Master‘s thesis

Impact assessment of invasive flora species in *Posidonia oceanica* meadows on fish assemblage: an influence on local fisheries? The case study of Lipsi Island, Greece.

Victor Buchet

Advisor: Michael Honeth

University of Akureyri
Faculty of Business and Science
University Centre of the Westfjords
Master of Resource Management: Coastal and Marine Management
Ísafjörður, September 2014

Supervisory Committee

Advisor:
- Micheal Honeth, Tobacco Caye Marine Station, Belize.

Reader:
- Zoi I. Konstantinou, MSc. - Ph.D.

Program Director:
- Dagný Arnarsdóttir, MSc.

Victor Buchet

Impact assessment of invasive flora species in Posidonia oceanica meadows on fish assemblage: an influence on local fisheries? The case study of Lipsi Island, Greece.

45 ECTS thesis submitted in partial fulfilment of a Master of Resource Management degree in Coastal and Marine Management at the University Centre of the Westfjords, Suðurgata 12, 400 Ísafjörður, Iceland

Degree accredited by the University of Akureyri, Faculty of Business and Science, Borgir, 600 Akureyri, Iceland

Printing: Háskólaprent, Reykjavík, September 2014

Declaration

I hereby confirm that I am the sole author of this thesis and it is a product of my own academic research.

Victor Buchet
Student‘s name

Abstract

Seagrasses are one of the most valuable coastal ecosystems with regards to biodiversity and ecological services, whose diminishing presence plays a significant role in the availability of resources for local communities and human well-being. At the same time, Invasive Alien Species (IAS) are considered as one of the biggest threats to marine worldwide biodiversity. In the Mediterranean, the issue of IAS is one which merits immediate attention; where habitat alteration caused by the human-mediated arrival of new species is a common concern. Indeed the Mediterranean Sea is considered to be one of the main hotspots of marine bio-invasions on earth. In this context, the present study examines the possible impacts of flora Invasive Alien Species on the fish assemblages associated with *Posidonia oceanica* seagrass habitat, and the possible impacts that any change might have on local fisheries. The setting for this study is Lipsi Island, in the Dodecanese, Greece. *In situ*, Underwater Visual Census's (UVC) were carried out at 14 sampling sites. Fish community parameters were estimated across three substrate types: dense *P. oceanica*, sparse *P. oceanica* and sparse invaded (by IAS flora) *P. oceanica*. External factors and percentage of flora cover were estimated for each substrate. Two flora IAS were found: *Halophila stipulacea*, one of the first species introduced in the region, which arrived after the Suez Canal opening (also known as a Lessepsian migrant), and *Caulerpa cylindracea*, a recent introduction through an unknown vector. The present study on the finfish assemblage around Lipsi Island supports the findings of similar studies undertaken in the Mediterranean. The results of the present study show that a low percentage of IAS does not have a significant impact on the finfish assemblage and thus does not seem to have had a significant impact on the local artisanal fishery. With little previous work in the region and no previous work on the Island, this study provides a baseline for future evaluation of changes produced by IAS and for potential management actions such as the creation of marine protected areas in the study region.

Keywords: *Posidonia oceanica, Greece, Management, fisheries, invasive species*

Acknowledgement

First I would like to thank Michael Honeth who accepted to help me finish my work in the middle of the summer and always helped me with kindness throughout the development of my thesis.

I would like to thank Richard Lilley and Dr. Richard Unsworth for having accepted advisory roles for this work.

I would like to thank Archipelagos NGO and its managers: Anastasia Miliou and Thodoris Tsimpidis who supported me to carry out this project.

I would like to thank the volunteers for their invaluable assistance in the November water of Lipsi Island: Tom Kerboul, Svenja Schönlau, Adam Coates, Yiannis Cetus and Kelli Maldre. I would like also to thank Rachel Seary and Kathryn Dawson for their help in designing my study, and finally Nizar Yaiche, for his daily help with the Lipsi team.

I would like to thank the University Centre of the Westfjord and Dagný Arnarsdóttir who aided in the processes of my research, especially with regards to their understanding of the multiple obstacles encountered.

I would also like to thank Dr. Marc Verlaque, Dr. Charles Francois Boudouresque, Michael Honeth, Albertína Friðbjörg Elíasdóttir and Brad Barr for their valuable knowledge and assistance.

And finally, I would like to thank my family for their support, the Dartevert family for daily hospitality, and Matija Drakulic, Charla Basran, Dinyar Minocher and Patrick Richard who kept me smiling and helped me to finalize and correct my work.

This project would not have been possible without their expertise and support.

Table of contents

Acknowledgement .... **v**

Table of contents .... **vi**

List of Figures .... **viii**

List of Tables .... **x**

List of Latin and Common names of species .... **xi**

1. Introduction **1**
- **1.1. Theoretical overview** **1**
- **1.2. Introduction in Seagrasses Meadows** **1**
  - 1.2.1. Ecological importance of the seagrasses 2
  - 1.2.2. Socio-economic importance of seagrasses 3
- **1.3. Introduction to Invasive Alien Species** **4**
  - 1.3.1. Introduction to Caulerpa cylindracea 4
  - 1.3.2. Introduction to *Halophila stipulacea* 7
- **1.4. Introduction to the Mediterranean Sea** **8**
  - 1.4.1. Introduction to the Dodecanese and Lipsi Island 10
- **1.5. Aims** **12**

2. Literature review **15**
- **2.1. The *Posidonia oceanica*** **15**
  - 2.1.1. Fish assemblage related to the *Posidonia oceanica* 17
  - 2.1.2. Threats to the *Posidonia oceanica* meadows 19
- **2.2. Invasive Alien Species** **20**
  - 2.2.1. Trends to Invasive Alien Species 20
  - 2.2.2. Impacts of the IAS on *Posidonia oceanica* and fish assemblage 21
    - 2.2.2.1. Impact of *Caulerpa cylindracea* 21
    - 2.2.2.2. Impact of *Halophila stipulacea* 23
- **2.3. Greek fisheries** **25**
  - 2.3.1. Introduction to artisanal fisheries 25
  - 2.3.2. Growth and development In Greece and Dodecanese 25
  - 2.3.3. On Lipsi 27
  - 2.3.4. Current fisheries management 28
- **2.4. Methods Literature review** **28**
  - 2.4.1. Percentage cover estimation 28
  - 2.4.2. UVCs method 29
  - 2.4.3. Data analysis tools 31
    - 2.4.3.1. Diversity and Evenness indices 31
    - 2.4.3.2. Analysis of Variance (ANOVA) 32

2.4.3.3. Cluster Analysis 33
2.4.3.4. Canonical correspondence analysis (CCA) 33

3. Methodology 35
3.1. The study Area 35
3.2. Data Collection 35
3.2.1. Record of the substrate and IAS characteristic 35
3.2.2. Underwater Visual Census (UVC) 38
3.2.3. Landings recording 39
3.2.4. Other data from the Survey 39
3.3. Data Analysis 40

4. Results 43
4.1. External factors and site conditions 43
4.2. Result of *Posidonia oceanica* and IAS percentage cover 45
4.2.1. Result of mean *Posidonia oceanica* percentage cover 45
4.2.2. Result of mean IAS percentage cover 46
4.3. Result of the fish assemblage UVC 48

5. Discussion 57
5.1. Finding on fish assemblages 57
5.2. Impact of *Posidonia oceanica* cover 58
5.3. Effect of Invasive Alien Species 58
5.5. Study limitation 60

6. Management recommendations 63
6.1. Proposed management guidelines 69
6.2. Future Studies 72

7. Conclusion 73

References 75

List of Figures

Figure 1. *Picture of a thallus of the invasive Caulerpa cylindracea from the Gulf of Marseille (-30 m). Herbarium specimen. Photograph: Klein J. (Klein & Verlaque, 2008)* 5

Figure 2. *Map of the Mediterranean distribution of Caulerpa cylindracea in 2006 (Galil, 2006a)*.. 6

Figure 3. *Picture of Halophila stipulacea collected in the Al-wahesh Lagoon, Libya. Photograph: Sghaier Y.R. (Sghaier et al., 2011)* 7

Figure 4. *Map of the Mediterranean distribution of Halophila stipulacea in 2006 (Galil, 2006b)* 8

Figure 5. *Map of the Mediterranean Sea showing the different seas, different straits and the 22 border countries of Mediterranean Sea (Graphicmaps.com, no date)* 9

Figure 6. *Map of Greece and Aegean sea, in red the Dodecanese and in the green circle Lispi island (Wikipedia, 2013)* 10

Figure 7. *Map of Lipsi Island with the locations of the 14 survey sites and Moschato Bay (Google Maps™)*.. 11

Figure 8. *Underwater picture of Posidonia oceanica seagrass. Mallorca, Balearic Islands, Spain; 6m depth on a sandy bottom. Photograph: Ordas E.I. (Guiry & Guiry, 2014)*....... 15

Figure 9. *Schematic representation of Posidonia meadow (matte, rhizomes and leaves) and its effect on sediment stabilization and reduction of hydrodynamism (Boudouresque & Meinesz, 1982)* 16

Figure 10. *Graphic of the vertical distribution of ichthyofauna in Posidonia oceanica meadows by day (A) and by night (B) (Harmelin-Vivien, 1982)*... 18

Figure 11. *Graphic of the number evolution of invasive marine species in Mediterranean Sea, Atlantic and Baltic sea over the 20 century (DAISIE data, presented in Hulme et al., 2010)* 21

Figure 12. *Map of grouping of Greek prefectures using as criteria the dependence on fisheries (D) and the number of fishermen (N) (Tzanatos et al., 2005).* 26

Figure 13. Example of the substrate estimation using 1m x 1m quadrat................................... 37

Figure 14. *Linear regression of the sites between the depth of the survey versus the percentage of Invasive Alien Species cover* 43

Figure 15. *Linear regression of the sites between the visibility of the survey versus the percentage of Invasive Alien Species cover* 44

Figure 16. *Map of Lipsi island. Red circles show sites invaded by IAS (n=10), green circle show intact site (n=4). Black stars (n=5) show the sites where dense P. oceanica has been surveyed (Google Maps™)* 45

Figure 17. *Graphic of the Posidonia oceanica % cover. In vertical axis the % cover form 0 to 100% and in horizontal axis each site studied groups by habitat types.* 47

Figure 18. *Graphic of the IAS % cover. In vertical axis the % cover form 0 to 35% and in horizontal axis each site studied invaded by IAS* 48

Figure 19. *Dendrogram showing the results of the cluster analysis of the abundance and species present among the habitats of each site surveyed (Euclidean distance). In green: dense P. oceanica sites. In blue: sparse P. oceanica sites. In red: sparse invaded (by IAS flora) P. oceanica sites - Lim.D – Limih Dense. Lim.IAS – Limih Sparse. Kim.D – Kimissi Dense. Kim.S – Kimissi Sparse. Kia – Kissiria. Mon – Mondendri. Cam.IAS – Campos Sparse. Pla.G – Platis Gailos. Ele.b.S – Elena beach Sparse. Ele.b.D – Elena beach Dense. Hoh – Hohlakoura. Kam.S – Kamares Sparse. Lal – Lalaouni. No.n.7 – No name 7. Kam.D – Kamares Dense. Tou – Tourcomnima. Mer – Mersini* 54

Figure 20. *A CCA plot of P.oceanica % cover, IAS % cover, depth variables and species present / abundance. C.chr – C.chromis. C.jul – C.julis. D.ann – D.annularis. D.sar – D.sargus. O.mel – O.melanura. S.lur – S.luridis. T.pav – T.pavo. S.cab – S.cabrilla. S.scr – S.scriba. S.cin – S.cinereus. S.tin – S.tinca. M.sur – M.surmuletus. A.boy – A.boyeri. G.buc – G.bucchichi. L.vir – L.viridi. D.vul – D.vulgaris. S.roi – S.roissali. S.ros – S.rostatus. S.sal – S.salpa*.. 56

Figure 21. *Graphic of the cooperative elements required to prevent present trend towards seagrass decline and efficiently converse seagrass ecosystems (Duarte, 2002)*................. 69

List of Tables

Table 1. *List of ranking of the 10 dominant species in terms of total biomass (up) and density (down) on P. oceanica meadows. Family, origin and ecological guild are indicated (Kalogirou et al., 2010)*........ 19

Table 2. *Table of the depth range and visibility of each site studied* 44

Table 3. *List of all the fish species and families recorded in the shallow sublittoral of the Lipsi Island during the present study.* 49

Table 4. *Table of the density (individual per m^2) calculated for each species in every sites studied* 51

Table 5. *Table of the result of the community indices (Abundance, Species richness, Diversity and Evenness) calculated for each habitat of the sites surveyed.* 52

List of Latin and Common names of species

The table lists all the Latin and Common names of the species cited in the present study. The common names are from the World Register of Marine Species (WoRMS) website.

Flora species		
Latin name	Family	Common name
Asparagopsis armata (Harvey, 1855)	Bonnemaisoniaceae	Harpoon weed
Caulerpa cylindracea (Sonder, 1845)	Caulerpaceae	Sea grapes
Caulerpa taxifolia (M.Vahl) C.Agardh, 1817	Caulerpaceae	Lukay-lukay
Codium vermilara (Olivi) Delle Chiaje, 1829	Codiaceae	
Cymodocea nodosa (Ucria) Ascherson, 1870	Cymodoceaceae	Little Neptune grass
Halophila stipulacea (Forsskål) Ascherson, 1775	Hydrocharitaceae	Broadleaf seagrass
Posidonia oceanica (Linnaeus) Delile, 1813	Posidoniaceae	Neptune Grass
Stypopodium schimperi (Kützing) M.Verlaque & Boudouresque, 1991	Dictyoataceae	
Zostera noltei (Hornemann, 1832)	Zosteraceae	Dwarf eelgrass
Fauna species		
Latin name	Family	Common name
Atherina boyeri (Risso, 1810)	Atherinidae	Big-scale sand smelt
Auxis rochei rochei (Risso, 1810)	Scombridae	Bullet tuna
Boops boops (Linnaeus, 1758)	Sparidae	Bogue
Bothus podas (Delaroche, 1809)	Bothidae	Wide-eyed flounder
Chromis chromis (Linnaeus, 1758)	Pomacentridae	Damselfish
Conger conger (Linnaeus, 1758)	Congridae	European conger
Coris julis (Linnaeus, 1758)	Labridae	Mediterranean rainbow wrasse
Dentex dentex (Linnaeus, 1758)	Sparidae	Common dentex
Diplodus annularis (Linnaeus, 1758)	Sparidae	Annular seabream
Diplodus sargus sargus (Linnaeus, 1758)	Sparidae	White seabream
Diplodus vulgaris (Geoffroy Saint-Hilaire, 1817)	Sparidae	Common two-banded seabream
Gobius bucchichi (Steindachner, 1870)	Gobiidae	Bucchich's goby
Gobius cobitis (Pallas, 1814)	Gobiidae	Giant goby
Gobius paganellus (Linnaeus, 1758)	Gobiidae	Rock goby
Hippocampus guttulatus (Cuvier, 1829)	Syngnathidae	Long-snouted seahorse
Hippocampus hippocampus (Linnaeus,	Syngnathidae	Short-snouted seahorse

1758)		
Labrus merula (Linnaeus, 1758)	Labridae	Brown wrasse
Labrus viridis (Linnaeus, 1758)	Labridae	Green wrasse
Lagocephalus sceleratus (Gmelin, 1789)	Tetraodontidae	Silver-cheeked toadfish
Lithognathus mormyrus (Linnaeus, 1758)	Labridae	Sand steenbras
Loligo vulagris (Lamarck, 1798)	Loliginidae	European squid
Mullus barbatus (Linnaeus, 1758)	Mullidae	Red mullet
Mullus surmuletus (Linnaeus, 1758)	Mullidae	Striped red mullet
Oblada melanura (Linnaeus, 1758)	Sparidae	Saddled seabream
Opeatogenys gracilis (Canestrini, 1864)	Gobiesocidae	
Pagellus acarne (Risso, 1827)	Sparidae	Axillary seabream
Pagellus erythrinus (Linnaeus, 1758)	Sparidae	Common Pandora
Pagrus pagrus (Linnaeus, 1758)	Sparidae	Red porgy
Parapenaeus longirostris (Lucas, 1847)	Penaeidae	Deep-water rose shrimp
Pinna nobilis (Linnaeus, 1758)	Pinnidae	noble pen shell / fan mussel
Sardina pilchardus (Walbaum, 1792)	Clupeidae	European pilchard
Sarpa salpa (Linnaeus, 1758)	Sparidae	Salema porgy
Scomber japonicus (Houttuyn, 1782)	Scombridae	Chud mackerel
Scorpaena porcus (Linnaeus, 1758)	Scorpaenidae	Black scorpionfish
Serranus cabrilla (Linnaeus, 1758)	Serranidae	Comber
Serranus hepatus (Linnaeus, 1758)	Serranidae	Brown comber
Serranus scriba (Linnaeus, 1758)	Serranidae	Painted comber
Siganus luridus (Rüppell, 1829)	Siganidae	Dusky spinefoot
Siganus rivulatus (Forsskål & Niebuhr, 1775)	Siganidae	Marbled spinefoot
Sparisoma cretense (Linnaeus, 1758)	Scaridae	Mediterranean parrotfish
Sphryaena chrysotaenia (Klunzinger, 1884)	Sphryaenidae	Yellowstripe barracuda
Sphryaena viridensis (Cuvier, 1829)	Sphryaenidae	Yellowmouth barracuda
Spicara maena (Linnaeus, 1758)	Centracanthidae	Blotched picarel
Spicara smaris (Linnaeus, 1758)	Centracanthidae	Picarel
Spondyliosoma cantharus (Linnaeus, 1758)	Sparidae	Black seabream
Symphodus cinereus (Bonnaterre, 1788)	Labridae	Grey wrasse
Symphodus mediterraneus (Linnaeus, 1758)	Labridae	Axillary wrasse
Symphodus melanocercus (Risso, 1810)	Labridae	Blackfin wrasse
Symphodus ocellatus (Linnaeus, 1758)	Labridae	Ocellated wrasse
Symphodus roissali (Risso, 1810)	Labridae	Five-spotted wrasse
Symphodus rostratus (Bloch, 1791)	Labridae	Pointed-snout wrasse
Symphodus tinca (Linnaeus, 1758)	Labridae	East Atlantic peacock wrasse
Syngnathus typhle typhle (Linnaeus, 1758)	Syngnathidae	Broadnosed pipefish
Thalassoma pavo (Linnaeus, 1758)	Labridae	Ornate wrasse
Xiphias gladius (Linnaeus, 1758)	Xiphiidae	Swordfish

"Adieu", dit le renard. "Voici mon secret. Il est très simple: on ne voit bien qu'avec le cœur. L'essentiel est invisible pour les yeux"

Antoine de Saint-Exupéry, Le Petit Prince

"Goodbye", said the fox. "And now here is my secret, a very simple secret: It is only with the heart that one can see rightly; what is essential is invisible to the eye"

Antoine de Saint-Exupéry, The Little Prince

1 Introduction

1.1 Theoretical overview

The Socio-Ecological study presented here examines the impact of flora Invasive Alien Species (IAS) on fish communities in *Posidonia oceanica* seagrass habitat, and possible impacts of any changes on the productivity of local artisanal fisheries. The study is defined as Socio-Ecological because of the intrinsic links between ecological systems and social processes. This scientific work can be seen as a baseline study, and a platform for future management and monitoring programmes. Within linked Socio-Ecological systems, the properties of each social system are influenced by the properties of the natural system on which the social system depends (Ash et al, 2010). Policy makers have started to highlight the key role within marine environments of human dependence on ecosystems for life support, well-being and socio-economic development (Cullen-Unsworth et al., 2013). However despite recent advances in our understanding of these Socio-Ecological processes, further ‘’place-based’’ research is required to investigate *“the connection between environmental issues and people”* (Ash et al, 2010: 10) and to understand the *“ecological and social characteristics of particular places and sectors”* (Potschin and Haines-Young, 2012: 3). This same set of connections operate at larger scale, so collection of data and place-based study at local level can offer the clearest example of the complex and dynamic interactions between ecology, society and economy. However, at the same time studies at larger geographic scales could also offer valuable insight.

1.2 Introduction in Seagrasses Meadows

Seagrasses are angiosperms, which are plants that are flowering, fruit-bearing, and where the ovules (and therefore seeds) of these plants develop within an enclosed ovary. Worldwide, these underwater flowering plants cover about 0.1–0.2 percent of the global ocean (Duarte, 2002), and despite the fact that they cover a tiny area of the ocean floor, they develop highly productive ecosystems which fulfil a key role in the coastal ecosystem and the entire ocean system (Duarte, 2002).

Seagrasses are marine plants found in shallow coastal areas. These plants often grow in large “meadow” or ‘’beds”. They grow on sheltered sandy or muddy substrates where they provide habitats, resources (food in direct and indirect ways) and shelter for many marine invertebrates and fish (Boudouresque et al., 2006, Diaz-Almela & Duarte 2008). They support a range of ecologically important marine species from all trophic levels (Orth et al. 2006) and are globally considered to be important for juvenile and larval stages of many commercial, recreational and subsistence fish and shellfish (Jackson et al., 2001, Bertelli & Unsworth, 2013). They appear to be one of the most valuable marine ecosystems in terms of goods and services (Costanza et al., 1997, Vassallo et al., 2013). They also have a multi-functional role in human well-being. Indeed, they are used for income generation, and as a source of food security through fisheries support; they support local tourism through the species that they host, for example with sea turtles observation and spearfishing spots and by indirect services, for example with the water quality due to the seagrass presence (Cullen-Unsworth et al., 2013).

This study focused on one species of seagrass, *Posidonia oceanica* a species endemic to the Mediterranean Sea. *Posidonia oceanica,* like other seagrasses, are under threat and are diminishing worldwide (Duarte, 2002, Boudouresque et al., 2006, Short et al., 2011). The threats are mainly anthropogenic; both directly and indirectly (Duarte, 2002), raising the issue as to how seagrass meadows can be more effectively protected.

1.2.1 Ecological importance of the seagrasses

Seagrasses are one of the richest and most valuable coastal ecosystems on the planet, supporting a range of keystone and ecologically important marine species (Costanza et al., 1997, Orth et al., 2006). They have high primary productivity and they provide organic carbon and nutrients to the entire oceans (Short et al. 2011). Seagrasses create a shelter for many marine species (Hemminga & Duarte, 2000). They are at the base of many marine food webs (Short et al., 2011) and they are an important food source for megaherbivores, like sea turtles, manatee and dugong (Orth et al., 2006).

Posidonia oceanica is seen as one of most complex and productive systems in the littoral zone of the Mediterranean Sea, with an ecological importance that is well documented (Guidetti et al., 1998, Díaz-Almela & Duarte 2008, Kalogirou et al., 2010, Short et al., 2011). *Posidonia oceanica* forms a complex habitat and supports high levels of biodiversity and numerous trophic interactions (Guidetti, 2000, Kalogirou, 2012, Guala et al., 2012). Furthermore it supports a rich fish community and invertebrate fauna and acts as a refuge and nursery to juvenile' fishes (Guidetti et al., 1998). In *P. oceanica* meadows are found a greater diversity and abundance of fish and larger numbers of juveniles than nearby bare/unvegetated substrata (Guidetti, 2000, Kalogirou, 2012). During daylight, *P. oceanica* meadows oxygenate the water with the leaf canopy increasing particle retention thereby increasing water transparency (Díaz-Almela & Duarte, 2008).

P. oceanica meadows are very good indicators of environmental quality as they mostly grow in relatively unpolluted waters. In addition, the rhizome concentrates radioactivity, synthetic chemicals and heavy metals, recording the environmental levels of such persistent contaminants on long-term scales (Díaz-Almela & Duarte, 2008). That is why the *P. oceanica* is often seen as an excellent bio-indicator for the Mediterranean Sea.

1.2.2 Socio-economic importance of seagrasses

In 2013, Vassallo et al., estimated the value of the services provided by seagrass ecosystems at 172€ m^2, which makes seagrass ecosystems as one of the most monetarily valuable ecosystem. Historically, the leaves of *P. oceanica* were used as packing material to transport fragile items. Furthermore, they were used to ship fresh fish from the harbour to inland cities. Respiratory infections and alleviation of skin diseases seemed to be prevented when sleeping in beds made of *P. oceanica,* parasites thrive less in *P. oceanica* leaves than in straw. So they were used as cattle bedding in stables and, later, as filling material for mattresses and cushions (Borum et al., 2004). The societal benefits derived from the *P. oceanica* have changed with time, but the biological attributes are just as important today. Geomorphologically, the leaves act as a filter, clearing the water of suspended sediments (Orth et al, 2006). By capturing the sediments, seagrasses stabilize them (Short et al., 2011) and by consequence, diminish the erosion of the coastline and assist in shore protection (Francour et al., 1999). Seagrass regulates the quality of coastal waters (Borum et al., 2004). It is used by many commercially important species, with ecosystem services including food supply for coastal populations (Unsworth & Cullen-Unsworth, 2011, Seitz et al., 2013), and indirectly produce a variety of goods, for instance finfish and shellfish (Borum et al., 2004). All these ecosystem services are directly used or beneficial to humans and economic development of coastal zones.

Seagrasses can be seen as a coupled social–ecological system (Cullen-Unsworth et al., 2013). Worldwide coastal communities rely on seagrasses and studies have shown the link between community decline and unsustainable methods of natural resource management (Adger, 2000). The social and ecological parts of the system need to be efficiently managed in order to ensure its sustainability.

> “*The threats to seagrass meadows are not only threatening an important resource, in many areas they are also threatening a way of life for those people closely associated with the system either directly or indirectly.*” (Cullen-Unsworth et al., 2013: 9)

1.3 Introduction to Invasive Alien Species

The Convention on Biological Diversity, (2009) defines Invasive Alien Species (IAS) as:

> “*Invasive alien species are plants, animals, pathogens and other organisms that are non-native to an ecosystem, and which may cause economic or environmental harm or adversely affect human health. In particular, they impact adversely upon biodiversity, including decline or elimination of native species - through competition, predation, or transmission of pathogens - and the disruption of local ecosystems and ecosystem functions.* ”

Biological invasions in marine habitats represent a worldwide threat to the dynamic of dependent coastal communities, through both the economy and social human well-being (Streftaris & Zenetos, 2006). They can accelerate the decline of native populations, and lead to population losses and extinctions on a local scale. IAS seems to be one of the biggest threats for the protection and preservation of worldwide biodiversity (Streftaris & Zenetos, 2006).

