time of day (three categories: night, day, and crepuscular), month,

area (survey area divided into subareas of  $\frac{1}{2}$  degree latitude and longitude), and year. A backward selection procedure was used to determine which variables were to be included into each submodel based on type 3 analyses with a level of significance for inclusion of  $\alpha = 0.05$ . If year was not significant then it was forced into each submodel in order to estimate least-squares means for each year that provides the interannual variability of larvae abundances. The GLIMMIX and MIXED procedures in SAS (v. 9.1, 2004) was used for the computation of the delta-lognormal model.

The fit of each of the submodels were evaluated using AIC, residual analysis for the lognormal submodel, and the area under a receiver operating curve (AUC) for the binomial submodel (Fielding & Bell 1997).

### **6.2.5 Model configuration of the “Standard Larval Index” (SLI)**

SLI was similar to the previous model but additional covariates were tested for inclusion in both submodels to develop abundance: geostrophic velocity (calculated at 5 meters depth from CTD profiles at the bongo fishing stations), the average salinity and temperature in the mixed layer. The model building procedure and evaluation was the same as described for the SLI.

### **6.2.6 Model configuration of the “Habitat corrected Larval Index” (HLI).**

Model design was the same as that for the SLI with one modification; a “potential habitat” (PHAB) variable, ranking the potential larval habitat quality, was included as additional covariate in the binomial submodel and the lognormal submodel.

For estimating the potential habitat quality indicator (PHAB) associated to each sampled station of a given year, the dataset (seven years of data), was split into two datasets, the prediction data set and the fitting dataset. The first one containing data from the considered year and the second one with data from the other five years. Using the fitting data set, a quasibinomial general additive model (GAM, Wood 2006) was designed to fit the larvae presence/absence to the following candidate explanatory variables: latitude, longitude, gear type, sea surface salinity, day of the year, geostrophic velocity and residual sea surface temperature. A backward stepwise model selection process was applied. Covariates significance was set at  $p < 0.05$  and model performance improvement was evaluated by their AUC. rSST was defined as the residual of SST against the day of the year, as both variables were strongly correlated. This variable accounted for stations where the temperatures were above or below the average for a specific time in the year. rSST was defined as the residual of SST against the day of the year, as both variables were strongly correlated. This



interannual cross-prediction approach provided the probability of larvae presences from the environmental information accounting for non linear relationships. This cross prediction approach is similar to that developed for the prediction of spawning areas in chapter 5. These predictions were used as the potential habitat quality indicator. This process was applied for each sampling campaign, so predictions of PHAB for each year were always based on data from the other six years. All calculations related to the PHAB were processed with the R software using the MGCV package (Wood 2006).

### **6.2.7 Comparison of larval index models and spawning stock biomass.**

In order to have an independent reference for evaluating the adequacy of the larval index as indicator of the SSB trends and which of the three larval index models could be a better estimator, results from the BLI,SLI and HLI models were compared against estimates of spawning stock biomass (SSB) obtained from the VPA of the ICCAT stock assessment in 2013 (ICCAT 2013). Time series of larval index from the three models and SSB from 2001 to 2005 and 2012 to 2013 were plotted to visualize trend patterns. Kendall and Spearman tau coefficients were calculated as indicators of correlation between larval index models and SSB.

It is well recognized that values of SSB obtained from VPA present high deviances for the most recent years, for that reason we run the same analysis with only the 2001-2005 period, for which VPA results can be considered more robust.

## **6.3 Results**

### **6.3.1 Data Summary**

Sizes of larvae collected in the Bongo-60 gear ranged from 1.39 to 8.5 mm and those from Bongo-90 between 1.74 and 11.49 mm (table 6.2). Length data for the Bongo-90 gear from 2004 and 2005 surveys are currently unavailable.

Table 6.2. Summary of data used in these analyses. B60 and B90 gear type indicate bongo-60 and bongo-90 gear, respectively.

Gear	Haul Type	Survey Year	Number of Specimens	Mean Length (mm)	Size Range (mm)
B60	deep oblique	2001	121	3.56	2.0 - 5.5
B60	deep oblique	2002	135	3.09	1.39 - 8.0
B60	deep oblique	2003	211	3.31	1.63 - 8.5
B60	deep oblique	2004	263	3.51	1.63 - 8.0
B60	deep oblique	2005	182	3.33	1.63 - 8.0
B90	subsurface	2004	3174	NA	NA
B90	subsurface	2005	831	NA	NA
B90	mixed layer oblique	2012	28761	3.83	2.06 - 8.74
B90	mixed layer oblique	2013	24728	3.66	1.74 - 11.49

### 6.3.2 PHAB model performance.

All covariates resulted significant for the model providing the PHAB. The PROC curve used for model validation is shown in figure 6.3. AUC was equal to 0.805, what shows a good model performance.

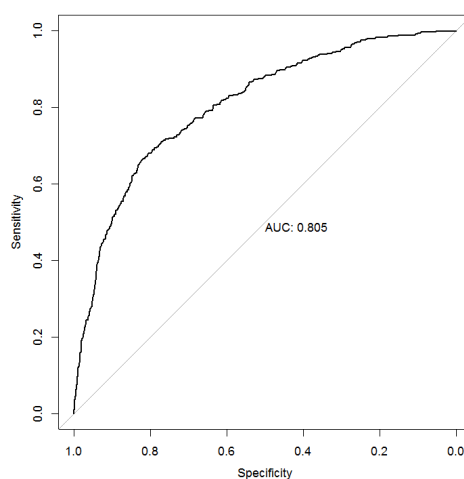


Figure 6.3: PROC curve of the PHAB presence/absence model.

### 6.3.3 Variable selection process for the BLI, SLI and HLI

The variable selection process of each of the three models analyzed is summarized in Table 6.3. The year variable was retained in each submodel in order to estimate least-squares means for each year to ensure the development of annual indices. For the BLI model with no environmental covariates included, the “survey area” was retained, but this variable was dropped when environmental variables were considered, showing that salinity and temperature accounted for much of the spatial deviance of the larval occurrence. For the BLI and SLI lognormal submodels, only the year variable was retained, while PHAB covariate was retained for the HLI lognormal submodel.

Graphical outputs of the BLI, SLI and HLI models are presented in Figure 6.4. All models predicted a relevant increase in larval abundance from 2005 to 2012 and a drop in 2013. However, the three models showed different patterns for the 2001-2005 time series. The BLI showed a constant decrease on the larvae index values during these years, while the SLI presented a small increase in larvae index till 2002, then dropped till 2005; and the HLI increased till 2003, then dropped till 2005.

Table 6.3: Summary of the variable selection process for each model and submodel.

Model	Submodel	Variables excluded	Variables included	AUC	AIC
BLI	Binomial	Time of the day	Year, month , geographic area	0.795	7172.7
BLI	Log-normal	Time of the day , month , geographic area	Year	--	1508.4
SLI	Binomial	Time of the day , geographic area	Year, month, salinity temperature	0.797	7423.6
SLI	Log-normal	Time of the day, geographic area, month, salinity temperature	Year	--	1605.4
HLI	Binomial	geographic area	Year, month, time of the day, PHAB	0.791	7294.7
HLI	Log-normal	Time of the day , month , geographic area	Year, PHAB	--	1594.8

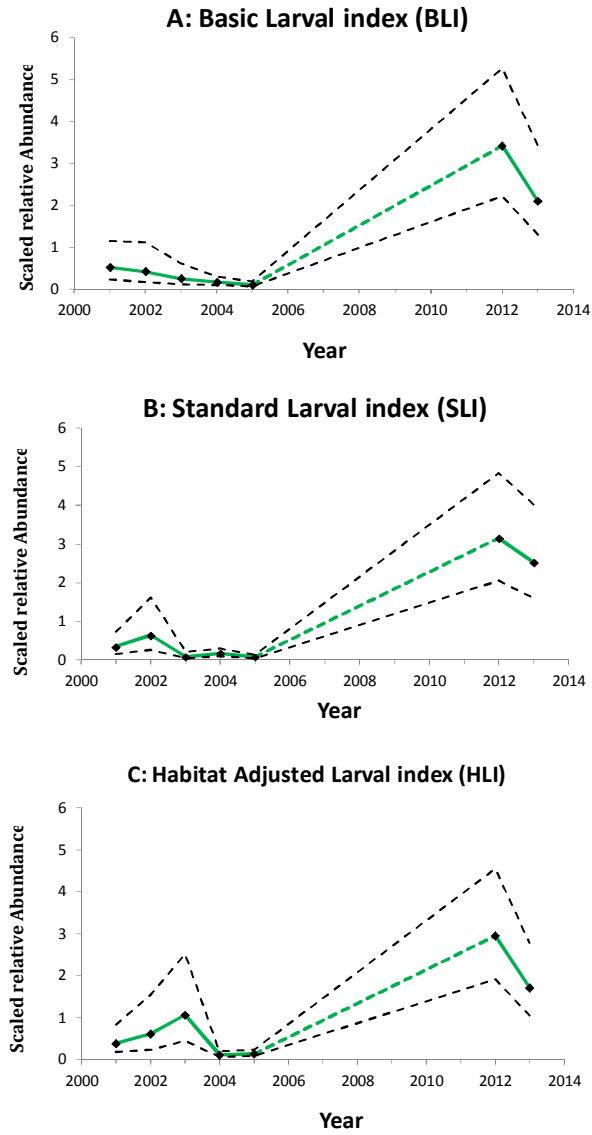


Figure 6.4. Relative abundances of BFT larvae scaled to the mean of one (green line) and 95% of confidence limits (black lines). Results from the three indexes, A: Basic larval indices (BLI); B: Standard larval index (SLI); C: Habitat adjusted larval index (HLI)

### 6.3.4 Comparison between larval indices and SSB.

Comparison between the three larval index models and values of SSB are presented in figure 6.5. The three larval indices follow the same general trend of the SSB, with higher values of larvae abundances in the period 2012-2013 than in the 2001-2005 years (Figure 6.5A-6.5B). Both correlation coefficients (Spearman and Kendall, Table 6.4) values show that the three models are well correlated with SSB trends. The HLI presented the highest coefficients. When we compared data from years 2001-2005 (see figure 6.5C-6.5D), the two correlations coefficients for the BLI and SLI were low while the correlation between larvae abundances from the HLI model and the SSB from VPA gets the higher value.

Table 6.4: Kendall and Spearman correlation coefficients for the three larval index models (BLI,SLI,HLI), against the spawning stock biomass (SSB). Correlations calculated for the 2001-2013 period (7 years of analysis), and for the 2001-2005 period, with more robust SSB data (5 years of analysis).

Kendall's coefficient			Spearman's coefficient		
2001-2013 analysis (n=7)			2001-2013 analysis (n=7)		
<i>Model</i>	<i>Kendall's</i>	<i>p-value</i>	<i>Model</i>	<i>Spearman's</i>	<i>p-value</i>
<b>BLI</b>	0.523	0.1361	BLI	0.785	0.0480
<b>SLI</b>	0.523	0.1361	SLI	0.678	0.1095
<b>HLI</b>	0.904	0.0028	HLI	0.964	0.0028
2001-2005 analysis (n=5)			2001-2005 analysis (n=5)		
<i>Model</i>	<i>Kendall's</i>	<i>p-value</i>	<i>Model</i>	<i>Spearman's</i>	<i>p-value</i>
<b>BLI</b>	0.200	0.8167	BLI	0.500	0.4500
<b>SLI</b>	0.200	0.8167	SLI	0.200	0.7830
<b>HLI</b>	0.999	0.1667	HLI	0.999	0.0167

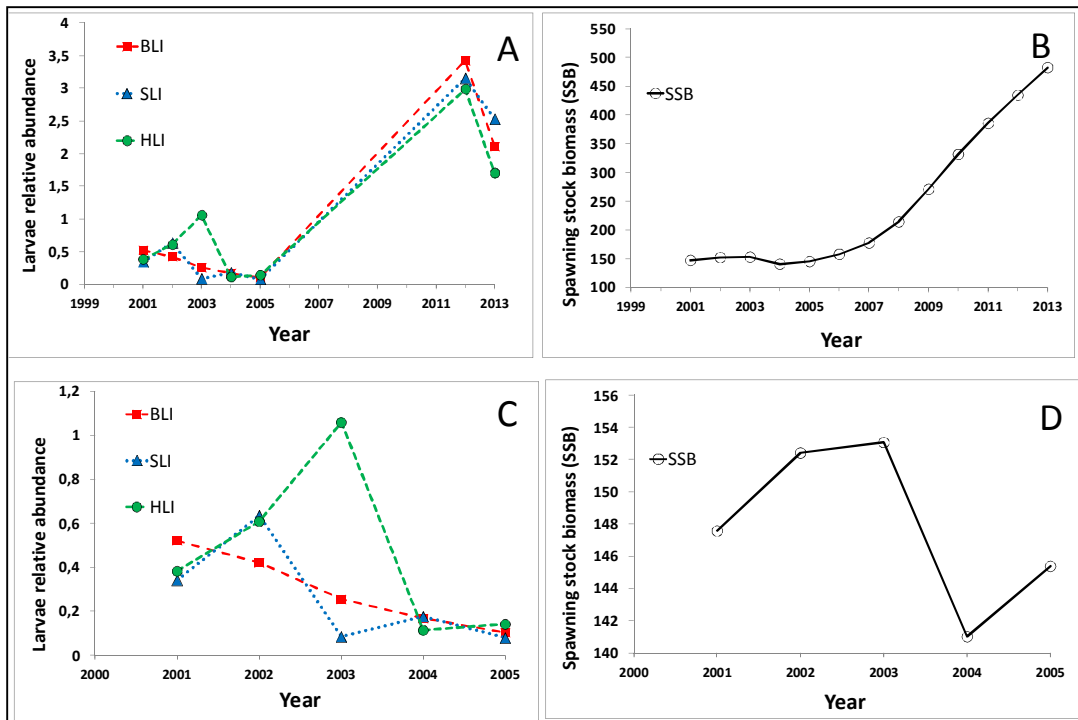


Figure 6.5. Comparisons between index values (BLI, SLI, and HLI) and spawning stock biomass (SSB) estimates from the 2013 stock assessment model.

