



Assessing the impacts of human activities on the fish assemblages from the Ebro Delta coastal lagoons: towards a sustainable management model

Sílvia Rodríguez Climent

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Universitat de Barcelona
Departament d'Ecologia



Centre Sant Carles de la Ràpita
Institut de Recerca i Tecnologia Alimentàries (IRTA)
Unitat d'Ecosistemes Aquàtics



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Avaluació de l'impacte de les activitats humanes en les comunitats de peixos de les llacunes costaneres del Delta de l'Ebre: cap a un model de gestió sostenible

TESI DOCTORAL

Sílvia Rodríguez Climent

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Programa de doctorat: Ecologia Fonamental i Aplicada

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Memòria presentada per

Sílvia Rodríguez Climent

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Vist-i-plau dels directors de la tesi

Dr. Nuno Caiola
Investigador del programa
d'Ecosistemes Aquàtics
IRTA

Dr. Adolf de Sostoa
Professor del Dept. de Biologia Animal
Universitat de Barcelona

Vist-i-plau de la tutora de tesi

Dra. Núria Bonada
Professora associada del Dept. d'Ecologia
Universitat de Barcelona

A la Kireta

Cover photograph / Foto portada:
Arts de pesca secant al sol (Delta de l'Ebre)
Courtesy of / Cortesia de Vanessa Castan Matamoros

*“En su pequeñez, cada semilla contiene
el espíritu del árbol que será después.
Cada semilla sabe cómo transformarse en árbol,
cayendo en tierra fértil,
absorbiendo los jugos que la alimentan,
llenándose de flores y frutos,
para poder dar lo que tiene que dar
[...]
Dentro de nosotros, innumerables sueños
esperan el tiempo de germinar,
echar raíces y darse a luz,
morir cómo semillas...
para convertirse en árboles”*

Jorge Bucay - *Cuentos para pensar* -

AGRAÏMENTS

Diuen que la vida s'assimila a un camí i que durant el seu recorregut innumerables persones s'hi creuaran. Amb algunes d'elles només hi coincidiràs uns instants, d'altres faran llargues etapes contigües al teu itinerari, mentre que una petita minoria caminarà sempre al teu costat. Aquí acaba una etapa del meu camí, la meua llarga etapa fins a acabar la tesi doctoral. Buffff, què puc dir...ha estat una etapa llarga, sí i dura també, però alhora satisfactòria. L'inici, el transcurs i la finalització d'aquesta no hagués estat possible sense tots i cadascun de vosaltres, a tots i totes **MOLTES GRÀCIES!!!**

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ABBREVIATIONS & ACRONYMS

ANOVA	Analysis of variance
CCA	Canonical Correspondence Analysis
CPUE	Capture Per Unit of Effort
DDT	Dichlorodiphenyltrichloroethane
EMP	Eel Management Plan
ENSO	El Niño Southern Oscillation
EPA	Environmental Protection Agency
ERSEM	European Regional Seas Ecosystem Model
EwE	Ecopath with Ecosim
FAO	Food and Agriculture Organization of the United Nations
GAM	Generalized Additive Models
IDEC	Infraestructura de Dades Espacials de Catalunya
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
MLS	Minimum Landing Size
MSE	Management Basis Evaluation
NPZD	Nutrient-Phytoplankton-Zooplankton-Detritus
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyl
SCI	Site of Community Importance
SEAPODYM	Spatial Ecosystem And Population Dynamics Model
SELECT	Share Each Length's Catch Total
SPA	Special Protection Area
SSEM	Shallow-Seas Ecological Model
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
WCED	World Commission on Environment and Development
WFD	Water Framework Directive
WFF	World Forum of Fish Harvesters & Fish Workers

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DIRECTOR'S REPORT

1

Report of the directors of the Ph.D. thesis in reference to its derived publications and the student's contribution to them

Dr. Nuno Caiola, Researcher at the Institute of Research and Technology, Food and Agriculture (IRTA), Sant Carles de la Ràpita, Catalonia, Spain, as supervisor and,

Dr. Adolf de Sostoa Fernández, Professor of the Department of Animal Biology (University of Barcelona), as co-supervisor of the Ph.D. thesis authored by *Silvia Rodríguez Climent* and entitled *Assessing the impacts of human activities on the fish assemblages from the Ebro Delta coastal lagoons: towards a sustainable management model*

INFORM

That the results and conclusions achieved in the research developed by *Silvia Rodríguez Climent* as part of her Ph.D. thesis have been organized in 3 chapters which correspond to 3 scientific papers (2 already published in SCI journals and 1 manuscript to be submitted). The list of publications and manuscripts is shown, indicating the journal impact factor (IF: according to SCI of ISI Web of Knowledge, Journal Citation Report-2012) as well as the median impact factor of the main subject categories and the position of the journal within the corresponding category.

1) *Rodríguez-Climent S, Alcaraz C, Caiola N, Ibáñez C, Nebra A, Muñoz-Camarillo G, Casals F, Vinyoles D, De Sostoa A. Gillnet selectivity in the Ebro Delta coastal lagoons and its implication for the fishery management of the sand smelt,*

***Atherina boyeri* (Actinopterygii: Atherinidae).** *Estuarine, Coastal and Shelf Science* (2012)114:41-49.

Impact factor: 2.324

This journal is reported in the Quartile 1 of “Marine & Freshwater Biology” subject category being in the 22th position of the 100 journals included, and in the Quartile 2 of “Oceanography” subject category being in the position 16th of the 60 journals included. “Marine & Freshwater Biology” category has a median IF of 1.5, and “Oceanography” has a median IF of 1.7.

II) Rodríguez-Climent S, Moreau J, Ibáñez C, Caiola N. **Assessing artisanal fisheries impact in a Ebro Delta coastal lagoon with an Ecopath with Ecosim mass-balanced model.** *ICES Journal of Marine Science* (to be submitted).

Impact factor: 2.277

This journal is reported in the Quartile 1 of “Fisheries” subject category being the 7th out of 49 journals; and in the Quartile 2 of the “Marine & Freshwater Biology” and “Oceanography” categories being in the 25th and 17th position of the 100 and 60 journals included in both categories respectively. Fisheries category has a median IF of 1.4, “Marine & Freshwater Biology” 1.5 and “Oceanography” 1.7.

III) Rodríguez-Climent S, Caiola N, Ibáñez C. **Salinity as the main factor structuring small-bodied fish assemblages in hydrologically altered Mediterranean coastal lagoons.** *Scientia Marina* (2013) 71(1): 37-45.

Impact factor: 1.006

This journal is reported in Quartile 3 of “Marine & Freshwater Biology” subject category, being in the 65th position of the 100 journals included, which have a median IF value of 1.5.

and CERTIFY

that Sílvia Rodríguez Climent contribution has been very active, as it is demonstrated by her first coauthoring of all the manuscripts that conform this Ph.D. thesis. Specifically, her participation included the following tasks:

- Definition of the objectives and focus of the research and its derived manuscripts.

- Field work, including water and habitat specific fish samples collection, and *in situ* physico-chemical measurements.
- Compilation of fishermen official landings.
- Sorting, counting and identification of fish species.
- Fish biometry.
- Data analysis, statistical and dynamic modelling and result's interpretation.
- Tables and Figures design and preparation.
- Main writing of the manuscripts, and contact person for the reviewing and editing process.

Barcelona, July 2nd 2014

Dr. Nuno Caiola and Dr. Adolf de Sostoa Fernández

SUMMARY

(catalan)

2

2.1 Introducció

Les llacunes costaneres i la seva composició de peixos

Les llacunes costaneres poden definir-se com a masses d'aigua salada o salobre, normalment someres i relativament isolades del mar obert (Barnes, 1980; Kjerve, 1989; Isla, 1995; Pérez-Ruzafa, 2007). Representen aproximadament el 13% de les àrees costaneres d'arreu del món, i són presents en tots els continents menys a l'Antàrtida (Barnes, 1980). D'entre tots els continents on apareixen, les llacunes mostren un ampli rang de mides, variant des d'una hectàrea a més de 10.000 km², e.g. Lagoa dos Patos a Brasil (Kennish i Paerl, 2010).

Diferents factors físics com les entrades d'aigua, la força dels vents, les variacions en les taxes d'evaporació-precipitació, l'escorrentia superficial o les condicions meteorològiques locals caracteritzen a les llacunes costaneres (Kennish i Paerl, 2010); però en general aquestes poden ser classificades en tres tipus geomorfològics segons l'aigua que intercanvien amb l'oceà (de major a menor grau): "ofegades" (choked), "restringides" (restricted) i "foradades" (leaky) (Kjerfve, 1994). Aquesta catalogació geomorfològica és un indicador de la variabilitat hidrològica i de les forces dominants a les llacunes (Kjerve, 1994).

Les llacunes costaneres són un dels sistemes biològics més productius del món (Kennish, 2002; Kennish i Pearl, 2010). Aquest fet, unit a la protecció física que ofereixen, les converteix en zones adequades per a la cria i alimentació de juvenils de multitud d'espècies, entre les quals destaquen els peixos (Yañez-Arancibia *et al.*, 1994; Elliott *et al.*, 2007).

A la regió Atlàntico-Mediterrània han estat reportades 199 espècies de peixos en 40 llacunes costaneres (Pérez-Ruzafa i Marcos, 2012), essent les famílies Mugilidae, Sparidae, Gobiidae, Atherinidae, Soleidae, Sygnathidae, Anguilidae, Moronidae i Blennidae les més abundants (Pérez-Ruzafa *et al.*, 2006, 2007, 2011; Maci i Basset, 2009, Verdiell-Cubedo, 2009).

Les llacunes costaneres són també molt importants des del punt de vista econòmic, essent la pesca, una de les activitats que més contribueix a aquesta riquesa (Kennish i Paerl, 2010). Aquesta activitat pot arribar a registrar valors de 100 kg ha⁻¹ any⁻¹ en mitjana de producció de peix (Yañez-Arancibia, 1994); en aquest sentit, les llacunes costaneres són generalment més productives en termes de rendiment pesquer que cap altre ecosistema aquàtic. D'altra banda, les llacunes també forneixen un elevat nombre de béns (recursos alimentaris, minerals i recursos medicinals, fauna i flora, font d'aigua per a ús domèstic, d'agricultura o industrial, provisió per a l'agricultura per dipòsit de terres fèrtils) i serveis (turisme/esbarjo, barreja de nutrients, biodiversitat, zones de cria per a peixos, mantenició hidrològica, oportunitats educatives, retenció de sediments, provisió d'hàbitats) als humans, fet que les ha convertit en ecosistemes històricament habitats (Gonenç i Wolfin, 2004; Seeram, 2008).

Principals amenaces a les llacunes costaneres i a les comunitats de peixos

La posició intermèdia de les llacunes - situades entre els ecosistemes terrestre i marí - les converteix en ecosistemes molt fràgils i sensibles tant a factors d'estrès naturals com antropogènics (Gonenç i Wolfin, 2004; Pérez-Ruzafa, 2005; Kennish i Pearl, 2010).

Factors d'estrès naturals. Encara que històricament els factors d'estrès naturals han rebut menys atenció que els antropogènics, el cert és que les conseqüències provocades per els primers poden ser de majors dimensions (Paerl *et al.*, 2009). Entre els factors ambientals que actuen com a components d'estrès, les llacunes costaneres són fortament dependents de: el vent, l'oscil·lació mareal, l'augment del nivell del mar, els balanços d'evaporació-precipitació, els balanços de calor superficial, les condicions climàtiques i el canvi climàtic (Kennish i Paerl, 2010). L'impacte d'aquests sobre les llacunes depèn principalment de característiques intrínseques de cada llacuna, com ho és la seva geomorfologia (Kjerfve, 1994; Kennish i Paerl, 2010). És important destacar, però que alguns factors d'estrès d'origen natural poden veure's agreujats per certes activitats humanes.

Factors d'estrès antropogènics. La sobre(explotació) humana per obtenir béns i serveis ha provocat la deterioració de les llacunes costaneres (Gonenç i Wolfin, 2004).

Es presenta a continuació una petita descripció de les principals causes i conseqüències d'aquests factors d'estrès sobre els ecosistemes indicats:

a) Alteració i pèrdua d'hàbitat. Normalment fomentat per factors físics (e.g. dragatge, modificació de la línia de costa i recuperació de zones humides), l'alteració de l'hàbitat crea problemes seriosos i persistents com modificacions a llarg plaç de les conques fluvials i estuarines que afecta als organismes que hi habiten (Kennish, 2002; Kennish i Pearl, 2010). La pèrdua d'hàbitat degut a la reclamació de terres per a l'agricultura i altres activitats humanes provoca grans pèrdues econòmiques, degut al fet que elimina valuoses zones per a la fresa, cria i alimentació (Kennish, 2002).

b) Eutrofització. Entès com el procés natural per el qual una massa d'aigua adquireix una alta concentració de nutrients, especialment fosfats i nitrats, l'eutrofització típicament promou el creixement excessiu d'algues (Art, 1993). A mesura que aquestes moren i es descomponen, nivells elevats de matèria orgànica, esgoten l'oxigen disponible a l'aigua, induint-ne la hipòxia (Art, 1993). La hipòxia ocasiona al seu torn, una disminució general de la biodiversitat, reduint l'abundància de la ictiofauna i d'altres poblacions animals (Horrigan *et al.*, 2002). Entre les activitats humanes que alliberen grans quantitats de nutrients al medi, promovent i accelerant el procés natural d'eutrofització destaquen: l'agricultura (*i.e.* fertilitzants, escorrentia de camps de cultiu, fertilitzants químics, fems), l'aqüicultura, les àrees urbanes i industrials (*i.e.* aigües residuals municipals, plantes de tractament) i la combustió de combustibles fòssils (*i.e.* alliberament de nitrogen a l'atmosfera) (Selman i Greenhalgh, 2009).

c) Aigües residuals i residus orgànics. Les aigües residuals són enteses com a matèria de rebuig provinent d'usos domèstics o industrials; mentre que els residus orgànics constitueixen només la part orgànica de la matèria de rebuig. Les activitats humanes que contribueixen a la descàrrega d'aigües residuals inclouen: descàrregues puntuals provinents d'usuaris industrials i de fàbriques, abocaments difusos locals (comercials i domèstics) o escorrenties després de la deposició aèria (Rogers, 1996). Els efectes adversos associats a un augment d'aigües residuals i residus orgànics són: blooms d'algues tòxiques o nocives, efectes d'ombra, aparició de toxines (e.g. sulfits), mortaldat d'espècies bentòniques i pelàgiques, reducció de la biodiversitat, disminució de la producció secundària, disminució de les pesqueries (tant recreatives com comercials) i alteració de la composició d'espècies, abundància i distribució dels organismes (Costello i Read, 1994; Alongi 1998; Holmer, 1999; Howarth *et al.*, 2000; Kennish, 2002).

d) Sobreexplotació pesquera. És ben sabut que unes de les principals conseqüències de la sobreexplotació dels recursos pesquers és el descens de les poblacions de peixos i mol·luscs d'importància recreativa i comercial (Botsford *et al.*, 1997). Al llarg de la història hi ha nombrosos exemples on la reducció de l'abundància de les poblacions ha dut al seu col·lapse, com és el cas de la sardina de Califòrnia (*Sardinops sagax* (Jenyns, 1842)) a Califòrnia i Japó a finals dels anys 1940; o de l'anxoveta del Perú (*Engraulis ringens* Jenyns, 1842) a les costes de Perú i Xile l'any 1972 (Botsford *et al.*, 1997; Hutchings, 2000; descrit en detall a la secció 3).

e) Augment del nivell del mar. L'escalfament global fomentat per activitats humanes com la crema de petroli i carbó, o la tala de boscos tropicals (*i.e.* increment de la concentració de gasos hivernacle), han estat reconegudes com les principals causes de l'escalfament del mar en el últim segle. L'escalfament de l'aigua de mar, provoca la fusió del gel dels glaciars i de les capes de gel a taxes més elevades que les de creixement, afegint d'aquesta manera aigua als oceans i provocant l'augment del nivell del mar (Nicholls i Cazenave, 2010; Union of Concerned Scientists, 2013). Aquest escalfament de l'aigua, amb l'augment de l'aigua associat, pot fomentar alteracions en la composició d'espècies i la distribució de la flora i la fauna de les zones humides (Kennish, 2002).

f) Contaminants químics. Els contaminants químics poden ser dividits en tres tipus depenent de l'efecte potencial que exerceixen sobre els organismes, classificats segons el seu grau de perillositat (de major a menor) de la manera següent: (i) hidrocarburs aromàtics policíclics (PAH's) (ii) hidrocarburs halogenats i (iii) metalls pesants (Mañosa *et al.*, 2001; Kennish, 2002). Les activitats humanes que promouen l'alliberament de PAH's inclouen els vessaments de petroli, les descàrregues municipals i industrials, marines, la combustió de combustibles fòssils i la crema de residus; mentre que els insecticides, herbicides i els productes químics industrials que entren en el sistema a través d'aigües residuals industrials i municipals, escorrenties urbanes i provinents de les granges o la deposició atmosfèrica, són fonts d'hidrocarburs halogenats (Kennish, 2002). Els metalls pesants per la seva banda, solen ser derivats d'activitats humanes com la mineria i la fundició, refinació i galvanoplàstia, fabricació de tints i pintures i crema de combustibles fòssils (Kennish, 2002). Els contaminants químics alteren tot l'ecosistema, provocant pèrdua d'espècies rares o sensibles, disminuint-ne la seva abundància, causant canvis en l'estructura d'edats de les poblacions, i finalment alterant-ne la seva fisiologia i relacions tròfiques. A més a més, degut a que alguns contaminants tendeixen a

bioacumular-se en organismes aquàtics, poden arribar als grans depredadors per efecte de la biomagnificació (Kennish, 2002).

g) Diversions d'aigua dolça (gestió de l'aigua). Hi ha múltiples processos que modifiquen el curs natural de l'aigua dolça, i conseqüentment alteren la dinàmica de les llacunes costaneres (descriu en detall en la secció 4, com a l'altre factor d'estrès que afecta a les llacunes costaneres estudiades): l'increment del desenvolupament urbà en les àrees costaneres, la canalització duta a terme per a fins municipals, industrials o d'agricultura, el control del cabal, la construcció de preses i d'estructures de retenció o la producció de sal i el turisme (Seeram, 1998; Ambrose i Meffert, 1999; Day Jr. *et al.*, 2002; Kennish, 2002; Lucena *et al.*, 2002; Lenzi *et al.*, 2003; De Stefano, 2004; Badosa *et al.*, 2007). Els impactes sobre l'ecosistema poden ser greus, degut a que l'alteració de les entrades d'aigua pot mudar els patrons de salinitat i els règims de sediments, així com les descàrregues de nutrients i contaminants; repercutint directament sobre l'abundància i distribució dels organismes a les zones afectades (Kennish, 2002).

h) Espècies introduïdes/invasores. La introducció d'espècies es defineix com el procés intencionat o accidental per el qual una espècie és importada a una regió fora del seu rang natiu (Moyle i Light, 1996; Moyle 1997). Les espècies introduïdes adquireixen caràcter invasiu quan es reproduïxen, es tornen abundants i s'expandeixen des del seu punt d'introducció (Richardson, 2000; Kolar i Lodge, 2001). Les espècies invasores poden provocar una disminució de la diversitat genètica i homogeneïtzació biòtica (Rahel, 2000). També promouen canvis en l'organització tròfica, infiltració de patògens perjudicials i alteració dels hàbitats (Kennish, 2002).

i) Subsidiència. La subsidiència és un procés irreversible on el nivell de la llacuna és deprimat respecte al nivell del mar. Tot i que la subsidiència geològica és un procés natural, la subsidiència antropogènica s'ha incrementat en els darrers temps (Brambati *et al.*, 2003). A l'ecosistema, provoca pèrdua de la qualitat de l'aigua i promou la presència de la cunya salina que inevitablement altera la flora i la fauna de l'ecosistema (Prat i Ibañez, 1995).

j) Entrada de sediments/turbulència. Un volum considerable de sediment pot entrar a les llacunes costaneres durant onades de tempesta o esdeveniments de rentat. A més a més, els sediments també poden provenir dels rius, esorrentia d'altres hàbitats i de processos interns (Kennish i Paerl, 2010). Nivells elevats de turbulència a l'aigua són perjudicials, vist que n'alteren la seva transparència (Kennish, 2002), en

redueixen l'activitat fotosintètica i conseqüentment modifiquen la seva producció primària, així com la producció de tot l'ecosistema.

k) Substàncies flotants i runes. Constitueixen el material de rebuig creat per l'home i alliberat al medi. Les runes tenen un impacte directe en els organismes, que moren al ingerir-les o ofegats al quedar-hi atrapats. S'ha pronosticat un increment de les runes i substàncies flotants en estuaris i àrees costaneres per l'any 2025, degut a l'augment esperat de població en aquestes àrees (Kennish, 2002).

Dos dels serveis de l'ecosistema més importants per al benestar del ésser humà: els peixos i l'aigua (Millenium Ecosystem Assessment, 2005) són sobreexplotats per factors d'estrès humà. Les pesqueries són de particular importància en països en desenvolupament, on els peixos molt sovint representen la primera font de proteïna animal (Béné *et al.*, 2007). Pel que respecta al recurs de l'aigua, les llacunes costaneres constitueixen importants magatzems d'aigua per a ús domèstic, industrial o d'agricultura, així com per a la regulació de caudals (Millenium Ecosystem Assessment, 2005). Atesa la importància de l'impacte humà sobre ambdós recursos, i específicament sobre la seva comunitat de peixos; aquests dos factors d'estrès antropogènics han estat escollits com a primordials a ser estudiats en la present tesi doctoral i seran tractats separatament en les seccions següents.

Pesqueries en llacunes costaneres

Les pesqueries en llacunes costaneres representen un 10% de la producció mundial (Quignard, 1984). Tot i la seva importància, el coneixement científic que se'n té és bastant escàs, difús o superficial (Pérez-Ruzafa i Marcos, 2012). Malgrat tot, un dels aspectes coneguts és que aquest tipus de pesqueries romanen en bona part artesanals (Kennish i Paerl, 2010).

Les pesqueries artesanals i de petita escala sovint competeixen amb les pesqueries industrials. No obstant, les primeres compten amb una sèrie d'avantatges tals com: (i) baixes despeses de funcionament i de consum de combustible, (ii) menor impacte ecològic, (iii) oportunitats més altes de contractació de personal, (iv) versatilitat, (v) despeses de construcció menors i (vi) tecnologies més econòmiques (FAO, 2005-2013). Tots aquests factors, afegits a la seva baixa taxa de descarts, les converteixen en pesqueries més rentables que les industrials des del punt de vista social, econòmic i ecològic. L'Organització de les Nacions Unides per a la Alimentació i l'Agricultura (FAO) ha reconegut l'important paper desenvolupat per la pesca de petita escala i les pesqueries artesanals en l'ocupació, els ingressos i la seguretat alimentària (FAO,

1995). Tot i aquest reconeixement internacional, el nivell semblant de productivitat (en desembarcaments totals), i la reconeguda necessitat del compliment de la regulació per a mantenir l'esforç pesquer dins de límits sostenibles; el desenvolupament de lleis per a regular les pesqueries artesanals ha rebut molta menys atenció al llarg de la història (Common Fisheries Policy, 2008; Diegues *et al.*, 2006).

Aquesta falta de regulació és crítica, tenint en compte que el terme "artesanal" no vol dir "sostenible". O dit d'una altra manera, les pesqueries artesanals tenen un efecte sobre els ecosistemes que exploten. Un dels efectes directes més coneguts és l'ocasionat sobre les espècies objecte. En les espècies objecte (*i.e.* comercials), la pesca pot provocar el descens de l'abundància, canvis en la composició d'espècies i la reducció de la variabilitat genètica de la comunitat (Goñi, 1998; Blaber, 2000). Per altra banda, la selectivitat de malla pot causar canvis en la mida, la mida-a-certa edat i la mida de maduresa sexual (Botsford, 1997; Goñi, 1998). Un dels potencials impactes de la pesca és la sobrepesca, que malgrat i considerada un problema a gran escala, també pot donar-se en les pesqueries artesanals (UNEP, 2006).

A les llacunes costaneres, les pesqueries artesanals estan fortament adaptades a les condicions del medi. En aquests sistemes, la pesca es basa en un ampli coneixement de l'ecologia i el comportament de les espècies objecte per part dels pescadors (Pérez-Ruzafa i Marcos, 2012). La conseqüència, és la convergència adaptativa en el tipus d'arts usades per pescadors a diverses llacunes d'arreu del món: totes elles s'aprofiten dels moviments migratoris dels peixos entre les llacunes i el mar (Pérez-Ruzafa i Marcos, 2012).

Gestió de l'aigua en les llacunes costaneres

L'aigua, com a recurs, és sovint sotmesa a moltes pressions derivades d'activitats humanes, esdevenint l'agricultura una de les més importants (veure secció anterior). L'elevada fertilitat dels sòls circumdants a les llacunes costaneres, les ha convertit en objecte del desenvolupament d'una intensa activitat agrícola. Tanmateix, la gran concentració de sals trobada en aquest tipus de sòls, torna quasi impossible el cultiu de qualsevol espècie (Galcerà, 2009). De fet, la planta més adequada (i potser la única) a cultivar en sòls deltaics és l'arròs, que requereix una inundació continua per a créixer. L'aigua dolça necessària per al cultiu d'aquest cereal és proporcionada per una extensa xarxa de canals de reg que transporten l'aigua del riu fins a cada parcel·la de cultiu (Chauvelon, 1998; Day Jr. *et al.*, 2002). Aquesta transformació implica no només canvis a nivell del cicle hidrològic -que queda totalment artificialitzat- sinó que també afecta a la fauna i flora habitant; alterant-ne els seus patrons d'abundància i distribució (Poizat *et*

al., 2004; Badosa *et al.*, 2007), fet que finalment acaba modificant l'estructura de tot l'ecosistema (Heurteaux, 1992). Específicament en els peixos, els canvis hidrològics poden implicar canvis en els patrons migratoris (Griffiths, 1999; Day Jr. *et al.*, 2002); alhora que ambients menys salins són capaços de promoure l'establiment i expansió d'espècies introduïdes en àrees on els seus depredadors són absents (Badosa *et al.*, 2006).

Com s'ha comentat prèviament i agreujat per l'augment global de la població, un elevat nombre de pressions antropogèniques està comprometent l'habilitat per a mantenir la qualitat i productivitat de les llacunes costaneres. Dit d'una altra manera, aquests sistemes no són capaços d'assolir les actuals demandes de creixement socioeconòmiques sense ser explotades d'una manera sostenible (Gonenç i Wolfen, 2004; Kennish i Paerl, 2010).

Gestió sostenible en llacunes costaneres

Segons el Brundtland Report, el desenvolupament sostenible és entès com “*aquell que satisfà les necessitats de les generacions presents sense comprometre'n el de les generacions futures*” (World Commission on Environment and Development, 1987). D'altra manera, s'entén com a ús sostenible, aquell en que un recurs és recol·lectat, extret o usat en una mesura en la que pot auto-regenerar-se (Clark, 1992). Per tant, una gestió sostenible és aquella que comprèn la planificació, el control i l'organització de les estratègies requerides per a un ús sostenible dels recursos naturals, assegurant d'aquesta manera el seu potencial biològic i renovable a llarg termini (Clark, 1992; Ritcher *et al.*, 1996).

Un pla de gestió sostenible ha de definir objectius concisos per a l'ús i l'explotació del medi, elaborar un inventari dels seus recursos naturals, estudiar els processos implicats en el funcionament de l'ecosistema, indexar les activitats humanes i les seves influències en els processos ecològics, classificar el territori segons els seus diferents usos, establir estàndards de qualitat ambiental i finalment dissenyar i monitoritzar plans d'actuació i mesures correctores (Pérez-Ruzafa *et al.*, 2005).

Amb tot, la gestió sostenible d'un recurs requereix el consens dels interessos de totes les parts implicades (naturalistes, polítics, terratinents, defensors del progrés, veïns de la zona i membres de la comunitat científica), així com de les necessitats de generacions presents i futures (Rigg, 1999; Suman *et al.*, 2005). A nivell Europeu, la inhabilitat per a conciliar els lucre de totes les parts involucrades ha estat assenyalada com la principal

causa de la degradació dels ecosistemes costaners i marins (Rigg, 1999; Suman *et al.*, 2005).

Una de les eines més usades per a la gestió de pesqueries costaneres és la implementació de models ecològics, que permeten integrar els interessos de tots els sectors afectats i obtenir d'aquesta manera la visió holística exigida. Donat que la modelització ecològica en la gestió de pesqueries és un tema molt ampli, aquest apartat centrarà la seva atenció en els anomenats "Models d'ecosistema globals" ("Whole ecosystem models"); models que tenen en consideració tots els nivells de l'ecosistema (*i.e.* des de productors primaris fins a superpredadors). Entre aquests models s'inclouen: Ecopath with Ecosim (EwE; Christensen *et al.*, 2008), ATLANTIS (Fulton i Smith, 2004), INVITRO (Gray *et al.*, 2006), European Regional Seas Ecosystem Model (ERSEM II; Baretta, Baretta-Bekker i Ruardij, 1996), i Spatial Ecosystem And Population Dynamics Model (SEAPODYM; ERSEM II; Baretta, Baretta-Bekker i Ruardij, 1996). D'entre tots els models enumerats, EwE va ser l'escollit en la present tesi doctoral per diverses raons: (i) la seva interfície fàcil d'usar, (ii) no requereix una recollida de dades intensiva i (iii) és mundialment utilitzat. No obstant, cal tenir en compte que qualsevol model és només una representació sistemàtica de la realitat, i el seu desenvolupament (dades usades, equacions, àrea d'estudi), defineix en part les qüestions científiques plantejades a respondre. Per tant, independentment de la seva complexitat, un model mai no reproduirà la realitat, essent només una rigorosa aproximació d'aquesta (Coll, 2006).

De la mateixa manera, la gestió sostenible de l'aigua requereix una visió integrada, on es contempli el subministrament, l'ús i el tractament de la mateixa. En conseqüència, la Unió Europea va adoptar recentment la Directiva Marc de l'Aigua (DMA o WFD sigles en anglès; Directiva 2000/60/EC) per a establir els principis bàsics per a una política de l'aigua sostenible a tots els estats membres. L'objectiu principal de la DMA és aconseguir un "bon estat ecològic i químic de les aigües" per a l'any 2015, incloent les aigües de transició de les llacunes costaneres i les seves conques (Unió Europea, 2000).

Considerant que, a les llacunes costaneres, l'agricultura és una de les principals activitats que influeix en la gestió del recurs de l'aigua (veure secció anterior); el desenvolupament d'una agricultura més sostenible comportarà un ús més sostenible de l'aigua a les llacunes costaneres (Bitelli, 2010). S'han reconegut cinc punts clau per al desenvolupament d'una agricultura més sostenible: (i) increment de la eficiència en l'ús de l'aigua (Tilman *et al.*, 2000); (ii) sembrar cultius amb major tolerància a la sequera (Tilman *et al.*, 2000); (iii) millorar la qualitat de l'aigua (*i.e.* impedir-ne la seva salinització

i contaminació per herbicides, nitrats i seleni); (iv) millorar la gestió de reg (Wichelns, 2002) i (v) reutilitzar l'aigua de drenatge per a regar camps amb una elevada tolerància a la salinitat (Rhoades, 1999).

Context d'estudi

El Delta de l'Ebre amb 320 km² d'extensió, representa l'àrea humida més important de Catalunya, la segona àrea més important d'Espanya (després del Parc Nacional de Doñana) i una de les zones estuarines més importants a Europa (RAMSAR, 1992; Fernández *et al.*, 1997; Day *et al.*, 2006). Des del punt de vista ecològic, el Delta de l'Ebre destaca per la seva elevada diversitat d'hàbitats en una àrea significativament reduïda (Ibañez *et al.*, 1999). La composició d'espècies, la seva fauna ornitològica i ictiològica, així com la seva vegetació halòfítica fan del delta un ecosistema singular (Ibañez *et al.*, 1999). A més a més, moltes de les espècies trobades al delta, són endèmiques, la conservació de les quals hauria de ser una tasca prioritària (Ibañez *et al.*, 1999).

Degut al seu elevat interès biològic, el 25% de l'àrea deltaica - incloent les llacunes costaneres i la línia de costa - estan protegides com a Parc Natural des de l'any 1983. Declarada també Zona d'Especial Protecció per a les Aus (ZEPA) l'any 1987, Lloc d'Importància Comunitària (LIC) i considerada una zona humida d'importància internacional en la convenció RAMSAR des de l'any 1993. A més a més des de l'any 2000, el delta està inclòs en la Xarxa Natura 2000 i recentment la UNESCO l'ha declarat com a Reserva Mundial de la Biosfera (2013).

Distribuïdes al llarg del delta, les 9 llacunes costaneres representen un 7.2% de l'àrea total: el Canal Vell, el Calaix de l'Illa de Buda, la Platjola, l'Aufacada, les Olles, el Garxal l'Encanyissada, el Clot i la Tancada.

En la present tesi doctoral, l'atenció s'ha centrat en tres llacunes situades a l'hemidelta sud: l'Encanyissada, el Clot i la Tancada. Els criteris usats per a l'elecció d'aquestes tres llacunes han estat els següents: (i) la seva representativitat en termes de variabilitat de mides, morfologia i connexions amb el mar i (ii) el fet d'estar sotmeses als dos estressors antropogènics estudiats en la present tesi doctoral: les pesqueries artesanals i la gestió de l'aigua. En aquest sentit, només en sis de les llacunes anteriorment enumerades s'hi practica la pesca artesanal: el Clot, la Tancada, l'Encanyissada, el Canal Vell, les Olles i la Platjola; essent la última d'aquestes descartada degut al seu caràcter privat. Atès que la gestió de l'aigua difereix entre els dos hemideltos (*i.e.* l'obertura i arribada de l'aigua dolça és progressiva, començant

primer en un hemidelta i després a l'altre; Comunicació personal del Parc Natural); d'entre la resta, la llacuna de la Tancada presenta una gestió de l'aigua diferent que mereixia ser estudiada i per tant, les llacunes situades en el seu mateix hemidelta varen ser escollides per tal d'obtenir resultats comparables.

Tot i la recent colonització humana, el Delta de l'Ebre està considerat com una àrea fortament impactada (Ibañez *et al.*, 1999; Mañosa *et al.*, 2001). De fet, la vegetació original (salobrars, absència d'arbres) ha mudat durant el darrer segle degut a l'agricultura (Espanya, 1997). L'agricultura a la zona ha afrontat (i encara afronta) un gran inconvenient: l'elevada salinitat dels sòls deltaics, que ha fet imprescindible la transformació de l'àrea per tal de fer-la cultivable (Galcerà, 2009). Per entendre bé la història del delta i la seva evolució, distingim quatre períodes atenent al tipus i intensitat d'explotació de la terra:

- 1) *Fins a 1900: Delta natural.* Fins a la primera meitat del segle XIX, la transformació del delta va ser minsa i les seves propietats naturals varen preservar-se: grans riuades i elevats fluxos de sediments que tenien com a resultat elevades taxes d'acreció, grans superfícies de zones humides i baixos nivells d'eutrofització (Mañosa *et al.*, 2001). La primera referència al cultiu de l'arròs la trobem al segle XVI, encara que no és fins alguns segles més tard, l'any 1860 quan la construcció del primer canal a l'hemidelta sud, va permetre la implementació d'aquest tipus de cultiu que va assolir la seva màxima expansió a finals del segle XIX-principis del XX quan aproximadament 4000 ha (12% de l'àrea total) eren cultivades (Mañosa *et al.*, 2001; De Sostoa i De Sostoa, 1985).
- 2) *1900-1960: Delta humanitzat ("revolució de l'arròs").* Aquest període està caracteritzat per la transformació de la plana deltaica com a conseqüència de l'expansió del cultiu de l'arròs (Ibañez *et al.*, 1999; Galcerà, 2009). Per tal de permetre aquesta expansió, una extensa infraestructura de canals que transporten aigua dolça als camps de cultiu va ser construïda. L'entrada d'aigua dolça va modificar la dinàmica hidrològica de tot el delta, que des d'aquell moment va passar a ser més regular i independent de la climatologia (Ibañez *et al.*, 1999). Les constants aportacions d'aigua dolça durant l'època de cultiu de l'arròs (Abril-Desembre), han invertit el patró hidrològic de les llacunes costaneres del Delta de l'Ebre. Tot i ser un període caracteritzat per una elevada transformació de la terra, pot dir-se que els sistemes naturals no van ser severament impactats degut a la manera tradicional en la qual va dur-se a terme l'explotació (*i.e.* sense la mecanització dels cultius i l'ús de pesticides/herbicides).

- 3) *1960-1980: Delta amenaçat (I)*. Durant aquestes dues dècades, el delta va passar de l'explotació extensiva i tradicional a una intensiva i tecnològica amb un clar objectiu en ment: maximitzar la producció de l'arròs (Ibañez *et al.*, 1999). Aquest fet va suposar la mecanització dels cultius i la introducció d'insecticides i pesticides en els conreus (Ibañez *et al.*, 1999). En el sector de la economia de mercat, les captures de pesca van començar a ser registrades, fet que ens indica una major activitat professional (1965 primer any de registres; Associació de Pescadors de St. Pere).
- 4) *1980-actualitat: Delta amenaçat (II)*. En aquest període, algunes mesures com la creació del Parc Natural (1983), o la implantació de la legislació ambiental Europea han fet l'explotació del delta més sostenible (Ibañez *et al.*, 1999). L'establiment d'aquestes mesures ha aturat la pèrdua d'hàbitats naturals, i actualment 25% de la superfície deltaica (7736 ha) correspon a llacunes i zones humides protegides pel Parc Natural (Mañosa *et al.*, 2001).

El progrés en la plana deltaica, ha transformat un ambient natural en un ambient completament depenent de la gestió humana. El preu que el delta ha hagut de pagar ha estat força elevat: reducció de poblacions de rèptils, peixos, amfibis i mamífers, reducció de la diversitat de la flora i deterioració de serveis i valors ecològics irrecuperables.

La present tesi doctoral està emmarcada en l'últim dels períodes descrits, i per tant s'enfronta a un delta amenaçat i en perill. De totes les activitats humanes desenvolupades a la plana deltaica, aquesta tesi està centrada en l'estudi de dos dels majors sectors econòmics a la zona: les **pesqueries artesanals** i la **gestió de l'aigua** associada al cultiu de l'arròs. S'analitza l'impacte d'aquests dos factors d'estrès en les comunitats piscícoles.

La ictiofauna del les llacunes costaneres del Delta de l'Ebre té tres orígens diferents: d'aigua dolça, marina i salobre; essent l'èxit de la colonització dependent del seu grau d'adaptació i la seva capacitat de suportar fluctuacions ambientals. L'entrada d'espècies d'origen marí a les llacunes és facilitada pels nombrosos canals que connecten les llacunes amb la Badia dels Alfacs (Mar Mediterrània); mentre que l'entrada de les espècies d'aigua dolça sol estar restringida als períodes en els quals els canals d'irrigació transporten aigua del riu.

Les espècies descrites a les llacunes costaneres amb anterioritat es troben enumerades a la Taula 5 de la introducció en anglès i seran resumides a continuació:

Entre les espècies marines citades anteriorment trobem les espècies accidentals com la sardina (*Sardina pilchardus* (Walbaum, 1752)) o el seitó (*Engraulis encrasicolus* (Linnaeus, 1758)) i espècies estacionàries com el peix agulla (*Belone belone* (Linnaeus 1761)). Les espècies estacionàries que entraven com a juvenils a les llacunes en busca d'un lloc segur on alimentar-se incloïen el llenguado (*Pegusa lascaris* (Risso, 1810)), el llenguado nassut (*Pegusa nasuta* (Pallas, 1814)) i el llenguado ver (*Solea solea* (Linnaeus, 1758)); mentre que altres espècies com el rèmol de riu (*Platichthys flesus* (Linnaeus, 1758)) només entraven a les llacunes en el seu estat adult (De Sostoa *et al.*, 1990). La resta de les espècies estacionàries trobades a les llacunes, tant en el seu estat adult com juvenil varen ser: mugílids (llissa vera (*Chelon labrosus* (Risso, 1827)), llissa calua (*Liza ramada* (Risso, 1827)), llissa petita (*Liza saliens* (Risso, 1810)), llissa galta-roja (*Liza aurata* (Risso, 1810)), llissa llobarrera (*Mugil cephalus* Linnaeus, 1758) i la llissa morruda (*Oedalechilus labeo* (Cuvier, 1829)), el llobarro (*Dicentrarchus labrax* (Linnaeus, 1758)) i l'orada (*Sparus aurata* Linnaeus, 1758). També era comú trobar espècies salobres/marines que vivien a les llacunes i eventualment anaven a reproduir-se al mar com el jovell (*Atherina boyeri* Risso, 1810) o el gòbit de sorra (*Pomatoschistus microps* (Krøyer, 1838)). Les espècies d'aigua dolça trobades a les llacunes eren o bé: sedentàries (fartet (*Aphanius iberus* (Valenciennes, 1821)), gambúsia (*Gambusia holbrooki* Girard, 1859)), estacionals (carpa (*Cyprinus carpio* Linnaeus 1758), carpí (*Carassius auratus* (Linnaeus 1758)) o accidentals (bavosa de riu (*Salaria fluviatilis* (Asso, 1801))). L'anguila (*Anguilla anguilla* (Linnaeus, 1758)) va ser la única espècie catàdroma trobada en aquella època (De Sostoa, 1983; De Sostoa i De Sostoa, 1985; De Sostoa i Lobón-Cerviá, 1989; De Sostoa *et al.*, 1990).

De les 24 espècies històricament trobades a les llacunes costaneres, tres varen resultar ser exòtiques (*G. holbrooki*, *C. carpio* i *C. auratus*). Addicionalment, dues de les espècies històricament descrites al delta: *A. iberus* i *A. anguilla* estan actualment catalogades com espècies en perill i en perill d'extinció respectivament per la Unió Internacional per a la Conservació de la Natura (ICUN); mentre que *A. boyeri* i *S. fluviatilis* estan considerades com espècies vulnerables a Espanya (Doadrio, 2001 i RD 139/2011 respectivament).

Principals amenaces per a la ictiofauna de les llacunes

Una de les grans amenaces a les quals la ictiofauna del Delta de l'Ebre ha de fer front són les **pesqueries artesanals**. Practicades de manera professional al delta des del segle XVII (Galcerà, 2009), les pesqueries artesanals són auto-gestionades per la Confraria de Pescadors de Sant Pere. Cada juliol (abans de l'inici de la temporada de

pesca), aquesta associació organitza un sorteig públic per a determinar els pescadors que podran pescar aquell any (Franch i Quintano, 2004; Fernández, 2007).

Els pescadors a les llacunes usen dos tipus d'arts passives: xarxes d'emmallar i trampes. Entre les xarxes d'emmallar, l'art de pesca usada a les llacunes són els tremalls; mentre que s'usen tres tipus de trampes diferents: el gànguil, el bussó i la pantena.

Encara que existeix un cert control, ja que el número de pescadors que pot pescar per temporada és limitat, el cert és que l'activitat no està sotmesa a mesures de control restrictives en quant al número d'arts de pesca per llacuna i mida de les espècies es refereix. Set de les espècies trobades històricament a les llacunes tenen interès comercial en els nostres dies: *D. labrax*, *S. aurata*, *A. boyeri*, angules i anguiles adultes (*A. anguilla*), adults d'espècies pertanyents a la família Mugilidae (*L. saliens*, *L. aurata*, *L. ramada*, *M. cephalus*, *C. labrosus*) i *C. caprio*.

L'altre gran amenaça a la qual la ictiofauna del Delta de l'Ebre ha de fer front, és la **gestió de l'aigua** derivada del cultiu de l'arròs. Per tal de sostenir aquest cultiu, dos canals principals, regulats en el seu punt d'origen (40 km riu amunt) varen ser construïts i dividits en nombrosos canals secundaris i comportes per tal de nodrir els camps d'arròs amb aigua dolça. L'aportació d'aigua dolça és estacional, sent tan sols interrompuda entre Gener i mitjans d'Abril quan els camps són preparats per a la propera sembra. Antigament, l'aigua dels arrossars era abocada a les llacunes, i des d'aquests sistemes al mar. Aquest sistema de gestió de l'aigua va causar l'eutrofització de les llacunes (Forés *et al.*, 2002). Problema que va ser solucionat als anys noranta amb la construcció de canals de circumval·lació que evitaven que l'aigua de drenatge dels arrossars fos abocada a les llacunes (Forés *et al.*, 2002). Actualment, les llacunes costaneres del Delta de l'Ebre segueixen rebent aigua dolça artificialment durant nou mesos a l'any; els primers sis mesos per a estimular el creixement de l'arròs (abril-setembre), i els tres últims per a fomentar habitats per a la reproducció de les aus (octubre-desembre; Taula 4 introducció en anglès).

2.2 Objectius

El principal objectiu de la present tesi doctoral va ser avaluar els efectes de dues de les activitats humanes dutes a terme a les llacunes costaneres del Delta de l'Ebre: les pesqueries artesanals i la gestió de l'aigua. A més a més, aquest estudi ha permès

investigar l'actual condició de la comunitat piscícola (tant a nivell de composició com d'estructura), desconeguda durant les últimes tres dècades. És esperat que els resultats i les conclusions presentades en aquesta tesi, identificant i quantificant les principals amenaces per a la ictiofauna en el Delta de l'Ebre, siguin d'utilitat per a poder aplicar un pla de gestió sostenible que permeti conservar els recursos naturals del Delta de l'Ebre.

La present tesi doctoral es presenta com un compendi de 3 publicacions científiques que corresponen als 3 capítols. Encara que l'ordre dels capítols no és cronològic (*i.e.* el segon capítol encara no ha estat publicat), aquesta estructura ha estat escollida per a una millor comprensió i seguiment de les idees exposades. Per tant, els primers dos capítols es centren en estudiar l'impacte de les pesqueries artesanals; mentre que l'últim capítol tracta sobre la gestió del recurs de l'aigua. En cada capítol en concret els següents objectius varen aconseguir-se:

Capítol I.

- Descripció de la composició i estructura de la actual comunitat de peixos a les llacunes costaneres del Delta de l'Ebre.
- Modelar la selectivitat de malla de les xarxes ganyeres usant el mètode SELECT (sigles en anglès per a "compartir cada longitud de les captures totals"), com a primera temptativa per a descriure els paràmetres de selecció de les xarxes ganyeres.
- Integrar i combinar la informació sobre els paràmetres reproductius amb l'obtinguda amb les xarxes ganyeres i els gànguils per a establir una talla mínima de captura per a *Atherina boyeri*, una espècie vulnerable amb creixent interès comercial.
- Avaluació de la mida de malla i de les xarxes de pesca com a possible enfocament per evitar la sobrepesca del jove (*A. boyeri*), una espècie vulnerable a les llacunes costaneres del Delta de l'Ebre.
- Proposar mesures de gestió i conservació per a *A. boyeri*.

Capítol II.

- Compilació bibliogràfica i estimació de dades necessàries per a la caracterització de l'ecosistema (composició de dietes, biomassa, producció, consum i captures de pesca).
- Construcció d'un model tròfic per la llacuna de la Tancada usant l'aplicació Ecopath.

- Simulació dels efectes de futurs escenaris amb esforços de pesca diferents a l'actual amb l'aplicació Ecosim, per tal d'assessorar el disseny d'una estratègia d'explotació sostenible dels recursos pesquers de la llacuna de la Tancada a llarg termini.

Capítol III.

- Descriure la composició de la comunitat de peixos de petita mida de les llacunes costaneres del Delta de l'Ebre.
- Investigar la relació entre l'estructura i patrons de distribució de la comunitat de peixos i els principals factors ambientals que depenen directament de la gestió artificial de l'aigua.
- Determinar el grau d'associació de cada una de les espècies amb els paràmetres fisicoquímics.
- Proposar un pla de gestió de l'aigua que afavoreixi espècies de peixos natives, importants tant des del punt de vista comercial com conservacionista.

2.3 Discussió general

La present tesi doctoral avalua els efectes de dos dels principals factors d'estrès antropogènics que afecten les llacunes costaneres del Delta de l'Ebre, la **pesca artesanal** i la **gestió de l'aigua**, sobre les comunitats de peixos que hi habiten. Per avaluar els efectes de la pesca artesanal, s'ha estudiat la selectivitat de la mida de malla de les arts de pesca usades (Capítol I). D'altra banda, el jovell (*Atherina boyeri*), una espècie vulnerable i amb un creixent interès comercial per la qual no està reconeguda legalment una mida mínima de captura (MLS, sigles en anglès per a "Minimum Landing Size"); va usar-se com a model biològic per tal de proposar una MLS fent servir una combinació de paràmetres de la biologia reproductiva i de la selectivitat de les arts de pesca (Capítol I). Les dades generades en els mostrejos realitzats per tal de dur a terme els estudis del Capítol I, també van utilitzar-se per fer una descripció acurada de la composició i estructura de les comunitats piscícoles actuals. A més, va utilitzar-se un enfocament de modelització ecològica per predir els efectes a llarg termini de diferents escenaris d'esforç de pesca sobre l'estructura de les comunitats biològiques de les llacunes costaneres del Delta de l'Ebre, amb especial èmfasi en la ictiofauna (Capítol II). Finalment, la influència de l'alteració hidrològica sobre les comunitats de peixos va avaluar-se a través del desenvolupament de models estadístics que permeten l'establiment de relacions entre la distribució i l'estructura de les comunitats piscícoles i

les principals característiques abiòtiques que depenen de la gestió de l'aigua (Capítol III).

Canvis recents en la ictiofauna de les llacunes costaneres del Delta de l'Ebre

En la present tesi doctoral, 25 espècies pertanyents a 15 famílies varen ser trobades en les tres llacunes estudiades (veure Taula 1 part amb anglès). D'aquestes, 18 ja havien estat reportades prèviament a les llacunes costaneres (De Sostoa, 1983; De Sostoa i De Sostoa, 1985; De Sostoa i Lobón-Cerviá, 1989; De Sostoa *et al.*, 1990). Per tant, 7 espècies van ser trobades per primer cop a les llacunes: l'alburn (*Alburnus alburnus* (Linnaeus, 1758)), el barb (*Barbus graellsii* (Steindachner, 1866)), l'*Aphanius fasciatus* (Valenciennes, 1821), el llenguado senegalès (*Solea senegalensis* Kaup, 1858), la pseudorasbora (*Pseudorasbora parva* (Temminck & Schlegel, 1846)), la lucioperca (*Sander lucioperca* (Linnaeus, 1758)) i el silur (*Silurus glanis* Linnaeus, 1758) (Capítol I i III). No obstant, les 3 últimes ja havien estat descrites com a presents en la plana deltaica. En aquest sentit, la introducció de *P. parva* va ser probablement deguda a un escapament involuntari d'una instal·lació d'aqüicultura (Caiola i De Sostoa, 2002); mentre que *S. lucioperca* i *S. glanis* varen ser intencionadament introduïdes degut al seu alt valor com espècies de pesca (Doadrio, 2002; CHE, 2009). La introducció incontrolada de les mencionades espècies ha instigat: (i) la seva expansió al riu Ebre (CHE, 2009) i (ii) la seva migració/desplaçament cap a altres ecosistemes com les llacunes costaneres facilitada per l'extensa xarxa de canals a la plana deltaica. Aquest fet és preocupant, degut a que les introduccions incontrolades poden comportar canvis en la xarxa tròfica i a més a més afectar seriosament a les poblacions natives. De fet, 4 de les 7 noves espècies citades a les llacunes són al·lòctones (les mencionades *P. parva*, *S. lucioperca*, *S. glanis*, i també *A. fasciatus*). Aquest fet, ha triplicat la ràtio d'espècies introduïdes/natives des dels anys 80. Alhora, aquest increment d'espècies introduïdes pot haver afectat les poblacions natives, degut a que 4 de les espècies trobades avui en dia a les llacunes, estan considerades com amenaçades per regulacions nacionals i/o internacionals: *A. iberus*, *S. fluviatilis*, *A. boyeri* i *A. anguilla*.