The situation in the Mediterranean merits attention; the alteration caused by the human-mediated arrival of new species is rapid. The Mediterranean Sea is considered to be one of the main hotspots of marine bio-invasions on earth (Rilov & Galil, 2009). In 2010, Zenetos et al. estimated the number of invasive or potentially invasive species in the Mediterranean Sea to 134 species (excluding microalgae). Moreover, the area the most impacted is the Eastern Mediterranean Sea (location of the present study) with 108 invasive or potentially invasive species in December 2010 (Zenetos et al., 2010). The present study focused on the two predominant flora IAS: the seagrass, *Halophila stipulacea* and the green alga *Caulerpa cylindracea*.

1.3.1 Introduction to Caulerpa cylindracea

Caulerpa cylindracea is from the Chlorophyta phylum of the order Bryopsidales belonging to the family Caulerpaceae. The genus *Caulerpa* includes approximately 85 species (Klein &

Verlaque, 2008). The previous name was *Caulerpa racemosa var. cylindracea*, but it has recently been changed to *Caulerpa cylindracea* (Belton et al., 2014).

In the Mediterranean Sea, three taxa of the same group (*racemosa* group) have been identified (Verlaque et al., 2000). The present study focused on the taxon found during the survey: *Caulerpa cylindracea* (Belton et al., 2014). Figure 1 below shows a picture of the species. Following the work of Verlaque et al., (2003) it has been shown that this species is endemic to South-West Australia. This green macroalgae with slender thallus has a uniaxial siphonous, mostly divided into a creeping axis (stolon) with rhizoids and erect shoots (Klein & Verlaque, 2008).

These fronds consist of leaf-like or grape or feather-like ramuli. *C. cylindracea* can typically grow up to 11cm (sometimes even to 19cm). As seen in Figure 1, the stolon is attached to the substrate by thin short rhizoids (Galil, 2006a, Klein & Verlaque, 2008).

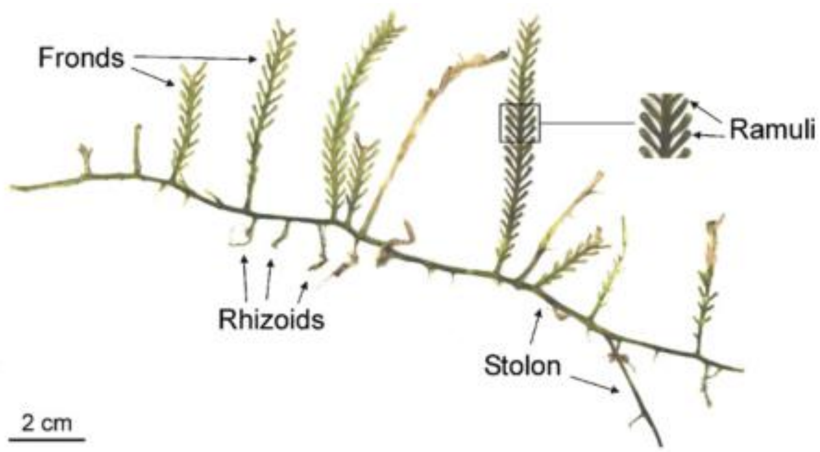


Figure 1. Picture of a thallus of the invasive Caulerpa cylindracea from the Gulf of Marseille (-30 m). Herbarium specimen. Photograph: Klein J. (Klein & Verlaque, 2008)

The first record of *Caulerpa cylindracea* in the Mediterranean Sea was in Libya in 1990 (Verlaque et al., 2003). The way of introduction of *C. cylindracea* into the Mediterranean Sea remains unclear. Ship traffic (ballast water, ship hull fouling) and aquaria are considered as different possibilities. Then, this green alga had nearly completely invaded the Mediterranean Sea (fig. 2), report of record in Italy – 1993, Greece - 1994, Cyprus – 1997, France – 1998, Turkey, Malta, Spain – 1999, Croatia – 2003, Algeria – 2007 (Klein & Verlaque, 2008). It was even recorded around the Canary Islands in the Atlantic Ocean (Verlaque et al., 2004).

Furthermore, in 2014 the latest records have shown that the coastlines of Algeria and Morocco have been invaded (Verlaque, personal communication, March 2014).

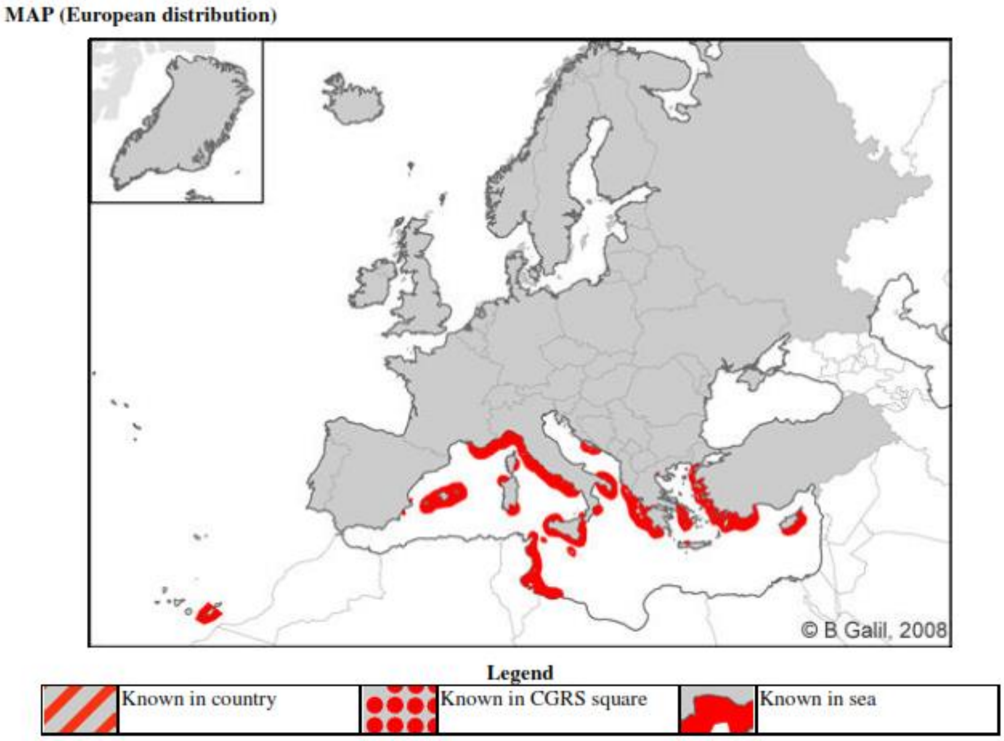


Figure 2. Map of the Mediterranean distribution of Caulerpa cylindracea in 2006 (Galil, 2006a)

Named as a ‘Blitzkrieg’ by Verlaque et al., (2004), the speed of the spread through the Mediterranean Sea of *C. cylindracea.* can be explained by these dispersal mechanisms. The zygotes, fragments or propagules are transported by shipping (ballast water, anchor gear), fishing gear (dredging, trawling, bottom nets and traps), and currents may play a major role in the dispersion of the species as well (Klein & Verlaque, 2008). Detached fragments can survive several days without affecting the capability of re-establishment (Piazzi et al., 2005).

Moreover, the high rates of elongation of the *C. cylindracea* stolons allow a rapid colonization of the substratum. The availability of numerous means of reproduction, multiplication and dispersion, give *C. cylindracea* the possibility to extend the colonized area rapidly. The size and growth dynamic vary with season, as well as external factors (temperature, weather, turbidity) and the location (Piazzi et al., 2005).

In its natural habitat, this green alga is a common species that grows from the intertidal zone down to only 6 meters depth on reef flats and intertidal pools. In the Mediterranean Sea, it thrives on all kinds of soft and hard substrata such as in tide pools, on pebbles, rock, dead 'matte', sand, mud, detrital and coralligenous assemblages. It is found in polluted and unpolluted areas. The depth range is from 0 to 70 meters, with highest abundance between 0 and 30 m (Klein & Verlaque, 2008). In 2010, Katsanevakis et al. found that the highest frond densities were observed on the dead matte and rocky habitat types, indicating their high vulnerability to colonization.

1.3.2 Introduction to Halophila stipulacea

Halophila stipulacea is from the Magnoliophyta phylum of the order Hydrocharitales, belonging to the family Hydrocharitaceae. This euryhaline seagrass is a Lessepsian migrant. These rhizomes are creeping, branched and fleshy, and the roots appear solitary at each node of the rhizome, unbranched and thick with dense soft root hairs (fig. 3). Pairs of leaves are distributed on petioles along a rhizome, rooted in the sand (Galil, 2006b). The rhizome is 0.5-2 millimetres wide and leaf blades are 3-6 centimetres long (Guiry & Guiry, 2014).

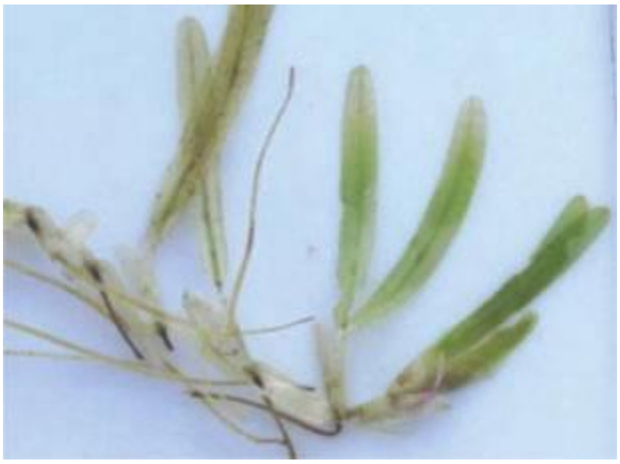


Figure 3. Picture of Halophila stipulacea collected in the Al-wahesh Lagoon, Libya. Photograph: Sghaier Y.R. (Sghaier et al., 2011)

In the Mediterranean Sea, it is found on sandy and muddy bottoms, the depth range is usually between 1 to 45 meters, but it has been found as deep at 65 meters (Galil, 2006b). *Halophila stipulacea* is native to the western Indian Ocean and is known as one of the first Lessepsian migrants (Sghaier et al., 2011). Indeed, the first record was reported in 1894 in Rhodos Island, Greece (Galil, 2006b). It invaded the Eastern Mediterranean Sea throughout the 20th century, report of record in Cyprus – 1895, Southern Aegean Sea – 1923, Egypt – 1941, Crete – 1955, Lebanon – 1961. However, recent records, Malta – 1970, Sicily – 1990, Tunisia – 2003, of the invasive seagrass (Galil, 2006b) show that it has recently started being recorded further West (fig. 4). The dispersal mechanisms of the *H. stipulacea* are the currents, ship transport (ballast water, anchor gear) and fishing gear (Galil, 2006b).

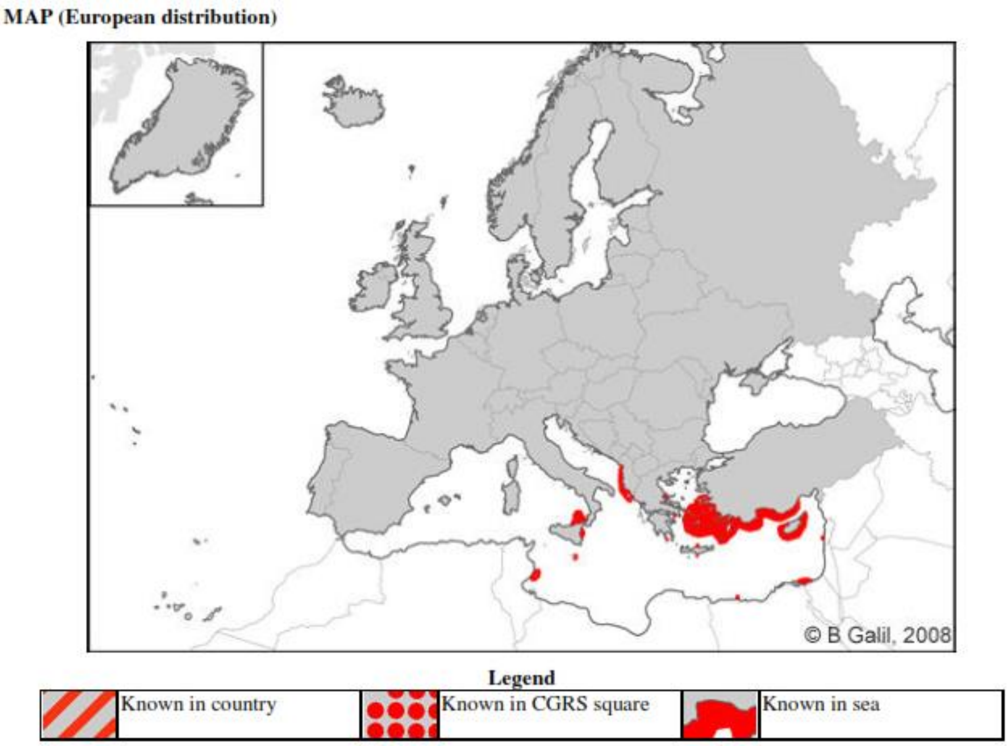


Figure 4. Map of the Mediterranean distribution of Halophila stipulacea in 2006 (Galil, 2006b)

1.4 Introduction to the Mediterranean Sea

The Mediterranean Sea is a sea between Europe and Africa. It covers a surface area of about 2.5 million km² (Sea Around Us, 2013). It is connected to the Atlantic Ocean by the Gibraltar Strait, to the Red Sea by the Suez Canal and to the Black Sea by the Dardanelles and the Bosporus. Twenty-two countries have a coastline on the Mediterranean Sea (fig. 5). In 2010, the population around the Sea was estimated at 466 million inhabitants and the population is predicted to reach 529 million by 2025 (UNEP/MAP, 2012)

Figure 5. Map of the Mediterranean Sea showing the different seas, different straits and the 22 border countries of Mediterranean Sea (Graphicmaps.com, no date)

The Mediterranean is defined by NOAA as a Large Marine Ecosystem (LME): a *"scientifically, technically and legally tractable regime for the conservation and management of living resources"* (Sherman et al. 1990: 3). This LME is one of the most diverse and stable LMEs in terms of fish species groupings and their share in the total catch (Garibaldi & Limongelli, 2003). However, the percentage of over-exploited fisheries had grown from 0 percent in 1958 to 25 percent in 2005 (Sea Around Us, 2013). The high anthropogenic pressure of the Mediterranean coastline affects the whole sea and especially the most productive habitats such as estuarine and coastal shallow waters (Sea Around Us, 2013). In terms of unsustainable coastal development projects and construction, it has been established by Seitz, et al. (2013) that 86 percent of the European coast is at a high or moderate risk. These pressures jeopardize the sustainability of Mediterranean Sea health, at the global scale and at the local scale, like on Lipsi Island.

Even if actions at a local level give good results, management decisions and programmes would be the most effective if they are implemented by all Mediterranean countries. That is why; the European Union fosters global management cooperation with different projects (European Commission, 2013). European projects attempt towards global cooperation in order to protect the Mediterranean resources. Two projects can be cited:

The project on Integrated Maritime Policy for the MEDiterranean (IMP-MED) seeks to provide opportunities to the European Neighbourhood Policy countries of the Mediterranean for engaging in and obtaining assistance for maritime-policy development and cooperation.

The MAritime REgions cooperation for the MEDiterranean (MAREMED) project, falling under the European Territorial Cooperation Programme Mediterranean for 2007-2013 seeks to encourage the integrated maritime management and the sustainable development of coastal zones for different levels of coastal governance.

Throughout history, the Mediterranean Sea has always been a resource provider for the numerous adjacent coastal communities. This protection and management is the keystone action in order to preserve the sensitive ecosystem and sustain the human well-being around this Sea.

1.4.1 Introduction to the Dodecanese and Lipsi Island

The Dodecanese is a group of 12 larger, and 150 smaller Greek islands in the Aegean Sea, of which 26 are inhabited (fig. 6). They are situated at the southeast of the Aegean Sea. They are politically under Greek administration, but as seen on the map, they are near the Turkish coastline. In 2005, the population was estimated at 200,452 inhabitants (Hellenic Statistical Authority, 2012).

Figure 6. Map of Greece and Aegean sea, in red the Dodecanese and in the green circle Lispi island (Wikipedia, 2013)

Historically, the region was under the domination of the Ottoman Empire until the end of the First World War and was ruled by Italy until the end of the Second World War. Following the war, the islands became a British military protectorate, and then despite objections from Turkey, they were formally united with Greece in 1947.

Lipsi Island is a part of the Dodecanese; the population was estimated at 790 in 2011 (Hellenic Statistical Authority, 2012). It has a surface area of 17.35 km² (fig. 7). Based on personal observation, the economy seems to be mainly based on tourism, construction, fisheries and local agricultural production; furthermore most of the inhabitants make their living from mixed occupations depending on the season (Archipelagos, unpublished report). Lispi Island hosts a dynamic coastal community of fishermen, who suffer from a global fish stock diminution over the last years (Milliou, personal communication, December 2013). Aquaculture had been introduced in secluded and protected bays like Moschato Bay (on the east of the Island), but has been removed. Currently the water in these bays is often very cloudy and the substrate is mostly dead matte of *P. oceanica*. In addition the presence of IAS has been detected around the Island (Milliou, personal communication, September 2013).

Figure 7. Map of Lipsi Island with the locations of the 14 survey sites and Moschato Bay (Google Maps™)

1.5 Aims

> "*Knowledge of ecological traits of invaders and assessment of damage are fundamental for predicting the consequences of invasions and finding effective methods to control and mitigate this form of pollution*" (Klein & Verlaque 2008: 10)

Assessments of ecosystem structures and functions are vital for scientific and decision makers to better understand them and implement efficient actions in order to ensure their ecological durability and their sustainable socio-economic development. Through quantitative analysis of seagrass meadow composition, and associated fish species assemblages, this project aims to increase the body of knowledge on fish assemblage structure and function of *Posidonia oceanica* in a coastal area of Lispi Island invaded by IAS. It also explores the potential impact of introduced flora species to the local ecology and its possible impact on local fisheries. Finally it gives management recommendations.

The over-arching aim of this research is to answer the following question:

How does a low percentage of flora Invasive Alien Species cover in the Posidonia oceanica meadows impact fish assemblages around Lipsi Island?

In order to address to this question, two research questions were answered:

1. What difference, if any, is there in fish assemblages and fish community indices between the habitats of Lipsi water (dense/sparse intact *Posidonia oceanica,* and *Posidonia oceanica* invaded by invasive flora species)?

2. How does the variation of density cover of IAS and *Posidonia oceanica* impact the fish assemblages in Lipsi waters?

Regarding these research questions, the hypotheses were:

1. Species richness and density of the fish assemblages are lower in the *Posidonia oceanica* invaded than in the *Posidonia oceanica* intact areas.

2. The higher the IAS density cover, the stronger the impact on the species richness and density of the fish assemblages.

2 Literature review

2.1 The *Posidonia oceanica*

Posidonia oceanica (fig. 8) is an endemic seagrass of the Mediterranean Sea. It is present in around 25 percent of the Mediterranean basin above 40 meters depth (Klein & Verlaque 2008), which represented around 2.5 - 4.5 million hectares in 1998 (Pasqualini et al., 1998). It forms dense meadows that grow on rocks and sandy bottoms. .

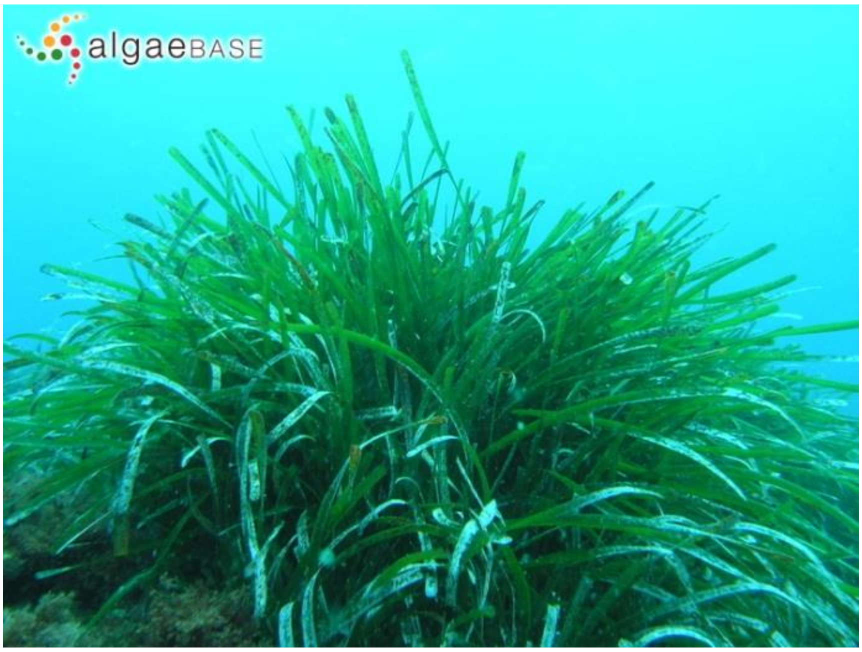


Figure 8. Underwater picture of Posidonia oceanica seagrass. Mallorca, Balearic Islands, Spain; 6m depth on a sandy bottom. Photograph: Ordas E.I. (Guiry & Guiry, 2014)

Posidonia oceanica is regarded as a key habitat within the Mediterranean Sea (Klein & Verlaque, 2008). On the 21st of May, 1992, *P. oceanica* meadows were defined as priority natural habitats recorded by Annex I of the EC Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora (Díaz-Almela & Duarte, 2008). Furthermore *P. oceanica* meadows are included in the reference list of priority habitats of the SPA/BIO Protocol of Barcelona Convention (Guala et al., 2012).

Posidonia oceanica is a large slow-growing seagrass. It forms a dense leaf canopy supported by a thick root-rhizome, called matte. In the meadows, leaves can attain 1 meter in height during the summer but appears shorter and sparser during winter and autumn (10 to 40 cm high). Meadow density is maximal in shallow water, where it may attain more than 1,000 shoots per m² and decreases exponentially with depth (Hemminga and Duarte, 2000). In very clear water, *P. oceanica* can be found at depths greater than 50 meters (Borum et al., 2004).

Enhanced sedimentation, combined with vertical rhizome growth, produces characteristic reefs called matte (fig. 9). The matte is a network of dead rhizomes with shell/organic debris and sediments, which accumulate over centuries to attain several meters in height (Hemminga and Duarte 2000).

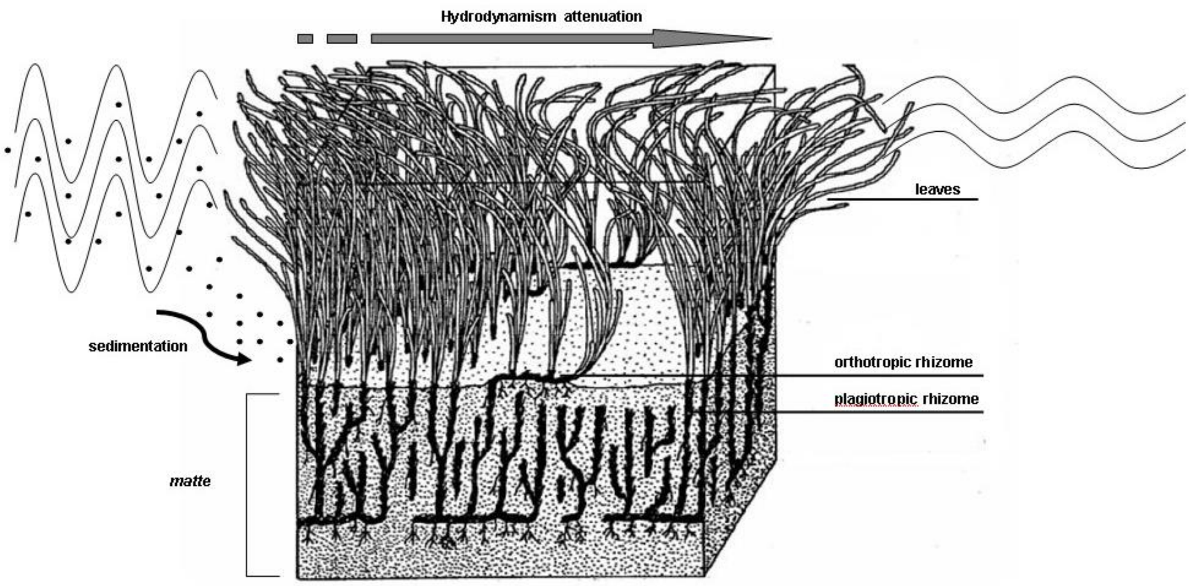


Figure 9. Schematic representation of Posidonia meadow (matte, rhizomes and leaves) and its effect on sediment stabilization and reduction of hydrodynamism (Boudouresque & Meinesz, 1982)

Death of *P. oceanica s*hoots leads to exposure of the underlying matte. It is called dead matte, and can remain for many years. Usually, dead matte is seen as a degraded habitat (Borg, 2006).

Posidonia oceanica has a vital function in the Mediterranean ecosystem. Most of the studies show a global diminution of the cover areas (Duarte, 2002, Jorda et al., 2012). However, Bonacorsi et al. (2013) showed a relatively stability in the areas cover by *P. oceanica*. This study was done in Corsica over more than 50 years. It suggests that the diminution of this endemic seagrass of Mediterranean Sea is not a global and certain phenomenon and local case studies can show variation.