## 6.4 Discussion

Larval index models analyzed in this study provided a time series of bluefin tuna larval relative abundances in the Western Mediterranean, based on three approaches: the basic larval index, standard larval index and the habitat-corrected larval index. Larval index trends followed a pattern similar to that of the spawning stock biomass, estimated using the virtual population analysis (VPA) from the previous ICCAT stock assessment model based primarily on fisheries-dependent data (ICCAT 2013).

Spawning stock biomass from VPA presented a strong increase when comparing the 2001-2005 and the 2012-2013 periods. This trend was well identified from the three larval index models, suggesting that sampling design, centered at the most relevant spawning areas, together with the modeling approach are able to capture significant changes in the spawning stock biomass. The model holding information about the spawning habitat quality sampled (HLI), presented significantly higher correlation coefficients with SSB data than the two other models (BLI and SLI). For the period 2001-2005, for which SSB data are more robust, but for which inter-annual variations on the SSB were lower, the HLI larval index was able to

accurately capture time series variations, while the BLI and SLI presented low correlation values. This result suggest that including larval habitat information in the models is crucial for identifying accurately SSB changes when inter-annual variability of spawning stock biomass is not very high.

Spatial distribution of larvae of BFT and other tuna species around the Balearic archipelago is tightly linked to mesoscale dynamic processes (Reglero *et al.* 2012; Muhling *et al.* 2013), as have been also observed in the GOM (Lindo-Atichatti *et al.* 2012, Muhling *et al.* 2013) . These mesoscale oceanographic processes, associated to the mixing of lower saline Atlantic waters incoming by the strait of Gibraltar with the resident surface waters, generate different oceanographic scenarios that vary among years (Mason & Pascual 2013; Balbín *et al.* 2014). For this reason, inter-annual variability of larvae abundances collected on a geographically systematic survey will be affected by this variability in hydrographic conditions, as the percentage of appropriated larval essential habitat sampled, that may vary among years. Results of this study confirm the need of weighting the results of larval abundances taking into account the spatial distribution of sampling stations in relation to that of the potential larval habitat in a given year. The latter can be only done, as this study shows, through sound knowledge about the environmental cues that drive the relation between larvae spatial distribution and hydrography, allowing the incorporation of hydrographic scenario inter-annual variability into the calculation of larval index. Summing up, the better the knowledge about which environmental descriptors provide more information on larvae distributions, the better the environmental standardization of the larval indices will be.

A significant increase of larvae abundances from the first years of the study (2001-2005) to the latest (2012-2013) was observed. Part of this increase could be attributable to the improvement of sampling methodologies (Bongo 90 fitted with 500 microns meshes vs BG 60 fitted with 333 microns meshes), but the results of the larval index calculations, which take into account the effect of the sampling gear, showed that this was mostly a reflect of the higher numbers of larvae at sea. Moreover, not only the BFT larval abundance changed between both periods, but their relative abundance in relation to the rest of tuna larva, since it was around 20% in the period 2001-2005 and higher that 95% for the period 2012-2013, suggesting an important shift in the structure of adult tuna populations in this region. This increase in BFT larval abundances confirms the improvement of the spawning stock biomass that has already been reported by the scientific commission on research and statistics of ICCAT (ICCAT 2013). This positive increase in larvae abundances along the study period was coterminous with the establishment of strong fisheries limitation and control measures by the

Countries members of ICCAT. Although bluefin tuna fisheries restrictions have been proposed since the 1990's for recovering the spawning stock in the Mediterranean, it was not till 2006 when a multiyear recovery plan was established, with strong limitations for the total allowable catch and minimum sizes regulations; and it was not till 2008 when this plan started to be fully implemented (ICCAT 2010). The results of our study suggest that measures taken started to take effect along the years 2008 to 2012. Continuous reductions in total allowable catches prevented the collapse predicted in 2009 in the case where no strong measures were taken (MacKenzie & Mariani 2012). However, this rapid recovery of the stock has been probably supported also by an extraordinary recruitment in 2003 resulting from higher larval survival rates associated to abnormally high sea surface temperatures (García *et al.* 2005a), which have been detected following the evolution of the age structure of BFT population in Spanish baitboat fleet catches (Rodriguez-Marin *et al.* 2013).

Apart from the increasing abundances between the two studied periods, deduced from the three different larval indices developed, it is relevant to highlight that all three models also showed a decrease in mean abundances in the year 2013 in relation to the abundances in 2012. This result could be showing a decrease or stabilization of the spawning stock biomass in the year 2013. This trend should be analyzed by extending the larval index time series along the following years, as differences among 2012 and 2013 are within the uncertainty limits of the model. It is worth to point out that larval index variation from 2012 to 2013 do not follow same pattern than spawning stock biomass obtained from virtual population models provided by ICCAT (ICCAT 2013) which indicated an increase from 2012 to 2013. However, the uncertainty associated to SSB values from the VPA for those years is very high (ICCAT 2013). In fact, to interpret properly the changes in BFT SSB from the first years of XXI century to those corresponding to the years 2012 and 2013, it must be taken into account that they do not represent a progressive and steady increase within an equilibrium situation, but they are mostly the direct consequence of an abrupt change in the exploitation pattern of the species. Only once a new equilibrium will reached the real trends in the temporal evolution of SSB, resulting from the combination of the new exploitation pattern and future environmental scenarios conditioning recruitment success and natural mortality, will become evident. Clarifying these aspects may be decisive for establishing the total allowable catches in following years. Some of the ICCAT integrating countries have already demanded an increase in TACs due to the strong pressure of the local fisheries sectors. Assignment of TACs became a political issue in 2013 and 2014. During 2013 ICCAT meeting, contracting countries showed different opinions about keeping or increasing fishing quota. Finally, ICCAT followed the



Standing Committee on Research and Statistics advice, keeping the same restrictions for fisheries as in years before.

In 2014 a new meeting of the ICCAT commission was held to establish TACs along following years. Conservationist groups had demanded keeping the TAC as years before (13500 tn), or at least not to increase it more than 10% per year, up to 20000 t along the next 5 years (WWF 2014). The SCRS reported a increasing trend of BFT population, but recommended to maintain the TAC of 2013, or to increase the catches moderately, but did not provide an specific value of what this rate of increase should be, due to uncertainties associated to the assessment, in part coming from the lack of stability of VPA results (Anonymous 2015b). Finally the ICCAT Commission decided a 20% of increase in annual TACs till 2017 (from 13500 tn in 2014 to 23500 in 2017, (Anonymous 2015a).

Improving accuracy and reducing uncertainties in stock assessment is paramount for the appropriate management of bluefin tuna populations. Within this framework, the time series of larval abundance indexes are of interest for BFT stock assessment (Anonymous 2015b), as it provided the unique already available fishery independent indicator of SSB. Unfortunately, field sampling campaigns for the collection of larvae abundance data is currently achieved by Spanish national funded research projects, and hence neither the continuity in the annual data collection nor appropriated spatial coverage can be guaranteed for the forthcoming years. Therefore, to keep the time series of larval abundance indexes, it is necessary to establish an internationally agreed larval monitoring program with and standardized sampling design for ensuring a reliable larval index calculation.

Going further, advances in the techniques for the larval index calculations could also provide an improvement in the assessment of spawning stock biomass. The application of this type of index in the Gulf of Mexico for the evaluation of trends in the SSB of the Western Atlantic BFT Stock demonstrated that the application of delta-lognormal models are an appropriate technique when larval CPUE data present an over dispersed, zero-inflated distribution (Ingram *et al.* 2010). This modeling technique was selected for developing the Balearic sea larval index. Nevertheless, future improvements on larval index calculations should consider that the best model, among the tree tested, was the one including an habitat parameter obtained from a nonparametric regression (Wood 2006) accounting for non linear relations between larvae abundances and environmental covariates. Techniques accounting for these effects or/and redefinition of covariates within the linear models could be explored to improve larval indices. Besides, here we used environmental covariates describing the hydrographic habitat conditions from vertical profiles of temperature and conductivity

collected *in situ*. Larval habitat parameterization could be improved by considering pelagic habitat seascape descriptors holding information about the dynamic nature of the pelagic realm. Seascapes derived from remote sensing or hydrodynamic models could provide additional information relevant for characterizing the pelagic habitat (Hobday & Hartog 2014), providing additional information for modeling larval habitat (Álvarez-Berastegui *et al.* 2014). Moreover, new operational oceanography platforms (Tintore *et al.* 2013) presently offer near real time information on the studied spawning area BFT, that may be applied for monitoring of these habitats, which could potentially aid in designing field sampling campaigns and improving larval index calculation.

## 6.5 Acknowledgements.

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# **CHAPTER 7**

## **General Discussion**

The results presented throughout this thesis represent an improvement of the characterization of fish essential habitats by applying seascape ecology techniques and concepts, as well as the transference of this knowledge to the current assessments and management for the conservation of exploited populations.

The objective of the following general discussion is twofold. It first underlines the main findings derived from the studies presented in chapters 2 to 7 and, second, it shows how these chapters link together setting an integrative research among the fields of seascape ecology, fish essential habitats and ecosystem based management. The selection of dusky grouper (*Epinephelus marginatus*) and Atlantic bluefin tuna (*Thunnus thynnus*) as study cases is related to the strong differences in their biology, ecology and management.

The initial motivation of this thesis raised from the findings obtained in the BIOMEX project. The results from this project showed how exploited fish abundances decline with distances to marine protected areas providing robust evidences of a gradient of fish biomass from marine protected areas (BIOMEX project, Planes 2005). The analytical tools used at that time were not appropriate to analyze the essential habitats of local fish species and how habitat distribution and exploitation patterns affect the spatial variability in the catch. Analyses of the temporal and/or spatial trends of Capture Per Unit Effort (CPUE) of low mobile coastal species such as dusky grouper needed to improve to incorporate appropriate information about their essential habitats, that could be defined as “*those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity*” (NOAA, 1996). A very relevant part of the observation patterns that one cannot explain is often related to the uncertainty associated to the spatial variability of essential habitats of these species (Coll et al. 2013, Stobart et al, 2011). This analytical challenge and its impact on assessment and management was, in essence, the trigger motivation for the research performed in this PhD.

Preliminary research of this thesis was performed on the different possibilities to try to get measurable parameters providing information about habitat characterization from bionomic and topographic maps. In 2008 Hinnchey et al. published the first review providing a rigorous evaluation of the potential of applying traditional landscape ecology concepts and techniques to the marine environment. From these studies, I developed integrative analytical tools to incorporate habitat information on to the current assessment and management approaches currently applied in two characteristic species from the coastal and pelagic ecosystems.

Dusky grouper was the first candidate, and actually a perfect one, as it is a highly resident species strongly linked to rocky habitats (Harmelin & Harmelin-Vivien 1999). The EPIMHAR project funded by the National Park's Network (Spanish Ministry of Environment, Maritime and Rural Affairs), had the objective of characterizing groupers essential habitats within the Cabrera Archipelago National Park to improve their conservation. Traditionally, characteristics of grouper essential habitats have been studied from data collected with underwater visual census, using information on habitat structure at small spatial scales (Harmelin & Harmelin-Vivien 1999; La Mesa *et al.* 2002; Vacchi *et al.* 2007). One of the main topics in landscape ecology is how the scale of observation affects our capability to recognize a habitat pattern affecting key ecological processes of a species under study (Turner, 1989). This is also a key issue in seascape ecology (Pittman & Brown, 2011; Wedding *et al.*, 2011). As a part of the team involved in the EPIMHAR project, I designed a way to use landscape ecology techniques, recently applied in coastal areas, to improve the definition of grouper essential habitats. To do that, I used habitat maps taking into account the spatial scales that provided the best information about how rocky habitat fragmentation patterns drives the distribution of the dusky groupers populations in the National Park.

During this study I faced strong limitations on the quality of the data available (mainly topographic and bionomic maps from various sources) and on the geographic information tools implemented in common software packages. For instance, all data cartographies had been developed using different geo-referenced systems (easy to correct) and different reference coastal lines (almost impossible to standardize). Methodological challenges such as this one forced investing time and resources in developing an appropriate technical framework for the study coastal marine habitats, and forced jumping from GUI software based tools (like ArcVIEW for example), to open source and programmable tools (as "R").

The combination of these advances, with the participation of a multidisciplinary team of researchers with strong knowledge on grouper biology and ecology from the field, allowed tackling the seascape multiscale analyses of dusky grouper essential habitats presented in chapter 2. The results contribute to improve the definition of dusky grouper habitats using seascape ecology metrics (Wedding *et al.* 2011), showing that habitat structure and topographic variables measured at different scales provides complementary sources of information to describe the essential habitats of this species. In addition, this study provides evidence of different scaled-drivers at different ontogenetic stages. Essential habitats of juvenile individuals are defined mainly by habitat descriptors providing information about rocky bottom complexity measured at small spatial scales. Rocky habitat patterns measured at

wider spatial scales and information about coastal dynamics provided additional information on juvenile habitat requirements. For adult individuals, both rock complexity at small scales and rocky structure patterns at wider spatial scales were highly relevant. Specific responses of densities of juvenile and adult individuals to each of the habitat descriptors analyzed helped to propose the ecological traits behind these relationships and to provide relevant information for MPAs design to protect this species, that should enclose: i) rocky habitats from surface down to 60 meters, ii) extensive areas of shallow rocky bottoms above 20 m with low-intermediate coastal exposures and with high complexity derived from the accumulation of small and medium rocky blocks, iii) cliffs with areas of intermediate slopes down to 30 meters with important rocky falls and high level of habitat complexity derived from big size blocks and surrounded by sea beds of fragmented habitats with relevant coastal hydrodynamics.