La comunitat de peixos trobada avui en dia a les llacunes costaneres, està formada per algunes espècies amenaçades, fet que difereix de la situació trobada fa 30 anys. Els resultats obtinguts en aquesta tesi, recalquen la necessitat de promoure polítiques conservacionistes i permeten fer algunes recomanacions per a millorar l'estat de les poblacions de *A. boyeri* i *A. anguilla*. La implementació en sincronia d'una talla mínima de captura i la regulació de la mida de malla per a l'amenaçada *A. boyeri* proposades

per primer cop en el present estudi, són d'importància per a la seva conservació a les llacunes costaneres del delta (Capítol I; veure secció següent); mentre que el model desenvolupat a la llacuna de la Tancada (Capítol III) s'espera que pugui contribuir a desxifrar el coneixement necessari per poder aplicar el pla d'avaluació i conservació que s'està duent a terme a nivell Europeu per a *A. anguilla*, degut al estat de perill crític d'extinció en el que es troba l'espècie (veure secció següent).

Adicionalment, 6 de les espècies mencionades prèviament no varen ser trobades en els mostrejos fets durant la realització d'aquesta tesi: el peix agulla (*Belone belone gracilis* (Linnaeus, 1761)), la llisa morruda (*Oedalechilus labeo* (Cuvier, 1829)), el rèmol de riu (*Platycthis flesus* (Linnaeus, 1758)), el llenguado (*Pegusa lascaris* (Risso, 1810)), el llenguado nassut (*Pegusa nasuta* (Pallas, 1814)) i el llenguado ver (*Solea solea* (Linnaeus, 1758)). Una possible hipòtesi és que l'actual gestió de l'aigua dolça durant el cultiu de l'arròs està condicionant la presència d'aquestes espècies marines que acostumaven a entrar de manera estacional o accidental a les llacunes (De Sostoa, 1983; De Sostoa i De Sostoa, 1985; De Sostoa i Lobón Cervià, 1989 i De Sostoa *et al.*, 1990). Es recomana un monitoratge regular i més exhaustiu per tal de confirmar la seva presència/absència en l'àrea d'estudi.

En termes de riquesa d'espècies, les dues comunitats de peixos (la passada i la contemporània) no difereixen gaire, amb només una espècie més trobada en l'actualitat.

Resumint, en els últims anys, 7 noves espècies han estat trobades a les llacunes. Cal destacar que 4 d'aquestes han resultat ser introduïdes. L'augment de les espècies introduïdes sembla haver estat en detriment de la població d'espècies natives (4 d'elles designades actualment com a amenaçades). Per últim, un suau increment en la riquesa d'espècies ha estat observat en els últims 30 anys.

Impactes de les activitats humanes en les comunitats de peixos

Tot i la controvèrsia envers si les captures poden o no reflectir l'abundància real dels peixos en una determinada àrea (Pauly *et al.*, 2013); el cert és que les captures poden proporcionar una idea de l'abundància relativa dels peixos, degut a la proporció existent entre les captures i l'autèntica abundància (Harley *et al.*, 2001). Al Delta de l'Ebre, des de l'any 1965 -any en que les captures van començar a ser registrades- s'ha donat una acusada disminució de les captures, i per tant de l'abundància de peixos.

La forta disminució és preocupant, si es té en compte, que l'esforç pesquer també ha disminuït des del primer any en el que es van començar a registrar les captures a les llacunes. Aquest fet és un dels que ha captat la nostra atenció respecte a la manera com les **pesqueries artesanals** es duen a terme a les llacunes costaneres del Delta de l'Ebre, però n'hi ha d'altres. Per exemple, el fet que l'*A. boyeri* no disposés d'una talla mínima de captura (Capítol I), degut a que la predicció per a una pesqueria sense regular és el seu col·lapse (Worm *et al.*, 2009). Alguns exemples de pesqueries que s'han col·lapsat recentment inclouen la sardina del Pacífic (*Sardinops sagax* (Jenyns, 1842)) a Califòrnia i Japó a finals dels anys 1940 o de l'anxoveta de Perú i Xile (*Engraulis ringens* Jenyns, 1842) l'any 1972 (Botsford *et al.*, 1997; Hutchings, 2000).

Una de les maneres més simples per a evitar el col·lapse de les poblacions és la regulació de la mida de la malla de les xarxes amb la que l'espècie és pescada (Wileman *et al.*, 1996; Stewart, 2008; Sterigiou *et al.*, 2009). D'aquesta manera s'incentiva un increment de la selectivitat i es redueix la captura accidental d'espècies no objecte (Worm *et al.*, 2009). El primer pas per a regular la mida de la malla de les xarxes, és conèixer la seva selectivitat. Per a estudiar la selectivitat de les arts de pesca usades al Delta de l'Ebre, varen emprar-se xarxes ganyeres i el mètode SELECT (sigles en anglès per a "compartir cada longitud de les captures totals"; Millar i Holst, 1997; Millar i Fryer, 1999; Millar, 2000). Aquestes xarxes varen escollir-se (tot i no ser les usades per als pescadors, Introducció general) degut a la seva semblança amb els tremalls (que són les xarxes usades) i al fet que són àmpliament utilitzades per a la recerca (Machelis *et al.*, 1994).

El mètode SELECT, és un model lineal generalitzat que assumeix una distribució Poisson de les captures de les xarxes ganyeres per a ajustar-les a un determinat model usant la màxima-versemblança (Millar, 2000). Implementat per Millar (Millar i Holst, 1997; Millar i Fryer, 1999; Millar, 2000), proporciona una aproximació coherent per a l'anàlisi de la selectivitat de mides. Les corbes de selectivitat són definides doncs com la probabilitat relativa d'un peix d'una determinada mida a ser capturat quan contacta una malla d'una determinada mida. Estudis previs han relacionat les corbes de selectivitat amb la manera com els peixos són capturats, tot basant-se en la classificació que en Baranov va fer l'any 1948. Ell, va reconèixer tres maneres principals en les que un peix pot quedar retingut a les xarxes ganyeres: a) encaixat b) agafat per les ganyes, o c) enredat. D'aquesta manera, els peixos que són encaixats o agafats per les ganyes serien millor ajustats amb models normals; mentre que els models esbiaixats, servirien per modelar els peixos que han quedat enredats a les xarxes (Hamley, 1975; Hovgård, 1996; Dos Santos *et al.*, 2003; Erzini *et al.*, 2003).

D'altre banda, els nostres resultats ressalten que el millor ajust de les corbes de selectivitat va estar relacionat amb la morfologia dels peixos, on espècies amb creixement més isomètric (*C. carpio*, *P. parva* o *A. alburnus*) varen aconseguir el seu millor ajust amb models normals; mentre que espècies amb creixement més al·lomètric (*C. auratus* i individus pertanyents a la família *Mugilidae*) o amb apèndixs corporals (*S. lucioperca* i *D. labrax*) varen aconseguir el millor ajust amb models esbiaixats com gamma, lognormal i invers Gaussià (Capítol I). Certament, aquests dos enfocaments són complementaris; degut a que la morfologia del peix influenciarà la manera com aquest és capturat (Reis i Pawson, 1999). Així, els resultats obtinguts, que relacionen la idoneïtat del model SELECT a la morfologia dels peixos, poden ser d'utilitat per a determinar la manera com aquests són capturats i les seves corbes de selectivitat millor ajustades.

Les corbes de selectivitat, un cop calculades poden usar-se per a preveure l'efecte que una regulació de la mida de malla pot tenir en la població de peixos (Psuty-Lipska *et al.* 2006). La regulació de les mides de malla és crucial alhora de protegir individus immadurs (Gulland, 1983; Wileman *et al.*, 1996; Millar i Holst, 1997). Diversos estudis recomanen l'establiment de les talles mínimes de captura -talla mínima en la que un peix pot ser capturat legalment- en conjunt amb estudis de selectivitat de xarxes (Wileman *et al.* 1996; Suuronen *et al.*, 2007).

Tenint en consideració el prèviament sostingut, l'enfocament d'aquesta tesi ha estat avaluar la selectivitat de les xarxes conjuntament amb la talla mínima de captura per a l'*A. boyeri*, tenint en compte també el seu procés de maduració (Capítol I). La conclusió que se'n pot extreure dels resultats obtinguts és que amb un increment de la mida mínima de la malla a 6.25 mm (actualment és de 5.00 mm), la població d'individus madurs d'*A. boyeri* seria propera al 100%, i permetria establir una mida mínima de captura de 54 mm de longitud total (LT). Aquesta proposta serà transferida als pescadors de la zona per a la seva possible aplicació.

La gestió uni-específica i més concretament les restriccions en la mida de malla, representen el primer pas per aturar una situació de poca vigilància de l'activitat de pesca. No obstant, degut a que la mortalitat dels peixos és un producte de l'esforç pesquer i de la selectivitat de les xarxes amb les que es pesca, les regulacions en la mida de malla no són suficients per a gestionar adequadament un estoc pesquer (Christensen *et al.*, 1996; Wileman *et al.*, 1996). A més a més, la captura dels peixos en si té un impacte a tot l'ecosistema. Per tant, les pesqueries no poden ser del tot ben gestionades si només es considera una sola espècie (FAO, 1995). Tot i això, el cert és

que la gestió multi-específica és molts cops difícil d'implementar degut a la dificultat d'obtenir suficient informació pel que fa a dades biològiques, ecològiques i d'interaccions tròfiques (Sainsbury *et al.*, 2000; Campos i Fonseca, 2003). A les llacunes costaneres del delta de l'Ebre, en comú amb altres llacunes costaneres mediterrànies tals com la llacuna de Venècia a Itàlia (Carrer i Opitz, 1999) o la llacuna de Thau a França (Palomares *et al.*, 1993), vàrem ser capaços de recol·lectar una llarga sèrie de dades de les captures de la zona, però ens mancava molta informació de la resta de les espècies. Per tant, un enfocament simplificat i més manejable per a la gestió multi-específica va realitzar-se (Capítol II) amb el software "*Ecopath with Ecosim*" (EwE; Christensen *et al.*, 2008; Coll *et al.*, 2009). D'entre les tres llacunes estudiades en la present tesi doctoral, va optar-se per construir un model de la llacuna de la Tancada per les raons següents: (i) la seva representativitat de tota la variabilitat present en les tres llacunes estudiades, (ii) l'aplicabilitat dels resultats obtinguts a la resta de les llacunes i (iii) la facilitat de modelització envers el sistema format per la llacuna de l'Encanyissada i el Clot. Degut a la connexió existent entre les llacunes de l'Encanyissada i el Clot (separades per un curt canal, Introducció General), en termes de modelització s'haurien de considerar com una sola unitat, fet que torna la tasca de modelització més complicada.

Els models Ecopath han estat àmpliament desenvolupats a l'àrea Mediterrània per entendre el funcionament i les interaccions tròfiques dels ecosistemes (Carrer i Opitz, 1999 i Palomares *et al.*, 1993). En aquest sentit, el model desenvolupat en la present tesi, ens ha permès entendre el funcionament i les interaccions tròfiques del nostre sistema, alhora que ha destacat el zoobentos, el joell i el gòbit de sorra com els grups més depredats, essent el primer d'aquests un grup molt important pel fet de sostenir dietes de nombrosos grups a l'ecosistema. Altres models desenvolupats a la Mediterrània també assenyalen el zoobentos com a un grup important a l'ecosistema (Brando *et al.*, 2004; Carrer i Opitz, 1999; Pinnegar i Polunin, 2004).

El mòdul Ecosim, va usar-se per a simular dues possible situacions futures a la llacuna de Tancada: una d'augment i l'altre de disminució del 50% de l'esforç pesquer actual. Mentre la simulació de l'augment de l'esforç pesquer va causar el descens de totes les espècies objecte menys les angules (juvenils *A. anguilla*; cas tractat més endavant); la simulació de la disminució de l'esforç va provocar l'efecte contrari tot i que diferent en dimensió. És a dir, el guany de biomassa quan l'esforç de pesca va ser disminuït va ser major que la pèrdua d'aquesta biomassa amb la simulació de descens d'esforç pesquer. Per tant, els resultats presentats suporten que una disminució gradual de l'esforç pesquer a la llacuna de la Tancada permetria una recuperació de les principals espècies a la llacuna.

El cas de *A. anguilla* (juvenils i adults) mereix una menció especial degut al seu descens a nivell mundial (FAO i ICES, 2006; IUCN, 2013). Amb el coneixement d'aquest estat crític en el que es troben les poblacions de *A. anguilla*, la Comissió Europea ha creat una Regulació (CE 1100/2007) en la que s'obliga als estats membres on aquesta espècie sigui nativa, a crear plans per a la seva recuperació global. L'objectiu principal del pla és permetre un retorn al mar de les anguiles d'almenys un 40% (valors de la població prístina). El pla desenvolupat a Espanya inclou 1 pla de gestió nacional i 12 regionals -11 plans autonòmics específics i 1 per a la conca del riu Ebre- (MARM, 2010). Aquest últim, inclou mesures per a la pesca professional com: restriccions d'arts de pesca, espacials i horàries per a la pesca de l'angula (<12 cm) i l'anguila adulta (> 35 cm); mentre que prohibeix la pesca de les anguiles compreses entre els 12 i els 35 cm. Per altra banda, també proposa mesures de repoblació a la zona (MARM i Generalitat de Catalunya, 2009; Taula 2 part en anglès).

Cal destacar que durant el desenvolupament de la regulació, que es troba en la seva primera fase d'execució (2010-2015); han sorgit alguns problemes en referència a l'estima de la biomassa real de les poblacions d'anguila (MARM i Generalitat de Catalunya, 2009). En aquest sentit, el model desenvolupat en la present tesi, és d'importància, ja que pot ajudar a dilucidar aquestes llacunes de coneixement. Per tant, és esperat que els resultats presentats puguin ser d'utilitat per a la proposta d'un pla de recuperació global.

No obstant, els esforços de la comunitat Europea de disminuir i controlar l'esforç pesquer sobre les angules, contrasten amb els resultats del nostre model que preveu un increment de les angules amb un increment de la pressió pesquera. *Està aquesta aparent paradoxa assenyalant un procés de denso-dependència com altres estudis han hipotetitzat amb anterioritat (De Leo and Gatto, 1996; Lobón-Cerviá and Iglesias, 2008; Bevacqua et al., 2011)?*. Serà necessari el desenvolupament d'estudis específics a l'àrea per tal de poder comprovar-ho.

Adicionalment, la construcció del model ens ha servit per a corroborar que l'ús d'arts de pesca com els bussons o la pantena, es basen en un profund coneixement de l'ecologia i el comportament dels peixos per part dels pescadors de la zona. Aquest fet, ha estat reportat en altres països com a Grècia, França, Tunísia i altres parts d'Espanya com el Mar Menor (De Sostoa and De Sostoa., 1985; Yañez-Arancibia et al., 1994; Koutrakis et al., 2005; Katselis et al., 2010; Pérez-Ruzafa i Marcos, 2012), i fa patent l'elevat coneixement ecològic que els pescadors tenen de la zona on operen. Conegut com Coneixement Ecològic dels Pescadors (Fisheries Ecological Knowledge (FEK)),

recentment s'ha remarcat la importància d'incloure'l en els plans pesquers per tal de gestionar els recursos de manera més sostenible (Freire i García-Allut, 1999, 2000; Neis *et al.*, 1999).

En resum, l'impacte de les pesqueries artesanals ha estat sobretot a dos nivells: el biològic i l'econòmic. Biològicament, la pesca ha provocat un descens de l'abundància de les espècies objecte, ha capturat individus immaturs i ha tret benefici dels moviments migratoris de les poblacions de peixos. Les conseqüències econòmiques venen derivades dels impactes a nivell biològic, ja que una eventual sobreexplotació dels recursos pesquers, provocada per exemple per la captura d'individus immadurs, pot causar pèrdues a nivell d'espècies comercials o disminuir el seu valor comercial degut a que no tenen la mida mínima per a ser venudes. Sostingut per els resultats trobats, les recomanacions a seguir per a desenvolupar una activitat pesquera més sostenible a les llacunes serien: (i) incrementar la mida mínima de les xarxes a 6.25 mm, (ii) seguir el principi de precaució de no incrementar o fins i tot si fos possible, disminuir l'actual esforç de pesca i (iii) establir plans de seguiment científic anuals de les captures/poblacions de peixos per tal d'avaluar l'adequació de les mesures proposades.

L'altre pressió antropogènica sobre les comunitats de peixos de les llacunes costaneres del Delta de l'Ebre és la **gestió de l'aigua** duta a terme per a permetre el cultiu de l'arròs a la zona. L'enfoc seguit en aquesta tesi per a estudiar aquest impacte, va ser mostrejar la fracció d'individus de petita mida, que comprenen els individus residents i els juvenils de les espècies migradores (Capítol III). Aquest enfoc va ser l'escollit degut a que la capacitat de moviment de les espècies de major mida i dels adults de les espècies migradores no els fa fiables com a descriptors de la distribució espacial (Mariani, 2001). A més a més, analitzant només les espècies de petita mida, no es consideren les espècies comercials i d'aquesta manera podem isolar l'efecte de la pesca. L'única excepció és l'*A. boyeri*, una espècie de petita mida que és comercial a la zona. No obstant, degut a que a la pesca a les llacunes es duu a terme només durant quatre mesos a l'any (octubre-març; Capítol III i Introducció general), pot considerar-se que existeix un període similar a una veda de pesca.

Com estudis precedents han reportat, la transformació duta a la plana deltaica per a permetre el cultiu del cereal - construint una extensa xarxa de canals que nodreixen els camps amb aigua fluvial -, ha originat l'artificialització del cicle hidrològic (De Sostoa i De Sostoa., 1985; Comin *et al.*, 1987; Forés *et al.*, 2002; Curcó, 2006). Aquesta artificialització, per la seva banda ha tingut repercussions econòmiques (construcció i manteniment dels canals, rescat de peixos atrapats), físiques (inversió del cicle

hidrològic de les llacunes, alteració de l'hàbitat) i biològiques (canvi dels patrons migratoris dels peixos, alteració de la composició de la ictiofauna).

A nivell físic, com ja s'ha comentat, el canvi més notable ha estat la inversió del cicle hidrològic natural de les llacunes; que ara reporta estius amb aigua dolça i hiverns d'aigua salada (De Sostoa i De Sostoa., 1985; Comin *et al.*, 1987; Forés *et al.*, 2002; Curcó, 2006). Aquesta artificialització del cicle de l'aigua, ha afectat a les comunitats biològiques que hi viuen (detallat més endavant). La deterioració de la qualitat de l'aigua també ha estat un fet notable i sol ser comú en llacunes costaneres afectades per activitats d'agricultura, degut a l'elevat nombre de nutrients, pesticides i herbicides usats per al seu conreu (Cloern 2001; Lucena *et al.*, 2002; Curcó, 2006; Badosa *et al.*, 2007). Concretament, l'excessiu abocament de nutrients a les llacunes va provocar episodis d'eutrofització en el passat, que van resoldre's amb dues mesures principals: primer amb la construcció de canals de circumval·lació que van evitar que la major part de l'aigua de drenatge provinent dels arrossars passés a través de les llacunes (Menéndez *et al.*, 1995; Forés *et al.*, 2002) i segon millorant la qualitat de l'aigua de riu (Nebra *et al.* 2011; Ibáñez *et al.* 2012a,b). Com a resultat, actualment l'aigua que entra a les llacunes és de major qualitat (*i.e.* és una barreja d'aigua provinent del riu i dels camps de cultiu), però encara conté quantitats elevades de nutrients. La solució passa per la reducció dels nutrients i herbicides usats, o com altres autors han proposat, promovent la transformació d'arrossars que ja no siguin operatius, per a transformar-los en zones pantanoses que puguin actuar com a "filtres verds", i d'aquesta manera millorar la qualitat de l'aigua (Ibañez *et al.*, 1999; Comín *et al.*, 2001).

A nivell biològic, tot i que no existeixen registres escrits en referència a la composició i abundància de les espècies de peixos abans de la "revolució de l'arròs", quan comparant l'actual composició de peixos amb altres llacunes costaneres que no pateixen alteració hidrològica (*i.e.* climàtic-depenents; Pérez-Ruzafa *et al.*, 2006; Maci i Basset, 2009; Verdiell-Cubedo, 2009), els nostres resultats destaquen que l'esmentada alteració hidrològica pot ser la responsable per a l'actual composició de peixos a les llacunes, de la modificació de la seva estructuració al llarg del gradient de salinitat i de l'augment d'espècies introduïdes. Les comunitats de peixos de les llacunes costaneres estan composades per famílies d'espècies marines i salobres (Gobiidae, Mugilidae, Atherinidae) i famílies d'espècies d'aigua dolça (Cyprinidae i Poecilidae).

Pel que respecta a l'estructuració al llarg del gradient de salinitat, va trobar-se que rangs de salinitats de 20-25 varen ser els preferits per a espècies marines (*L. aurata*, *L. saliens* i *C. labrosus*), mentre que les espècies salobres varen preferir rangs al voltant

de 10-20 (*A. boyeri*, *P. microps* i *A. iberus*) i l'espècie al·lòctona d'aigua dolça (*G. holbrooki*) va trobar-se a salinitats per sota de 5. En conseqüència, es recomana l'establiment de salinitats majors a 15-20 durant tot l'any. És esperat que l'adopció d'aquestes salinitats, permetrà obtenir una comunitat de peixos amb dominància d'espècies natives, importants des del punt de vista comercial i conservacionista.

La presència d'espècies introduïdes va semblar ser perjudicial per a algunes poblacions d'espècies natives, com el reportat impacte negatiu que l'invasora *G. holbrooki* provoca sobre l'amenaçada *A. boyeri* (Caiola i De Sostoa, 2005). A més a més, la condició al·lòctona de les espècies d'aigua dolça, ha alterat la ràtio d'espècies introduïdes/natives. Addicionalment, la ràtio d'espècies introduïdes/natives va semblar estar relacionada amb la gestió de l'aigua aplicada a cada una de les tres llacunes estudiades, reflectint alhora diferents graus d'alteració hidrològica. D'aquesta manera, la llacuna de la Tancada (que rep la menor quantitat d'aigua dolça durant el cultiu de l'arròs) va ser la que va registrar les proporcions més elevades d'espècies natives no només durant els períodes de més abundància d'aigua salada, sinó també en els d'aigua dolça. Llavors, podem concloure que la millor gestió és aquella aplicada a la llacuna de la Tancada, caracteritzada per menys entrades d'aigua dolça durant el període de cultiu de l'arròs (maig-desembre). La recomanació a seguir seria aplicar aquest mateix model de gestió a les altres dues llacunes estudiades, l'Encanyissada i el Clot.

Basades en els resultats obtinguts, les recomanacions a seguir per a desenvolupar una gestió més sostenible de l'aigua al les llacunes costaneres del Delta de l'Ebre són: (i) aplicar les polítiques que es duen a terme actualment a la llacuna de la Tancada, caracteritzades per menys aportacions d'aigua dolça durant el període de cultiu d'arròs a les llacunes de l'Encanyissada i el Clot i (ii) establir salinitats mínimes superiors a 15-20 al llarg de tot l'any.

Els resultats obtinguts en aquesta tesi han elucidat l'estat de l'art de les llacunes costaneres del delta de l'Ebre i han caracteritzat els impactes que dues de les principals activitats dutes a terme a la zona: les pesqueries artesanals i la gestió de l'aigua provoquen en les comunitats de peixos. Tres mesures clau per a l'assoliment d'un model de gestió més sostenible a les llacunes costaneres del Delta de l'Ebre són proposades: (i) incrementar la mida mínima de malla de les xarxes usades a 6.25 mm, (ii) reduir l'esforç de pesca actual i (iii) establir salinitats mínimes de 15-20 al llarg de l'any. Així mateix és recomana un seguiment anual de les comunitats de peixos a les llacunes per tal de

garantir l'efectivitat de les mesures proposades i permetre així el seu funcionament a llarg termini.

2.4 Conclusions

1. La composició actual de la ictiofauna a les llacunes del Delta de l'Ebre difereix considerablement de la existent als anys vuitanta.
2. La idoneïtat del model SELECT va estar relacionada amb la morfologia dels peixos. Els peixos amb creixement menys al·lomètric (*i.e.* més isomètric) varen ser millor ajustats amb models "normal scale", mentre que els peixos amb creixement més al·lomètric o amb més apèndixs en el seu cos van ser millor ajustats amb models "skewed" (*i.e.* esbiaixats) com el gamma, el lognormal i l'invers Gaussià.
3. La gestió de les pesqueries a les llacunes del Delta de l'Ebre sembla ser inadequada per espècies com el jovell (*Atherina boyeri*). L'enfocament de combinar estudis de selectivitat i de maduresa sexual va ser d'utilitat per establir noves mesures de gestió per a l'espècie.
4. Les anàlisis de la selectivitat de xarxes varen servir per dilucidar que l'actual mida de malla usada per pescar l'*Atherina boyeri* no és la més adequada; així les malles de mida menor (actualment 5.00 mm) haurien de ser augmentades a almenys 6.25 mm per a permetre l'establiment d'una talla mínima de captura de 54 mm LT (L_{75}).
5. La construcció del model amb "Ecopath with Ecosim" (EwE) ens ha permès identificar l'estructura i les principals interaccions tròfiques per als 18 grups funcionals definits a la llacuna de la Tancada. El zoobentos, el jovell i el gòbit de sorra varen ser els grups que presentaren major pressió de predació a la llacuna.
6. Els resultats obtinguts amb el model permeten confirmar la hipòtesi que les pesqueries artesanals a la zona aprofiten els moviments migratoris d'algunes de les espècies de la llacuna com el llobarro (*Dicentrarchus labrax*), l'anguila adulta (*Anguilla anguilla*) i l'orada (*Sparus aurata*).
7. Les simulacions dutes a terme amb l'Ecosim suggereixen una disminució gradual de l'esforç pesquer a la llacuna de la Tancada com la millor estratègia de gestió, assegurant conjuntament la sostenibilitat de la pesca i la conservació de les llacunes costaneres del Delta de l'Ebre.
8. Les mesures implementades per a millorar la qualitat de l'aigua a les llacunes varen assolir-se als anys noranta. Tanmateix, les entrades d'aigua dolça

continuen essent un focus de preocupació degut a l'elevada concentració de nutrients associada a aquestes.

9. L'alteració hidrològica de les llacunes costaneres del Delta de l'Ebre caracteritzada per entrades d'aigua dolça durant el procés de cultiu de l'arròs, té un gran impacte en la ictiofauna a diferents nivells: incrementant la presència d'espècies d'aigua dolça (majoritàriament espècies introduïdes), modificant l'estructuració de la població de peixos al llarg d'un gradient de salinitat, i alterant la composició de la comunitat de peixos.
10. L'actual esquema de gestió de l'aigua a les llacunes costaneres ha afectat negativament les espècies natives i n'ha afavorit les introduïdes. Les espècies natives amenaçades com l'*Aphanius iberus* i l'*Anguilla anguilla* o vulnerables com és el cas de la *Salaria fluviatilis* i l'*Atherina boyeri*, mereixen una especial consideració.
11. Les diferents polítiques de gestió aplicades a les tres llacunes costaneres, han comportat diferents nivells de pertorbació hidrològica.
12. La millor política de gestió a seguir per a la comunitat de peixos ha estat la duta a terme a la llacuna de la Tancada, caracteritzada per un nombre menor d'entrades d'aigua dolça durant el període de cultiu de l'arròs que han permès assolir salinitats al voltant dels 15-20 durant tot l'any. L'adopció de polítiques similars, serien recomanables per a les llacunes de l'Encanyissada i el Clot per tal de poder mantenir nivells de salinitat adequats per a sustentar les espècies natives, importants tant des del punt de vista comercial com conservacionista.

PUBLICATIONS SUMMARY (catalan)

3

Article I. Selectivitat de malla de les xarxes ganyeres a les llacunes costaneres del Delta de l'Ebre i la seva implicació per a la gestió de la pesca del jovell, *Atherina boyeri* (Actinopterygii: Atherinidae)

Paraules clau: pesqueries artesanals, xarxes ganyeres, gànguils, regulacions de la mida de la malla, mètode SELECT, talla mínima de captura.

L'objectiu d'aquest estudi va ser prevenir un possible estat de sobrepesca de la nostra espècie d'estudi: el jovell, *Atherina boyeri* Risso, 1810. Per a tal efecte, van considerar-se la regulació de la mida de la malla i l'elecció de l'art de pesca més adequat per a la seva captura. Els objectius específics de l'estudi varen ser: (i) modelar la selectivitat de malla de les xarxes ganyeres usant el mètode SELECT, com a primer enfoc per a descriure els patrons de selectivitat de les xarxes ganyeres per a vuit espècies d'interès comercial a tres llacunes costaneres del Delta de l'Ebre; (ii) Combinar la informació obtinguda amb les xarxes ganyeres i els gànguils per tal d'establir una talla mínima de captura per al jovell. L'àrea d'estudi elegida van ser tres llacunes costaneres del Delta de l'Ebre on es van establir diferents punts de mostreig. En cadascun d'ells es va mostrejar amb dues arts de pesca diferents: els gànguils i les xarxes ganyeres. Aquestes últimes eren xarxes multi-malla compostes per dotze panells amb llum de malla diferents (amb obertures de malla compreses entre 5.0 i 55.0 mm). El mètode SELECT (sigles en anglès per a "compartir cada longitud de les captures totals") va ser usat per estudiar les corbes de retenció mitjançant cinc models diferents: "model location", "normal scale", "gamma", "lognormal" i "Inverse Gaussian". Cada model va ser ajustat dues vegades, sota les assumpcions d'esforç de pesca igual o proporcional a la mida de la malla. No varen trobar-se diferències entre els diferents enfoc utilitzats per a l'anàlisi de la selectivitat de malles. Cada espècie es va ajustar millor a un model

diferent, no trobant-se per tant un patró general. Els resultats suggereixen però, les proporcions del cos del peix com a determinants per a la selectivitat de malla. L'explotació amb gànguils i amb xarxes ganyeres amb obertures de malla per sota dels 6.25 mm van resultar no ser sostenibles per a l'explotació del jove. Es proposa l'ús de xarxes amb una obertura mínima de 6.25 mm que permetin establir una talla mínima de captura de 54 mm de longitud total (L_{75}).

Article II. Avaluació de l'impacte de les pesqueries artesanals en una llacuna costanera Mediterrània usant el model de masses balancejades Ecopath amb Ecosim

Paraules clau: Modelització ecosistèmica, Ecopath, Ecosim, comunitat de peixos, xarxa tròfica.

Es va construir un model tròfic de masses balancejat per a la llacuna de la Tancada (Delta de l'Ebre, NE Espanya). El software utilitzat per a la construcció del model va ser "Ecopath with Ecosim" (EwE), un model àmpliament usat per avaluar l'estat d'explotació dels sistemes aquàtics i que en la última dècada ha experimentat un increment exponencial en el seu ús en tot tipus d'ecosistemes aquàtics (marins, estuarins, lèntics i lòtics). Els objectius específics d'aquest estudi varen ser (i) construir un model de xarxa tròfica (trophic-web model) per a la llacuna de la Tancada usant l'aplicació Ecopath, (ii) avaluar els impactes de diferents esforços de pesca a l'ecosistema i (iii) simular escenaris amb esforços de pesca diferents a l'actual amb l'aplicació Ecosim per tal de poder dissenyar una estratègia d'explotació sostenible a llarg plaç dels recursos pesquers de la llacuna de la Tancada. El model estava compostat per 18 grups funcionals, incloent peixos, productors primaris, invertebrats, ocells i detritus. Les dades referents a la biomassa, als quocients de producció/biomassa i consum/biomassa, i composició de les dietes per cada grup funcional varen ser, depenent de cada cas, estimades o adaptades a partir de la bibliografia consultada. Les estadístiques de desembarcaments oficials per a la llacuna de la Tancada per al període comprés entre 1965 i 2010 varen ser facilitades per l'associació de pescadors de St. Pere i usades per calibrar les dades de captures amb l'Ecosim. Les dues simulacions, una pronosticant un augment (50%) i l'altre una disminució (50%) de l'actual esforç de pesca desenvolupat a les llacunes es varen dur a terme sense l'ajustament per a la sèrie de dades durant 10 anys.

Els resultats del model ens han ajudat a identificar l'estructura de la xarxa tròfica i de totes les interaccions biològiques que succeeixen a l'ecosistema. Alhora, els nostres resultats reforcen la hipòtesi que les pesqueries artesanals a la llacuna de la Tancada aprofiten els moviments migratoris d'algunes espècies de la zona com el llobarro (*Dicentrarchus labrax* (Linnaeus 1758)), l'anguila adulta (*Anguilla anguilla* (Linnaeus 1758)) i l'orada (*Sparus aurata* Linnaeus, 1758). Un augment de l'esforç de pesca a l'ecosistema provocaria disminucions en la biomassa de la major part de les espècies objecte (comercials), essent l'orada, el llobarro i l'anguila adulta les espècies més afectades. Per contra, una disminució de l'esforç de pesca faria augmentar la biomassa de les espècies objecte i disminuir la de les no objecte per efecte indirecte, alhora que provocaria un augment general de la biomassa de l'ecosistema. El model representa un important esforç per a integrar tota la informació biològica disponible a la zona d'una forma coherent. Algunes mancances i deficiències han estat detectades durant la construcció d'aquest, la correcció futura de les quals serà un pas endavant cap a una millor caracterització de l'ecosistema, així com una millor gestió de les pesqueries dutes a terme a la zona. Es recomana la reducció progressiva de l'actual esforç pesquer com a la millor estratègia a seguir per tal d'assegurar una pesca sostenible a la llacuna costanera del Delta de l'Ebre.

Article III. La salinitat com el principal factor estructurant la comunitat de peixos de petita mida en llacunes costaneres hidrològicament alterades

Paraules clau: comunitat de peixos, salinitat, gestió de l'aigua, Delta de l'Ebre, llacunes costaneres.

Una de les majors pressions antropogèniques sobre les llacunes costaneres del Delta de l'Ebre és l'entrada artificial d'aigua dolça. Cada llacuna costanera presenta mesures de gestió diferents que alteren profundament les seves propietats fisicoquímiques. Aquesta gestió de l'aigua té com a objectiu afavorir algunes espècies d'aus amb interès per a la caça i/o conservació. El present estudi avalua la influència de l'alteració hidrològica en la comunitat de peixos de tres llacunes costaneres del Delta de l'Ebre: l'Encanyissada, el Clot i la Tancada. D'acord amb el règim hidrològic artificial dut a terme a la zona, van escollir-se dos períodes de mostreig: març i setembre de 2008, representatius dels mínims i màxims aportats d'aigua dolça respectivament. A cada punt de mostreig van recol·lectar-se peixos i es van mesurar els descriptors ambientals. Els peixos varen ser recol·lectats amb gànguils, i les captures expressades com a Captures

per Unitat d'Esforç (CPUE). Els descriptors ambientals mostrejats varen ser: paràmetres fisicoquímics de l'aigua (oxigen dissolt (mgL^{-1}), temperatura ($^{\circ}\text{C}$), salinitat (psu) i pH); nutrients dissolts (fosfats (P-PO_4), nitrats (N-NO_3), nitrits (N-NO_2) i amoni (N-NH_4) medits en mgL^{-1}) i profunditat (cm). Els descriptors fisicoquímics entre llacunes i períodes van ser comparats mitjançant una anàlisi de la variança (ANOVA) seguit d'una anàlisi post-hoc Games-Howell (anàlisi GH). La variació de les espècies explicada per els paràmetres fisicoquímics va ser analitzada a través d'una anàlisi de correspondència canònica (CCA) i la resposta individual de les espècies a la salinitat va ser avaluada usant Models Additius Generalitzats (GAMs sigles en anglès per a "Generalized Additive Models"). Totes les anàlisis estadístiques van dur-se a terme amb els programes SPSS 17.0 i CANOCO 4.5. Van trobar-se diferències significatives entre les tres llacunes per als descriptors ambientals de temperatura, profunditat, pH i salinitat. La temperatura al març va resultar ser més freda que al setembre; la profunditat i el pH varen ser majors al setembre; mentre que la salinitat va ser significativament major al març. La fracció de la ictiofauna integrada per individus de petita mida va estar majorment composta per cinc famílies: Gobiidae, Poeciliidae, Cyprinodontidae, Atherinidae i Mugilidae. La salinitat va ser el factor que més va contribuir a estructurar la comunitat de peixos a les llacunes. Amb valors màxims de salinitat, l'espècie dominant a les llacunes va ser el burret (*Pomatoschistus microps* (Krøyer 1838)), mentre que la gambúsia (*Gambusia holbrooki* Girard, 1859), una espècie invasora, va ser dominant durant el període caracteritzat pels aportats d'aigua dolça. Els juvenils de la família Mugilidae presentaren valors baixos de CPUE, especialment durant el període de menor salinitat. El mateix patró va ser trobat per al fartet (*Aphanius iberus* (Valenciennes 1846)), espècie en perill d'extinció. En general, les espècies introduïdes van ser afavorides per una baixa salinitat, fet que remarca la importància de modificar l'actual gestió de l'aigua reduïnt els aportats d'aigua dolça. Una salinitat superior a 15-20 durant tot l'any, asseguraria una comunitat de peixos amb dominància d'espècies natives, importants des del punt de vista comercial i conservacionista. Aquest hauria de ser un criteri clau per establir un futur esquema de gestió de l'aigua a les llacunes costaneres del Delta de l'Ebre.

GENERAL INTRODUCTION

4

4.1 Coastal lagoons and their fish assemblages

Coastal lagoons can be defined as salt or brackish areas, usually shallow and relatively isolated from the open sea (Barnes, 1980; Kjerfve, 1989; Isla, 1995; Pérez-Ruzafa, 2007). Worldwide, they represent approximately 13% of coastal areas, and are found in every continent except Antarctica (Barnes, 1980). Among all the continents, coastal lagoons vary greatly in size: from as small as a hectare to more than 10.000 km², e.g. Lagoa dos Patos in Brazil (Kennish and Paerl, 2010).

Different physical factors such as water inlets, wind conditions, variation in precipitation and evaporation rates, surface runoff or local meteorological conditions characterize coastal lagoons (Kennish and Paerl, 2010); but in general coastal lagoons can be classified in three different geomorphic types according to water exchange with

the coastal ocean as follows: choked, restricted and leaky (Kjerfve, 1994; Fig. 1). **Choked** lagoons are series of elliptical cells connected by a single narrow channel. They are characterized by long flushing times, dominant wind forcing, intermittent stratification and by insignificant tides. **Restricted** lagoons are larger bodies of water, usually oriented shore-parallel and with two or more channels that connect them to the sea. They are also characterized by a well-defined tidal circulation, influenced by winds and with shorter flushing times than choked lagoons. **Leaky** lagoons are elongated shore-parallel water bodies with many ocean entrances. In this kind of lagoons, tides are very important and salinities are close to

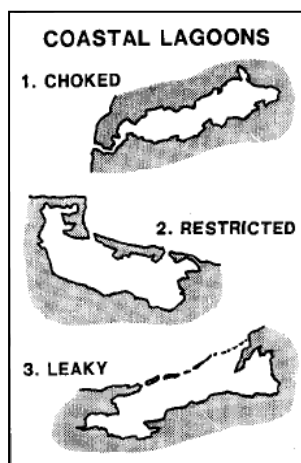


Figure 1. Geomorphic types of coastal lagoons (from Kjerfve, 1994).

the seawater (Kjerfve, 1994). Thus, the geomorphological classification of a lagoon is an indicator of the dominant forcing function(s) and the time-scale of hydrologic variability (Kjerfve, 1994).

Coastal lagoons are one of the most biologically productive systems on Earth (Kennish, 2002; Kennish and Pearl, 2010). Their high biological productivity is driven in part by its shallowness, which allows the photic zone to be extended to most of the seafloor, thus a high fraction of the primary production is supported by the benthic primary producers. Furthermore, because many shallow lagoon systems have relatively long water residence times, nutrient inputs can be recycled many times before their exit into sea environments (Kennish and Pearl, 2010). Because of this high productivity, in addition to the physical refuge from predation that they offer, coastal lagoons are used as a nursery and feeding habitats for a wide variety of organisms (Yañez-Arancibia *et al.*, 1994; Elliott *et al.*, 2007).

Fish are one of the taxonomic groups that take advantage of the high productivity and physical refuge of coastal lagoons. The narrow suite of fish species of a fish community that coexist in the same habitat at the same time (*i.e.* in a coastal lagoon) is known as a fish assemblage (FAO, 1997; Miller, 2002). In the Atlantic-Mediterranean region, Pérez-Ruzafa and Marcos (2012) reported 199 fish species in 40 coastal lagoons. The fish species richness in these lagoons ranged between 6 and 48, with a mean of 23.4 species (Pérez-Ruzafa and Marcos, 2012). The more common species (present in more than 50% of the lagoons) comprised marine migrants such as European sea bass (*Dicentrarchus labrax* (Linnaeus, 1758)), European eel (*Anguilla anguilla* (Linnaeus, 1758)), flathead grey mullet (*Mugil cephalus* Linnaeus, 1758), thinlip grey mullet (*Liza ramada* (Risso, 1827)), golden grey mullet (*Liza aurata* (Risso, 1810)), leaping mullet (*Liza saliens* (Risso, 1810)), gilthead seabream (*Sparus aurata* Linnaeus, 1758) and common sole (*Solea solea* (Linnaeus, 1758)), and the brackish species sand smelt (*Atherina boyeri* Risso, 1810). At family level, most dominant families in Atlantic-Mediterranean coastal lagoons include: Mugilidae, Sparidae, Gobiidae, Atherinidae, Soleidae, Syngnathidae, Anguillidae, Moronidae and Blennidae (Pérez-Ruzafa *et al.*, 2006, 2007, 2011; Maci and Basset, 2009, Verdiell-Cubedo, 2009); and among all these families, Mugilidae and Sparidae constitute the main catch in the coastal lagoons worldwide (Pérez-Ruzafa *et al.*, 2012).

Coastal lagoons are also very important from the economical point of view. Fisheries are one of the many ecosystem services and resources that lagoons can provide (Kennish and Paerl, 2010). Rich recreational and commercial fisheries are granted by

the high productivity from both water column and benthos, which allow coastal lagoons to support quantitatively important food chains in the benthic and detrital compartments (Alongi, 1998). In this sense, coastal lagoons are generally more productive in terms of fisheries yields than any other aquatic ecosystem, registering fish productivity averages of about $100 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Yañez-Arancibia, 1994). Aside from fisheries, coastal lagoons also provide high amount of goods (food resources, mineral and medicinal resources, wildlife, source of water for domestic, agricultural or industrial use, provision of land for agriculture by deposition of fertile soils) and services (tourism/recreation, nutrient mixing, biodiversity, nursery of fishes, hydrological maintenance, educational opportunities, sediment retention, habitat provision) to humans, which have made them prone to be historically inhabited (Gonenç and Wolfin, 2004 ; Seeram, 2008).

4.2 Main threats to coastal lagoons and their fish assemblages

The interface position of coastal lagoons -situated between continental and marine ecosystems- makes them very fragile and sensitive environments to both **natural** and **anthropogenic** stressors (Gonenç and Wolfin, 2004; Pérez-Ruzafa, 2005; Kennish and Pearl, 2010). Stressors are defined as natural or anthropogenic disturbances, depending on the main originating force. However, it must be taken into consideration that some natural stressors can be magnified by human activity. The main causes and effects of those disturbances are listed below.

- *Natural stressors*

Although historically natural stochastic events have received less attention than anthropogenic stressors; the consequences of the natural perturbations might be of higher dimensions (Paerl *et al.*, 2009). Among all environmental factors acting as stressors, coastal lagoons are highly dependent on wind stress, tide oscillation, sea-level rise, evaporation-precipitation balances, surface heat balances, climate conditions and climate-change (Kennish and Paerl, 2010). **Wind forcing**, reported as a key element in the wave genesis on coastal lagoons, affect differently each lagoon type (Kjerfve, 1989; Kennish and Paerl, 2010). In restricted and leaky lagoons, the wind propagates readily into the system and can cause large changes in water level and currents; whereas in choked lagoons wind effect is most often due to local winds which can promote current flows, flooding of adjacent low-lying lands and either enhance or retard gravitational fluctuation (Kjerfve, 1989). **Tides** in coastal lagoons are driven by tides in the coastal

ocean and give rise to non-linear flow dynamics within the entrance channel (Kjerfve, 1989). In restricted and leaky lagoons, flow variability is mostly due to oscillating tidal currents. In choked lagoons, however, the entrance channel serves as a dynamic filter that alternates or eliminates currents and water level fluctuations (Kjerfve, 1989). Thus, depending on the geomorphological lagoon type, the intensity of wind effect and the magnitude of the tides would be greatly different. Coastal lagoons are also susceptible to **sea-level rise** because their vertical accretion rates are limited (FitzGerald *et al.*, 2008). The Intergovernmental Panel on Climate Change (IPCC) predicts a sea-level rise of 40 cm by year 2100. General predicted effects of a plausible sea-level rise scenario in coastal lagoons include: inundation of low-lying coastal regions, redistribution of sediment along sandy coasts, alteration in salinity gradients and alteration in inter-tidal organisms distribution and abundances (Morris *et al.*, 2002; FitzGerald *et al.*, 2008). However, the magnitude, rate and effects of relative sea-level rise will vary regionally because of differences in subsidence (resulting from sediment loading, groundwater withdrawal or oil extraction), tectonic uplift, isostatic rebound, and the compactation of muddy soils (Anthony *et al.*, 2009).

Evaporation-precipitation and **heat balances** are also very important natural stressors in coastal lagoons. Equilibrium between evaporation and precipitation processes will determine periods of drought or strong rainfalls; whereas heat balances might induce stratification. Both processes will influence ecological functioning of ecosystems by altering the natural hydrologic cycle (Kjerfve, 1994; Kennish and Paerl, 2010). Coastal lagoons are also notably influenced by **regional climate conditions** (Kjerfve, 1994). For instance, Patos Lagoon in Brazil is highly influenced by the El Niño Southern Oscillation (ENSO) phenomenon, that alters precipitation patterns (freshwater discharges) and long-term salinity regimes in the area (Garcia *et al.*, 2001; Kennish and Paerl, 2010). Those changes seem to play a major role in life cycle patterns, recruitment, growth, biomass and dynamics (immigration-emigration) of most fish species (Garcia *et al.*, 2001; Kennish and Paerl, 2010). In Mediterranean lagoons, such as the case of the Venice lagoon in Italy, wind, flooding events and water temperature were reported to be the most influent stressors (Kennish and Paerl, 2010). Moreover, under a **global climate change** scenario, the dynamic occurrence of some of the already commented natural stressors (sea-level rise, temperature and precipitation) are expected to suffer significant changes that would directly impact on coastal lagoons with uncertain consequences (FitzGerald *et al.*, 2008; Anthony *et al.*, 2009; Kennish and Paerl, 2010).

To sum up, the impact that natural stressors can have on coastal lagoon dynamics seems to be dependent on their intrinsic characteristics such as their geomorphology

(choked, restricted and leaky) (Kjerfve, 1994; Kennish and Paerl, 2010). Moreover and as commented before, some natural stressors can be modified by human intervention. As a case in point, physical stress due to high residence time can have a natural origin, but its effects or pressure can be changed by anthropogenic activity such as the construction of a dam or the digging up of a canal to increase water circulation (Gamito *et al.*, 2008).

- *Anthropogenic stressors*

As a result of coastal lagoons goods and services (over)exploitation by human population, many of these systems are deteriorating (Gonenç and Wolfin, 2004). The principal anthropogenic stressors and their major effects in estuarine areas (including coastal lagoons) are listed in Table 1. A brief description of the main causes and consequences of those stressors is presented below.

a) Habitat loss and alteration. Usually fomented by physical factors (*e.g.* dredging, shoreline modification and wetland reclamation); habitat alteration creates persistent and serious environmental problems, such as large-scale modifications of watersheds and estuarine basins that affect inhabiting organisms (Kennish, 2002; Kennish and Pearl, 2010). However, not all the anthropogenic activities and structures affect equally coastal habitats; for example, poorly planned domestic and industrial construction can increase the non-point source pollution, whereas the construction in catchments may hasten sediment loading that can impact benthic communities (Kennish, 2002). Nevertheless, sediment retention occasioned by dam construction has been recognized as a very serious problem in many deltas of the world, suffering from regression (Ibáñez *et al.*, 1997; Kennish, 2002; Vörösmarty *et al.*, 2003). In the Ebro Delta case, dam sediment retention has implied the delta's growth arrest (Ibáñez *et al.*, 1997). Furthermore, habitat loss due to land reclamation for agriculture and other human needs results in critical habitat and economic losses, as eliminates valuable finfish spawning, feeding and nursery grounds (Kennish, 2002).

b) Eutrophication. Understood as the process by which a body of water acquires high concentration of nutrients, especially phosphates and nitrates, eutrophication typically promotes excessive growth of algae (Art, 1993). As the algae die and decompose, high levels of organic matter deplete the available oxygen in the water, inducing hypoxia (Art, 1993). Hypoxia induces general decrease of biodiversity by reducing abundance of specific fish and other animal population (Horrigan *et al.*, 2002). In addition,

eutrophication can promote the growth of toxic algal blooms, which can be lethal for many organisms, including humans (Anderson *et al.*, 1994). Although eutrophication is a natural process, it has been included in this section as human activity can accelerate the release of nutrients. Several examples of human activities promoting the release of high amount of nutrients include: agricultural activities (*i.e.* fertilizers, runoff from agricultural fields, chemical fertilizers, manure), aquaculture, urban and industrial areas (*i.e.* municipal wastewater, treatment plants) and fossil fuel combustion (*i.e.* release of nitrogen to the atmosphere) (Selman and Greenhalgh, 2009).

Lagoon systems characterized by restricted water circulation, poor flushing, shallow depths and heavily populated watersheds will be more prone to suffer nutrient enrichment impacts (Kennish, 2002). Although choked lagoons (with less water circulation) are the more susceptible to suffer from eutrophication (*e.g.* Patos lagoon, Brazil or Mar Menor in Spain), restricted lagoons (*e.g.* Tancada lagoon, Spain) and leaky lagoons (*e.g.* Venice lagoon, Italy; Wadden sea, the Netherlands) are not exempt either of suffering from this stressor (Forés *et al.*, 2002; Suman *et al.*, 2005; Pérez-Ruzafa *et al.*, 2005; Kennish and Paerl., 2010).

c) Sewage and organic wastes. Sewage and organic wastes are both refuse matter. Sewage is understood as the refuse matter from domestic or industrial establishments carried away in sewers; whereas organic waste constitutes just the organic part of the refuse matter (*i.e.* derived from animal and plant materials). Point source discharges from industrial users or manufacturers, and diffuse discharges from commercial and domestic premises, contribute to the organic loading in sewage (Rogers, 1996). Adverse effects associated with an increase of sewage and organic wastes are: periodic toxic or nuisance algal blooms, shading effects, build up of toxins (*e.g.* sulphides), mortality of benthic and pelagic species, reduced biodiversity, diminished secondary production, diminution of recreational and commercial fisheries, as well as altered species composition, organisms abundance and distribution (Costello and Read, 1994; Alongi 1998; Holmer, 1999; Howarth *et al.*, 2000; Kennish, 2002). Moreover, the release of untreated or partially treated sewage can significantly increase nutrient levels in coastal lagoons. This waste input also degrades water quality by raising the biochemical oxygen demand, promoting eutrophication and delivering chemical contaminants and pathogenic microorganisms (*i.e.* bacteria, viruses, protists and helminthes) to the systems (Chapman *et al.*, 1996; Kennish, 1997; Eganhouse and Sherblom, 2001). Pathogens from sewage pose a serious threat to human health, being responsible for cholera, hepatitis, as well as gastroenteric diseases; hence, they

are targets in water quality monitoring programs for the assessment of the closure of bathing beaches and shellfish growing waters (Kennish, 2002).

d) Fisheries overexploitation. It is well-known that one of the main consequences of fisheries resources overexploitation is a decline in finfish and shellfish populations of recreational and commercial importance (Botsford *et al.*, 1997). Moreover, since

Table 1. List of anthropogenic stressors on estuarine environments.