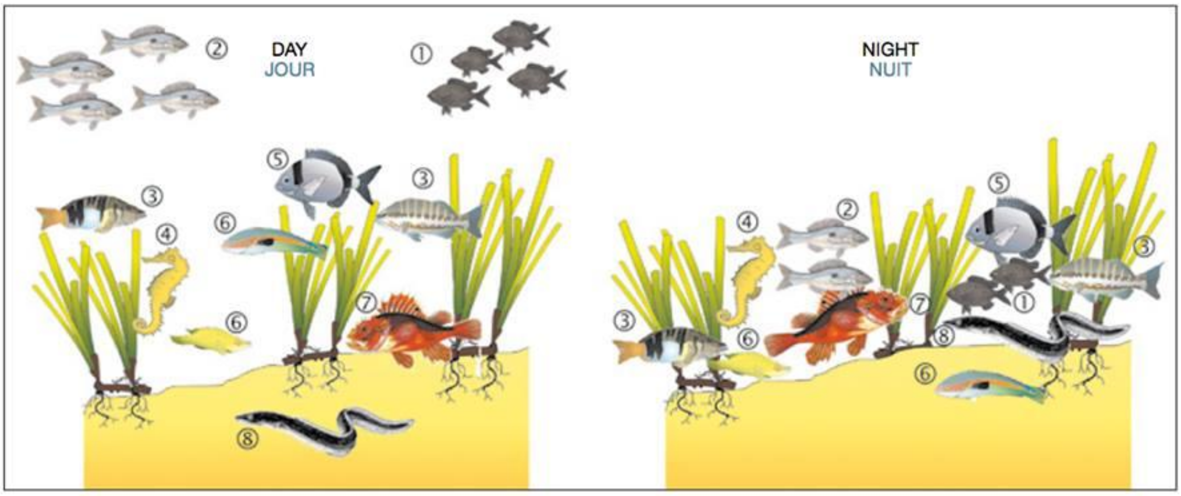
2.1.1 Fish assemblage related to the *Posidonia oceanica*

Many studies have been done on the fish assemblage associated with *Posidonia oceanica*, across a number of regions of the Mediterranean. Spatial and temporal variations in fish community structures are influenced by biological (predation, competition, larval dynamics, recruitment variability) and environmental (light and nutrient availability, habitat structure, substrate complexity, depth, current) factors (Giakoumi & Kokkoris, 2013). Moreover anthropogenic disturbances (e.g. fisheries) also affect the fish community structure (Pauly et al., 2005)

Posidonia oceanica is the habitat for a rich fish community and provides food and shelter to juvenile fishes of commercial interest (Guidetti et al., 1998). Moreover it is a habitat that provides complexity needed for a rich biodiversity (Gratwicke & Speight, 2004, Giakoumi & Kokkoris, 2013). Figure 10 shows that the vertical distribution of the fish community of *P. oceanica* meadows changes between day and the night (Harmelin-Vivien, 1982).

Posidonia oceanica is the habitat for a lot of different species, some of the main species found are the Gobius sp. (living on rhizomes), as well as Labrus merula, Labrus viridis, Symphodus sp., Diplodus sp., Sarpa salpa, Coris julis and Chromis chromis (Diaz-Almeta & Duarte, 2008). There are also some obligate species living within the leaf canopy, like the cryptic species Opeatogenys gracilis and Syngnathus typhle typhle (Díaz-Almela & Duarte, 2008). The endangered species Hippocampus hippocampus can be found within the canopy (Díaz-Almela & Duarte, 2008). A lot of small encrusting algae use the surface of the leave for attachment (Short et al., 2011).

In their study in the Cyclades Archipelago, Giakoumi & Kokkoris, (2013), found *Coris julis* and *Sarpa salpa* as the most abundant species in rocky habitats with patches of *P. oceanica* and sand. The study was done by underwater survey along a 75m X 5m transect line, placed approximately parallel to the shore at 3 meters depth.



Vertical distribution of ichthyofauna (teleosts) in *Posidonia oceanica* meadows by day (A) and by night (B) (from Harmelin-Vivien, 1982 simplified).

Distribution verticale de l'ichtyofaune (téléostéens) dans les herbiers à *Posidonia oceanica* le jour (A) et la nuit (B) (d'après Harmelin-Vivien, 1982 simplifié).

1 – Pomacentridae *(Chromis chromis)*,
2 – Centracanthidae *(Spicara maena)*,
3 – Serranidae *(Serranus scriba, Serranus cabrilla)*,
4 – Syngnathidae *(Hippocampus guttulatus)*,
5 – Sparidae *(Diplodus vulgaris)*,
6 – Labridae *(Coris julis, Symphodus ocellatus)*,
7 – Scorpaenidae *(Scorpaena porcus)*,
8 – Congridae *(Conger conger)*.

Figure 10. Graphic of the vertical distribution of ichthyofauna in Posidonia oceanica meadows by day (A) and by night (B) (Harmelin-Vivien, 1982)

Around the coasts of Rhodes Island, Kalogirou et al., (2010), found in the *P. oceanica* meadows 88 species within 34 families (Table 1). This study was conducted with boat seining at a depth range from 5 to 35 meters around the coasts of Rhodes Island.

Outside of the Aegean Sea, in the Adriatic Sea, a study using diver visual census (20m) transect line 2 meters wide, depth not recorded) found 24 different species in the *P.oceanica* meadows. *Symphodus ocellatus, Diplodus annularis* and *Spondyliosoma cantharus* were the most common species in the *P. oceanica* seagrass meadows (Guidetti, 2000). In the Ligurian Sea, Guidetti et al., (1998) found 28 species within 9 families. The results show that the beds were numerically dominated by planktivorous species (*Chromis chromis, Spicara smaris, Spicara maena* and *Boops boops*) and Labridae and Sparidae were the most species-rich families. These results have been recorded with transect lines placed from about 6 to 28-30 meters depth. Even if the exact results change for each study, certainly due to the different methods used and the temporal and spatial change among studies, it is clear that the *P. oceanica* has important diverse fish community associated with it.

Table 1. List of ranking of the 10 dominant species in terms of total biomass (up) and density (down) on P. oceanica meadows. Family, origin and ecological guild are indicated (Kalogirou et al., 2010)

Rank	Species	Family	Origin	Ecological guild	Biomass (kg)
1	*Boops boops**	Sparidae	I	SR	527·69
2	*Spicara smaris**	Centracanthidae	I	SR	231·75
3	*Chromis chromis*	Pomacentridae	I	SM	148·48
4	*Spicara maena**	Centracanthidae	I	SR	80·62
5	*Oblada melanura**	Sparidae	I	JM	33·41
6	*Sphyraena viridensis**	Sphyraenidae	I	SR	32·79
7	*Lagocephalus sceleratus*	Tetraodontidae	NI	SR	27·27
8	*Sparisoma cretense**	Scaridae	I	JM	22·08
9	*Coris julis*	Labridae	I	SR	16·78
10	*Pagrus pagrus**	Sparidae	I	JM	12·17
	Species	**Family**	**Origin**	**Ecological guild**	**Density**
1	*Spicara smaris**	Centracanthidae	I	SR	40056
2	*Boops boops**	Sparidae	I	SR	31202
3	*Chromis chromis**	Pomacentridae	I	SM	16646
4	*Sardina pilchardus**	Clupeidae	I	JM	2845
5	*Spicara maena**	Centracanthidae	I	SR	2148
6	*Coris julis*	Labridae	I	SR	2103
7	*Siganus rivulatus**	Siganidae	NI	JM	2055
8	*Sphyraena chrysotaenia**	Sphyraenidae	NI	JM	1325
9	*Mullus surmuletus**	Mullidae	I	JM	1312
10	*Sparisoma cretense**	Scaridae	I	JM	1077

2.1.2 Threats to the *Posidonia oceanica* meadows

Even if Bonacorsi et al., (2013) showed stability in the seagrass of Corsica, a recent meta-analysis of studies showed that seagrass habitat disappeared worldwide at a rate of 110km² per year between 1980 and 2006 (Waycott et al., 2009). In recent decades, the total area under the *Posidonia oceanica* meadows in the Mediterranean Sea has decreased by more than 30 percent (Terrados & Borum, 2004). Indeed like the other seagrass species *P. oceanica* total cover is decreasing. The decline in seagrass communities can be caused by both natural processes (geological, meteorological, biological) and anthropogenic activities.

The most common human activity responsible for the decline of seagrass habitats, according to Francour et al., (1999), is eutrophication as a result of nutrient loading and a subsequent reduction in water quality and increased turbidity. Indeed *P. oceanica* meadows are very sensitive to water and sediment enrichment with organic matter and nutrient (Díaz-Almela & Duarte, 2008). Aquaculture has been shown to produce major environmental impacts, especially due to shading, eutrophication and sediment deterioration through excess organic inputs. Seagrass meadows as far as 100 meters from fish cages can be impacted (Terrados & Borum, 2004).

Coastal urbanization leads to a fragmentation and reduction of the habitats available for the seagrass meadows (Terrados & Borum, 2004). Moreover, coastal development may increase seagrass reduction. The disruption of the sedimentation / erosion balance causes by coastal or inland construction might alter the pattern of coastal current and lead to erosion or siltation of the seabed (Díaz-Almela & Duarte, 2008). Dredging and reclamation of coastal ecosystems either for extraction of sediments or as part of coastal engineering or construction, can damage seagrass meadows as well (Terrados & Borum, 2004).

Propeller damage, boat-trawling and boat anchoring have been proven to negatively affect seagrass communities (Francour et al., 1999, Díaz-Almela & Duarte, 2008). Boat anchoring leaves scars in *Posidonia oceanica* meadows, as do boat moorings (Terrados & Borum, 2004). These scars create favourable microhabitats for the settlement and expansion of *C. cylindracea* (Katsanevakis et al., 2010). Fishing gear, particularity bottom trawling on the deep meadows impact *P. oceanica* and the repeated use of such trawl gear have dramatically reduced plant density and cover (Díaz-Almela & Duarte, 2008). Moreover, fishing gear contributes to the spread of *C. cylindracea*. Its fragments can be collected by fishing gear and subsequently discarded by fishermen together with other by-catch and debris (Katsanevakis et al., 2010).

There are also indirect impacts that result in the reduction of the seagrass meadows and could increase the spread of IAS: global warming, sea level rise, CO^2 increased, UV penetration, and anthropogenic impacts on marine biodiversity change the natural conditions of the *P. oceanica* (Terrados & Borum, 2004). *Posidonia oceanica* is a fragile habitat and therefore damages in its meadows increase the potential of IAS implantation into it. Also, the potential implications of degraded coastal regions extends further than lost habitat, as it would decrease fishery landings based on the fleeting presence of species dependent on *P. oceanica* meadows (Seitz et al., 2013). Management measures need to be taken in order to diminish each of these threats.

2.2 Invasive Alien Species

2.2.1 Trends to Invasive Alien Species

Invasive alien species are plants, animals, pathogens and other organisms that are non-native to an ecosystem. They may cause economic or environmental harm or adversely affect human health (Convention on Biological Diversity, 2009). The IAS may cause different problems. They are considered one of the biggest threats to marine biodiversity (Streftaris & Zenetos, 2006) and are recognized as one of the five pressures directly driving biodiversity loss. The other four are habitat change, overexploitation, pollution and climate change (Shine et al., 2010).

Worldwide globalization has opened new trade routes, increased trade between continents and, expanded tourism. These factors among others have increased the opportunities for potential IAS to expand. As it is possible to see on the Figure 11, at the European level, over the period 1970-2007 the total area of IAS grew by 76 percent (Hulme et al., 2010). The Mediterranean Sea is the most affected European Sea (Shine et al., 2010). In 2008, the monetary cost of IAS in Europe was estimated close to €10 billion annually (Hulme et al., 2010).

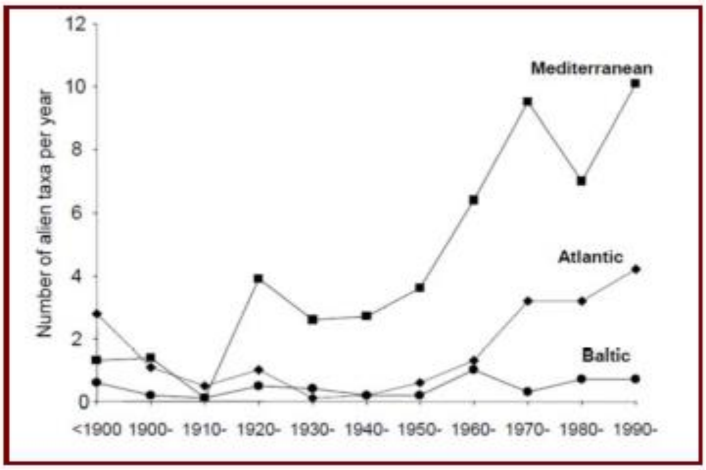


Figure 11. Graphic of the number evolution of invasive marine species in Mediterranean Sea, Atlantic and Baltic sea over the 20 century (DAISIE data, presented in Hulme et al., 2010)

Six flora species have been recorded as IAS (Invasive Alien Species) in the Eastern part of the Aegean Sea: *Halophila stipulacea, Caulerpa taxifolia, Codium vermilara, Stypopodium schimperi, Caulerpa cylindracea* and *Asparagopsis armata* (Archipelagos, unpublished data, 2012). Two of these invasive flora alien species were found during the present study.

2.2.2 Impacts of the IAS on *Posidonia oceanica* and fish assemblage

2.2.2.1 Impact of Caulerpa cylindracea

In 2006, *C. cylindracea* was listed as one of the 100 'Worst Invasives' Alien Marine Species in the Mediterranean (Streftaris & Zenetos 2006). There are two different impacts, the one directly on the *P. oceanica* and the one on the other species.

The effect of *C. cylindracea* on the vegetative cycle and phenolic compounds of *Posidonia oceanica* have been assessed. Leaf length and leaf area index was found reduced in the presence of *C. cylindracea*. At the same time an increase in primary foliar production and in the number of leaves produced annually was observed, leading to a higher turnover rate (Dumay et al., 2002).

In 2000, Ceccherelli et al., found that the susceptibility of the seagrass community to invasion of *C. cylindracea* is related to the availability of sand habitat ground. The health of seagrass meadows assessed by the shoot density influences the invasive success of *C. cylindracea.* It has been found that low shoot density of *P. oceanica*, correlated to a relatively high *C. cylindracea* growth rate. At the same time a higher shoot density seems to reduce the green alga growth (Ceccherelli et al., 2000). In fragmented *P. oceanica* meadows, the sandy and dead matte patches represent the spots of high vulnerability to invasion (Katsanevakis et al., 2010). Moreover the dense patches of *P. oceanica* seem to never be penetrated by the *C. cylindracea* while it is often seen creeping in on the rhizomes at the margins or in sparse mats (Deudero et al., 2011). Also in the presence of the *C. cylindracea, P. oceanica* has a lower shoot density, which can be indicative of a more degraded seagrass ecosystem (Shepperson et al., 2013).

Caulerpa cylindracea does not only impact *P. oceanica*, In 2007, Raniello et al., have shown that its cohabitation with *Cymodocea nodosa*, triggered alterations in photosynthesis of the seagrass and decrease in shoot density. However, the shoot density increased for the *Zostera noltei* in contact with the *C. cylindracea* (Klein & Verlaque, 2008). When the green alga has invaded all the available substrata, it constitutes a multi-layered structure up to 10 – 15 centimetres wide. This structure traps sediment and leads to the creation of an anoxic layer that kills sessile benthic organisms (De Biasi et al., 1999). The cover and the species number of macroalgal communities invaded by *C. cylindracea* decline and the structure of the assemblages changes (Piazzi et al., 2001). This leads to a homogenization of the flora assemblage.

Due to these impacts on habitat architecture and sediment trapping, *C. cylindracea* is seen to be a habitat modifier and it has been classified as a new ecosystem engineer (Klein & Verlaque, 2008).

> *"An ecosystem engineer is an organism that modifies, creates or destroys habitat and directly or indirectly modulates the availability of resources to other species, causing physical state changes in biotic or abiotic materials."*
> (Jones, 1994: 3).

Posidonia oceanica and *C. cylindracea* seem to be impacted by their coexistence. *Caulerpa cylindracea* affected the characteristic of *P. oceanica* and flora associated with it, and consequently has the potential for large cascading effects on associated fauna biota.

Firstly, the rapid spread of the *C. cylindracea* may be due to the fact that it is not of interest to many consumers (Ruitton et al., 2006). Few herbivorous fish species seem to feed on the *C. cylindracea*. *Boops boops, Pagellus acarne, Sarpa salpa* and the Lessepsian species *Siganus luridus* have been recorded grazing on *C. cylindracea* (Azzurro et al., 2004, Ruitton et al., 2006).

Few studies have been done on the impact on *C. cylindracea* on fauna. It is noteworthy, that Ulas et al., (2011) states that, this genus does not seem to affect fish species and diversity. However, the presence of *C. cylindracea* changes the composition of the phytobenthos, which brings a modification of the macrobenthos: a proliferation of polychaetes, bivalves and echinoderms and a reduction in the numbers of gastropods and crustaceans (Galil, 2006a). Moreover, studies have shown an increase in densities, diversity and evenness of meiofauna in assemblages invaded by *C. cylindracea*. The results showed an increase of crustaceans and annelids (Carriglio et al., 2003, Piazzi et al., 2005, Galil, 2006a). The green alga has been recorded to invaded various kinds of macrobenthic animals such as sponges, gorgonian corals and sea anemones (Zuljevic et al., 2004), and based on personal observation on living *Pinna nobilis.*

So far, only one study has been done on the effect of *C. cylindracea* on landings (Ulas et al., 2011) but no study has been done on the impact on human activity and the economy (Piazzi et al., 2005, Klein & Verlaque, 2008). Only one impact is known on the fisheries by the obstruction of fishing nets by the uprooted alga. However through trophic cascade, the known impact of *C. cylindracea* on the benthic community and on the grazing species could cause a decrease of predators, within some of the most valuable targeted commercial species (Piazzi et al., 2005).

2.2.2.2 Impact of Halophila stipulacea

Even if the *Halophila stipulacea* is considered as one of the 100 worst IAS in the Mediterranean Sea, there is much less literature about this seagrass than about *C. cylindracea* (Streftaris & Zenetos, 2006). One of the explanations may be that invasive behaviour has not been observed in the southern Mediterranean marine areas. Almost all the recorded presence of

H. stipulacea in the southern Mediterranean is limited to the areas near or within harbours or marinas (Sghaier et al., 2011). However, the Lessepsian migrant competes with the native Mediterranean seagrass and cohabits with it. It was recorded on the eastern coast of Cyrenaica (Libya) within meadows dominated by the *P. oceanica* (UNEP / MAP-RAC / SPA, 2009).

Halophila stipulacea seems to be an opportunistic species. With the coastal environment routinely subjected to more and more disturbance, the tropical seagrass seem to colonize the available space created by the disturbances (boat anchoring for instance) and out-competes the slow-growing *P. oceanica* (Procaccini et al., 1999). The comparison between the associated algal assemblages of an invaded meadow and two pristine meadows dominated by *Posidonia oceanica* and *Cymodocea nodosa* revealed significant differences in floral species composition (Galil, 2006b).

During a study off Vulcano Island (Eolian Islands, western Mediterranean), thirty-six species of macroalgae were found associated with the species. In comparison with other Mediterranean seagrasses, *H. stipulacea* has a poor epiphytic flora. The virtual absence of encrusting corallines was noteworthy. It can be explained by the fast turnover rate of the leaves (Rindi et al., 1999). Like *C. cylindracea*, these impacts on flora have the potential for cascading effects on associated fauna biota.

One study has been done on the fish community associated with *H. stipulacea* in the Mediterranean Sea (Di Martino et al., 2007). The study was done by Underwater Visual Census. Seasonal variation of the fish community was observed, but the fish community does not seem to be affected by shoot density variation. A total of 30 fish species from eight families have been recorded. The most abundant species were *Sarpa salpa*, *Coris julis*, *Oblada melanura, Mullus surmuletus* and *Thalassoma pavo*. Regarding the family, three Labrids and Sparids, two Serranids and one Mullid were detected with high frequency of occurrence, > 75 percent (Di Martino et al., 2007).

Like *P. oceanica*, the Lessepsian migrant seems to be used by the fish as a nursery area. Indeed a high number of small-sized specimens of *Scorpaena porcus, Serranus hepatus, Diplodus vulgaris* and *Oblada melanura*, have been observed. However, the limited occurrence of planktivorous species may differ in the other Mediterranean sea-beds (Di Martino et al., 2007).

Even if *Halophila stipulacea* is considered an invasive species, and it impacts the Mediterranean ecosystem, it seems to play the seagrass-equivalent roles of a nursery. However, its morphology being substantially different than *P. oceanica*, it might impact the fish community in other indirect ways.

2.3 Greek fisheries

2.3.1 Introduction to artisanal fisheries

The small-scale fisheries, also called the artisanal fishery or traditional fisheries are, defined by fishing households (as opposed to commercial companies), using relatively small amount of capital and energy, relatively small fishing vessels (if any), making short fishing trips, close to shore and mainly for local consumption. However this definition varies between countries (FAO, 2014).

Small-scale or traditional fisheries are a complex system identified by a great spatio-temporal variation, diversity of gear and target species, scattering of fishing activity on the coastal zone and direct supply of the catch to market (Tzanatos et al., 2005). As a result, it is very hard to efficiently manage the small-scale fisheries. Indeed its complexity makes it difficult to predict the allocation of fishing effort among alternative target species in mixed fisheries (Salas et al., 2004).

Compared to the industrial fisheries, artisanal fisheries are seemed as more selective, using less destructive fishing gear, taking less by-catch, and using less fuel (Fabio & Hazin, 2005). Small-scale fisheries provide more job opportunity and therefore the benefits and fish stocks are shared with more people. Higher dependency due to much lower mobility leads to a more responsible/respectful uses and due to the fishing gear limitation, they do not impact sensible marine ecosystem (deep sea, for instance). Finally they are often part of the cultural heritage of local community and environmental popular knowledge (Fabio & Hazin, 2005).

2.3.2 Growth and development In Greece and Dodecanese

Historically, and through to the present, small-scale fisheries have an enormous socio-economic importance to the Greek coast. During the 1964-1989 period, it contributed 87.5 percent to the mean number of boats, 63.7 percent to the mean number of fishers and 47.4 percent to the mean wholesale value of catch (Stergiou et al., 1996). The social importance of the artisanal fishery varies between the Greek prefectures, but it is worth mentioning, as seen in Figure 12, that it is relatively higher in the insular ones (Tzanatos et al., 2005). The official data recognize that 46.8 percent of the total fisheries production is from this way of fishing (Tzanatos et al., 2005).

In 2002, the official data of the Greek small-scale fishing fleet totalled 19,052 vessels. The average total length was 6.8 meters with more than 70 percent of the vessels ranging between 5 to 10 meters (Tzanatos et al., 2005).

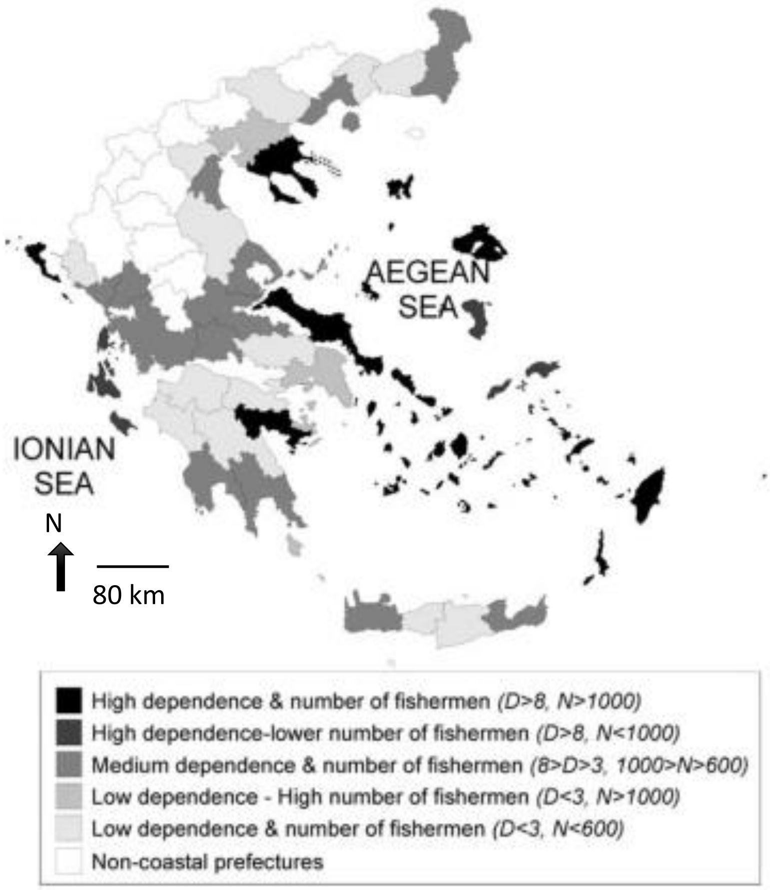


Figure 12. Map of grouping of Greek prefectures using as criteria the dependence on fisheries (D) and the number of fishermen (N) (Tzanatos et al., 2005).

Tzanatos et al., (2005) define the typical Greek small scale fisherman as:

> *"[person who] goes fishing for 209.2 days a year, following a seasonal pattern. He typically uses nets and longlines. In most cases, he alters seasonally the usage of fishing gear and the targeting of species to a lesser or higher extent to attain higher yields."*

On the Greece scale, the most important target species, in 2002, they were *Mullus surmuletus, M. barbatus*, *Pagrus pagrus*, *Pagellus erythrinus* and *Diplodus* sp. In the Dodecanese, the most targeted species throughout the year are: *Boops boops, Mullus surmuletus, Scorpaena sp.*, *Pagellus erythrinus* and *Diplodus sp.* All year except summer the species targeted are: *Pagrus pagrus, Dentex dentex,* and *Spondyliosoma cantharus*. Finally during the summer and autumn *Sparisoma cretense* is targeted (Tzanatos et al., 2005).

2.3.3 On Lipsi

As already stated, Lipsi Island is part of the Dodecanese. While there are no sources documenting the relationship between local fisheries and the Lipsi community, it appears that the fishery seems to be highly important economically and socially on the island. Daily life is based on and around the harbour. For example most of the restaurants on Lipsi buy their fish daily directly from the local fishermen (Vassilis, personal communication, November 2013). The fishery provides a full income for only few fishermen, but it provides an extra-income and an important leisure activity for a lot of inhabitants. Indeed, there are 25 fishing licenses but only 11 regular 'full time' fishermen (<6 meters boats), known to mostly use Trammel, Gill and Longline gear, they fish largely on seagrass and rocky-algal habitats. These boats use static gear (Lilley, personal communication, March 2014).