While main objective of this thesis was not to provide recommendations about how MPAs should be designed, I do aimed at providing effective analytical tools that were in addition able to link with and improve assessment and management procedures. If the identification of essential habitats can be structured starting from information at wide spatial scales to end with improved information at small scales, it would be possible to design an approach to assess habitat locations starting from map analysis and finishing with *in situ* data collection. Therefore, findings from the study of dusky grouper essential habitat and the improvement on habitat definition by using seascape metrics (from chapter 2) were applied to develop new methods for MPA design in the framework of Rapid Assessment Programs (Alonso *et al.* 2011). This idea allowed designing a novel method for rapid multidisciplinary environmental assessment of coastal areas of direct application for the design and management of Marine Protected Areas (chapter 3), providing tools for the selection design and management of coastal MPAs when time, budget or potential human pressures, create an urgent need for prioritization. The methodology developed allows maximizing results and minimizing cost by re-evaluating existing information on the study area, integrating physical, environmental and socioeconomic indicators, and generating outputs in the form of thematic maps to support managers. The final products obtained inform planners and managers about the study areas across multiple aspects that all need to be considered in integrated coastal management, complementing the approach of other existing rapid assessment techniques (Kramer 2003; Romero *et al.* 2007; Ballesteros *et al.* 2007). Although originally proposed for widespread use in the Mediterranean, this methodology can be flexibly adapted, with modifications in the selection of indicators, for its use in other regions. The results show its potential for merging and synthesizing information not only as a tool in Rapid Assessment

Programs but also as a tool for facing management of wide coastal areas integrating aspects of the complex coastal socio-ecological systems.

The challenge successfully faced in finding how habitats segregate spatially different developmental stages of a species, and applying that concept to improve MPA design (Chapter 3), demonstrated that identifying key ecological processes strongly dependent on the essential habitats could provide a path to advance on ecosystem base management without the need of setting extremely complex end-to-end ecosystem models. Assimilating this concept and seeing the potential for its application to improve the conservation of many other exploited species is a transversal conceptual and analytical approach that can be applied to different marine ecosystems, from the benthic to the pelagic realm. The pelagic ecosystems have obviously differences to the benthic systems, both in the habitat modeling and on the way resources are assessed and managed. Tackling with these differences to adapt the application of essential habitats into the conservation of an emblematic pelagic species, Atlantic bluefin tuna (*Thunnus thynnus*), is the focus of the chapters 4, 5 and 6.

This thesis evidences how pelagic seascapes can be applied to improve studies on the ecology of Atlantic bluefin tuna, a highly migratory pelagic species with a wide geographical distribution along the Atlantic waters. This species travels long distances to reach the Mediterranean and Gulf of Mexico during the reproductive season. Management of Atlantic bluefin tuna is approached mainly by minimum weight regulation and limitations in the total allowable catches (TACs), established as a function of the status of the adult stock populations calculated from virtual population analysis (Fromentin & Powers 2005). These management approaches completely differs from those applied for groupers in coastal areas. However, the same concept of improving conservation through the analysis of essential habitats can be applied for this species (Chapters 4, 5 and 6).

Nevertheless, techniques developed from landscape ecology, already applied for coastal seascape ecology, could not solve the characterization of the pelagic realm, where there are no clear boundaries delimitating habitats (Bostrom *et al.* 2011; Pittman *et al.* 2011). Besides the spatial scale is also a key issue when trying to identify dependencies of pelagic species at particular developmental stages with oceanographic conditions (Álvarez, 2015). Therefore, I propose a technical framework to parameterize the species surrounding habitat patterns and allowing investigating the effect of the spatial scale of observation by changing the “grain” (spatial definition) of the input habitat data (Chapter 4). Results show that pelagic seascape metrics, defined as a combination of hydrographic variables and their spatial gradients calculated at an appropriate spatial scale, improve our ability to model pelagic fish distribution.

These pelagic seascape metrics have been applied to study the spawning locations of the Atlantic bluefin tuna, showing that the quality of habitat models incorporating the proposed seascape metrics increases significantly when compared with models that do not consider these metrics. These results suggest that seascape techniques developed in the present thesis are relevant for the study of essential habitats of species whose ecology is dependent on highly variable local oceanographic scenarios as it is the case of bluefin tuna. The study shows general consistency with previous studies on spawning habitat on tuna specie (Alemany *et al.* 2010; Reglero *et al.* 2012; Muhling *et al.* 2013) while providing considerable advances on the understanding of how species perceive their habitat.

This study has been a first step in the application of seascape ecology to the pelagic realm and evidences that pelagic seascape ecology differs enough from coastal seascape ecology. It is very likely that they will both evolve along two different but complementary field of research in marine ecology. This is why I used the term “pelagic seascape ecology”. Pelagic seascape ecology is strongly linked to satellite remote sensing, hydrodynamic models and the development of algorithms for identification of specific oceanographic processes such frontal areas (i.e. Hobday and Hartog, 2014), while coastal seascape ecology is linked to high quality benthic habitat and topographic maps (i.e. Bostrom *et al.*, 2011).

The results from Chapter 4 showing how pelagic seascape ecology provided relevant information on spawning habitats of bluefin tuna around the Balearic Islands, establishing the basis for the improvement of the current management of this species (Chapters 5 and 6). The possibility of establishing closure areas as management approach in the open sea to protect pelagic species has been hanging over the scientific literature for a while (Game *et al.* 2009). Indeed, dynamic pelagic marine protected areas, which adapt to the location of dynamic essential spawning habitat of the southern bluefin tuna (*Thunnus maccoyii*), are already a reality in eastern Australia (Hobday *et al.* 2011). The main objective of these pelagic MPAs is to avoid the bycatch of bluefin tuna in long lines targeting other species. Non dynamic closure areas (fixed in space), with the same objective, have been established in the Gulf of Mexico the past year (US DOC/NOAA/NMFS 2014). In 2002, closure areas were also declared for the management of small pelagic species in the Mediterranean (BOE Num 313, 2012). These examples show that the approach of the spatial management, widely applied in coastal MPAs, is arriving to the institutions managing pelagic fisheries. In this framework, predicting location of spawning areas of bluefin tuna could be a very valuable tool for management, as could provide the scientific basis for designing closure areas to reduce bluefin tuna mortality in other fisheries than those with TACs assigned.



The prediction of spawning areas has been developed along this thesis (Chapter 5) following-up the techniques and concepts analyzed in Chapter 4. Chapter 5 provides strong evidences of modeling and predicting capabilities of spawning habitats of bluefin tuna in the Western Mediterranean. The bluefin tuna spawning habitats were characterized using only operational oceanography data from remote sensing and hydrodynamic models. The output of this study provides high quality spawning habitat maps around the Balearic Islands, complementing other techniques based on adult presences, as a tool for bluefin tuna assessment and management (Druon *et al.* 2011). These predictions provide the scientific basis to establish potential closure areas and the following and logical question is how to use this tool to manage fisheries. The bluefin tuna fisheries is strongly regulated and purse seiners directly targets the spawning aggregations during fishing, so closure areas are not appropriate to manage this type of fishing, while onboard observers may be the best control in that case. Nevertheless, it is worth noticing that no reported mortality of bluefin tuna by-catches exists. When a bluefin tuna is caught accidentally (in recreational or professional fisheries), the capture cannot “officially” be retained and, therefore, the capture will never be reported. While there is not information about the relevance that non-reported catch may have in the current assessment performed by the International Commission for the Conservation of Atlantic Tunas (ICCAT), I suggest that it would be relevant to have these estimates. Total numbers could be surprisingly high and the need of tackling the problem of accidental by catch will emerge. In that case, closure areas will be a relevant management tool.

In addition to assess closure areas, identification of spawning locations provides initial conditions for evaluating Atlantic bluefin tuna larval survival, that is strongly dependent on environmental conditions (Reglero *et al.* 2011). Coupling spawning ecology with survival models and environmental forcing to assess inter-annual variability on recruitment is one of the more relevant potential of the spawning habitat models, and it has been proposed as one of the mechanisms to implement modern “ecosystem fisheries oceanography” (Cury *et al.* 2008).

In addition to the development of tools allowing the application of new approaches for assessment and management of bluefin tuna, I investigated how essential habitat information could play an active role in improving techniques applied at present within ICCAT. Currently, the National Oceanography and Atmospheric Administration (NOAA, EEUU) applies larval abundance indices (Ingram *et al.*, 2010) to contrast and validate the results from the virtual population analysis, used to assess the status of the western population of Atlantic bluefin tuna. In recent assessments, the western stock larval index has been improved by including

environmental information on the modeling approach. If the spatial distribution of larvae is dependent upon a specific water masses and this water masses can be tracked, larval index models will improve substantially. Thus, techniques developed in Chapters 4 and 5 can be used to improve important assessment estimates of this species. This thesis presents a larval index for the evaluation of the eastern population of Atlantic bluefin tuna, which reproduces in the Mediterranean. The improvement of the larval index by incorporating information on essential habitats is presented in Chapter 6. Larval index models for the Mediterranean have still considerable room for improvements. However, first results provide robust steps to move forward in the assessment of this species incorporating essential habitat information (ICCAT, 2014).

The results obtained along the different research tasks presented from chapter 2 to 6 demonstrate that understanding the relationships among species and essential habitats provides key information for understanding species ecology, and allowing new techniques for improving species assessment and management. Here, I focused on two top predator species as study cases, but same principles and techniques could be applied to other species of interest across contrasting marine ecosystems. Seascape metrics of benthic habitats could provide valuable information for the study and conservation of many species in marine protected areas in the Balearic sea, and could improve the performance of novel techniques based on essential habitat information used for assessing the status of exploited populations in coastal areas (Coll *et al.* 2013). The pelagic environment around the Balearic islands is an important spawning area for many species and the hydrographic scenario determine relevant ecological processes during the larval stages of fishes (Torres *et al.* 2011; Rodriguez *et al.* 2013; Álvarez *et al.* 2015), crustaceans (Carbonell *et al.* 2014; Mallol *et al.* 2014) or cephalopods (Zaragoza *et al.* 2015) that in general affects the whole larval community (Hidalgo *et al.* 2014; Hidalgo *et al.* 2015). Therefore, the techniques and concepts applied for the analysis of pelagic seascapes in this thesis could also improve the identification of the links between oceanography and their ecological processes. Finding how this information needs to be ultimately transferred to management, as it has been done here, will require further investigation involving multidisciplinary groups of research.

# **CHAPTER 8**

## **General Conclusions**

- 1- Essential habitats of dusky grouper (*Epinephelus marginatus*) were defined by merging variables providing information about rocky habitat structure collected at small spatial scales from visual censuses and variables calculated at wider spatial scales from habitat maps. This multiscale approach increased the deviance explained in abundance models when compared to single scale approaches.
- 2- Essential habitats of dusky grouper change along the species ontogeny. Both, the variables and the spatial scales that best described the essential habitats were different for juvenile and adult individuals
- 3- Metrics traditionally applied in landscape ecology can be used in benthic coastal areas to define habitat structure patterns. Nevertheless, the application of these metrics in the marine environment needs to tackle specific issues. The inappropriate calculation of metrics that do not consider the effect of the terrestrial-marine border and the quality of the habitat georeferenciation derive in misidentification of dusky grouper essential habitats.
- 4- The spatial scale is a key issue in benthic coastal seascape ecology, as it is in traditional landscape ecology. In our case, the selection of appropriate spatial scales allowed to identify environmental variables influencing the distribution of adult and juvenile dusky groupers that otherwise would have not been considered as relevant.
- 5- Rapid assessment programs can be developed to assist marine protected areas design and management when information and/or budget are limited. Evaluation of coastal areas within a rapid assessment framework was possible by i) taking the most of already existing information in the area, ii) following a specific *in situ* sampling design, iii) involving local stakeholders, and iv) involving local experts in the data interpretation. These four requirements were essential to ensure the adequacy of the final output products for managers.
- 6- The traditional landscape ecology metrics, based on the patch mosaic concept and successfully applied in coastal areas, are not valid to parameterize dynamic processes in the pelagic realm. Thus, a new approach was developed based on the application of the gradient concept of seascape structure, combining means, gradients and spatial scales. This new approach was useful for describing the spatio-temporal dynamics of

the pelagic environment, allowing to identify the relation between bluefin tuna spawning habitats and frontal processes in the Balearic Sea.

- 7- The reproductive ecology of Atlantic bluefin tuna in the Balearic Sea is highly dependent on the local mesoscale oceanographic processes characterized by spatial gradients of surface salinities and geostrophic velocities. On the contrary, the bullet tuna (*Auxis rochei*), a resident species smaller in size than bluefin, is less dependent on these processes and functional responses are opposite to that showed by bluefin.
- 8- The relationships between the ecology of tuna species and the dynamic oceanographic processes only emerged when seascapes were observed, measured and parameterized at the appropriate spatial scales. The effect of the extent of the geographical area covered and the grain or spatial definition of the input data is a key issue in pelagic seascape ecology as it is in benthic coastal ecology and in terrestrial ecology.
- 9- Regarding the relation between bluefin tuna spawning ecology and the local mesoscale oceanography, operational oceanography data sources provided the possibility of monitoring and forecasting the location of reproductive areas of this species, through the development of models based exclusively on operational products. The spatial distribution of the bluefin spawning areas off the Balearic islands varies among years depending on the oceanographic scenarios.
- 10- Monitoring larval abundances in the Balearic Sea allowed developing larval indices providing information about the status of the Eastern population of Atlantic bluefin tuna. Incorporating information about essential spawning habitat in the larval indices calculations improved the correlation between such indices and the spawning stock biomass estimated from fishery data based population dynamics models.
- 11- The analyses developed to determine how marine habitats affects key ecological processes of a littoral resident species such as dusky grouper, and a highly migratory pelagic species such as bluefin tuna, allowed the improvement of the current approaches used for their management and assessment by applying a “minimum-realistic” approach. This approach was based on the combination of several simple but realistic models within the framework of the ecosystem based management.