Stressor	Impact
a) Habitat loss and alteration	Elimination of usable habitat for estuarine biota
b) Eutrophication	Overgrowth of primary producers, harmful algal blooms, hypoxia and anoxia, increased benthic invertebrate mortality, fish kills, altered community structure, increased turbidity and shading, reduced seagrass biomass, degraded water quality
c) Sewage and organic wastes	human pathogens delivery and overgrowth, increased nutrient and organic matter loading, increased eutrophication, increased hypoxia, degraded water and sediment quality, reduced biodiversity
d) Fisheries overexploitation	Depletion or collapse of fish and shellfish stocks, altered food webs, changes in the structure, function, and controls of estuarine ecosystems
e) Sea-level rise	Shoreline retreat, loss of wetlands habitat, widening of estuary mouth, altered tidal prism and salinity regime, changes in biotic community structure
f) Chemical contaminants	Adverse effects on estuarine organisms including tissue inflammation and degeneration, neoplasm formation, genetic derangement, aberrant growth and reproduction, neurological and respiratory dysfunction, digestive disorders and behavioral abnormalities; reduced population abundance; sediment toxicity
g) Freshwater diversions and water management	Altered hydrological, salinity, and temperature regimes; changes in abundance, distribution, and species composition of estuarine organisms
h) Introduced / invasive species	Changes in species composition and distribution, shifts in trophic structure, reduced biodiversity, introduction of detrimental pathogens
i) Subsidence	Modification of shoreline habitat, degraded wetlands, accelerated fringe erosion, expansion of open water habitat
j) Sediment input / turbidity	Habitat alteration, reduced primary production, shading impacts on benthic organisms
k) Floatables and debris	Increased mortality of seabirds, marine mammals, reptiles, and other animals

(from Kennish and Paerl, 2010).

fisheries are size selective, removing preferentially larger fish, overfishing modify size (*i.e.* reducing average size, size-at-age and size-at-maturity), age structure and genetic diversity of targeted species (Botsford *et al.*, 1997; Goñi, 1998). Along history, there are numerous examples of stocks abundance reduction until population collapse such as the case of Pacific sardine (*Sardinops sagax* (Jenyns, 1842)) in California and Japan in the late 1940's or the anchovy off Peru and Chile (*Engraulis ringens* Jenyns, 1842) in 1972 (Botsford *et al.*, 1997; Hutchings, 2000). While many of these consequences have been demonstrated in open sea fisheries, in coastal lagoons (maybe because fishing remains largely artisanal; Kennish and Paerl, 2010), no clear effects of overfishing have been detected (Pérez-Ruzafa and Marcos, 2012). Since fisheries impact in coastal lagoons is a main subject of this thesis, it will be described in detail in **section 2.3**).

e) Sea-level rise. Global warming fomented by human activities such as burning oil and coal, cutting down tropical forests (*i.e.* increased concentrations of heat-trapping gases), has been recognized as the main cause for sea warming in the last century. With a warmer sea, land ice (*i.e.* glaciers, ice caps and ice sheets) melts at higher rates than it grows, adding water to the oceans and causing a global sea-level rise (Nicholls and Cazenave, 2010; Union of Concerned Scientists, 2013). Along history, periods of sea-level rise and accretion had allowed the formation of numerous coastal lagoons (Kjerfve, 1994). However, the formation process is dependent on the achievement of an accretionary balance, which can be attained if: (i) the long-term filling rate exactly equals the relative sea-level rise, or (ii) the lagoon fills rapidly to capacity (*i.e.* catches up to the rate of sea-level rise) and the accretion surface remains at the base level for a long time (Kjerfve, 1994). Nonetheless, already in 1994, Kjerfve prognosticated that accretion would not be sufficient to keep pace or maintain the equilibrium with the sea-level rise during next century. Kjerfve was not very inaccurate in his proposal, and according to Eisma (1998), relative sea-level rise along the Louisiana coast (from a net of 1-2 mm yr⁻¹ to 1cm yr⁻¹ or more during the XXth century), accounted for an annual loss of up to 73 km² of wetlands area (Kennish, 2002). Moreover, it is estimated that in the Northern Gulf of Mexico 60% of the wetland loss can be attributed to this process (White and Tremblay, 1995). This global sea-level change, with the associated warm water, would foment changes in species composition and distribution of both flora and fauna in wetland ecosystems (Kennish, 2002).

f) Chemical contaminants. Chemical contaminants can be divided in three types depending on the potential effect over organisms, and listed in order of danger as follows: (i) polycyclic aromatic hydrocarbons (PAH's), (ii) halogenated hydrocarbons,

and (iii) heavy metals (Mañosa *et al.*, 2001; Kennish, 2002). Anthropogenic activities that promote release of PAH's include oil spills, municipal and industrial discharges, marinas, fossil-fuel combustion and waste incineration; whereas insecticides, herbicides and industrial chemicals that enter into the system through industrial and municipal wastewater, urban and farmland runoff or atmospheric deposition, are sources of halogenated hydrocarbons (Kennish, 2002). Further, heavy metals are usually derived from anthropogenic activities such as mining and smelting operations, refining and electroplating, dye and paint manufacture and fossil-fuel burning (Kennish, 2002).

Organisms exposed to high levels of chemical contaminants exhibit an array of adverse responses. As a result, biotic communities affected by the contaminants can experience significant changes, including the loss of rare or sensitive species, decreased species abundance, shifts in the age structure of populations, altered physiology and altered trophic interactions. These changes can disrupt entire ecosystems. Moreover, some of the pollutants tend to bioaccumulate in aquatic organisms, and thus biomagnifications could take place in food chains, reaching the highest concentration in top-predators (Kennish, 2002). Certain toxic substances as dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyls (PCB's) are a real threat to humans that could consume contaminated organisms (*i.e.* seafood) (Kennish, 2002).

g) Freshwater diversions and water management. There are many processes that alter freshwater main courses, and consequently alter hydrologic dynamics of coastal lagoons (described in detail in **section 2.4**, as the other main anthropogenic stressor affecting coastal lagoons studied in the present thesis). One of them is the increase of urban development in coastal areas, which leads to a greater impervious cover that accelerates storm water runoff. Other common processes that alter natural freshwater courses are canalization; usually done to satisfy agricultural, municipal or industrial demands or just for flood control over those activities (Kennish, 2002; Badosa *et al.*, 2007). Other human activities such as salt production, construction of catchment and retention structures (*e.g.* dams) and tourism (*i.e.* increase of wastewater discharges), may also alter coastal lagoons hydrological dynamics (Ambrose and Meffert, 1999; Day Jr. *et al.*, 2002; Lucena *et al.*, 2002; Lenzi *et al.*, 2003; De Stefano, 2004; Seeram, 2008).

Impacts over the ecosystem may be acute, as an alteration in a water inflow can change salinity patterns and sediment regimes, as well as nutrient and pollutants loadings, which directly affects habitat areas and organism's abundance and

distribution (Kennish, 2002). In Mediterranean coastal lagoons, some studies have already demonstrated the strong influence of hydrology on nutrient dynamics, water quality and zooplankton community (Comín *et al.*, 1987; Oltra and Miracle, 1992; Quintana *et al.*, 1998; Ortega *et al.*, 2000).

h) Introduced invasive species. Species introduction is understood as the intentional (deliberate) or unintentional (accidental) process by which some species is imported into a region beyond its native range. Introduced species turn into invasive when they spread from the point of introduction and became abundant (Richardson, 2000; Kolar and Lodge, 2001).

Invasive species lead to a reduction of genetic biodiversity and biotic homogenization through a variety of mechanisms (Rahel, 2000). They also promote shifts in trophic organization, infiltration of detrimental pathogens and alteration of habitats (Kennish, 2002). Moreover, and due to the lack of natural control in the areas where they are introduced, invasive species often outcompete with native species to dominate communities, and might drive them to extinction (see review by Drake *et al.*, 1989). Human activities have accelerated intentionally and/or accidentally the introduction and distribution of certain species further than their natural environments (ICES, 1988; Courtenay and Robins, 1989; Winfield, 1992; Cowx, 1997). Some examples of invasive fish species in Europe are: freshwater bream (*Abramis brama*, Linnaeus 1758), common carp (*Cyprinus carpio* Linnaeus 1758), wels catfish (*Silurus glanis* Linnaeus, 1758) and European mosquitofish (*Gambusia holbrooki* Girard, 1859) (www.europe-aliens.org).

i) Subsidence. Subsidence is an irreversible process where the level of the lagoon is lowered with respect to the sea. Although geological subsidence is a natural process, subsidence due to anthropogenic activities has increased in recent times (Brambati *et al.*, 2003). Examples of subsidence problems can be found around the world. In Galveston Bay (Texas), land submergence were attributed to gas and oil withdrawal (Kennish *et al.*, 2002). In the Mediterranean area, Venice lagoon subsidence problems were reported to begin after the 2nd World War due to a boom in the groundwater withdrawals. Since then, a sunk of 12 cm has been observed (Brambati *et al.*, 2003; Suman *et al.*, 2005). Dam construction in Ebro's watershed during last century is considered the responsible of sediment loss. About 99% of the sediment is calculated to be retained by dams; being the largest ones in the lower Ebro River (Mequinensa and Ribaroja), the ones that retain most of the solid discharge (Ibáñez and Prat, 1996; Muñoz, 2013). In the Ebro Delta, wave erosion and this loss of new sediments are

inducing subsidence. In the ecosystem, subsidence process can trigger loss of water quality and promote presence of salt wedge (Prat and Ibáñez, 1995). Those effects can consequently alter flora and fauna inhabiting affected areas and might induce a decrease in ecosystem productivity (Prat and Ibáñez, 1995).

j) Sediment input / turbidity. A substantial volume of sediment can enter into coastal lagoons during storm surge and overwash events. Aside from these sources, sediments accumulate in lagoonal basins from rivers draining the mainland (above-mentioned), runoff of tidal marsh and other habitats bordering the basin and internal processes (*i.e.* organic carbon production, chemical precipitation, as well as erosion and resuspension of older sediments; Kennish and Paerl, 2010). As noted by Nichols and Boon (1994), coastal lagoons are net sediment sinks, but changes in the volume of river inflow, frequency of storms, in-basin wave and flow action contribute to considerable variation in the rate of biogenic activity and sediment accumulation in these shallow systems (Kennish and Paerl, 2010). Run-off of different human activities such as agriculture, construction, mining, silviculture or urban run-off can contribute to increase turbidity in coastal lagoons (Kennish, 2002).

Although some turbulence in water body can occur naturally, high amount of sediment input can favor greater water turbidity. Nevertheless, low levels of water turbidity might be beneficial for the ecosystem, promoting internal recycling (Chapelle *et al.* 2000, Ofanidis *et al.* 2005, Badosa *et al.* 2007); while high levels of turbidity can be prejudicial as they alter water transparency (Kennish, 2002), reducing the photosynthetic activity and then, the primary production, which at the same time might diminish the whole ecosystem productivity. Moreover, high turbidity may impair reproduction and osmoregulation, reduce the exchange rate of oxygen and alter respiration in molluscs and fish (Widdows *et al.*, 1979; Daon and Gouletquer, 1988; Servizi *et al.*, 1991).

l) Floatables and debris. Debris can be defined as the human-created waste material released to the environment deliberately or accidentally; meanwhile the part of the debris that remain suspended on the surface of a water body without sinking, constitute the floatables (EPA, 2002).

Debris have a more direct impact by increasing the mortality of organisms which become entangled in it and subsequently, drown or ingest it, which might involve a suffocation or starving process later on. Organisms entangled in debris are also more vulnerable to predation. It has been predicted that floatable debris, especially plastics,

could increase substantially in estuaries and coastal areas by 2025, with the increase of population on those areas and its recreational use (Kennish, 2002).

Two of the most important coastal ecosystem services for human well-being: **fish** and **water resources** (Millenium Ecosystem Assessment, 2005) are overexploited by humans. Fisheries are of particular importance in developing countries as they may be the primary source of animal protein to which rural communities have access (Béné *et al.*, 2007). In reference to water resources, coastal lagoons are important as storage of water for domestic, industrial and agricultural uses and for the hydrological flow regulation (Millenium Ecosystem Assessment, 2005). Due to the importance that the impact induced by human exploitation of those two resources have in coastal lagoon ecosystems, and specifically on coastal lagoons fish assemblages; they were chosen as the main impacts to be studied in the present PhD thesis, and for this reason will be treated separately in the following sections.

4.3 Fisheries in coastal lagoons

Fishing activity in coastal lagoons represents the 10% of fish worldwide production (Quignard, 1984). Despite its importance, scientific knowledge concerning fisheries coastal lagoons mechanisms and ecological functioning is quite scant, scattered and/or superficial (Pérez-Ruzafa and Marcos, 2012). However, one widely known characteristic of fisheries developed in coastal lagoons is that they remain largely artisanal (Kennish and Paerl, 2010). In brief, **artisanal fisheries** definition is a complicated issue, as it varies along history and among countries. A consensus definition could be the fishery that operates using traditional fishing techniques, small amount of capital and energy, and relatively small fishing boats. Although this last characteristic seems to be quite variable, as can range from a one-man canoe in poor developed countries, to a more than 20 m trawlers, seiners or long-liners in developed ones (Jacquet and Pauly, 2008; Mathew, 2003; Food and Agriculture Organization of the United Nations (FAO) 2005-2013). Additionally, the term “artisanal fisheries” must not be confounded with “**small-scale fisheries**”. Although subtle, the difference relies in size (scale) and level of technology (“artisanality”). Then, small-scale fisheries include fisheries of small vessel dimensions that might have (or not) associated high levels of technology (*e.g.* GPS, Sonar, diesel motor, etc), whereas artisanal fisheries mean low levels of technology with little reference to the size (FAO, 2005-2013).

Small-scale and artisanal fisheries often compete with industrial fisheries. However, the former ones have several advantages over the last, as its (i) lower running costs and fuel consumption, (ii) lower ecological impact, (iii) higher employment opportunities and (iv) versatility, (v) lower construction costs and (vi) less expensive technology (FAO, 2005-2013). All these facts, in addition to its usually low percentage of discards, turn out artisanal and small-scale fisheries activity more profitable than industrial fisheries from social, economical and ecological points of view. As an example, small-scale and artisanal fisheries catch the same amount of fish for human consumption (30 million tons) that commercial large-scale fisheries, but employing 25 times the number of fishermen (over 12 million people) and using an eighth part of the fuel used annually by the commercial fisheries (Jacquet and Pauly, 2008). FAO have recognized the important contributions of artisanal and small-scale fisheries to employment, income and food security (FAO, 1995). Despite this international recognition, the similar level of productivity (in terms of total landings) and the recognized universal need of regulation compliance to maintain fishing effort within sustainable limits; the development of policies regulating artisanal fisheries have received much less attention (Common Fisheries Policy, 2008; Diegues *et al.*, 2006). This lack of regulation is critical taking into consideration that “artisanal” does not necessarily mean “sustainable”. Although artisanal and small-scale fishing units practice a seasonal multi-species fishery, using selective and passive gears; just like industrial fisheries, they also have an impact in the exploited systems (Mathew, 2003; World Forum of Fish Harvesters & Fish Workers (WFF)). Fishing activity effects on ecosystems are multiple and diverse, but can generally be divided into direct and indirect effects (Goñi, 1998). The most widely known direct effects are the ones over target species. On target species, fishing activities might decrease fish abundance, change species composition and reduce genetic variability of fish communities (Goñi, 1998; Blaber, 2000). Moreover, fishing gear selectivity arouses changes in size, size-at-age and size at maturity (Botsford, 1997; Goñi, 1998). One of the most notable potential direct effect of fisheries is overfishing. Although considered a big-scale problem, overfishing could also occur in artisanal fisheries scale (UNEP, 2006). Its main effects in target populations include: (i) decreased larval and fry recruitment, (ii) increased mortality risk of new recruits and (iii) diminution of genetic diversity due to the stock reduction (Pérez-Ruzafa and Marcos, 2012). Fisheries also impact directly on non-target species which are most of the times fished as bycatch, and due to its non-commercial value or its unwanted sizes are subsequently discarded (Goñi, 1998). Artisanal fisheries also imply a direct physical disturbance and habitat destruction, especially with trawl gears. The increase of turbidity and changes in the sediment, also have an impact on water quality (Goñi, 1998; Blaber, 2000). Regarding the indirect

effects of fisheries, it is important to highlight effects through biological interactions (trophic effects). Trophic effects include changes in predation and competition relationships, which at the same time lead to changes on the whole trophic community (Goñi, 1998). Other indirect effects include the ones provoked by thrown discards, which can alter community structure and the impact by lost gears that can continue fishing and entangling fish, seabirds or other animals (Goñi, 1998).

Artisanal fisheries in coastal lagoons are strongly adapted to the conditions of their environment. In those systems, fishing is based on a deep cultural understanding by fishermen of the ecology and behavior of target species (Pérez-Ruzafa and Marcos, 2012). As a consequence, there is a substantial adaptive convergence in the fishing gears used by coastal lagoons fishermen throughout the world: they take advantage of the migratory movements of fish between the lagoon and the sea (Pérez-Ruzafa and Marcos, 2012). Examples can be found in Greece, Spain, France and Tunisia, where different fishing gears, but with similar shape and objectives, block the connections with the sea, capturing the seaward migrating fish (De Sostoa and De Sostoa, 1985; Yañez-Arancibia *et al.*, 1994; Koutrakis *et al.*, 2005; Katselis *et al.*, 2010). The efficiency of this kind of fishing gear is very high, as it traps almost all the migratory fish in their way back to the sea. The main target species with those gears are Sparidae, Mugilidae, Anguillidae and Moronidae families, which are present in more than 75% of the Mediterranean coastal lagoons (Pérez-Ruzafa *et al.*, 2011; see section 4.1). In the Western-Central Mediterranean area (*i.e.* Morocco, Algeria, Tunisia, Libya, Italy, France and Spain), the most important gears are gillnets and entangling nets, representing 53% of the total gear used followed by hooks and lines with 33%, and other gears used in less percentage (traps, surrounding nets, seine nets and dredges; Coppola, 2001). In this area, the most important caught fish families include: Sparidae (31%), Serranidae (6%), Scombridae (6%), Mullidae (6%), Carangidae (6%) and Sepidae (5%) (Coppola, 2001). Nevertheless, artisanal fisheries in some coastal lagoons also include among their catches crustaceans (especially prawns), shrimps, crabs and bivalves (Pérez-Ruzafa and Marcos, 2012).

4.4 Water management in coastal lagoons

The water resource is often subjected to pressures derived from different human activities, being agriculture one of the most important ones (see section 4.2). The high fertility of soils surrounding coastal lagoons, have made them devoted to the development of intensive agriculture. However, the high concentration of salts found in

those soils, made almost impossible a general and wide agriculture in terms of plant species cultivation (Galcerà, 2009). In fact, the most adequate (and maybe the only one) plant to cultivate in those conditions - and where a permanent inundation is needed - is rice (Galcerà, 2009). However, a transformation is required for rice-cultivation, implying the canalization of freshwater to the cultivated areas (Chauvelon, 1998; Day Jr. *et al.*, 2002), which alters the natural hydrology of coastal lagoons situated nearby cultivated areas (Heurteaux, 1992; Vicente and Miracle, 1992; Chauvelon, 1998; Menéndez *et al.*, 1998; Menéndez, 2009).

Consequences of those hydrological alterations in coastal lagoons are diverse. The main abiotic consequence is the change of coastal lagoons hydrological pattern, which become artificially controlled by man (Badosa *et al.*, 2007). This artificialization alters some water proprieties as the salinity (Menéndez *et al.*, 1998), and promotes water quality degradation through the high amount of pollutants and nutrients that wastewaters supply to the lagoons (Cloern, 2001; Lucena *et al.*, 2002; Badosa *et al.*, 2006, 2007). A well-known effect of the increase of water nutrients is eutrophication - as previously commented in section 4.2 -, which can become widely extent among lagoons where hydrological alteration has occurred (Forés *et al.*, 2002; Badosa *et al.*, 2007). Direct effects on the organisms living in such altered lagoons consist of changes in its abundance and distribution (Poizat *et al.*, 2004; Badosa *et al.*, 2007). At ecosystem level, those changes modify the trophodynamics and end up altering the structure and composition of the whole ecosystem (Heurteaux, 1992). Specifically in fish, altered hydrologic conditions of lagoons can imply changes in their migratory patterns (Griffiths, 1999; Day Jr. *et al.*, 2002). Moreover, environments with less saltier conditions could promote the establishment and expansion of introduced species in areas where they natural predators are absent (Badosa *et al.*, 2006).

As it has been reviewed above, and aggravated by the global increasing population, higher numbers of anthropogenic impacts in coastal lagoons, and with higher frequency are compromising the sustainability of the quality and productivity of these systems. Thus, they cannot meet the growing demands of the socioeconomic system without their sustainable use and development (Gonenç and Wolfen, 2004; Kennish and Paerl, 2010). The above mentioned human activities require key management strategies pursuing a balance between the finite production and capacity of coastal lagoons and the high socioeconomic demand over their resources (Kennish and Paerl, 2010), in order to reduce their impacts on the ecosystem, if not to avoid them.

4.5 Sustainable management approach in coastal lagoons

The Brundtland Report, released by the United Nations in 1987, included the definition of **sustainable** development as “the development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (World Commission on Environment and Development, 1987). Moreover, the criterion for a sustainable use is that the resource to be harvested, extracted or utilized is not done in excess, and this amount can be self regenerated (Clark, 1992). Thus, under these two definitions, sustainable management would compile the planning, controlling and organizing strategies needed for the achievement of a sustainable use of natural resources, assuring its biological and long-term renewable potential (Clark, 1992; Ritcher *et al.*, 1996).

Any planned measure for sustainable management should define concise objectives for the use and exploitation of the environment; make an inventory of its natural resources; study the processes involved in the functioning of the ecosystem and of different subsystems (terrestrial, marine, interchanges with the open sea and surrounding systems); index human activities and their influences on the ecological processes; classify the territory on the basis of previously defined objectives and its ability to sustain the different activities; establish the standards of environmental quality and design monitoring plans and correction measures (Pérez-Ruzafa *et al.*, 2005). Effective sustainable management promotes a general increase of the conservation and long-term productivity of the managed area by: (i) facilitating sustainable economic growth based on natural resources, (ii) conserving natural habitats and species, (iii) controlling pollution, (iv) controlling watershed activities, (v) controlling excavation, mining and other land alterations, (vi) rehabilitating degraded resources and (vii) providing mechanisms and tools for rational resource allocation (Clark, 1992 ; Day Jr. *et al.*, 2002).

However, the achievement of a sustainable management is not straightforward as it requires gathering the interests of stakeholders (nature protectionists, policy makers, managers, local people and the scientific community) and both present and future generation needs (Rigg, 1999; Suman *et al.*, 2005). Although several Mediterranean and Black Sea coastal lagoons nowadays are examples of successful sustainable processes, where natural dynamics and man development have interacted for centuries (GCFM, 2013; Galcerà, 2009); in all management plans there are always a concern to know until which level exploitation of natural resources can be maintained and conserved (Comín and Menéndez, 2001). Inability to conciliate all the required needs and interests of the

area to be managed and/or to set adequate exploitation limits, can lead to a lack of effective sustainable management (Suman *et al.*, 2005). This, in fact, has been recognized, at European level, as the principal factor responsible for coastal and marine environments degradation (Rigg, 1999; Suman *et al.*, 2005).

One of the most useful tools for coastal fisheries management is the implementation of **ecological models**, which integrate the gathering interest of all implicated parts, providing the required holistic view. Given that ecological modelling in fisheries management is a large topic on its own, the second part of this section will be only focused on models that attempt to take into account all levels of the ecosystem (*i.e.* from primary producers to top predators), the named “Whole ecosystem models”. Models classified as whole ecosystem models (Table 2) include: Ecopath with Ecosim (EwE; Christensen *et al.*, 2008), ATLANTIS (Fulton and Smith, 2004), INVITRO (Gray *et al.*, 2006), European Regional Seas Ecosystem Model (ERSEM II; Baretta, Baretta-Bekker and Ruardij, 1996), Shallow-Seas Ecological Model (SSEM; Sekine *et al.*, 1991) and Spatial Ecosystem And Population Dynamics Model (SEAPODYM; Bertignac, Lehoey and Hampton, 1998). A brief description of these models, highlighting its main advantages and drawbacks is given hereinafter.

Table 2. Comparison of the different whole ecosystem models.

Research question/model	EwE	ATLANTIS	INVITRO	ERSEM II	SSEM	SEPODYM
Understanding - subset of ecosystem						
Understanding - complete ecosystem						
Impact of target species						
Effect of top predators						
Competition: marine mammals- fisheries						
Rebuilding depleted fish stocks						
Biases in single-species assessment						
Ways to distribute fishing effort among fisheries						
Under-exploited species						
Change in ecosystem state						
Spatial concentration of fishing						
Environmental/physical effects						
Effects of habitat modification						
Effects of by-catch						
Introduction of non-native species						

Shared regions highlight appropriateness of the model to answer the research question proposed (adapted from Plagányi, 2007).

The **EwE** modelling tool is composed of a core mass balance model (Ecopath) which stands for Ecological Pathways Model (Christensen *et al.*, 2008), from which temporal and spatial dynamic simulations can be developed with the Ecosim and Ecospace modules, respectively (Christensen and Walters, 2004). EwE highlights from the rest for

its user friendly interface, structured parameterisation framework, well-balanced level of conceptual realism, and a novel representation of predator-prey interaction terms. In many respects, EwE achieves a good balance in model structure between simplicity and the level of complexity inherent to other ecosystem model representations. All those properties converted EwE model into the most widely used worldwide (Plagányi, 2007). **ATLANTIS** is an ecosystem model that considers all parts of marine ecosystems: biophysical, economic and social. Originally focused on the biophysical world, but after in fisheries, its use has grown due to multiple use and climate questions (Fulton and Smith, 2004). Among advantages, the spatially and explicit biomass dynamics in response to different management is highlighted, in addition to the fact that it includes mixotrophy. However, the main drawbacks of ATLANTIS are the high amount of data required and not having a user friendly interface (Plagányi, 2007). Regarding the **INVITRO** model, it uses the basis of Management Basis Evaluation (MSE) focusing on multiple-use ecosystem-level management questions (Gray *et al.*, 2006). This model is agent-based, so it uses a targeted representation across multiple scales and sectors, which constitutes one of its advantages; but similarly to ATLANTIS, it has not an easy user interface (Plagányi, 2007). **ERSEM II** is the improved version of ERSEM, a model developed to simulate the annual cycles of carbon, nitrogen, phosphorus and silicon in the pelagic and benthic components of the North Sea (Baretta, Baretta-Bekker and Ruardij, 1996). The model is related to Nutrient-Phytoplankton-Zooplankton-Detritus (NPZD) models, but includes several refinements necessary to correctly represent the key processes of temperate shelf ecosystems (Baretta, Baretta-Bekker and Ruardij, 1996). Main advantages include the capacity of exploring hydrographic and planktonic conditions impacting juvenile fish and its capacity to be coupled with different physical models. However, it requires intensive data (Plagányi, 2007). **SSEM** is a modeling approach developed to predict the impact of coastal activities on fisheries (Sekine *et al.*, 1991). The model includes a detailed representation of processes such as swimming, advection and diffusion, and requires inputs in the form of water temperature, currents and nutrient loads from surrounding land masses. The main disadvantage of this model is that it is not suitable for answering broader questions related to the ecosystem impacts of fisheries and the fact that requires an intensive data (Plagányi, 2007). Finally, **SEAPODYM** is a two-dimensional coupled physical-biological interaction model at the ocean basin scale, developed for tropical tunas in the Pacific Ocean (Bertignac, Lehoev and Hampton, 1998). Its advantages comprise the attempt to incorporate environmental data directly into spatial population dynamics, and its disadvantage is an insufficient resolution of mid-trophic levels to explore trophic interactions at all levels (for a detailed description of model types see Plagányi, 2007).

Among all the listed whole ecosystem models, EwE was chosen in the present PhD thesis mainly for three reasons: (i) its user friendly interface, (ii) intensive data is not required, and (iii) its worldwide use. However, it must be remained that any model is a schematic representation of the reality, and its development (used data, equations, study area) define in part the kind of scientific questions that can be answered. Thus, and independently of its complexity, a model will never reproduce the reality, being always an accurate approximation (Coll, 2006).

Likewise, more sustainable approaches in coastal lagoons water management, require an integrated view to involve all the facets of water-supply, use and treatment. Moreover, an integrated approach to water management is recommended to address current water challenges, which are often interrelated with other environmental, economic and social issues (Mchugh, 2011). Accordingly, the European Union adopted few years ago, the Water Framework Directive (WFD; Directive 2000/60/EC) to establish basic principles of sustainable water policy in member states. The WFD is aiming for all European waters to have achieved a 'good ecological and chemical status' by 2015, which includes coastal lagoon transitional waters and the catchment boundary of coastal lagoons (European Union, 2000). From a scientific point of view, the WFD provides a challenge to develop new ecological knowledge and new and accurate methodologies for impact assessment and ecosystem restoration. Whereas, from a management point of view, it addresses the assessment of Ecological Quality Status within European rivers, lakes, groundwaters, estuaries and coasts; requiring an integrated management (Pérez-Ruzafa, 2008).

Considering that in coastal lagoons the main anthropogenic activity influencing the water management is agriculture (see section 4.4); the development of a more sustainable agriculture activity would led to a more sustainable use of the water resources in coastal lagoons (Bitelli, 2010). In this way, five key points for the development of a more sustainable agricultural activity have been emphasized: (i) increase water-use efficiency (Tilman *et al.*, 2000); (ii) cultivate crops with greater drought tolerance (Tilman *et al.*, 2000); (iii) improve water quality (*i.e.* avoid salinization and contamination of ground and surface waters by pesticides, nitrates and selenium); (iv) improve irrigation management (Wichelns, 2002) and (v) reuse drainage water for irrigation of appropriate salt-tolerant crops (Rhoades, 1999).

4.6 Study context

The Ebro Delta with 320 km² of extension, represents the most important wetland area in Catalonia, the second most important in Spain (after the Doñana National Park) and one of the most important estuarine zones in Europe (RAMSAR, 1992; Fernández *et al.*, 1997; Day *et al.*, 2006) (Fig. 2). From the ecological point of view, the Ebro Delta stands out for its high diversity of habitats in a significant reduced area (Ibáñez *et al.*, 1999). The species composition, its ornithological and ichthyological fauna, and also its halophilic flora make the delta a singular ecosystem (Ibáñez *et al.*, 1999). Moreover, most of those species, are endemic, thus its preservation and conservation should be a priority task (Ibáñez *et al.*, 1999).

Due to its high biological interest, 25% of its total area, including coastal lagoons, bays and adjacent coastline, is protected as a Natural Park from 1983. Also declared Special Protection Area (SPA) in 1987, Site of Community Importance (SCI) and considered Wetland of International Importance by the RAMSAR convention since 1993. Moreover, in 2000, Ebro Delta was included in the Natura 2000 Network, a European

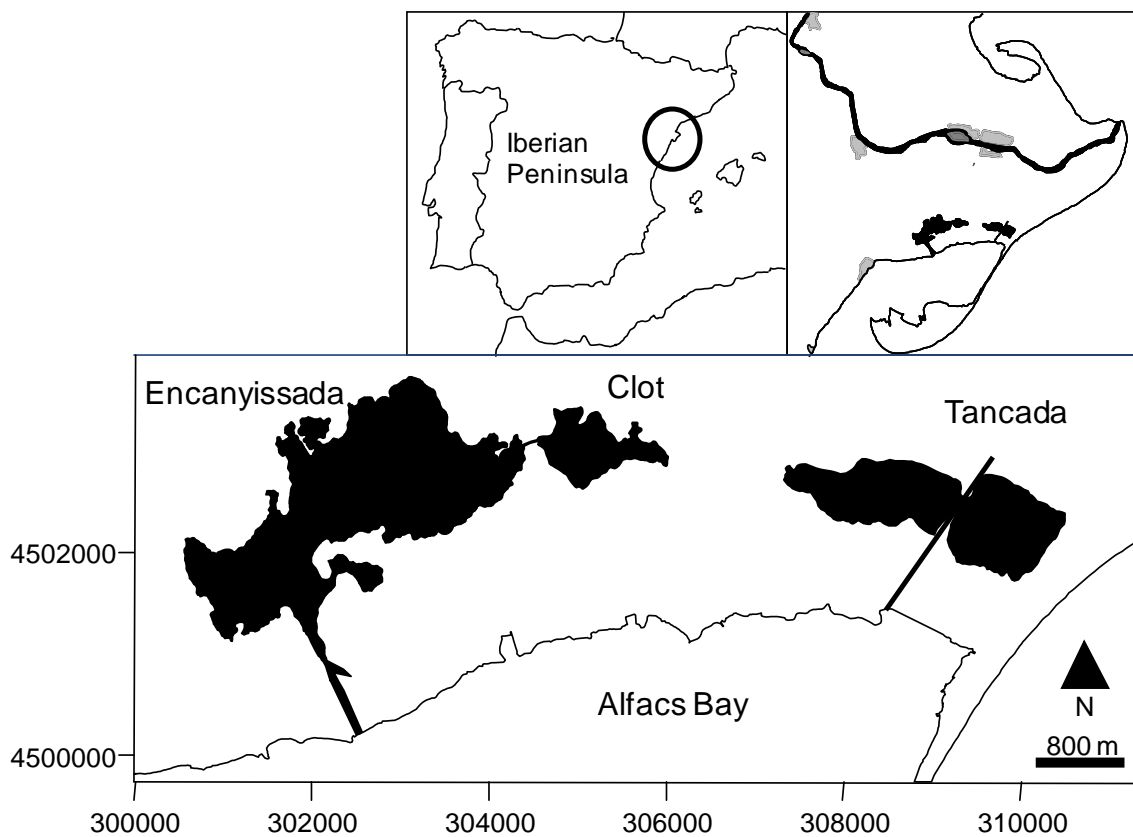


Figure 2. Map showing the Ebro Delta location in Iberian Peninsula (top) and the three coastal lagoons studied in this PhD thesis (bottom).

network of nature protection with the aim to assure the long-term survival of the Europe's most valuable and threatened species and habitats (http://ec.europa.eu/environment/nature/natura2000/index_en.htm). Recently, the United Nations Educational, Scientific and Cultural Organization (UNESCO) declared the Ebro deltaic plain as a World Biosphere Reserve (2013).

Scattered along the delta, **coastal lagoons** represent the 7.2% of its total area and are numerous: Canal Vell, Calaix de l'Illa de Buda, Platjola, Aufacada, Olles, Garxal, Encanyissada, Clot and Tancada. Their origin is diverse; some of them are the result of sand bars that have delimited a sea portion, while others are just old river branches that become isolated from the present river course (Fernández *et al.*, 1997).

In the present PhD thesis, attention has been focus on three lagoons situated in the southern hemidelta: Encanyissada, Clot and Tancada (Fig.2). Its choice was based in two main criteria: (i) their representativeness of the whole variability present in the delta in terms of size, morphology and sea connections and (ii) their exposure to the two anthropogenic stressors in which this thesis is centered: artisanal fisheries and water management. In this regard, artisanal fisheries are carried out only in six of the above mentioned lagoons: Clot, Tancada, Encanyissada, Canal Vell, Olles and Platjola; being this last one discarded due to the fact that is a private property. Concerning the water management, although all coastal lagoons in the delta plain have been affected by this impact carried out to develop rice agriculture in the area; the strategies followed differ among some lagoons and the two hemidelta's. Tancada lagoon stands out from the rest, because the water management carried out is closer to its natural state, and thus deserved to be studied. The difference among hemidelta's is due to the fact that the opening and arrival of freshwater is progressive, starting first for one hemidelta and then the other (Natural park personal communication). Then, in order to obtain comparable results, lagoons for the same hemidelta as Tancada lagoon were chosen.

Encanyissada lagoon is a long coastal lagoon with its longest axis running parallel to the coastline. Located in the southern part of the Ebro, is the largest lagoon in the delta. The lagoon is 4.1 km long and 1.14 km mean wide, with a total surface area of 4.18 km². Average depth is 77 cm, with a maximum depth of 121 cm, and its bottom has a uniform relief. Lagoon perimeter is constantly changing depending on water level, being marked by a belt of vegetation formed mainly by reed beds (majorly: *Arundo donax*, *Phragmites australis* and *Typha sp*). In one side, the lagoon is connected to the drainage canals of the irrigation system that during the rice cultivation season supply freshwater; meanwhile on the other side, it is also connected through different artificial canals and natural

channels to the Alfacs bay, a large natural bay located in the southern lobe of the Ebro Delta, between the coastal border of the deltaic plain and the southern sand spit.

Encanyissada lagoon is separated from the **Clot** lagoon by a floodgate in a short canal of 100 m long. The last is a small (0.56 km² surface with 1.6 km long and 0.8 km wide) and shallow (53 cm of mean depth; maximum depth of 90 cm) lagoon that has no direct connection with the sea (it has an indirect connection through the Encanyissada lagoon).

Finally, at the east of both Encanyissada and Clot lagoons is situated the **Tancada** lagoon. This lagoon is nearly rectangular in shape and its main axis is parallel to the coastline. It has a 3.01 km of length and a total surface of 1.85 km². It presents a strait that clearly delimits two basins: the eastern (with a total length of 1.16 km and mean wide of 783 m) and the western (total length of 1.85 km and mean wide of 485 m). With an average depth of 58 cm and maximum depth 80 cm; as the previous described lagoons, Tancada's bottom relief is uniform. Tancada lagoon is mainly surrounded by a belt of helophytic vegetation except on the western flank where it is halophilous. Several drainage canals connect this lagoon to the network of the rice irrigation system; whereas a bunch of artificial canals (concretely 5) link the lagoon with the Mediterranean Sea through the Alfacs bay (Fig. 2).

The Ebro Delta has been target of various types of **human pressures and impacts** during its recent history. Agricultural activity developed in this region (mainly rice cultivation) provoked an artificialization of its hydrological cycle; meanwhile artisanal fisheries were carried out with a low surveillance. Moreover, human pressures resulting from the development and expansion of tourism increased in last decades. Many of these activities led to environmental changes that affected the biota and altered the main configuration of their lagoons. Most changes occurred in recent times in the Ebro Delta and the main impacts are described below.

Although the Ebro Delta was relatively recently colonized by human population, it is considered a highly impacted area (Ibáñez *et al.*, 1999; Mañosa *et al.*, 2001). Indeed, the original vegetation landscape (*i.e.* saltwort, absence of trees) has been greatly transformed due to the agriculture carried out for more than a century (Espanya, 1997). In this century, agriculture in the area faced (and still confronts) a huge inconvenient: the high salinity of the deltaic soils, which made almost impossible the cultivation of anything (Galcerà, 2009). In order to allow soil use, a high transformation of the deltaic plain by constructing an extended network of canals to distribute freshwater to the area was

performed (Ibáñez *et al.*, 1999; Galcerà, 2009). To better understand the modern history of the Ebro Delta and how this transformation took place, four periods taking into account the kind and intensity of land exploitation at each time were differentiated (Table 3):

1) *Until 1900: Natural delta.* Until the first half of the XIXth century, the transformation of the delta was minor and its natural properties preserved: large floods and high-suspended sediment flux, which gave high accretion rates, large wetland surface and low eutrophication (Mañosa *et al.*, 2001). The harsh characteristics of the area with regular floods (basin not regulated), high salinity (which made the exploitation of the soil difficult) and wetlands insalubrity (where the mosquitoes could transmit the malaria) made the life for human population difficult. However, the few inhabitants of the region knew how to take advantage of those hard conditions by cultivating punctually in the period when floods allowed a reduction of the salt content of the soil. This temporality in the exploitation did not allow a continuous transformation of the deltaic plain, which conserved its natural state (Galcerà, 2009). Moreover, hunting and fishing activities were also practiced in the area as a subsistence economy (Ibáñez *et al.*, 1999). The first reference of rice farming in the area dates back to the XVIth century, although it was not until some centuries later, in 1860, when the construction of the first canal in the southern hemidelta allowed the implementation of this type of crop, which reached its great expansion by the end of the XIXth century-beginning of the XXth century, when approximately 4000 ha (12% of the area) were cultivated (Mañosa *et al.*, 2001; De Sostoa and De Sostoa, 1985).

2) *1900-1960: Humanized delta ("Rice revolution").* This period is characterized by the agriculture transformation of the deltaic plain, mostly due to the greatest rice farming expansion (Ibáñez *et al.*, 1999; Galcerà, 2009). To allow this expansion, an extensive infrastructure of canals to bring freshwater to the crops was constructed (Fig. 3). The entrance of freshwater modified the hydrological dynamics of the delta, becoming more regular and less climate dependent (Ibáñez *et al.*, 1999). Furthermore, continuous freshwater inputs during the rice cultivation cycle (from April to December, see Table 4) inverted the hydrological pattern in the Ebro Delta coastal lagoons; being the summer season the period with more freshwater input along the year instead of winter (De Sostoa and De Sostoa, 1985; Comín *et al.*, 1987). The human control of the hydrological cycle implied changes in the ecological functioning of the aquatic ecosystems as well as the landscape transformation (Ibáñez *et al.*, 1999). From 1910 to 1960 - the period of fastest development of rice farming in the delta -, rice crops spread over the most part of the delta plain, reaching 17400 ha in 1960. This

Table 3. Transformation process in the Ebro Delta since the beginning of the XVIIIth century.

		Until 1900	1900-1960	1960-1980	1980-present
		NATURAL DELTA	HUMANIZED DELTA	ENDANGERED DELTA	
DELTA	Human Population	Sparse	High increase	Low increase	Stabilization
	Predominant habitat	Saline habitats	Freshwater habitats (more canals)	Freshwater habitats	Few saline habitats
RIVER	General condition	No contamination		Moderate contamination	Moderate-high contamination
		High sediment loading		Very low sediment loading	
	Basin	Low regulation	Moderate regulation	High regulation	
ECONOMY	Cultivation method	Vegetable patches	Traditional agriculture rice	Mechanization of crops Use of pesticides	Chemical and mechanic agriculture. Artificial hydrology
	Subsistence economy	Hunting, fishing and extensive livestock	Hunting, fishing and extensive livestock	Intensive hunting	Controlled hunting
	Market economy	Salt			
		Agriculture			
		Fishing			
		Services			
		Industry			
		Tourism			
	Aquaculture				
	Hunting				

(adapted from Ibáñez *et al.*, 1999)

Table 4. Main activities developed in the Ebro Delta along the year.

		J	F	M	A	M	J	J	A	S	O	N	D
Rice fields	Activity	Land preparation (harrowing and levelling)				Rice cultivation						Rest	
	Period					Sowing	Vegetative period (heading and maturation)				Harvesting	Postharvest	
Water canals	Activity	Closed. Cleaning and maintaining tasks			Flood-ing	Open for rice cultivation (maximum flow)					Open for environmental purposes (minimum flow)		
	Period	Brackish water period				Freshwater period							
Fisheries	Activity							Fishing draw					
	Period	Fishing period			Fishing rest-period						Fishing period		

expansion was in detriment of the natural habitats, which declined from 80% (27000 ha) to 33% of the total surface (11000 ha), and with the lost of littoral lagoons, marshes and pastures (Mañosa *et al.*, 2001). As a result, the economy of the Ebro Delta population changed from a mere subsistence to a more complex agricultural society (Ibáñez *et al.*, 1999). Although this period was characterized by a high land transformation, the natural system was not severely impacted because of the traditional way of land exploitation (*i.e.* without crop mechanization and use of pesticides/herbicides).

3) *1960-1980: Endangered delta (I)*. During these two decades, the delta passed from being exploited in a traditional and extensive manner to a highly technological and intensive way, with one clear objective: maximize the rice production (Ibáñez *et al.*, 1999). The intensive agriculture implied the mechanization of the crops and the introduction of insecticides and pesticides in the fields (Ibáñez *et al.*, 1999). The change on the cultivation system allowed the reduction of the cultivated area in the first years (1960-1965), to approximately 12000 ha of extension (Mañosa *et al.*, 2001). At the end of this first decade (late 1960's), the construction of two big dams in the lower part of the Ebro basin (Riba-roja and Mequinença) aggravated the situation of the already regulated river basin by reducing the solid river discharge (sediments) and turning the accretion rates in the delta plain insignificant (Ibáñez *et al.*, 1999). In the market economy sector, the total landings of the artisanal fisheries developed in the area started to be registered (1965; St. Pere Fishermen association), which indicate a more active professional fishing. Moreover, by the end of this period (1975-1985), a recovery of the crop area was observed, reaching 18500 ha of the cultivated area (Mañosa *et al.*, 2001).

4) *From 1980 to present: Endangered delta (II)*. In this period, some measures such as the creation of the Natural Park (1983) or the implementation of the European environmental legislation, have been undertaken to improve the ecological status of the river and the delta, and made their exploitation more sustainable (Ibáñez *et al.*, 1999). The implementation of all these measures stopped the loss of natural habitats and nowadays 25% of the delta surface (7736 ha) is still occupied by lagoons and marshes, being those areas protected by the Natural Park (Mañosa *et al.*, 2001). In the economic sector, fishing activity experimented a reduction of the effort and in the total catches (Franch and Quintano, 2004), while tourism related activities like urbanization increased exponentially, thus adding a new threat to the already endangered delta (Ibáñez *et al.*, 1999).

The deltaic plain with the mechanization of the crops, the use of pesticides, freshwater canalizations, ditches and the artificialization of the hydrologic cycle have been totally transformed from a natural environment to an environment completely dependent of the human management. The price that the natural delta had to pay for the “rice revolution” (or technological advance) has been quite high: reduction of the reptilian, fish, amphibians and mammals populations, decline of the flora diversity and the detriment of unrecoverable ecological values and services. Evenmore, the current impact of the artisanal fisheries developed in the area remains largely unknown, since no more studies regarding the ichthyofauna were done from the eighties (De Sostoa, 1983).

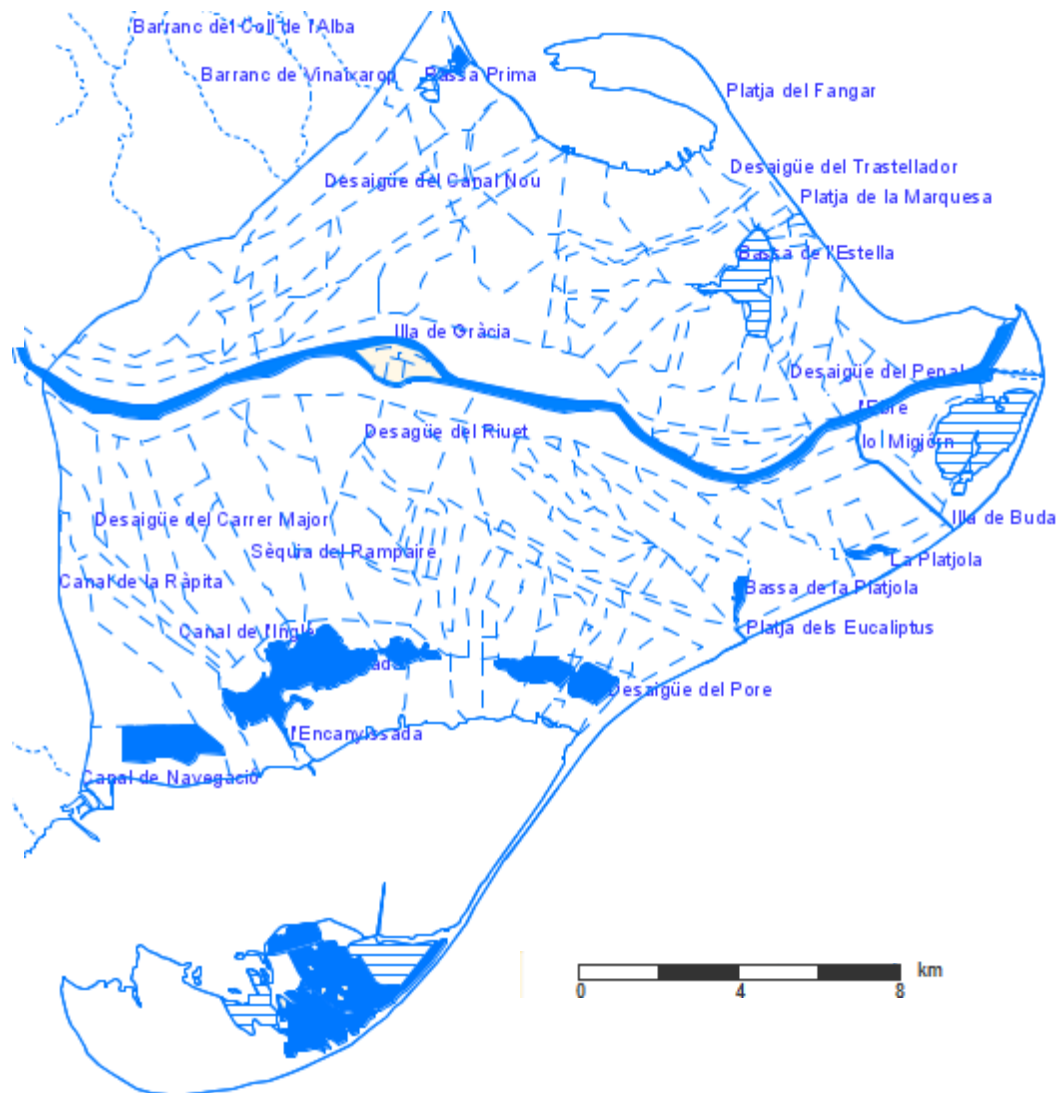


Figure 3. Map of the Ebro Delta plain showing the irrigation canals network, the river and the coastal lagoons. The most important irrigation canals are also highlighted (from Infraestructura de Dades Espacials de Catalunya (IDEC): www.geoportal-idec.cat).

The present PhD thesis is settled in the last of the four periods presented, so is facing a threatened and endangered delta. From all the mentioned human activities developed in the deltaic plain, two of the major economic sectors (and the ones where this PhD thesis will be focus on) are: the **artisanal fisheries** and the **water management** derived from the rice cultivation. The impact of both anthropogenic stressors on the ichthyofauna was considered, evaluated and discussed in the present PhD thesis.

The ichthyofauna of the Ebro Delta coastal lagoons has three different origins: freshwater, marine and brackish environments; being the success in colonization by different species dependent on their degree of adaptation and their capacity to withstand environmental fluctuations. The entrance of marine species in the lagoons is quite easy due to the numerous natural channels and artificial canals that connect the lagoons with the sea allowing the pass of this species; whereas the entrance of the freshwater species is restricted to the periods when the irrigation canals are open.

Table 5 shows the fish species present in the lagoons before the present PhD thesis. This information was gathered from previous studies in the area: De Sostoa (1983); De Sostoa and De Sostoa (1985); De Sostoa and Lobón-Cerviá (1989); and De Sostoa *et al.* (1990). Following the classification suggested by De Sostoa and De Sostoa (1985), listed fish species can be organized according to their origin (marine/brackish/freshwater) and permanence (seasonal/sedentary/accidental/migratory) in the lagoons. Among the marine species, accidental species such as European anchovy (*Engraulis encrasicolus* (Linnaeus, 1758)) and European pilchard (*Sardina pilchardus* (Walbaum, 1792)) or seasonal species that regularly colonized those waters as the case of garfish (*Belone belone* (Linnaeus, 1761)) exclusively for reproduction were found in the past. Seasonal species present at that time in the lagoons that entered in the juvenile stage looking for a safe and rich in food place (De Sostoa and De Sostoa, 1985) were sand sole (*Pegusa lascaris* (Risso, 1810)), blackhand sole (*Pegusa nasuta* (Pallas, 1814)) or *S. solea*; while others entered only in the adult stage such as the case of European flounder (*Platichthys flesus* (Linnaeus, 1758)) (De Sostoa *et al.*, 1990). The rest of the seasonal species found in the lagoons, either in their adult or juvenile stage, were: mullets (thicklip grey mullet (*Chelon labrosus* (Risso, 1827)), *L. ramada*, *L. saliens*, *L. aurata*, *M. cephalus* and boxlip mullet (*Oedalechilus labeao* (Cuvier, 1829))), *D. labrax* and *S. aurata* (De Sostoa and De Sostoa, 1985). Also, brackish/marine species that lived in the lagoons and eventually went to reproduce at the sea (although most of them could close their life-cycle in the lagoons; thus, seasonal-sedentary species) like *A. boyeri*, sand goby (*Pomatoschistus*

Table 5. List of fish species cited in previous studies in the Ebro Delta coastal lagoons.