Regardless of the gear use, *Posidonia oceanica* was always the most targeted habitat for fishing effort (Archipelago, unpublished report). According to Savva et al. (2013), the most targeted species of Lipsi's artisanal fisheries as a percentage of total catch (kg) were *Sparisoma cretense* (29.03%), *Loligo vulgaris* (14.9%), *Scorpaena* sp. (9.39%), *Pagellus erythrinus* (9.01%), *Octopus* sp. (5.19%), *Scomber japonicus* (4.14%), *Auxis rochei rochei* (3.43%), *Serranus cabrilla* (3.315%), *Mullus surmuletus* (2.88%). Compared to the Mediterranean catch database (Sea Around Us, 2013), there is similarity between the fish targeted on both scales (*Mullus surmuletus* and *Scomber sp.* for instance). Most of the main species targeted at the Mediterranean scale are also targeted on Lispi Island. However there are also some differences. They might be explained by the fact that Lipsi fishermen are mostly artisanal fishermen, who practice fishing close to the shore. They do not target species far away from the coastline which require heavy fishing gear, for example: *Xiphias gladius* and *Parapenaeus longirostris*. Fishermen from Lipsi Island are highly dependent on the fish in the *Posidonia oceanica* meadows and the species found in shallow waters. This is at the same depth of where *C. cylindracea* and *H. stipulacea* are found. So if the IAS affect the fish assemblage in the *P. oceanica* meadows, it could equally affect the local fisheries.

The fishing pressure in the waters surrounding Lipsi is not only from professional and recreational Lipsi fishermen. Fishermen from other islands sometimes come to fish in Lipsi waters. Small-scale fishermen from close by islands visit regularly; three are based on Arki and around 40 on Patmos. Also there are two purse seine boats operating out of Leros, and one out of Patmos. Trawlers are also active in the area. Based on Lilley (personal communication, May 2014), the last listing records one based on Patmos, two on Leros, two on Samos, and four on Kalymnos. All this fishing effort generates big pressure on the fishery resources as fishermen can stay at sea for days and fish in Lipsi waters (Lilley, personal communication, May 2014). Finally spearfishing is popular on the island in summer and autumn. Spear fishermen mainly target big fish like Grouper, which are threatened (Lilley, personal communication, May 2014). Moreover, there are no regulations concerning spearfishing activity around the Island (personal observation).

2.3.4 Current fisheries management

Fisheries management strongly depends on information that can be obtained via fish landings. However, this kind of information is usually absent in Greece (Tsikliras et al., 2007). The explanations can be the difficulty to record catch data from small scale fishermen disseminated on the 18,000km coastline of Greece and a mistrust for official authorities (Savva et al., 2013). Currently Lipsi fisheries are governed independently from Patmos and Arki. However fishing pressure is much more interlinked geographically between Lipsi and Arki than Arki/Lipsi and Patmos (Lilley, personal communication, May 2014). It seems that there is a need for change in the governance of fisheries management in order to be more efficient (Milliou, personal communication, December 2014).

2.4 Methods Literature review

2.4.1 Percentage cover estimation

Seagrass beds are one of the most vulnerable coastal habitats worldwide, and policy makers are under pressure to protect these valuable habitats (Kirkman, 1990). There is a need to map and monitor the meadows of seagrass over a range of spatial and temporal scales (McKenzie et al. 2003). The first seagrass monitoring programmes started at the beginning of the 1980's in Australia, USA and France (Borum et al., 2004). Since seagrass monitoring programmes have become more popular over the last decades. As UVC, some approach uses are a cost-effective way to collect data and involve stakeholders in management. There are a lot of different protocols for monitoring seagrass using a range of approaches from in situ observation to remote sensing (McKenzie et al., 2003). The choice of technique and protocols are scale and site dependent, and may include a range of approaches. The structure and goals of monitoring programmes are diverse, as some of them are conducted exclusively by scientists or technical personnel, and others, such as Seagrass-Watch, rely on volunteers (Borum et al., 2004). With volunteer-based protocol, the limitations and biases are mostly form the surveyors, and can be reduced by their training.

Two of most used seagrass monitoring protocol are seagrass-watch (McKenzie et al., 2003) and SeagrassNet (Short et al., 2006). Both use quadrats in order to assess the seagrass cover characteristics. In the SeagrassNet protocol, 0.50 m^2 quadrat is used to estimate percentage cover. For each site studied, the protocol demands 3 transects of 50 meters each, where 12 quadrats are randomly measured along the each transect (Short et al., 2006). Seagrass-watch protocol uses the same technique of the quadrat (McKenzie et al., 2003).

However, these two protocols differ in techniques, canopy height, shoot density, and biomass measurements, and laboratory procedure for SeagrassNet, Photography and multiplication of the different data collection techniques for seagrass-watch.

2.4.2 UVCs method

The Underwater Visual Census is to date, the most commonly used technique for studying littoral fish communities, due to its minimal environmental impact (Harmelin-Vivien et al. 1985). The visual census is defined as survey that records all the species encounter or only some species in a limited area.

Underwater Visual Census (UVC) seemed to be one of the best non-destructive methods to assess fish communities in shallow water. It seems relatively accurate and cost effective (Halford & Thompson, 1994), but it always records only a percentage of the total fish assemblage (Harmelin-Vivien & Francour, 1992).

It is a recent technique popularized in the 80’s mostly by Harmelin-Vivien et al., (1985). Before that fish censuses were mostly performed using small trawl nets or by poisoning with rotenone. However the increase in marine protected areas and parks, where destructive methods could not be implemented, and with a need for long term monitoring, led to new ways to study fish communities (Harmelin-Vivien & Francour, 1992).

The biases and errors during an UVC can be influenced by three factors, the surveyor, the surveyed (the fishes) and external factors as weather, current and site location (Harmelin-

Vivien et al., 1985). Surveyors are a source of error, by their presence in the studied environment, and by miss-recording (Harmelin-Vivien et al., 1985). Furthermore an UVC tries to record an instantaneous estimate of abundance for species present within the area of the transect. Unfortunately this theoretical goal can never be perfectly realised due to two factors: the time taken to count and record each individual species and the inability to scan the entire transect area at any one time (Halford & Thompson, 1994).

The behaviour of the fish is also a source of error. It leads to an over-estimation of the fish attracted by the surveyor (c.f *Atherina boyeri*, *Boops boops, Oblada melanura, Diplodus* sp.) and an underestimation of the cryptic species, for instance *Syngnathidae* sp. (Harmelin-Vivien et al., 1985). The only way to decrease the bias of this factor is in training. Even if the fish will always be attracted to the surveyor, the training can minimize mistakes.

However, even if this method is becoming more and more popular, there are always questions about its effectiveness. Indeed, the volunteer-based monitoring approaches have recently received great attention as a cost-effective way to collect data and involve stakeholders in management (Leopold et al., 2009). These works give data for policy makers; in order to be able to have a clearer idea of the consequences of any environmental policies they might adopt (Schmeller et al., 2009). However, the analyses of the results of volunteer-based work may be less precise. Fish community estimates from UVC are weakened by numerous biases that ultimately reduce the diagnostic power of the data (Bernard et al., 2011).

Precision is a function of the number of monitored sites and the number of sites is maximized by volunteer involvement (Schmeller et al., 2009). Under Visual Census needs people involvement and it is time-consuming, which can be problematic in remote areas. Volunteer-based UVC needs to be calibrated as well. Leopold et al., (2009), compared local volunteer-based results and scientific results in three underwater reefs in Fiji. Their conclusion was that a calibration of community-based monitoring was needed in order to ensure appropriate management action. Also, many UVC protocols are a mix between different techniques, UVC transect, stationary point and fyke net survey (McKenzie et al., 2003). The aim is to increase the robustness of the results.

Stereo-DOV (Jind, 2012), large/vulnerable-fish monitoring, socio-economic monitoring or water-quality monitoring (Wilkinson et al., 2003) can in some cases be a better option to monitor marine protected areas. However, with limited human resources and time available for the present study, UVC with transect seemed to be the best option for this study. It allowed the team to perform replicate substrate cover estimations within the same area of the fish data collection, while maintaining a space big enough to record fast-swimming fishes (Harmelin-Vivien et al., 1985). It also allowed getting the best estimation of commercial and non-commercial fish communities, contrary to estimations made from landings or from large/vulnerable-fish data collections, and to assess the potential consequences of IAS on each fish species.

To conclude, UVC is a good non-destructive method to monitor shallow water and provide data for policy makers. It is an attractive method for volunteers. It is used by a lot of NGOs, Blue Venture in Belize and Madagascar, and Archipelagos in Greece, for example. However the biases are numerous, it is time-consuming, and it requires involvement of volunteers (local or not) and scientists.

2.4.3 Data analysis tools

2.4.3.1 Diversity and Evenness indices

In order to compare the fish community among the sites, different community indices were used in the present study. A diversity index is a measurement of the heterogeneity of species, there are in a dataset. It also looks at how evenly the individuals are distributed among the dataset. The value of a diversity index is directly linked to the species richness and the evenness. There are numerous diversity indices, the present study used one of the most known among ecological studies: The Shannon-Wiener Index.

Shannon-Wiener Index is denoted by $\mathbf{H = -SUM\ [(p_{\mathit{i}}) \times ln(p_{\mathit{i}})]}$

SUM = summation
p_i = proportion of total sample represented by species i

Species evenness is a measurement of biodiversity referring to how close the numbers of each species is in a known environment. The present study used Pielou's evenness index.

Pielou's index is denoted by $\mathbf{J = \mathit{H'} / ln(\mathit{S})}$

H' = Shannon Weiner diversity
S = the total number of species in a sample, across all samples in dataset

2.4.3.2 Analysis of Variance (ANOVA)

$$F = \frac{MST}{MSE} = \frac{\frac{SST}{p-1}}{\frac{SSE}{N-p}} = \frac{\frac{\sum n(x-\bar{x})^2}{p-1}}{\frac{\sum (n-1)S^2}{N-p}}$$

Where:

F = ANOVA Coefficient, MST = Mean sum of squares due to treatment. MSE = Mean sum of squares due to error.

SST = Sum of squares due to treatment, SSE = Sum of squares due to error, p = Total number of populations, n = Total number of samples in a population, S = Standard deviation of the samples, N = Total number of observations

The analysis of variance, ANOVA, is a statistical method in which the variation in a set of observations is divided into distinct components. In the present study, only one-way analysis of variance (one-way ANOVA) was used. This technique can be used only for numerical data (Howell & David, 2002). It compares the means of two or more groups to determine if at least one group mean is different from the others. The F-ratio is used to determine statistical significance (NCSS Statistical Software, 2012). The results of a one-way ANOVA can be considered reliable as long as the following assumptions are met:

- Observations were randomly and independently chosen from the populations.
- Population distributions are normal for each group.
- Population variances are equal for all groups.

There are a few limitations to using one-way ANOVA. If no significant difference is found in the data-set, it does not mean that the samples are the same. ANOVA only indicates a difference between groups, not which group(s) are different (Gaten, 2000). Also the power of this test is influenced by the sample size. If the sample size is small, (less than 25 per group), the power of the normality test can be questionable (NCSS Statistical Software, 2012).

2.4.3.3 Cluster Analysis

Cluster analysis groups objects (observations, events) based on the information found in the data describing the objects or their relationships. It groups a set of objects in such a way that objects in the same group (called a cluster) are more similar to each other than to those in other groups (Kaufman & Rousseeuw, 1990). It is often used as an exploratory data mining technique. A cluster analysis has two objectives; first to discover types among the data-set, and then reduce the number of cases by enabling consideration of several types instead of numerous records (Sclove, 2001). In order to generate a reliable analysis, some assumptions need to be met before carrying out a cluster analysis:

- The data need to be pre-processed by outlier detection and standardization.
- Data need to give a reliant representativeness of the sample.

It is worthy to mention that there are no rules-of-thumb about the sample size necessary for cluster analysis (Dolcinar, 2002). However, cluster analysis has several limitations; it imposes hierarchical structure on data, whether it is real or not. It does not depict data with multiple, independent underlying controls well and because these are based on algorithms rather than formal mathematics, solutions can be non-unique (Olszewski, 2007).

2.4.3.4 Canonical correspondence analysis (CCA)

Canonical correspondence analysis (CCA) is a multivariate method to elucidate the relationships between biological assemblages of species and their environment (ter Braak & Verdonschot, 1995). So it contains two sets of variables, the biological assemblage and the environmental characteristic. CCA can help to disentangle how a multitude of species simultaneously respond to external factors, such as environmental variables. It is often used to identify environmental gradients in ecological data-sets in particular to determine which environmental variables are important in the determination of the community composition (ter Braak & Verdonschot, 1995). For cluster analysis, data-sets for the CCA need to meet some assumptions in order to be robust:

- The distributions of the variables in the population are normal.
- The relations among the variables are linear.
- The sample size needs to include 40 to 60 times as many cases as variables.
- Data are independent (no redundancy).

Regardless the assumption, one of main limitation is the fact that there is always some inherent variability or "noise" in vegetation and biological data, even for plots identical environmental conditions. This noise could arise from errors in data collection, stochastic variation in the location of individuals within a stand, or site-specific variation in history (Gauch, 1982). Finally as a multivariate analysis, some information is lost in the process and the results are mostly based on their interpretation (Baccini & Gonzalez, 2006).

3 Methodology

3.1 The study Area

The study took place on Lipsi Island (Λειψοί, 37°18′N 26°45′E). This island is part of the Greek administrative region of the South Aegean, which is the part of the regional unit of Kalymnos (fig. 6). This study was done with the support of a Greek NGO named ARCHIPELAGOS. This NGO works on marine and land conservation in Greece and in the whole Mediterranean Sea. With the help of this NGO, an Underwater Visual Census and Seagrass cover estimation were carried out, that took place in the coastal zone of Lispi. The survey took place in November 2013 (4 weeks), (n=14) sites were studied (fig. 7). Archipelagos'' volunteers assisted in all these tasks.

Based on time limitation (a month), equipment limitation (no car), coast accessibility (the west of the island is mountainous and the island has few roads), team issues (3 days the team was not able to go survey, due to sickness) and weather limitation (9 days of thunder storm), the maximum of site were studied. In order to generate a clear picture of the Lipsi Island fish assemblage, sites were selected all around the island. Following Giakoumi & Kokkoris, (2013) and La Mesa et al., (2011) methodologies, the different sites were spaced at least by 500 meters (it was five kilometres for Giakoumi & Kokkoris, (2013) and one kilometres for La Mesa et al., (2011)). Finally some sites were not survey because of a danger due to the boat traffic (around the harbour), due to submarine electricity cable proximity (South-East of the island) and because of a poor visibility underwater (Moschato bay and biological sewage exit).

3.2 Data Collection

3.2.1 Record of the substrate and IAS characteristic

Limited by time (a month) and equipment (only transects and one quadrat), a simple and easy seagrass estimation protocol was designed based on other protocols (Archipelagos protocol and SeagrassNet protocol, Guala et al., 2012) and with the help of Dr Marc Verlaque (Verlaque, personal communication, October & November, 2013), from the Mediterranean Institute of Oceanology of Aix Marseille Université. To estimate the percentage cover characteristics of each transect substrate, a quadrat 1m x 1m was used and 25 replicates per transect were surveyed. So 25 percent of the total surface of each transect was assessed in order to have a statistically reliable estimation of the substrate cover characteristic.

The quadrat was divided into four squares of 50cm x 50cm. In each 50cm x 50cm square, the percentage cover of *Posidonia oceanica*, dead *Posidonia oceanica* meadows, rock and sand were assessed by visual observation accurate to 25 percent. (0% / 25% / 50% / 75% / 100%). The IAS percentage cover was assessed in the same way, except a > 5 percent class: (0% / >5% / 5-25% / 25-50% / 50-75% / 75-100%). The average of the four 50cm x 50cm squares provided the information for the quadrat. The depth was recorded every 5 meters along each transect (6 measurements per 25 meters transect line).

The figure 13 below shows an example of the substrate assessment. The 1m x 1m quadrat has been divided in 50cm x 50cm square (1, 2, 3, 4). In the present example, the assessment would be:

For the *Posidonia oceanica*: Square 1 = 25%, S2 = 75%, S3 = 50%, S4 = 100%

25 + 75 + 50 + 100 = 250 %. 250 / 4 = 62.5 %

The estimation result of the *Posidonia oceanica* for this quadrat would be to 62.5 percent.

For the Caulerpa cylindracea: S1 = > 5%, S2 = 5 – 25%, S3 = 5 – 25%, S4 = 0%

2.5 + 15 + 15 + 0 = 32.5 32.5 / 4 = 8.1 %

The estimation result of the Caulerpa cylindracea for this quadrat would be to 8.1 percent.

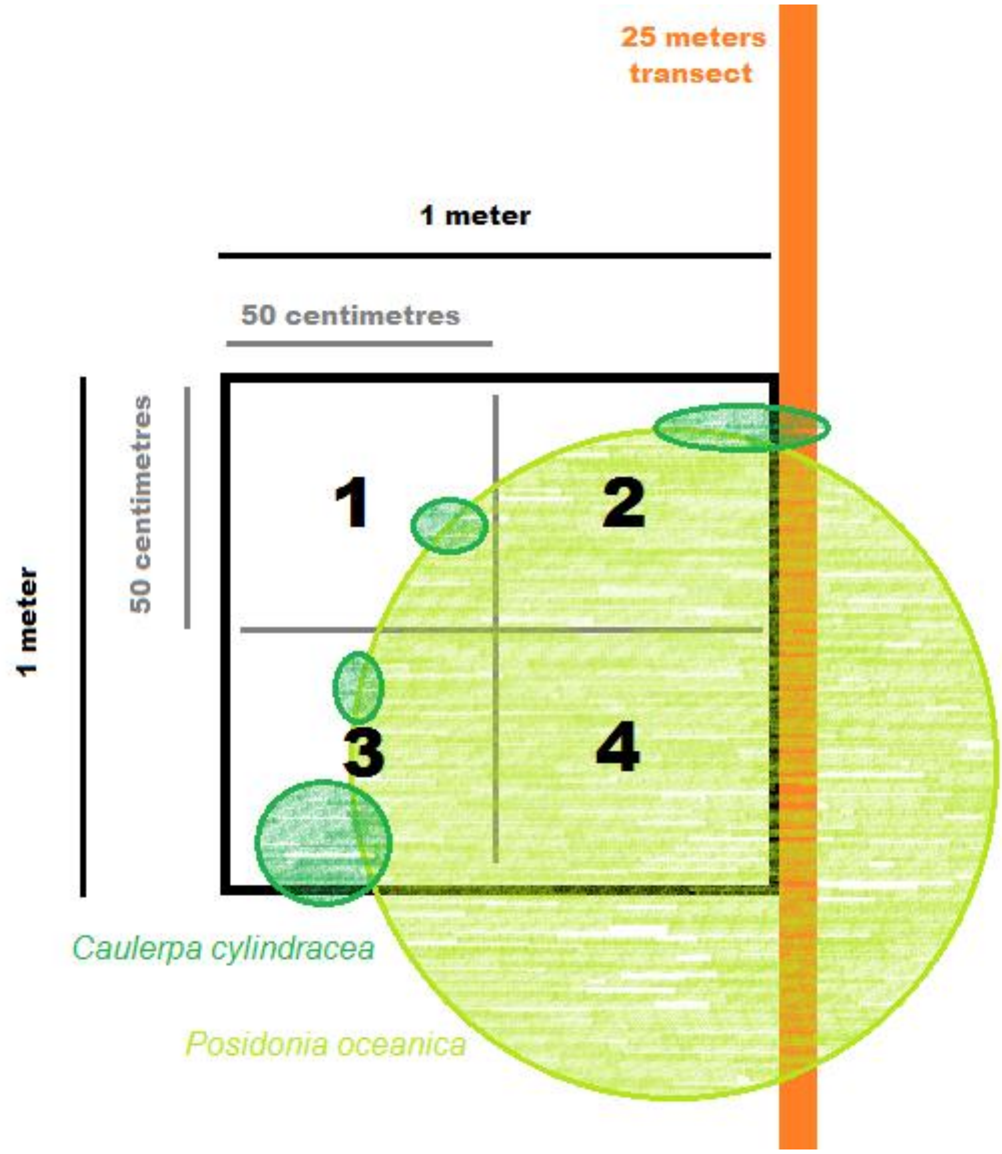


Figure 13. Example of the substrate estimation using 1m x 1m quadrat

Based on personal observations and Giakoumi & Kokkoris, (2013) methodology, a very basic habitat differentiation was used to assess the possible impact of IAS. Spatial and habitat-related patterns were examined, if it was possible, in three habitats:

- Dense *Posidonia oceanica* (n=5).
- Sparse *Posidonia oceanica* (n=4).
- Sparse *Posidonia oceanica* affected by IAS (n=10).

Posidonia oceanica is known to form dense homogeneous patches, but at the same time can also forms very sparse meadows on sand and sometimes rock. From this observation, 3 different habitat types were identified. Dense *P. oceanica* meadows were defined has homogenous meadows contrary to the heterogeneous substrate cover of the sparse *P. oceanica*. The transect on dense *P. oceanica* meadows were only place when the team was sure of the homogeneity of the meadows and the *P. oceanica* percentage cover would be above 90 percent. If it was not possible to find dense *P. oceanica* habitat, only Sparse *P. oceanica* habitat was surveyed always in the depth range 2 to 5 meters. If IAS was found during the data collection of Sparse *P. oceanica* habitat, the habitat was seen as Sparse *Posidonia oceanica* affected by IAS.

3.2.2 Underwater Visual Census (UVC)

Based on others UVC protocols (Archipelago protocol, Harmelin-Vivien et al., 1985, Guidetti, 2000, Giakoumi & Kokkoris, 2013), it was created as a simple specific protocol in order to fit into time and equipment limits. In this study, the visual census was done underwater; one surveyor was at the surface of the water and recorded all fish species seen in the limited area. This limited area was defined by a strip-transect 25 meters long and 4 meters wide ($100m^2$ total surface area per transect). To diminish the biases, the fish and algae surveyors were always the same person in order to avoid inconsistencies between the surveys, and the team trained before the data collection, and had to pass an UVC test. Moreover, a few things were done regarding the presence impact: not making any sudden moves, swimming gently, and use of a no-purge snorkel.

Following the methodology of La Mesa et al., (2011), UVCs were carried out between 13:00 and 16:00 and with good sea-weather conditions. Based on the standardized procedures of Harmelin-Vivien et al., (1985), on each site, for each habitat type, three transects were done to have enough replicates in order to have a reliant estimation of the fish assemblage. Each transect was done between 2 to 5 meters depth. This depth range was selected, in order to not be too close to the fish and disturb them too much, and at the same time be close enough to be able to identify the fish species. This technique was selected among the available visual methods as the most appropriate for smaller specimens and fast swimming species (Harmelin Vivien & Francour, 1992).

Individuals were recorded to assess the species richness. Abundance of fishes was recorded by counting single specimens, and schools of larger than 10 individuals were recorded within one of six abundance classes (11–30, 31–50, 51–100, 101–200, 201–500, >500 individuals). The midpoint of each abundance class gave estimates of fish density (Harmelin-Vivien et al. 1985).

The snorkel surveys were carried out in water of around 2 to 5 meters depth, whereas the fishery landings are from depths averaging around 30 meters (Savva et al., 2013). As the present study was restricted by the availability of diving equipment, it was unable to survey the true habitats of the specific fishing locations. Nonetheless, the fact remains that the impact of the AIS in the shallow water ecosystems is of great importance to the fishery since they support the beginning of life-history pattern of a lot of fish as they provide a nursery for juveniles, *Pagrus pagrus* for example. (Zeller & Pauly, 2001).

3.2.3 Landings recording

One of main question of the present study was to assess the possible impact of the IAS on the local fisheries. To answer this question, the present study used the result of an Archipelagos project, which did focused on the recording of the landing of Lipsi local fishermen. It lasted from late July to end of November 2013. Every day, Lipsi local fishermen were interviewed about the type of their fishing gear, size of nets, hooks, mesh and the fishing techniques they use. Landing's information was noted at daily basis upon fishermen's arrival at the port at 08:00 a.m. to 13:00 p.m. (Greek time), including identification and measurement (biomass in kilogram, and length in centimetre) of the catch, time of departure, the type of habitat, depth, and the fishing technique/gear used. The recorded information was then input in a database in respect to the gear used at a chronological order.

Only the local fishermen willing to collaborate were interviewed daily and only from the main port of Lipsi Island. Based on personal observation, some fishermen moored at different spots around the island and they were not able to be interviewed. The interviews were done in English and Greek. The present study followed the Savva et al., (2013) report; the Archipelagos report made from this data collection.

3.2.4 Other data from the Survey

To assess the possible impact of the external factor on the results, site weather conditions were recorded at the start of each fish survey. The wind direction was estimated (S, SE, E, NE, N, NW. W, SW), the Beaufort rating, using the NOAA Beaufort scale (NOAA, no date). In the water, at the beginning of each fish survey, the visibility was estimated using a secchi disk to record vertical visibility. Finally the GPS coordinate of the land starting point of each site was recorded with Google Maps™ (fig. 7).

3.3 Data Analysis

The data analysis process was designed to answer the two research questions. In order to answer the first research question, community indices were compared among the habitats to check if any differences were present. These community indices were chosen following the methodology of Giakoumi & Kokkoris, (2013). Shannon-wiener diversity index, Pielou's evenness index and one-way ANOVA were selected along others, because they are some of the most used unvaried statistical tools in ecological data analysis and have been applied in many studies on fish assemblages; therefore the results of the present study were comparable to other studies. Also they are easy to process and there is a great quantity of literature about them. Then to answer the second research question, a cluster analysis and a canonical correspondence analysis were carried out. These two multivariate analyses were used as data mining tools. The cluster analysis explored the possible link between sites regarding fish assemblage features, and the canonical correspondence analysis investigated the relationships between fish species and environmental variables.

All mean summary statistics were calculated with their standard error. In order to meet assumptions of the data analyses tools used all set of data were tested for Normality using Anderson-Darling test and for homogeneity using Levene's test. Moreover for the multivariate analyses (cluster analysis and canonical correspondence analysis) data was standardised, z-scores was used to carry out the cluster analysis and data were transformed to log (x+1) for the correspondence analysis.