# **CHAPTER 9**

## **Bibliographic References**

- Afonso, P., J. Fontes, and R. S. Santos. 2011. Small marine reserves can offer long term protection to an endangered fish. *Biological Conservation* 144:2739-2744.
- Agardy. 2010. *Ocean Zoning, Making Marine Management More Effective*. Earthscan, London, UK.
- Aguilar R., X. Pastor and M. J. de Pablo. 2006. *Habitats in danger: Oceana's proposal for protection*. Oceana, Madrid, Spain.
- Akaike, H. 1981. Likelihood of a model and information criteria. *Journal of econometrics* 16:3-14.
- Aleman, F., L. Quintanilla, P. Velez-Belchi, A. Garcia, D. Cortés, J. M. Rodriguez, M. L. Fernández de Puellas, C. González-Pola and J. L. López-Jurado. 2010. Characterization of the spawning habitat of Atlantic bluefin tuna and related species in the Balearic Sea (western Mediterranean). *Progress in Oceanography* 86:21-38.
- Allen, C. R., J. J. Fontaine, K. L. Pope and A. S. Garmestani. 2011. Adaptive management for a turbulent future. *Journal of Environmental Management* 92:1339-1345.
- Allen R. 2010. *International management of tuna fisheries: arrangements, challenges and a way forward*. Food and Agriculture Organization of the United Nations.
- Alonso L. E., J.L.Deichmann, S.A.McKenna, P.Naskrecki and S.J.Richards. 2011. *Still Counting: Biodiversity Exploration for Conservation – The First 20 Years of the Rapid Assessment Program*. Conservation International, Arlington,VA,USA.
- Álvarez Berastegui, D., A. Aparicio-Gonzalez, L. Rueda, O. Navarro, J. Coll, A. M. Grau and S. Deudero. 2010. Marine habitat mapping estimation of wave exposure in Cabrera Archipelago National Park for identification of essential *fish habitats*. XVI Simposio Ibérico de Estudios de Biología Marina (SIEBM), Alicante (Spain).
- Álvarez, I., J. M. Rodríguez, I. A. Catalán, M. Hidalgo, D. Álvarez-Berastegui, R. Balbín, A. Aparicio-Gonzalez and F. Alemany. 2015. Larval fish assemblage structure in the surface layer of the northwestern Mediterranean under contrasting oceanographic scenarios. *Journal of plankton research* .
- Álvarez-Berastegui, D., L. Ciannelli, A. Aparicio-Gonzalez, P. Reglero, M. Hidalgo, J. L. López-Jurado, J. Tintoré and F. Alemany. 2014. Spatial Scale, Means and Gradients of Hydrographic Variables Define Pelagic Seascapes of Bluefin and Bullet Tuna Spawning Distribution. *PLoS one* 9:e109338.
- ICCAT, 2015a, Report for biennial period, 2014-15 PART I - Vol.1. 1-61. 2015. Madrid, Spain, The International Commission for the Conservation of Atlantic Tunas. 2-3.
- ICCAT, 2015b, Report of the 2015 ICCAT Atlantic Bluefin Tuna Data Preparatory Meeting. 1-61. Madrid, Spain, The International Commission for the Conservation of Atlantic Tunas. 2-3-2015.
- Aparicio-González, A., J. L. López-Jurado, R. Balbín, J. C. Alonso, B. Amengual, J. Jansá, M. C. García, F. Moyá, R. Santiago and M. Serra. 2015. IBAMar Database: Four Decades of Sampling on the Western Mediterranean Sea. *Data Science Journal* 13:172-191.
- Aranda, G., F. J. Abascal, J. L. Varela and A. Medina. 2013. Spawning Behaviour and Post-Spawning Migration Patterns of Atlantic Bluefin Tuna (*Thunnus thynnus*) Ascertained from Satellite Archival Tags. *PLoS one* 8:e76445.

- Arvanitidis, C., G. Chatzigeorgiou, D. Koutsoubas, T. Kevrekidis, C. Dounas, A. Eleftheriou, P. Koulouri and A. Mogias. 2005. Estimating lagoonal biodiversity in Greece: comparison of rapid assessment techniques. *Helgoland Marine Research* 59:177-186.
- Assis, J., K. Narváez and R. Haroun. 2007. Underwater towed video: a useful tool to rapidly assess elasmobranch populations in large marine protected areas. *Journal of Coastal Conservation* 11:153-157.
- Baillie J., C. Hilton-Taylor and S. N. Stuart. 2004. 2004 IUCN red list of threatened species: a global species assessment. IUCN.
- Bakun, A. 2013. Ocean eddies, predator pits and bluefin tuna: implications of an inferred 'low risk-limited payoff' reproductive scheme of a (former) archetypical top predator test. *Fish and Fisheries* 14:424-438.
- Balaguer, P., A. Diedrich, R. Sardá, M. Fuster, B. Cañellas and J. Tintoré. 2011. Spatial analysis of recreational boating as a first key step for marine spatial planning in Mallorca (Balearic Islands, Spain). *Ocean & Coastal Management* 54:241-249.
- Balbín, R., J. L. López-Jurado, M. M. Flexas, P. Reglero, P. Vélez-Velchí, C. González-Pola, J. M. Rodríguez, A. García and F. Alemany. 2014. Interannual variability of the early summer circulation around the Balearic Islands: driving factors and potential effects on the marine ecosystem. *Journal of Marine Systems* 138:70-81.
- Balbín, R., M. d. M. Flexas, J. L. López-Jurado, M. Peña, A. Amores and F. Alemany. 2012. Vertical velocities and biological consequences at a front detected at the Balearic Sea. *Continental Shelf Research* 47:28-41.
- Ballesteros, E. 1989. Production of seaweeds in Northwestern Mediterranean marine communities: its relation with environmental factors. *Scientia Marina* 357-364.
- Ballesteros, E. 1991. Structure and dynamics of North-Western Mediterranean phytobenthic communities: a conceptual model. *Oecologia aquatica* 10:223-242.
- Ballesteros, E., X. Torras, S. Pinedo, M. Garcia, L. Mangialajo and M. De Torres. 2007. A new methodology based on littoral community cartography dominated by macroalgae for the implementation of the European Water Framework Directive. *Marine Pollution Bulletin* 55:172-180.
- Barragán Muñoz J. M. 1997. Medio ambiente y desarrollo en áreas litorales: Introducción a la planificación y gestión integradas. Oikos Tau, Barcelona, Spain.
- Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley and P. A. Doherty. 2003. Collapse and conservation of shark populations in the Northwest Atlantic. *Science* 299:389-392.
- Belhabib, D., V. Koutob, A. Sall, V. W. Lam and D. Pauly. 2014. Fisheries catch misreporting and its implications: The case of Senegal. *Fisheries Research* 151:1-11.
- Beliaeff, B. and D. Pelletier. 2011. A general framework for indicator design and use with application to the assessment of coastal water quality and marine protected area management. *Ocean & Coastal Management* 54:84-92.
- Berx, B., M. Dickey-Collas, M. D. Skogen, Y. H. De Roeck, H. Klein, R. Barciela, R. M. Forster, E. Dombrowsky, M. Huret and M. Payne. 2011. Does operational oceanography address the needs of fisheries and applied environmental scientists? *Oceanography* 24:166-171.



- Bird E. C. F. 2008. Coastal geomorphology: an introduction., Second edition. John Wiley & Sons Inc, Chichester; West Sussex, England.
- Blank, J. M., J. M. Morrisette, A. M. Landeira-Fernandez, S. B. Blackwell, T. D. Williams and B. A. Block. 2004. *In situ* cardiac performance of Pacific bluefin tuna hearts in response to acute temperature change. *Journal of Experimental Biology* 207:881-890.
- Block, B. A., I. D. Jonsen, S. J. Jorgensen, A. J. Winship, S. A. Shaffer, S. J. Bograd, E. L. Hazen, D. G. Foley, G. A. Breed and A. L. Harrison. 2011. Tracking apex marine predator movements in a dynamic ocean. *Nature* 475:86-90.
- Block, B. A., S. L. Teo, A. Walli, A. Boustany, M. J. Stokesbury, C. J. Farwell, K. C. Weng, H. Dewar and T. D. Williams. 2005. Electronic tagging and population structure of Atlantic bluefin tuna. *Nature* 434:1121-1127.
- BOE-Num-313, 2012, Orden AAA/2808/2012, de 21 de diciembre, por la que se establece un Plan de Gestión Integral para la conservación de los recursos pesqueros en el Mediterráneo afectados por las pesquerías realizadas con redes de cerco, redes de arrastre y artes fijos y menores, para el período 2013-2017. <https://www.boe.es/boe/dias/2012/12/29/pdfs/BOE-A-2012-15740.pdf>
- Boström, C., S. J. Pittman, C. Simenstad and R. T. Kneib. 2011. Seascape ecology of coastal biogenic habitats: advances, gaps and challenges. *Marine Ecology Progress Series* 427:191-217.
- Boudouresque, C. F. and M. Verlaque. 2002. Biological pollution in the Mediterranean Sea: invasive versus introduced macrophytes. *Marine Pollution Bulletin* 44:32-38.
- Bouffard, J., F. Nencioli, R. Escudier, A. M. Doglioli, A. A. Petrenko, A. Pascual, P. M. Poulain and D. Elhmaidi. 2014. Lagrangian analysis of satellite-derived currents: Application to the North Western Mediterranean coastal dynamics. *Advances in Space Research* 53:788-801.
- Bricheno, L. M., J. M. Wolf and J. M. Brown. 2014. Impacts of high resolution model downscaling in coastal regions. *Continental Shelf Research* 87:7-16.
- Burnham K. P. and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach., 2n edition. Springer-Verlag, New York.
- Burrows, M. T., R. Harvey and L. Robb. 2008. Wave exposure indices from digital coastlines and the prediction of rocky shore community structure. *MARINE ECOLOGY-PROGRESS SERIES- 353*:1.
- Butterworth, D. S. and R. A. Rademeyer. 2008. Statistical catch-at-age analysis vs. ADAPT-VPA: the case of Gulf of Maine cod. *ICES Journal of Marine Science: Journal du Conseil* 65:1717-1732.
- Caddy, J. F. and K. L. Cochrane. 2001. A review of fisheries management past and present and some future perspectives for the third millennium. *Ocean & Coastal Management* 44:653-682.
- Calleja, M. L., N. Marba and C. M. Duarte. 2007. The relationship between seagrass (*Posidonia oceanica*) decline and sulfide porewater concentration in carbonate sediments. *Estuarine Coastal and Shelf Science* 73:583-588.
- Carbonell, A., A. Tor, D. Álvarez-Berastegui, P. Vélez-Belchi, A. dos Santos, R. Balbín, and F. Alemany. 2014. Environmental driving forces determining the epipelagic decapod larval community distribution in the Balearic Sea (western Mediterranean). *Crustaceana* 87:686-714.
- Chauvet, C. and P. Francour. 1989. Les merous *Epinephelus guaza* du Parc National de Port-Cros (France): aspects socio-demographiques. *Bull.Soc.zool.Fr* 114:5-13.

- Ciannelli, L., K. Bailey, and E. M. Olsen. 2015. Evolutionary and ecological constraints of fish spawning habitats. *ICES Journal of Marine Science: Journal du Conseil* 72:285-296.
- Claudet J. 2011. *Marine protected areas: a multidisciplinary approach*. Cambridge University Press.
- Coll, J., A. Garcia-Rubies, G. Morey and A. M. Grau. 2012. The carrying capacity and the effects of protection level in three marine protected areas in the Balearic Islands (NW Mediterranean). *Scientia Marina* .
- Coll, J., A. Garcia-Rubies, G. Morey, O. Reñones, D. Álvarez-Berastegui, O. Navarro and A. M. Grau. 2013b. Using no-take marine reserves as a tool for evaluating rocky-reef fish resources in the western Mediterranean. *ICES Journal of Marine Science: Journal du Conseil* 70:578-590.
- Colton, M. A. and S. E. Swearer. 2010. A comparison of two survey methods: differences between underwater visual census and baited remote underwater video. *Marine Ecology Progress Series* 400:19-36.
- Cowan Jr, J. H., J. C. Rice, C. J. Walters, R. Hilborn, T. E. Essington, J. W. Day Jr and K. M. Boswell. 2012. Challenges for implementing an ecosystem approach to fisheries management. *Marine and Coastal Fisheries* 4:496-510.
- Cressey D. *Eyes on the ocean*. 2015. Nature Publishing Group Macmillan building, 4 Crinan st, London n1 9xw, England.
- Cury, P. M., Y. J. Shin, B. Planque, J. I. M. Durant, J. M. Fromentin, S. Kramer-Schadt, N. C. Stenseth, M. Travers and V. Grimm. 2008. Ecosystem oceanography for global change in fisheries. *Trends in Ecology & Evolution* 23:338-346.
- Cushman, S. A., K. Gutzweiler, J. S. Evans and K. McGarigal. 2010. The gradient paradigm: a conceptual and analytical framework for landscape ecology. Pages 83-110 in F. Huettmann and S. Cushman editors. *Spatial complexity, informatics and wildlife conservation*. Springer, Tokyo.
- Cushman, S. A., K. McGarigal and M. C. Neel. 2008. Parsimony in landscape metrics: strength, universality and consistency. *Ecological Indicators* 8:691-703.
- Dahlgren, C. 2014. *Review of the Benefits of No-take Zones: A Report to the Wildlife Conservation Society*. -104.
- de Juan, S., J. Moranta, H. Hinz, C. Barberá, C. Ojeda-Martinez, D. Oro, F. Ordines, E. Ólafsson, M. Demestre and E. Massutí. 2012. A regional network of sustainable managed areas as the way forward for the implementation of an Ecosystem-Based Fisheries Management in the Mediterranean. *Ocean & Coastal Management* .
- de la Gándara F., Ortega A., Blanco E., Viguri F.J., and Reglero P.. 2013. La flexión de la notocorda en larvas de atún rojo, *Thunnus thynnus* (L, 1758) cultivadas a diferentes temperaturas. Pages 181-182 in *Sociedad Española de Acuicultura editor. XIV Congreso Nacional de Acuicultura*. Universidad Laboral de Gijón, Gijón.
- DeVantier, L. M., G. De'ath, T. J. Done and E. Turak. 1998. Ecological assessment of a complex natural system: A case study from the Great Barrier Reef. *Ecological Applications* 8:480-496.
- Diaz-Almela, E., N. Marba, E. Álvarez, R. Santiago, M. Holmer, A. Grau, S. Mirto, R. Danovaro, A. Petrou, M. Argyrou, I. Karakassis and C. M. Duarte. 2008. Benthic input rates predict seagrass (*Posidonia oceanica*) fish farm-induced decline. *Marine Pollution Bulletin* 56:1332-1342.