Family	Species	Common name	State	Described by
Anguillidae	<i>Anguilla anguilla</i>	European eel	Native	Linnaeus, 1758
Atherinidae	<i>Atherina boyeri</i>	Sand smelt	Native	Risso, 1810
Belonidae	<i>Belone belone</i>	Garfish	Native	Linnaeus, 1761
Blennidae	<i>Salaria fluviatilis</i>	Freshwater blenny	Native	Asso, 1801
Clupeidae	<i>Sardina pilchardus</i>	European pilchard	Native	Walbaum, 1792
Cyprinidae	<i>Cyprinus carpio</i>	Common carp	Introduced	Linnaeus, 1758
	<i>Carassius auratus</i>	Goldfish	Introduced	Linnaeus, 1758
Cyprinodontidae	<i>Aphanius iberus</i>	Spanish toothcarp	Native	Valenciennes, 1846
Engraulidae	<i>Engraulis encrasicolus</i>	European anchovy	Native	Linnaeus, 1758
Gobiidae	<i>Pomatoschistus microps</i>	Sand goby	Native	Krøyer, 1838
Moronidae	<i>Dicentrarchus labrax</i>	European sea bass	Native	Linnaeus, 1758
Mugilidae	<i>Chelon labrosus</i>	Thicklip grey mullet	Native	Risso, 1827
	<i>Liza ramada</i>	Thinlip grey mullet	Native	Risso, 1827
	<i>Liza aurata</i>	Golden grey mullet	Native	Risso, 1810
	<i>Liza saliens</i>	Leaping mullet	Native	Risso, 1810
	<i>Mugil cephalus</i>	Flathead grey mullet	Native	Linnaeus, 1758
	<i>Oedalechilus labeo</i>	Boxlip mullet	Native	Cuvier, 1829
Pleuronectidae	<i>Platichthys flesus</i>	European flounder	Native	Linnaeus, 1758
	<i>Pegusa lascaris</i>	Sand sole	Native	Risso, 1810
Poeciliidae	<i>Gambusia holbrooki</i>	Eastern mosquitofish	Introduced	Girard, 1859
Soleidae	<i>Pegusa nasuta</i>	Blackhand sole	Native	Pallas, 1814
	<i>Solea solea</i>	Common sole	Native	Linnaeus, 1758
Sparidae	<i>Sparus aurata</i>	Gilthead seabream	Native	Linnaeus, 1758
Syngnathidae	<i>Syngnathus abaster</i>	Black-striped pipefish	Native	Risso, 1827

Data collected from De Sostoa (1983); De Sostoa and De Sostoa (1985); De Sostoa and Lobón-Cerviá (1989) and De Sostoa *et al.* (1990).

microps (Krøyer, 1838)) and black-striped pipefish (*Syngnathus abaster* Risso, 1810) were found. Freshwater species found in the lagoons were either sedentary (Spanish toothcarp (*Aphanius iberus* (Valenciennes, 1846)); considered brackish/freshwater; Froese and Pauly, 2013) and Eastern mosquitofish (*Gambusia holbrooki* Girard, 1859); seasonal like common carp (*Cyprinus carpio* Linnaeus, 1758) and goldfish (*Carassius auratus* (Linnaeus, 1758)); or accidental as the freshwater blenny (*Salaria fluviatilis* (Asso, 1801)). Their abundance depended on the freshwater inputs drained into the lagoons as well as on freshwater permanence and the evaporation index. *A. anguilla* was the only catadromous species colonizing the lagoons at that time.

From the total 24 species historically found in the Ebro Delta coastal lagoons, three were exotic (*G. holbrooki*, *C. carpio* and *C. auratus*). Moreover, two of the cited species:

A. iberus and *A. anguilla* are currently catalogued as endangered and critically endangered species by the International Union for Conservation of Nature (IUCN) (<http://www.iucnredlist.org>), respectively; meanwhile *A. boyeri* and *S. fluviatilis* are presently considered as a vulnerable species in Spain (Doadrio, 2001 and RD 139/2011, respectively).

4.6.1 Main threats to coastal lagoons' ichthyofauna

One of the main threats for the Ebro Delta coastal lagoons's ichthyofauna is the **artisanal fisheries** carried out in the area. Artisanal fisheries in the Ebro Delta coastal lagoons are performed since XIIth century, although it was not until the XVIIth century when this activity started to be developed in a professional way (Galcerà, 2009). Current fisheries management in the area is done and self-managed by the Sant Pere Fishermen association (Cofradia de Pescadors de Sant Pere). Every July (before the fishing season starts), this association makes a public draw (see Table 4) to determine whose members of the association will be able to fish that year. Fishers elected to fish one year, do not enter in the draw on the following year to ensure the chance of all members to fish (Franch and Quintano, 2004; Fernández, 2007). Each of the selected fishermen practice fishing in an artisanal manner, using traditional boats and passive gears. Two different passive fishing gears types are used: entangling nets and traps. Among entangling nets, the fishing gear used by fishermen in the lagoons are **trammel nets**. Trammel nets are made with three walls of netting, the two outer walls being of a larger mesh size than the loosely hung inner netting panel. The fish get entangled in the inner small meshed wall after passing through the outer wall (Nédélec and Prado, 1990). This fishing gear has been historically used to capture mostly: *D. labrax*, mullets (*M. cephalus*, *L. ramada*, *L. aurata*, *L. saliens* and *C. labrosus*), *P. flesus*, *S. solea*, *P. lascaris* Risso, 1810), *S. aurata*, *C. auratus* and *C. carpio* (De Sostoa, 1983). Regarding the traps, three different ones are used in the Ebro Delta lagoons: fyke nets, glass eel fishing traps (local name: Bussó) and "Pantena" fishing trap. **Fyke nets** consist of cylindrical or cone-shaped bags mounted on rings or other rigid structures, covered by netting and completed by wings or leaders which guide the fish towards the opening of the bags (Nédélec and Prado, 1990). Its shape made this fishing gear suitable for capturing fish species of small size (Brazner *et al.*, 1998; Pampoulie *et al.*, 2001; Uzarski *et al.*, 2005; Fredberg *et al.*, 2009; Wedderburn *et al.*, 2012). However, captures of *A. anguilla* and sometimes *D. labrax*, but juvenile forms of *C. auratus*, *C. carpio* and mullets (*L. ramada*, *L. saliens*, *L. aurata*, *M. cephalus* and *C. labrosus*) have been also historically registered (De Sostoa, 1983). **Glass eel fishing traps** consist of a trapezoidal metallic or wooden structure, covered

with a 1 mm mesh net. At the trap's entrance, there is a funnel directed inwards that prevents fish escape (De Sostoa, 1983; Gibsert and López, 2008). This kind of fishing gear is designed to catch glass eel (juveniles of *A. anguilla*) - where it gets its English name - and takes advantage of the migratory movements of this species. The last, "**Pantena**" fishing gear, is a special kind of trap with a funnel shape settled permanently in the canals that communicate the lagoons with the sea. Although the "pantena" structure is permanently settled, the nets are only set during the fishing season. As the fishing season coincides with the migrating period of most of the species in the lagoon, this fishing gear is specially designed to capture the migratory fish in their way back to the sea (Fig. 4). "Pantena" is considered one of the oldest fishing gears used in the Ebro Delta. New materials with longer duration and equal efficiency replaced the old wood and cotton with which this fishing gear was originally constructed, but the main structure and functioning remain the same (De Sostoa, 1983). There is one "Pantena" in Encanyissada lagoon (Fig. 4) and another one in the Tancada lagoon.

Although there is a sort of control by limiting the number of fishermen authorized to fish every year, the activity is not submitted to any restrictive control measures regarding the number of fishing gears per lagoon and species' size reporting. Moreover, most captured fish are directly sold without intermediaries. Thus, the self-control management carried out for the Fishermen association seems to be an inadequate manner to administer fish resources at the moment. Seven species out of the total found historically in the lagoon are of commercial interest nowadays: *D. labrax*, *S. aurata*, *A. boyeri*, glass and adult eels (*A. anguilla*), adults of species from the Mugilidae family (*L. saliens*, *L. aurata*, *L. ramada*, *M. cephalus*, *C. labrosus*) and *C. caprio*.

The second major threat for coastal lagoons' ichthyofauna, **water management** associated with rice cultivation activity, caused a profound transformation of the Ebro Delta regarding its landscape and ecological functioning as already stated in the beginning of this section. To sustain rice cultivation agricultural activity, two main canals, regulated at their origin point (40 km upstream the river mouth) were constructed and split into numerous secondary canals and ditches to bring freshwater to all rice fields in the delta (see Fig. 3). However, this freshwater supply to rice fields is seasonal, being interrupted between January and mid April, when fields are prepared for the next crop (see Table 4). In the past, waste water from the rice fields was drained to the lagoons, and from these systems to the sea through the outlets. This water management scheme caused eutrophication in the coastal lagoons (Forés *et al.*, 2002). This problem was overcome in the nineties with the construction of circumvallation canals, avoiding

drainage water passing through the lagoons (Forés *et al.*, 2002). At present, the Ebro Delta coastal lagoons still receive freshwater artificially during nine months per year; the first six months to stimulate rice growth (April-September), and the last three months to promote habitat management for game birds (October-December). The freshwater comes both from the river, through the irrigation canals, and from the rice fields drainages. The quality of the river water increased significantly in the last 15 years (Nebra *et al.*, 2011, Ibáñez *et al.*, 2012a,b) and the water input from the rice fields drainages is done only in the periods of low inputs of pesticides and fertilizers (Forés *et al.*, 2002). As a result, water quality in the lagoons has improved, although its hydrology is still severely altered.

The construction of the extensive canals network had severe consequences to coastal lagoons such as: diminution of the average annual salinity, increase of the water nutrient concentration that can lead to eutrophication, increase of chemical contaminant inputs, decrease of the coastal lagoons total surface (and even disappearance of some of them), decrease of water transparency and inversion of the hydrological cycle of the coastal lagoons (Cloern 2001; Lucena *et al.*, 2002; Curcó, 2006; Badosa *et al.*, 2007). One of the most stressful consequences for the ichthyofauna inhabiting the coastal lagoons has been the inversion of its hydrological cycle, that now register the saltier conditions in winter, contrary to the rest of non-hydrological altered Mediterranean coastal lagoons, where the saltier conditions are taking place on summer (Comín *et al.*, 1987; De Sostoa and De Sostoa, 1985; Forés *et al.*, 2002; Curcó, 2006). The effects of this water management over the coastal lagoons' fish assemblages remain unknown.

4.7 Thesis justification

The present thesis is focused on analyzing the impact of two the main human activities in the Ebro Delta (artisanal fisheries and water management derived from rice cultivation) on the fish communities from three coastal lagoons located in the southern hemidelta: Encanyissada, Clot and Tancada. Fish communities were chosen as they are good elements to integrate information about the whole ecosystem (van der Oost *et al.*, 2011). First of all, because they live exclusively in aquatic environments, which made them very sensitive to changes in their environment, either physical or chemical (De Sostoa *et al.*, 2005). Secondly, they have life-cycles that allow them to integrate changes in a long time period (De Sostoa *et al.*, 2006). Last but not least, fish communities include different trophic levels (omnivorous, insectivorous, planktivorous, piscivorous) and they are usually in the upper part of the trophic webs. Then, the composition and structure of the

fish community might provide information from lower trophic levels (especially algae and invertebrates) and reflects the quality state of the whole community (De Sostoa *et al.*, 2005). Moreover, the fish fauna from the coastal lagoons is a natural resource that has been exploited by the local fishermen for more than fifty years. Although a constant decrease of the landings has been observed from the mid 1970s to recent times, there are not studies analyzing the causes of this decline. Moreover, studies focusing in the fish fauna are very scarce and not updated at all (De Sostoa, 1983). Thus, the current species composition and the impact of the self-controlled artisanal fisheries on target species remain unexplored. In addition, during this lapse of time (30 years) without studies focusing on Ebro Delta ichthyofauna; water management in the area has changed and the impact of this activity on the fish assemblages remains also unstudied.

Studies focusing on the impact of the artisanal fisheries in the Ebro Delta and in the lagoons are scarce. The only published works in this issue belong to the Natural Park, and most of the times are published just in local journals (or just in the Natural Park journal), being their diffusion to the scientific community reduced (Franch and Quintano, 2004). The effects of the fisheries in the ichthyofauna were studied for the first time by the already mentioned doctoral thesis of Sostoa (De Sostoa, 1983). Since then, no more studies have been done.

In contrast, the number of studies analyzing the consequences of the water management in the Ebro Delta and particularly over its lagoons is extent. However, along the years the aims/needs of research have changed. In the seventies and eighties the studies were focused on analyzing the rice fields as an ecosystem and to understand how all the biological and biochemical processes worked out (Forés and Comín, 1986, Comín *et al.*, 1987, Forés, 1989). These first studies (after the called “rice revolution” period) alerted from the eutrophicated state of the lagoons. At that time, waste water from the rice fields was drained to the lagoons, and from these systems, to the sea through the outlets. Therefore, the construction of circumvallation canals to prevent the drainage water to pass through the lagoons solved the problem in the 1990’s (Forés *et al.*, 2002). Then, the studies were centered on analyzing how the hydrological alteration of the lagoons could affect the living organisms (Menéndez and Comín, 2000; Forés *et al.*, 2002). In 2001 a multidisciplinary study in the Ebro Delta –including the study of the sediment problematic, microbial mats, biogeochemical processes, benthos, flora, pesticides, virus, molluscs, vertebrates and land management among others- was



Figure 4. “Pantena” in the Encanyissada lagoon during the fishing period (Photograph: Sílvia Rodríguez).

edited. Nevertheless, none of those studies were focus on analyzing how the Ebro Delta’s human activities affected the inhabiting fish community (Institut d’Estudis Catalans, 2001). Only the aforementioned PhD thesis from Sostoa (De Sostoa, 1983) has focused in the characterization of the fish communities in the Ebro Delta and the main factors that explain its distribution and abundance.

All together, the extensively self-controlled artisanal fisheries and the new hydrological management carried out nowadays in the Ebro Delta, makes that the study of the state of the fish community from the coastal lagoons is an imperative requirement to know, and if needed preserve, the native ichthyofauna; and to suggest different measures to avoid/reduce their impacts in order to reach a suitable management of these priceless ecosystems. Therefore, this PhD thesis has been developed to describe the current state of the fish community in the coastal lagoons, how human activities might impact on this ecosystem, and propose a better management that may integrate, combine and pace human activities with nature conservation.

OBJECTIVES

5

The main objective of the present PhD thesis was to assess the effects of two important human induced stressors on the fish fauna from Ebro Delta coastal lagoons: artisanal fisheries and water management. Moreover, this study aimed to describe the state of the art of the fish community composition and structure that remained unstudied for over 3 decades. It is expected that the results and conclusions presented in this PhD thesis, identifying and quantifying main human threats in the Ebro Delta, can assist for the implementation of future sustainable management schemes in this natural and endangered ecosystem.

The PhD thesis is organized in 3 chapters that results from 3 publications. Although the order of the chapters is not chronological (*i.e.* the second chapter is not published yet), this structure was chosen for the better comprehension and following of the ideas presented in the thesis. Thus, the first two chapters cover the artisanal fisheries impact, while the last chapter is dealing with the water management issue. Individually, each chapter pursues the following specific objectives:

Chapter I.

- Description of the state of the art of the fish community composition and structure from the Ebro Delta coastal lagoons.
- Modelling gillnet selectivity using the Share Each Length Catch Total (SELECT) method, as the first attempt to describe gillnet selection patterns.
- Integration of reproductive parameters combined with fishing gears (gillnet and fyke net) mesh size information to establish a Minimum Landing Size (MLS) for *Atherina boyeri*, a vulnerable species with a growing commercial interest.
- Evaluation of mesh size and fishing nets as a possible approach to prevent overfishing of a vulnerable species (*A. boyeri*) in the Ebro Delta coastal lagoons.
- Proposal of management and conservation measures for *A. boyeri*.

Chapter II.


- Data gathering (literature and estimations) of species diet composition, biomass, production, consumption and fisheries landings data for the ecosystem characterization.
- Construction of a trophic-web model for the Tancada lagoon using Ecopath.
- Simulation of the effects of different future fishing effort scenarios with the Ecosim application in order to assist on the designing of a long-term sustainable exploitation plan of fisheries resources in the Tancada lagoon.

Chapter III.

- Description of the composition of the small-bodied fish assemblages from the Ebro Delta coastal lagoons.
- Investigating the relationship of fish assemblages' structure and distribution patterns with the main environmental features that depend on the water management.
- Assessing the degree of association of each species with the studied physicochemical parameters.
- Proposal of a water management scheme that favours the native fish species that are important for both conservation and commercial purposes.

CHAPTERS

6



Gillnet selectivity in the Ebro Delta coastal lagoons and its implication for the fishery management of the sand smelt, *Atherina boyeri* (Actinopterygii: Atherinidae)

Rodríguez-Climent S, Alcaraz C, Caiola N, Ibáñez C, Nebra A, Muñoz-Camarillo G, Casals F, Vinyoles D, De Sostoa A

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Gillnet selectivity in the Ebro Delta coastal lagoons and its implication for the fishery management of the sand smelt, *Atherina boyeri* (Actinopterygii: Atherinidae)

Sílvia Rodríguez-Climent¹, Carles Alcaraz¹, Nuno Caiola^{1,*}, Carles Ibáñez¹, Alfonso Nebra¹, Gloria Muñoz-Camarillo¹, Frederic Casals², Dolors Vinyoles³ and Adolfo de Sostoa³

¹ IRTA Aquatic Ecosystems, Crta. de Poble Nou, Km 5.5. E-43540 Sant Carles de la Ràpita, Catalonia, Spain

² Universitat de Lleida, Catalonia, Spain

³ Universitat de Barcelona, Catalonia, Spain

Abstract

Multimesh nylon gillnets were set in three Ebro Delta (North-East of Spain) lagoons to determine mesh selectivity for the inhabiting fish community. Each gillnet consisted on a series of twelve panels of different mesh size (ranging from 5.0 to 55.0 mm bar length) randomly distributed. The SELECT method (Share Each Length's Catch Total) was used to estimate retention curves through five models: normal location, normal scale, gamma, lognormal and inverse Gaussian. Each model was fitted twice, under the assumptions of equal and proportional to mesh size fishing effort, but no differences were found between approaches. A possible situation of overfishing in the lagoons, where artisanal fisheries are carried out with a low surveillance effort, was assessed using a vulnerable species inhabiting these brackish waters as case study: the sand smelt, *Atherina boyeri*. The minimum size for its fishery has not been established, thus remaining under an uncontrolled exploitation situation. Therefore, a Minimum Landing Size (MLS) is proposed based on sexual maturity data. The importance of establishing an adequate MLS and regulate mesh sizes in order to respect natural maturation length is discussed, as well as, the proposal of other measures to improve *A. boyeri* fishery management.

Keywords: artisanal fisheries, multimesh gillnets, fyke nets, mesh regulations, SELECT method, Minimum Landing Size.

* Corresponding author E-mail: nuno.caiola@irta.cat (Nuno Caiola).Tel: +34 902 789 499 Ext. 1821

1. Introduction

Fisheries have altered and degraded marine ecosystems through direct (e.g. overfishing, bycatch and discards) and indirect (e.g. community changes, mortality caused by lost gears, etc.) effects, especially in coastal regions where fishing and other anthropogenic perturbations are most intense (Botsford *et al.*, 1997; Goñi, 1998). Nowadays, the main threat of fisheries is overfishing, with numerous examples of stocks abundance reduction until population collapse (Botsford *et al.*, 1997; Hutchings, 2000). Of major concern is the fact that in industrial fisheries, the 10 most fished marine species that account for the 30% of the world production are overexploited (FAO, 2009). Artisanal, small-scale fisheries, although are less productive, contribute to more than of the 25% of total world catch (Mathew, 2003). Because fisheries are size selective, removing preferentially larger fish, overfishing modify size distribution pattern (*i.e.* reducing average size, size-at-age and size-at-maturity), age structure and genetic diversity of targeted species (Botsford *et al.*, 1997; Goñi, 1998).

One of the main problems related to fisheries management, relies on the fact that large levels of natural variability frequently mask the effects of overexploitation. In such cases, initial overexploitation is not detectable until it is severe and often irreversible (Ludwig *et al.*, 1993). Therefore, it is necessary the establishment of long term monitoring, including studies on community and populations to ensure a sustainable management. Fisheries management measures to avoid overfishing include closures (temporary or seasonally) reducing fishing effort and limiting the harvest (Watson *et al.*, 1993), catch number limitation to a more sustainable level (Frid *et al.*, 2003), and establish Minimum Landing Sizes (hereafter MLS) to allow fish to spawn at least once (Jennings *et al.*, 1998; Stewart, 2008; Stergiou *et al.*, 2009). Furthermore, fisheries management should be complemented with other ecosystem-based measures (Stergiou *et al.*, 2009).

Gillnets are fishing gears widely used for commercial purposes (Machelis *et al.*, 1994) and in research activities, since they are useful to monitor catch-length distribution (Hamley, 1975) to determine a population size structure, and to assess fishing effects on exploited stock (McAuley *et al.*, 2007). Gillnet selectivity can be estimated through direct or indirect (*i.e.* species size distribution is unknown) methods, by comparison of the observed catch frequencies across various mesh sizes (Millar and Holst, 1997; Millar and Fryer, 1999). Nevertheless, indirect methods for estimates are more common (Hamley, 1975; Millar and Holst, 1997). The SELECT (Share Each Length's Catch Total) method, is an indirect method implemented by Millar (Millar and Holst, 1997; Millar and

Fryer, 1999; Millar, 2000) that provides a cohesive approach to size selectivity analysis. SELECT is the most widely used in gillnet selectivity studies (Reis and Pawson, 1999; Stergiou and Karpouzi, 2003; Carol and García-Berthou, 2007).

The Ebro River ends up in the Mediterranean Sea forming a high valued Delta in terms of biodiversity and productivity (Day *et al.*, 2006). Nutrient inputs from the river allow the development of a prosperous bivalve aquaculture activity and establish one of the richest fishing areas in the western Mediterranean Sea (Navarro *et al.*, 2009). Artisanal, small-scale fisheries are carried out in the Ebro Delta coastal lagoons. This activity is not submitted to restrictive control measures regarding number of fishing gears per lagoon and species' size reporting, meaning that a high amount of captured fish are directly sold without intermediaries. Although it has been pointed by Sostoa (1983), the effects of this low surveillance effort have not been previously studied and may culminate in an overfishing situation with uncertain consequences. Of major concern is the sand smelt (*Atherina boyeri* Risso, 1810) fishery, a small, short-lived and euryhaline fish that inhabits coastal and estuarine waters (Leonardos, 2001; Patimar *et al.*, 2009). Although *A. boyeri* is catalogued as a vulnerable species in Spain (Doadrio, 2001) and without a MLS established (RD 1615/2005), its fisheries are among the most important on the Spanish Mediterranean coast and with increasing interest (Andreu-Soler *et al.*, 2006). This high interest is probably due to the strong restrictions and high monitoring on the catches of juveniles (with a similar length of *A. boyeri*) of other commercial species such as the sardine (*Sardina pilchardus*), the anchovy (*Engraulis encrasicolus*) and the Atlantic horse mackerel (*Trachurus* spp.), among others, very much appreciated in Spain. In the Murcia region (SE Spain), from 2004 to 2009 catches were 13155 ± 7957 kg/year, with a market value of 5.8 ± 1.55 €/kg (Ministry of Agriculture and Water; Government of Murcia); and in the Ebro Delta coastal lagoons were 2355 kg/year for the last five years (2006-2011), with a minimum of 583 kg and a maximum of 5012 kg, for 2006 and 2008 respectively (St. Pere Fishermen's Association). Thus, the development of a MLS for this species is the first essential step in order to establish an appropriate management action.

The aim of this paper was to consider mesh regulation and choice of fishing nets as a possible approach to prevent overfishing of a vulnerable species in Spanish lagoons. Therefore, we used the following approaches: (i) modelling gillnet selectivity using the SELECT method, as the first attempt to describe gillnet selection patterns for eight species with commercial interest in three Ebro Delta coastal lagoons; (ii) combining

Gillnet and Fyke net information to establish a MLS for *A. boyeri*. This allowed us to develop conservation and management guidelines.

2. Material and methods

2.1 Study area

The Ebro River with 910 km long and a drainage area of 85362 km², is one of the most important tributaries to the Mediterranean Sea, where originates one of the largest deltas (320 km²) in the north-western Mediterranean Basin (Fig. 1). Although it is the Spanish river with the highest mean annual flow (426 m³/s), it shows a high variability between dry (118 m³/s) and wet (569 m³/s) years. This environmental variability is usual in the Mediterranean regions but it is exacerbated by a long history of human induced pressures (Caiola *et al.*, 2001a, b; Ferreira *et al.*, 2007a, b). The majority of the delta plain is devoted to rice agriculture (65% of the total surface) and natural areas cover only about 80 km² (25%). These areas include salt marshes, reed-type marshes, sand dunes, coastal lagoons, natural springs, and bays.

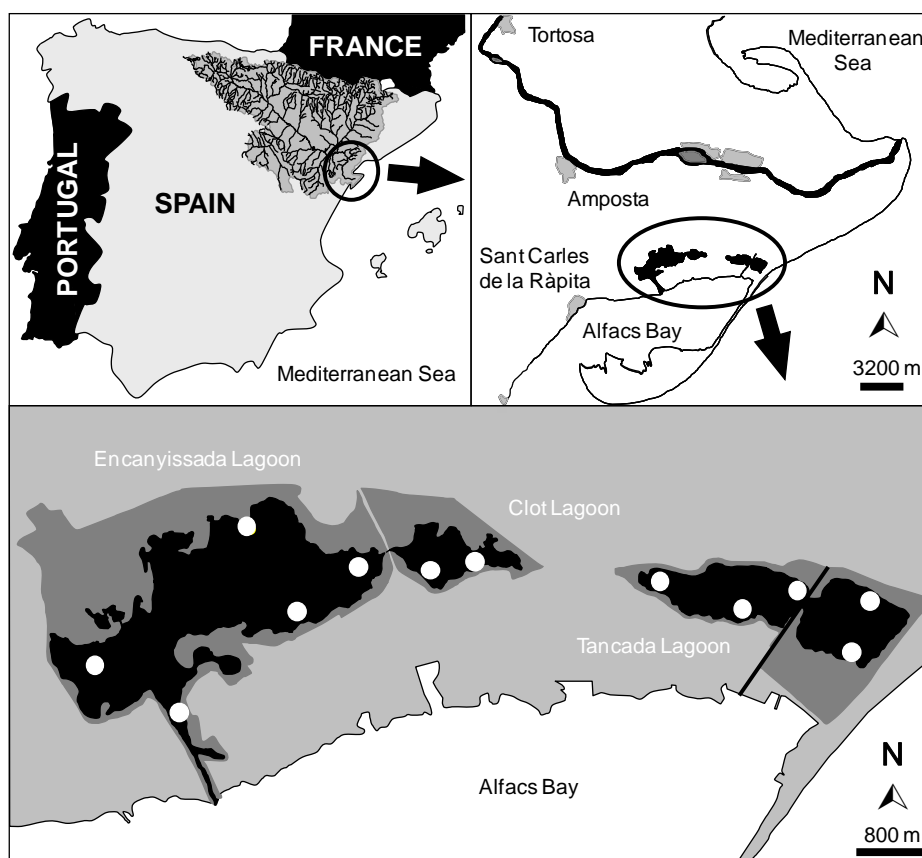


Figure 1. Location of the study area and map of the three lagoons with the sampling points for gillnets and fyke nets (white circles).

The study was conducted in three coastal lagoons from the Ebro River Delta: Encanyissada, Clot and Tancada (Fig. 1). In the Encanyissada lagoon the largest one in the Ebro Delta (4.18 km²), salinity ranges between 3 and 30 and its mean depth is approximately 50 cm. The Clot lagoon (0.56 km² surface area; 0.8-12 salinity range and 30 cm mean depth) although it is part of the Encanyissada lagoon system, is separated by a floodgate showing a different water management scheme which determines differences in the hydrology and ecology between both lagoons. The Tancada lagoon has a 1.85 km² surface area, 37 cm of mean depth and a salinity range between 8 and 36. Both the Encanyissada and Tancada lagoons are connected with the Alfacs bay (Fig.1).

2.2 Sampling methods and data collection

A total of twenty-four multimesh nylon gillnets of 30 x 1.5 m in length and height respectively, were set in the lagoons during two different sampling times (March and September 2008) (Fig. 1). Each gillnet consisted on a series of twelve panels (2.5 meters width) composed by different randomly distributed meshes ranging from 5.0 to 55.0 mm bar length (see Table 1 for mesh sizes) to avoid confusion of the mesh size with environmental gradients, and following a geometric progression to optimize efficiency (Kurkilahti *et al.*, 2002). The nets were made of monofilament twine (ranging from 0.10 to 0.20 mm depending on the panel) and the hanging ratio oscillated between 0.493 and 0.5 depending on the panel mesh size. Nets were set on late afternoon and hauled the next morning, hence being an average soak time of 12 hours. Twenty-four fyke nets of 2 meters long with a hoop diameter of 80 cm and 5.0 mm mesh size, were set and hauled at the same time and sampling points of the gillnets. All fish specimens were identified to species level sorted by mesh size, and fork length (hereafter FL) was measured to the nearest millimetre.

A total of 164 females of *Atherina boyeri* from the Ebro Delta coastal lagoons were collected in 1982 during the reproduction months (Sostoa, 1983). In the same year, the fish specimens were measured for total body length (to the nearest mm) and dissected. The gonads were kept in Gilson's fluid (Bagenal & Braum, 1978) so oocytes, once separated, could be counted and classified according to their development stage. The presence of ripe oocytes in the ovaries of each female, which indicates maturity, was then recorded. This dataset was used to establish the relationship between body length and maturity stage in order to determine the MLS of the species.

2.3 Data analysis

Gillnet selectivity for each species (Table 1) was estimated by the SELECT method (Share Each Length's Catch Total) through R (2.8.1 version) code developed by Russell Millar (available at: <http://www.stat.auckland.ac.nz/~millar/selectware/R>). The SELECT method is a generalized linear model that assumes a Poisson distribution of the gillnet catch data to fit them to a specified model using maximum-likelihood (Millar, 2000). Selectivity curves are defined as the relative probability of a fish of a given length to be captured when contacting to a mesh of a determinate size (Millar, 2000). Five models were tested, the four previously available in the R scripts of SELECT (Millar, 2000) (normal location, normal scale, gamma and lognormal) and the inverse Gaussian model, developed and added by us (Table 2). All five models are unimodal and consist of two parameters describing the location and dispersion of the curves. The normal location and normal scale models are based on the normal distribution, whereas the other three are skewed curves with positive asymmetry. In addition to its statistical accuracy and availability of several models, SELECT method analyses the data of all meshes within a single model, increasing statistical precision and power (Carol and García-Berthou, 2007). All models were fitted under the assumption of equal effort of mesh size and assuming fishing power proportional to mesh size. The goodness of fit was performed by referring model deviances to a chi-squared distribution with df degrees of freedom (Madsen *et al.*, 1999), with the lowest deviance values corresponding to the best fitting model ($p < 0.05$ denotes lack of fit). The Baranov's principle of geometric similarity (Baranov, 1948) is the main used assumption of gillnet selectivity. This principle interprets gillnet captures as a mechanical process that depends only on the relative geometry of the mesh and the fish, stating that "*Since all meshes are geometrically similar and all fish of the same species (within a reasonable size range) are also geometrically similar, the selectivity curves for different mesh sizes must be similar*" (Hamley, 1975). Four of the models fitted here showed geometric similarity with both length and spread of the curves increasing proportionally to mesh size and only the normal location curve (with the modal length proportional to mesh size but the spread fixed over mesh sizes) (Millar and Fryer, 1999), did not present the geometric similarity.

Atherina boyeri minimum landing size (MLS) was estimated by a logistic regression model relating the total body length as the quantitative predictor with the maturation stage as a qualitative response variable ("mature female" vs. "immature female"). This model allowed the estimation of the female probability to be mature according to its total

length. The χ^2 value (and its associated p value) that compares the full model with the constant-only model and Nagelkerke's R^2 , which is similar to the ordinary determination coefficient (Tabachnick and Fidell, 2007), are reported for the logistic regression. Size-at-maturity (L_x) of a population, is the total length (TL) at which $X\%$ of the individuals are mature and was estimated from the fit of logistic regression (Stergiou *et al.*, 1996; Somarakis *et al.*, 2005). The percentage of *A. boyeri* mature individuals were compared among gears and mesh sizes with a G-test of independence (Sokal and Rohlf, 1995). Analysis of variance (ANOVA) was used to compare model deviances between both approaches and to compare deviances among models and fished species, followed by Games-Howell post-hoc tests (hereafter, GH tests). The GH tests are among the most powerful and most robust to unequal variances of post-hoc multiple comparison methods (Day and Quinn, 1989). The association of model deviances with catch number and number of meshes in which one species is captured was analysed with Spearman's correlation coefficient (r_s). All statistical analyses were performed using SPSS v.17.0 software.

3. Results

3.1 Fitting selectivity curves

A total of 2297 fish belonging to 20 different fish species were gillnet caught (24 nets) and valid for SELECT analysis. Nevertheless, for gillnet selectivity estimation only fish species captured in at least four different panels and in enough number of individuals were considered (Tables 1 and 2). The mean length of captured fish increased with mesh size (Fig. 2). Although wedged and entangled fish were excluded from gillnet selectivity analysis, there were a few fish in mesh sizes larger than expected (e.g. one 199 mm *Mugil cephalus* in the 15.5 mm mesh) or in smaller meshes than expected (e.g. one 316 mm *Liza ramada* in the 8 mm mesh or several *Liza saliens*) (Fig. 2). Also quite apparent was the increased size variability of catches with increasing meshes (*i.e.* geometric similarity) particularly in fish species with a higher number of captures (e.g. *Atherina boyeri*, *L. saliens*, *L. ramada*, *Pseudorasbora parva*) (Fig. 2). A contrasting pattern was found related to fish length range, so, the smaller *A. boyeri*, *L. ramada*, *P. parva*, and *Alburnus alburnus* were mainly captured in smaller mesh sizes, while larger *Cyprinus carpio* and *Carassius auratus* were principally caught in larger mesh sizes. The rest of fish species, with larger length ranges, were captured in a wider number of meshes (Table 1).

Table 1. Total gillnet captures (*N*) by species and mesh size. Species mean size (fork length or total length (*)) depending on the species) and range (mm) is also shown.

Fish species		Mesh size (mm)													Fish length	
Species name	Common name	<i>N</i>	5.0	6.2 5	8.0	10. 0	12. 5	15. 5	19. 5	24. 0	29. 0	35. 0	43. 0	55. 0	Mean	Range
<i>Anguilla anguilla</i>	Eel	1									1				320 *	320 *
<i>Atherina boyeri</i>	Sand smelt	1172	361	500	303	8									65	40-90
<i>Engraulis encrasicolus</i>	European anchovy	3		3											68	67-69
<i>Alburnus alburnus</i>	Bleak	29	2		14	2	6	5							91	45-136
<i>Barbus graellsii</i>	Barbel	3				1	1		1						139	90-185
<i>Carassius auratus</i>	Goldfish	12						1	1	2	2	3		3	177	82-324
<i>Cyprinus carpio</i>	Carp	36						3	5		5	13	5	5	187	81-331
<i>Pseudorasbora parva</i>	Topmouth gudgeon	175	1	45	75	48	5			1					70	41-96
<i>Chelon labrosus</i>	Thicklipped grey mullet	16			1	7	1	1		2	2	1	1		190	72-360
<i>Liza aurata</i>	Golden mullet	29		4		1	6	6	4	5		1	1	1	206	55-387
<i>Liza ramada</i>	Thinlip mullet	389		3	14	86	130	44	43	48	10	6	3	2	184	64-395
<i>Liza saliens</i>	Leaping mullet	341	15	65	53	56	78	51	17	6					140	32-310
<i>Mugil cephalus</i>	Flathead mullet	26				1		1	1	12	3	2	5	1	247	83-495
<i>Dicentrarchus labrax</i>	Sea bass	12					1	2	5		2		2		198	110-337
<i>Pomatoschistus microps</i>	Common goby	25	25												40 *	35-49 *
<i>Sparus aurata</i>	Gilthead seabream	6								4	2				144	127-169
<i>Solea senegalensis</i>	Senegalese sole	1							1						244*	244*
<i>Sander lucioperca</i>	Pikeperch	18					1	5	3	5	4				251	215-281
<i>Silurus glanis</i>	Wels catfish	2								1	1				205 *	196-213 *
<i>Syngnathus abaster</i>	Black-striped pipefish	1		1											80 *	80 *

Table 2. Fitting parameters of the five models tested with the SELECT method by species (in **bold** best model fit) in the Ebro Delta lagoons. Parameters 1 and 2 are k and σ for normal location model; k_1 and k_2 for normal scale model (spread proportional to mesh size); α and k for gamma model; μ_1 and σ for lognormal model and k_1 and k_2 for inverse Gaussian model. Deviance statistic measure goodness of fit. Significant results $p < 0.05$ indicate lack of fit.

Species	Model	Equal fishing power					Fishing power relative to mesh size				
		Par.1	Par. 2	Dev	df	p	Par.1	Par. 2	Dev	df	p
<i>A. boyeri</i>	Normal fixed	9.57	4.21	82.69	142	1.000	9.62	4.22	85.04	142	1.000
	Normal scale	9.74	0.41	42.02	142	1.000	9.78	0.41	42.02	142	1.000
	Gamma	225.68	0.04	41.74	142	1.000	226.6	0.04	41.74	142	1.000
	Lognormal	3.88	0.07	42.56	142	1.000	3.89	0.07	42.56	142	1.000
	Inverse Gaussian	9.72	2158.6	42.62	142	1.000	9.76	2168.1	42.62	142	1.000
<i>C. labrosus</i>	Normal fixed	10.19	63.21	45.78	96	1.000	10.80	63.26	46.86	96	1.000
	Normal scale	11.71	11.11	46.83	96	1.000	12.58	9.67	47.47	96	1.000
	Gamma	13.92	0.81	42.11	96	1.000	14.92	0.81	42.11	96	1.000
	Lognormal	4.45	0.25	40.10	96	1.000	4.51	0.25	40.10	96	1.000
	Inverse Gaussian	11.16	163.45	40.04	96	1.000	11.94	173.13	39.92	96	1.000
<i>L. saliens</i>	Normal fixed	11.77	33.68	712.02	1020	1.000	12.41	34.53	686.8	1020	1.000
	Normal scale	12.62	10.50	766.22	1020	1.000	13.43	9.55	775.2	1020	1.000
	Gamma	16.66	0.75	672.19	1020	1.000	17.66	0.75	672.1	1020	1.000
	Lognormal	4.09	0.24	640.07	1020	1.000	4.15	0.24	640.0	1020	1.000
	Inverse Gaussian	12.41	200.11	640.27	1020	1.000	13.19	210.99	638.6	1020	1.000
<i>L. aurata</i>	Normal fixed	10.60	102.27	106.46	230	1.000	12.57	117.33	110.1	230	1.000
	Normal scale	14.84	25.16	103.93	230	1.000	16.32	20.11	106.5	230	1.000
	Gamma	8.31	1.76	93.64	230	1.000	9.31	1.76	93.64	230	1.000
	Lognormal	4.45	0.37	90.47	230	1.000	4.59	0.37	90.47	230	1.000
	Inverse Gaussian	14.89	100.03	90.01	230	1.000	17.30	110.83	89.69	230	1.000
<i>L. ramada</i>	Normal fixed	9.84	55.98	993.76	1418	1.000	10.83	59.40	962.2	1418	1.000
	Normal scale	10.69	5.22	782.44	1418	1.000	11.17	4.91	784.1	1418	1.000
	Gamma	19.21	0.56	771.94	1418	1.000	20.21	0.56	771.9	1418	1.000
	Lognormal	4.18	0.24	787.13	1418	1.000	4.24	0.24	787.1	1418	1.000
	Inverse Gaussian	10.81	173.98	795.73	1418	1.000	11.49	183.47	797.2	1418	1.000
<i>M. cephalus</i>	Normal fixed	8.23	25.00	37.67	173	1.000	8.32	25.23	38.22	173	1.000
	Normal scale	8.57	0.60	30.09	173	1.000	8.64	0.59	30.08	173	1.000
	Gamma	116.11	0.07	30.44	173	1.000	117.1	0.07	30.44	173	1.000
	Lognormal	4.45	0.09	30.68	173	1.000	4.45	0.09	30.68	173	1.000
	Inverse Gaussian	8.56	959.04	30.68	173	1.000	8.63	967.36	30.68	173	1.000
<i>A. alburnus</i>	Normal fixed	8.93	10.26	30.76	102	1.000	9.03	10.32	31.69	102	1.000
	Normal scale	9.26	0.82	24.34	102	1.000	9.35	0.82	24.34	102	1.000
	Gamma	99.81	0.09	24.40	102	1.000	100.8	0.09	24.40	102	1.000
	Lognormal	3.83	0.10	24.48	102	1.000	3.84	0.10	24.48	102	1.000
	Inverse Gaussian	9.28	898.95	24.46	102	1.000	9.38	907.97	24.46	102	1.000
<i>C. carpio</i>	Normal fixed	5.41	22.24	49.27	173	1.000	5.48	22.41	48.76	173	1.000
	Normal scale	5.48	0.30	37.99	173	1.000	5.54	0.30	37.98	173	1.000
	Gamma	92.97	0.06	39.14	173	1.000	93.97	0.06	39.14	173	1.000
	Lognormal	4.44	0.11	39.81	173	1.000	4.45	0.11	39.81	173	1.000
	Inverse Gaussian	5.49	486.32	39.83	173	1.000	5.55	491.67	39.84	173	1.000
<i>C. auratus</i>	Normal fixed	4.87	15.84	14.47	58	1.000	4.94	15.99	14.70	58	1.000
	Normal scale	5.08	0.36	14.27	58	1.000	5.15	0.35	14.27	58	1.000
	Gamma	72.13	0.07	14.19	58	1.000	73.13	0.07	14.19	58	1.000
	Lognormal	4.36	0.12	14.18	58	1.000	4.37	0.12	14.18	58	1.000
	Inverse Gaussian	5.09	363.48	14.17	58	1.000	5.16	368.49	14.17	58	1.000
<i>P. parva</i>	Normal fixed	8.23	10.73	212.43	233	0.829	8.43	11.04	210.3	233	0.855
	Normal scale	8.59	0.82	107.77	233	1.000	8.68	0.81	107.7	233	1.000
	Gamma	83.15	0.10	114.58	233	1.000	84.15	0.10	114.5	233	1.000
	Lognormal	3.76	0.11	121.40	233	1.000	3.77	0.11	121.4	233	1.000
	Inverse Gaussian	8.60	661.77	123.50	233	1.000	8.72	670.13	123.6	233	1.000
<i>S. lucioperca</i>	Normal fixed	11.13	71.00	47.81	62	0.908	12.20	79.83	48.02	62	0.904
	Normal scale	12.20	25.52	48.54	62	0.894	14.10	22.41	48.61	62	0.893
	Gamma	9.40	1.43	48.08	62	0.903	10.40	1.43	48.08	62	0.903
	Lognormal	5.10	0.32	47.83	62	0.907	5.20	0.32	47.83	62	0.907
	Inverse Gaussian	13.78	131.87	47.81	62	0.907	15.28	144.43	47.79	62	0.908
<i>D. labrax</i>	Normal fixed	8.45	12.49	6.48	46	1.000	8.52	12.48	6.48	46	1.000
	Normal scale	8.58	0.54	6.11	46	1.000	8.64	0.54	6.11	46	1.000
	Gamma	137.23	0.06	6.02	46	1.000	138.2	0.06	6.02	46	1.000
	Lognormal	4.67	0.09	5.99	46	1.000	4.68	0.09	5.99	46	1.000
	Inverse Gaussian	8.58	1179.5	5.99	46	1.000	8.64	1188.0	5.99	46	1.000

Fitting parameters estimated by the SELECT method for all models and fish species are given in Table 2 (also see Fig. 3 for best model fitting the eight most captured species). Overall, it was not observed a common pattern, since best fitting model varied among fish species. Assuming equal fishing power for all meshes, the normal scale (proportional spread) model had the lowest deviance value (*i.e.* better fit) for *Mugil cephalus*, *Alburnus alburnus*, *Cyprinus carpio* and *Pseudorasbora parva* (Fig. 3). For the *Chelon labrosus*, *Liza aurata*, *Carassius auratus* and *Dicentrarchus labrax* (but only on the third decimal place), the best fit model was the inverse Gaussian model (Fig. 3). The gamma distribution model had the lowest deviance for the *Atherina boyeri* and *Liza ramada*; thus inverse Gaussian model was the best fit when having low number of captures, while gamma was the best fit with higher captures. For *Liza saliens* the best fit model was the lognormal model; and for *Sander lucioperca* was the normal location model (fixed spread) but only on the third decimal place (Table 2; Fig. 3). For all fish species, goodness of fit tests (Table 2) indicated no deviation of the observed catch for the best model predictions ($p > 0.90$). None of the models showed a lack of fit ($p < 0.05$) indicating the accuracy of results obtained. But, the normal location (for 9 species) and the normal spread model (for the rest of the species) were a worse fit than the rest of the models (Table 2). Similar results were obtained when fishing power relative to mesh size was assumed; only for *L. saliens* and *S. lucioperca* the best fit model changed to inverse Gaussian model due to smaller deviances values changes (Fig. 2). Thus, both approaches were suitable to estimate gillnet selectivity. Model deviance (Table 2) also did not show significant differences between both gillnet estimation approaches (ANOVA, $F_{1, 44} = 0.75$, $p = 0.39$) but significantly depended on fish species ($F_{11, 44} = 51776.4$, $p < 0.0001$) mostly because an effect of sample size (correlation between deviance and catches; Spearman's $r_s = 0.71$, $p < 0.001$) and mainly due to the number of meshes in which one species was captured (Spearman's $r_s = 0.52$, $p < 0.0001$), since species with larger samples sizes and captured by a wider range of meshes had larger deviance than less captured species or present in only a few number of panels (Tables 1 and 2). There was also a significant model \times species interaction ($F_{44, 44} = 121.92$, $p < 0.001$) because the fit of different models showed opposite patterns in different species; for instance when the normal scale model was the best fit (lowest deviance), normal location was the worst (*e.g.* *P. parva* and *A. alburnus*) when the normal location or lognormal were the best, normal scale was the worst (*e.g.* *S. lucioperca* and *L. saliens*). There was no significant evidence for both approach \times model or approach \times species interactions ($p > 0.16$) effects.

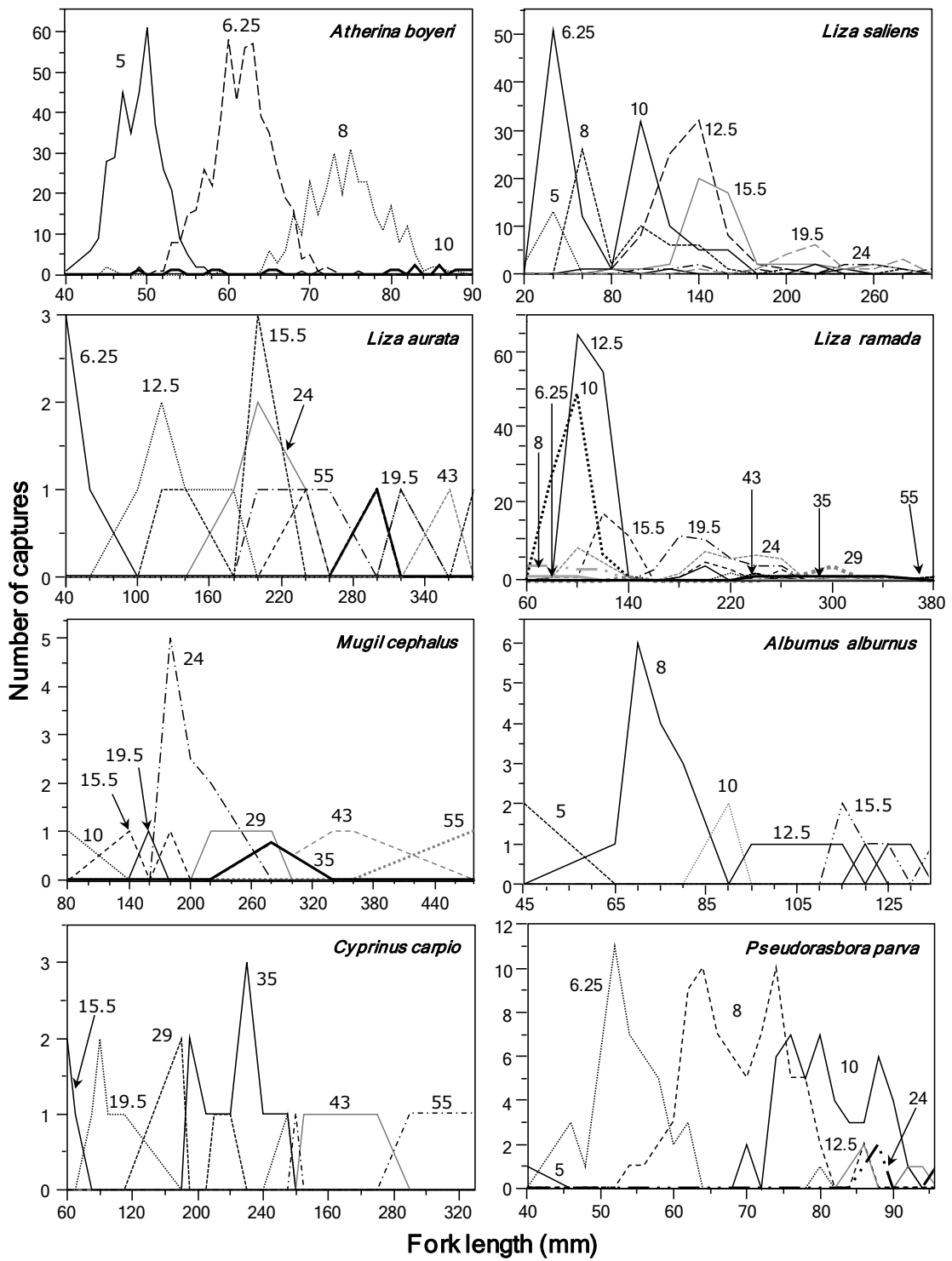


Figure 2. Catch frequency per fork length class of the eight most captured fish species in the Ebro Delta lagoons (Figures shown above lines = bar length in mm).

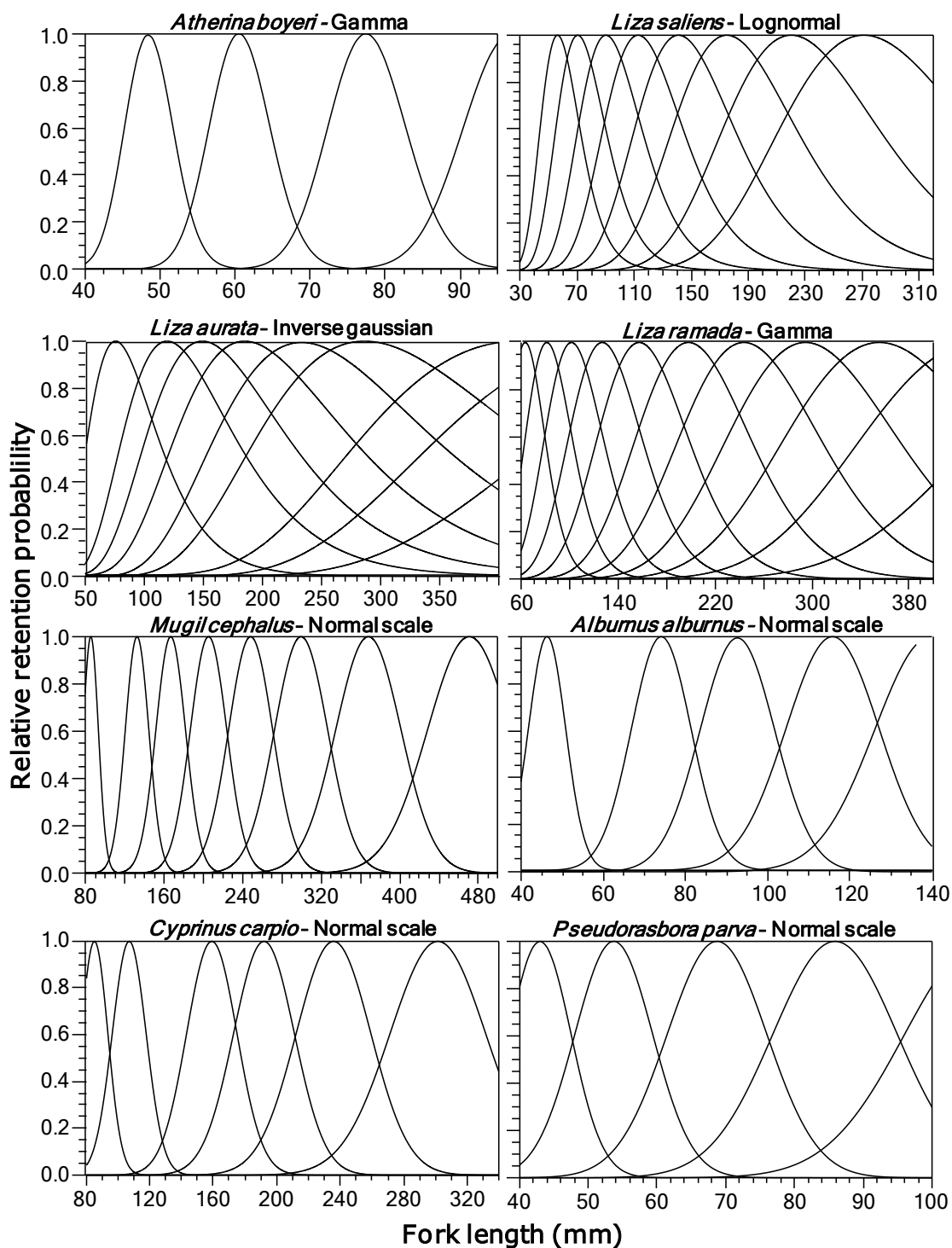


Figure 3. Fitted selective curves for the eight most captured fish ($N \geq 20$) assuming equal fishing power to mesh size: Only model with best fit for each species is shown (Table 2). Meshes are shown in size ascending order consecutively from 5.0 mm mesh size (see Table 1).