One-way ANOVA was used to compare community indices among habitat types for each site (n=3). The community indices calculated and compared were: species richness (number of species), and diversity (Shannon – Wiener H'), which is a measure of the diversity of a community and the evenness (Pielou's J), which refers to how close in numbers species in an environment are (Giakoumi & Kokkoris, 2013). All these community indices were calculated using Minitab 16.

Multi-variant analysis was done to analyse any differences found in fish assemblage between habitats, different percentage of *Posidonia oceanica* and IAS cover and depth using PAST 2. To investigate the possible difference between habitats, a cluster analysis was carried out. It was chosen because of the facts that it does not need certain amount of data to be relevant and for its quality as data mining tool. The cluster analysis aimed to distinguish differences in *Posidonia oceanica* and IAS cover regarding the abundance and the composition of species between habitats. The hypothesis was that if the habitats were different, the sites of the same habitat would cluster together. This analysis was carried out using paired group algorithm and Euclidean distances among habitats.

Canonical correspondence analysis (CCA) was used to describe the relationship between environmental variables (*P. oceanica* cover, IAS cover and depth) and individual fish species. It was selected as a weighted averaging method that directly relates community data to environmental variables by constraining species ordination to a pattern that correlates maximally with environmental variables. The intention was to check if any fish species were impacted by the different variables, and to know which variable affected which species. This analysis was conducted only on abundances of the most common fish species with occurrence frequency >20 percent -19 species- (Giakoumi & Kokkoris, 2013).

4 Results

4.1 External factors and site conditions

The data collection took place in November 2013. 14 sites were studied (n=14). Following the methodology, data collection took place only on good weather days (Beaufort equal or below 4, no rain) in order to minimize the influence of the external factors. The Beaufort range for all data collection is ranged from one to four (NOAA, no date). The depth range and the visibility of each site were recorded (table. 2). The depth range for the all data collection is from 2 to 5.25 meters (±0.13 SE). The visibility range was from 9 to 21 meters (±0.96 SE), the highest visibility was at Platis Gialos site (21m), and the lowest was at Limih and Lalaouni (9m). The 2 figures below show that it does not seem to have a correlation between depth and IAS presence (fig. 14) (all the surveys were done at the same depth range), or between visibility and IAS presence (fig. 15).

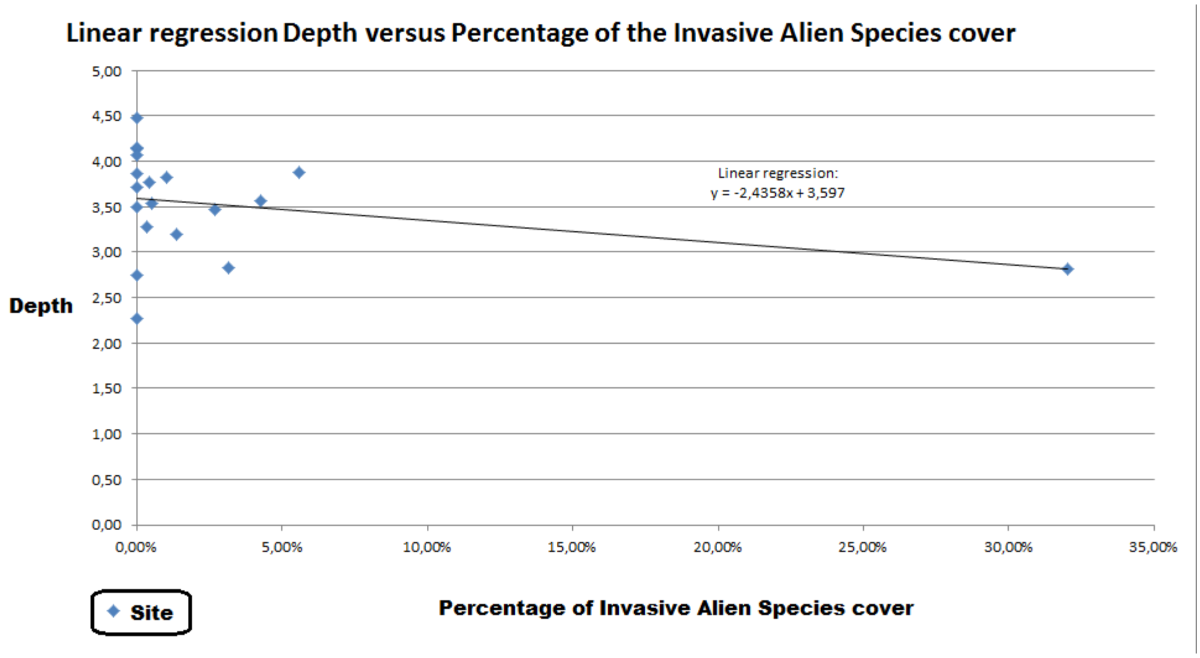


Figure 14. Linear regression of the sites between the depth of the survey versus the percentage of Invasive Alien Species cover

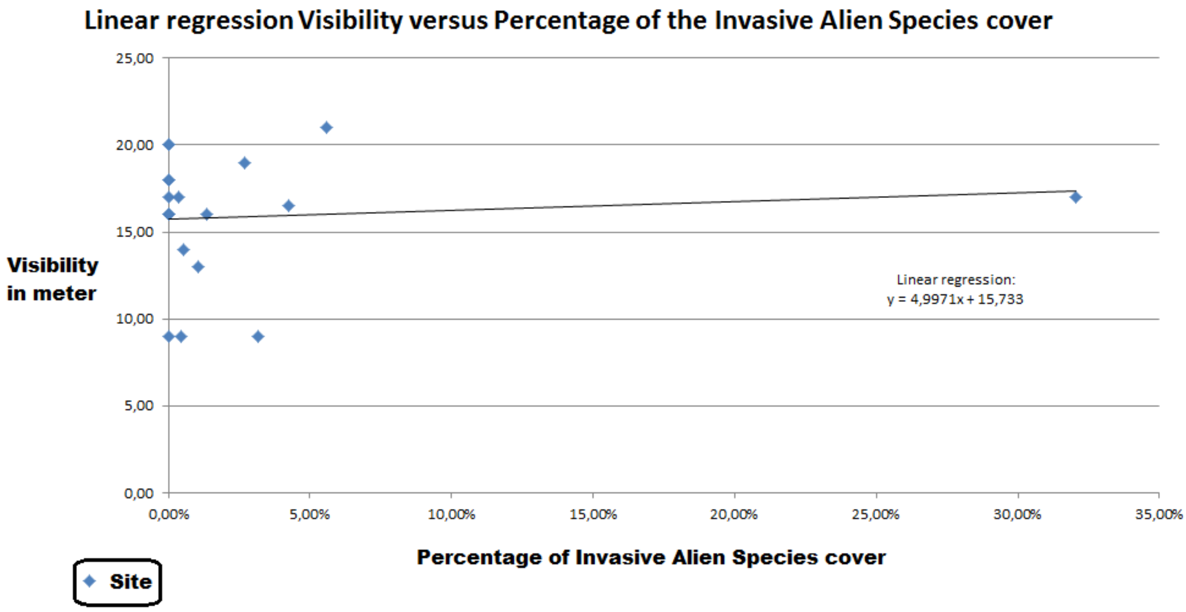


Figure 15. Linear regression of the sites between the visibility of the survey versus the percentage of Invasive Alien Species cover

Table 2. Table of the depth range and visibility of each site studied

Habitat	Site	Depth (m)	SE depth (m)	Visibility (m)
Dense *Posidonia oceanica*	Elena Beach	2.50 - 5.25	± 0,26	16
	Campos	2.25 - 3.25	± 0,11	17
	Kamares	2.50 - 4.75	± 0,19	20
	Limih	2.75 - 4.50	± 0,12	9
	Kimissi	3.25 - 4.75	± 0,11	18
Sparse *Posidonia oceanica*	Elena Beach	3.50 - 5.00	± 0,14	16
	Monodendri	3.50 - 5.25	± 0,13	16
	Kamares	2.00 - 2.75	± 0,04	20
	Kimissi	3.00 - 5.00	± 0,14	18
Sparse *Posidonia oceanica* affected by IAS	Kissiria	2.75 - 4.00	± 0,13	19
	Mersini	2.50 - 5.00	± 0,21	16.50
	Hohlakoura	2.75 - 4.00	± 0,14	14
	Katsadia	2.75 - 5.00	± 0,21	13
	Campos	2.50 - 4.00	± 0,12	17
	Tourcomnima	2.00 - 3.25	± 0,12	17
	Limih	3.00 - 4.25	± 0,11	9
	Platis Gialos	2.00 - 5.00	± 0,23	21
	Lalaouni	2.50 - 3.00	± 0,06	9
	No name 7	2.50 - 4.25	± 0,16	16

4.2 Result of *Posidonia oceanica* and IAS percentage cover

Of the 14 sites studied: five sites had dense and sparse *P. oceanica*; three sites with intact sparse *P. oceanica* (Kamares, Kimissi, Elena beach), and two sites with sparse invaded with IAS (Campos, Limih). Only one site was found with only sparse intact *P. oceanica* (Monodendri). Eight Sites demonstrated only sparse *P. oceanica* affected by IAS (Kissiria, Mersini, Lalaouni, No name 7, Hohliakoura, Katsadia, Tourcomnima, and Platis Gialos). In total, 19 habitats were studied: five dense *P. oceanica* (n=5), four Sparse intact *P. oceanica* (n=4) and 10 Sparse *P. oceanica* invaded by IAS (n=10) (fig. 16).

Figure 16. Map of Lipsi island. Red circles show sites invaded by IAS (n=10), green circle show intact site (n=4). Black stars (n=5) show the sites where dense P. oceanica has been surveyed (Google Maps™)

4.2.1 Result of mean *Posidonia oceanica* percentage cover

The mean seagrass percentage cover in the 19 habitats studied was 50.17% (±7.09% SE). In the dense *P. oceanica* mean seagrass percentage cover was 94.13% (±1.57% SE), in the sparse intact *P. oceanica* 32.31% (±10.51% SE), and in the invaded *P. oceanica* 35.33% (±5.44% SE). The *P. oceanica* percentage cover of each habitat surveyed was calculated (fig. 17). The highest by percentage cover was Campos dense, 99.67% (±0.33% SE) following by Kamares

94.67% (±2.19% SE). The lowest was Campos sparse, 18.50% (±2.22% SE) followed by Elena beach sparse 18.92% (±3.60% SE).

As can be seen in Figure 17, *P. oceanica* percentage cover among the different habitats demonstrates that there is a highly significant difference between Dense *P. oceanica* cover sites and Sparse *P. oceanica* cover sites (intact and invaded). However it does not seem to have a significant difference between intact sparse and invaded sparse *P. oceanica* cover sites.

4.2.2 Result of mean IAS percentage cover

Firstly, it is noteworthy that the present study did not draw differences between the two IAS species. The 10 sites were invaded by the two species. The mean IAS percentage cover in the 10 invaded sites studied was 5.14% (±3.04% SE). The IAS percentage of each affected site studied was calculated (fig. 18). The highest by percentage cover was Tourcomnima, 32.03% (±0.75% SE) followed by Platis Gialos, 5.57% (±0.51% SE). The lowest was Limih, 0.43% (±0.06% SE) followed by Hohlakoura 0.53% (±0.06% SE).

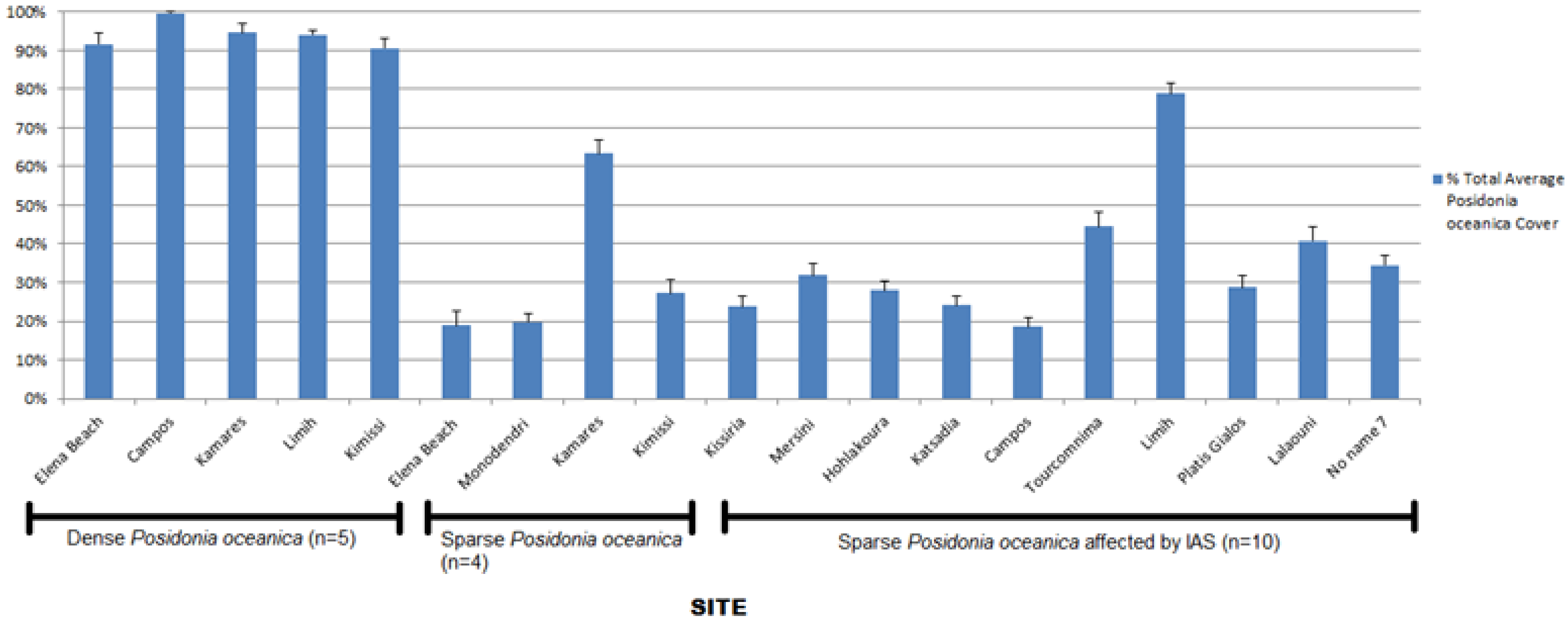


Figure 17. Graphic of the Posidonia oceanica % cover. In vertical axis the % cover form 0 to 100% and in horizontal axis each site studied groups by habitat types.

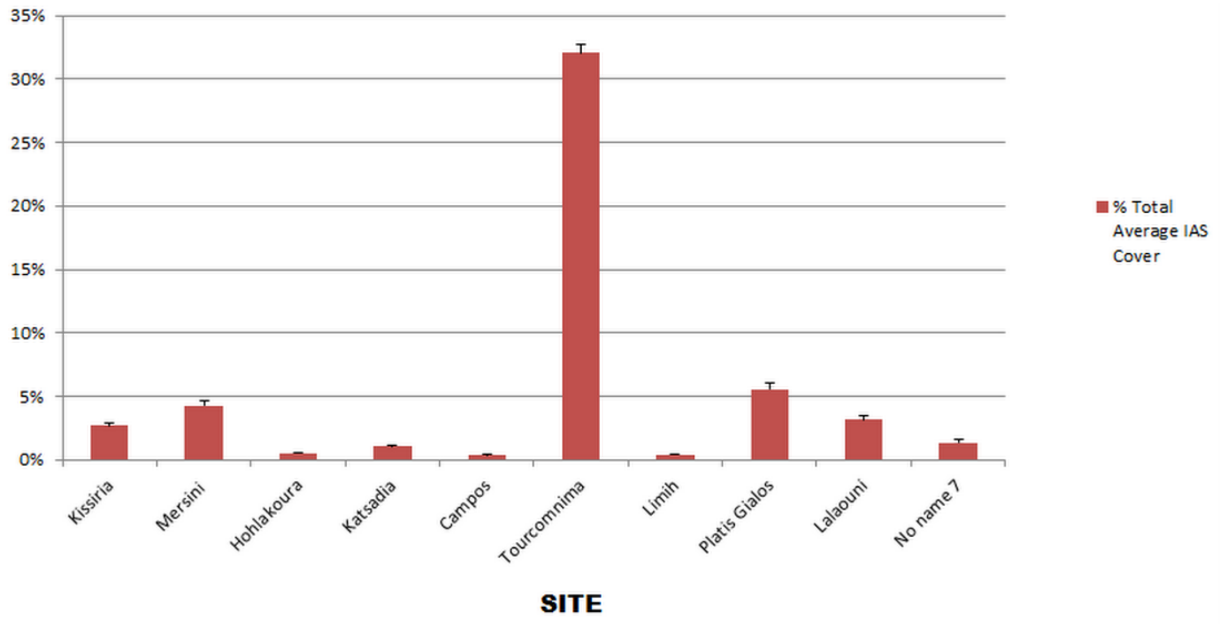


Figure 18. Graphic of the IAS % cover. In vertical axis the % cover form 0 to 35% and in horizontal axis each site studied invaded by IAS

4.3 Result of the fish assemblage UVC

The methods implemented in the present study were comparable to other studies done on fish assemblage in the *P. oceanica* and IAS (Giakoumi & Kokkoris, 2013) in the Aegean Sea, (Guidetti, 2000) in Adriatic Sea and Di Martino, et al., (2007) next to Sicily Island. Regarding the species presented in the results, 25 percent were found in the four different studies (12 species). There were three species (*Spicara maena Symphodus ocellatus Scorpaena porcus*) that were found in three other studies (Guidetti, 2000, Di Martino, et al., 2007, Giakoumi & Kokkoris, 2013) but not in the present one. At the same time, one species (*Siganus luridus*) was found only in the present study. However this can be explained by the fact that *S. luridus* is an invasive species only found in the Eastern Mediterranean Sea (Galil, 2006c). The comparison of the different fish assemblage studies with the present study found a similarity in the species of 42% with the Guidetti, (2000), 37.5% with Di Martino et al., (2007) and 71.9% with Giakoumi & Kokkoris, (2013). This can be seen as a confirmation that the sampling methodology seemed to have been implemented effectively during the data collection and therefore that the results are relevant with respect to the results of other similar studies. It also confirms that Lipsi Island hosts fish communities similar to other fish communities found in the region and in Mediterranean shallow waters.

Overall, 11.350 (±5.7 SE) individuals were counted. Thirty species, belonging to 11 families, were identified (table. 3). In each habitat studied, density was calculated for each species present during the survey (table. 4). In all habitat types, the most abundant species was *A. boyeri (*9,790 individuals, ±128.74 SE). The second and third most abundant species were in the dense *P. oceanica*: *C. chromis* (114, ±13.54 SE) and *D. annularis* (92, ± 3.01 SE). In the sparse *P. oceanica* intact: *C. chromis* (111, ±22.68 SE) and *C. julis* (66, ±4.35 SE) and in the *P. oceanica* affected by IAS: *C. julis* (240, ±5.47 SE) and *C. chromis* (207, ±11.42 SE).

Table 3. List of all the fish species and families recorded in the shallow sublittoral of the Lipsi Island during the present study.

Family – Species	
Atherinidae	***Mullidae***
Atherina boyeri	*Mullus surmuletus*
Bothidae	***Pomacentridae***
Bothus podas	*Chromis chromis*
Gobiidae	***Scaridae***
Gobius bucchichi	*Sparisoma cretense*
Gobius cobitis	***Serranidae***
Gobius paganellus	*Serranus cabrilla*
Labridae	*Serranus hepatus*
Labrus merula	*Serranus scriba*
Labrus viridis	***Siganidae***
Symphodus cinereus	*Siganus luridis*
Symphodus mediterraneus	***Sparidae***
Symphodus melanocercus	*Boops boops*
Symphodus roissali	*Diplodus annularis*
Symphodus rostratus	*Diplodus sargus*
Symphodus tinca	*Diplodus vulgaris*
Thalassoma pavo	*Sarpa salpa*
Coris julis	*Oblada melanura*
Lithognathus mormyrus	***Syngnathidae***
	Sygnathus typhle typhle

The community indices have been calculated for each site surveyed (table. 5). The total abundance, the total number of fish count among transects which has been recorded for each site is presented as well as the species richness, the number of species found during the survey. The table shows also the result for each site of the Shannon-Wiener Diversity and Pielou's

Evenness, indices explained in the literature review section. ANOVA’s results for each of the community indices did not find significant differences between habitats: species richness ($F = 0.16$, $p < 0.851$), Shannon-Wiener diversity ($F = 0.28$, $p < 0.756$), Pielou’’s evenness ($F = 0.42$, $p < 0.664$). These results did not allow showing the impact of IAS and the importance of the dense *P. oceanica* on the species richness, Diversity and Evenness.

Table 4. Table of the density (individual per m^2) calculated for each species in every sites studied

Site	Elena beach	Campos	Kamares	Limih	Kimissi	Elena beach	Monodendri	Kamares	Kimissi	Kissiria	Mersini	Hohliakoura	Katsadia	Campos	Tourcomnima	Limih	Platis Gialos	Lalaouni	No name 7
HABITATS	Dense PO	Dense PO	Dense PO	Dense PO	Dense PO	Intact Sparse PO	Intact Sparse PO	Intact Sparse PO	Intact Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO	Invaded Sparse PO
C. chromis	0.75	0.12	0.02	0.03	0.22	0.95	0	0.01	0.15	0	0	0.95	0	0	0	0	0.37	0	0.75
C. julis	0.07	0	0.04	0.16	0.11	0.12	0.18	0.08	0.28	0.07	0.05	0.4	0.5	0.26	0.06	0.2	0.07	0.39	0.4
D. annularis	0.28	0.14	0.22	0.17	0.11	0.11	0.01	0.19	0.08	0.03	0.04	0.01	0.03	0	0	0	0.15	0.08	0.05
D. sargus	0.01	0.08	0.03	0	0.03	0.02	0	0.13	0	0.01	0.01	0.02	0.02	0	0.04	0	0.03	0	0.03
O. melanura	0.01	0.11	0.17	0.15	0	0	0	0.18	0	0	0	0	0	0	0	0	0	0	0.1
S. luridis	0.08	0.13	0	0.07	0.07	0	0	0	0	0	0	0	0	0	0	0	0.05	0.02	0.06
T. pavo	0.07	0	0.03	0	0.01	0	0.02	0.05	0	0	0	0.11	0.02	0	0	0	0	0.01	0.09
S. cabrilla	0.01	0.02	0	0.01	0.01	0.02	0.06	0.01	0.03	0.02	0	0.05	0.05	0.03	0.01	0	0	0.04	0.07
S. scriba	0.02	0.01	0.03	0.04	0.04	0	0.01	0.03	0.01	0.01	0.02	0.08	0.04	0	0.01	0	0.01	0.01	0.04
S. cinereus	0.01	0	0	0.01	0	0.03	0.02	0.02	0	0.03	0.05	0.02	0.04	0.12	0.03	0	0.02	0	0.05
S. typhle	0.01	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0.01	0	0	0	0
S. tinca	0.02	0.07	0.02	0.07	0.02	0	0.01	0.05	0.04	0.01	0	0.05	0.05	0.1	0.08	0.05	0.15	0.12	0.02
M. surmuletus	0	0	0	0	0.01	0.01	0	0	0.02	0	0	0.03	0.03	0.03	0.01	0	0	0	0.01
A. boyeri	0	3.5	16	7.5	5	0	3.5	0	4.25	4.25	18.5	0	3.5	3.5	15	8.25	3.5	0.9	0.75
G. paganellus	0	0	0	0	0	0	0.01	0	0	0.01	0	0	0	0	0	0	0	0	0
G. bucchichi	0	0	0	0	0	0	0.05	0	0.01	0.01	0	0.01	0.01	0	0	0	0.04	0	0
L. viridis	0	0	0.01	0.04	0	0	0.04	0.01	0	0.02	0.04	0.03	0.01	0.03	0	0.02	0.02	0	0
D. vulgaris	0	0	0	0	0	0	0	0.02	0	0.02	0.01	0	0.01	0	0.01	0	0.04	0	0.02
L. mormyrus	0	0	0	0	0	0	0	0	0	0.02	0.05	0	0	0	0	0	0	0	0
B. podas	0	0	0	0	0	0.01	0	0	0	0.01	0	0	0	0	0	0	0.01	0	0
S. roisalli	0	0	0	0.03	0.02	0	0	0	0	0.01	0	0.03	0.01	0	0.01	0	0	0.02	0.03
S. rostatus	0	0	0.01	0	0	0	0	0	0	0.02	0.02	0.02	0	0	0	0	0.03	0	0.03
S. salpa	0	0	0	0	0.06	0	0	0	0	0	0	0.03	0	0	0	0	0.01	0	0.15
G. cobitis	0	0	0	0	0	0.01	0	0	0	0	0	0.01	0	0	0	0	0	0	0
S. mediterraneus	0	0	0	0	0	0	0	0	0	0	0	0.01	0	0	0.01	0	0	0	0
B. boops	0	0	0	0	0.16	0	0	0	0	0	0	0	0.05	0	0	0.05	0	0	0
S. melanocercus	0	0	0.01	0.04	0	0	0	0	0	0	0	0	0	0	0.01	0	0	0	0
L. merula	0	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0
S. cretense	0	0	0	0.07	0.01	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0
S. hepatus	0	0	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0

Table 5. Table of the result of the community indices (Abundance, Species richness, Diversity and Evenness) calculated for each habitat of the sites surveyed.