- Dicenta, A. 1977. Zonas de puesta del atún (*Thunnus thynnus*, L) y otros túnidos del Mediterráneo occidental y primer intento de evaluación del stock de reproductores de atún. *Bol.Inst.Esp.Oceanogr* 234:109-135.
- Dicenta, A. and C. Piccinetti. 1978. Desove de atún (*Thunnus thynnus* L.) en el Mediterráneo occidental y evaluación directa del stock de reproductores, basado en la abundancia de sus larvas. *ICCAT.Col.Vol.Sci.Pap* 7:389-395.
- Dickey-Collas, M. 2014. Why the complex nature of integrated ecosystem assessments requires a flexible and adaptive approach. *ICES Journal of Marine Science: Journal du Conseil* fsu027.
- Dodge, K. L., B. Galuardi, T. J. Miller and M. E. Lutcavage. 2014. Leatherback turtle movements, dive behavior and habitat characteristics in ecoregions of the Northwest Atlantic Ocean. *PLoS one* 9:e91726.
- Druon, J. N. 2010. Habitat mapping of the Atlantic bluefin tuna derived from satellite data: Its potential as a tool for the sustainable management of pelagic fisheries. *Marine Policy* 34:293-297.
- Druon, J. N., J. M. Fromentin, F. Aulanier and J. Heikkonen. 2011. Potential feeding and spawning habitats of Atlantic bluefin tuna in the Mediterranean Sea. *Marine Ecology-Progress Series* 439:223-240.
- Duarte C.M., Marba N., Agawin N., Cebrian J., Enriquez S., Fortes M.D., Gallegos M.E., Merino M., Olesen B., Sandjensen K., Uri J. & Vermaat J. (1994) Reconstruction of Seagrass Dynamics - Age-Determinations and Associated Tools for the Seagrass Ecologist. *Marine Ecology-Progress Series* 107, 195-209
- Dungan, J. L., J. N. Perry, M. R. T. Dale, P. Legendre, S. Citron-Pousty, M. J. Fortin, A. Jakomulska, M. Miriti and M. S. Rosenberg. 2002. A balanced view of scale in spatial statistical analysis. *Ecography* 25:626-640.
- Dutra, G. F., G. R. Allen, T. Werner and S. A. McKenna. 2006. A rapid marine biodiversity assessment of the Abrolhos Bank, Bahia, Brazil. *RAP bulletin of biological assessment* 38.
- Ebert, U. and H. Welsch. 2004. Meaningful environmental indices: a social choice approach. *Journal of Environmental Economics and Management* 47:270-283.
- Ekebom, J., P. Laihonon and T. Suominen. 2003. A GIS-based step-wise procedure for assessing physical exposure in fragmented archipelagos. *Estuarine, Coastal and Shelf Science* 57:887-898.
- Essington, T. E. and A. E. Punt. 2011. Implementing Ecosystem-Based Fisheries Management: Advances, Challenges and Emerging Tools. *Fish and Fisheries* 12:123-124.
- EU. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy. European Parliament and Council. [http://ec.europa.eu/environment/water/marine/index\\_en.htm](http://ec.europa.eu/environment/water/marine/index_en.htm) Official Journal of the European Union. 2008.
- FAO. 2014. The state of world fisheries and aquaculture, Opportunities and challenges. Rome.
- Fielding, A. H. and J. F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental conservation* 24:38-49.

- FISGRW, 1998. Stream Corridor Restoration: Principles, Processes and Practices by The Federal Interagency Stream Restoration Working Group (15 Federal agencies of the Us gov't) GPO Item 0120-A ; Su Docs No.A 57.6/2=EN3/PT.653.0-93421-359-3
- Forcada, A., C. Valle, P. Bonhomme, G. Criquet, G. Cadiou, P. Lenfant and J. L. Sánchez-Lizaso. 2009. Effects of habitat on spillover from marine protected areas to artisanal fisheries. *Marine Ecology Progress Series*, 379: 197-211.
- Forman R. T. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge, UK.
- Fromentin, J. M. and J. E. Powers. 2005. Atlantic bluefin tuna: population dynamics, ecology, fisheries and management. *Fish and Fisheries* 6:281-306.
- Fromentin, J. 2009. Lessons from the past: investigating historical data from bluefin tuna fisheries. *Fish and Fisheries* 10:197-216.
- Fulton, E. A., J. S. Link, I. C. Kaplan, M. Savina-Rolland, P. Johnson, C. Ainsworth, P. Horne, R. Gorton, R. J. Gamble, A. D. M. Smith and D. C. Smith. 2011. Lessons in modelling and management of marine ecosystems: the Atlantis experience. *Fish and Fisheries* 12:171-188.
- Fulton, E. A., A. D. Smith and C. R. Johnson. 2003. Effect of complexity on marine ecosystem models.
- Game, E. T., H. S. Grantham, A. J. Hobday, R. L. Pressey, A. T. Lombard, L. E. Beckley, K. Gjerde, R. Bustamante, H. P. Possingham and A. J. Richardson. 2009. Pelagic protected areas: the missing dimension in ocean conservation. *Trends in Ecology & Evolution* 24:360-369.
- García, A., F. Alemany, J. M. De la Serna, I. Oray, S. Karakulak, L. Rollandi, A. Arigò and S. Mazzola. 2005a. Preliminary results of the 2004 bluefin tuna larval surveys off different Mediterranean sites (Balearic Archipelago, Levantine Sea and the Sicilian Channel). *Collective Volume of Scientific Papers ICCAT* 58:1261-1270.
- García, A., F. Alemany, P. Velez-Belchí, J. L. Jurado, D. Cortés, J. M. De la Serna, C. G. Pola, J. M. Rodríguez, J. Jansá and T. Ramírez. 2005b. Characterization of the bluefin tuna spawning habitat off the Balearic Archipelago in relation to key hydrographic features and associated environmental conditions. *ICCAT Collected Volume of Scientific Papers* 58:535-549.
- García S. M. 2003. The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook. *Food & Agriculture Org.*.
- García, S. M. and K. L. Cochrane. 2005. Ecosystem approach to fisheries: a review of implementation guidelines. *ICES Journal of Marine Science: Journal du Conseil* 62:311-318.
- García-Charton, J. A., A. Pérez-Ruzafa, C. Marcos, J. Claudet, F. Badalamenti, L. Benedetti-Cecchi, J. M. Falcon, M. Milazzo, P. J. Schembri, B. Stobart, F. Vandeperre, A. Brito, R. Chemello, M. Dimech, P. Domenici, I. Guala, L. Le Direach, E. Maggi and S. Planes. 2008. Effectiveness of European Atlanto-Mediterranean MPAs: Do they accomplish the expected effects on populations, communities and ecosystems? *Journal for Nature Conservation* 16:193-221.
- García-Charton, J. A., A. Pérez-Ruzafa, P. Sánchez-Jerez, J. T. Bayle-Sempere, O. Reñones and D. Moreno. 2004. Multi-scale spatial heterogeneity, habitat structure and the effect of marine reserves on Western Mediterranean rocky reef fish assemblages. *Marine Biology* 144:161-182.

- García-Charton, J. A., I. D. Williams, A. P. Ruzafa, M. Milazzo, R. Chemello, C. Marcos, M. S. Kitsos, A. Koukouras and S. Riggio. 2000. Evaluating the ecological effects of Mediterranean marine protected areas: habitat, scale and the natural variability of ecosystems. *Environmental conservation* 27:159-178.
- García-Rubies A. 1997. Estudi ecològic de les poblacions de peixos litorals sobre substrat rocós a la Mediterrània Occidental: efectes de la fondària, el substrat, l'estacionalitat i la protecció. Universitat de Barcelona, Barcelona, Spain.
- García-Rubies, A., B. Hereu and M. Zabala. 2013. Long-term recovery patterns and limited spillover of large predatory fish in a Mediterranean MPA. *PloS one* 8(9): e73922. doi:10.1371/journal.pone.0073922.
- Gillanders, B. M., K. W. Able, J. A. Brown, D. B. Eggleston and P. F. Sheridan. 2003. Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: an important component of nurseries. *Marine Ecology Progress Series*, 247: 281-295.
- Gobert, S., S. Sartoretto, V. Rico-Raimondino, B. Andral, A. Chery, P. Lejeune and P. Boissery. 2009. Assessment of the ecological status of Mediterranean French coastal waters as required by the Water Framework Directive using the *Posidonia oceanica* Rapid Easy Index: PREI. *Marine Pollution Bulletin* 58:1727-1733.
- Gordoa, A. and G. Carreras. 2014. Determination of Temporal Spawning Patterns and Hatching Time in Response to Temperature of Atlantic Bluefin Tuna (*Thunnus thynnus*) in the Western Mediterranean. *PloS one* 9:e90691.
- Gökyer E. 2013. Understanding Landscape Structure Using Landscape Metrics, *Advances in Landscape Architecture*, Dr. Murat Ozyavuz (Ed.), ISBN: 978-953-51-1167-2, InTech, DOI: 10.5772/55758. Available from: <http://www.intechopen.com/books/advances-in-landscape-architecture/understanding-landscape-structure-using-landscape-metrics>
- Gregg, E. J. and K. M. A. Chan. 2011. Making science relevant to marine ecosystem-based management. *Biological Conservation* 144:670-671.
- Guidetti, P. and E. Sala. 2007. Community-wide effects of marine reserves in the Mediterranean Sea. *Marine Ecology Progress Series* 335:43-56.
- Gustafson, E. J. 1998. Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems* 1:143-156.
- Halpern, B. S., C. V. Kappel, K. A. Selkoe, F. Micheli, C. M. Ebert, C. Kontgis, C. M. Crain, R. G. Martone, C. Shearer and S. J. Teck. 2009. Mapping cumulative human impacts to California Current marine ecosystems. *Conservation Letters* 2:138-148.
- Halpern, B. S. and R. R. Warner. 2003. Review paper. Matching marine reserve design to reserve objectives. *Proceedings of the Royal Society of London B: Biological Sciences* 270:1871-1878.
- Hamner, W. M., M. S. Jones, J. H. Carleton, I. R. Hauri and D. M. B. Williams. 1988. ZODPlankton, planktivorous fish and water currents on a windward reef face: Great Barrier Reef, Australia. *Bulletin of Marine Science* 42:459-479.
- Hargrave, B. T. 2002. A traffic light decision system for marine finfish aquaculture siting. *Ocean & Coastal Management* 45:215-235.