3.2 A. *boyeri* selectivity and minimum landing size

A total of 1625 *Atherina boyeri* individuals were captured in the three lagoons through fyke net and gillnet gears. By number, the *A. boyeri* only represented the 3.8% of the total catches from the fyke nets, being the fourth most important species ($N = 453$), after

Pomatoschistus microps (6161), *Gambusia holbrooki* (3890) and *Aphanius iberus* (694); but it was the main catch species by gillnets (1172) (Table 1) corresponding to the 51% of the gillnet total captures. *A. boyeri* was only captured by the four smaller meshes (from 5.0 to 10.0 mm mesh; see Table 1). *A. boyeri* mean fork length was significantly different between gears (ANOVA, $F_{1, 1413} = 444.1$, $p < 0.001$), and fish length from fyke nets was smaller ($\mu \pm$ standard deviation = 46.66 ± 12.52) than those from gillnets (61.14 ± 10.50). Mean fork length was also significantly different among gillnet meshes ($F_{4, 1410} = 870.89$, $p < 0.001$); with fish length related to mesh size 5.0 mm (49.08 ± 3.21); 6.25 mm (61.56 ± 3.92); 8.0 mm (74.21 ± 5.48) and 10.0 mm (80.80 ± 8.62). Thus fyke net and the 5.0 mm mesh size panel captured the smallest individuals, while the 8.0 and 10.0 mm panels caught the largest fish (GH post-hoc tests; fyke nets < 5.0 mm $\ll 6.25$ mm $\ll 8.0$ mm = 10.0 mm). Summarizing, fyke nets captured less and smaller individuals than gillnets, and the 6.25 mm mesh size panel and higher, were the best for the fishery of *A. boyeri*; since an increase in both catches and fish length were observed.

From the total *Atherina boyeri* individuals examined for maturity ($N = 164$), all fish with a total length below 45 mm were sexually immature; and fish with a length over 55 mm were all mature, so maturity was significantly related to fish length ($p < 0.0001$) (Fig. 4). The L_{50} of the *A. boyeri* (estimated from the logistic model) was 52.27 mm TL (Fig. 4), and less than 2 mm more were necessary to reach the L_{75} (53.92 mm TL). The L_{95} and L_{99} were estimated from the logistic model at sizes of 56.67 and 59.12 mm total length respectively. Interestingly, the percentage of mature individuals captured significantly varied among meshes ($G_4 > 1495.393$; $p < 0.0001$ for L_{50} , L_{75} and L_{95}); and the percentage of mature fish caught were positively related to mesh size ($r_s > 0.542$; $p < 0.0001$ for L_{50} , L_{75} and L_{95}) (Fig. 5). Therefore, fyke nets and 5.0 mm mesh gillnets captured less and smaller individuals and with a higher percentage of immatures, than larger meshes. Increasing mesh size only 1.25 mm (*i.e.* up to 6.25 mm), the number of captures increased with a percentage of mature individuals close to 100% (Fig. 5).

4. Discussion

4.1 Gillnet-fit and selectivity curves

The normal scale model had the best fit in four of the twelve fish species. The lognormal and the normal location models had the best fit for one fish species; only on equal-effort approach combined with inverse Gaussian model on unequal-effort approach, with the best fit for four different fish species. The gamma model, showed the

Figure 4. Relationship between maturity stages of *Atherina boyeri* with total length. Circles shown the maturity of the individuals (0; immature and 1; mature) and black line illustrates the predicted probability to be mature at a determined length.

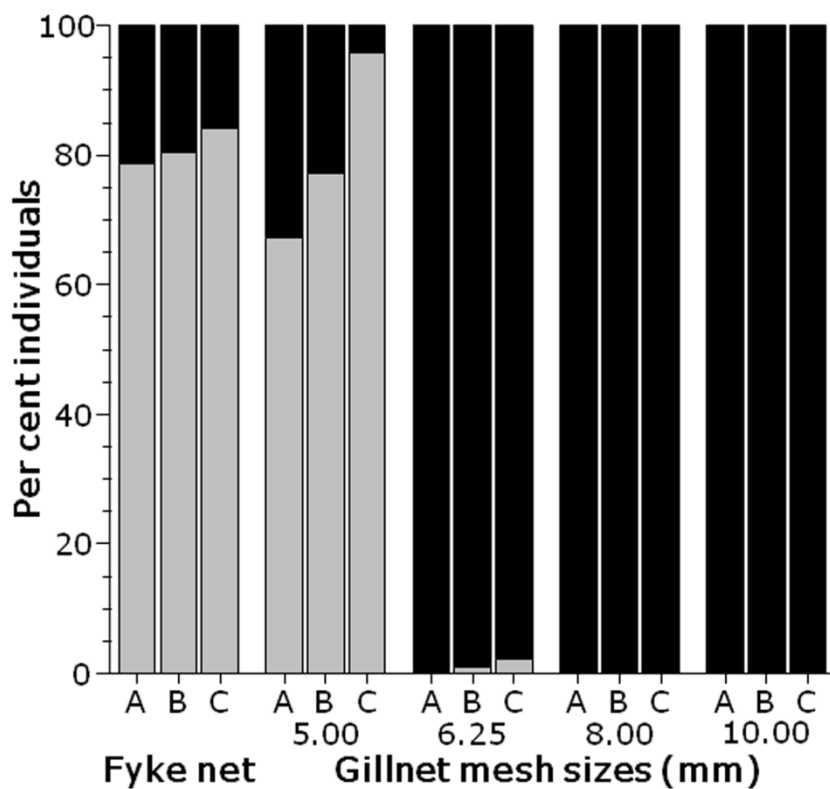
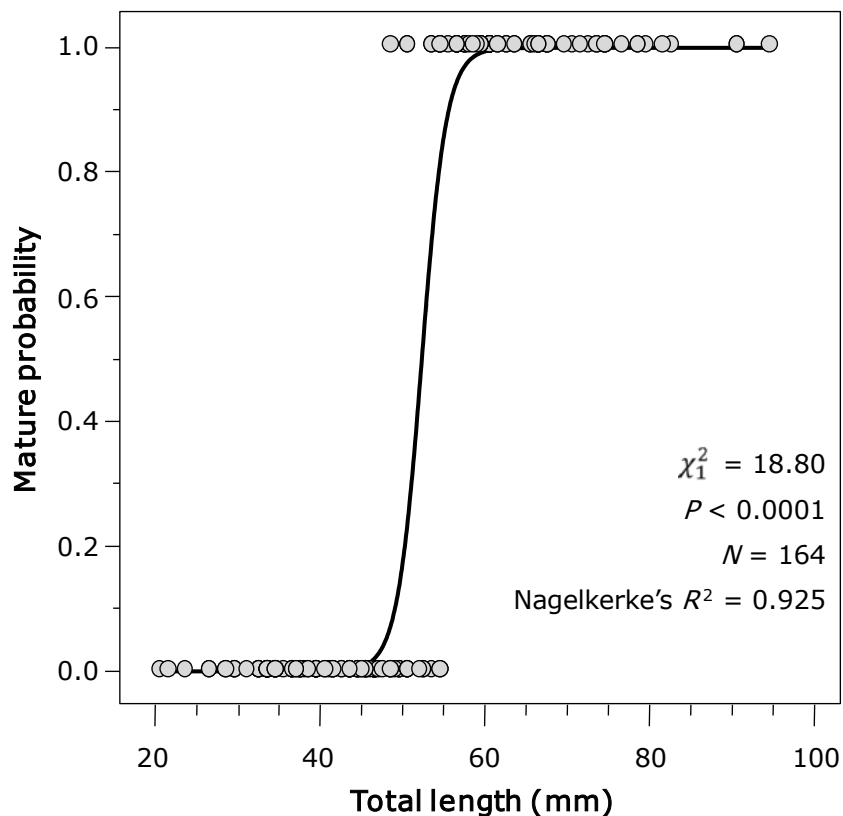


Figure 5. Percentage of mature (■) and immature (□) individuals of *A. boyeri* by fyke nets and gillnets (by mesh size) per different lengths at maturity (A = L_{50} , B = L_{75} , C = L_{95}) estimated from the fit of logistic regression (see Material and Methods).

best fit for two fish species; any model, also, showed a lack of fit indicating the accuracy of the results. In general, the worst approach was achieved by the normal fixed model mostly due to its fixed spread, because assumes no geometrical symmetry (McAuley *et al.*, 2007). Overall, gillnet selectivity curves approaches normal curves when most fish are wedged or gilled, but when many fishes are entangled or snagged, catch data are skewed to the right and are better fitted by gamma, lognormal, inverse Gaussian models or multimodal mesh models (Hamley, 1975; Hovgård, 1996; Dos Santos *et al.*, 2003; Erzini *et al.*, 2003). According to this, some studies have reported that the binomial model may better fit than unimodal models, when fishes are caught by a combination of different processes (*i.e.* gilled, entangled, wedged or snagged) and therefore smaller or larger fish than expected are found in a determinate mesh (Holt, 1963; Hovgård, 1996; Erzini *et al.*, 2003). Nevertheless, although the binomial model was not available in the R-scripts of SELECT, in the present study the presence of entangled or wedged fish were lower than the 2%, so bimodality was not evident in our catch data.

Model fit behaved opposite pattern in different species; when the normal location or lognormal models were the best (lowest deviance), normal scale was the worst. Residual symmetry analysis provides an explanation for this opposite behavior (Carol and García-Berthou, 2007). All models, except normal location model, incorporate positive asymmetry, thus when normal location model had the lower fit (*e.g.* *Pseudorasbora parva*), deviance length residuals had a positive asymmetry, disappearing in the rest of the models that incorporated it. Nevertheless, for species best fitted with the normal location model, uniform and symmetric residuals are expected and a length variability reduction with increasing mesh size; meanwhile models with positive asymmetry would produce larger residuals with negative asymmetry (see Millar and Holst, 1997) since the variability increase with increasing mesh size (*i.e.* geometric similarity). This variability may have been influenced by different factors, for instance if fishes were entangled or wedged; also, differences in body shape, allometric growth, girth at the point of capture, different behavior towards gillnets and the presence of appendages are important in determining how fish are caught by gillnets (Campos and Fonseca, 2003), and if selection curves are skewed or bell-shaped (Hamley, 1975).

Species with typical fish body shape such as the cyprinids *Cyprinus carpio*, *Pseudorasbora parva* or *Alburnus alburnus* have lesser allometry, existing a clear linkage between fish length and shape and both increasing with gillnet mesh size. Thus they were best fitted with normal scale model. However, species that have greater allometry, for instance the *Carassius auratus* (higher body depth - fish length

relationship) and mugilids (higher fish length - body girth relationship), or species with gill bony appendages such as *Sander lucioperca* and *Dicentrarchus labrax*, were best fitted with skewed models (gamma, lognormal and inverse Gaussian models).

4.2 Gear selectivity implication on the *A. boyeri* fisheries management

According to the Code of Conduct for Responsible Fisheries (FAO, 1995) fisheries management strategy should be developed from a multispecific point of view. However, multispecies management is arduous to implement due to the difficulty of having enough information on species biology, ecology and interactions among species (Sainsbury *et al.*, 2000; Campos and Fonseca, 2003). *Atherina boyeri* is a widespread and locally abundant species along the Mediterranean Sea, and it is commercially exploited in different Mediterranean areas such as Croatia, Greece or Italy (Maci and Basset, 2010) being one of the most important fisheries in the Spanish Mediterranean coast (Andreu-Soler *et al.*, 2003). For instance its captures triplicate in a period of three years in Mar Menor (1998-2000), being one of the commercial species of major interest in this area (Andreu-Soler, 2006). This commercial interest contrast with the lack of studies on its biology and ecology mainly when referred to the Spanish Mediterranean coast, where only Sostoa (1983) and Andreu-Soler *et al.* (2003) have described some aspects of its biology in the Ebro River Delta and Mar Menor lagoon respectively. This lack of knowledge is translated into an absence of the fisheries regulation. Although the establishment of an adequate MLS is the most basic measure in fisheries management, (Stewart, 2008; Stergiou *et al.*, 2009), *A. boyeri* was the only commercial species present in the lagoon without a MLS regulated by the European Union (CE 1967/2006) or the Spanish government (RD 1615/2005). In order to enhance fish recruitment and juvenile survival, it is necessary to establish a valid MLS and to improve the knowledge of the fisheries gears selectivity (Wileman *et al.*, 1996). In the Ebro Delta coastal lagoons, *A. boyeri* represented only 3.8% of total fyke nets catches but accounted for 51% of gillnets catches; although it was present in only four of the 12 gillnet panels. For the three smaller gillnet panels (5.0 mm, 6.25 mm and 8.0 mm) separately, *A. boyeri* accounted for 89%, 81% and 66% respectively. Fyke nets fished all the *A. boyeri* size range and significantly captured smaller individuals (*i.e.* higher proportion of immature fish) than gillnets, even when compared with gillnet 5.0 mm mesh size. Different gillnet meshes captured a narrow size range with fish mean length increasing on larger panels and reducing the proportion of immature fish captured. Thus, in the case of *A. boyeri*, gillnets demonstrated its higher specific and size selectivity usefulness, suggesting that they could be a better tool for the assessment of *A. boyeri* fisheries allowing, thus, obtaining

more suitable data for management recommendations when compared to fyke nets. Since the selectivity of the gear is the major factor determining the exploitation of a fish stock (Suuronen *et al.*, 2007) should be related to a valid MLS. Furthermore, an ecological MLS should be close to L_{50} , but if the management aim is to protect juveniles, then the L_{75} criteria would be applied (Stewart, 2008). In our study we found that mesh sizes over 6.25 mm had a similar effect in *A. boyeri* catch, independently of both L_{50} and L_{75} approaches, even if a L_{95} was assumed. Thus, from a fisheries management point of view mesh sizes lower than 6.25 mm should be avoided, together with fyke nets of the same mesh sizes. The use of 6.25 mm bar length mesh appeared to be the most appropriate tool for the exploitation and management of *A. boyeri* fishery, since it captured higher number of individuals allowing the establishment of a L_{75} MLS (near 54 mm TL) in order to enhance juvenile survival (Stewart, 2008).

Gear selectivity and valid MLS are commonly used to improve fisheries management, restricting the size of fish captured (Machelis *et al.*, 1994; Dos Santos *et al.*, 2003; Fonseca *et al.*, 2005). As far as we are concern, this is the first time that those are applied to the fisheries of a small fish species, and particularly to *A. boyeri* (in our study $TL \leq 100$ mm). This framework must be the first step to stop the low surveillance fishing situation. Thus, future research should focus on exploring how management actions may affect *A. boyeri*, the rest of fish community and the relationship among them in order to establish a global management plan.

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Assessing artisanal fisheries impact in a Ebro Delta coastal lagoon with an Ecopath with Ecosim mass-balanced model

Rodríguez-Climent S, Moreau J, Ibáñez C, Caiola N

(to be submitted)

Assessing artisanal fisheries impact in a Ebro Delta coastal lagoon with an Ecopath with Ecosim mass-balanced model

Rodríguez-Climent, Sílvia ^{1*}; Moreau, Jacques ²; Ibáñez, Carles ¹ and Caiola, Nuno ^{1*}

¹ *IRTA Aquatic Ecosystems, Crta. de Poble Nou, Km 5.5. E-43540 Sant Carles de la Ràpita, Catalonia, Spain*

² *ENSAT- École Nationale Supérieure Agronomique de Toulouse, France*

Abstract

A mass balanced trophic model for the Tancada lagoon (Ebro Delta, NE Spain) using the Ecopath with Ecosim software (EwE) was constructed and used to simulate two different fisheries scenarios. The model was composed by 18 functional groups including fish, primary producers, invertebrates, birds and detritus. Data on biomass, production/biomass, consumption/biomass, and diet composition for each group were either estimated or adapted from the literature. Official landings statistics from the period 1965-2010 concerning Tancada lagoon artisanal fisheries were provided by the St. Pere Fisherman association and used to calibrate the catch data with Ecosim. Further simulations one predicting an increase and another a decrease of the current fishing effort were done without time-series fitting.

The model outcomes allowed identifying the structure of the food web and all the biological interactions taking place in the ecosystem. Our results reinforce the hypothesis that artisanal fisheries in the area take advantage of the migratory movements of some species as Sea bass (*Dicentrarchus labrax*), adult eel (*Anguilla anguilla*) and gilthead seabream (*Sparus aurata*). Future predictions point out gradual reduction of the current fishing effort as the best strategy to ensure both fishery sustainability and conservation of the Ebro Delta coastal lagoons.

Keywords: Ecosystem modelling, Ecopath, Ecosim, fish community, food web

* Corresponding author E-mail: silviarodriguezcliment@gmail.com

1. Introduction

Coastal lagoons are shallow bodies of brackish or sea water, partially separated from an adjacent coastal sea (Barnes, 1980). These systems are highly productive and function as nursery areas and feeding grounds for marine and estuarine species that support important fisheries (Pérez-Ruzafa *et al.*, 2004; Rodríguez-Climent *et al.*, 2013). Moreover, coastal lagoons are usually considered priority habitats (see Habitats Directive 92/43/EEC) due to their high importance in ensuring biodiversity conservation (European Parliament, 1992). Fishing activity is one of the major human disturbances to coastal lagoons, affecting target and non-target fish species, some of these endangered and protected. Fisheries can also cascade down across trophic levels affecting the functioning of the whole ecosystem (*e.g.*, Coll *et al.*, 2006a). Therefore, an integrative approach needs to be adopted to understand how fishing activity may impact the complex food web structure of ecosystems (Coll *et al.*, 2006a; Pauly *et al.*, 2002). In this context, ecosystem modelling is considered a useful tool, mostly because it can assess the trophic interactions of an ecosystem resulting from direct or indirect human activities (*e.g.* fisheries), and predict the ecological impacts that fisheries may cause on the trophic web (Cury and Christensen, 2005; Piroddi *et al.*, 2010). Among modelling packages, Ecopath with Ecosim (EwE; Christensen *et al.*, 2008), has been widely used to assess the status of exploited aquatic ecosystems (Christensen and Walters, 2004). In the last decade, the number of studies using EwE for fisheries assessment has increased exponentially, and models have been developed for different kinds of ecosystems including marine and estuarine habitats and lentic and lotic waters (Christensen and Pauly, 2004; Christensen and Walters, 2004; Morissette, 2007; Piroddi *et al.*, 2010). In the Mediterranean Sea, EwE models have also been applied to diverse types of ecosystems (Coll and Libralato, 2012) including: coastal zones (Albouy *et al.*, 2010; Libralato *et al.*, 2006; Pinnegar, 2000; Pinnegar and Polunin, 2004; Piroddi *et al.*, 2010; Tsagarakis *et al.*, 2010; Valls, 2009), shelf and slope areas (Coll *et al.*, 2006a; Coll Montón, 2006; Coll *et al.*, 2007; Coll *et al.*, 2008) and coastal lagoons (Brando *et al.*, 2004; Carrer and Opitz, 1999; Libralato *et al.*, 2002; Libralato and Solidoro, 2009; Palomares *et al.*, 1993; Pranovi *et al.*, 2003). Yet, an ecotrophic model using EwE has never been constructed for coastal lagoons of the Ebro Delta.

The Ebro Delta (NE Spain) spreads over *ca.* 320 km² and is one of the most important estuarine systems in Europe (Curcó, 2006; Day *et al.*, 2006). Due to its high biological and ecological interest, one quarter of its surface (77, 36 km²) has been protected as a Natural Park, including coastal lagoons, bays and the adjacent coastline

(Day *et al.*, 2006). Since the 19th century, anthropogenic impacts have greatly altered the dynamics of the area (Curcó, 2006). On one hand, rice agriculture and the water management associated to this practice have altered fish communities inhabiting the coastal lagoons (Rodríguez-Climent *et al.*, 2013). On the other hand, artisanal fisheries carried out in the area have also been reported to affect the fish community of the coastal lagoons (Rodríguez-Climent *et al.*, 2012; Sostoa, 1983). Moreover, important changes in landings have been registered in the last decades (Fig. 1; data from St. Pere Fishermen association), that show a dramatic decrease of the landings from the mid 1990s to recent times. This situation alerts for the urgent necessity of a different management approach of the local fisheries within the lagoon. The present study aimed to: (i) construct a trophic-web model for the Tancada lagoon using Ecopath, (ii) to evaluate the impacts of artisanal fisheries on the ecosystem and, (iii) to simulate future fishing effort scenarios with the Ecosim application in order to assist on the design of a sustainable, long-term exploitation of fisheries resources in the Tancada lagoon.

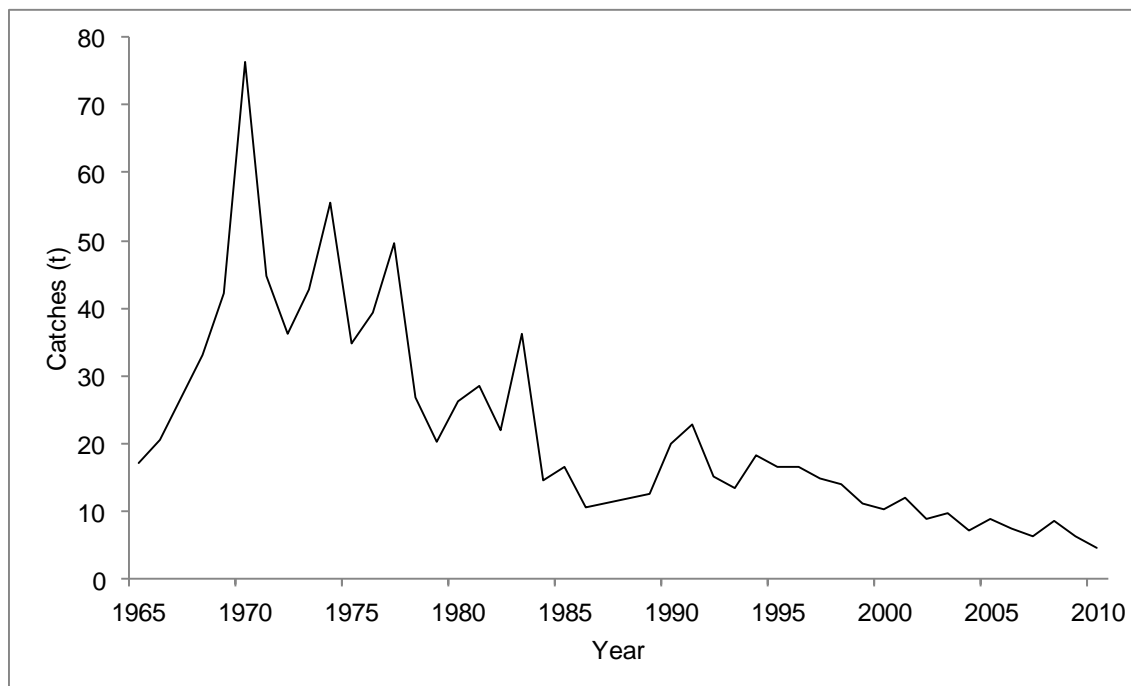


Figure 1. Total catches per year in the Tancada lagoon from the period 1965-2010. (Data from St.Pere Fishermen association).

2. Materials and methods

2.1 Study area

An Ecopath with Ecosim (EwE) mass-balanced model was developed in the Tancada lagoon, situated in the Ebro Delta, Spain (Fig. 2). With a total surface of 1.85 km², mean depth of 0.58 m and mean temperature of 19.87 °C, Tancada lagoon is one of the largest coastal lagoons in the Ebro Delta system. It is connected to the Mediterranean Sea (Alfacs Bay) by several artificial water channels that allow both water exchange and fish-migrations between the lagoon and the sea (Fig. 2). The lagoon also receives, in its Northern flank, fresh water inputs from the rice fields during

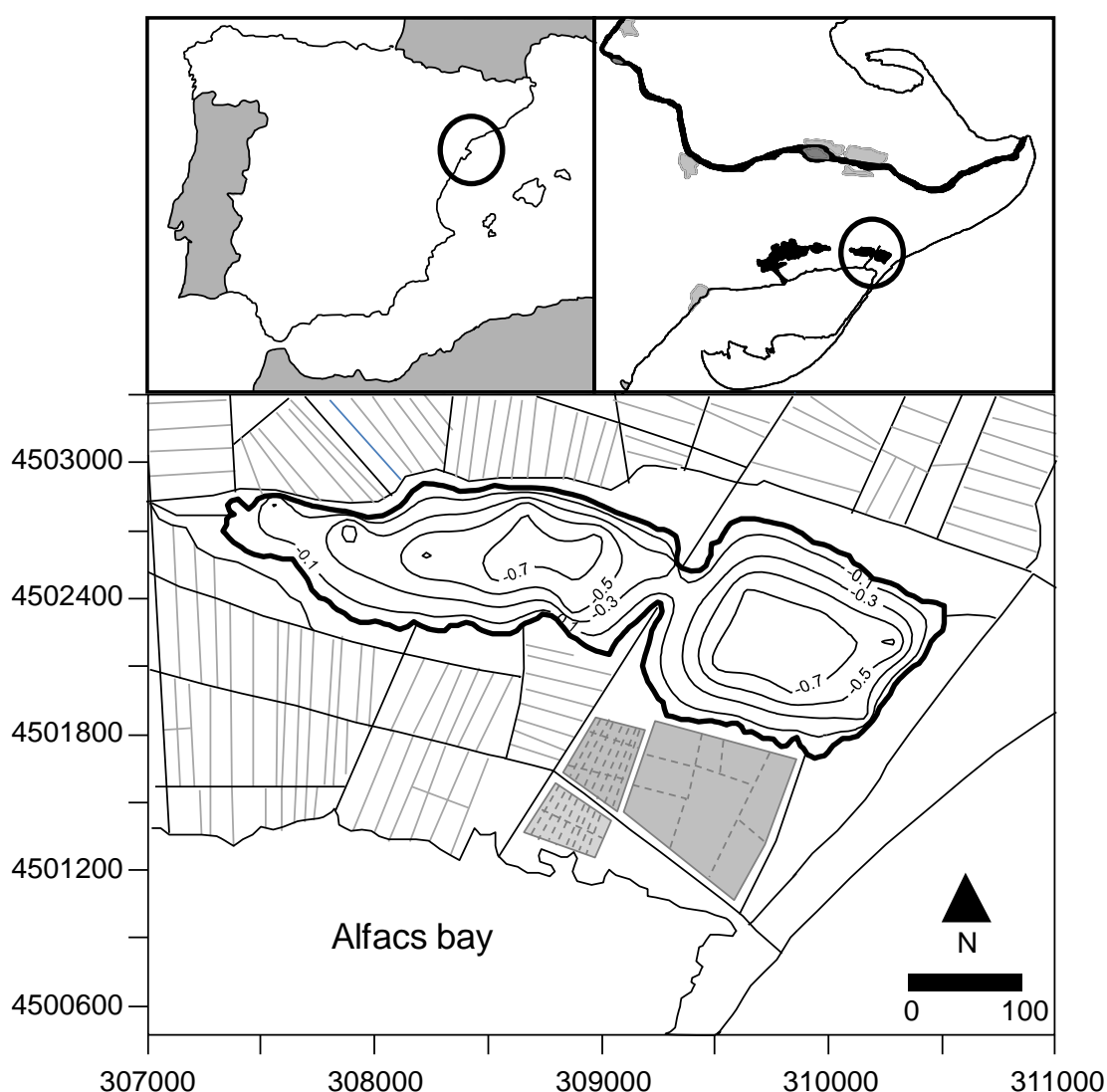


Figure 2. Map showing the location of the Ebro Delta and the Tancada coastal lagoon. Isobaths indicated the mean depth in the lagoon (m). Black lines are water channels. Grey lines indicate separation among rice fields. Grey shaded areas are St. Antoni salt works.

the rice cultivation period (April-September) that result on great salinity fluctuations (from 8 to 36) and at the same time allows the colonization of fish from marine, brackish and freshwater environments (Rodríguez-Climent *et al.*, 2013). The high diversity of fish fauna and the migratory movements of some marine species in the area, are used by the fishermen who set fishing nets all around the lagoon and in the canals that connect the lagoon with the sea through the Alfacs Bay (Fig. 2). Seven functional groups out of the total found in the lagoon are of commercial interest (target species): sea bass (*Dicentrarchus labrax*), gilthead seabream (*Sparus aurata*), sand smelt (*Atherina boyeri*), glass and adult eels (*Anguilla anguilla*), Mullet species (*Liza* sp., *Mugil cephalus* and *Chelon labrosus*) and carps (*Cyprinus carpio*).

2.2 Modelling approach

2.2.1 Ecopath

The EwE software version 6 (Christensen *et al.*, 2008; www.ecopath.org) was used to develop the Tancada mass-balanced model. EwE software is derived from the work initially developed by Polovina (1984) who aimed to estimate the biomass and consumption of various elements of an aquatic ecosystem using a network theory to analyse flows among the elements of the ecosystem (Ulanowicz, 1986). The software was later modified and expanded by Christensen and Pauly (1992). The parameterization of an Ecopath model is based on solving two main equations. The first equation splits the production (P_i) of each functional group i of the ecosystem, into the sum of the total fishery catch (Y_i), the total predation ($M2_i$), the net migration rate (E_i) (expressed as emigration – immigration), the Biomass accumulation rate (BA_i) and ‘other mortality’; in which EE_i is the Ecotrophic efficiency defined as the proportion of the production utilized in the system. This first master equation is formulated as:

$$P_i = Y_i + B_i \cdot M2_i + E_i + BA_i + P_i (1 - EE_i) \quad (1)$$

Nevertheless, Eq. (1) is usually expressed as:

$$B_i \cdot \left(\frac{P_i}{B_i}\right) \cdot EE_i - \sum_{j=1}^n B_j \cdot \left(\frac{Q}{B}\right)_j \cdot DC_{ji} - EX_i = 0 \quad (2)$$

where B_i is the biomass of group i ; P_i/B_i is the production/biomass ratio (or Z under steady-state conditions (Allen, 1971)); EE_i is the ecotrophic efficiency; B_j is the biomass of predator j ; Q_j/B_j is the consumption/biomass ratio of predator j ; DC_{ji} is fraction of prey i in the diet of predator j and EX_i is the export of group i . Based on the principle of matter

conservation, the energy balance within each group is ensured using the second master equation:

$$C_i = P_i + R_i + \left(\frac{UN}{Q}\right)_i \quad (3)$$

Where consumption (C_i) equals the production (P_i), respiration (R_i) and unassimilated food (UN/Q) $_i$ for each functional group i . Thus, for a system with n functional groups, n linear equations are calculated by Ecopath. In general, only one of the four basic parameters: B_i , $(P/B)_i$, $(Q/B)_i$ or EE_i may be unknown, whereas DC_{ij} , Y_i and E_i are always required. When the system equations are solved, they provide a snapshot of the trophic flow within the system (Christensen and Walters, 2004; Christensen *et al.*, 2008).

To ensure consistence between ontogenetic groups, the multiple stanza representation (Christensen and Walters, 2004) was used to model the European eel (*Anguilla anguilla*) and the Mugilidae family (*Liza* sp., *Mugil cephalus* and *Chelon labrosus*). For the European eel two groups were identified: glass eel (< 36 months; Gómez-Juaristi and Salvador, 2011) and adult eel (> 36 months); diet composition was provided for both groups (Arias and Drake, 1990; Fernández-Delgado *et al.*, 2000; Froese and Pauly, 2012); whereas B_i and $(Q/B)_i$ were only introduced for the leading stanza group: adult eel. For the Mugilidae family two groups were also differentiated: juvenile Mugilidae (< 24 months or length of maturity (Lm) < 22 cm; Kottelat and Freyhof, 2007) and adult Mugilidae (> 24 months); diet composition was provided for both groups (Brando *et al.*, 2004) and B_i and $(Q/B)_i$ entered for adult Mugilidae (leading group).

2.2.2 Ecosim

The Ecosim module was used to analyse the temporal dynamics of the Tancada lagoon. The model was calibrated for the period 1965-2010 with catch data time series (data provided by the St. Pere Fishermen association). Ecosim is a time-dynamic simulation model that takes the set of linear equations used to construct Ecopath models and sets up a series of differential equations. Its master equation is:

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M0_i + F_i + e_i)B_i \quad (4)$$

where dB_i/dt represents the growth rate during the time interval dt of group i in terms of its biomass, B_i , g_i is the net growth efficiency (production/consumption ratio), $M0_i$ the non-predation ('other') natural mortality rate, F_i is the fishing mortality rate, e_i is emigration rate, I_i is immigration rate. The consumption rates Q_{ji} are calculated based on the 'foraging arena' concept, where B_i are divided into vulnerable and invulnerable

components (Walters *et al.*, 1997), and it is the transfer rate (v_{ij}) between these two components that determines if control is top-down, bottom-up or an intermediate type (Christensen *et al.*, 2008).

2.3 Ecopath model parameters and functional groups

The Ecopath model was built with an average of data from three years (2008-2010). Biomass values for the functional groups of: cormorants, sand smelt, adult eel, zooplankton, phytoplankton and macrophytes were calculated with our own data (see appendix A). P/B and Q/B ratios were taken from the literature or obtained from the application of empirical equations (Palomares and Pauly, 1998) using length frequency data, when available. A bottom-up control regime was set up with low vulnerabilities ($v=1.1$; Christensen *et al.*, 2008). The diet composition for all functional groups was compiled from published studies, as no data from the stomach contents was available. Other necessary parameters such as K and L at maturity were obtained from Fish Base (Froese and Pauly, 2012).

Literature data sources used were prioritized as follows: i) Mediterranean coastal lagoons (Brando *et al.*, 2004; Carrer and Optiz, 1999; Libralato and Solidoro, 2009; Libralato *et al.*, 2002; Palomares *et al.*, 1993; Pinnegar and Polunin, 2004; Pranovi *et al.*, 2003); ii) Western Mediterranean (Gucu, 2002; Orek, 2000; Tsagarakis *et al.*, 2010); iii) outside the Mediterranean (Blume, 2003; Yunkai-Li *et al.*, 2009). When necessary, values were corrected to consider differences in temperature between data from the adopted value and our study site using the Optiz correction formula (Opitz, 1996). Input data is listed in Table 1 and data sources and estimation methods are compiled in appendix A. The model included 18 functional groups comprising target and non-target fish species and the main trophic items of the ecosystem.

2.4 Ecosim fit and model predictions

Official landings statistics from the period 1965-2010 concerning the Tancada lagoon artisanal fisheries were provided by the St. Pere Fishermen association and used to calibrate the catch data with Ecosim. During the process of catch data calibration (predicted catch versus observed catch), the fishing effort of glass eel was multiplied per three because the original fishing mortality was too low. This fact could be explained because about 1/3 of the captured glass eel were not reported in the landings statistics, since it is sold before going to the fish market (local fishermen comment). Moreover, during the adult eel calibration process, its catch was divided by half in order to obtain a

better fit. This could be explained by the fact that this species migrates every year, and thus the total of specimens captured (registered in the landings statistics from which Ecosim was fitted) is not the “real” resident population in the lagoon, but just a part of it which migrates.

Two fishing effort scenarios (50% increase and 50% decrease of the present fishing effort) were simulated with Ecosim with no loaded time-series for a 10 years period in order to see the response of the ecosystem to a possible situation of a gradual increase or decrease of the fishing effort. 50% was chosen, because it is plausible to happen in the future. The suitability of both fisheries’ management scheme in order to ensure the sustainability of this activity, and its compatibility with the ecosystem conservation is discussed.

3. Results

3.1 Ecopath results

A balanced Ecopath model was constructed for the Tancada lagoon. The model consisted of 18 compartments and a total biomass (excluding detritus) of 542.25 t/km² (Tables 1 and 2). The Biomass of the fish groups accounted for 21.77 t/km². The P/Q (P/B ÷ Q/B) values were in the expected range of 0.05-0.3 (Christensen *et al.*, 2008) and the Ecotrophic efficiencies for all groups were below 1. We found the highest values of EE in sand goby (0.950), sand smelt (0.944) and zoobenthos (0.900); whereas the lowest values were observed for macrophytes (0.023), glass eel (0.219) and phytoplankton (0.233) (Table 2). Diet composition for the 18 groups in the Tancada model is shown in Table 3. The total fisheries catch accounted for 3.56 t/km² per year and operated a mean trophic level of 2.856, with adult mugilidae registering the highest catches per year (Table 4).

The diagram of the trophic flow across the aggregated trophic levels in the Tancada lagoon is shown in Figure 3. Functional groups were illustrated by their trophic levels (TL) ranging from 1 to 3.473, with the highest values for cormorants, sea bass, gilthead seabream, sand goby, adult eel and mosquitofish with trophic levels above 3. All other fish groups had TL’s between 2.203 (adult mugilidae) and 2.885 (Glass eel). The lowest trophic groups by definition were primary producers and detritus (TL=1). Each functional group was represented by a circle proportional to its biomass (Figure 3).

Direct and indirect interactions within the ecosystem were analysed by means of the mixed trophic impact routine (MTI) (Figure 4). Direct negative effects were found in

Table 1. Species identified in the Tancada lagoon system and their aggregation into the trophic groups.

	Taxa
Cormorants	<i>Phalacrocorax carbo</i>
Sea bass	<i>Dicentrarchus labrax</i>
Sand goby	<i>Pomatoschistus microps</i>
Gilthead seabream	<i>Sparus aurata</i>
Sand smelt	<i>Atherina boyeri</i>
Glass eel	<i>Anguilla anguilla</i> (juveniles)
Adult eel	<i>Anguilla anguilla</i>
Mosquitofish	<i>Gambusia holbrooki</i>
Spanish toothcarp	<i>Aphanius iberus</i>
Juvenile mugilidae	<i>Liza aurata</i> (juveniles), <i>Liza ramada</i> (juveniles), <i>Liza saliens</i> (juveniles), <i>Mugil cephalus</i> (juveniles), <i>Chelon labrosus</i> (juveniles)
Adult mugilidae	<i>Liza aurata</i> , <i>Liza ramada</i> , <i>Liza saliens</i> , <i>Mugil cephalus</i> , <i>Chelon labrosus</i>
Carps	<i>Cyprinus caprio</i>
Zoobenthos	'Suspension feeders', 'deposit feeders'
Zooplankton	-
Phytoplankton	-
Benthic algae	-
Macrophytes	<i>Ruppia cirrhosa</i> , <i>Najas marina</i> , <i>Pomatogeton pectinatus</i> , <i>Chaetomorpha capillaries</i>
Detritus	-

cases of predation (*i.e.* predators over their preys, for example cormorants over carps or sea bass over Spanish toothcarp (see Table 3 for specific diet compositions)) and complementarily positive direct effects were found between preys and their predators (*e.g.* phytoplankton over zooplankton). Artisanal fisheries showed a direct negative impact on cormorants and all target species (except in glass eel), and an indirect positive impact on the non-target species such as sand goby, mosquitofish, Spanish toothcarp and juvenile mugilidae (Figure 4). Trophic chain indirect effects were also found in cases of competition for resources as the case of adult eel and sea bass (Figure 4).

Table 2. Ecopath basic input estimates. *Italic values were computed by the model.* B: Biomass ($t\ km^{-2}$); P/B: Production /Biomass ($year^{-1}$); Q/B: Consumption / Biomass ($year^{-1}$); EE: Ecotrophic efficiency; Un/Q: Unassimilated consumption.

Functional group	B	P/B	Q/B	EE	Un/Q
1 Cormorants	0.021	0.300	58.400	<i>0.000</i>	0.200
2 Sea bass	<i>1.056</i>	0.750	5.623	0.700	0.200
3 Sand goby	<i>0.499</i>	2.460	18.560	0.950	0.200
4 Gilthead seabream	<i>2.149</i>	0.700	4.404	0.800	0.200
5 Sand smelt	0.120	2.100	13.769	<i>0.944</i>	0.200
6 Glass eel	<i>2.383</i>	1.170	8.368	<i>0.219</i>	0.200
7 Adult eel	2.686	0.700	4.000	0.840	0.200
8 Mosquitofish	<i>0.709</i>	2.040	24.850	0.800	0.200
9 Spanish toothcarp	<i>0.189</i>	3.920	23.568	0.800	0.200
10 Juvenile mugilidae	<i>2.613</i>	1.700	19.044	<i>0.623</i>	0.200
11 Adult mugilidae	9.200	0.700	7.200	<i>0.236</i>	0.200
12 Carps	<i>0.142</i>	0.500	6.540	0.840	0.200
13 Zoobenthos	<i>28.071</i>	5.000	20.000	0.900	0.200
14 Zooplankton	2.250	23.470	80.000	<i>0.664</i>	0.200
15 Phytoplankton	26.400	40.000		<i>0.233</i>	
16 Benthic algae	<i>22.463</i>	15.000		0.500	
17 Macrophytes	441.300	5.000		<i>0.023</i>	
18 Detritus	2.000			<i>0.090</i>	

3.2. Ecosim results

Time series fitting between predicted and observed catch for a period of 45 years are shown in Figure 5. From all the fished species in the lagoon, the model was able to predict the catch for glass eel, sand smelt, adult mugilidae and gilthead seabream but failed to reproduce carp's trend. Moreover, the model did not reproduce the trend for the first decades of the time series data for the two piscivorous species in the lagoon: adult eel and sea bass, although it seemed to reproduce quite well the catch for the last decades (1987 onwards for adult eel and 1998 onwards for sea bass) (Figure 5).

The predicted effect of simulated changes in the fishing effort for 10 years is shown in Figure 6. There was no evidence that any new equilibrium was reached for any group in those 10 years for either of the two situations (*i.e.* increase or decrease of the fishing effort). A gradual increase of fishing effort of 50% (Figure 6a) caused decreases in the biomass of most of the target species with gilthead seabream (18.63% of biomass

Table 3. Diet composition matrix for the functional groups in the model

Predator														
Prey	1	2	3	4	5	6	7	8	9	10	11	12	13	14
1 Cormorants														
2 Sea bass	0.020													
3 Sand goby	0.010	0.100					0.052							
4 Gilthead seabream	0.174													
5 Sand smelt	0.050						0.010							
6 Glass eel							0.052							
7 Adult eel	0.012													
8 Mosquitofish		0.100					0.052	0.001						
9 Spanish toothcarp		0.100												
10 Juvenile mugilidae	0.346	0.300					0.052							
11 Adult mugilidae	0.346													
12 Carps	0.034													
13 Zoobenthos		0.400	0.530	1.000	0.420	0.600	0.573	0.499	0.750	0.210	0.180	0.100	0.100	
14 Zooplankton			0.470		0.300	0.200		0.450		0.050		0.200	0.015	0.040
15 Phytoplankton													0.150	0.900
16 Benthic algae												0.050	0.300	
17 Macrophytes					0.050	0.100	0.104	0.050	0.002	0.410	0.410			
18 Detritus					0.230	0.100	0.104		0.248	0.330	0.410	0.650	0.435	0.060
19 Import	0.008													
20 Total	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 4. Total landings of the target species in the Tancada lagoon (average of the years 2008-2010).

Functional group	Artisanal fisheries (t km ⁻² year ⁻¹)
Sea bass	0.530
Gilthead seabream	0.990
Sand smelt	0.065
Glass eel	0.051
Adult eel	0.810
Adult mugilidae	1.096
Carps	0.018
Total catch	3.560
Trophic level	2.856

decrease), sea bass (17.43%) and adult eel (7.74%) being the fish species showing the higher loss of biomass. The rest of the target species showed a decrease of their biomass below 8%; and glass eel was the only target species which experienced an increase of its biomass in 1.10% (Figure 6a). From the remaining groups, cormorants

were the most affected, showing a decrease of 12.08% of its initial biomass. The decline in the biomass of target species led to a substantial increase in the biomass of the non-target fish species; however the percentage of biomass increment was below 3% in all cases: Spanish toothcarp (2.76%), sand goby (2.35%), mosquitofish (2.02%) and juvenile mugilidae (1.04%).

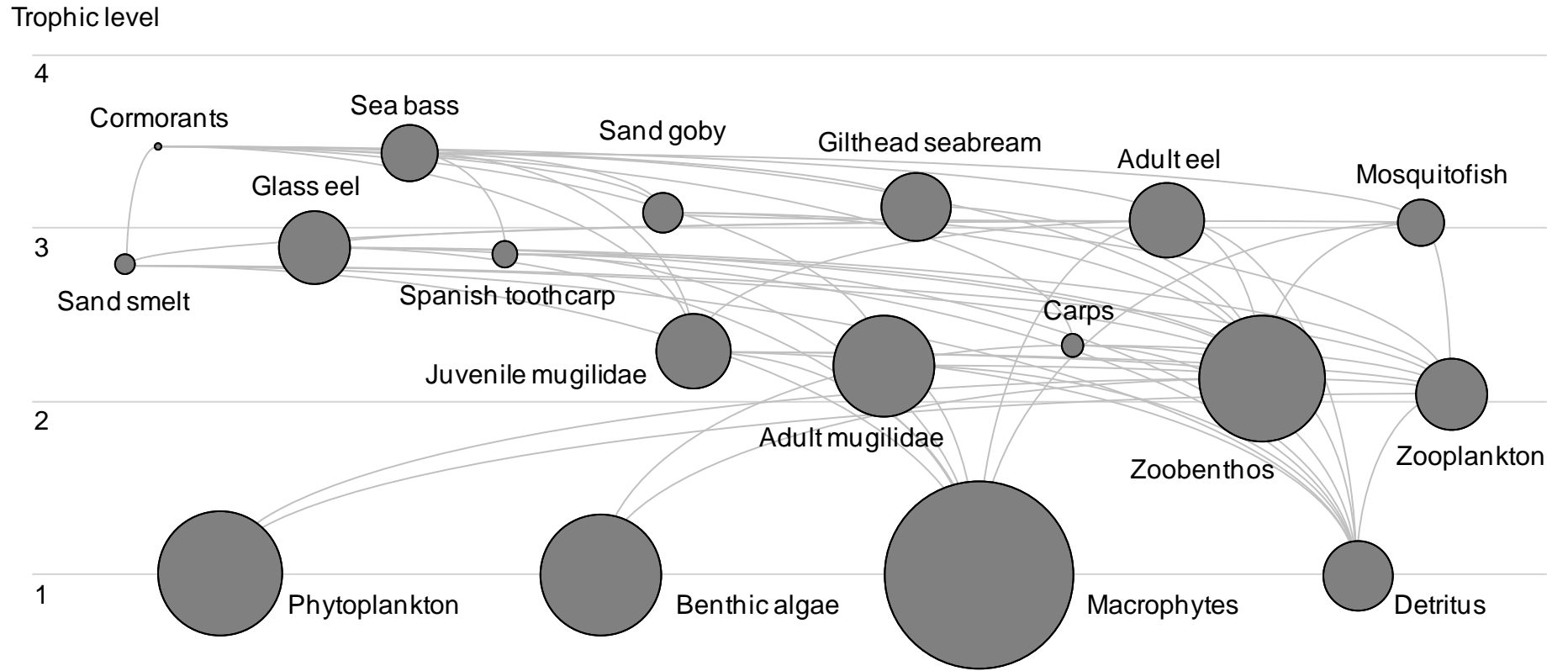
Figure 6b shows the result of the second simulation scenario with a gradual decrease of the fishing effort by 50% in the same period (10 years). Direct and indirect responses to the reduction in the fishing effort were opposite to those observed when fishing pressure was increased, but different in dimension. The non-target fish species showed biomass decreases, being Spanish toothcarp the species that reported the highest decrease in abundance (1.51%), followed by sand goby (1.43%), mosquitofish (1.24%) and juvenile mugilidae (0.68%). All target fish species reported increases of their biomass (gilthead seabream 13.12%, sea bass 11.94%, adult eel 6.96%), except glass eels that showed a 0.81% decrease of its biomass. Moreover, the group of cormorants showed an increase of 18.85% of its initial biomass (Figure 6b).

4. Discussion

4.1 Ecopath model

Ecopath with Ecosim was used to model the Tancada lagoon in the Ebro Delta to better understand the structure and trophic interactions of the main functional groups in the area and to assess the impact of the artisanal fisheries. Although EwE has been widely used to analyse the impact of the fisheries worldwide (Albouy *et al.*, 2010; Coll *et al.*, 2006a; Coll *et al.*, 2007; Gucu, 2002; Pinnegar and Polunin, 2004; Piroddi *et al.*, 2010

Figure 3. Flow diagram of the Tancada lagoon ecosystem. Each functional group is shown as a circle and its size is proportional to its biomass. The trophic links between functional groups are also presented.



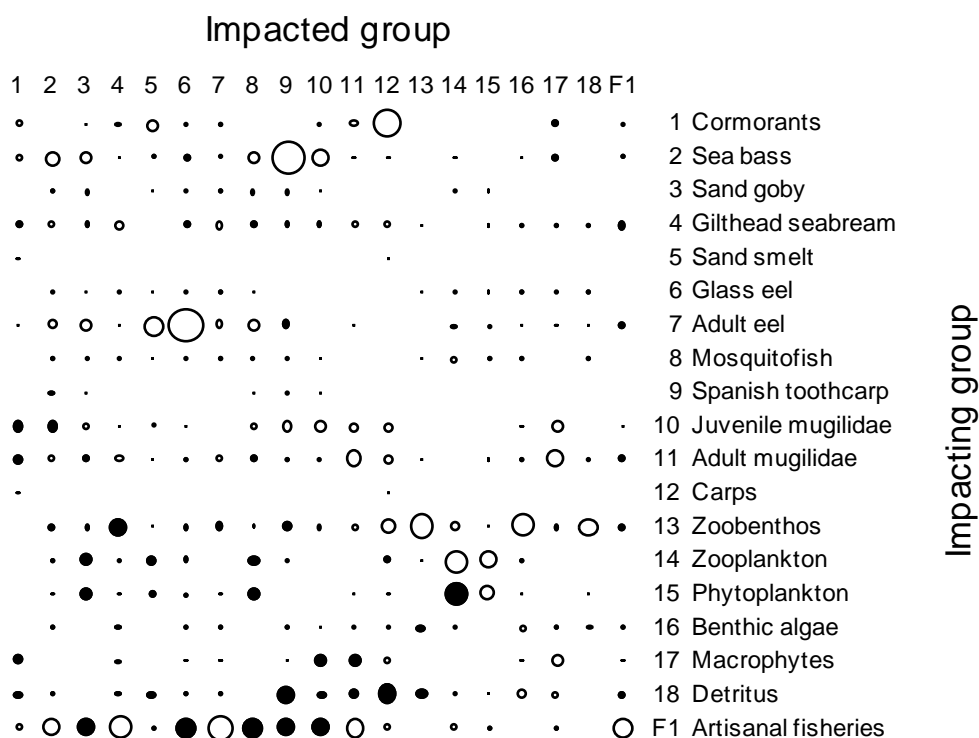


Figure 4. Mixed trophic impact analysis. Circle size is proportional to the exerted impact. Black circles indicate positive impact and white circles indicate negative impact. Impact by artisanal fisheries (F1) is also shown.