SITE	Habitat	Abundance	Standard error abundance	Species richness	Standard error richness	Diversity (S-W)	Standard error diversity	Evenness (P's)	Standard error evenness
Elena Beach	Dense PO	134	18.19	12	1.00	1.08	0.38	0.58	0.18
	Intact Sparse PO	128	20.00	9	0.88	0.90	0.34	0.52	0.17
Kimissi	Dense PO	588	95.73	15	0.33	0.85	0.23	0.36	0.09
	Intact Sparse PO	489	101.12	11	0.67	0.73	0.34	0.37	0.16
Kamares	Dense PO	1659	252.38	12	0.33	0.56	0.42	0.28	0.22
	Intact Sparse PO	81	8.57	14	0.88	1.80	0.08	0.84	0.05
Monodendri	Intact Sparse PO	391	117.85	11	0.88	0.97	0.43	0.62	0.25
Campos	Dense PO	408	117.48	9	0.33	1.34	0.50	0.67	0.25
	Invaded Sparse PO	407	115.18	7	0.33	1.02	0.39	0.60	0.24
Limih	Dense PO	839	239.29	14	1.20	1.38	0.65	0.61	0.29
	Invaded Sparse PO	857	237.74	5	0.58	0.52	0.29	0.54	0.26
Kissiria	Invaded Sparse PO	455	107.58	16	0.33	0.88	0.57	0.43	0.27
Mersini	Invaded Sparse PO	1879	132.22	10	1.86	0.12	0.05	0.06	0.02
Lalaouni	Invaded Sparse PO	159	24.88	9	1.20	1.11	0.19	0.62	0.09
No name beach 7	Invaded Sparse PO	265	22.70	17	0.67	1.46	0.22	0.60	0.08
Hohliakoura	Invaded Sparse PO	186	29.14	17	2.00	1.54	0.07	0.69	0.08
Katsadia	Invaded Sparse PO	437	119.69	15	1.00	1.03	0.33	0.52	0.18
Tourcomnima	Invaded Sparse PO	1529	251.36	13	1.86	0.56	0.46	0.34	0.29
Platis Gialos	Invaded Sparse PO	450	120.03	15	2.08	1.03	0.44	0.53	0.19

In order to better understand the relationship between the different variables, multivariate analyses were carried out as data mining. A cluster analysis was conducted on the abundance (density) and species present (species richness) between each habitat of the sites studied. The result of the cluster analysis shows the similarity of fish faunas among habitats and between sampling locations. The result can be seen in the graph below (fig 17). The first comment is that dense *P. oceanica* habitats had clustered together, except Limih dense *P. oceanica* habitat. Different hypotheses can be made to explain this, the fact that the site was invaded by IAS (but Campos was also invaded), the fact that the site had the worst visibility (9m) of the all data collection or perhaps the situation of the site, located at the South-East of the Island, isolated to the other dense *P. oceanica* sites.

Secondly, the sites seem to be clustered around geographical location and proximity between the sites. Katsadia, Lalaouni and Limih (sparse) are all located in the bay of the South of the Island. Campos (sparse), which clustered next with the three previous sites, is also located in a site protected of the sea, in the bay of the harbour. Then the next, which clustered with this group was Kimissi (dense), site relatively protected. These five sites are all located on the South of the island.

A third group contains Monodendri, Kissiria and Mersini sites, all located on the North of the island. However, at the same time Hohlakoura and Tourcomnima, two sites close to each other, and located on the East of the island, did not cluster together. It could be explained by the fact that Tourcomnima is the only site significantly affected by IAS (32.02% of IAS cover). Platis Gialos, No name 7 and Kimissi (sparse) did not cluster with any other sites. A hypothesis can be that these sites are remote from the others. Platis Gialos and Kimissi (sparse) were the two only sites on the West of the island surveyed. No name 7 is the only site located on this part of the coastline.

To summarize the result of the cluster analysis, Dense *P. oceanica* habitats did cluster together and so, seemed to show a difference in the abundance and species composition found in this habitat compared to the sparse (invaded and not) *P. oceanica* habitats surveyed. The second result is that the determinant factor for the clustering of the other sites seems to have been the location around the island. The IAS presence or absent did not seem to have had any influence on the analysis among the sparse habitats.

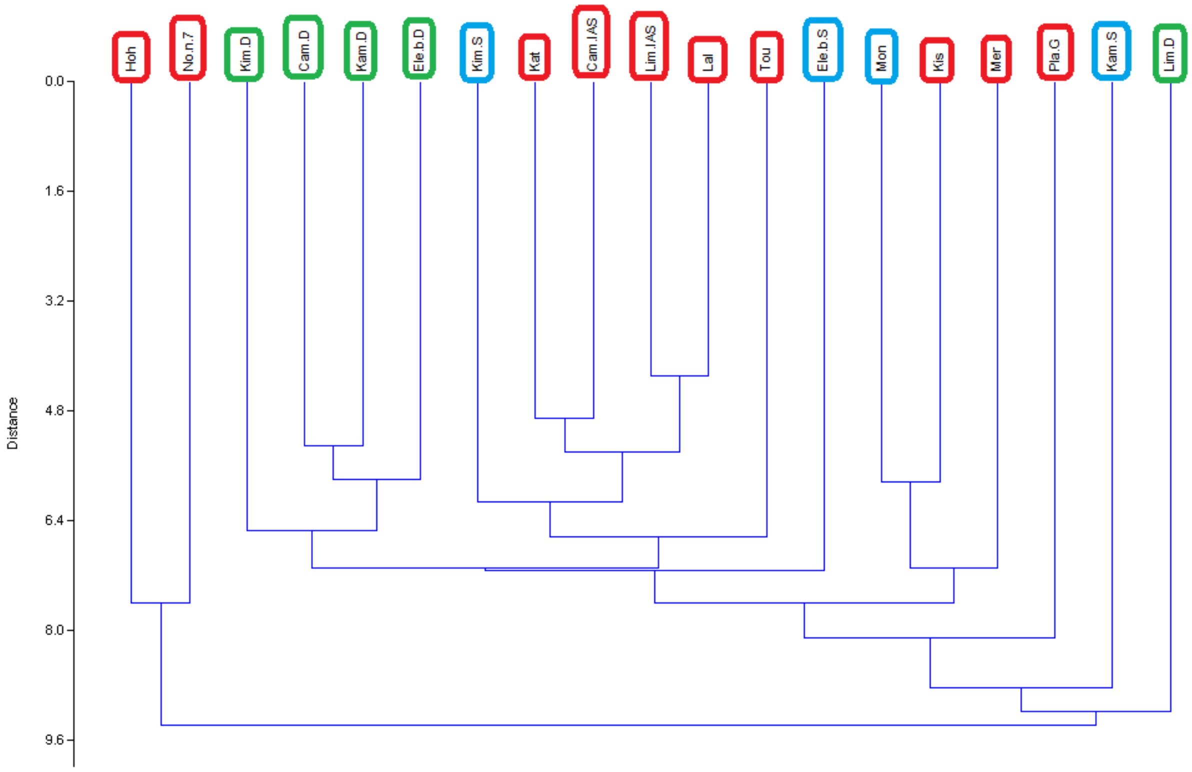


Figure 19. Dendrogram showing the results of the cluster analysis of the abundance and species present among the habitats of each site surveyed (Euclidean distance). In green: dense P. oceanica sites. In blue: sparse P. oceanica sites. In red: sparse invaded (by IAS flora) P. oceanica sites - Lim.D – Limih Dense. Lim.IAS – Limih Sparse. Kim.D – Kimissi Dense. Kim.S – Kimissi Sparse. Kia – Kissiria. Mon – Mondendri. Cam.IAS – Campos Sparse. Pla.G – Platis Gailos. Ele.b.S – Elena beach Sparse. Ele.b.D – Elena beach Dense. Hoh – Hohlakoura. Kam.S – Kamares Sparse. Lal – Lalaouni. No.n.7 – No name 7. Kam.D – Kamares Dense. Tou – Tourcomnima. Mer – Mersini

The cluster analysis focused on site similarities and differences. It characterized the fish community data by density and species richness. The canonical correspondence analysis focused on the impact of the environmental variables on fish species. It aims to see if the IAS could have an impact on one or more species. It was carried out between species present (species richness), above an occurrence of 20 percent for the entire survey, abundance (density) and depth, and percentage of *P. oceanica* and IAS cover (fig. 20). In a canonical correspondence analysis, the axes are a derived of the environmental information (Depth, *P. oceanica* and IAS percentage cover) used for the analysis. Axis one summarized 78.87 percent of the total environmental information. Axis two summarized 21.13 percent. So, the graphic below (fig. 20) is a summary of almost 100 percent of the environmental information used to create this chart.

The results of the analysis, based on the interpretation of the graphic (fig. 20), showed that *A. boyeri, D. vulgaris, S. roissali* and *S. tinca* seem to be slightly positively affected by IAS presence. *O. melanura, S. luridis, S. scriba* and *D. annularis* form another species group, which seemed to prefer areas with the highest percentage cover of *P. oceanica*. Then *S. rostatus, S. cabrilla, C. julis, L. viridis, M. surmuletus* and *G. bucchichi* form a third group, which gather the species that seem to be the most impacted by the depth. Finally, *C. chromis, T. pavo* and *S. salpa* seemed to be negatively impacted by the presence of IAS.

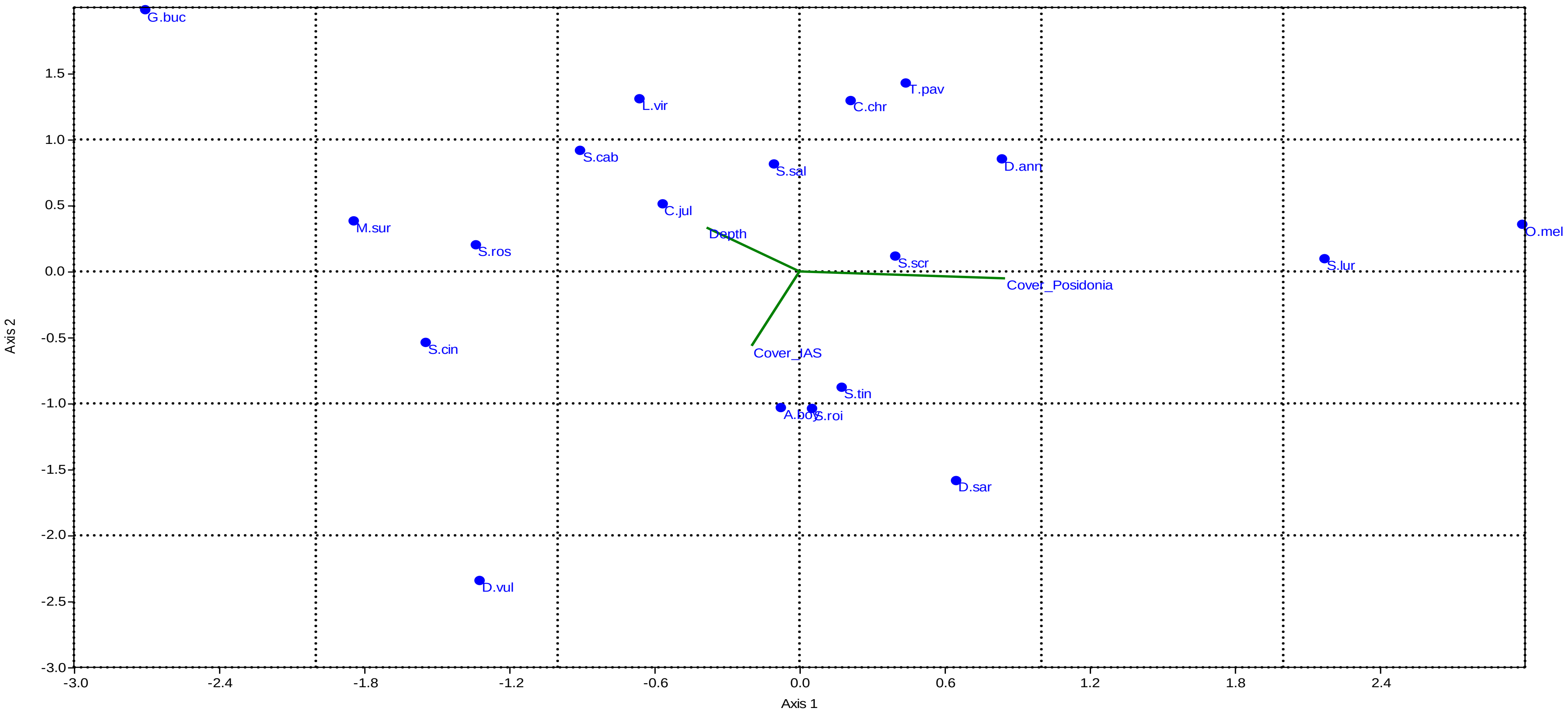


Figure 20. A CCA plot of P.oceanica % cover, IAS % cover, depth variables and species present / abundance. C.chr – C.chromis. C.jul – C.julis. D.ann – D.annularis. D.sar – D.sargus. O.mel – O.melanura. S.lur – S.luridis. T.pav – T.pavo. S.cab – S.cabrilla. S.scr – S.scriba. S.cin – S.cinereus. S.tin – S.tinca. M.sur – M.surmuletus. A.boy – A.boyeri. G.buc – G.bucchichi. L.vir – L.viridi. D.vul – D.vulgaris. S.roi – S.roissali. S.ros – S.rostatus. S.sal – S.salpa.

5 Discussion

5.1 Finding on fish assemblages

The fish assemblages found in the seagrass of Lipsi's shallow water is similar to what was found in previous studies in the area (Guidetti, 2000, Di Martino, et al., 2007, Giakoumi & Kokkoris, 2013). Moreover, the fish community census on the *Posidonia oceanica* meadows of Lipsi Island showed species richness similar to that which has been evaluated by visual census in other *P. oceanica* meadows in the region (Giakoumi & Kokkoris, 2013).

From a quantitative point of view, the strong dominance of the *A. boyeri* (86.3% of the total individuals observed), can be explained by the behaviour of *A. boyeri* to form a big schooling group above the *P. oceanica* in shallow waters and its behaviour to be attracted to the surveyor (Harmelin-Vivien et al. 1985). The abundance of *C. chromis* and Labridae (mainly represented by *C. julis*) could be determined by the heterogeneity of the substrate (*P. oceanica* / Sand / Rock). For the *C. chromis,* the presence of rocky habitats and consequently, shelter, is one of the main explanations for the abundance of this species (Guidetti et al., 1998). Regarding the Labridae species, the heterogeneity of the habitats including seagrasses and rocky substrate colonized by arborescent algae where juveniles and adults of Labridae actively feed, is one of the main factors that determined the abundance of this family (Guidetti et al., 1998).

At the same time, the low abundance of Sparidae, Serranidae and Mullidae, including several species of commercial interest, could be attributable to the effects of fishing activities actively exerted in the studied area. Indeed, the species of these families are targeted by the local fishermen (Savva et al., 2013). Finally, some individuals of Scorpaenidae family were seen outside of the survey, but like other cryptic / small species, (Trachinidae, Tripterygiidae, and Blennidae); they were not seen during the surveys and therefore not recorded. For the same reason Syngnathidae, Bothidae, Gobiidae may have been under-estimated (Harmelin-Vivien et al. 1985). Also, it is worth mentioning the presence of one Lessepsian migrant: *Siganus luridus*.

To sum up, the fish community associated with the Lipsi *P. oceanica* meadows showed general features similar to those observed by other authors in other Mediterranean areas. Species richness and abundance do not depend only on the shoot density, but also on the substrate type on which phanerogams have settled, and on the influence of adjacent habitats (Guidetti et al, 1998, Giakoumi & Kokkoris, 2013).

5.2 Impact of *Posidonia oceanica* cover

The result of the cluster analysis suggested a possible difference between the dense *P. oceanica* habitat and the sparse. Same results were found in others studied (Guidetti et al., 1998, Giakoumi & Kokkoris, 2013), and confirm the ecological importance of the dense patches of *P. oceanica*. The canonical correspondence analysis showed that some fish species seemed to be positively affected by the percentage cover of *P. oceanica*. It is noteworthy that two of these species (*O. melanura* and *D. annularis*) are part of the Sparidae family, and are actively targeted by the local fishermen (Savva et al., 2013). At the same time, based on personal observation *S. luridis*, which also seem to be positively affected by *P. oceanica* percentage cover, is also caught by the local fishermen, but it is seen as a by-catch by them. Indeed this fish is occasionally poisonous (with all spines slightly venomous), with a very painful but non-lethal sting. Several cases of ciguatera-like effects have been attributed to consumption of *S. luridus* (Streftaris & Zenetos, 2006). This observation leads to the question of whether protection of *P. oceanica* ensures the sustainability of local fisheries since it also favours an invasive species.

5.3 Effect of Invasive Alien Species

Unlike most of other studies on the impact of IAS on fish assemblage (Relini et al., 1998, Streftaris & Zenetos, 2006, Di Martino, et al., 2007, Klein & Verlaque, 2008), the results of this study do not show differences between intact and invaded habitats among the fish assemblages. At the same time, regarding the numerous limitations (explained after) of the present study, the two hypotheses cannot be confidently answered. However, regardless these limitations, some conclusions can be drawn with some confidence.

Contrary to other studies, which surveyed homogeneous invaded habitats (IAS cover around 100%), this study analysed invaded habitats with a low percentage of IAS. When the substrate is heterogeneous with a dominance of the native species (in this case *P. oceanica* with *C. cylindracea* and *H. stipulacea*), the fish community does not seem to be affected. The results of the present study seem to support the finding of Ulas et al., (2011). Changes in species cover, number of species and diversity have only been recorded when *C. cylindracea* has overgrown the substrata and impoverished the algal assemblages (Streftaris & Zenetos, 2006). An explanation can be that even if the IAS impacts some species and structure (Piazzi et al., 2005, Galil, 2006a, Galil 2006b), the fish assemblages may be robust enough to accommodate these changes through mechanisms like prey switching and modifications to feeding behaviour.

Heterogeneity of the substrate is an important factor for biodiversity (Guidetti, 2000, Gratwicke & Speight, 2005, Giakoumi & Kokkoris, 2013). It seems that when *C. cylindracea* and *H. stipulacea* constitute a low proportion of the total substrate percentage cover, they participate in the heterogeneity of the habitat and seem to not impact the fish community.

However, this is the first study done on the presence and the impact of IAS in the shallow water of Lispi. There was no information on the date of the IAS introduction. It may be recent. Without baseline data, it is impossible to know what will be the comportment of the IAS in the future, but it seems logical that it will expand. In order to manage IAS, a follow-up study on fish assemblage over the next few years should be conducted, to evaluate possible changes, and to monitor the growth of IAS in the shallow water of Lipsi. Furthermore a *Posidonia oceanica* monitoring programme could be implemented.

5.4 Effect on Artisanal fisheries

Worldwide, significant amounts of human-exploited species are facing population declines. This is not only a direct result of overfishing, but habitat changes and degradation have also been identified as a potential large scale influence of this decline (Seitz et al., 2013). The results of this study, however, did not prove that IAS have an impact on local fish communities, therefore on local Lipsi fisheries. A similar result was found in the region (North of Samos Island) for *C. cylindracea* impacts on catch by Ulas et al., (2011).

However, in the present study, the lack of previous local data on IAS presence and fish catch landings make it impossible to estimate a temporal evolution. The local fishermen complained about a diminution of the stock quantity and length of individuals over the last years but without data, and taking the local situation into account, it seems to be the result of a long-term over-exploitation of the local fish resource (Milliou, personal communication, September 2013). With this decrease in fish abundance, the fishermen must to work more time and with more nets to make up for the low fish level. The situation seems to be a case of tragedy of the commons, and only efficient management seems to be able to change the current situation. Finally IAS do not appear to have a significant importance for the local fishery so far.

5.5 Study limitation

The present study contained a lot of limitations of different kinds, which diminish the power and the robustness of the ecological data results and do not allow refuting or confirming the hypotheses. First, with the theoretical assumption, there was a difference of depth between the snorkel surveys -2 to 5meters- and fishing location -around 30meters- (Savva et al., 2013). The comparison had been possible only because the present study assume the life-history pattern of fish has totally scientifically true and affecting all the fish community. However, difference among fishes exists in their life-history patterns, and this theory is not accepted by all the scientific community (Verlaque, personal communication, November 2013).

The UVC protocol used was simplified with respect to other UVC protocols (McKenzie et al., 2003, Short et al., 2006). Advanced techniques mix cover estimation done with quadrats (canopy height, shoot density, biomass measurements) with UVCs done with transect and fyke nets to improve the robustness of the results. The methodology used for the present study did not mix different techniques of measurement. Moreover, a survey about fish pattern of each species would have helped to get a better picture of the situation and estimate if some species were missing from the present survey through breeding season pattern and specific behaviour. However, as it is impossible to survey each species at the best time of the year and as fishes forms a community, this community needs to be survey as a whole. The survey took place in November. It was not the best time in the year to get a clear picture of the fish community of Lipsi, as the CIESM recommends performing UVC between June to October (CIESM, 2012). However an informal comparison with precedent results of UVC done on Lipsi during the summer 2013 shown strong similarity and the results obtained were similar to other studies done in the Mediterranean Sea (Guidetti, 2000, Di Martino, et al., 2007, Giakoumi & Kokkoris, 2013). These similarities could be interpreted as a confirmation that even though a simplified sampling methodology was used, not at the best time of the year, it gave relevant results.

The study area might not have been a relevant place to study the impact of IAS on fish assemblage. Indeed as the figure 18 showed, the percentage cover of IAS on the sites affected was very low, eight sites below five percent of IAS cover, one with 5.57 percent (Platis Gialos) and only one with a significant percentage (32.02%) of IAS cover (Tourcomnima). The project had been framed by Archipelago NGO, and they choice of Lipsi Island as the location for the present study was one requirement of the NGO. Form the Greek NGO knowledge, the shallow waters of the island were greatly invaded by IAS and the present study was a part of a project to assess the situation.

The UVCs method has some limitations; one of them is the bias due to the surveyor. The only way to decrease the bias of this factor is in training. Even if the fish will always be attracted to the surveyor, the training can minimize mistakes. Before, data collection, the team has had to pass a test, with a 95 percent pass rate, assessing fish species knowledge before starting the survey. In addition, the team trained the whole October 2013, helping other students with their projects. However, this training might not have been enough and the data collection was done by volunteers and not supervised by scientific on the field. Also, the time of the UVC was not the most adapted of the survey, early morning would have been much better time to do UVC (Verlaque, Personal communication, April 2014).

The duration of the data collection of the present study did not allow development of a clear picture of the fish community around Lipsi Island. Indeed, due to numerous limitations the data collection lasted only a month. This time frame did not allow observation of inter-seasonal variation and even less annual variation (Harmelin-Vivien et al., 1985). At the same time, *C. cylindracea* reaches its maximum development in autumn; the season of high growth lasts from June to October in Mediterranean Sea (Ruitton et al., 2005). So, it is possible to hypothesise that the data collection of the *C. cylindracea* was done at the best time of year to be able to identify it.

The daily variations, environmental variations, and the human frequentation of the site, impact fish behaviour and bias the data collection (Harmelin-Vivien et al., 1985). To diminish errors due to these factors, as explained in the methodology, surveys on each site, were done at the same hour of the day (between 14:00 to 15:00) and surveys were only done if there were good weather conditions. Also, sites were not surveyed if there were other persons in the water, but occasionally boats passed next to the site during the survey.

Finally, the main limitation was the data analysis. Even if the data were checked for normality, homogeneity and were transformed for the multivariate analysis, numerous assumptions were not check and only presumed (independency, linearity, random). Moreover, the very small sample size (n=19) greatly diminished the robustness of the analytic tools used. That is why; the results of the data analysis of the present study were too weak and did not allow refuting or confirming the hypotheses.

6 Management recommendations

When it comes to ecosystem service assessments, there are three vital consideration that need to be addressed to allow for successful dependability and adoption: credibility, legitimacy, and relevance to decision makers' needs (Ash et al, 2010). The assessment acts as a bridge between the scientific and decision making realms, and therefore has a responsibility to be adequately explicit and technically proficient. The achievement of social and economic goals is dependent on the reliability and robustness of scientific information that helps to formulate ecosystem management. A sound document is capable of addressing the needs and concerns of both the development and environmental communities. The opportunity for informed decision-making based on assessment findings greatly improves a planner's ability to weigh alternatives, understand the implications of their actions, and engage a variety of services and tools related to management. This is a direct result of describing and valuing the benefits of ecosystem services. In terms of integrating the results of an ecosystem services assessment, opportunities are plentiful for both the environmental perspective as well as that of the development planners. Room for incorporation exists at all levels of governance, irrespective of personal roles (Ash et al, 2010).

As it was been shown in the result section, IAS do not look like to impact fish community around Lipsi Island at the moment. The reason of the fish community degradation seems to be from fishery through an over-exploitation of the resource over the last years according to the local fishermen. At the same time, it is now widely known that IAS invasions in marine habitats represent a threat to the integrity of native communities, the local economy, and therefore local human well-being. Moreover worldwide, the global economic costs of IAS are estimated by IUCN to be about US$400,000 million annually (UNEP 2003). They are believed to accelerate the decline in native populations already under environmental stress, leading to population losses and extinctions on a local scale (Streftaris & Zenetos, 2006). These information and the results of the other studies should be used for support action in order to avoid similar situation, as now the possible degradation of the fish community by the IAS is not yet scientifically visible around Lipsi Island. These actions need to be taken regarding the local situation and what has already been done and tested in other case studies. Also the results seemed to show the important of the dense and healthy *P. oceanica*, results support by the scientific knowledge. Regarding the works, which have been done in other places and the current local situation of *P.oceanica* and IAS around Lipsi Island, the main recommendations at the local level could include:

1. Teaching fishermen and locals to identify IAS. Local population and stakeholder acknowledgement of the scientific work done in their environment can be seen as one of the first steps of an efficient management plan. Indeed, to make them realize the current situation and potential impacts of the status quo is critical for a management project. In the present case, the stakeholders are the sea users (fishermen, recreational fishermen, ferrymen, and sailors). Based on personal observation, these different groups did not know about IAS and did understand the work of volunteer during the data collection. Spread the information should be done by public Greek institutions and local NGOs like Archipelagos.

2. Teaching fishermen and locals how to clean their boats and fishing gear. The spread of the *C. racemosa* and *H. stipulacea* is mostly done by anthropogenic factors (Galil 2006a, Galil 2006b). At the local level, the cleaning of boat and fishing gear could be a good way to diminish the spread of IAS around Lipsi and on other islands. Indeed, based on Klein & Verlaque, (2008) and personal observations, fishing gear, anchors, ballasts and vessel' hulls carry zygotes and fragments or propagules of IAS. If all this equipment is not cleaned between the trips, it will lead to the spread of IAS from invaded sites to pristine locations.

3. Monitoring the spread with mapping. After the local population is briefed about the current situation and know how to recognize the different IAS, public Greek institutions or local NGOs like Archipelagos need to monitor the long-term spread. This can be done by annual surveys in different locations in order to assess the presence of IAS and its temporal and geographical evolution. Local populations can help as well; they can report where they saw IAS and contribute to the mapping and monitoring of the spread.