- Harmelin, J.-G. 1988. Structure et variabilité de l'ichtyofaune d'une zone rocheuse protégée en Méditerranée. *PSZNI Marine Ecology* 8:263-284.
- Harmelin, J. G. 1987. Structure and Variability of the Ichthyofauna in A Mediterranean Protected Rocky Area (National-Park-Of-Port-Cros, France). *Marine Ecology-Pubblicazioni Della Stazione Zoologica di Napoli I* 8:263-284.
- Harmelin, J. G., P. Robert, M. Cantou and M. Harmelin-Vivien. 2007. Long term changes in the dusky grouper (*Epinephelus marginatus*) population from a NW Mediterranean marine protected area, the national park of Port-Cros (France). in Nice University publ., May.
- Harmelin, J. G. and M. Harmelin-Vivien. 1999. A review on habitat, diet and growth of the dusky grouper *Epinephelus marginatus* (Lowe, 1834). *Marine Life* 9:11-20.
- Heinisch, G., A. Corriero, A. Medina, F. J. Abascal, J.-M. de la Serna, R. Vassallo-Agius, A. B. Ríos, A. García, F. De la Gándara and C. Fauvel. 2008. Spatial-temporal pattern of bluefin tuna (*Thunnus thynnus* L. 1758) gonad maturation across the Mediterranean Sea. *Marine Biology* 154:623-630.
- Hereu, B., D. Diaz, J. Pasqual, M. Zabala and E. Sala. 2006. Temporal patterns of spawning of the dusky grouper *Epinephelus marginatus* in relation to environmental factors. *Marine Ecology Progress Series* 325:187-194.
- Heslop, E. 2015. Unravelling high frequency and sub-seasonal variability at key ocean circulation 'choke' points: a case study from glider monitoring in the western Mediterranean sea. University of Southampton, Ocean and Earth Science, Doctoral Thesis , 248pp.
- Hidalgo, M., Y. Gusdal, G. E. Dingsor, D. Hjermann, G. Ottersen, L. C. Stige, A. Melsom and N. C. Stenseth. 2012. A combination of hydrodynamical and statistical modelling reveals non-stationary climate effects on fish larvae distributions. *Proceedings of the Royal Society B: Biological Sciences* 279:275-283.
- Hidalgo, M., P. Reglero, D. Álvarez-Berastegui, A. P. Torres, I. Álvarez, J. M. Rodriguez, A. Carbonell, N. Zaragoza, A. Tor, and R. Goñi. 2014. Hydrographic and biological components of the seascape structure the meroplankton community in a frontal system. *Mar Ecol Prog Ser* 505:65-80.
- Hidalgo, M., P. Reglero, D. Álvarez-Berastegui, A. P. Torres, I. Álvarez, J. M. Rodriguez, A. Carbonell, R. Balbín, and F. Alemany. 2015. Hidden persistence of salinity and productivity gradients shaping pelagic diversity in highly dynamic marine ecosystems. *Marine Environmental Research* 104:47-50.
- Hijmans, R. J. and J. van Etten. 2012. raster: Geographic analysis and modeling with raster data. R package version 1:9-92.
- Hilborn, R. 2011. Future directions in ecosystem based fisheries management: a personal perspective. *Fisheries Research* 108:235-239.
- Hilbe J. 2011. Negative binomial regression., Second edition. Cambridge University Press, New York.
- Hilton-Taylor C. C. 2000. 2000 IUCN red list of threatened species. IUCN, Gland, Switzerland and Cambridge, UK.
- Hinchey, E. K., M. C. Nicholson, R. N. Zajac and E. A. Irlandi. 2008a. Preface: marine and coastal applications in landscape ecology. *Landscape Ecology* 23:1-5.

- Hobday, A. J. and K. Hartmann. 2006. Near real-time spatial management based on habitat predictions for a longline bycatch species. *Fisheries Management and Ecology* 13:365-380.
- Hobday, A. J. and J. R. Hartog. 2014. Derived Ocean Features for Dynamic Ocean Management. *Oceanography* 27:134-145.
- Hobday, A. J., J. R. Hartog, T. Timmiss and J. Fielding. 2010. Dynamic spatial zoning to manage southern bluefin tuna (*Thunnus maccoyii*) capture in a multi-species longline fishery. *Fisheries Oceanography* 19:243-253.
- Hobday A. J., Maxwell S. M., Forgie J., and McDonald J. 2014. Dynamic ocean management: integrating scientific and technological capacity with law, policy, and management. *Stanford Environmental Law Journal*. Vol. 33: 125-165.
- Huntington, B. E., M. Karnauskas, E. A. Babcock and D. Lirman. 2010. Untangling natural seascape variation from marine reserve effects using a landscape approach. *PloS one* 5:e12327.
- ICCAT,2010, Summary of measures taken historically by iccat for bluefin tuna. CoP14 Doc. 68 Annex 4, 4-5. Madrid. 1-11-2010
- ICCAT. 2013. Report of the 2012 Atlantic Bluefin Tuna Stock Assessment Session. Pages 1-198 in ICCAT editor. The International Commission for the Conservation of Atlantic Tunas, Madrid-Spain.
- ICCAT. 2014. Meeting of the Standing Committee on Research and statistics, Report for biennial period 2014-15 PARTI (2014)- vol. 2. SCRS
- Ingram, G. W., F. Alemany, D. Álvarez-Berastegui and A. García. 2013. Development of indices of larval bluefin tuna (*Thunnus thynnus*) in the Western Mediterranean Sea. *ICCAT,Collect.Vol.Sci.Pap.* 69:1057-1076.
- Ingram, G. W., W. J. Richards, J. T. Lamkin and B. Muhling. 2010. Annual indices of Atlantic bluefin tuna (*Thunnus thynnus*) larvae in the Gulf of Mexico developed using delta-lognormal and multivariate models. *Aquatic Living Resources* 23:35-47.
- Ingram, G. W., W. J. Richards, C. E. Porch, V. Restrepo, J. T. Lamkin, B. Muhling, J. Lyczkowski-Shultz, G. P. Scott and S. C. Turner. 2008. Annual indices of bluefin tuna (*Thunnus thynnus*) spawning biomass in the Gulf of Mexico developed using delta-lognormal and multivariate models. *ICCAT Working Document SCRS/2008/086* .
- Itziar Álvarez. 2015. Multiscale environment-ichthyoplankton assemblages relationships in the balearic sea. PhD Thesis, Universitat de les Illes Balears. Palma de Mallorca, Spain
- Jackson, J. B. 1997. Reefs since columbus. *Coral Reefs* 16:S23-S32.
- Jackson, J. B. 2001. What was natural in the coastal oceans? *Proceedings of the National Academy of Sciences* 98:5411-5418.
- Jackson, J. B., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson and J. A. Estes. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293:629-637.
- Juza, M., B. Mourre, J. M. Lellouche, M. Tonani and J. Tintoré. 2015. From basin to sub-basin scale assessment and intercomparison of numerical simulations in the Western Mediterranean Sea. *Journal of Marine Systems* 149:36-49.

- Juza, M., L. Renault, S. Ruiz and J. Tintoré. 2013. Origin and pathways of Winter Intermediate Water in the Northwestern Mediterranean Sea using observations and numerical simulation. *Journal of Geophysical Research: Oceans* 118:6621-6633.
- Kelleher G. 1999. Guidelines for marine protected areas. IUCN, Gland, Switzerland and Cambridge, UK.
- Kendall, M. S., T. J. Miller and S. J. Pittman. 2011. Patterns of scale-dependency and the influence of map resolution on the seascape ecology of reef fish. *Marine Ecology-Progress Series* 427:259-274.
- Kenny, A. J., I. Cato, M. Desprez, G. Fader, R. T. E. Schttünhelm and J. Side. 2003. An overview of seabed-mapping technologies in the context of marine habitat classification. *ICES Journal of Marine Science: Journal du Conseil* 60:411-418.
- Kiflawi, M. and A. Genin. 1997. Prey flux manipulation and the feeding rates of reef-dwelling planktivorous fish. *Ecology* 78:1062-1077.
- Kimmel J.J. (1985) A new species-time method for visual assessment of fishes and its comparison with established methods. *Environmental Biology of Fishes* 12, 23-32
- Koched, W., A. Hattour, F. Alemany, A. Garcia, and K. Said. 2013. Spatial distribution of tuna larvae in the Gulf of Gabes (Eastern Mediterranean) in relation with environmental parameters. *Mediterranean Marine Science* 14:5-14.
- Koeck, B., J. Pastor, G. Saragoni, N. Dalias, J. Payrot and P. Lenfant. 2014. Diel and seasonal movement pattern of the dusky grouper *Epinephelus marginatus* inside a marine reserve. *Marine Environmental Research* 94:38-47.
- Kramer, P. A. 2003. Synthesis of coral reef health indicators for the western Atlantic: Results of the AGRRA program(1997-2000). *Atoll Research Bulletin* 496:1-57.
- La Mesa, G., P. Louisy and M. Vacchi. 2002. Assessment of microhabitat preferences in juvenile dusky grouper (*Epinephelus marginatus*) by visual sampling. *Marine Biology* 140:175-185.
- La Violette, P. E., J. Tintoré and J. Font. 1990. The surface circulation of the Balearic Sea. *Journal of Geophysical Research* 95:1559-1568.
- Lauck, T., C. W. Clark, M. Mangel and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. *Ecological Applications* 8:S72-S78.
- Lehodey, P., I. Senina and R. Murtugudde. 2008. A spatial ecosystem and populations dynamics model (SEAPODYM) Modeling of tuna and tuna-like populations. *Progress in Oceanography* 78:304-318.
- Lembo, G., I. Fleming, F. Okland, P. Carbonara and M. T. Spedicato. 1999. Homing behaviour and site fidelity of *Epinephelus marginatus* (Lowe, 1834) around the island of Ustica: Preliminary results from a telemetry study. *Biología Marina del Mediterraneo* 6:90-99.
- Lembo, G., M. T. Spedicato, F. Okland, P. Carbonara, I. A. Fleming, R. S. McKinley, E. B. Thorstad, M. Sisak and S. Ragonese. 2002. A wireless communication system for determining site fidelity of juvenile dusky groupers *Epinephelus marginatus* (Lowe, 1834) using coded acoustic transmitters. *Hydrobiologia* 483:249-257.



- Levin, S. A. 1992. The problem of pattern and scale in ecology: the Robert H. MacArthur award lecture. *Ecology* 73:1943-1967.
- Lewison, R. L., S. A. Freeman and L. B. Crowder. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. *Ecology letters* 7:221-231.
- Linde, M., A. M. Grau, F. Riera, and E. Massutí-Pascual. 2004. Analysis of trophic ontogeny in *Epinephelus marginatus* (Serranidae). *Cybium* 28:27-35
- Lindo-Atichati, D., F. Bringas, G. Goni, B. Muhling, F. E. Muller-Karger and S. Habtes. 2012. Varying mesoscale structures influence larval fish distribution in the northern Gulf of Mexico. *Marine Ecology Progress Series* 463:245-257.
- Link, J. S. 2002. Ecological considerations in fisheries management: when does it matter? *Fisheries* 27:10-17.
- Liquete, C., C. Piroddi, E. G. Drakou, L. Gurney, S. Katsanevakis, A. Charef and B. Egoh. Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review.
- Llopiz, J. K., and A. J. Hobday. 2015. A global comparative analysis of the feeding dynamics and environmental conditions of larval tunas, mackerels, and billfishes. *Deep Sea Research Part II: Topical Studies in Oceanography* 113:113-124.
- Louisy, P., Ganteaume A. and Francour P. 2007. Les relations des espèces de mérous à leur habitat *Epinephelus marginatus*, *E. costae* et *Mycteroperca rubra* dans la région de Kas, Turquie, Méditerranée Orientale. Pages 121-123, 2nd Symposium on Mediterranean Groupers , Francour P., Gratiot J. (eds). Nice, May 10th – 13th 2007
- Louzao, M., D. Pinaud, C. Peron, K. Delord, T. Wiegand and H. Weimerskirch. 2011. Conserving pelagic habitats: seascape modelling of an oceanic top predator. *Journal of Applied Ecology* 48:121-132.
- Luckhurst B.E. & Luckhurst K. (1978) Analysis of the influence of substrate variables on coral reef fish communities. *Marine Biology* 49, 317-323
- Machado, A. 2004. An index of naturalness. *Journal for Nature Conservation* 12:95-110.
- Machado, L. F., Á. A. Bertoncini, M. Hostim-Silva and J. P. Barreiros. 2003. Habitat use by the juvenile dusky grouper *Epinephelus marginatus* and its relative abundance, in Santa Catarina, Brazil. *aqua, Journal of Ichthyology and Aquatic Biology* 6:133-138.
- MacKenzie, B. R. and P. Mariani. 2012. Spawning of Bluefin Tuna in the Black Sea: Historical Evidence, Environmental Constraints and Population Plasticity. *PloS one* 7:e39998.
- Mallol, S., Á. Mateo-Ramírez, F. Alemany, D. Álvarez-Berastegui, D. Díaz, J. L. López-Jurado, and R. Goñi. 2014. Abundance and distribution of scyllarid phyllosoma larvae (Decapoda: Scyllaridae) in the Balearic Sea (Western Mediterranean). *Journal of Crustacean Biology* 34:442-452.
- Manderson, J., L. Palamara, J. Kohut and M. J. Oliver. 2011. Ocean observatory data are useful for regional-habitat modeling of species with different vertical habitat preferences. *Marine Ecology Progress Series* 438:1-17.

- Mannocci, L., S. Laran, P. Monestiez, G. Dorémus, O. Van Canneyt, P. Watremez and V. Ridoux. 2013. Predicting top predator habitats in the Southwest Indian Ocean. *Ecography* 37:261-278.
- Marba N., Duarte C.M., Holmer M., Martinez R., Basterretxea G., Orfila A., Jordi A. & Tintore J. (2002) Effectiveness of protection of seagrass (*Posidonia oceanica*) populations in Cabrera National Park (Spain). *Environmental Conservation* 29, 509-518
- Mason, E. and A. Pascual. 2013. Multiscale variability in the Balearic Sea: An altimetric perspective. *Journal of Geophysical Research: Oceans* 118:3007-3025.
- Massuti, E. and O. Renones. 2005. Demersal resource assemblages in the trawl fishing grounds off the Balearic Islands (western Mediterranean). *Scientia Marina* 69:167-181.
- Mather F. J., J. M. Mason and A. C. Jones. 1995. Historical document: life history and fisheries of Atlantic bluefin tuna. US Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.
- Mayol J., Grau A.M., Riera F. & Oliver J. Llista vermella dels peixos de les Balears. [4], -126. 2011. Balears, Spain, Conselleria d'Agricultura i Pesca. Govern de les Illes Balears. Quadern de pesca.
- McGehee, M. A. 1994. Correspondence Between Assemblages of Coral-Reef Fishes and Gradients of Water Motion, Depth and Substrate Size Off Puerto-Rico. *Marine Ecology-Progress Series* 105:243-255.
- McKenna, S. A. 2011. History and overview of Marine RAP. Pages 74-79 in *Still Counting: Biodiversity Exploration for Conservation – The First 20 Years of the Rapid Assessment Program*. Conservation International, Arlington, VA, USA.
- McKenna, S. A., G. R. Allen and S. Suryadi. 2002. A marine rapid assessment of the Raja Ampat islands, Papua Province, Indonesia. *RAP bulletin of biological assessment* 22:1-193.
- McLeod K. and H. Leslie. 2009. *Ecosystem-based management for the oceans*. Island Press, Washington.
- Medina, A., F. J. Abascal, C. Megina, and A. Garcia. 2002. Stereological assessment of the reproductive status of female Atlantic northern bluefin tuna during migration to Mediterranean spawning grounds through the Strait of Gibraltar. *Journal of Fish Biology* 60:203-217.
- Mellin, C., Andréfouët, A., and Ponton, D. 2007. Spatial predictability of juvenile fish species richness and abundance in a coral reef environment. *Coral Reefs*, 26: 895-907.
- Mouillot, D., J. M. Culioli and T. D. Chi. 2002. Indicator species analysis as a test of non-random distribution of species in the context of marine protected areas. *Environmental conservation* 29:385-390.
- Muhling, B. A., J. T. Lamkin and M. A. Roffer. 2010. Predicting the occurrence of Atlantic bluefin tuna (*Thunnus thynnus*) larvae in the northern Gulf of Mexico: building a classification model from archival data. *Fisheries Oceanography* 19:526-539.
- Muhling, B. A., S. K. Lee, J. T. Lamkin and Y. Liu. 2011. Predicting the effects of climate change on bluefin tuna (*Thunnus thynnus*) spawning habitat in the Gulf of Mexico. *ICES Journal of Marine Science: Journal du Conseil* 68:1051-1062.