Sánchez and Olaso, 2004), in Mediterranean coastal lagoons it has been mostly employed to describe the ecosystem functioning and structure, and its applications to implement better management strategies (Brando *et al.*, 2004; Libralato *et al.*, 2002; Palomares *et al.*, 1993). Moreover, only some studies considered the role of fisheries, but not with the objective of a better fisheries management proposal (Carrer and Opitz, 1999; Pranovi *et al.*, 2003).

One of the indicators of the community structure of a EwE model is the Ecotrophic efficiency (EE). EE measures the degree in which the production of one group is exported out to the system or consumed by predators and is expected to be close to 1 for groups with considerable predation pressure (Albouy *et al.*, 2010; Christensen *et al.*, 2008). Our reported high EE values for the zoobenthos, sand smelt and sand goby might indicate that these groups are suffering from a big predation pressure in the ecosystem; whereas other groups such as macrophytes, phytoplankton and glass eel are not very consumed in the Tancada lagoon. Specifically, zoobenthos stood out to be a very important group in the lagoon, as it is sustaining diets of numerous groups. This was in

accordance with some previous models developed in the Mediterranean area (Brando *et al.*, 2004; Carrer and Opitz, 1999; Pinnegar and Polunin, 2004). Sand goby ended to be the group with the highest EE value, which could be explained by the fact that all top-predators in the lagoon (cormorants, adult eel and sea bass) fed on this group.

Feeding interactions among species is one of the straightforward ways to describe ecosystems structure and dynamics (Pauly *et al.*, 1998). In EwE models, the trophic level (TL) of a group, defined as the default values of 1+ the weighted average TL of their preys, provide this information (Christensen, 1998; Christensen *et al.*, 2008; Coll *et al.*, 2009). TL's reported in this study were in consensus with TL ranges found in previous models developed in the Mediterranean area. TL for top-predators such as cormorants, was close to the average trophic level found for seabirds in studies focusing on seabird feeding ecology (Hobson *et al.*, 1994) and other studies in the Mediterranean area (TL = 3.5-4; Brando *et al.*, 2004; Coll *et al.*, 2006; Piroddi *et al.*, 2010). While, TL recorded for the other top predators in the lagoon (piscivores): adult eel and sea bass were lower than the ones registered in other Mediterranean studies (Brando *et al.*, 2004; Pinnegar and Polunin, 2004); however they were close to the TL found for the group of pelagic and benthic feeders fish in the Atlantic sea (Gamito and Erzini 2005). In the case of the sea bass, this fact could be explained by the large amount of zoobenthos that this species consumed in the lagoon. The TL values found for the adult mugilidae group were in accordance with other studies developed both in the Mediterranean and the Atlantic sea (Brando *et al.*, 2004; Gamito and Erzini, 2005; Pinnegar and Polunin, 2004). Furthermore, the pattern in the mugilidae multi-stanza group, with higher TL for juveniles than for adults, was also found by Brando (2004). Those differences in TL might indicate that the ecological strategy followed by each stanza group was different, hence reducing their niche overlapping (Gisbert *et al.*, 1995). The TL for the zoobenthos was similar to values found in the Orbetello lagoon and the Bay of Calvi (Brando *et al.*, 2004; Pinnegar and Polunin, 2004) and TL for the zooplankton was inside the observed rank (2-2.5) in other Atlantic (Gamito and Erzini, 2005) and Mediterranean studies (Albouy *et al.*, 2010; Brando *et al.*, 2004; Pinnegar and Polunin, 2004; Piroddi *et al.*, 2010).

Calculated as the weighted average of the TL of fisheries target species (Pauly *et al.*, 1998; Piroddi *et al.*, 2010), the mean trophic level of the catches (TLc) is a great indicator of the impact that fisheries exert on a system ('Fishing down marine food web', Pauly *et al.*, 1998). The low TLc found in this study when compared with other models developed

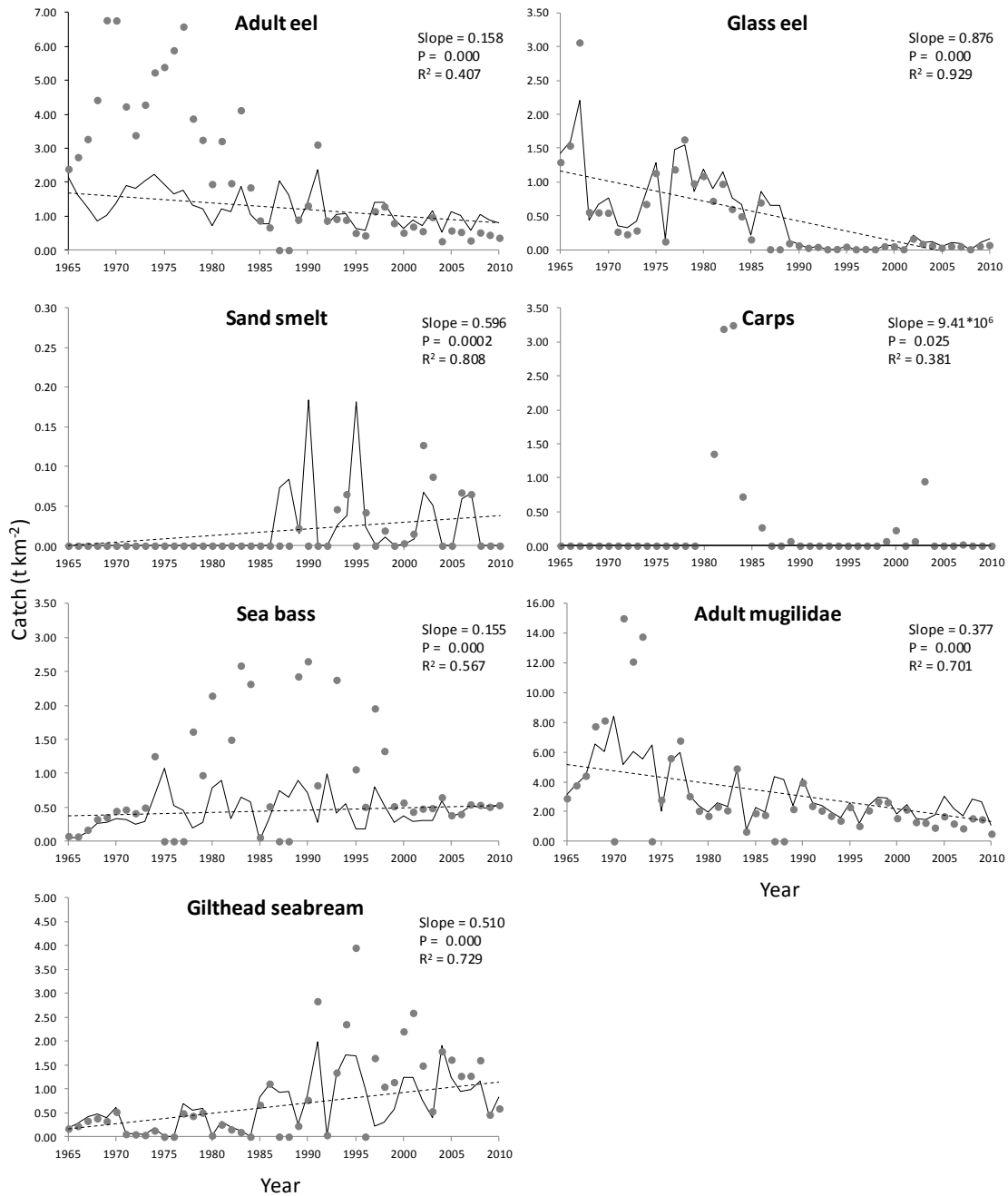


Figure 5. Time series fitting between predicted (line) and observed catch in t/km² for the period 1965-2010. The estimated trends (dashed line) are shown with the value of the slope, p-value and coefficient of variation (R²) for the regression model.

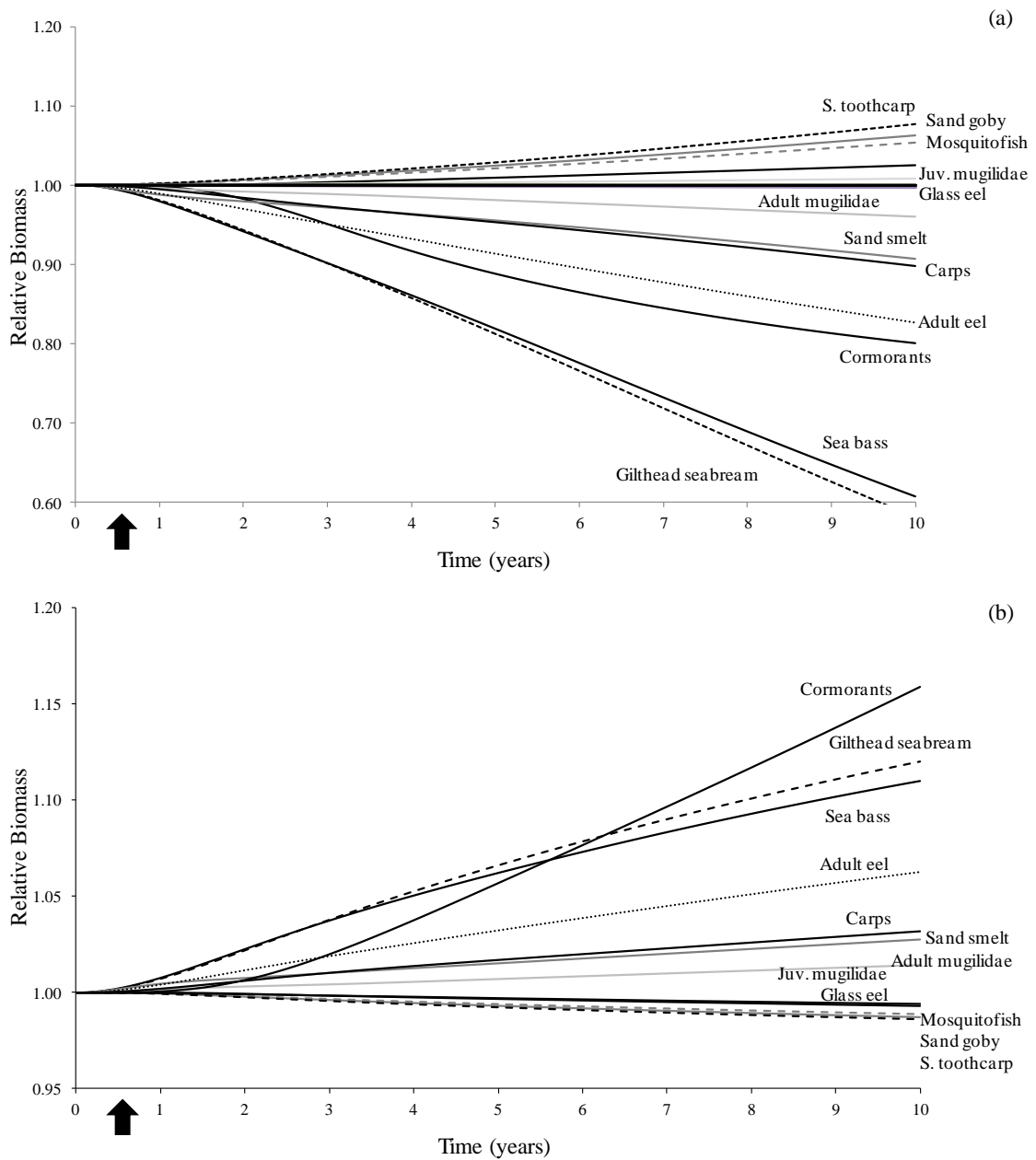


Figure 6. Effect of simulated changes in the fishing effort for 10 years, only fish groups are shown. (a) increase of 50% of the fishing pressure. (b) decrease of 50% of the fishing pressure. Arrow indicates steady-state fishing pressure.

in the North and South Catalan Sea (TLc 3.04-3.12; Coll *et al.*, 2006), Ionian Sea (TLc 3.1; Piroddi *et al.*, 2010) or some Mediterranean lagoons as the Orbetello lagoon in Italy (TLc 4.006-4.1; Brando *et al.*, 2004), could be explained by the dominance of the adult mugilidae (TL=2.203) in the catches, and might indicate a more severe exploitation of the fisheries in the Tancada lagoon, *i.e.*, the higher the fisheries exploitation, the lower the TL of the caught fish (Christensen *et al.*, 1996; Pauly *et al.*, 1998). The effect that an increase or decrease in fisheries, or in any other group in the model may have on the other ecological groups of the system, can be evaluated with the mixed trophic impact routine (MTI) (Carrer and Opitz, 1999). In this study, the positive impact of artisanal fisheries on the target species glass eel, might be explained by two reasons: firstly by the sub-estimated landings for this species in the lagoon (approximately 1/3 of the captured glass eel in the area is declared). This fact is not exclusive from the Ebro Delta region, as some studies from the Northern Ionian Sea have reported the unreliability of the landings facilitated by the fishers, who misreport their catches to avoid stricter regulations or higher taxation (Bearzi, 2006; Briand, 2000; Stergiou *et al.*, 1997; Stergiou *et al.*, 1998). Secondly, artisanal fisheries are removing the only natural predator that glass eel has in the Tancada lagoon: adult eel (Arias and Drake, 1985). Then, the unreported catches from the fishermen added to the fact that fisheries subtracted the predation of their main predator, could explain why glass eel were favoured by artisanal fisheries in the Tancada lagoon model.

4.2 Time-dynamic simulations

The model was able to predict catch for species with long time-series data available, whereas it failed to reproduce the trend for species with less complete time-series data (*e.g.* carps). This fact was in line with some previous studies who noticed that the reliability of simulations is determined by the quantity and quality of the data (Coll *et al.*, 2009; Walters *et al.*, 1997). However, the inability of the model to predict carp catches might indicate that dynamics of the species in the area might be affected by other factors apart from artisanal fisheries. In the Ebro delta, rice cultivation agriculture forced to construct a dense chain of channels in order to bring freshwater to the paddies (De Sostoa and De Sostoa, 1985). At present, in non-regular periods some freshwater inputs into the lagoons are promoted in order to favour game birds (Rodríguez-Climent *et al.*, 2013). Then, peaks of high catches for carps might be related to peaks of freshwater inputs in the lagoon, as salinity has been documented to be one of the main factors affecting the distribution of small-bodied fish in the lagoon (Rodríguez-Climent *et al.*, 2013). In the present study, environmental factors were not considered due to the lack of

time-series data. So, a precautionary approach must be taken when interpreting results of the Ecosim simulations. The worst fit of the model found for adult eel might be more related to its biology than to the lack of data, since it is a catadromus species (freshwater resident who migrates to spawn at the sea) (Kottelat and Freyhoff, 2007). Therefore, fishery effect was not enough to explain coastal lagoons stock fluctuations for this species, which also depends on fish migratory movements. However, the model was able to predict the trend for glass eel, thus reinforcing the role developed for coastal lagoons in general and Tancada lagoon in particular as nurseries' for the first life stages of fish (De Sostoa and De Sostoa., 1985; Pérez-Ruzafa *et al.*, 2004; Verdiell Cubedo, 2009). Moreover the stock decline in glass eel found in this study since the eighties, was in accordance with the decline for glass eel recruitment found worldwide (FAO and ICES, 2006). In gilthead seabream and sea bass groups the fit of the model was better than the one found for adult eel. However, the non-adjustment found for some years could be explained by the dependence that those two species have on sea dynamics as they are sea spawners (Kottelat and Freyhof, 2007). Results obtained for adult mugilidae might suggest a possible land-locked population in the area, as already documented for some species belonging to this family in Portugal (Fratel reservoir, Kottelat and Freyhof, 2007). General increasing trend of higher captures in recent times found for sand smelt was in agreement with previous studies in the area (Rodríguez-Climent *et al.*, 2012).

The recover capacity after pulses of increased fishing pressure was reported to be dependent on the model configuration and on the vulnerability value assumed (Pinnegar and Polunin, 2004). Our simulations have demonstrated that the Tancada lagoon ecosystem was not able to assume a new equilibrium in either a situation of increase or decrease of the fishing pressure before year 10. In those 10 years, the potential trophic cascade effects were also evaluated. Responses of target species to both an increase or decrease of the fishing effort were predicted to occur very shortly after the initiation of a new fishing regime, which is in agreement with other Mediterranean studies (Pinnegar and Polunin, 2004). On one hand, a gradual increase of actual fishing effort, showed a biomass decline for all target species in the lagoon except for the glass eel. The indirect effect of the artisanal fisheries, feeding on glass eel predators could explain this fact (Pinnegar 2000; Pinnegar and Polunin, 2004). Moreover, an increase of the actual fishing effort ended in a total loss of the ecosystem biomass; mostly due to the decrement of the target species fish biomass. On the other hand, our results highlighted that a gradual decrease of the actual fishing effort would provoke: a minimum biomass loss in some species (mostly preys of the target species), an increase of the biomass of the target species, an increase of the biomass of cormorants (due to a decrease of their

main competitor, as seen in other Mediterranean studies (Pinnegar and Polunin, 2004)), and a general increase of the total biomass of the ecosystem. Then, our results support that a gradual decrease of the fishing effort in the Tancada lagoon would let to a recover of the main fish species in the lagoon. This is in accordance with Christensen (1996) who pointed out that in order to implement a management model plan; the changes to be made in an ecosystem should be gradual, avoiding drastic changes. Our simulations indicated a better condition for target species in the system in comparison with its present state, reinforcing the hypothesis that reduction of fishing effort would be a step further towards the achievement of a more sustainable fisheries in the lagoon.

The present model constitutes the first mass-balance model constructed to characterise the Tancada lagoon and the impact of its artisanal fisheries. It represents an important effort to integrate all the available biological data from the area in a coherent format. A long time-series data for the artisanal fisheries (45 years) was collected. However, some deficiencies in available biological data have been identified such as a lack of biomass data for some fish species in the lagoon and a lack of a time-series of environmental data (salinity, temperature, pH...). The correction of the catches of some species due to the non-reported landings (e.g. glass eel) alerted us that the impact that artisanal fisheries may have in this species might be higher than the initially suspected, and must be taken into consideration for future management actions. This is also of relevance if we consider that eel fishing has a high economic importance in some countries as in France, where eel fishing accounts for more than half of the income of the professional lagoon fisheries (Crivelli *et al.*, 2008).

The results obtained in this study have clearly highlighted that artisanal fisheries developed in the Tancada lagoon have a negative impact into the cormorants and all the target species, except glass eels. With a gradual decrement of the current fishing effort in the lagoon, a more sustainable exploitation of the fisheries resources in the Tancada lagoon would be done, allowing to a long-term use of the natural resources. Further efforts to better characterise the weak parts of the model could turn out the simulations predicted more realistic and could be an important step forward towards a better characterization of the ecosystem and a better future management of the fisheries carried out in the area.

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Functional groups	Value	Equations	Source
1. Cormorant			
	Mean body weight (kg)	2.085	Albert Bertolero personal data
	Biomass (t/km ²)	0.021	Empirical calculation B = average number * average weight Auteri <i>et al.</i> , 1993
	P/B (year -1)	0.300	Moreau personal communication
	Q/B (year -1)	58.400	Brando <i>et al.</i> , 2004
	Trophic data	Data extracted from the literature	Olmos <i>et al.</i> , 2000
2. Sea bass			
	P/B (year -1)	0.750	Moreau personal communication
	Q/B (year -1)	5.623	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \text{log}Z - 0.152 * \text{log}W_{inf} - 1.360 * T' + 0.062 * A + 0.510 * h + 0.390 * d$ Christensen <i>et al.</i> , 2008
	EE	0.700	Moreau personal communication
	Trophic data	Data extracted from the literature	Costa <i>et al.</i> , 1988; Brando <i>et al.</i> , 2004
3. Sand goby			
	P/B (year -1)	2.460	Empirical calculation with Fostat; assuming Z= P/B (Beverton and Holt, 1957) Fisat II (vs 1.2.2) 2000-2005 FAO, Rome, Italy
	Q/B (year -1)	18.560	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \text{log}Z - 0.152 * \text{log}W_{inf} - 1.360 * T' + 0.062 * A + 0.510 * h + 0.390 * d$ Pauly, 1989; Christensen <i>et al.</i> , 2008
	EE	0.950	Pinnegar and Polunin, 2004
	Trophic data	Data extracted from the literature	Thorman, S. and Wiederholm, A.M, 1986
4. Gilthead seabream			
	P/B (year -1)	0.700	Moreau personal communication
	Q/B (year -1)	4.404	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \text{log}Z - 0.152 * \text{log}W_{inf} - 1.360 * T' + 0.062 * A + 0.510 * h + 0.390 * d$ Christensen <i>et al.</i> , 2008
	EE	0.800	Brando <i>et al.</i> , 2004
	Trophic data	Data extracted from the literature	Rosecchi, E., 1987; Pita, C. <i>et al.</i> , 2002
5. Sand smelt			
	Biomass (t/km ²)	0.120	Empirical calculation with own data with the formula B= C/F Jul-Larsen <i>et al.</i> , 2003
	P/B (year -1)	2.100	Empirical calculation with Fostat Fisat II (vs 1.2.2) 2000-2005 FAO, Rome, Italy
	Q/B (year -1)	13.769	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \text{log}Z - 0.152 * \text{log}W_{inf} - 1.360 * T' + 0.062 * A + 0.510 * h + 0.390 * d$ Christensen <i>et al.</i> , 2008
	Trophic data	Data extracted from the literature	Blanco, S., <i>et al.</i> 2003

6. Glass eels				
Biomass (t/km ²)	8.144	Estimated from Ecopath (multi-stanza group)		
P/B (year -1)	1.170	Empirical calculation with Fostat		Fisat II (vs 1.2.2) 2000-2005 FAO, Rome, Italy
Q/B (year -1)	6.026	Estimated from Ecopath (multi-stanza group)		
Trophic data		Data extracted from the literature		Arias & Drake, 1990 ; Fernández-Delgado <i>et al.</i> , 2000
7. Adult eel				
Biomass (t/km ²)	2.686	Empirical calculation with own data with the formula B= C/F		Jul-Larsen <i>et al.</i> , 2003
P/B (year -1)	0.700			Moreau personal communication
Q/B (year -1)	4.000	Data extracted from the literature		Palomares <i>et al.</i> , 1993; Brando <i>et al.</i> 2004
EE	0.840			Brando <i>et al.</i> , 2004
Trophic data		Data extracted from the literature		Arias & Drake, 1990 ; Fernández-Delgado <i>et al.</i> , 2000
8. Mosquitofish				
P/B (year -1)	2.040	Empirical calculation with Fostat		Fisat II (vs 1.2.2) 2000-2005 FAO, Rome, Italy
Q/B (year -1)	24.850	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \log Z - 0.152 * \log W_{inf} - 1.360 * T^{\circ} + 0.062 * A + 0.510 * h + 0.390 * d$		Christensen <i>et al.</i> , 2008
EE	0.800	Data extracted from the literature		Blume, 2003
Trophic data		Data extracted from the literature		García-Berthou, 1999
9. Spanish toothcarp				
P/B (year -1)	3.920	Empirical calculation with Fostat		Fisat II (vs 1.2.2) 2000-2005 FAO, Rome, Italy
Q/B (year -1)	23.568	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \log Z - 0.152 * \log W_{inf} - 1.360 * T^{\circ} + 0.062 * A + 0.510 * h + 0.390 * d$		Christensen <i>et al.</i> , 2008
EE	0.800			Brando <i>et al.</i> , 2004
Trophic data		Data extracted from the literature		Alcaraz and García-Berthou, 2007
10. Juv. mugilidae				
Biomass (t/km ²)	8.713	Estimated from Ecopath (multi-stanza group)		
P/B (year -1)	1.740			Moreau personal communication
Q/B (year -1)	19.965	Estimated from Ecopath (multi-stanza group)		
Trophic data		Data extracted from the literature		Brando <i>et al.</i> , 2004
11. Adult mugilidae				
Biomass (t/km ²)	9.200	Data extracted from the literature		Palomares <i>et al.</i> , 1993
P/B (year -1)	0.700			Moreau personal communication
Q/B (year -1)	7.200			Moreau personal communication

	Trophic data		Data extracted from the literature	Brando <i>et al.</i> , 2004
12. Carps				
	P/B (year -1)	0.500		Moreau personal communication
	Q/B (year -1)	6.540	Empirical calculation $\text{Log}(Q/B) = 5.847 + 0.28 * \log Z - 0.152 * \log W_{inf} - 1.360 * T' + 0.062 * A + 0.510 * h + 0.390 * d$	
	EE	0.840		Yunkai-Li <i>et al.</i> , 2009
	Trophic data		Data extracted from the literature	Blanco <i>et al.</i> , 2003
13. Zoobenthos				
	P/B (year -1)	5.000		Moreau personal communication
	Q/B (year -1)	20.000		Moreau personal communication
	EE	0.900	Data extracted from the literature	Brando <i>et al.</i> , 2004
	Trophic data		Data extracted from the literature	Brando <i>et al.</i> , 2004 and Moreau personal communication
14. Zooplankton				
	Biomass (t/km ²)	2.250	Estimation with own data	
	P/B (year -1)	23.470	Adapted the PB value with Optiz formula (1996) from a value from Palomares <i>et al.</i> 1993	Palomares <i>et al.</i> , 1993; Optiz, 1996
	Q/B (year -1)	80.000		Moreau personal communication
	Trophic data			Moreau personal communication
15. Phytoplankton				
	Biomass (t/km ²)	26.400	Estimation with own data	
	P/B (year -1)	40.000	Data extracted from the literature	Palomares <i>et al.</i> , 1993; Brando <i>et al.</i> , 2004
16. Benthic algae				
	P/B (year -1)	15.000		Moreau personal communication
	EE	0.500		Palomares <i>et al.</i> , 1993; Moreau personal communication
17. Macrophytes				
	Biomass (t/km ²)	441.300	Estimation with own data	
	P/B (year -1)	5.000		Moreau personal communication

Appendix A. Input data and references by functional group for the Tancada lagoon model.

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Salinity as the main factor structuring small-bodied fish assemblages in hydrologically altered Mediterranean coastal lagoons

Rodríguez-Climent S, Caiola N, Ibáñez C

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Salinity as the main factor structuring small-bodied fish assemblages in hydrologically altered Mediterranean coastal lagoons

Running title: Small-bodied fish in hydrological altered lagoons

Sílvia Rodríguez-Climent, Nuno Caiola* and Carles Ibáñez

IRTA Aquatic Ecosystems. E-43540 Sant Carles de la Ràpita, Catalonia, Spain

Abstract

In the Ebro Delta coastal lagoons, one of the main anthropogenic pressures is the artificial freshwater input. Each coastal lagoon has different water management schemes causing profound changes in its physicochemical characteristics. The main objective of this water management is to favor some bird species with interest either for conservation or hunting activities. The present study assesses the influence of hydrological alteration on the fish assemblages of three coastal lagoons in the Ebro Delta. The small-bodied fish fauna was mainly composed by five families: Gobiidae, Poeciliidae, Cyprinodontidae, Atherinidae and Mugilidae. Salinity was found to be the main factor structuring fish community in the lagoons. The dominant species was the common goby (*Pomatochistus microps*) when the lagoons reached higher salinity values, whereas the invasive Eastern mosquitofish (*Gambusia holbrooki*) dominated during the period of higher freshwater inputs. The juveniles of the family Mugilidae accounted with low Captures per Unit of Effort (CPUEs), especially during the period of lower salinity. This same pattern was found for the endangered Spanish toothcarp (*Aphanius iberus*). Overall, introduced species were favored by low salinity, which highlights the importance of changing the present water management by reducing the freshwater inputs in order to maintain adequate levels of salinity to favor native species, important both for commercial and conservation purposes.

Key words: fish community, salinity, water management, Ebro Delta, coastal lagoons.

* Corresponding author E-mail: nuno.caiola@irta.cat (Nuno Caiola).Tel: +34 902 789 499 Ext. 182

1. Introduction

Coastal lagoons are located at the land-sea interface and are generally characterized by high biological productivity and shallowness (Barnes 1980, Brehmer *et al.* 2011). They are used by many fish species either as feeding, nursery or spawning grounds (Pérez-Ruzafa *et al.* 2004, Oertli *et al.* 2005, Ribeiro *et al.* 2006, Verdiell-Cubedo 2009). Usually dominated by marine and estuarine species, coastal lagoon fish assemblages may present different composition and structures, depending principally on the balance between marine and freshwater inputs (Gordo and Cabral 2001). The Mediterranean coastal lagoons are highly modified ecosystems due to the impacts of human activities such as intensive agriculture, urban and industrial land uses, canal construction and impoundments, dredging and tourism. These activities cause serious changes in the natural ecological cycles, particularly water regime alterations caused by artificial freshwater inputs (Pérez-Ruzafa *et al.* 1991, Day *et al.* 2000, Caiola *et al.* 2001a, b, Badosa *et al.* 2007, Verdiell-Cubedo 2009). The most common consequences of these hydrological alterations are the decline in salinity and the increase in both eutrophication and environmental pollution as a result of increased nutrient and chemical contaminant inputs (Cloern 2001, Lucena *et al.* 2002, Badosa *et al.* 2007). Despite the severe impacts that these hydrological alterations can cause in the fish communities, the studies focusing on this issue in the Mediterranean area are scarce (Poizat *et al.* 2004, Badosa *et al.* 2007).

Situated in Catalonia (North-East of Spain), the Ebro Delta, with 320 km² of extension, is one of the most important estuarine systems in Europe. Due to the high ecological interest, 25% of its area, including the coastal lagoons, bays and adjacent coastline, is protected as a Natural Park since 1983. The Natural Park conservation status means that the protection of the natural areas must be compatible with the use of their resources and activities of their inhabitants (such as fisheries, hunting, agriculture, etc.). The rest of the area is used for agriculture, mainly rice cultivation (Day *et al.* 2006). To sustain this agricultural activity two main canals, regulated at their origin point (40 km upstream the river mouth), are split into numerous secondary canals and ditches to bring freshwater to all rice fields in the delta. Freshwater supply is only interrupted between January and April to prepare the fields for the next crop. In the past, waste water from the rice fields was drained to the lagoons, and from these systems to the sea through the outlets. This water management scheme caused eutrophication in the coastal lagoons (Forés *et al.* 2002). This problem was overcome in the nineties with the construction of circumvallation canals, avoiding drainage water to pass through the lagoons (Forés *et al.* 2002). At present the Ebro Delta coastal lagoons still receive freshwater artificially,

mainly for habitat management to favor game birds. The freshwater comes both from the river, through the irrigation canals, and from the rice fields drainages. The quality of the river water increased significantly in the last 15 years (Nebra *et al.* 2011, Ibáñez *et al.* 2012a,b) and the water input from the rice fields drainages is done only in the periods of low inputs of pesticides and fertilizers (Forés *et al.* 2002). As a result, water quality in the lagoons has improved, but its hydrology is still severely altered. The effects of water management on the coastal lagoons' fish assemblages remain unknown. The approach of sampling small-sized resident fish species and juveniles of migratory species was chosen because the capacity of movement of larger species and the adults of migratory species do not make them reliable as descriptors of spatial distribution (Mariani, 2001). Moreover, by analyzing only small-bodied fishes, target species are not taken into account and, therefore, the effect of fisheries on the fish community is isolated.

The present study aims to describe the composition of the small-bodied fish assemblages from the Ebro Delta coastal lagoons and to investigate the relationship between assemblages' structure and distribution patterns with the main environmental features that depend on the artificial water management. The degree of association of each species with the studied physicochemical parameters is also assessed.

2. Material and Methods

2.1 Study area

Three coastal lagoons located in the Ebro Delta were selected for this study: Encanyissada, Clot and Tancada (Fig.1). The Encanyissada lagoon with an area of 4.18 km², is the largest lagoon in the delta; it has a mean depth of 77 cm (121 cm maximum depth) and the salinity varies between 2 and 36 psu (practical salinity units). It is connected naturally to the Alfacs bay (Mediterranean Sea) by the Sant Pere canal (see Fig. 1) and to the Clot lagoon by a floodgate. The Clot is a small (0.56 km²) and shallow lagoon, with a mean depth of about 53 cm (maximum depth around 90 cm). The salinity of this lagoon ranges between 1 and 29. Closest to the sea, the Tancada lagoon has a surface area of 1.85 km², 58 cm mean depth (80 cm maximum depth) and its salinity ranges between 8 and 37. It is connected to the Alfacs bay by several artificial canals (Fig.1).

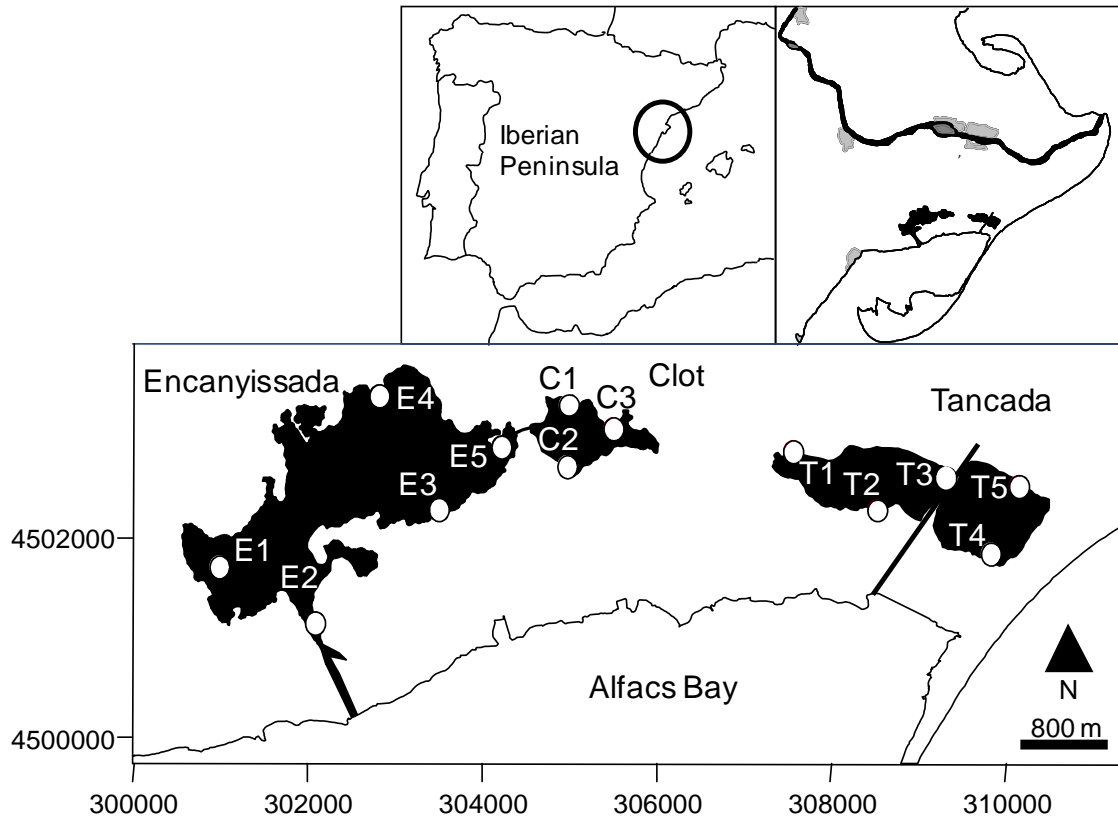


Figure 1. Map showing the location of the Ebro Delta and the sampled coastal lagoons. White spots represent sampling sites.

2.2 Sampling procedures

Two sampling periods were selected according to the artificial freshwater input scheme: March and September 2008 coinciding with the periods of minimum and maximum artificial freshwater inputs, respectively. Fish were collected with fyke nets, a passive fishing gear being increasingly used for scientific surveys (Poole *et al.* 2007) and widely used to survey small-bodied fish (Brazner *et al.* 1998, Pampoulie *et al.* 2001, Uzarski *et al.* 2005, Fredberg *et al.* 2009, Wedderburn *et al.* 2012). Fyke nets used were 2 m long with a hoop diameter of 80 cm and 5.0 mm of mesh size. A total of 26 fishing sets were settled in 13 different sampling locations in a way that the maximum spatial gradient was covered in each lagoon (5 in the Encanyissada and Tancada lagoons, 3 in the Clot lagoon) (Fig.1). As coastal lagoons are shallow water bodies with a quite constant depth, and therefore aquatic habitat conditions, the fyke nets were set in the littoral zone where there is a higher probability of capturing fish due to habitat characteristics (higher macrophyte coverage and presence of helophytic and littoral vegetation). In fact, this same procedure is done by local fishermen. Moreover, setting the fyke nets in the littoral zone allowed maintaining a constant depth between all the sampling points of approximately 80 cm, corresponding to the fyke nets' hoop diameter (the whole water

column was sampled for small bodied fish). All nets were set on late afternoon and hauled the next morning, hence having an average soak time of 12 hours. The fish were sorted, identified at species level and counted. This procedure allowed the accomplishment of the fish specimens capture requirements as stated in the scientific capture permit issued by the Catalan government: eliminate the exotic species according to the current law (in this case with excess of anesthetic MS-222) and release the native species after handling them. Each sampling site was also sampled for environmental descriptors that are directly affected by the water management. These descriptors are related with water physicochemical parameters, water nutrient loading variables, and water level expressed as depth (cm). Concerning the physicochemical parameters, dissolved oxygen (mgL^{-1}), temperature ($^{\circ}\text{C}$), salinity and pH were measured with an YSI 556 multi-parameter probe. To determine the water nutrient loading water samples were collected, preserved on ice in the absence of light, transported to the laboratory and stored at -20°C until a dissolved nutrient concentration analysis was done. The nutrients analyzed were ammonium (N-NH_4), nitrite (N-NO_2), nitrate (N-NO_3) and phosphate (P-PO_4), following Koroleff (1977).

2.3 Statistical analysis

Fish abundance was expressed as Captures Per Unit Effort (CPUE). CPUE is an index of relative abundance that represents the success of capture. CPUE values were calculated as number of captured fish per fishing set. Fishing time was assumed to be equal, since soaking time was roughly constant. Rare species ($\text{CPUE} < 0.833$ and/or present in just one sampling site) were excluded from the analyses. Normality and homogeneity of variances of all physicochemical variables were tested with Shapiro-Wilk test and Levene statistic (Clifford and Taylor 2008). Physicochemical variables were log-transformed and species data were square-root transformed when necessary in order to improve linearity as well as the normality of variances (Quinn and Keough 2002). Analysis of variance (ANOVA) was used to compare physicochemical descriptors between lagoons and sampling periods, followed by Games-Howell post-hoc test (GH test). A Canonical Correspondence analysis (CCA) was performed in order to extract the species variation explainable by the measured physicochemical parameters (Lepš and Šmilauer 2003). To describe the individual species response to salinity Generalized Additive Models (GAMs) were fitted assuming a Poisson distribution and a log link function. GAMs are an extension of the general lineal models that do not require the assumption of a relationship between environmental variables and the species distribution (Hastie and Tibshirani 1986, Lepš and Šmilauer 2003). The model complexity

of GAMs was selected by the stepwise selection procedure using the Akaike Information Criterion (AIC). AIC considers not only the goodness of fit but also parsimony, penalizing very complex models (Lepš and Šmilauer 2003). Thus the relationship between the predictors and the dependent variable can take on any shape, from a straight line to non-parametric curves of increasing complexity (Alcaraz *et al.* 2011). Tolerance ranges of the species were analyzed by data attribute plots with salinity as the environmental variable. The statistical analyses were performed with SPSS 17.0 and CANOCO 4.5.

3. Results

3.1 Environmental descriptors

The environmental descriptors of the three lagoons were compared between the two sampled periods (Table 1). Significant differences were found in: temperature ($F_{1,92}=95.57$; $p<0.01$), depth ($F_{1,92}=47.63$; $p<0.01$), pH ($F_{1,92}=19.15$; $p<0.01$) and salinity ($F_{1,94}=100.33$; $p<0.01$); whereas no differences were found in oxygen concentration ($F_{1,91}=0.22$; $p=0.64$). The mean temperature in March was significantly lower than in September (GH , $p<0.01$); Tancada lagoon being the coldest and Clot lagoon the warmest one (GH , $p<0.01$). Regarding depth, all three lagoons presented significantly higher mean values in September, with Encanyissada lagoon being the deepest one (GH , $p<0.01$). The pH was highest in September, being statistically significant only for Clot (GH , $p<0.05$) and Tancada lagoons (GH , $p<0.01$). Tancada lagoon showed the lowest pH values. In contrast, the salinity was significantly higher in March than in September (GH , $p<0.01$). The maximum observed salinity was in the Tancada lagoon in March (30.30; GH , $p<0.01$), whereas the Clot lagoon demonstrated the lowest water salinity (9.99; GH , $p<0.01$). In addition, temperature, depth and pH were negatively correlated with salinity ($r=-0.87$, $p<0.01$; $r=-0.37$, $p<0.01$ and $r=-0.58$, $p<0.01$, respectively).

Regarding the dissolved nutrients concentration (Table 1), significant differences in nitrite and phosphate were found between sampling periods ($F_{1,42}=8.21$ and $F_{1,41}=102.54$, respectively $p<0.01$). Nitrite concentrations were significantly higher in September, Tancada lagoon being the one reaching higher concentrations (0.085 ± 0.035). The phosphate was significantly higher in September (GH , $p<0.05$) with Clot showing the highest values. Moreover a negative correlation was found between salinity and phosphate ($r=-0.53$, $p<0.01$). Furthermore, although no differences were found in ammonium and nitrate ($F_{1,42}=2.92$ and $F_{1,41}=1.09$ respectively; $p>0.09$) their concentrations were higher in September, except for nitrate in Encanyissada.

Table 1. Environmental descriptors for the three lagoons in the two sampled periods. Mean \pm standard deviation is shown. DO: dissolved oxygen. ** statistical significance among seasons. *** statistical significance among lagoons. 'ns' non significant differences.

		March 2008						September 2008					
		Clot		Encanyissada		Tancada		Clot		Encanyissada		Tancada	
T (°C)	*	19.74 \pm 0.63	**	16.97 \pm 0.99	**	11.87 \pm 0.34	**	21.52 \pm 0.47	**	20.15 \pm 1.31	**	19.78 \pm 0.96	**
DO (mg l ⁻¹)	ns	8.28 \pm 0.82		7.96 \pm 1.12		8.30 \pm 1.17		7.93 \pm 0.84		8.00 \pm 1.18		7.95 \pm 0.76	
Salinity	*	9.99 \pm 1.01	**	18.40 \pm 4.91	**	30.30 \pm 0.89	**	1.20 \pm 0.16	**	9.04 \pm 2.00	**	10.29 \pm 1.78	**
pH	*	7.93 \pm 0.07	**	8.11 \pm 0.16	ns	7.72 \pm 0.09	**	8.23 \pm 0.18	**	8.10 \pm 0.11	ns	8.21 \pm 0.18	**
P-PO ₄ (mg l ⁻¹)	*	>0.001	**	0.012 \pm 0.004	**	0.006 \pm 0.003	**	0.045 \pm 0.022	**	0.039 \pm 0.011	**	0.040 \pm 0.011	**
N-NH ₄ (mg l ⁻¹)	ns	0.003 \pm 0.003		0.037 \pm 0.023		0.156 \pm 0.176		0.055 \pm 0.092		0.100 \pm 0.071		0.194 \pm 0.059	
N-NO ₂ (mg l ⁻¹)	*	0.003 \pm 0.000	ns	0.006 \pm 0.003	ns	0.008 \pm 0.003	**	0.021 \pm 0.019	ns	0.007 \pm 0.002	ns	0.085 \pm 0.035	**
N-NO ₃ (mg l ⁻¹)	ns	0.010 \pm 0.006		0.157 \pm 0.165		0.075 \pm 0.077		0.191 \pm 0.320		0.023 \pm 0.044		0.374 \pm 0.265	
Depth (cm)	*	39.29 \pm 11.67	**	65.07 \pm 15.23	**	56.50 \pm 10.31	**	68.43 \pm 7.57	**	91.30 \pm 17.35	**	67.50 \pm 11.63	**

Table 2. List of species captured in the Ebro Delta lagoons and its abundance in CPUE. Lagoon: (C) Clot; (E) Encanyissada; (T) Tancada. Status: (N) native; (I) introduced. Species with code in **bold** were selected for the analysis.

Family	Common name	Species	Code	State	March 2008			September 2008		
					C	E	T	C	E	T
Anguillidae	European eel	<i>Anguilla anguilla</i>	AAN	N	0.167	0.083	0.250		0.083	0.083
Atherinidae	Sand smelt	<i>Atherina boyeri</i>	ABO	N	0.500	3.417	7.417	4.000	21.667	0.167
Cyprinidae	Carp	<i>Cyprinus carpio</i>	CCA	I			0.083			
Cyprinodontidae	South European toothcarp	<i>Aphanius fasciatus</i>	AFA	I		0.083				
Cyprinodontidae	Spanish toothcarp	<i>Aphanius iberus</i>	AIB	N	3.167	36.083	15.333		2.417	0.417
Gobiidae	Common goby	<i>Pomatoschistus microps</i>	PMI	N	30.917	148.91	221.75	77.083	22.083	10.833
Mugilidae	Thicklip grey mullet	<i>Chelon labrosus</i>	CLA	N		3.083	1.583		0.833	0.083
Mugilidae	Golden mullet	<i>Liza aurata</i>	LAU	N		0.167	6.167		1.250	
Mugilidae	Thinlip grey mullet	<i>Liza ramada</i>	LRA	N			4.083			
Mugilidae	Leaping mullet	<i>Liza saliens</i>	LSA	N	0.333	0.167	17.333		2.000	0.500
Mugilidae	Flathead mullet	<i>Mugil cephalus</i>	MCE	N			0.833			
Poecilidae	Eastern mosquitofish	<i>Gambusia holbrooki</i>	GHO	I	5.333	3.250	0.333	218.33	95.917	0.417
Syngnathidae	Black-striped pipefish	<i>Syngnathus abaster</i>	SAB	N		0.250	0.083			

3.2 Fish assemblages

Overall, 13 species belonging to 8 different families were captured in the three lagoons during the two sampling periods (Table 2). The small bodied fish composition in the lagoons was composed by three introduced species (23.08%) and ten native species (76.92%). Although the native species dominated in terms of richness in all lagoons and sampling seasons, in terms of abundance the introduced species (*Gambusia holbrooki*) was dominant in September (69%). This pattern was observed in all the lagoons except in the Tancada where the native species dominated in the two periods (Table 2). For further analysis, the rare species were excluded, so finally 7 species belonging to 5 families were analyzed (Fig. 2). The small-bodied fish community in March was dominated by *Pomatoschistus microps*, *Aphanius iberus* and *Liza saliens*. *P. microps* was present in all lagoons and sampling sites, whereas *A. iberus* was present in all sampling points, except T2 and T3. *L. saliens* was mostly present in saltier habitats, mainly in the Tancada lagoon (Fig. 2). The presence of *L. aurata* was low and always concentrated in the saltier environments. The invasive *Gambusia holbrooki* occurred in more freshwater areas (Clot and Encanyissada lagoons). *Chelon labrosus* in March was found almost exclusively in sampling points with sea connections (Encanyissada and Tancada lagoons). In September the community was dominated by freshwater and brackish species such as *G. holbrooki*, *P. microps* and *Atherina boyeri*. In this period, *P. microps* was dominant in the Tancada lagoon, whereas both the Clot and Encanyissada lagoons were dominated by *G. holbrooki*. The species *A. boyeri* was abundant in the Encanyissada lagoon. The endangered *A. iberus* was present in all lagoons except in Clot. The mugilids *L. saliens* and *C. labrosus* occurred in the saltier habitats of this period (Tancada lagoon), whereas the presence of *L. aurata* in this period was restricted to the Encanyissada lagoon.

3.3 Effects of environmental descriptors on the fish assemblages

The first two axes of the CCA accounted for 27.4% of the total variability of the species data. The CCA summary statistics resulted in a significant first axis species-environment relationship ($r=0.836$, $p<0.01$). Salinity ($F=5.17$, $p<0.01$) and depth ($F=2.42$; $p<0.05$) were the only significant variables in the CCA analysis (Fig. 3). The first axis suggested a salinity gradient derived from the differences in the artificial freshwater inputs, which split September samples (fresher and deeper) from March samples (shallower and saltier). Regarding the species scores, brackish species as

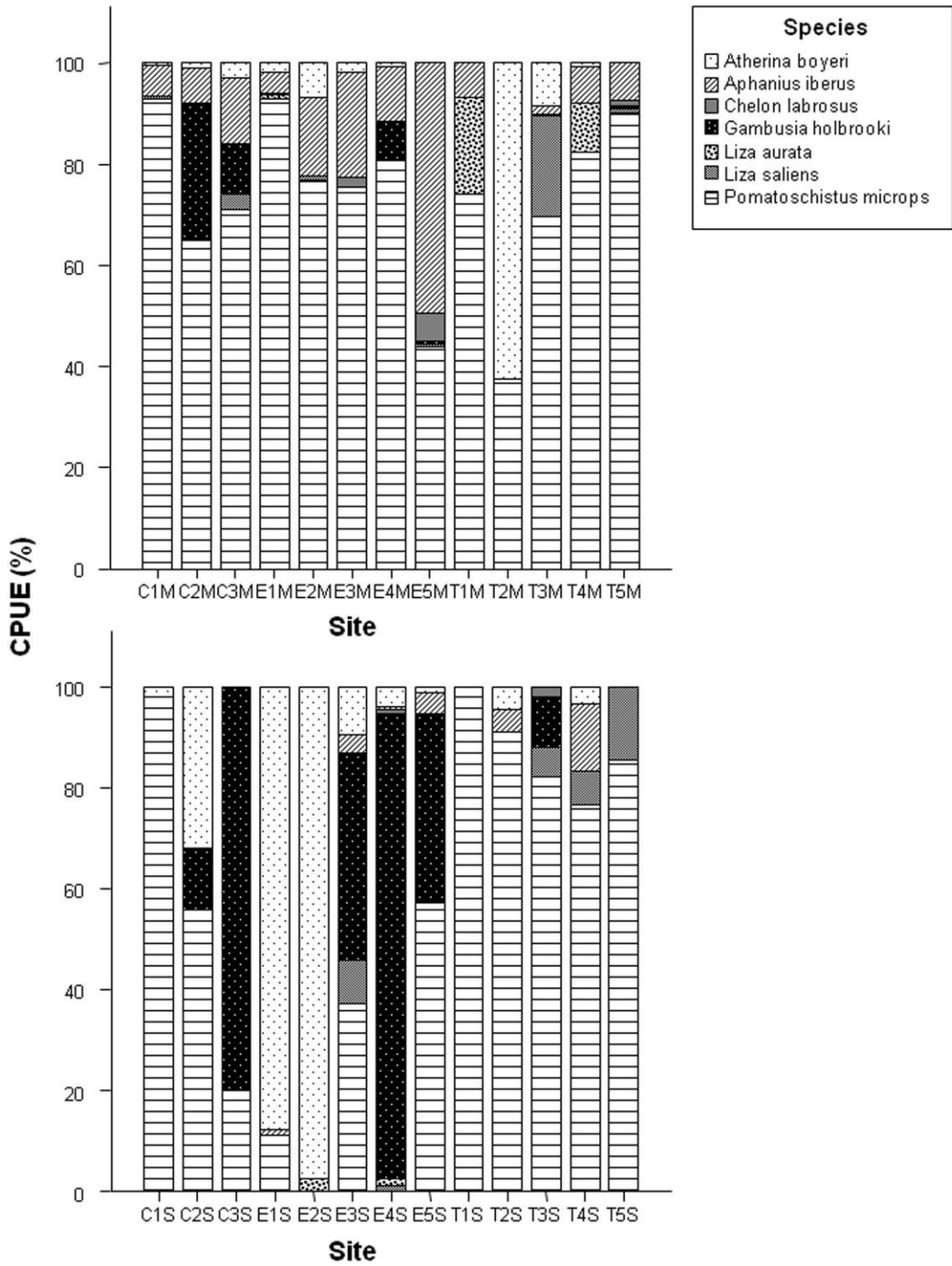


Figure 2. Percentage contribution of species to the overall abundance in the three lagoons sampled in March (above) and September (below). See figure 1 for sampling sites codes.