4. Organize removal operations in the affected areas regularly and protect the pristine areas. Form the mapping, the situation of spread around Lipsi will be known and understood. Long term monitoring will give geographical trends of the spread. Public meetings with local stakeholders should frame the different locations that the population wish to protect in priority regarding the IAS spread evolution. Removal operations can be one of actions that can be developed. No take zones in some areas would also diminish the spread of IAS while sustaining local fisheries and increase catch size (Vandeperre, et al., 2011).

5. Enhancing *P. oceanica* protection. As the results of the present study and others (Klein & Verlaque, 2008, Deudero et al., 2011, Shepperson et al., 2013) showed, the IAS surveyed did not creep into dense/healthy *P. oceanica*. Its meadows prevent IAS colonisation. However, *P. oceanica* is impacted by numerous anthropogenic actions, which need to be explained to the local population to allow its protection.

And at the national / European level:

6. Forbidding the sale of IAS, for personal aquarium for instance (like *Caulerpa* species). Based on Verlaque, personal communication, (April 2014), the commercialisation of IAS and their international sales is a means of spreading IAS. Indeed, these IAS can be released, intentionally or not, in a pristine environment, and contribute to the spread. Forbidding the sale of the most damaging IAS could be a first step for biodiversity protection.

7. Following the guidelines of the European Union Strategy on Invasive Alien Species. The European Union has already framed a strategy on IAS (Shine et al., 2010). As said before, the action against the spread of IAS has to involve all the stakeholders, and public administrations should push for the acceptance of this strategy.

Invasive Alien Species issues involve complex interactions between political, economic, social and technical factors. Moreover, as Cullen-Unsworth et al., (2013) explain, seagrass meadows represent a coupled social–ecological system and the degradation of the resource can directly impact the well-being and sustainability of coastal communities. With the challenges of managing and eradicating IAS invasions, especially *Caulerpa cylindracea* invasions (Zuljevic et al., 2007, Klein & Verlaque, 2008), prevention seems to be the best way to protect areas. Indeed, even if it is too late in the study area to prevent the spread of the IAS. Only a mix of prevention / control of the spread /eradication actions could diminish the spread of IAS around Lipsi. If the results of the present study are assumed to be correct, IAS around Lipsi at low percentage cover do not seem to impact the fish community and therefore the local fisheries. Following the previous recommendations the managing plan could be simplified as:

- 1. Prevent the future colonisation of the pristine locations of Lipsi's shallow water.
- 2. Monitor the spread of the invaded sites in order to keep a low IAS percentage.
- 3. Eradicate the IAS if their spread starts to affect the fish community.
- 4. Protect the *P. oceanica*, as a barrier against the spread of IAS and key-ecosystem of the shallow waters of Lipsi.

Take into action these recommendations will need resources and local help. Regarding *C. cylindracea*, eradication attempts have been performed only at small scales and have been limited to very high value areas (Zuljevic et al., 2007, Klein & Verlaque, 2008). Manual eradication is time-consuming, needs the involvement of a large number of SCUBA divers, and the results are random (Galil, 2006a).

> *"Eradication of the Caulerpa racemosa var. cylindracea colonies was exceptionally difficult. Small fragments are usually invisible until stolon reaches length of 10 - 20 centimetres. Due to constant eradication efforts number of colonies and covered area was significantly reduced. Caulerpa racemosa var. cylindracea is an exceptionally invasive species. Its eradication is difficult due to small fragments, successful reproduction and fast growing thalli."* (Zuljevic al., 2007: 1)

Other methods have been tried. Covering colonies with black PVC plastic and removal by suction pump were ineffective (Galil, 2006a). Injecting liquid or solid chlorine or coarse sea salt to sealed-off areas was tried. Off the Montenegrin coast copper-sulphate solution and lime were injected under PVC foil – with no success (Galil, 2006a). Research in the natural habitat of *C. cylindracea* (South-West Australia) on its natural predators, diseases or parasites could provide a basis for understanding the biology of this green alga and help find a possible control mechanism (Klein & Verlaque, 2008), although this could lead to the introduction of a new invasive species.

Regarding *H. stipulacea*, no work seems to have been published regarding eradication attempts (Galil, 2006b, Guiry & Guiry, 2014). However eradication attempts mean that the IAS have already invaded. Monitoring IAS is a long-term effort and an expensive strategy. Control of the spread and prevention seem to be the best strategies against IAS. One proposition to prevent the spread of *H. stipulacea* in the Mediterranean basin is to erect a salinity barrier in the Suez Canal in order to reduce the number of Lessepsian migrants arriving (Galil, 2006b). As a natural phenomenon, IAS do not respect political boundaries. Invasive Alien Species management strategies need to be put into action encompassing all countries affected by the problem. It is the only way for them to be effective.

Public awareness needs to be improved in order to communicate to the public good and bad behaviours. For example a project of large-scale communication on *Caulerpa taxifolia* was done and gave good results:

> *"Multi-language leaflets and posters as well as a video were produced and distributed in eight Mediterranean countries (Spain, France, Italy, Malta, Croatia, Tunisia, Algeria and Turkey). The effectiveness of this campaign was remarkable: tourists and residents contributed to the discovery of new colonies of C. taxifolia which were subsequently removed, thereby slowing-down the spread of the "killer alga" in the Mediterranean Sea."* (Díaz-Almela & Duarte, 2008: 13).

Another way to increase the public awareness is by the ecosystem assessments. The results can clearly help to establish and show the connections between people and the environment, by investigating the balance between the supply and demand for ecosystem services (Potschin and Haines-Young, 2012). For the *C. cylindracea* and more generally IAS; partnerships with fishermen, diving clubs and other sea professionals have been and can be created to promote working practices that reduce the risk of spread. These partners can also report their incidental observations of new points of growth to a coordinating agency. Annual eradication activities can decrease the rate of spread (Díaz-Almela & Duarte, 2008).

In this case study, one way to protect *Posidonia oceanica* against IAS would be to protect the seagrass meadows themselves. Studies and personal observation have shown that dense, homogeneous *P. oceanica* meadows are not invaded by IAS (in the present study by *C. cylindracea* and *H. stipulacea*). Furthermore *C. cylindracea* seems to prefer dead *P. oceanica* matte for aggressive growth (Klein & Verlaque, 2008). In 2010, Katsanevakis et al. found that the highest frond densities were observed on the dead matte and rocky habitat types, indicating their high vulnerability to colonization. Dead matte is seen as a degraded habitat (Borg, 2006), and can be created by anthropogenic impacts, for instance boat-anchoring (Francour et al., 1999). The area invaded could also impact diving tourism. Indeed the large homogenization of the bottom by the *C. cylindracea* could decrease the biodiversity and richness of some sites and by consequence, their economic value (Zuljevic, unpublished data).

Preventive action concerning boat anchoring, boat-trawling, eutrophication, urbanization, aquaculture farming, and fishing gear impacts needs to be taken in order to protect these meadows. The creation of a Marine Protected Area seems to be a good and realistic option in order to ensure the long-term protection of the fauna and flora of the area (Borum et al., 2004, Guala et al., 2012). However, the creation of a MPA could be implemented only with the local stakeholder's acceptance, with an understanding of the long term benefits of a MPA around Lipsi.

Regarding the local protection of the *Posidonia oceanica*, measures need to be taken. Over-exploitation and coastal development represent both immediate and future threats to the meadows. Also as it has been said before, the value of the service provided by the seagrass meadows is estimated at 172€/m². Artificial compensation would certainly be very expensive and would not be as efficient as natural *P. oceanica*. The loss of the *P. oceanica* meadows would definitely be dramatic for the fauna and flora of Lipsi's waters. Moreover they host numerous commercial species important for the local fishermen and restaurants. The loss of these meadows could seriously impact the socio-economic situation of the Island as the sea resources are vital for Lispi. So far the Island is still a relatively pristine place, removed from Greek tourism hotspots. However, the economy is growing and construction is expanding on the Island. Some measures can be taken following guidelines already made in order to protect *P. oceanica* (Boudouresque et al., 2006):

- Prohibition of mooring in the *P. oceanica*.
- Prohibition of coastal construction.
- Limiting sediment run-off.
- Prohibition of trawling in the *P. oceanica*.
- Installation of seagrass-friendly mooring.
- Managing stranded seagrass litter.
- Prohibition of sewage disposal directly to the sea.

Monitoring the *P. oceanica* can help to better understand the situation. The seagrass meadow area, species composition, and canopy height, can be different factors that can give an idea of the health of local meadows, boat frequentation can be also assessed (Guala et al., 2012) Increased public awareness is also an important step for a better protection.

Unfortunately, despite the high ecological / societal / economic value of the seagrasses, as one part of the social–ecological system, knowledge of and attention to these ecosystems are still insufficient (Duarte, 2002, Seitz et al., 2013). Seagrasses are important ecosystem service providers, but are often disregarded or missing from the global conservation agenda (Cullen-Unsworth et al., 2013). Scientific knowledge is one of the main priorities for efficient seagrass protection Duarte (2002) illustrated the interconnecting ways to protect seagrasses and by consequence decrease IAS spread in *Posidonia oceanica* (fig. 21).

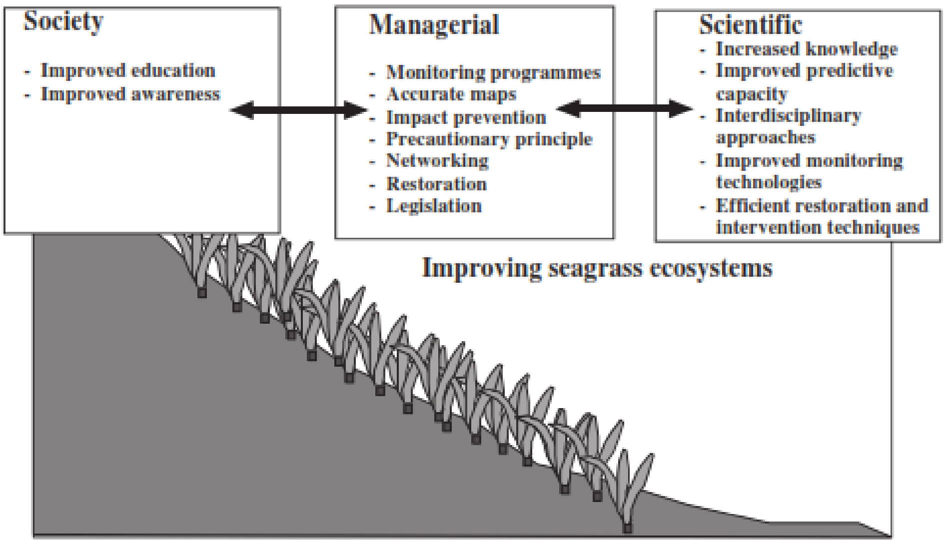


Figure 21. Graphic of the cooperative elements required to prevent present trend towards seagrass decline and efficiently converse seagrass ecosystems (Duarte, 2002)

6.1 Proposed management guidelines

As a socio-ecological system, the management of Lipsi Island local community and its seagrasses ecosystem should be thought of as a single and unique system, which is under a threat. This is to say that the properties and processes of the community and surrounding environments are linked in a way that greatly influences the well-being or the degradation of the other. The most common understanding of this is the influence that the presence of extractable resources (ex. fish for eating) from the natural system has on the dependent social system (Ash et al, 2010). As explained by Folke, et al (2010), the dynamics of these linked systems are often determined by feedback loops that operate between them.

With this, resource and ecosystem conservation requires insight from multiple perspectives – both the human and natural dimensions– for the problem to be appropriately assessed and addressed. The integration of both realms allows for a more holistic understanding and allows for the introduction of a well-researched conservation strategy (Valdés-Pizzini et al., 2012). Adopting and implementing all the recommendations said above could be done following an ecosystem-based approach to fishery management. It could be a solution to decrease and then stop the spread of the IAS and to protect fish assemblage, the habitats, and the Lipsi community. Indeed this approach tries to speak to the multitude of needs and desires of societies, while at the same time ensuring future generations the opportunity to share in and benefit from all goods and services provided by the natural coastline and associated habitats (Valdés-Pizzini et al., 2012). Even if this management strategy has been implemented mostly in large areas (Grumbine, 1994) and the fact that it is encompassing an array of interactions, which often make it looks as a complex process, difficult to implement, this would make sense to be implemented on Lipsi Island, and monitor by a local institute or local NGO like Archipelagos. Ecosystem-based management is normally used at the "bioregions" scale as management units (Aberley, 1993), but as this management strategy recognizes the full array of interactions within an ecosystem, including humans (Christensen et al. 1996), a small scale implementation on Lipsi Island as one socio-ecological system could be tested, framed and monitored by Archipelagos NGO with the support of local Greek authorities. This could allow testing this management system at a local scale directly in connexion with local regarding fisheries and IAS monitoring and action measures. Moreover this small-scale management would permit to be quickly flexible to make the adjustment needed in short term and be reactive to take action against IAS.

It advances the participation of the local community and stakeholders, who are concerned about fish stock and sea protection. It also advances the creation of Marine Protected Area, and currently Archipelagos is proposing a project involving MPA creation around Lipsi Island. Moreover it considers the effects of anthropogenic activities on habitat quality, the use of marine protected areas based on habitat characteristics and the effects of habitat availability on fishery yield (Seitz et al., 2013). These three factors align with Lipsi Island case. This management strategy would link the biological sustainable objectives with the local community wishes and needs. Furthermore, Tallis et al. (2010) suggest setting thresholds for each indicator and setting targets that would represent a desired level of health for the ecosystem. Examples from Lispi Island may include species composition within an ecosystem or IAS percentage cover over the time on local observations. These indicators and thresholds could be managed and assess by local community and volunteers frame by Archipelagos NGO.

Beyond, *P. oceanica* and IAS threat, there is a need for protection of the entire coastal habitat to ensure the protection of the social-economic features of Lipsi community and its human well-being. A lot of species rely on different coastal habitats to provide all necessary components of their life cycle. With that, the habitat quality and connectivity of all processes within it are to be viewed as essential characteristics of coastal ecosystems that need to be protected (Lipcius et al., 2008). Future fishery management efforts cannot solely address the goals of maintaining fish stocks, but must also see to habitat reconstruction, to IAS monitoring and human concerns, as they are environmentally critical for the fauna and flora, and financially important for the local population through the ecological services provided, as on Lipsi Island, where habitats and human well-being seem to be already damaged. With this engagement of ecosystem-based management coastal areas will see a return of fish and invertebrate populations, resulting from a holistic management approach (Seitz et al., 2013). Furthermore, by viewing an ecosystem at this larger scale, fisheries managers will be able to take into consideration and understand larger transformations and changes in the ecosystems, which alter the availability of varied species and the structure of the trophic web as a whole (Valdés-Pizzini et al., 2012).

In order to implement this management, there are several stages of ecosystem-based management that need to be present to be considered an effective management strategy. First, some form or historical data regarding the area under question would need to be incorporated to understand patterns and processes. This data is still missing on Lipsi Island, but Archipelagos could create and monitor this data on a long term strategy. Next, it should be a democratic and participatory process, speaking to multiple parties and stakeholders to ensure the use of 'best available knowledge' (both science and local forms of knowledge). Indeed as McConney & Salas (2011) explain, its success would depend mostly on the incorporation of the social dimensions. This part would fit for Lipsi Island, as it is a small island with a small population, after ensuring the trust of local stakeholders and local community, Archipelagos would easily organize meeting to inform person and list the desires and needs of the local population. This will ensure a holistic approach (encompassing all behavioural dimensions: biological, economic, social, cultural, governance), which has already been discussed as a method by which larger scale concerns are understood and addressed. Finally, the system should engage an adaptive and flexible strategy as Lipsi Island hosts artisanal fishermen that modify their fishing behaviour quickly and in short-term. This strategy would enable managers (and participants in the fishery) to make adjustments and develop new strategies for maintaining sustainability in real time (Valdés-Pizzini et al., 2012). The first signs of the Lipsi's underwater resources degradation have started to be seen. Actions against IAS spread need to be taken now, otherwise negative impact on fish community will certainly start to be recorded soon. However, as it has been said, Lipsi community and its surrounding underwater resources (fauna and flora) should be seen as an unique system, that is why the management of IAS should to be encompassed in global management plan for Lipsi.

6.2 Future Studies

This present study can be seen as a preliminary work about IAS and seagrass in the shallow water of Lipsi Island. This work pushes future researches to be carried out in order to better understand the different fields covered by the study. Regarding the IAS and *P. oceanica*, work should be done to help identify potential percentage thresholds of invasion of IAS, and damage of *P. oceanica* that would likely impact fish assemblages and therefore local fisheries. This work would be more robust and relevant if the data collection is done in the area of active fishing in order to be able to assess the impact of IAS on the fish assemblage directly targeted by the local fishermen. Future studies should have appropriate diving equipment in order to not be limited by this.

Future research is also needed on the effectiveness of management strategy implemented on Lipsi Island; indeed ecosystem-based management is not the only management strategy available, and perhaps experience will show that another management plan would be better suited to the case of Lipsi. These results could allow managers and policy makers to stay flexible and choose the best strategy for their area. As this study suggests, habitat connectivity is also a field that needs further study. Fishes move between the different habitats (seagrass, rock, band sand) in the shallow water, and researches need to show a better understanding of the interdependence among these habitats and the importance of each of them. The present study aims to understand the social–ecological system, created by the seagrass habitat and the local community through the sustainable use of fishery resources and other indirect benefits. Future studies should look at this system, to better understand the relationships between each member of this system. These studies could help to better understand and assess the socio-ecological resilience of these interdependent systems (Folke et al., 2010) and could help to foster resilience in a new desired development trajectory.

7 Conclusion

The present study on the fish community around Lipsi Island supports the findings of similar studies undertaken in the region regarding the composition of fish community around the Island and the impact of low percentage IAS cover on them. This study suggests that a low percentage of IAS does not significantly impact the fish community around Lispi Island, but numerous limitations had to be taken into account in the analyses of the results. The literature review showed the importance of *Posidonia oceanica* in the Mediterranean Sea and the negative potential of the IAS. This study can be seen as a baseline and first preliminary assessment of the resources in shallow waters of Lipsi. This work could be used by Archipelagos in order to create a new management plan for Lipsi Island and the creation of an MPA. Scientists could also use the data about *P. oceanica*, IAS, fish assemblages and landings to compare temporal change over years. This study has also shown the importance of preliminary works, before a study of this kind. In this case, IAS cover importance should have been checked before the start of the data collection to ensure its real cover importance. Furthermore, diving equipment and volunteer skills should have been checked before as well in order to diminish the limitation and bias of this data collection.

Two flora invasive alien species were recorded around Lispi Island. Even if this study had not shown any impacts on fish assemblage, action to limit the spread of IAS and to protect *P. oceanica* meadows should be implemented. Local NGOs like Archipelagos and public authorities could organize regular manual removal with local fishers or SCUBA diver clubs. However, in the case study of Lipsi Island, SCUBA diving equipment and skilled persons are lacking for large manual removal operation. It seems to be better and more realistic to prevent further degradation of *P. oceanica* and impede the spread of IAS. Increased public awareness about this threat could be implemented with locals and tourists in order to explain the impacts of bad behaviour and map IAS locations. Monitoring programmes should be undertaken to follow and prevent damage caused by IAS. Protection and preservation projects need the involvement of local and regional stakeholders and the continuance of scientific work in order to increase the local and regional body of knowledge. The human and societal dimensions in management still need to be encouraged in order to ensure the efficiency of sustainable development.

At the Mediterranean and global scales, data about the rise and impacts of IAS and the decrease of *P. oceanica* meadows need to be gathered. They directly impact the economic and therefore social features and human well-being. Scientific studies are the base of effective policy decisions and data about social and economic impacts of IAS are still lacking.

Finally, an ecosystem-based management programme could be undertaken at the local scale mixed as with MPAs or restricted fishery areas, this management strategy known as holistic and community-based could be worthy to be tested on Lipsi Island. A numerous limitations could make this management plan hard to implement, but with the continuous work of Archipelagos NGO and local community support, this plan could be very efficient and ensure Lipsi sustainable development. However, it would only protect the small area of the island. As noted before, management projects could be highly effective if they are undertaken at the global scale, following ecosystem-based management. In this case of larger scale, government and public administration need to take over and give the resource needed for the implementation of this kind of plan. Fishery / ecological scientists and social scientists have to come together due to the complexity of the task, by thinking innovatively of a global socio-ecological management framework. The advantages and benefits of adaptive, socially grounded forms of ecosystem management and planning are now widely accepted. Understanding of circumstantial and site by site analysis is acknowledged as one of the most efficient methods by which the connection, issues and concerns between people and the environment around them can be understood.

Future researches should be done on the effectiveness of different management strategies, on environmental processes and events, the habitat connectivity and social–ecological system seagrass connectivity. Furthermore work on the artisanal fishery, its advantages and disadvantages for the sustainable use of fishery resources and social consideration should be done as the Greek fishing activity is mainly based on small-scale fishery and this kind of fisheries is, since recently, seen as much more sustainable than industrial fishery. It is also seen as much more respectful to the local coastal community.

To conclude, the ocean is one ecosystem and each part of this complex system needs to be protected if we want to ensure the protection of the hand that feeds us.

References

Aberley D., 1993. Boundaries of Home: mapping for local empowerment. Philadelphia, PA: New Society Publications; 1 – 7 pp.

Adger, W.N., 2000. Social and ecological resilience: are they related''. Programme Human Geography 24, 347 – 364 pp.

Archipelagos, unpublished report, no date. Previous regional fisheries management in Lipsi island, Dodecanese, Greece.

Archipelagos, unpublished data, 2012. Flora of the Aegean Sea updated August 2012. Interne document for the formation of the volunteer for the underwater flora identification.

Ash N., Blanco H., Brown C., Garcia K., Henrichs T., Lucas N., Raudsepp-Hearne C., Simpson R.D., R. S., Tomrich T.P., Vira B., and Zurek M., 2010. Ecosystems and Human Well-being. Washington: Island Press.

Azzurro, E., Fanelli, E., Andaloro, F., 2004. Preliminary data on feeding habits of dusky spinefoot *Siganus luridus* in the Sicily Channel (central Mediterranean). Biologia Marina Mediterranea 11 (3), 145 pp.

Baccini A. & Gonzalez I., 2006, An extension of Canonical Correlation Analysis to study of the relationship between gene expression data and biological variables. Online support. Searched on 16 May, 2014. http://www.math.univ-toulouse.fr/~igonzal/Enc_France_Vzla.pdf

Belton G.S., Prud'homme van Reine W., Huisman J.M., Draisma S.G.A., Gurgel C.F.D. Resolving phenotypic plasticity and species designation in the morphologically challenging *Caulerpa racemosa* – peltata complex (Chlorophyta, Caulerpaceae). Journal of Phycology 50, 32 - 54 pp.

Bernard A.T.F., Götz A., 2011. Development of a volunteer diver based marine monitoring programme for the Table Mountain National Park. Technical Report prepared for WWF South Africa, Stellenbosch, South Africa.

Bertelli C.M & Unsworsth R.K.F., 2013. Protecting the hand that feeds us: Seagrass (*Zostera marina*) serves as commercial juvenile fish habitat. Marine Pollution Bulletin.

Bonacorsi M., Pergent-Martini C., Breand N., Pergent G., 2013. Is *Posidonia oceanica* regression a general feature in the Mediterranean Sea? Mediterranean Marine Science 14 (1), 193 - 203 pp.

Borg J.A., Rowden A.A., Attrill M.J., Schembri P.J., Jones M.B., 2006, Wanted dead or alive: high diversity of macroinvertebrates associated with living and 'dead'' *Posidonia oceanica* matte. Marine Biology 146, 667 – 677 pp.

Borum J., Duarte C.M., Krause-Jensen D. & Greve T.M., 2004, European seagrasses: an introduction to monitoring and management. European Commission, M&Ms project.

Boudouresque C.F., Meinesz A., 1982. Découverte de l'herbier de Posidonie. Cahier Parc National Port-Cros, France, 4 : 1 - 81 pp.

Boudouresque C.F., Bernard G., Bonhomme P., Charbonnel E., Diviacco G., Meinesz A., Pergent G., Pergent-Martini C., Ruitton S., Tunesi L. 2006. Préservation et conservation des herbiers à *Posidonia oceanica*. Ramoge publication.

Carriglio D., Sandulli R., Deastis S., Gallo d''Addabbo M., Grimaldi de Zio S., 2003. *Caulerpa racemosa* spread effects on the meiofauna of the Gulf of Taranto. Biologia Marina Mediterranea 10 (2), 509 – 511 pp.

CBD, Convention of Biological Diversity, 2009. What are Invasives Alien Species. CBD website; searched on 13 May, 2014. http://www.cbd.int/idb/2009/about/what/

Ceccherelli, G., Piazzi, L., Cinelli, F., 2000. Response of the non-indigenous Caulerpa racemosa (Forsskål) J. Agardh to the native seagrass Posidonia oceanica (L.) Delile: effect of density of shoots and orientation of edges of meadows. Journal of Experimental Marine Biology and Ecology 243, 227 – 240 pp.

Christensen N. L., Bartuska A., Brown J.H., Carpenter S., D'Antonio C., Francis R., Franklin J.F., MacMahon J.A., Noss R.F., Parsons D.F., Peterson C.H., Turner M.G., and Moodmansee R.G., 1996. The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecological Applications 6 ; 665 – 691 pp.

CIESM, 2012. CIESM Tropical Signals Programme: The Fish Census Protocol.

Clarke K.R. and Warwick R.M. 1994. *Changes in marine communities: an approach to statistical analysis and interpretation.* Plymouth Marine Laboratory, Plymouth: National Environmental Research Council.

Costanza R, D.Arge D, De Groot R., Farber S., Grasso M., Hannon B., Limburg K, Naeem S., O'Neill R.V., Paruelo J., Raskin R.G., Sutton P. & Van Den Belt M. 1997. The value of the world''s ecosystem services and natural capital. Nature 387, 253 – 260 pp.

Cullen-Unsworth L.C., Mtwana Nordlund L., Paddock J., Baker S., McKenzie L.J., Unsworth R.K.F., 2013. Seagrass meadows globally as a coupled social–ecological system: Implications for human well-being. Marine Pollution Bulletin.