- Muhling, B. A., P. Reglero, L. Ciannelli, D. Álvarez-Berastegui, F. Alemany, J. T. Lamkin and M. A. Roffer. 2013. Comparison between environmental characteristics of larval bluefin tuna *Thunnus thynnus* habitat in the Gulf of Mexico and western Mediterranean Sea. *Marine Ecology Progress Series* 486:257-276.
- Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. da Fonseca and J. Kent. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403:853-858.
- Myers, R. A. and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283.
- Nobre, A. M. and J. G. Ferreira. 2009. Integration of ecosystem-based tools to support coastal zone management. *Journal of Coastal Research* 2:1676-1680.
- NOAA (National Oceanic and Atmospheric Administration). 1996. Magnuson-Stevens Fishery Management and Conservation Act amended through 11 October 1996. NOAA Technical Memorandum NMFS-F/SPO-23
- Ojeda-Martinez, C., F. G. Casalduero, J. T. Bayle-Sempere, C. B. Cebrian, C. Valle, J. L. Sanchez-Lizaso, A. Forcada, P. Sanchez-Jerez, P. Martin-Sosa, J. M. Falcon, F. Salas, M. Graziano, R. Chemello, B. Stobart, P. Cartagena, A. Perez-Ruzafa, F. Vandeperre, E. Rochel, S. Planes and A. Brito. 2009. A conceptual framework for the integral management of marine protected areas. *Ocean & Coastal Management* 52:89-101.
- Oray, I. K. and F. S. Karakulak. 2005. Further evidence of spawning of bluefin tuna (*Thunnus thynnus* L., 1758) and the tuna species (*Auxis rochei* Ris., 1810, *Euthynnus alletteratus* Raf., 1810) in the eastern Mediterranean Sea: preliminary results of TUNALEV larval survey in 2004. *Journal of Applied Ichthyology* 21:236-240.
- Ordines, F., J. Moranta, M. Palmer, A. Lerycke, A. Suau, B. Morales-Nin and A. M. Grau. 2005. Variations in a shallow rocky reef fish community at different spatial scales in the western Mediterranean Sea. *Marine Ecology Progress Series* 304:221-233.
- Ortiz de Urbina, J., J. M. Fromentin, V. R. Restrepo, H. Arrizabalaga and J.-M. de la Serna. 2007. Standardized CPUE of Bluefin Tuna (*Thunnus thynnus*) caught by Spanish traps for the period 1981-2004. *Collective Volume of Scientific Papers* 60:913-927.
- Ostrom, E. 2009. A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science* 325:419-422.
- Pascual, A., J. Bouffard, S. Ruiz, B. Buongiorno Nardelli, E. Vidal-Vijande, R. Escudier, J. M. Sayol and A. Orfila. 2013. Recent improvements in mesoscale characterization of the western Mediterranean Sea: synergy between satellite altimetry and other observational approaches. *Scientia Marina* 77:19-36.
- Pascual, A., M. I. Pujol, G. Larnicol, P. Y. Le Traon and M. H. I. n. Rio. 2007. Mesoscale mapping capabilities of multisatellite altimeter missions: First results with real data in the Mediterranean Sea. *Journal of Marine Systems* 65:190-211.
- Palumbi, S. R. 2004. Marine reserves and ocean neighbourhoods: the spatial scale of marine populations and their management. *Annu. Rev. Environ. Resour.* 29: 31-68.
- Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends in ecology and evolution* 10:430.

- Pauly, D., V. Christensen, J. Dalsgaard, R. Froese and F. Torres. 1998. Fishing down marine food webs. *Science* 279:860-863.
- Pauly, D., V. Christensen, S. Guénette, T. J. Pitcher, U. R. Sumaila, C. J. Walters, R. Watson and D. Zeller. 2002. Towards sustainability in world fisheries. *Nature* 418:689-695.
- Pérez-Ruzafa, A., C. Marcos, J. A. García-Charton and F. Salas. 2008. European marine protected areas (MPAs) as tools for fisheries management and conservation. *Journal for Nature Conservation* 16:187-192.
- Phillips, A. J., L. Ciannelli, R. D. Brodeur, W. G. Pearcy and J. Childers. 2014. Spatio-temporal associations of albacore CPUEs in the Northeastern Pacific with regional SST and climate environmental variables. *ICES, Journal of Marine Science* in press.
- Piccinetti, C. 1994. Distribution des larves de Thonides en Méditerranée. *FAO Fisheries Report (FAO)*. FAO.
- Piccinetti, C., G. Piccinetti-Manfrin and S. SoRO. 1997. Résultats d'une campagne de recherche sur les larves de thonidés en Méditerranée. *Collective volume of scientific papers-international commission for the conservation of Atlantic tunas* 46:207-214.
- Piccinetti, C., G. Piccinetti-Manfrin and S. SoRO. 1999. Larve di tunnidi in Mediterraneo. *Biologia marina mediterranea* 6:229.
- Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty and B. Heneman. 2004. Ecosystem-based fishery management. *Science (Washington)* 305:346-347.
- Pitcher, T. J., D. Kalikoski, K. Short, D. Varkey and G. Pramod. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy* 33:223-232.
- Pittman, S. J. and K. A. Brown. 2011. Multi-Scale Approach for Predicting Fish Species Distributions across Coral Reef Seascapes. *PloS one* 6.
- Pittman, S. J., R. T. Kneib and C. A. Simenstad. 2011. Practicing coastal seascape ecology. *Marine Ecology-Progress Series* 427:187-190.
- Pittman, S. J., C. A. McAlpine and K. M. Pittman. 2004. Linking fish and prawns to their environment: a hierarchical landscape approach. *Marine Ecology-Progress Series* 283:233-254.
- Pollnac, R., P. Christie, J. E. Cinner, T. Dalton, T. M. Daw, G. E. Forrester, N. A. J. Graham and T. R. McClanahan. 2010. Marine reserves as linked social-ecological systems. *Proceedings of the National Academy of Sciences of the United States of America* 107:18262-18265.
- Polunin, N. V. C. and C. M. Roberts. 1993. Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. *Marine Ecology-Progress Series* 100:167.
- Pomeroy R. S., J. E. Parks and L. M. Watson. 2004. *How is your MPA doing?: a guidebook of natural and social indicators for evaluating marine protected area management effectiveness*. IUCN Gland, Switzerland.
- Posidonia-LIFE-Project. LIFE 00/NAT/E/7303. [http://lifeposidonia.caib.es/user/index\\_en.htm](http://lifeposidonia.caib.es/user/index_en.htm). 2001.
- Post J. C., C. G. Lundin and B. Mundial. 1996. *Guidelines for integrated coastal zone management*. World Bank Washington, DC.

- Planes, S. 2005. Final report BIOMEX (Assessment of biomass export from marine protected areas and its impacts on fisheries in the Western Mediterranean Sea) Project: QLRT-2001-0891. BIOMEX Perpignan. BIOMEX Perpignan, available at .  
[http://www.medmpaforum2012.org/sites/default/files/final\\_report\\_biomex.pdf](http://www.medmpaforum2012.org/sites/default/files/final_report_biomex.pdf)
- R Development Core Team. 2008. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Rayner, R. 2010. The us integrated ocean observing system in a global context. *Marine Technology Society Journal* 44:26-31.
- RDevelopment, C. O. R. E. 2011. TEAM. 2008. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051.
- Reglero, P., L. Ciannelli, D. Álvarez-Berastegui, R. Balbín, J. L. López-Jurado and F. Alemany. 2012. Geographically and environmentally driven spawning distributions of tuna species in the western Mediterranean Sea. *Marine Ecology Progress Series* 463:273-284.
- Reglero, P., D. P. Tittensor, D. Álvarez-Berastegui, A. Aparicio-González and B. Worm. 2014. Worldwide distributions of tuna larvae: revisiting hypotheses on environmental requirements for spawning habitats. *Marine Ecology Progress Series* 501:207-224.
- Reglero, P., A. Urtizberea, A. P. Torres, F. Alemany and Ö. Fiksen. 2011. Cannibalism among size classes of larvae may be a substantial mortality component in tuna. *Marine Ecology Progress Series* 433:205-219.
- Rempel, R. S., D. Kaukinen and A. P. Carr. 2012. Patch analyst and patch grid. Ontario Ministry of Natural Resources. Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario .
- Reñones O., Álvarez-Berastegui D., Coll J., Morey G., Rueda L., Grau A., Stobart B., Díaz D., Box A., Deudero S., Maria Grau A. and Goñi R., 2012. Identificación del patrón de movimientos y factores ambientales que determinan la distribución espacial del mero *epinephelus marginatus* en el parque nacional marítimo-terrestre del archipiélago de Cabrera: aplicaciones para su conservación. Pages 407-430 in *Proyectos de Investigación en Parques Nacionales 2008-2011*. Organismo Autónomo de Parques Nacionales, Madrid.
- Reñones, O., C. Piñeiro, X. Mas and R. Goñi. 2007. Age and growth of the dusky grouper *Epinephelus marginatus* (Lowe 1834) in an exploited population of the western Mediterranean Sea. *Journal of Fish Biology* 71:346-362.
- Reñones, O., N. V. C. Polunin, and R. Goñi. 2002. Size related dietary shifts of *Epinephelus marginatus* in a western Mediterranean littoral ecosystem: an isotope and stomach content analysis. *Journal of Fish Biology* 61:122-137.
- Reñones, O., R. Goñi, M. Pozo, S. Deudero and J. Moranta. 1999. Effects of protection on the demographic structure and abundance of *Epinephelus marginatus* (Lowe, 1834). Evidence from Cabrera Archipelago National Park (West-central Mediterranean). *Marine Life* 9.
- Reñones, O., J. Moranta, M. Coll and B. Morales-Nin. 1997. Rocky bottom fish communities of Cabrera Archipelago national park (Mallorca, western Mediterranean). *Scientia Marina* 61:495-506.
- Rice, J. 2011. Managing fisheries well: delivering the promises of an ecosystem approach. *Fish and Fisheries* 12:209-231.

- Riera, F., S. Pou and Grau A.M. 1993. La Ictiofauna. Pages 244-263 in J.A.Alcover, E.Ballesteros and J.J.Fornós editors. Història Natural de L'Arxipèlag de Cabrera. CSIC-Moll. Mallorca.
- Rietkerk, M. G., J. v. d. Koppel, L. Kumar, F. v. Langevelde and H. H. Prins. 2002. The ecology of scale. *Ecological Modelling* 149:1-4.
- Rio, M. H., A. Pascual, P.-M. Poulain, M. Menna, B. Barceló-Llull and J. Tintoré. 2014. Computation of a new mean dynamic topography for the Mediterranean Sea from model outputs, altimeter measurements and oceanographic *in situ* data. *Ocean Science* 10:731-744.
- Roberts, C. M., S. Andelman, G. Branch, R. H. Bustamante, J. C. Castilla, J. Dugan, B. S. Halpern, K. D. Lafferty, H. Leslie, J. Lubchenco, D. McArdle, H. P. Possingham, M. Ruckelshaus and R. R. Warner. 2003a. Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications* 13:S199-S214.
- Roberts, C. M. 2000. Selecting marine reserve locations: optimality versus opportunism. *Bulletin of Marine Science* 66:581-592.
- Roberts, C. M., S. Andelman, G. Branch, R. H. Bustamante, J. Carlos Castilla, J. Dugan, B. S. Halpern, K. D. Lafferty, H. Leslie and J. Lubchenco. 2003b. Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications* 13:199-214.
- Roberts, C. M., J. A. Bohnsack, F. Gell, J. P. Hawkins and R. Goodridge. 2001. Effects of marine reserves on adjacent fisheries. *Science* 294:1920-1923.
- Rodriguez, J. M., I. Álvarez, J. L. Lopez-Jurado, A. Garcia, R. Balbín, D. Álvarez-Berastegui and F. Alemany. 2013. Environmental forcing and the larval fish community associated to the Atlantic bluefin tuna spawning habitat of the Balearic region (Western Mediterranean), in early summer 2005. *Deep Sea Research Part I: Oceanographic Research Papers* 77:11-22.
- Rodriguez-Marin, E., M. Ruiz, B. Pérez, P. Quelle, P. L. Luque and J. O. de Urbina. 2013. Have the atlantic bluefin tuna management measures influenced the age composition of the bay of biscay baitboat catches? *Collect.Vol.Sci.Pap.ICCAT* 69:252-258.
- Romero, J., B. Martinez-Crego, T. Alcoverro and M. Perez. 2007. A multivariate index based on the seagrass *Posidonia oceanica* (POMI) to assess ecological status of coastal waters under the framework directive (WFD) (vol 55, pg 196, 2007). *Marine Pollution Bulletin* 54:631.
- Rooker, J. R., D. H. Secor, G. De Metrio, R. Schloesser, B. A. Block and J. D. Neilson. 2008. Natal homing and connectivity in Atlantic bluefin tuna populations. *Science* 322:742-744.
- Rooker, J. R., J. R. Simms, R. D. Wells, S. A. Holt, G. J. Holt, J. E. Graves and N. B. Furey. 2012. Distribution and habitat associations of billfish and swordfish larvae across mesoscale features in the Gulf of Mexico. *PloS one* 7:e34180.
- Rooper, C. N. and M. Zimmermann. 2007. A bottom-up methodology for integrating underwater video and acoustic mapping for seafloor substrate classification. *Continental Shelf Research* 27:947-957.
- Rosenberg, A., T. E. Bigford, S. Leathery, R. L. Hill and K. Bickers. 2000. Ecosystem approaches to fishery management through essential fish habitat. *Bulletin of Marine Science* 66:535-542.
- Ruiz, J., D. Macías, M. M. Rincón, A. Pascual, I. A. Catalán and G. Navarro. 2013. Recruiting at the edge: kinetic energy inhibits anchovy populations in the Western Mediterranean. *PloS one* 8.