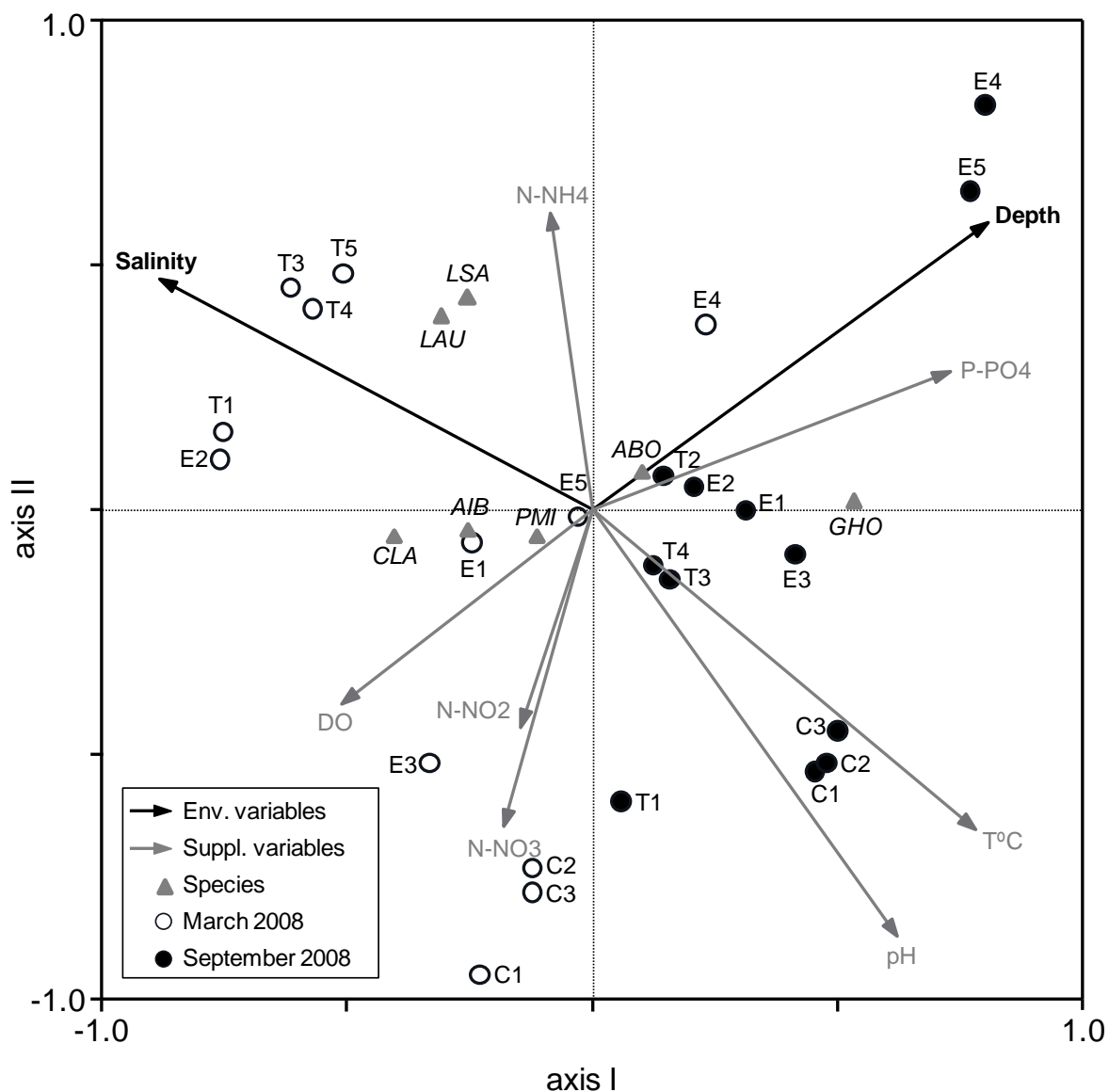


Figure 3. Correspondence canonical analysis triplot showing the relationships between physicochemical parameters, sampling sites and species. Significant and non significant (supplementary) physicochemical parameters are plotted. Species abbreviations: ABO (*Atherina boyeri*), AIB (*Aphanius iberus*), CLA (*Chelon labrosus*), GHO (*Gambusia holbrooki*), LAU (*Liza aurata*), LSA (*Liza saliens*) and PMI (*Pomatoschistus microps*). See figure 1 for sampling codes.

Aphanius iberus, *Pomatoschistus microps*, *Liza saliens*, *Liza aurata* and *Chelon labrosus*, showed negative coordinate values, i.e. associated with higher salinity; whereas species with freshwater origin as *G. holbrooki* were situated in positive coordinates. Other species as *Atherina boyeri* was positioned near the center of the plot. The samples ordination in the second axis (4.1% of the total variability) indicated nitrogen loading with positive coordinate values for the ammonium and negative values for nitrate and nitrite. Then, high ammonium concentration samples were situated in the upper part of the plot (Tancada in March) and low ammonium loaded samples at the bottom (Clot in March).

3.4 Species response to salinity

The response curves of species abundances (expressed as CPUE square root transformed) to salinity are shown in Figure 4. The significance of the response curves are shown in Table 3. *Gambusia holbrooki* decreased drastically with salinity, whereas *Pomatoschistus microps* increased with salinity values higher than 20. The mugilids, *Liza saliens* and *Liza aurata* increased linearly with salinity; whereas the endangered *Aphanius iberus* showed salinity preferences around 15, with a peak of abundance at 20. *Chelon labrosus* and *Atherina boyeri* were not plotted as they were not selected during the GAM iterative process. The salinity range for each species is shown in Figure 5. *G. holbrooki* has its optimum in freshwater environments, whereas *A. boyeri* is more adapted to mesohaline waters (around 15). The rest of the species have their optimums in brackish to marine waters (18-30).

Table 3. Results of the final Generalized Additive Models (GAM) applied to all the selected species showing the explained deviance of the fitted model (Deviance), the residual degrees of freedom (DF residual) the Akaike's Information Criterion (AIC), and the model significance (F value). Asterisks denote significance levels: **, $p < 0.01$; *, $p < 0.05$.

Species	Deviance	DF	AIC	F	
<i>Pomatoschistus microps</i>	127.55	21	161.66	3.83	*
<i>Gambusia holbrooki</i>	230.58	22	278.27	12.18	**
<i>Aphanius iberus</i>	70.87	21	87.36	10.73	**
<i>Liza saliens</i>	63.54	21	86.00	4.83	*
<i>Liza aurata</i>	44.99	22	62.37	4.06	*
<i>Chelon labrosus</i>	28.40	21	37.75	6.02	**

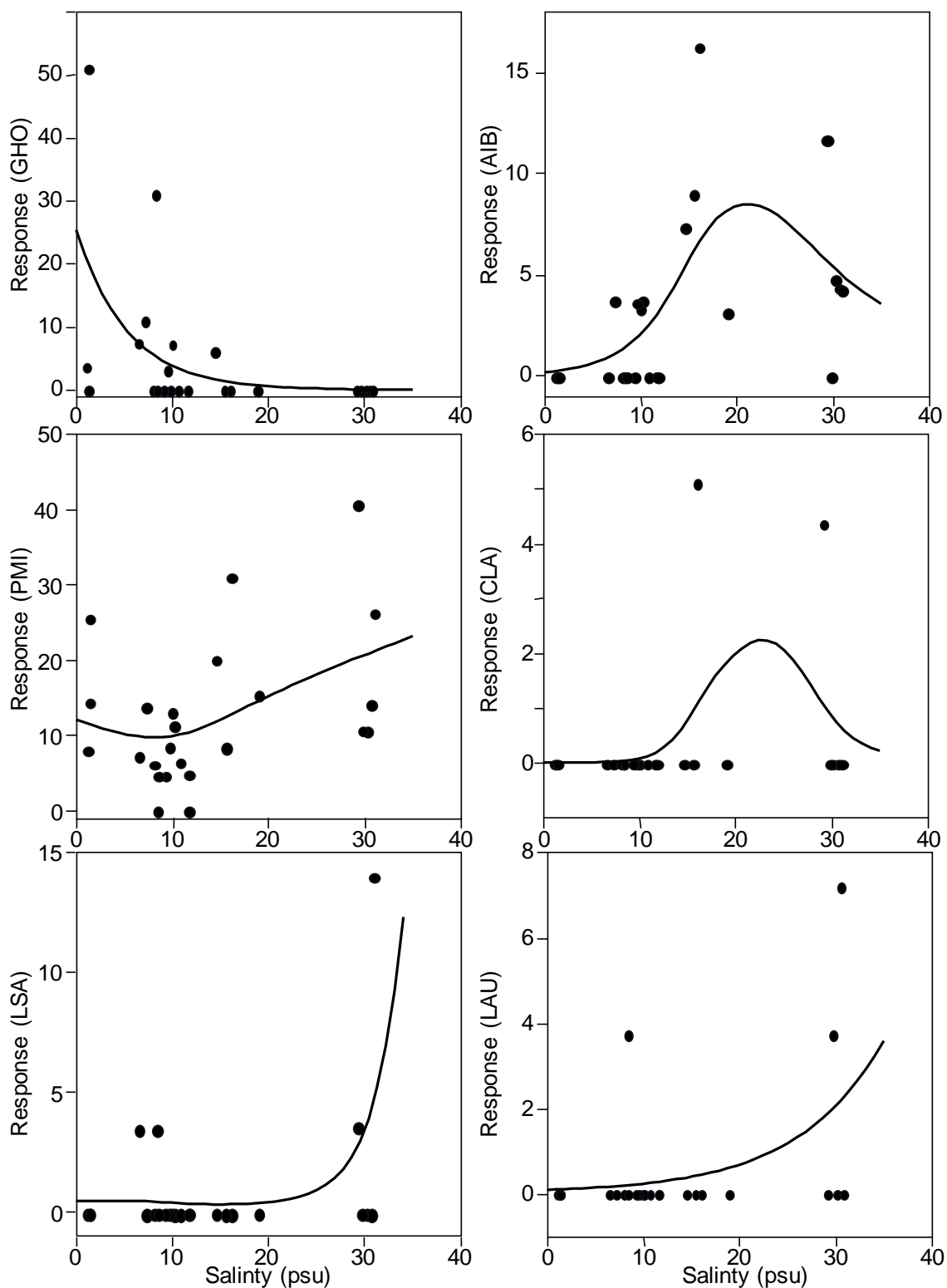


Figure 4. Response curves of species abundance (species selected by the model) to salinity. The curves are the generalized additive models (GAM) selected by the Akaike information criteria (AIC). Dots represent sampling sites. See Fig. 3 for species abbreviations.

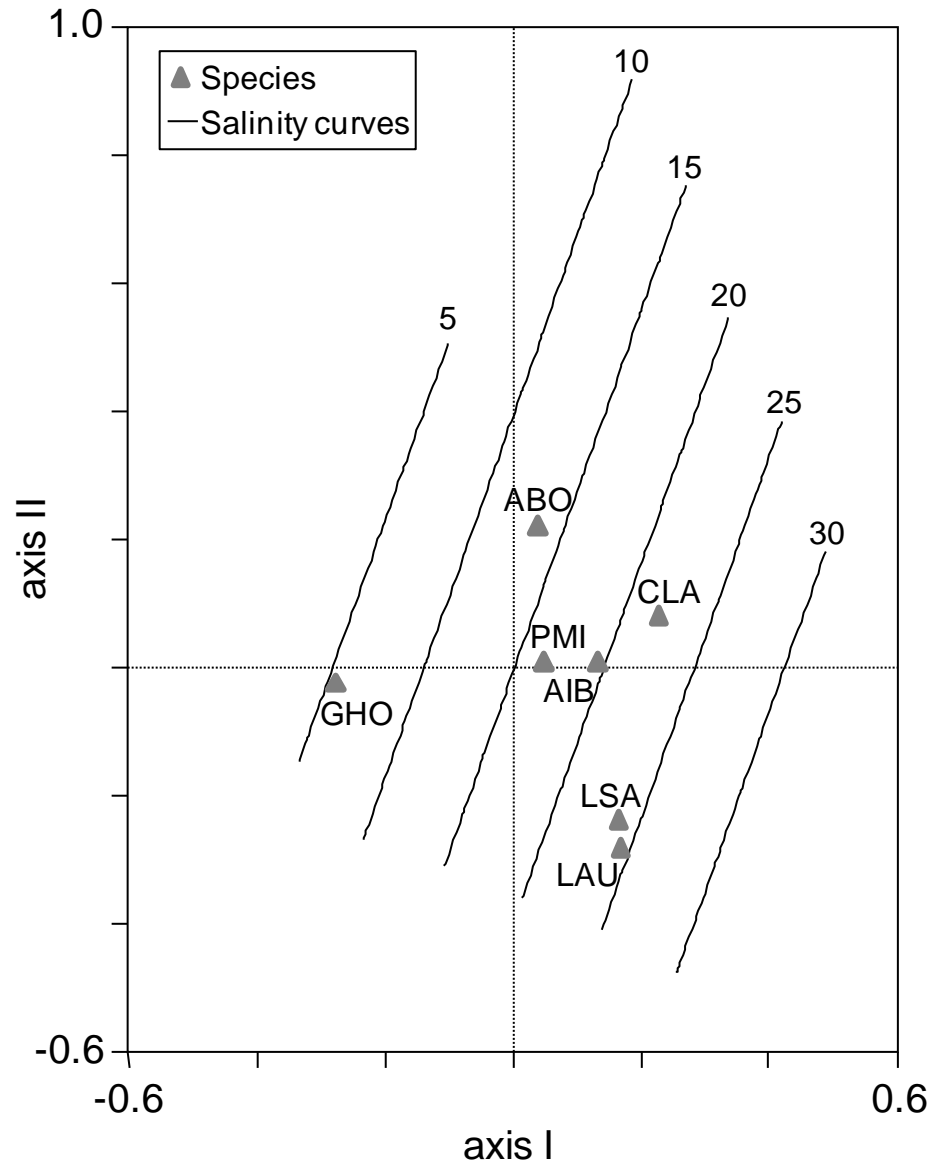


Figure 5. Species attribute plot showing salinity curves and the species' preferences. Species abbreviations: ABO (*Atherina boyeri*), AIB (*Aphanius iberus*), CLA (*Chelon labrosus*), GHO (*Gambusia holbrooki*), LAU (*Liza aurata*), LSA (*Liza saliens*) and PMI (*Pomatoschistus microps*).

4. Discussion

The continuous freshwater inputs during nine months per year have inverted the hydrological pattern in the Ebro Delta coastal lagoons (De Sostoa and De Sostoa 1985, Comín *et al.* 1987). The inverted pattern was further confirmed in the present study by the opposite salinity cycle, since lagoons were found to have fresher water during dry seasons, when lowest freshwater inputs took place (Badosa *et al.* 2007). September

was not only characterized by lagoons with lower salinity, but also because was the period with higher pH values. This results, contrasted with previous studies which reported higher pH values in March (De Sostoa and De Sostoa 1985, Comín *et al.* 1987). These differences could be explained by the different management applied at each time, as the mentioned previous studies were carried out before the circumvallation canals construction (Forés *et al.* 2002). Furthermore, besides the inversion of the hydrological pattern, the continuous freshwater supplies, tended to reduce the differences among the lagoons in September; whereas in March, the differences between the lagoons were increased due to the cessation of the freshwater flow. Although the continuous freshwater inputs acted as homogenizer agent, some differences were still found among the lagoons within the same period, which could be attributed to a different water management scheme in each lagoon. This was the case of the Tancada lagoon, the one receiving less freshwater inputs and the Clot lagoon receiving the largest freshwater supplies. Then, these differences in freshwater inputs, in addition to the proximity to the sea, might explain the extreme differences in salinity values found between the Tancada and Clot lagoons.

The freshwater inputs in the lagoons are a mixture of drainage water from the rice fields and irrigating water from the Ebro River. During its entire course, the Ebro River collects nutrients from all the agricultural activities developed in the basin (Mañosa *et al.* 2001). Furthermore, though rice fields are known to act as ecological filters, the drainage water still contains a high amount of nutrients (Forés and Comín 1992). As a result, water drained into the lagoons is high-loaded in nutrients. This fact, might explain the negative correlation between nutrient loading and salinity, which is common in coastal systems affected by agriculture activities (Chapelle *et al.* 2000, Lucena *et al.* 2002, Pérez-Ruzafa *et al.* 2005a). Moreover, this freshwater input could promote turbulence, favoring the internal recycling process in the lagoons, as has been previously suggested for phosphate and ammonium (Chapelle *et al.* 2000, Orfanidis *et al.* 2005, Badosa *et al.* 2007).

The amount of freshwater/marine inputs have been pointed out as the key factor in determining the fish assemblages in an ecosystem (Gordo and Cabral 2001; Mariani 2001). Therefore, any change that could modify the natural inputs of freshwater/marine water (as the artificialization of the hydrological cycle), would consequently modify the existing fish assemblage. Fish assemblages of Mediterranean coastal lagoons with climatic-dependent hydrology (where climatic variation is the only source of freshwater inputs e.g. rain, floods), have been previously described to be characterized by marine

and brackish species with the dominance of the families Sparidae, Mugilidae, Gobiidae, Atherinidae, Soleidae, Sygnathidae and Blennidae (Pérez-Ruzafa *et al.* 2006, Maci and Basset 2009, Verdiell-Cubedo 2009). While, fish assemblages of coastal lagoons with artificial freshwater supplies, as is the case of the Vaccarès lagoon (France; Poizat *et al.* 2004) and the Albufera lagoon (Spain; Blanco *et al.* 2003, Blanco and Romo 2006), presented families typical of freshwater environments, in addition to the aforementioned brackish and marine species. Our results suggest that the hydrological alteration of the cycle might be the responsible for the current fish community in the Ebro Delta coastal lagoons composed by brackish and marine families as Gobidae, Mugilidae and Atherinidae and by freshwater species belonging to the families Cyprinidae and Poeciliidae (Blanco *et al.* 2003, Poizat *et al.* 2004, Blanco and Romo 2006). Moreover, the fact that the freshwater species (mostly the introduced *Gambusia holbrooki*) present in those lagoons were introduced (Doadrio 2001), lead to an alteration of the native species assemblage. The effect of the altered hydrological pattern over the native/introduced species proportion was correlated with the management carried out in each lagoon; i.e. the higher proportion of native species found in the Tancada lagoon, even in the fresher period (September), reflects a lower perturbation of the fish community by the water management. In summary, data confirms that the small-fish community composition of the Ebro Delta coastal lagoons was affected by the altered hydrological pattern, which is in concordance with other studies on coastal ecosystems suffering from hydrological alterations (Flower 2001, Gordo and Cabral 2001, Poizat *et al.* 2004, Pérez-Ruzafa *et al.* 2005b, Badosa *et al.* 2007).

The hydrological alteration was also responsible for structuring the small-bodied fish community through salinity. The alternation of a freshwater input period with a non-input period, creates a wide salinity range in the coastal lagoons that fish have to cope with. Fish have the capacity to keep their plasma osmotic concentration constant by osmoregulation process (Rigal *et al.* 2008). However, this capacity is different among species, some being more or less tolerant to salinity variations (Blaber 1997, Marshall and Elliott 1998, Rigal *et al.* 2008). The abrupt decrease in abundance of the freshwater species *Gambusia holbrooki* with increasing salinities was probably due to the fact that its upper salinity tolerance is at 25 (Nordlie and Mirandi 1996). This salinity tolerance could also explain its absence in Tancada lagoon in March, where the average salinity raised up to 30. Moreover, a peak of abundance of the endangered *Aphanius iberus* was observed with the decrease of the invasive *G. holbrooki*. This could be attributed to the interaction of these two species, as some recent studies pointed out. While some authors suggested that a drastically decrease of *A. iberus* in the presence of the *G.*

holbrooki occur not only due to the salinity preferences but also to the fact that *G. holbrooki* is more competitive due to its higher foraging ability, especially when this species outnumber the native (Caiola and De Sostoa 2005). Other authors hypothesized a possible predation of *G. holbrooki* over *A. iberus* juveniles (Rincón *et al.* 2002). In whatever circumstances, our observed higher presence of the endangered *A. iberus* species in more saline habitats seems to be a response to avoid competition with the invasive *G. holbrooki* (Doadrio 2001, Rincón *et al.* 2002, Caiola and De Sostoa 2005).

The high osmoregulation capacity of *Pomatoschistus microps* (Rigal *et al.* 2008), was probably the reason of its presence in all three lagoons in both periods. Although this species shows a tolerance to a wide salinity range, a peak of abundance was found around 27, thus suggesting a higher preference for euhaline environments. Although *Atherina boyeri* was also found in all lagoons and periods, this species was not selected in the GAM analysis procedure due to a lack of response of its CPUE to the salinity gradient. A possible hypothesis to explain this fact could be that *A. boyeri* is the only small-bodied fish from the Ebro Delta coastal lagoons that is a target species (Rodríguez-Climent *et al.* 2012). The period of higher salinity coincides with the professional fishing season (from October to March). Therefore, the expected increase in the abundance of this species during the period of higher salinity is offset by the effect of fishery. Although members of the Mugilidae family spawn at sea (Kottelat and Freyhof 2007, Cardona *et al.* 2008), during their juvenile stages they enter into coastal lagoons looking for a protected area with abundant food that will provide all the favorable conditions for their development (Verdiell-Cubedo 2009). In these habitats, salinity has been pointed out to be a key factor in determining the distribution pattern of young mugilids (Lasserre and Gallis 1975, Cardona 2006, Mićković *et al.* 2010). The low osmoregulation capacity of the juveniles of *Liza saliens* and *Liza aurata*, together with their optimization of growth at high salinity levels (Cardona 2006, 2008), might explain its distribution in the present study, being absent in freshwater areas and showing preference for high salinity areas. Moreover, the low abundance of *Chelon labrosus* in September was in accordance to its reduced ability of osmoregulation in freshwaters, where the species is not able to survive for more than a few months (Lasserre and Gallis 1975). Furthermore, the presence of *C. labrosus* in sites closest to the sea, supported the hypothesis that marine connectivity plays an important role in the abundance of this species (Cardona *et al.* 2008). Although no doubt exists about the role salinity played in the distribution patterns of the family; some authors have hypothesized the high trophic overlap found among young mugilids, to determine its survival in low resources conditions (Gisbert *et al.* 1996). Thus, the dominance of *L. saliens* above *L. aurata* (the

only other species with preference for high salinity ranges) found in our study, could be explained by a possible trophic competition (Gisbert *et al.* 1995, 1996, Cardona *et al.* 2008).

5. Conclusions

The present study is the first research approach describing the impact of the hydrological alteration on the fish assemblages of the Ebro Delta lagoons. It can be concluded that water management practices provoking hydrological alterations have a direct impact on the structure and dynamics of the coastal lagoons' fish assemblages. The current water management scheme developed in the Ebro Delta Natural Park is favoring the introduced species in detriment of the native ones. Of special mention was the inverse abundance trends found between the endangered *Aphanius iberus* and the invasive *Gambusia holbrooki*. This finding not only evidences two opposite salinity preferences but also corroborates previous studies that demonstrate that the invasive species affects negatively the native. In addition, the different water management applied to each lagoon allowed us to conclude that the best management scheme regarding the fish assemblages was the one undertaken in the Tancada lagoon, characterized by less freshwater inputs during the rice cultivation period (May-December). Moreover, with salinities higher than 15-20 throughout all the year, a fish community with a dominance of native species important both for commercial and conservation purposes is assured. This should be set as a key criterion in the Ebro Delta coastal lagoons future water management schemes.

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GENERAL DISCUSSION

7

The present PhD thesis evaluates the effects of two of the main anthropogenic stressors affecting the Ebro Delta coastal lagoons, **artisanal fisheries** and **water management**, on the inhabiting fish assemblages. To assess the artisanal fisheries effects, studies on fishing gears mesh size selectivity were carried out (Chapter I). Moreover, a vulnerable species with a growing commercial interest without an established Minimum Landing Size (MLS), the sand smelt (*Atherina boyeri*), was used as case study to propose a MLS based on a combination of reproductive biology parameters and fishing gears' mesh sizes (Chapter I). The field data gathered to carry out the studies of Chapter I, was also used for a detailed description of the state of the art of the fish assemblages. In addition, an ecological based modelling approach was used to predict the effects of different long term fishing efforts scenarios on the Ebro Delta coastal lagoons biological communities' structure (with special emphasis on fish) (Chapter II). Finally, the influence of hydrological alteration on the fish assemblages was assessed throughout the development of statistical models that allow the establishment of relationships between the fish assemblages' structure and distribution patterns and the main environmental features that depend on the water management (Chapter III).

7.1 Recent changes in the Ebro Delta coastal lagoons fish fauna

Since last studies regarding Ebro Delta coastal lagoons' ichthyofauna are from the 80's (De Sostoa, 1983; De Sostoa and De Sostoa, 1985), the results presented in this thesis allowed the analysis of recent changes in the fish community. Comparing data from the 80's, 25 fish species belonging to 15 families were found in the three studied coastal lagoons in the present thesis (Table 1). From those, 18 species were already present in the Ebro Delta coastal lagoons in the 80's (De Sostoa, 1983; De Sostoa and De Sostoa, 1985; De Sostoa and Lobón-Cerviá, 1989; De Sostoa *et al.*, 1990). Thus, **7 new fish species** were found for the first time in the coastal lagoons' by the present thesis: bleak

(*Alburnus alburnus* (Linnaeus, 1758)), Ebro barbel (*Barbus graellsii* (Steindachner, 1866)), South European toothcarp (*Aphanius fasciatus* (Valenciennes, 1821)), Senegalense sole (*Solea senegalensis* Kaup, 1858), topmouth gudgeon (*Pseudorasbora parva* (Temminck & Schlegel, 1846)), pike-perch (*Sander lucioperca* (Linnaeus, 1758)) and wels catfish (*Silurus glanis* Linnaeus, 1758) (Table 1; Chapter I and III). The last 3 of these 7 new found species in the lagoons were already cited to be present in the delta by recent studies. *P. parva*, an allocthonous fish species in this region, was first described in the Ebro Delta canals by Caiola and De Sostoa (2002), although the origin is not clear, the authors pointed out an involuntary escapement from an aquaculture installation in the area as the most probable hypothesis for its presence. Then, the species reached the lagoons thanks to the artificial Ebro Delta canals network (Chapter III).

In contrast *S. lucioperca* and *S. glanis* are **allocthonous freshwater species** intentionally introduced to the Ebro River in 1990 and 1974 respectively, for its high value in sport fishing (Doadrio, 2001; CHE, 2009). Once in the river and again through the vast extent network of canals (Chapter III), their arrival to the coastal lagoons was a question of time. The uncontrolled introduction of the aforementioned species in the Ebro Delta has instigate: (i) their spread on the Ebro River (CHE, 2009) and (ii) its migration/displacement to other ecosystems such as the coastal lagoons. This deliberated or accidental introduction of allocthonous fish species is of concern, since uncontrolled introductions can lead not only to unexpected changes in the trophic web, but can also affect native populations seriously, driving them to the extinction at worst-case scenario (Drake *et al.*, 1989; Mooney and Cleland, 2001; Blanco and Romo, 2006). In the Ebro Delta coastal lagoons, as a result of those fish species introductions, the introduced/native species ratio has been triplicated since the eighties, from 0.14 to 0.47, since three allocthonous species (*Carassius auratus* (Linnaeus, 1758); *Cyprinus carpio* Linnaeus, 1758 and *Gambusia holbrooki* Girard, 1859) found in the lagoons, were already reported at that time (De Sostoa, 1983; De Sostoa and Lobón-Cerviá, 1989; De Sostoa *et al.*, 1990). Those results are in agreement with the ones found in the Albufera lagoon (Valencia), another Spanish lagoon suffering from the rice cultivation drawbacks, where a notable increase of allocthonous species has been observed in recent times (Blanco and Romo, 2006). Thus, suggesting that rice irrigation canals act as a source of spreading allocthonous fish species downstream the river to the coastal lagoons.

Meanwhile, in the eighties some allocthonous species were found, and an increase in recent times have been observed; no records regarding the **threatened status** of the native species were found at that time. Contrary, nowadays 4 out of the total native fish species are considered threatened by international and/or national regulations: Spanish

toothcarp (*Aphanius iberus* (Valenciennes, 1846)), European eel (*Anguilla anguilla* (Linnaeus, 1758)), freshwater blenny (*Salaria fluviatilis* (Asso, 1801)) and sand smelt (*Atherina boyeri* Risso, 1810) (Doadrio, 2001; International Union for Conservation of Nature (IUCN), 2013). Although all of them are currently present in the lagoons, their permanence is different, being *A. iberus* and *A. boyeri* sedentary; *A. anguilla* migratory and *S. fluviatilis* considered as an accidental species; fact that must be taken into consideration in the designing of measures for their conservation accounting its threatened status. The denomination of this threatened status is however, controversial for some of the mentioned native species. While there is a consensus regarding the threatened status of *A. iberus* and *A. anguilla* by international and national legislation (considered as endangered species by the International Union for Conservation of Nature (IUCN) red list and the Spanish regulation (RD 139/2011)); some divergences exist in relation to the cases of *S. fluviatilis* and *A. boyeri*. In fact, those species are considered as least concern by the IUCN, but vulnerable by Spanish legislation (RD 139/2011 and Doadrio, 2001 respectively; Chapter I). Even more, such divergences are further regionally, being *S. fluviatilis* considered as endangered species only in some autonomous communities in Spain (*i.e.* Extremadura, Catalonia, Aragon, La Rioja and Basque Country; Doadrio, 2001).

The divergences found on some of the species threatened status, might suggest deficiencies on monitoring programs or differences of the environmental status among regions. Nevertheless, monitoring programs should be undertaken to assess the status of the threatened population first, and to provide a recovery process for them once the threatened status of a population is confirmed (Campbell *et al.*, 2002). However, due to the fact that it is an expensive and time-consuming process (Ross, 1997), is not always straightforward to assess the status of a threatened species. Further, recovery plans vary in their specifications of criteria for recovery, depending on factors that imperil endangered species (Campbell *et al.*, 2002). This is the case of the Ebro Delta coastal lagoons, where the aforementioned species are threatened by different stressors and have different permanence time in the lagoons, factors that must be considered in the designing of conservation plans for the species. With respect to the sedentary *A. iberus* species, main threats in the area include habitat degradation and introduction of other fish species like *G. holbrooki* or the mummichog (*Fundulus heteroclitus* Linnaeus, 1766) that have been reported to displace the native populations (Doadrio, 2001; Caiola and De Sostoa, 2005; Alcaraz, 2006; Chapter I). Regarding the migratory species *A. anguilla*, which is considered a global concern, as its population has been decreasing from 1980, and since 2000 is at its historical low, with just 1-5% of the pre-1980 levels showing a 95

Table 1. Comparative list between fish species found in the coastal lagoons during the current PhD thesis and the 80's.

Family	Species	Common name	State	Described by
Anguillidae	<i>Anguilla anguilla</i>	European eel	Native	Linnaeus, 1758
Atherinidae	<i>Atherina boyeri</i>	Sand smelt	Native	Risso, 1810
Blennidae	<i>Salaria fluviatilis</i>	Freshwater blenny	Native	Asso, 1801
Clupeidae	<i>Sardina pilchardus</i>	European pilchard	Native	Walbaum, 1792
Cyprinidae	<i>Alburnus alburnus</i>	Bleak	Native	Linnaeus, 1758
	<i>Barbus graellsii</i>	Ebro barbel	Native	Steindachner, 1866
	<i>Carassius auratus</i>	Goldfish	Introduced	Linnaeus, 1758
	<i>Cyprinus carpio</i>	Common carp	Introduced	Linnaeus, 1758
	<i>Pseudorasbora parva</i> *	Topmouth gudgeon	Introduced	Temminck & Schlegel, 1846
	<i>Sander lucioperca</i> *	Pike-perch	Introduced	Linnaeus, 1758
	Cyprinodontidae	<i>Aphanius fasciatus</i>	South European toothcarp	Introduced
<i>Aphanius iberus</i>		Spanish toothcarp	Native	Valenciennes, 1846
Engraulidae	<i>Engraulis encrasicolus</i>	European anchovy	Native	Linnaeus, 1758
Gobiidae	<i>Pomatoschistus microps</i>	Sand goby	Native	Krøyer, 1838
Moronidae	<i>Dicentrarchus labrax</i>	European sea bass	Native	Linnaeus, 1758
Mugilidae	<i>Chelon labrosus</i>	Thicklip grey mullet	Native	Risso, 1827
	<i>Liza aurata</i>	Golden grey mullet	Native	Risso, 1810
	<i>Liza ramada</i>	Thinlip grey mullet	Native	Risso, 1827
	<i>Liza saliens</i>	Leaping mullet	Native	Risso, 1810
	<i>Mugil cephalus</i>	Flathead grey mullet	Native	Linnaeus, 1758
Poecilidae	<i>Gambusia holbrooki</i>	Eastern mosquitofish	Introduced	Girard, 1859
Siluridae	<i>Silurus glanis</i> *	Wels catfish	Introduced	Linnaeus, 1758
Soleidae	<i>Solea senegalensis</i>	Senegalense sole	Native	Kaup, 1858
Sparidae	<i>Sparus aurata</i>	Gilthead seabream	Native	Linnaeus, 1758
Syngnatidae	<i>Syngnathus abaster</i>	Black-striped pipefish	Native	Risso, 1827

Name species in **bold** were found in both periods. * Species cited before in the Ebro Delta, but not in the coastal lagoons studied in the present PhD.

to 99% decline (FAO and ICES, 2008; IUCN, 2013); most important threats constitute: dams blocking migration routes, pollution, loss of wetlands, climate change and juvenile overfishing (*i.e.* glass eels) (IUCN, 2013; Dekker, 2003). Due to the migratory character of this species and the fact that its juveniles enter into the lagoons looking for a safe place to eat and grow, its conservation at the lagoonal stage is critical for the survival of the species. In the case of *S. fluviatilis*, and similarly to the aforementioned *A. iberus* species, allochthonous species also represents a threat, in this case by predated on adults and nests. Nevertheless, gravel extraction and habitat destruction stand out as the factors affecting the most the species population (Doadrio, 2001; IUCN, 2013). However, the accidental presence of this species in the lagoons is due to the freshwater inputs in those systems during the rice cultivation period, and thus although is a native species in the Ebro basin, its existence in the lagoons is artificial. Then, measures to mitigate its

endangered state should be applied at the Ebro's basin level where it is resident (CHE, 2009). In the case of *A. boyeri*, main threats in our country might include overfishing and predation by other fish species, being habitat alteration also included in the list of perils affecting the most the population trends (Doadrio, 2001) of this sedentary species in the lagoons.

As a result, some of the species forming the coastal lagoons fish community are considered threatened species, fact that differs from the fish community found 30 years ago; thus highlighting the necessity of promoting conservation policies. In this sense, research work carried out in the present thesis, allows giving some recommendations to improve *A. boyeri* and *A. anguilla* fish populations, and to evaluate the success of those recommendations. In one hand, the implementation in synchrony of the Minimum Landing Size (MLS) and the mesh size regulation for *A. boyeri* proposed for the first time in the presented work, might help on the species conservation in the Ebro Delta coastal lagoons (Chapter I; see next section); on the other hand, the model developed in the Tancada lagoon (Chapter II) might elucidate if the evaluation and conservation plan that it is being carried out at European level for the endangered *A. anguilla* (see next section) is effective.

The previous observations leads directly to the question: *Can the rise of introduced species be related with the threatened status of some of the native species found nowadays in the lagoons?* However, the answer is not as direct as the question arises. Plenty of studies have reported the negative effects of introduced species when they become invasive on native populations - by reducing genetic biodiversity, promoting biotic homogenization, competing for food, by hybridization and competitive exclusion (Huxel, 1999; Rahel, 2000, Caiola and De Sostoa, 2005) -, but none have highlighted the threatened status as one of them. Albeit considering that at worst, invasive species can lead to the extinction of native ones (*i.e.* they are listed as the second cause of North American fish and world fish extinction (Miller *et al.*, 1989; Harrison and Stiassny, 2004)), and reinforced by some historical examples such as the introduction of the Nile perch (*Lates niloticus* (Linnaeus, 1758)) into lake Victoria which resulted in the loss of hundreds of native cichlid fish species (Moyle and Light, 1996) or the regional notable decline of native species such as *A. iberus* and Valencia toothcarp (*Valencia hispanica* (Valenciennes, 1846)) with the introduction of the invasive *G. holbrooki* in the Ebro delta plain (Caiola and De Sostoa, 2005); it might be hypothesized that a relationship between the current threatened status and the introduction of allochthonous species might exist. Nevertheless, the influence of many other factors (*e.g.* temperature, salinity...) - and their change in this 30 years lapse of time - that might have affect the native species

population and their dynamics, must not be discarded. In this sense, more studies to corroborate the veracity of this hypothesis are recommended in the area.

In contrast to the presence of the above-mentioned endangered fish species, 6 of the previously cited fish species in the lagoons, did not appear in our samplings: garfish (*Belone belone gracilis* (Linnaeus, 1761)), European flounder (*Plathycthis flesus* Linnaeus, 1758), sand sole (*Pegusa lascaris* (Risso, 1810)), blackhand sole (*Pegusa nasuta* Pallas, 1814), common sole (*Solea solea* (Linnaeus, 1758)) and boxlip mullet (*Oedalechilus labeo* (Cuvier, 1829)) (Table 1). One plausible hypothesis for its currently absence, is that contemporary freshwater management during the rice cultivation period, determines the presence of those marine species which were used to enter seasonally (the first 5) or accidentally (last mentioned species) into the lagoons in the past (De Sostoa, 1983; De Sostoa and De Sostoa, 1985; De Sostoa and Lobón-Cerviá, 1989 and De Sostoa *et al.* 1990). Nevertheless, a regular and more exhaustive monitoring of the fish community in the Ebro Delta coastal lagoons should be implemented in order to confirm their presence/absence in the study area or to unveil threatened statement of fish species before they totally and irreversibly disappear from the natural environment.

In terms of **species richness**, both fish communities (the past and the contemporary) do not differ very much; with just one more fish species found nowadays. Moreover, the species richness found in both periods is in agreement with the mean richness found in Atlanto-Mediterranean coastal lagoons: 23.4 (Pérez-Ruzafa *et al.*, 2007). Thus, a slight increase on the diversity of fish species since last studies has been noticed in the present thesis. Notwithstanding, conservation policies must be undertaken in order to maintain native species and richness.

Summarizing all the above-mentioned, in the last years, while 7 new fish species were found in the lagoons, other 6 were not. Of importance is the fact that 4 of the 7 new found species are allocthonous species. The increase of allocthonous species seemed to be an indicative of the detriment of the general well-being of ecosystem, as it is also confirmed by the status of the native population (*i.e.* 4 native species are recently designed to be endangered species); a fact that is not reflected by the species richness that has been slightly increased. In this sense, it must be taken into account that species richness is not a good tool to analyse changes in fish communities (*i.e.* to describe if a fish community is at worst or better state than it was) as it only takes into consideration the count of individuals in a determinate area (Hill, 1972). For this purpose, previous studies highlighted that a definition of what a natural ecosystem without perturbations it is was needed first, in order to determine then if that ecosystem is deteriorating or not

with time. They coined the term *ecosystem health* - as analogous to humans - to define this state (Costanza *et al.*, 1992; Patil *et al.*, 2001). A healthy ecosystem would be then defined as the one that is productive, have resilience and organization (including biodiversity). Although the concept is not exempt of criticism of authors that defend that the analogy with human health is not accurate (Lancaster, 2000), or that some relationships as the ones among species richness and productivity (Adler *et al.*, 2011), or resilience and diversity (Ives and Carpenter, 2007) are not always true; the term itself is very useful as allows to have a reference status to predict in which direction an ecosystem is changing, by the time that is a very intuitive concept for the society to understand it (Lackey, 2003). In this sense, the aim of this section was to give some hints about recent changes observed in the Ebro Delta coastal lagoons fish fauna, but in order to have a clearer picture and to see exactly in which direction the ecosystem is evolving, more studies taking into consideration the above-explained concept of ecosystem health which considers looking at the ecosystem in a more holistic way, would be recommended.

7.2 Impacts of human activities on the fish assemblages

Despite the controversy regarding whether catches might or not reflect the real abundance of living fish in a particular area (Pauly *et al.*, 2013); total catch landings can proportionate an idea of the relative abundance of the fish, since a proportion between catches and true abundance has been reported (Harley *et al.*, 2001). Since 1965 - first year of registered landings (General introduction) - a general decrease of the catches has taken place in all delta coastal lagoons, and in particular in the three studied lagoons (Figure 1). However, judging the general state of **artisanal fisheries** in the Ebro Delta coastal lagoons without considering fishing effort - an important variable to explain the variability of yields - could lead us to inaccurate conclusions (Bayley 1988). Fishing effort in the studied lagoons, calculated as the ratio between the number of fishermen per lagoon (Table 2) and its area - following the Bayley (1988) approach - has diminished along the whole data series (general decrease of 46% from 1965). Taking into consideration the decline in the catches and the decreasing fishing effort separately, it could be at first glance considered that if fewer fishermen are fishing in the area, less catches are expected. Thus, artisanal fisheries would be “free of guilt” for the decrease of the catch landings. Then, the evolution of the Catch per Unit Effort (CPUE), which takes into consideration the landings regarding the fishing effort employed to catch those

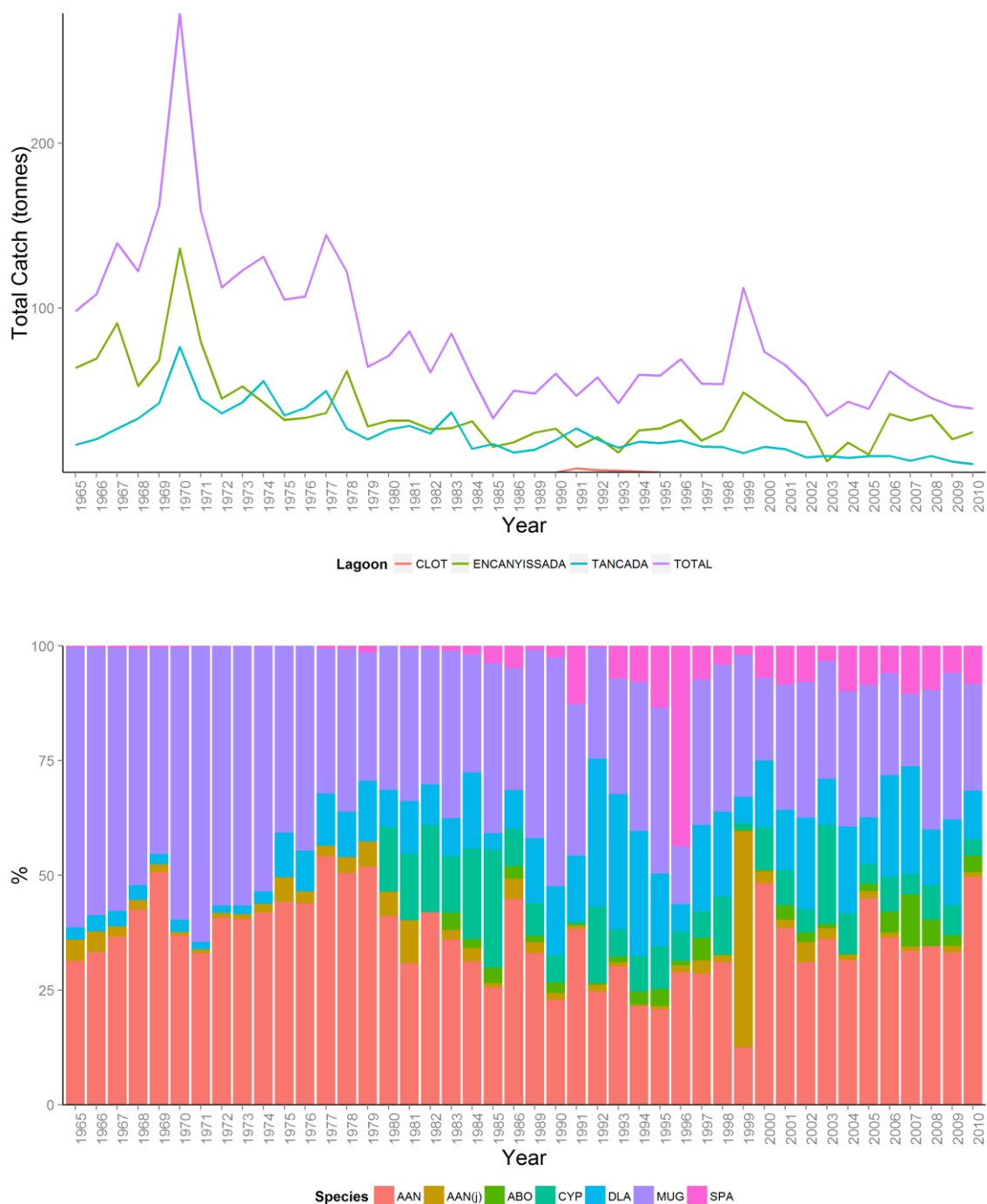


Figure 1. Total and detailed catches (in tonnes) per year for all (total) and each of the studied coastal lagoons in the Ebro Delta plain in the present thesis for the period 1965-2010 (top). Total catches per year (in %) for the main target species in the Ebro Delta coastal lagoons (bottom). Data from St.Pere Fishermen association, no data was available for years 1987 and 1988. Species and family abbreviations: AAN: *A. anguilla*, AAN(j): juveniles *A. anguilla*, ABO: *A. boyeri*, CYP: Cyprinidae (*C. carpio* and *C. auratus*), DLA: *D. labrax*, MUG: Mugilidae (*L. ramada*, *L. saliens*, *L. aurata*, *M. cephalus* and *C. labrosus*), SPA *S. aurata*.

fish, along the years can proportionate a clearer picture to analyze the role developed for the artisanal fisheries in the lagoons. Whereas a clear decrease of the CPUE along the data series was seen for the Tancada lagoon, catches remained more or less constant but with a slightly decrease trend in the Encanyissada lagoon (figures not shown). Assuming catchability to be constant for all the species, a declining trend of the fish stocks in the lagoons is then observed. Furthermore, when looking at the species composition of the catches (Fig. 1), while the whole data series showed a quite steady target species composition throughout the years, being *A. anguilla* and Mugilidae

Table 2. Number of fishermen authorized to fish per lagoon in the different periods.

Lagoon	until 1975	1976-2009	2010-present
Encanyissada	30	13	12
Canal Vell	20	11	10
Olles	10	5	4
Tancada	18	11	10

(from Franch and Quintano, 2004 and own data collection).

family (golden grey mullet (*Liza aurata* (Risso, 1810)), thinlip grey mulley (*Liza ramada* (Risso, 1827)), leaping mullet (*Liza saliens* Risso, 1810), flathead grey mullet (*Mugil cephalus* Linnaeus, 1758) and thicklip grey mullet (*Chelon labrosus* (Risso, 1827)) the most captured species; their relative abundances varied along the years.

Declining trends were found for *A. anguilla* - both adult and juveniles - and Mugilidae family, while increasing trends were found for *A. boyeri*, European sea bass (*Dicentrarchus labrax* (Linnaeus, 1758)) and gilthead seabream (*Sparus aurata* Linnaeus, 1758). No clear trend was found for the cyprinids (*C. auratus* and *C. carpio*). Taking into account all the aforementioned, it seems that the the *A. anguilla* and the Mugilidae family fish stock from the Ebro Delta coastal lagoons are in a decreasing trend. Moreover, the increasing trend for some of the mentioned species, might indicate that in front of the decrease of the main interest fishing species, fishermen are catching other species in bigger proportions and/or that other factors apart from artisanal fisheries, such as the water management might play an important role for explaining the dynamics of the species in the Ebro Delta coastal lagoons (discussed later on).

The cited CPUE decreasing trend, was the first impact detected on the Ebro Delta fish assemblages. However, other causes of concern regarding the way that artisanal fisheries are currently carried out in the lagoons were identified in the present thesis. The second cause of concern was the lack of a specific regulation to set a Minimum Landing Size (MLS; Chapter I) in *A. boyeri* fisheries. As pointed out by Worm *et al.* (2009), any non-regulated fishery will end in a collapse sooner or later. Some examples of worldwide fisheries which have collapsed recently include the case of the Pacific sardine

(*Sardinops sagax* (Jenyns, 1842)) in California and Japan in the late 1940's or the anchovy off Peru and Chile (*Engraulis ringens* Jenyns, 1842) in 1972 (Botsford *et al.*, 1997; Hutchings, 2000). To avoid fisheries collapse, expert fisheries managers agree on the necessity of adjusting captures level to the reproductive capacity of the exploited target species, thus reducing its exploitation rate (Botsford *et al.*, 1997; Worm *et al.*, 2009). Some measures to achieve this goal include gear restrictions, closed areas, reduction of fishing capacity, or reductions in total allowable catch and catch shares (Worm *et al.*, 2009).

Gear restrictions constitute one of the simplest ways to regulate fisheries by restricting the amount and/or type of fishing gear that can be used (Wileman *et al.*, 1996; Stewart, 2008; Stergiou *et al.*, 2009). In order to propose a gear restriction in a determinate fishery, studies of gear selectivity are needed first. Gear selectivity is the quantitative expression of the proportion of fish that is retained in a determinate gear, and thus represents the probability of capture of a certain size of fish in a certain size of mesh. By regulating gear size, an increase of gear selectivity is incentivized and by-catch of non-target species is reduced (Worm *et al.*, 2009). The approach chosen to study gear selectivity in the Ebro Delta coastal lagoons, was to use multimesh nylon gillnets although this fishing gear is not used by the local fishermen (General introduction). This approach was selected for two reasons:

(i) The similarity with trammel nets - the fishing gear used in the coastal lagoons - since the catching principle of gillnets and trammel nets is comparable (Cochrane, 2002), as both are passive gears that caught fish encountering the net during their feeding or migratory movements. The difference lies on the fact that in gillnets only one panel is fishing, whereas in trammel nets there are three panels (Fig. 2).

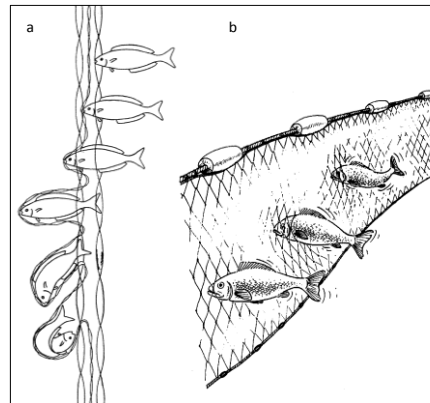


Figure 2. Diagrammatic drawing to show how fish are entrapped in: a) trammel nets (from Clark, 1930) and b) gillnets (from Rosman and Mageri, 1980).

(ii) Gillnets are widely used in research activities, since they are useful to monitor catch-length distribution (Hamley, 1975), to determine a population

size structure, and to assess fishing effects on exploited stock (McAuley *et al.*, 2007; Chapter I).

According to FAO (Cochrane, 2002), an ideal fishing gear has to accomplish the following requisites: (i) be highly selective for the target species and sizes, with negligible direct or indirect impact on non-target species, sizes and habitats, (ii) be effective, giving high catches of target species at lowest possible cost and (iii) be quality orientated, producing catches of high quality. The ideal fishing gear does not exist. However, the achievement of any of the mentioned requisites in a quotidian gear, will be a boost towards the sought of the ideal fishing gear. Moreover, an ideal fishing gear (*i.e.* highly selective, effective and quality oriented on the catches) would presumably lead to a more sustainable fishing activity; reason why promoting the use of highly selective fishing gears has become one of the priorities of the Fisheries Common Policy (EU 1380/2013 11 December 2013).

The selectivity of gillnets was modelled using the Share Each Length's Catch Total (SELECT) method in the Ebro Delta coastal lagoons. The SELECT method is a generalized linear model that assumes a Poisson distribution of the gillnet catch data to fit them to a specified model using maximum-likelihood (Millar, 2000). Selectivity curves are defined as the relative probability of a fish of a given length to be captured when contacting to a mesh of a determinate size (Millar, 2000). The method is based in the Baranov's principle of geometric similarity who stated that "*Since all meshes are geometrically similar and all fish of the same species (within a reasonable size range) are also geometrically similar, the selectivity curves for different mesh sizes must be similar*" (Baranov, 1948; Hamley, 1975). Implemented by Millar (Millar and Holst, 1997; Millar and Fryer, 1999; Millar, 2000), SELECT provides a cohesive approach to size selectivity analysis. It was chosen, due to the fact that it is an indirect method (*i.e.* do not require to know the size distribution of the target population; Chapter I), and it is widely used in gillnet selectivity studies (Reis and Pawson, 1999; Stergiou and Karpouzi, 2003; Carol and García-Berthou, 2007). Five different models were tested: normal location, normal scale, gamma, lognormal and the inverse Gaussian. All five models are unimodal and consist of two parameters describing the location and dispersion of the curves. The differences among them are that whereas the normal location and the normal scale models are based on the normal distribution, the other three are skewed curves with positive asymmetry (Figure 3 and Chapter I).