De Biasi A.M., Gai F. & Giaccone G., 1995. La dispersione in Mediterraneo di alghe tropicali del genere Caulerpa. Bolletino delle sedute dell'' Accademia Gioenia di Scienze Naturali 28 (349), 693 – 705 pp.

Deudero S., Box A., Alós J., Arroyo N.L., Marbà N., 2011. Functional changes due to invasive species: Food web shifts at shallow *Posidonia oceanica* seagrass beds colonized by the alien macroalga *Caulerpa racemosa*. Estuarine, Coastal and Shelf Science 93, 106 – 116 pp.

Díaz-Almela E. & Duarte C.M., 2008. Management of Natura 2000 habitats. 1120, Posidonia beds (Posidonion oceanicae). European Commission.

Di Martino V., Stancanelli B., Molinari A., 2007. Fish community associated with *Halophila stipulacea* meadow in the Mediterranean Sea. Cybium 31 (4), 451 – 458 pp.

Dolnicar S., 2002. A Review of Unquestioned Standards in Using Cluster Analysis for Data-driven Market Segmentation. Australian and New Zealand Marketing Academy Conference 2002 (ANZMAC 2002), Deakin University, Melbourne.

Duarte C.M., 2002. The future of seagrass meadows. Environmental Conservation 29, 192 - 206 pp.

Dumay O., Fernandez C., Pergent C., 2002. Primary production and vegetative cycle in *Posidonia Oceanica* when in competition with the green algae *Caulerpa taxifolia* and *Caulerpa racemosa*. Marine Biological Association UK 82, 379 - 387 pp.

European Commission, 2013. Website of the European Maritime Affairs. Article on the Mediterranean Sea Basin. Searched on 27 February, 2014. http://ec.europa.eu/maritimeaffairs/ policy/sea_basins/mediterranean_sea/index_en.htm

Fabio H.V. & Hazin D., 2005. Fisheries and their contribution to sustainable development: Small scale and artisanal fisheries. The Sixth Meeting of the United Nations Open-ended Informal Consultative Process on Oceans and the Law of the Sea. Online support. Searched on 19 May, 2014. http://www.un.org/Depts/los/consultative_process/documents/6_hazin.pdf

FAO, 2014. Fisheries and Aquaculture topics. Small-scale and artisanal fisheries. Topics Fact Sheets. Text by Jan Johnson. In: FAO Fisheries and Aquaculture Department [online]. Rome. Updated 27 May 2005. Searched on 3 February, 2014. http://www.fao.org/fishery /topic/14753/en

Folke C., Carpenter S.R., Walker B., Scheffer M., Chapin, T. and Rockstrom J., 2010. Resilience Thinking: Integrating Resilience, Adaptability and Transformability. *Ecology and Society* 15 (4), p. 20.

Francour P., Ganteaume A., Poulain M., 1999. Effects of boat anchoring in *Posidonia oceanica* seagrass beds in the Port-Cros National park (North-Western Mediterranean Sea). Aquatic conservation: Marine Freshwater Ecosystem 9, 391 – 400 pp.

Galil B.S., 2006a, Fact Sheet: *Caulerpa racemosa* var. *cylindracea.* European Commission. Delivering Alien Invasive Species In Europe (DAISIE) project.

Galil B.S., 2006b, Fact Sheet: *Halophila stipulacea*. European Commission. Delivering Alien Invasive Species In Europe (DAISIE) project.

Galil B.S., 2006c, Fact Sheet: *Siganus rivulatus*. European Commission. Delivering Alien Invasive Species In Europe (DAISIE) project.

Garibaldi L. & Limongelli L., 2003. Trends in oceanic captures and clustering of large marine ecosystems: Two studies based on the FAO capture database. FAO Fisheries Technical Paper 435. Food and Agriculture Organization of the United Nations. Rome.

Gaten T., 2000. On-line statistic, Analysis of Variance (ANOVA). Department of Biology, University of Leicester. Searched on 16 May, 2014. http://www.le.ac.uk/bl/gat/virtualfc/Stats/anova.html

Gauch H.G.Jr., 1982. Multivariate Analysis and Community Structure. Cambridge University Press, Cambridge.

Graphicmaps.com, no date. Searched on 27 February, 2014. http://www.worldatlas.com/aatlas/infopage/medsea.gif

Gratwicke B. and Speight M.R, 2004. The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of Fish Biology 66, 650 - 667 pp.

GBRMPA, 1978. Great Barrier Reef Marine Park Authority Workshop on reef fish assessment and monitoring, Workshop Series 1, heron Island, Australia, 64 pp.

Giakoumi S. and Kokkoris G.D., 2013. Effects of habitat and substrate complexity on shallow sublittoral fish assemblages in the Cyclades Archipelago, North-eastern Mediterranean sea. Mediterranean Marine Science 14 (1), 58 - 68 pp.

Grumbine R. E., 1994. What is Ecosystem Management? Conservation Biology. 8; 27 – 38 pp.

Guala I., Di Carlo G., Jakl Z., et al. 2012. Monitoring of *Posidonia oceanica* meadows in Croatian Protected Areas. Association Sunce Technical Report, January 2012, 40 pp.

Guidetti P., 2000. Differences Among Fish Assemblages Associated with Nearshore *Posidonia oceanica* Seagrass Beds, Rocky–algal Reefs and Unvegetated Sand Habitats in the Adriatic Sea. Estuarine, Coastal & Shelf Science 109, 1 - 10 pp.

Guidetti P., Bussotti S., Conti M., 1998. Fish fauna of the Genoa-Quinto *Posidonia oceanica* bed (Ligurian sea, north – western Mediterranean). Rapport de la Commission internationale sur la Mer Méditerranée 35, 546 – 547 pp.

Guiry, M.D. & Guiry, G.M. 2014. AlgaeBase. World-wide electronic publication, National University of Ireland, Galway. http://www.algaebase.org; searched on 17 January, 2014.

Jackson, E.L., Rowden, A.A., Attrill, M.J., Bossey, S., Jones, M., 2001. The importance of seagrass beds as a habitat for fishery species. Oceanography Marine Biology 39, 269 - 304 pp.

Jind S., 2012. A comparison of two underwater visual sampling techniques used to estimate tropical reef fish communities. Unpublished bachelor dissertation. Dalhousie University.

Jones C.G., Lawton J.H. and Shachak M., 1994. Organisms as ecosystem engineers. Oikos 69, 373 - 386 pp.

Jorda G., Marba N., Duarte C.M., 2012. Mediterranean seagrass vulnerable to regional climate warming. Natural Climate change 2. 821 – 824 pp.

Kalogirou S., Corsini-Foka M., Sioulas A., Wennhage H. & Pihl L., 2010. Diversity, structure and function of fish assemblages associated with *Posidonia oceanica* meadows in Eastern Mediterranean and the role of non-indigenous species. Journal of fish biology 77, 2338 - 2357 pp.

Katsanevakis S.,Issaris Y., Poursanidis D., Thessalou-Legaki M., 2010. Vulnerability of marine habitats to the invasive green alga *Caulerpa racemosa var cylindracea* within a marine protected area. Marine Environmental Research 70, 210 – 218 pp.

Kaufman L. & Rousseeuw P.J., 1990. Finding Groups in Data: an Introduction to Cluster Analysis, John Wiley and Sons.

Klein J. & Verlaque M. 2008. The *Caulerpa racemosa* invasion - A critical review. Marine Pollution Bulletin 56, 205-225 pp.

Halford A. R. & Thompson A. A., 1994. Visual Census Surveys of Reef Fish, Long-term Monitoring of the Great Barrier Reef. Standard Operational Procedure Number 3. Australian Institute of Marine Science.

Harmelin-Vivien, M. L. 1982 Ichtyofaune des herbiers de Posidonies du Parc National de Port-Cros: I. Composition et variations spatio-temporelles. Travaux scientifics du Parc national de Port Cros 8, 69 – 92 pp.

Harmelin-Vivien M. L. & Francour P. 1992. Trawling or visual censuses? Methodological bias in the assessment of fish populations in seagrass beds. P.S.Z.N.I: Marine Ecology 13 (1), 41 - 51 pp.

Harmelin-Vivien, M. L., Harmelin, J. G., Chauvet, C., Duval, C., Galzin, R., Lejeune, P., Barnabe, G., Blanc, F., Chevalier, R., Duclerc, J. & Lasserre, G. 1985. Evaluation des peuplements et populations de poissons. Méthodes et problemes. Rev. Ecol. (Terre Vie) Vol.40 : 467 - 539 pp.

Hellenic Statistical Authority, 2012. Announcement of the results of the 2011 Population Census for the Resident Population. http://www.statistics.gr/portal/page/portal/ESYE/BUCKET/A1602/PressReleases/A1602_SAM01_DT_DC_00_2011_02_F_EN.pdf

Hemminga M. A. & Duarte C. M. 2000. Seagrass Ecology. Cambridge University Press, Cambridge.

Howell, David (2002). Statistical Methods for Psychology. Duxbury.324 – 325 pp.

Hulme P.E., Pysek P., Nentwig W., and Vila M., 2010. Will Threat of Biological Invasions Unite the European Union? Science, 4 - 5 pp.

Kirkman H., 1990. Seagrass distribution and mapping. In: Seagrass Research Methods - Monographs on oceanographic methodology. Eds R C Phillips and C P McRoy. pp 19 - 25. UNESCO, Paris

La Mesa G., Molinari A., Gambaccini S. & Tunesi L. 2010. Spatial pattern of coastal fish assemblages in different habitats in North-western Mediterranean. P.S.Z.N.I: Marine Ecology 32 (1), 104 - 114 pp.

Leopold M., Cakacaka A., Meo S., Sikolia J., & Lecchini, D. 2009. Evaluation of the effectiveness of three underwater reef fish monitoring methods in Fiji. Biodiversity and Conservation 18 (13), 3367 - 3382 pp.

Lilley R.J., 2013. Implications for the future of fisheries as extractive industries in the creation of sustainable places. A cross-sectional study of nearshore seagrass meadows in The Turks and Caicos Islands. Unpublished doctoral dissertation. Cardiff Business School - Cardiff University.

Lipcius R.N., Eggleston D.B., Schreiber S.J., Seitz R.D., Shen J., Sisson M., Stockhausen W.T., et al., 2008. Importance of metapopulation connectivity to restocking and restoration of marine species. Reviews in Fisheries Science 16, 101 – 110 pp.

McConney P. and Salas S. 2011. "Why Incorporate Social Considerations into Marine EBM?" In Towards Marine Ecosystem-Based Management on the Wider Caribbean, edited by L. Fanning et al. Amsterdam University Press.

McKenzie L.J., Campbell S.J., and Roder C.A. 2003. Seagrass-Watch: Manual for Mapping & Monitoring Seagrass Resources by Community (citizen) volunteers Cairns: QFS, NFC.

Moutopoulos, D.K., Stergiou, K.I., 2002. Length-weight and length-length relationships of fish species from the Aegean Sea (Greece). Journal of Applied Ichthyology 18 (3), 200 - 203 pp.

NCSS Statistical Software, 2012. Chapter 210 One-Way Analysis of Variance. Online support. Searched on 15 May, 2014. http://www.ncss.com/wp-content/themes/ncss/pdf/Procedures/NCSS/One-Way_Analysis_of_Variance.pdf

NOAA, National Oceanic and Atmospheric Administration, no date, Beaufort Wind scale, Strom Prediction Center / NOAA / National Weather Service website; searched on 27 January, 2014. http://www.spc.noaa.gov/faq/tornado/beaufort.html

Olszewski T., 2007. Cluster Analysis class, PBDB Intensive Summer Course 2007, Paleoecology Section. Online support, Search on 16 May, 2014. http://paleodb.org/public/summercourse07/ olszewski_clustering.pdf

Orth R.J., Carruthers T.J.B., Dennison W.C., Duarte C.M., Fourqurean J.W., Heck Jr K.L., Randall A.H., Kendrick G.A., Kenworthy W.J., Olyarnik S., Short F.T., Waycott M., Williams S.L., 2006. A global crisis for seagrass ecosystems. Bioscience 56, 978 – 996 pp.

Pasqualini V., Pergent-Martini C., Clabaut P. & Pergent G. 1998. Mapping of *Posidonia oceanica* using aerial photographs and side scan sonar : application off the Island of Corsica (France). Estuarine, Coastal and Shelf Science 47, 359 - 367 pp.

Pauly, D, Watson R., and Alder J., 2005. Global trends in world fisheries: impacts on marine ecosystems and food security. Philosophical Transactions of the Royal Society B360, 5 – 12 pp.

Piazzi L., Ceccherelli G., Cinelli F., 2001. Threat to macroalgal diversity: the effects of the introduced green alga *Caulerpa racemosa* in the Mediterranean. Marine Ecology Progress Series 210, 149 – 159 pp.

Piazzi L., Meinesz A., Verlaque M., Akcali B., Antolic B., Argyrou M., Balata D., Ballesteros E., Calvo S., Cinelli F., Cirik S., Cossu A., D''Archino R., Djellouli A S., Javel F., Lanfranco E., Mifsud C., Pala D., Panayotidis P., Peijrano A., Pergent G., Petrocello A., Ruitton S., Zuljevic A., Ceccherelli G., 2005. Invasion of *Caulerpa Racemosa var. cylindracea* (Caulerpales, Chlorophyta) in Mediterranean Sea - Assessment of the spread. Cryptogamie, Algologie 26, 189 - 202 pp.

Potschin M. and Haines-Young R., 2012. Landscapes, sustainability and the place-based analysis of ecosystem services. *Landscape Ecology* 28 (6), 1053 – 1065 pp.

Procaccini G., Acunto S., Fama P., Maltagliati F., 1999. Structural, Morphological and genetic variability in *Halophila stipulacea* (Hydrocharitaceae) populations in the western Mediterranean. Marine Biology 135, 181 – 189 pp.

Raniello R., Mollo E., Lorenti M., Gavagnin M., Buia M.C., 2007. Phytotoxic activity of caulerpenyne from the Mediterranean invasive variety of *Caulerpa racemosa*: a potential allelochemical. Biological Invasions 9, 361 – 368 pp.

Relini G., Relini M., Torchia G., 1998 Fish biodiversity in a *Caulerpa taxifolia* meadow in the Ligurian Sea. Italian Journal of Zoology 65 (1), 465 – 470 pp.

Rilov G, Galil B (2009) Marine Bio-invasions in the Mediterranean Sea – History, Distribution and Ecology. In: Rilov G, Crooks JA (eds) Biological Invasions in Marine Ecosystems. Springer, Berlin Heidelberg, 549 – 575 pp.

Rindi, F.; Maltagliati, F.; Rossi, F.; Acunto, S.; Cinelli, F., 1999. Algal flora associated with a *Halophila stipulacea* (Forsskal) Ascherson (Hydrocharitaceae, Helobiae) stand in the western Mediterranean. Oceanologica Acta 22 (4), 421 - 429 pp.

Ruitton S., Verlaque M., Boudouresque C.F., 2005. Seasonal changes of the introduced *Caulerpa racemosa var. cylindracea* (Caulerpales, Chlorophyta) at the northwest limit of its Mediterranean range. Aquatic Botany 82, 55 - 70 pp.

Ruitton S., Verlaque M., Aubin G., Boudouresque C.F., 2006. Grazing *on Caulerpa racemosa* var. *cylindracea* (Caulerpales, Chlorophyta) in the Mediterranean Sea by herbivorous fish and sea urchins. Vie et Milieu 56 (1), 33 – 41 pp.

Ruiz J.M., Marín-Guirao L., Bernardeau–Esteller J., Ramos–Segura A., García–Muñoz R. & Sandoval–Gil J.M., 2011. Spread of invasive species *Caulerpa racemosa* along Murca region SPAIN. Animal Biodiversity and Conservation 34 (1), 73 - 82 pp.

Salas, S., U.R. Sumaila, and T. Pitcher. 2004. Short-term decisions of small-scale fishers selecting alternative target species: a choice model. Canadian Journal of Fisheries & Aquatic Sciences. Vol. 61 (3), 374 – 383 pp.

Savva I., Dawson K., Seary R., Miliou A., 2013. Identifying Small Scale Fisheries at Lipsi Island, Dodecanese, Greece and their seasonal production during mid-summer and autumn. Internal Archipelagos works/data. Unpublished report.

Schmeller D. S., Henry, P. Y., Julliard R., Gruber B., Clobert J., Dziock F., et al., 2009. Advantages of volunteer-based biodiversity monitoring in Europe. Conservation Biology 23 (2), 307 - 316 pp.

Sclove S.L, 2001. Statistics for Information Systems and Data Mining, Notes on Cluster Analysis. University of Illinois, College of Business Administration, Department of Information & Decision Sciences. Online support. Searched on 16 May, 2014. http://www.uic.edu/classes/idsc/ids472/clustering.htm

Sea Around Us, 2013. A global database on marine fisheries and ecosystems. Fisheries Centre, University of British Columbia. Searched on 27 February, 2014. http://www.seaaroundus.org/lme/26.aspx

Seitz R.D., Wennhage H., Bergstrom U., Lipcius R.N. & Ysebaert T., 2013. Ecological value of coastal habitats for commercially and ecologically important species. ICES Journal of Marine Science.

Sghaier Y.S., Zakhama-Sraieb R., Benamer I. & Charfi-Cheikhrouha F., 2011. Occurrence of the seagrass *Halophila stipulacea* (Hydrocharitaceae) in the southern Mediterranean Sea. Botanica Marina 54, 575 – 582 pp.

Shepperson, J., Jeffery A., Taylor L., Miliou A., Thornton H.. and Acasuso C. 2013. A preliminary analysis into the relationship between *Posidonia oceanica* morphology and *Caulerpa racemosa*, internal report of Archipelagos. Greece.

Sherman, K. and Alexander, L. M. and Gold, B. D. 1990. Large Marine Ecosystems: Patterns, processes and yields. Washington D.C. American Association for the Advancement of Science.

Shine C., Kettunen M., Genovesi P., Essl F., Gollasch S., Rabitsch W., Scalera R., Starfinger U. and ten Brink P. 2010. Assessment to support continued development of the EU Strategy to combat invasive alien species. Final Report for the European Commission. Institute for European Environmental Policy (IEEP), Brussels, Belgium.

Short F.T., McKenzie L.J., Coles R.G., Vidler K.P., Gaeckle J.L. 2006. SeagrassNet Manual for Scientific Monitoring of Seagrass Habitat, Worldwide edition. University of New Hampshire Publication. 75 pp.

Short, F. T., Beth P., Livingstone S. R., Carpenter K. E., Salomão B., Bujang J. S., Calumpong, H. P., Carruthers T. J.B., Coles R. G., Dennison W. C., Erftemeijer P. L.A., Fortes M. D., Freeman A. S., Jagtap T.G., Kamal A. H. M., Kendrick G. A., Kenworthy W. J., La Nafie Y. A., Nasution I. M., Orth R. J., Prathep A., Sanciangco J. C., van Tussenbroek B., Vergara S.

G., Waycott M. and Zieman J. C., 2011. Extinction risk assessment of the world‘s seagrass species. Biological Conservation 144 (7), 1961 - 1971 pp.

Stergiou K.I., Petrakis G., Politou C-Y., 1996. Small-scale fisheries in the South Euboikos Gulf (Greece) : species composition and gear competition. Fisheries Research 26, 325 - 336 pp.

Streftaris N. & Zenetos A., 2006. Alien Marine Species in the Mediterranean the 100 worst invasive species and their impacts. Mediterranean Marine Science 7 (1), 87 - 118 pp.

Tallis H., Levin P.S., Ruckelshaus M., Lester S.E., McLeod K.L., Fluharty D.L. and Halpern B.S., 2010. The many faces of ecosystem-based management: Making the process work today in real places. Marine Policy 34, 340 – 348 pp.

ter Braak C.J.F. & Verdonschot P.F.M., 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57, 255 – 289 pp.

Terrados, J. and Borum J., 2004. Why are seagrasses important? Goods and services provided by seagrass meadows. European Commision, M&MS project, Copenhagen: 8 – 11 pp.

Tsikliras A., Moutopoulos D. & Stergiou K., 2007. Reconstruction of Greek marine fisheries landings: national versus FAO statistics. Fisheries Centre Research Reports 15, 121 – 137 pp.

Tzanatos, E., Dimitriou, E., Katselis, G., Georgiadis, M., Koutsikopoulos, C., 2005. Composition, temporal dynamics and regional characteristics of smallscale fisheries in Greece. Fisheries Research 73, 147 – 158 pp.

Ulas A., Duzbastilar F.O., Aydin C., Lok A., Metin C., 2011. Determining Density of *Caulerpa racemosa* (Forsskål) J. Agardh and its Effects on Catch Compositions of Fishing Gears. Turkish Journal of Fisheries and Aquatic Sciences 11, 385 - 391 pp.

UNEP, 2003.Treaty on International Trade in GMOs to become Law. Press Release, 13 June. United Nations Environment Programme, Nairobi. http://www.unep.org/Documents. Multilingual/Default.asp?DocumentID=321&ArticleID=4039&l=en

UNEP/MAP, 2012. State of the Mediterranean Marine and Coastal Environment, UNEP/MAP – Barcelona Convention, Athens, 2012.

UNEP/MAP-RAC/SPA. 2009. Report of the MedPosidonia Project-Libya (Project to develop the inventorying, mapping and monitoring of *Posidonia* meadows in four Mediterranean countries: Algeria, Libya, Tunisia and Turkey. RAC/SPA edition, Tunis; 29 pp.

Unsworth R.K.F & Cullen-Unsworth L.C., 2011. Seagrass meadows vital for marine conservation. The oceans 52 Pacific Ecologist, Winter 2011, 51 – 54 pp.

Valdés-Pizzini M., García-Quijano C.G. & Schärer-Umpierre M.T., 2012. Connecting Humans and Ecosystems in Tropical Fisheries: Social Science and Ecosystem-Based Fisheries Management in Puerto Ricoand the Caribbean. *Caribbean Studies* 40, 2, 95 – 128 pp.

Vandeperre F., Higgins R. M., Sánchez-Meca J., Maynou F., Goñi R., Martín-Sosa P., Pérez-Ruzafa A., Afonso P., Bertocci I., Crec'hriou R., D'Anna G., Dimech M., Dorta C., Esparza O., Falcón J. M., Forcada A., Guala I., Le Direach L., Marcos C., Ojeda-Martínez C., Pipitone C., Schembri P. J., Stelzenmüller V., Stobart B. and Santos R. S., 2011, Effects of no-take area size and age of marine protected areas on fisheries yields: a meta-analytical approach. Fish and Fisheries 12, 412 – 426 pp.

Vassallo P., Paoli C., Rovere A., Montefalcone M., Morri C., Bianchi C.N. 2013. The value of the seagrass *Posidonia oceanica*: A natural capital assessment. Marine Pollution Bulletin 75, 157 - 167 pp.

Verlaque, M., Boudouresque, C.F., Meinesz, A., Gravez, V., 2000. The *Caulerpa racemosa* complex (Caulerpales, Ulvophyceae) in the Mediterranean Sea. Botanica Marina 43, 49 – 68 pp.

Verlaque, M., Durand, C., Huisman, J.M., Boudouresque, C.F., Le Parco,Y., 2003. On the identity and origin of the Mediterranean invasive *Caulerpa racemosa* (Caulerpales, Chlorophyta). European Journal of Phycology 38, 325 – 339 pp.

Verlaque, M., Afonso-Carrillo, J., Gil-Rodriguez, M.C., Durand, C.,Boudouresque, C.F., Le Parco, Y., 2004. Blitzkrieg in a marine invasion: *Caulerpa racemosa var. cylindracea* (Bryopsidales, Chlorophyta) reaches the Canary Islands (north-east Atlantic). Biological Invasions 6, 269 – 281 pp.

Waycott, M., Duarte C.M, Carruthers T.J.B, Orth R.J, Dennison W.C, Olyarnik S, Calladinea A, Fourqurean J.W, Heck K.L, Hughese A.R, Kendrick G.A, Kenworthy W.J, Short F.T and Williams S.L. 2009. Accelerating loss of seagrass across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences 106, 12377 – 12381 pp.

Wikipedia, 2013, Kalymnos (regional unit), searched on 27 January, 2014. http://en.wikipedia.org/wiki/Kalymnos_(regional_unit)

Wilkinson C., Green A., Almany J., Dionne S., 2003. Monitoring Coral Reef Marine Protected Areas. A practical guide on how monitoring can support effective management of MPAs. Australian Institute of Marine Science and the IUCN Marine Program.

Zeller D. & Pauly D., 2001. Visualisation of standardized life-history patterns. Fish and Fisheries 2, 344 - 355 pp.

Zenetos A. , Gofas S., Verlaque M., Çinar Me, García Raso E., Azzurro E., Bilecenolu M., Froglia C., Siokou I., Bianchi Cn, Morri C., Sfriso A., San Martin G., Giandgrande A., Kataan T., Ballesteros E., Ramos-Esplá A., Mastrototaro F., Ocaña O., Zingone A., Gambi Mc., Streftaris N., 2010. Alien species in the Mediterranean Sea by 2010. A contribution to the application of European Union’’s Marine Strategy Framework Directive (MSFD). Part I. Spatial distribution. Mediterranean Marine Science 11 (2), 381 - 493 pp.

Zuljevic A., Antolic B., Despalatovic M., Onofri V., 2004. The spread of the invasive variety of *Caulerpa racemosa* in the Adriatic Sea. Rapports de la Commission Internationale pour l’’Exploration Scientifique de la Mer Méditerranée 37, 649 pp.

Zuljevic A., Nikolic V., Onofri V., Srsen P., Tefan A.Å., Kružic P., Fabrio K., Obarcanin A., Petric M., 2007. Eradication of Invasive Alga *Caulerpa racemosa var. cylindracea* in National

Park Mljet (CROATIA). Rapport Commission International Mer Méditerranée, 38 - 39 pp. Vassallo et al., 2013

Zuljevic, unpublished data, cited in Piazzi et al., 2005. Invasion of *Caulerpa Racemosa var. cylindracea* (Caulerpales, Chlorophyta) in Mediterranean Sea - Assessment of the spread. Cryptogamie, Algologie 26, 189 - 202 pp.