- Sabatés, A. and L. Recasens. 2001. Seasonal distribution and spawning of small tunas (*Auxis rochei* and *Sarda sarda*) in the northwestern Mediterranean. *Scientia Marina* 65:95-100.
- Sala, E. 2004. The past and present topology and structure of Mediterranean subtidal rocky-shore food webs. *Ecosystems* 7:333-340.
- Sala, E., E. Ballesteros, P. Dendrinou, A. Di Franco, F. Ferretti, D. Foley, S. Fraschetti, A. Friedlander, J. Garrabou and G. Harun. 2012. The structure of Mediterranean rocky reef ecosystems across environmental and human gradients and conservation implications. *PLoS one* 7:e32742.
- Sala, E., C. Costello, D. Dougherty, G. Heal, K. Kelleher, J. H. Murray, A. A. Rosenberg and R. Sumaila. 2013. A general business model for marine reserves. *PLoS one* 8:e58799.
- Sala, E. and N. Knowlton. 2006. Global marine biodiversity trends. *Annu.Rev.Environ.Resour.* 31:93-122.
- Scales, K. L., P. I. Miller, L. A. Hawkes, S. N. Ingram, D. W. Sims and S. C. Votier. 2014. On the Front Line: frontal zones as priority at-sea conservation areas for mobile marine vertebrates. *Journal of Applied Ecology* 51:1575-1583.
- Schiermeier, Q. 2002. Fisheries science: how many more fish in the sea? *Nature* 419:662-665.
- Schipper, J., J. S. Chanson, F. Chiozza, N. A. Cox, M. Hoffmann, V. Katariya, J. Lamoreux, A. S. Rodrigues, S. N. Stuart and H. J. Temple. 2008. The status of the world's land and marine mammals: diversity, threat and knowledge. *Science* 322:225-230.
- SCICOM (Steering Group On Ecosystem Pressures And Impacts ) 2014, Interim Report of the Working Group on Multispecies Assessment Methods (WGSAM), ICES WGSAM REPORT 2014, ICES CM 2014/SSGSUE:11
- Scott, G. P., S. C. Turner, G. B. Churchill, W. J. Richards and E. B. Brothers. 1993. Indices of larval bluefin tuna, *Thunnus thynnus*, abundance in the Gulf of Mexico; modelling variability in growth, mortality and gear selectivity. *Bulletin of Marine Science* 53:912-929.
- Shannon, C. E and W. Weaver. 1948. A mathematical theory of communication. *Bell Syst. Technol. J.*, 27, 379-423
- Sheehan, E. V., T. F. Stevens and M. J. Attrill. 2010. A quantitative, non-destructive methodology for habitat characterisation and benthic monitoring at offshore renewable energy developments. *PLoS one* 5:e14461.
- Shillinger, G. L., D. M. Palacios, H. Bailey, S. J. Bograd, A. M. Swithenbank, P. Gaspar, B. P. Wallace, J. R. Spotila, F. V. Paladino and R. Piedra. 2008. Persistent leatherback turtle migrations present opportunities for conservation. *PLoS biology* 6:e171.
- Sluka, R., M. Chiappone, K. M. Sullivan and R. Wright. 1997. The benefits of a marine fishery reserve for Nassau grouper *Epinephelus striatus* in the central Bahamas. Pages 1961-1964 in *Citesee*.
- Spedicato, M. T., P. Carbonara and G. Lembo. 2005. Insight into the homing behaviour of the dusky grouper (*Epinephelus marginatus* Lowe, 1834) around the island of Ustica, Italy. Pages 103-109 in.
- Steele, J. H. 1989. The ocean landscape. *Landscape Ecology* 3:185-192.
- Steele, J. H. and M. Schumacher. 2000. Ecosystem structure before fishing. *Fisheries Research* 44:201-205.

- Steiner R. Lebanon oil spill rapid assessment and response mission. Final Report, September 11, 2006. 2006.
- Stewart D. W., P. N. Shamdasani and D. W. Rook. 2007. Focus groups: Theory and practice., 20 edition. Sage Publications, Inc.
- Stobart, B., J. A. García-Charton, C. Espejo, E. Rochel, R. Goñi, O. Reñones, A. Herrero, R. Crec'hriou, S. Polti and C. Marcos. 2007. A baited underwater video technique to assess shallow-water Mediterranean fish assemblages: Methodological evaluation. *Journal of Experimental Marine Biology and Ecology* 345:158-174.
- Stobart, B., D. Álvarez-Berestegui and R. Goñi. 2012. Effect of habitat patchiness on the catch rates of a Mediterranean coastal bottom long-line fishery. *Fisheries Research* .
- Swan, J. and D. Gréboval. 2005. Overcoming factors of unsustainability and overexploitation in fisheries: selected papers on issues and approaches: international workshop on the implementation of international fisheries instruments and factors of unsustainability and overexploitation in fisheries, Siem Reap, Cambodia, 13-16 September 2004. *FAO fisheries report* (ISSN 0429-9337).
- Tegner, M. J. and P. K. Dayton. 1999. Ecosystem effects of fishing. *Trends in Ecology & Evolution* 14:261-262.
- Tintoré, J., B. Casas, E. Heslop, G. Vizoso, A. Pascual, A. Orfila, S. Ruiz, L. Renault, M. Juzá and P. Balaguer, et al. 2013. The Impact of New Multi-platform Observing Systems in Science, Technology Development and Response to Society Needs; from Small to Large Scales. Pages 341-348 in *Computer Aided Systems Theory-EUROCAST 2013*. Springer.
- Tintore, J., G. Vizoso, B. Casas, E. Heslop, A. Pascual, A. Orfila, S. Ruiz, M. Martinez-Ledesma, M. Torner, S. Cusi. et al. 2013. SOCIB: The Balearic Islands Coastal Ocean Observing and Forecasting System, Responding to Science, Technology and Society Needs. *Marine Technology Society Journal* 47:101-117.
- Tonani M., A.Teruzzi, G.Korres, N.Pinardi, A.Crise, M.Adani, P.Oddo, S.Dobricic, C.Fratianni, M.Drudi, S.Salon, A.Grandi, G.Girardi and V.Lyubartsev and S.Marino. 2014. The Mediterranean Monitoring and Forecasting Centre, a component of the MyOcean System. Page 13628 in H.Dahlin and N.C.Fleming and S.E.Petersson editors. Eurogoos, Sopot, Poland.
- Torres, A. P., P. Reglero, R. Balbin, A. Urtizberea and F. Alemany. 2011. Coexistence of larvae of tuna species and other fish in the surface mixed layer in the NW Mediterranean. *Journal of plankton research* 33:1793-1812.
- Turner, M. G. 1989. Landscape Ecology - the Effect of Pattern on Process. *Annual Review of Ecology and Systematics* 20:171-197.
- Turner, M. G., R. V. O'Neill, R. H. Gardner and B. T. Milne. 1989. Effects of changing spatial scale on the analysis of landscape pattern. *Landscape Ecology* 3:153-162.
- Turner M. G., R. H. Gardner and R. V. O'Neill. 2001. *Landscape ecology in theory and practice: pattern and process*. Springer Verlag.
- Tveit M., Ode A. & Fry G. (2006) Key concepts in a framework for analyzing visual landscape character. *Landscape Research* 31, 229-255



- US DOC/NOAA/NMFS. Final Amendment 7 to the 2006 Consolidated Highly Migratory Species Fishery Management Plan. 2014. Highly Migratory Species Management Division-Office of Sustainable Fisheries-National Marine Fisheries Service.
- Vacchi M., Montamari B., La Mesa G. & Cattaneo-Viatti R. The dusky grouper of the Portofino marine reserve: a first assessment of size distribution, habitat preferences and other biological features. Francour, P. & Gradiot J. 2007. Nice, Nice University publ. Symposium on the Mediterranean Groupers.
- Valavanis V. D. 2009. Essential fish habitat mapping in the Mediterranean., 203 edition. Springer Science & Business Media.
- Vella, P., R. E. Bowen and A. Frankic. 2009. An evolving protocol to identify key stakeholder-influenced indicators of coastal change: the case of Marine Protected Areas. ICES Journal of Marine Science: Journal du Conseil 66:203-213.
- Villa, F., L. Tunesi and T. Agardy. 2002. Zoning Marine Protected Areas through Spatial Multiple-Criteria Analysis: the Case of the Asinara Island National Marine Reserve of Italy. Conservation Biology 16:515-526.
- Volpe, G., R. Santoleri, V. Vellucci, M. R. d'Alcalá, S. Marullo and F. d'Ortenzio. 2007. The colour of the Mediterranean Sea: Global versus regional bio-optical algorithms evaluation and implication for satellite chlorophyll estimates. Remote Sensing of Environment 107:625-638.
- Watson, R. and D. Pauly. 2001. Systematic distortions in world fisheries catch trends. Nature 414:534-536.
- Wedding, L. M., C. A. Lepczyk, S. J. Pittman, A. M. Friedlander and S. Jorgensen. 2011. Quantifying seascape structure: extending terrestrial spatial pattern metrics to the marine realm. Marine Ecology-Progress Series 427:219-232.
- Wheeler, W. N. 1980. Effect of Boundary-Layer Transport on the Fixation of Carbon by the Giant-Kelp *Macrocystis-Pyrifera*. Marine Biology 56:103-110.
- White M., Haxhiu I., Kouroutos V., Gace A., Vaso A., Beqiraj S., Plytas A. & Dedej Z. Rapid Assessment Survey of important marine turtle and monk seal habitats in the coastal area of Albania, October-November 2005. Report to UNEP MAP RAC/SPA . 2005.
- Wiens, J. A. 1989. Spatial Scaling in Ecology. Functional ecology 3:385-397.
- Wilkinson C., D. Souter and J. Goldberg. 2006. Status of coral reefs in tsunami affected countries: 2005. Australian Institute of Marine Science, Townsville, Queensland.
- Wood S. N. 2006. Generalized additive models: an introduction with R., 66 edition. Chapman & Hall, CRC, Boca Raton, Florida.
- Worm, B., M. Sandow, A. Oschlies, H. K. Lotze and R. A. Myers. 2005. Global patterns of predator diversity in the open oceans. Science 309:1365-1369.
- Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. Jackson, H. K. Lotze, F. Micheli and S. R. Palumbi. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314:787-790.

- Wu, J. G. and H. B. Li. 2006. Perspectives and methods of scaling. Pages 17-44 in Jianguo Wu, K. Bruce Jones, Harbin Li And Orié L. Loucks editors. *Scaling and Uncertainty Analysis in Ecology*. Springer, Netherlands.
- Wu, J. J. 2013. Landscape ecology. Pages 179-200 in Rik Leemans editor. *Ecological Systems*. Springer, New York.
- WWF. WWF position paper on the Meeting of International Commission for the Conservation of Atlantic Tunas (ICCAT). World Wildlife Fund for Nature . 2014.
- Zabala, M., A. García-Rubies, P. Louisy and E. Sala. 1997. Spawning behaviour of the Mediterranean dusky grouper *Epinephelus marginatus* (Lowe, 1834)(Pisces, Serranidae) in the Medes Islands Marine Reserve (NW Mediterranean; Spain). *Scientia Marina* 61:65-77.
- Zaragoza, N., A. Quetglas, M. Hidalgo, D. Álvarez-Berastegui, R. Balbín, and F. Alemany. 2015. Effects of contrasting oceanographic conditions on the spatiotemporal distribution of Mediterranean cephalopod paralarvae. *Hydrobiologia* 749:1-14.
- Ziegeler, S. B., J. D. Dykes, and J. F. Shriver. 2012. Spatial error metrics for oceanographic model verification. *Journal of Atmospheric and Oceanic Technology* 29:260-266.
- Zuur A., E. N. Ieno, N. Walker, A. A. Saveliev and G. M. Smith. 2009. *Mixed effects models and extensions in ecology with R*. Springer Science & Business Media.
- Zuur, A. F., E. N. Ieno and C. S. Elphick. 2010. A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution* 1:3-14.