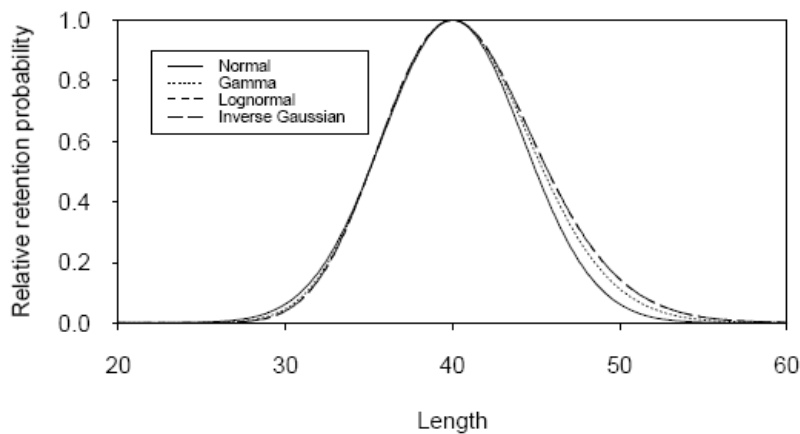


Figure 3. Schematic figure showing the size selection curves modelled by four different models (from Millar and Fryer, 1999).

Selectivity curves have been previously related to the manner the fish is caught based on the classification that Baranov did in 1948. He recognized three ways in which fish can be retained by a gillnet: (a) wedged (*i.e.* being held by a mesh around the body); (b) gilled (*i.e.* having too great a girth to pass through the mesh); or (c) tangled, (*i.e.* held in the net by the teeth, maxillaries or other projections without its body necessarily having penetrated the mesh). Ergo, gillnet selectivity curves would approach normal curves when most fish are wedged or gilled, but when many fishes are entangled or snagged, catch data are skewed to the right and are better fitted by gamma, lognormal, inverse Gaussian models or multimodal mesh models (Hamley, 1975; Hovgård, 1996; Dos Santos *et al.*, 2003; Erzini *et al.*, 2003).

However our outcomes seemed to be more associated to the fish morphology than to the manner the fish were caught. Fish growth can be described as allometric when fish shape growth proportionally to the body size, or isometric when the shape of the fish remains the same irrespective of fish size. On the one hand, our results showed that species with lesser allometry such as *C. carpio*, *P. parva* or *A. alburnus* were best fitted with normal scale model. On the other hand, species with greater allometry, for instance the *C. auratus* (higher body depth - fish length relationship) and mugilids (higher fish length - body girth relationship), or species with gill bony appendages such as *S. lucioperca* and *D. labrax*, were best fitted with skewed models (gamma, lognormal and inverse Gaussian models). Then, our results highlighted that selectivity curves best fit was related to fish morphology (Chapter I).

Indeed both approaches are complimentary, because fish morphology and appendages would influence the particular way in which fish are caught (Reis and Pawson, 1999). In these sense, other studies defended girth as the main factor determining the size of the fish caught by different mesh sizes because girth at the point of capture has to be equal or slightly higher than the mesh perimeter (Stergiou and Karpouzi, 2003; Carol and García-Berthou, 2007). Nevertheless, as fish of the same length can have different girths (Stergiou and Karpouzi, 2003); both morphological parameters must be taken into account in gear size selectivity studies (Hamley, 1975). In this regard, the results obtained here which related the suitability of the SELECT model to fish morphology (that takes into account both length, girth and its relationship), can be useful in determining the way fish might be caught and its selectivity curves best fitted. Selectivity curves and knowledge of the size selectivity can be used to predict the effect of mesh size regulation on fish populations (Psuty-Lipska *et al.* 2006). This knowledge is crucial to management of a fishery for purposes of maximizing yield and protecting juvenile fish, because can prevent of fishing undersized individuals (Gulland, 1983; Wileman *et al.*, 1996; Millar and Holst, 1997). Nevertheless, to enhance fish recruitment and juvenile survival, it has been recommended that gear selectivity and MLS - which set the smallest size at which a particular species can be legally retained - should always be addressed simultaneously (Wileman *et al.* 1996; Suuronen *et al.*, 2007). This approach was followed in this thesis, assessing gear selectivity and MLS simultaneously for *A. boyeri* species, but also taking into account its maturation process (Chapter I). Accordingly, the recommendation for the species would be to promote an increase of the smallest mesh size up to 6.25 mm (nowadays is 5.00 mm). With this proposed mesh size, an increase of the percentage of *A. boyeri* mature individuals caught would be up to 100%, allowing the establishment of a MLS of 54 mm of total length (TL).

Although SELECT method has been widely used to evaluate, propose and modify MLS for a variety of species around the world (Wileman *et al.* 1996; Dos Santos *et al.*, 2003; Gray *et al.*, 2004; Fonseca *et al.*, 2005; Broadhurst *et al.* 2007; Suuronen *et al.*, 2007); as far as we are concern, is the first time that the method is applied to the fisheries of a small fish species, and particularly to *A. boyeri*. Moreover, is also the first time that the approaches followed of modelling gillnet selectivity beforehand, and using this information to establish an MLS in combination with maturity studies is used. Then, this procedure is a novelty and represents a step towards the establishment of a fisheries regulation in the area and a sustainable long-term monitoring.

These results (*i.e.* the proposed MLS for *A. boyeri*) were transferred to the local fishermen. The transference of scientific knowledge to fishermen communities is not

always a straightforward process, as questions such as: *how large a change in mesh size is possible? How easily can the fishers manipulate selective properties of a gear, legally and illegally? Will fishers manipulate their gears?* have to be considered and addressed to guarantee an accomplishment of long-term benefits (Suuronen *et al.*, 2007). In the Ebro Delta case, although we do not know how large a change in mesh size is possible, considering the modest increase of the mesh size (just 1.25 mm), and the fact that the species composition would be pretty analogous (Chapter I), it is foreseen that the proposed change might be feasible in the near future. Nevertheless, it must be considered that artisanal fisheries in the Ebro Delta coastal lagoons are self-managed by the St. Pere Fishermen Association. The concept of self-management or self-governance fisheries is quite new, and in the last years some successful examples around the world - such as United States, Canada, New Zealand or Japan to cite some - have been reported (Townsend *et al.*, 2008). Clues for the success seemed to be based on the small scale of the fisheries, and the flexibility that fishermen have when deciding about determining closing seasons, fixing catch limits or managing research among other range of activities which fishermen are responsible for. Although self-governance has emerged in relatively few of the world's fisheries and it is quite variable among regions, the economical benefits of the activity for the fishermen have been already announced (Uchida *et al.*, 2010). However there are two well-known objections for allowing fishermen to make decisions regarding for instance the settlement of Total allowable catches (TAC) or gear regulations. One is that, compared to government regulations, fishermen will be short-sighted. The other is that they will be too little concerned with the survival of low-value bycatch species (Shotton R., 2000). So, *it is self-management fisheries a good option in reality?* In light of the above mentioned, a middle term where fishermen could manage the resources but considering the voice of the scientific community seems to be a good compromise. Then, in the Ebro Delta case the recommendation proposed would be that the system continues in a regime of self-management control - as it has always been -, but taking into consideration the opinions of scientific community and *vice versa*. In this way, both communities (fishermen and scientific) would be able to express their concerns, investigate possible remedies, and propose final solutions for the encountered problematic. In this sense, in order to figure out the acceptance of the proposed measure, it is recommended to carry out a survey among the local fishermen. Only with their support and understanding that a more sustainable fishing is needed in the delta, the effectiveness of the measures here proposed will be 100% assured.

Single-species management and more specifically gear restrictions, represent the first step to stop a low surveillance fishing situation. Notwithstanding, accounting that fishing mortality is a product of both fishing effort and selectivity, controlling gear selectivity alone is insufficient to manage a stock at a target exploitation rate (Christensen *et al.*, 1996; Wileman *et al.*, 1996). Moreover, fish capture itself has an impact on the aquatic ecosystem that often goes beyond than just removing part of the species population. Then, according to the Code of Conduct for Responsible Fisheries (FAO, 1995), most fisheries cannot be well-managed when only considering the target species; and thus, the management strategy should be developed from a multispecific or ecosystemic point of view. Nonetheless, multispecies management is arduous to implement due to the difficulty of having enough information on biology, ecology and interactions among species (Sainsbury *et al.*, 2000; Campos and Fonseca, 2003).

From the Ebro Delta coastal lagoons, in common with other Mediterranean coastal lagoons such as the Venice lagoon in Italy (Carrer and Opitz, 1999) or the Thau lagoon in France (Palomares *et al.*, 1993), we were able to collect a large series of landing data from the St. Pere Fishermen association, which provided useful information on the Ebro Delta coastal lagoons' target species. Despite, a lack of information regarding the rest of the species (non-commercial fish species, birds, primary producers, etc.) and their trophic interactions in the ecosystem was detected. Therefore, a simplified and more tractable multispecies management approach has been performed (Chapter II) with the ecosystem modelling software system, Ecopath with Ecosim (EwE; Christensen *et al.*, 2008; Coll *et al.*, 2009). The EwE modelling tool is composed of a core mass balance model: Ecopath, which stands for Ecological Pathways Model (Polovina, 1984; Christensen *et al.*, 2008), Ecosim and Ecospace, from which temporal and spatial (respectively) dynamic simulations can be developed (Walters *et al.*, 1997; Christensen and Walters, 2004; Coll *et al.*, 2009). For instance, by simulating possible future scenarios of increase and decrease of the current fishing effort, Ecosim module may allow to evaluate the likely consequences/benefits of different management options in a fishing region at an ecosystemic level, as it has been done in the present thesis.

The approach followed in this thesis was to model the Tancada lagoon. Among the three studied lagoons, the election of the last was due to: (i) it is representative of the whole variability present in the three studied coastal lagoons in terms of size, morphology and sea connections, (ii) the applicability of the obtained results to the rest of the lagoons and (iii) it was easier to model than the system formed by the Encanyissada and Clot lagoons. Due to the connection existing between the Encanyissada and the Clot lagoons (separated by a short canal, General introduction), in

terms of modelling they should be considered as a unique unit, fact that would turn the modelling task more complicated.

Ecopath models are very useful to collect ecological and biological information of a determinate ecosystem and set it in a coherent framework. In this sense, Ecopath models are especially effective to disentangle the functioning of systems with lack of information regarding non-target fish species in the Mediterranean area such as the aforementioned Venice and Thau lagoons (Carrer and Opitz, 1999 and Palomares *et al.*, 1993, respectively). Accordingly, the Ecopath model developed for the Tancada lagoon in this thesis, allowed us to understand the ecological structure and functioning, as well as to analyze the trophic interactions of the main functional groups. The outputs of the model highlighted that zoobenthos, sand smelt and sand goby were the groups presenting higher predation pressures in the lagoon. Furthermore, zoobenthos stood out to be a very important group in the lagoons; fact that was in accordance with other models developed in the Mediterranean area that highlighted the important role developed by this group for the whole ecosystem (Brando *et al.*, 2004; Carrer and Opitz, 1999; Pinnegar and Polunin, 2004).

Two different scenarios (one predicting an increase and the other a decrease of the current fishing effort by 50%) were simulated because of their plausibility to occur in the near future. With an increase of the current fishing effort in the lagoons, the model predicted biomass decreases for all the target species except for the glass eel. Indirectly, increases of the non-target species biomass were also predicted. Conflicts between cormorants and human fishing activity (which compete for the fish resource) were additionally notable with this simulation, being in agreement with previous studies developed in the Mediterranean area (Pinnegar and Polunin, 2004; Brando *et al.*, 2004). Contrary, the second simulated scenario (*i.e.* 50% decrease of the current fishing effort), caused direct increases in the target species biomass but led to a substantial decrease in the non-target species. Notwithstanding, the biomass loss of the non-target species was of minor dimensions (1.22% in average). Furthermore, results from both simulations pointed out that the total ecosystem biomass was higher with the decreasing simulation effort. This means that if the general trend of decreasing fishing effort observed during the whole data series of catch landings (Figure 1 and Table 2) is maintained, an increase of the total biomass of the ecosystem is predicted in ten years. Moreover, an increase of the biomass of all the target species - except the glass eel - it is also expected. Then, a gradual decrease of the fishing effort in the Tancada lagoon would be an efficient measure for the recovery of the main species on the lagoon.

The eel case (glass and adult eels), is of concern and deserved more attention because of its worldwide decline (FAO and ICES, 2008; IUCN, 2013). With aware of its critical situation, the European Commission has created a Council Regulation (EC 1100/2007) which constrains all member states with European eel native populations (which includes Spain) to create management plans for its global recovery. The aim is to reduce the mortality related to anthropogenic activities, in order to reach the escapement to the sea of at least 40% of the pristine European eel population with a high probability and in a long-term. The management plan in Spain is based on a 1 national Eel Management Plan (EMP) and 12 regional EMPs -11 autonomic EMP and 1 EMP specific for the Ebro River Basin- (MARM, 2010). The Ebro River Basin EMP included main management actions to be developed specifically in the Catalan part of the Ebro Basin (summarized in Table 3) and in the whole Ebro River Basin.

Table 3. EMP for the Catalan part of the Ebro River Basin.

Eel size	Professional fishing	Recreational fishing	Restocking measures *
< 12 cm	<ul style="list-style-type: none"> Limited sites (346 points) Limited period (4 months and 20 days/year: 1 nov to 20 march) 15 hours/day (during the night) Personal licenses Fishing gears: "bussó" 	<ul style="list-style-type: none"> Forbidden by specific legislation 	<ul style="list-style-type: none"> It will be carried out in those river basins where glass eels are captured, to avoid the possible transmission of parasites and pests between the different basins. Restocking sites selection has been done based on studies on the current eel situation.
> 12 cm - < 35 cm	<ul style="list-style-type: none"> Banned 		
> 35 cm (MLS)	<ul style="list-style-type: none"> * <i>In the bays</i>: banned * <i>In the lagoons (Tancada, Olles, Encanyissada and Canal Vell)</i>: <ul style="list-style-type: none"> As a bycatch Limited period (5 months/year) Fishing gear: fyke nets or "pantena" 	<ul style="list-style-type: none"> Catch-release fishing Effort to reduce unregulated or illegal recreational eel fishing (IUU fishing) 	<ul style="list-style-type: none"> Restocking will be carried out with individuals of different sizes to ensure a greater survival of the individuals and to compensate the sex-ratio. It will be done according to the percentages specified in the Regulation.

* Restocking measures (5% of the catches) are carried in the area since 1996 for conservation purposes. (adapted from Ebro Basin EMP, 2010).

In the whole Ebro River Basin the measures were focused on the habitat quality improvement (*i.e.* river connectivity, water quality, fight against predators, protected sites, etc.) and control and surveillance measures (*i.e.* by developing an inter-regional Ebro River basin evaluation plan). The plan is now in the first phase (2010-2015), where coastal communities with available data might implement their proposed measures based on the estimates of the pristine and current situation of the European eel in Spain. At the end of this first phase (2016), the newly available data will allow a re-assessment of the stock situation and to launch a second phase (2016-2050), where a management

plan for the population will be applied (MARM, 2010). During the development of this first phase and specifically in the Ebro River Basin, some difficulties while estimating the real biomass of the eel population were found (MARM and Generalitat de Catalunya, 2009). Also during this first phase, measures at European level are towards the reduction and control of the glass eel fishing effort. In this regard, the model developed in the current thesis is of importance because it contributes to the knowledge of the eel population biomass in one coastal lagoon from the Ebro Delta, and could be used to decipher the biomass in the rest of the lagoons. Thus it is foreseen, that the results presented in this thesis, could be useful for the proposal of a recovery management plan in the Ebro River Basin and would contribute to the recovery of the European eel population. Moreover, if successful, this methodology could be applied in similar basins to solve problems of biomass estimation. Unexpectedly, our model predicted an increment of the glass eel population with the increasing fishing simulation. This apparent paradox, could be explained by the fact that approximately 1/3 of captured glass eel is not reported in the landings statistics because it is sold before going to the fishing market and also as a result of indirect artisanal fisheries effect, which are removing its main predator in the lagoons. A question regarding this last hypothesis arises: *Is it this last fact indicating some kind of density-dependent regulatory process in the *A. anguilla* population as some previous studies have postulated (De Leo and Gatto, 1996; Lobón-Cerviá and Iglesias, 2008; Bevacqua et al., 2011)?* More studies are needed in the area to investigate if this hypothesis can be based on facts.

The model construction, helped us to elucidate one of the hypotheses about the artisanal fisheries in the area: local fishermen take advantage of the migratory species such as the European eel with the “pantena” and “bussó” fishing gears. As described in detail in the General introduction, “pantena” is a special kind of trap with a funnel shape that is permanently settled in the canals that communicate the lagoons with the sea. Although the “pantena” structure is permanently settled, the nets are only set during the fishing season. This type of fishing gear is based on the fact that migratory fish enters the lagoons in spring as juveniles, looking for a safe place to grow, and are trapped when they attempt to return to the sea as adults in the autumn. “Bussons”, trapezoidal structures specially designed to capture glass eels, also take advantage of the migratory movements. The principle of glass eel capture is the migration of glass eels from marine and brackish waters to freshwater environments (Gisbert and López, 2008), then and contrary to the “pantena”, this fishing trap captures the individuals (*i.e.* glass eels) in their route to the lagoons and the river. The adoption of the fishing methods to the ecology and behaviour of target species has been also found in Greece, France, Tunisia and in

other parts of Spain, as the Mar Menor lagoon, in Murcia (De Sostoa and De Sostoa., 1985; Yañez-Arancibia *et al.*, 1994; Koutrakis *et al.*, 2005; Katselis *et al.*, 2010; Pérez-Ruzafa and Marcos, 2012). This ecological knowledge of the system that fishermen from the area retain has been named as Fishers' Ecological Knowledge (FEK). Recently some studies have recognized the importance of incorporating FEK in fisheries management plans, in order to manage the resources in a more sustainable way (Freire and García-Allut, 1999, 2000). In order to acquire traditional knowledge, methodologies and research tools must include: a) analysis of discourse, b) selection of information, c) semi-guided open interviews, d) surveys on specific points of knowledge, e) analysis of the distribution maps of the resources and habitats drawn up by the fishers, and (f) use of different documents of a functional nature that fishers may have (notebooks, graph interpretations (*i.e.* depth sounder, radar), notes, etc.) (Freire and García-Allut, 1999; 2000). For instance, when data are aggregated, for local and regional fisheries where little scientific data exists, personal interviews with a career-history format can generate a baseline of information for a particular fisher (Neis *et al.*, 1999). Moreover, since the usefulness and validity of the data that may be get from the interviewers are based on the mutual trust among fishers and biologists, performing the interviews in person is more appropriate rather than by email or phone.

Despite the fact that fishermen knowledge can be certainly valuable in the sustainable management of the natural resources, management measures often fail due to limited scientific knowledge (Suman *et al.*, 2005). In this sense, by defining the precautionary approach for fisheries, FAO (1995) stated that absence of adequate scientific information should not be used as a reason for postponing or failing to take conservation and management measures. For that reason and based on the obtained results, the recommendation in reference to artisanal fisheries, should be to follow the precautionary principle, and thus not to increase, or if possible, decrease the current fishing effort carried out at the lagoons, which the model predicted to be more beneficial in terms of ecosystem biomass win. Reducing fishing effort in the rebuilding of the stocks has been reported to have positive effects for the whole ecosystem and to allow a more sustainable exploitation of the fisheries resources (Bostford *et al.*, 1997; Worm *et al.*, 2009). Accordingly, the reduction of the current fishing effort in the Ebro coastal lagoons would be a step towards the development of a more sustainable fishing activity.

Taken all together, artisanal fisheries in the Ebro Delta coastal lagoons seemed to have impacted the lagoons' fish fauna in various forms (schematized in Figure 4): (i) at biological level, by decreasing the target fish abundance, through the capturing of immature individuals for example (*A. boyeri* case); and (ii) at economical level, by

reducing the quantity of marketable fish derived from the changes above mentioned at biological level. An eventual overexploitation of the fisheries resources or changes of fish size population caused for example by capturing undersized individuals (*i.e.* immatures), could led to the loss of marketable species, or to a diminution of their commercial value due to the fact of not attaining the minimum commercial size. Some recommendations for the development of a more sustainable fishing activity in the lagoons, supported by our results, would be: (i) promote an increase of the smallest mesh size from 5.00 mm to a minimum of 6.25 mm, (ii) a precautionary principle of not to increase, or if possible, decrease the current fishing effort carried out at the lagoons and (iii) a yearly scientific follow up of the fishing captures/fish stocks in order to see the adequateness of the proposed measures. As none previous studies analyzing the impact of the fisheries and further, considering the coastal lagoons as a whole ecosystem, were carried out in the area; the results presented in the current thesis are of novelty and will bring useful data for the sustainability of the fisheries from a commercial point of view and for the conservation of such priceless ecosystem from a social and scientific point of view.

Although the model construction has enlightened the structure, the functioning, the main trophic relationships and the effects of future fishing scenarios in the Tancada lagoon; a few uncertainties regarding the environment effects and its interactions with the ecosystem still exists. Moreover, the dynamics of some species in the lagoons seemed to depend on other factors apart from the artisanal fisheries (Chapter II). In this regard, the effect of the other main anthropogenic pressure on the fish assemblages from the Ebro Delta coastal lagoons, the **water management**, was studied considering the small-bodied fraction of the fish community. The approach of sampling small-bodied fish species, comprising resident fish species (*i.e.* complete the reproductive cycle inside the lagoon) and juveniles of migratory species (*i.e.* spawn at the sea), was chosen because the capacity of movement of larger species and the adults of migratory species do not make them reliable as descriptors of spatial distribution (Mariani, 2001). Moreover, by analyzing only small-bodied fishes, target species are not taken into account and, therefore, the effect of fisheries on the fish community could be isolated. The case of *A. boyeri* is different, as is the only resident species being fished in the area. However, in our studied lagoons, the fishing activity is restricted to four months a year (October-March; Chapter III and General introduction), remaining the rest of the year with no fishing activity. If we take in consideration that the reproductive period of the species *A. boyeri* is comprised between the months of April-July (Andreu-Soler, 2006), the fishing activity developed in the lagoons does not affect the reproductive capacity of the species, and moreover is acting similar to a fishing closure period.

Water management in the Ebro Delta is carried out to allow rice cultivation in the area. The pristine delta (*i.e.* prior to the named “rice revolution”; General introduction), was very different from the one found nowadays. Characterized by rugged hills, large dunes and beaches, numerous saline or hypersaline lagoons and extreme contrasts between dry and wet (*i.e.* marshes and ponds) areas; the original delta was composed by flora (*i.e.* saltwort, absence of trees) and fauna typically adapted to coastal marine environments (Espanya, 1997; Fabregat, 2009). The “rice revolution” involved the construction of an extensive canals network to bring freshwater to the rice paddies, which completely altered this original landscape. As previous studies reported in the area, this transformation caused an artificialization of the natural hydrological cycle (De Sostoa and De Sostoa., 1985; Comin *et al.*, 1987; Forés *et al.*, 2002; Curcó, 2006). In turn, the indicated artificialization had many severe consequences in the whole deltaic plain at economical (construction and maintenance of the canals, rescue of trapped fish), physical (inversion of the coastal lagoons’ hydrological cycle, habitat alteration) and biological level (change of fish migratory patterns, alteration of fish composition) (Fig. 4).

Among the physical consequences that canals infrastructure establishment has caused, the most notable one has been undoubtedly the inversion of the coastal lagoons’ hydrological cycle (Chapter III), with the subsequent loss of water quality and habitat alteration (Fig. 4). As previous studies reported in the area, with the inversion of the hydrological cycle, the lagoons nowadays register freshwater summers and saltier winters (De Sostoa and De Sostoa., 1985; Comin *et al.*, 1987; Forés *et al.*, 2002; Curcó, 2006). This artificialization of the system had in turn consequences at biological level (discussed below). Unfortunately, the problem of water deterioration is also common in coastal lagoons affected by agriculture activities, because of the high amount of nutrients, pesticides and herbicides that water drained from the paddies contains (Cloern 2001; Lucena *et al.*, 2002; Curcó, 2006; Badosa *et al.*, 2007). Concretely, the excessive amount of nutrients drained into the Ebro Delta lagoons from the rice fields, caused episodes of eutrophication in the past that led to: a decrease in biological diversity, reduced submerged macrophytes production and to a lower fish and waterfowl populations (Comín *et al.*, 1989; Forés *et al.* 2002). Two principal measures to alleviate this eutrophic situation in the lagoons were taken in the last decades. First, by constructing circumvallation canals that avoided major part of the waste water pass from the rice fields to the sea through the lagoons (Menéndez *et al.*, 1995; Forés *et al.*, 2002; General introduction); and second by improving the river water quality (Nebra *et al.*, 2011, Ibáñez *et al.*, 2012a,b). As a result, nowadays water drained into the coastal lagoons is of higher quality (*i.e.* it is a mixture of water from the river and the rice fields;

from the irrigation and drainage canals, respectively), but still contain high amount of nutrients (Chapter III). Moreover, this improvement of the lagoon's water quality was in detriment of the Alfacs bay quality that nowadays receives the major part of the waste water from the rice fields. This fact has converted the bay in a very productive system due to the high organic matter loadings coming from the rice fields, but also into a system that occasionally suffers from anoxia, especially in summer with the elevated temperatures, the lower water circulation (*i.e.* freshwater inputs are scarce) and the increase of the evaporation rates that favours processes of organic matter degradation (Ibáñez *et al.*, 1999; De Pedro, 2007). This is of importance, not only because it is altering the entire ecosystem, but also considering the important economical role that the bay plays by supporting prosperous aquaculture activities (mainly of mussels, clams and oysters) and establishing one of the richest fishing areas in the Western Mediterranean Sea (Navarro *et al.*, 2009).

Accordingly, although some improvements have been accomplished in the last decades, the amount of nutrients that are still being drained to the lagoons and bays during the rice cultivation period can cause eutrophication of those systems. The solution passes for the reduction of the amount of nutrients entering to the system, by reducing the volume of fertilizers being used, using fertilizers without phosphates or by using organic load of waste water from farms as fertilizers (Ibáñez *et al.*, 1999). In order to be effective and due to the non-point origin of the eutrophication focus, these measures must be applied at basin level (*i.e.* Ebro River basin), considering the Ebro and its delta as a whole (PIPDE, 2006). In this regard, it is important to remind the endangered status of the Ebro Delta (General introduction). Threatened by the loss of sediments and the consequent decreasing accretion rates, the deltaic plain is suffering from a subsidence process (Prat and Ibáñez, 1995; Ibáñez *et al.*, 1999). Moreover, the expected increase of the temperatures (*i.e.* evaporation rates) and the predicted global sea-level rise of about 40 cm by the year 2100 (IPCC, General introduction), under the scenario of global climate change, will do nothing but aggravate this critical situation. One of the consequences of the subsidence process in the ecosystem is the loss of water quality and the promotion of salt wedge that can trigger the salinization of fields under cultivation, being in this manner no longer cultivable (Prat and Ibáñez, 1995; Ibáñez *et al.*, 1997). Considering all the above mentioned, "green filters", a concept used to describe the use of some of the non-cultivable wetlands as waste water treatment systems for non-point source pollution such as fertilizers, seems to be a plausible solution in the Ebro Delta case (Ibáñez *et al.*, 1999; Comín *et al.*, 2001). The proposal of using green filters is two-folded: (i) revaluation of less productive fields due to an

increase of soil salinity, that otherwise would be abandoned and (ii) improvement of water quality by using the fields as biological filters. Efficiencies up to 50-95% of the total nitrogen and to approximately 50% of phosphorous retention have been reported in some experimental fields in the Ebro Delta (Comín *et al.*, 2001), highlighting its appropriateness to be used as filters for agriculture runoff in the area. Then, the implementation of green filters to act as biological filters, would let to a long-term and more sustainable water management in the area, and thus it is recommended to guarantee the future of valuable ecosystems such as the coastal lagoons and the bay.

Available data from which information regarding fish composition and abundance could be extracted - catch landings (first year registration 1965) and preceding studies (De Sostoa (1983); De Sostoa and De Sostoa (1985); De Sostoa and Lobón-Cerviá (1989) and De Sostoa *et al.* (1990) -, are posterior to the “rice revolution” period (1900-1960), and thus to the inversion of the coastal lagoons’ hydrological cycle. Artisanal fisheries in the Ebro Delta coastal lagoons are performed since the XIIth century, but we do not have information about the catches. The only reference about that time is that in 1340 some registers indicated the existence of the ancestral “Pantena” fishing gear, which allows us to think that founder fishermen were already taking advantage of the fish migratory movements (Fabregat, 2009). Then, although the composition and abundance of the pristine community (the one existing before the “rice revolution”) in the lagoons is unknown; considering that the deltaic plain previous to the agricultural transformation was characterized by rugged hills, large dunes and beaches, numerous saline or hypersaline lagoons; fish communities adapted to coastal marine environments might be the most plausible to exist at that moment (Espanya, 1997; Fabregat, 2009). In whatever circumstances, when compared with coastal lagoons without hydrological alteration (*i.e.* climatic-dependent hydrology; Pérez-Ruzafa *et al.*, 2006, Maci and Basset, 2009; Verdiell-Cubedo, 2009), our results confirmed that at biological level the hydrological alteration has subsequently affected the dynamics and composition of the ichthyofauna. In this sense, outcomes suggested that the hydrological alteration might be responsible for the current fish community in the Ebro Delta coastal lagoons, composed by brackish and marine families (Gobiidae, Mugilidae, Atherinidae) and freshwater species (belonging to Cyprinidae and Poeciliidae families). Those results are in concordance with other studies on coastal ecosystems suffering from hydrological alterations (Flower 2001, Gordo and Cabral 2001, Poizat *et al.* 2004, Pérez-Ruzafa *et al.* 2005, Badosa *et al.* 2007). Hereof, the present study represents the first research approach describing the impact of the hydrological alteration on the fish assemblages from the Ebro Delta lagoons.

In addition, the hydrological alteration promoted peaks of species abundances in unusual periods and was responsible for structuring the fish community along a salinity pattern. Then, higher abundances of brackish and marine species during winter-spring - with the highest salinity conditions were found -; whereas freshwater species abundance where higher in summer-autumn, coinciding with the freshwater inputs. The alternation of a freshwater input period with a non-input period creates a wide salinity range in the lagoons that fish have to cope with. Thus, depending on their osmoregulatory capacity (Rigal *et al.*, 2008), fish were distributed in the lagoons accordingly to their salinity tolerances and preferences. In summary, the results obtained in the present thesis highlighted preferred salinity ranges of 20-25 for marine species (*L. aurata*, *L. saliens* and *C. labrosus*); ranges of 10-20 for brackish species (*A. boyeri*, *P. microps* and *A. iberus*); and salinities below 5 for the presence of allocthonous freshwater species (*G. holbrooki*). Then, salinities higher than 15-20 throughout the year seem to assure the dominance of native species in the coastal lagoons, and thus are recommended.

The presence of allocthonous freshwater species seemed to be perjudicial for some native species. Although the introduction of an organism into a new environment always provides risks and surprises as to the impact it will have on other organisms, it is particularly disconcerting when organisms that are introduced to control the activities of an unwanted invader instead do collateral damage to other species, even driving them to extinction (Mooney, 2001). This is apparently the case of the freshwater invasive species *G. holbrooki* in the Ebro Delta. First introduced in Spain in 1921 to fight against malaria (De Buen, 1935), its spread in the Ebro Delta was easy due to the continuous freshwater inputs - that allowed its survival - and the lack of predators in the basin - which allowed its expansion -. Earlier studies in the region have reported the presence of this species in the lagoons (Table 1), as well as its negative impact to native species in the area such as the endangered *A. iberus* (Caiola and De Sostoa, 2005). The inverse abundance trends found between the endangered (*A. iberus*) and the invasive (*G. holbrooki*) in the present thesis, evidences not only two opposite salinity preferences, but also suggested that the native species is still being threatened by the invasive. Moreover our observed higher presence of the endangered species *A. iberus* in more saline habitats seems to be a response to avoid competition with the invasive *G. holbrooki* (Doadrio, 2001; Rincón *et al.*, 2002; Caiola and De Sostoa, 2005). The current water management scheme developed in the Ebro Delta Natural Park seems to favor the introduced species in detriment of the native ones. In this sense measures to act against this invasive species and to conserve the threatened native populations such the above mentioned

instauration of minimum salinities higher than 15-20 throughout the year are strongly recommended in the area.

The allochthonous status of the freshwater species in the Ebro Delta coastal lagoons altered at the same time the introduced/native species ratio. Moreover, the introduced/native species ratio seemed to be correlated with the management carried out in each lagoon. From the three studied lagoons, Clot is the one receiving the largest freshwater supplies, and Tancada the one receiving the lower freshwater inputs during the season of cultivation. The higher proportion of native species found in the Tancada lagoon even in the period of more freshwater inputs (*i.e.* September), reflects a lower perturbation of the fish community by the water management. Based on all those results, the recommendation in reference to the water management to promote fish communities with native species dominance - important both for commercial and conservation purposes -, would be to: (i) follow the policies carried out currently in the Tancada lagoon, characterized by less freshwater inputs during the rice cultivation period (May-December) and (ii) set minimum salinities higher than 15-20 throughout all the year. In order to facilitate this last task, a network of “smart buoys” - autonomous buoys that observe and record meteorological and water quality data (including air temperature, relative humidity, barometric pressure, water temperature, salinity and dissolved oxygen among others) in real-time - could be settled in the different Ebro Delta coastal lagoons. Several countries are monitoring their oceans, coastal waters and bays using these systems with satisfactory results (Mills *et al.*, 2003; Mills *et al.*, 2005; Wilson, 2012). Then, it is expected that would be a very useful system to establish in the Ebro Delta coastal lagoons. Likewise, a fish community dominated by native species, would beneficiate not only the conservation of the natural environment, but also the fishermen community and the general Ebro Delta’s population, since autochthonous fish species (such as *D. labrax*, *S. aurata*, *A. boyeri*, *A. anguilla*) reach highest values in the market than freshwater alien species (*C. carpio*, *C. auratus*).

In the deltaic plain, water management and artisanal fisheries are carried out alternatively along the year (*i.e.* higher salinity period in the lagoons coincides with the professional artisanal fishery season). Then, fish populations might be impacted not only by one or the other main developed activity in the area, but also by the combination of the two. Moreover, the two activities seemed to affect differently depending on the species. For example, the brackish species *A. boyeri* which high abundances are expected during the period of higher salinity, because of its salinity preferences (Leonardos, 2001); was offset by the fisheries effect (Chapter I and II). Whereas for the

freshwater *C. carpio*, the fluctuation of the population was not only dependent on the fishing pressure exerted by the artisanal fisheries in the area, but also by the water management who dictates the periods of freshwater dominance favourable for the survival of the species (Chapter II and III). Those two examples illustrate the difficulty of analysing these two human impacts independently, and emphasized the importance of investigate them from a wider point of view. Moreover, and due to the different approaches used to investigate the impact of two of the main human activities - each one of them designed as the most suitable to analyze a determinate pressure - in this thesis; it is challenging to decide which one of the two is affecting the most the fish assemblages from the Ebro Delta coastal lagoons, if any.

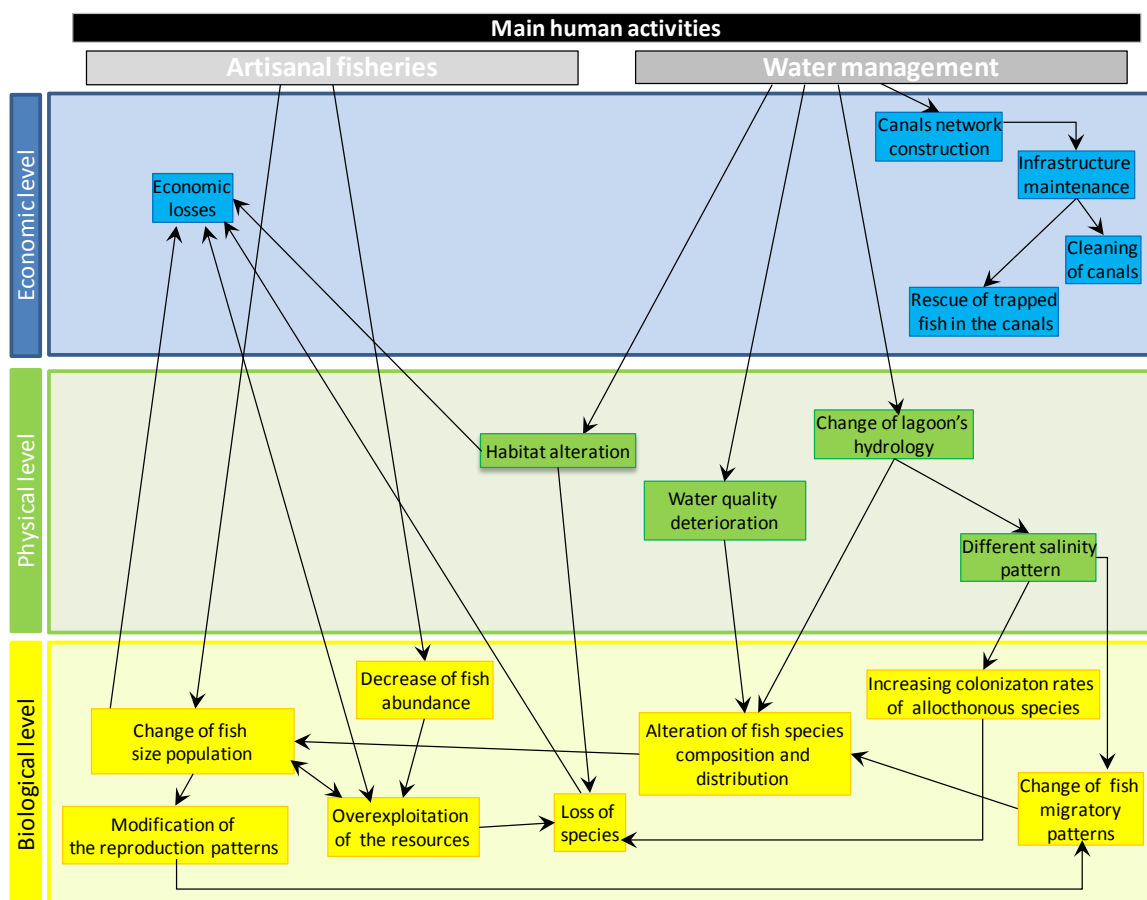


Figure 4. Resume of the main effects, their relationships and consequences of the two studied impacts in the coastal lagoons ecosystem.

The results obtained in this thesis have elucidated the current state-of-art of the coastal ecosystems from the Ebro Delta plain and characterize the impacts on this ecosystem of two of the main human activities carried out on it: **artisanal fisheries** and **water management**; highlighting simultaneously the different levels at which

consequences could be noticed: economic, physical and biological (schematized in Fig. 4). Overall, the presented work, allowed to describe the manner that both impacts altered the **ichthyofauna** in the coastal lagoons and provided enough awareness for the proposal of three key mitigation measures as the: (i) increase of the minimum mesh size used up to 6.25 mm, (ii) reduction of the artisanal fishery fishing effort and (iii) settlement of minimum salinities higher than 15-20 throughout all the year; to go towards a more **sustainable management model** in the Ebro Delta coastal lagoons. In this sense, the achievement of a long-term sustainable management model of the **Ebro Delta coastal lagoons** needs regular monitoring to guarantee the effectiveness of the proposed measures.

CONCLUSIONS

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1. The current fish composition in the Ebro Delta coastal lagoons differ considerably from the one found in the eighties.
2. The suitability of the SELECT model was related to fish morphology. Fish with less allometric growth (*i.e.* more isometric) were best fitted with normal scale models, whereas fish with more allometric growth or with body appendages were best fitted with skewed models (gamma, lognormal and inverse Gaussian).
3. The fisheries management for the endangered and vulnerable species *Atherina boyeri* seems to be inadequate. The approach of combining gear selectivity and maturity studies was useful for the establishment of better management proposals for this species.
4. The analysis of gear selectivity suggested that the current mesh size used to fish *Atherina boyeri* is not the most adequate. The smallest mesh size (5.00 mm) should be increased to at least 6.25 mm to allow the population to reach a Minimum Landing Size of 54 mm TL (L_{75}).
5. The construction of the Ecopath with Ecosim (EwE) model allowed to identify the structure and the main trophic interactions for the 18 functional groups defined in the Tancada lagoon. Zoobenthos, sand smelt and sand goby were the groups presenting higher predation pressure in the coastal lagoons.
6. Model results confirmed the hypothesis that artisanal fisheries in the area take advantage of the migratory movements of some species in the lagoon as European seabass (*Dicentrarchus labrax*), adult eel (*Anguilla anguilla*) and gilthead seabream (*Sparus aurata*).

7. Ecosim simulations suggested the gradual reduction of the current fishing effort in the Tancada lagoon as the best strategy to ensure both fishery sustainability and conservation of the Ebro Delta coastal lagoons.
8. Measures implemented to improve water quality in the lagoons were accomplished in the nineties. However, freshwater inputs are still a source of concern since higher concentrations of nutrients in the lagoons are associated with freshwater inputs coming from rice fields.
9. The altered hydrological pattern in the Ebro Delta coastal lagoons- characterized by freshwater inputs during the rice cultivation period- has a big impact on the ichthyofauna at different levels: increased the presence of freshwater species (mostly introduced species), modified the structuring of fish populations along a salinity pattern and altered the fish community composition.
10. The current water management scheme in the coastal lagoons negatively affected native species and favoured the introduced ones. Special consideration must be taken with some native species which conservation status is threatened such as *Aphanius iberus* and *Anguilla anguilla* or vulnerable in Spain such as *Salaria fluviatilis* and *Atherina boyeri*.
11. Different water management policies applied to the three coastal lagoons, ended up to have different levels of hydrological perturbation.
12. The best policy management for the fish assemblages was the one done in the Tancada lagoon, characterized by less freshwater inputs during the rice cultivation period that allowed salinities higher than 15-20 throughout all the year. The adoption of similar management policy would be recommendable for the Encanyissada and Clot lagoons in order to maintain adequate levels of salinity and sustain their native species important both for commercial and conservation purposes.

FUTURE PERSPECTIVES

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Taken into account the principal findings and main conclusions of the present PhD thesis, and although it is expected that this thesis will contribute to increase the knowledge regarding the main impacts of human activities on the fish assemblages from the Ebro Delta coastal lagoons, further research work remains to be done. In this sense, future research and monitoring needs and perspectives are:

- Regular monitoring of ichthyofauna from the Ebro Delta coastal lagoons, in order to check the state of the ecosystem and the efficiency of the applied measures, if any.
- Increase surveillance and research to identify potential new impacts and develop corrective measures that restore and/or protect natural environmental conditions.
- Study the viability of incorporating Fishers Ecological Knowledge (FEK) in future fisheries management plans.
- Carry out more studies in the area focusing on the fish migratory movements.
- Carry out studies on eel's recruitment. *Is this process density-dependent? Can the Ebro Delta populations give some clues to decipher it?*
- Increase and promote the communication and interactions between scientists, resource managers, and policy makers to ensure consensus decisions regarding estuaries and coastal lagoons, by for example organizing national and international meetings and conferences that would allow the exchange of opinions, points of view and future plans or research needs.
- Develop educational and outreach programs to inform students and general public about the importance of maintaining healthy and sustainable estuarine and coastal environments.

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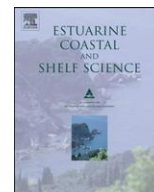
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APPENDIX

(original publications)

11



Gillnet selectivity in the Ebro Delta coastal lagoons and its implication for the fishery management of the sand smelt, *Atherina boyeri* (Actinopterygii: Atherinidae)

Sílvia Rodríguez-Climent^a, Carles Alcaraz^a, Nuno Caiola^{a,*}, Carles Ibáñez^a, Alfonso Nebra^a, Gloria Muñoz-Camarillo^a, Frederic Casals^b, Dolors Vinyoles^c, Adolfo de Sostoa^c

^aIRTA Aquatic Ecosystems, Ctra. de Poble Nou, Km 5.5, E-43540 Sant Carles de la Ràpita, Catalonia, Spain

^bUniversitat de Lleida, Catalonia, Spain

^cUniversitat de Barcelona, Catalonia, Spain

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ABSTRACT

Multimesh nylon gillnets were set in three Ebro Delta (North-East of Spain) lagoons to determine mesh selectivity for the inhabiting fish community. Each gillnet consisted on a series of twelve panels of different mesh size (ranging from 5.0 to 55.0 mm bar length) randomly distributed. The SELECT method (Share Each Length's Catch Total) was used to estimate retention curves through five models: normal location, normal scale, gamma, lognormal and inverse Gaussian. Each model was fitted twice, under the assumptions of equal and proportional to mesh size fishing effort, but no differences were found between approaches. A possible situation of overfishing in the lagoons, where artisanal fisheries are carried out with a low surveillance effort, was assessed using a vulnerable species inhabiting these brackish waters as case study: the sand smelt, *Atherina boyeri*. The minimum size for its fishery has not been established, thus remaining under an uncontrolled exploitation situation. Therefore, a Minimum Landing Size (MLS) is proposed based on sexual maturity data. The importance of establishing an adequate MLS and regulate mesh sizes in order to respect natural maturation length is discussed, as well as, the proposal of other measures to improve *A. boyeri* fishery management.

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1. Introduction

Fisheries have altered and degraded marine ecosystems through direct (e.g. overfishing, bycatch and discards) and indirect (e.g. community changes, mortality caused by lost gears, etc.) effects, especially in coastal regions where fishing and other anthropogenic perturbations are most intense (Botsford et al., 1997; Goñi, 1998). Nowadays, the main threat of fisheries is overfishing, with numerous examples of stocks abundance reduction until population collapse (Botsford et al., 1997; Hutchings, 2000). Of major concern is the fact that in industrial fisheries, the 10 most fished marine species that account for the 30% of the world production are overexploited (FAO, 2009). Artisanal, small-scale fisheries, although are less productive, contribute to more than of the 25% of total world catch (Mathew, 2003). Because fisheries are size selective, removing preferentially larger fish, overfishing modify

size distribution pattern (i.e. reducing average size, size-at-age and size-at-maturity), age structure and genetic diversity of targeted species (Botsford et al., 1997; Goñi, 1998).

One of the main problems related to fisheries management, relies on the fact that large levels of natural variability frequently mask the effects of overexploitation. In such cases, initial over-exploitation is not detectable until it is severe and often irreversible (Ludwig et al., 1993). Therefore, it is necessary the establishment of long term monitoring, including studies on community and populations to ensure a sustainable management. Fisheries management measures to avoid overfishing include closures (temporary or seasonally) reducing fishing effort and limiting the harvest (Watson et al., 1993), catch number limitation to a more sustainable level (Frid et al., 2003), and establish Minimum Landing Sizes (hereafter MLS) to allow fish to spawn at least once (Jennings et al., 1998; Stewart, 2008; Stergiou et al., 2009). Furthermore, fisheries management should be complemented with other ecosystem-based measures (Stergiou et al., 2009).

Gillnets are fishing gears widely used for commercial purposes (Machelis et al., 1994) and in research activities, since they are

* Corresponding author.

E-mail address: nuno.caiola@irta.cat (N. Caiola).

useful to monitor catch-length distribution (Hamley, 1975) to determine a population size structure, and to assess fishing effects on exploited stock (McAuley et al., 2007). Gillnet selectivity can be estimated through direct or indirect (i.e. species size distribution is

unknown) methods, by comparison of the observed catch frequencies across various mesh sizes (Millar and Holst, 1997; Millar and Fryer, 1999). Nevertheless, indirect methods for estimates are more common (Hamley, 1975; Millar and Holst, 1997). The SELECT (Share Each Length's Catch Total) method, is an indirect method implemented by Millar (Millar and Holst, 1997; Millar and Fryer, 1999; Millar, 2000) that provides a cohesive approach to size selectivity analysis. SELECT is the most widely used in gillnet selectivity studies (Reis and Pawson, 1999; Stergiou and Karpouzi, 2003; Carol and Garcia-Berthou, 2007).

The Ebro River ends up in the Mediterranean Sea forming a high valued Delta in terms of biodiversity and productivity (Day et al., 2006). Nutrient inputs from the river allow the development of a prosperous bivalve aquaculture activity and establish one of the richest fishing areas in the western Mediterranean Sea (Navarro et al., 2009). Artisanal, small-scale fisheries are carried out in the Ebro Delta coastal lagoons. This activity is not submitted to restrictive control measures regarding number of fishing gears per lagoon and species' size reporting, meaning that a high amount of captured fish are directly sold without intermediaries. Although it has been pointed by Sostoa (1983), the effects of this low surveillance effort have not been previously studied and may culminate in an overfishing situation with uncertain consequences. Of major concern is the sand smelt (*Atherina boyeri* Risso, 1810) fishery, a small, short-lived and euryhaline fish that inhabits coastal and estuarine waters (Leonardo, 2001; Patimar et al., 2009). Although *A. boyeri* is catalogued as a vulnerable species in Spain (Doadrio, 2001) and without an MLS established (RD 1615/2005), its fisheries are among the most important on the Spanish Mediterranean coast and with increasing interest (Andreu-Soler et al., 2006). This high interest is probably due to the strong restrictions and high monitoring on the catches of juveniles (with a similar length of *A. boyeri* of other commercial species such as the sardine (*Sardina pilchardus*), the anchovy (*Engraulis encrasicolus*) and the Atlantic horse mackerel (*Trachurus* spp.), among others, very much appreciated in Spain. In the Murcia region (SE Spain), from 2004 to 2009 catches were $13,155 \pm 7957$ kg/year, with a market value of 5.8 ± 1.55 V/kg (Ministry of Agriculture and Water; Government of Murcia); and in the Ebro Delta coastal lagoons were 2355 kg/year for the last five years (2006–2011), with a minimum of 583 kg and a maximum of 5012 kg, for 2006 and 2008 respectively (St. Pere Fishermen's Association). Thus, the development of an MLS for this species is the first essential step in order to establish an appropriate management action.

The aim of this paper was to consider mesh regulation and choice of fishing nets as a possible approach to prevent overfishing of a vulnerable species in Spanish lagoons. Therefore, we used the following approaches: (i) modelling gillnet selectivity using the SELECT method, as the first attempt to describe gillnet selection patterns for eight species with commercial interest in three Ebro Delta coastal lagoons; (ii) combining Gillnet and Fyke net information to establish an MLS for *Atherina boyeri*. This allowed us to develop conservation and management guidelines.

2. Material and methods

2.1. Study area

The Ebro River with 910 km long and a drainage area of 85,362 km², is one of the most important tributaries to the Mediterranean Sea, where originates one of the largest deltas (320 km²)

in the north-western Mediterranean Basin (Fig. 1). Although it is the Spanish river with the highest mean annual flow (426 m³/s), it shows a high variability between dry (118 m³/s) and wet (569 m³/s) years. This environmental variability is usual in the Mediterranean regions but it is exacerbated by a long history of human induced pressures (Caiola et al., 2001a, b; Ferreira et al., 2007a, b). The majority of the delta plain is devoted to rice agriculture (65% of the total surface) and natural areas cover only about 80 km² (25%). These areas include salt marshes, reed-type marshes, sand dunes, coastal lagoons, natural springs, and bays. The study was conducted in three coastal lagoons from the Ebro River Delta: Encanyissada, Clot and Tancada (Fig. 1). In the Encanyissada lagoon the largest one in the Ebro Delta (4.18 km²), salinity ranges between 3 and 30 and its mean depth is approximately 50 cm. The Clot lagoon (0.56 km² surface area; 0.8–12 salinity range and 30 cm mean depth) although it is part of the Encanyissada lagoon system, is separated by a floodgate showing a different water management scheme which determines differences in the hydrology and ecology between both lagoons. The Tancada lagoon has a 1.85 km² surface area, 37 cm of mean depth and a salinity range between 8 and 36. Both the Encanyissada and Tancada lagoons are connected with the Alfacos bay (Fig. 1).

2.2. Sampling methods and data collection

A total of twenty-four multimesh nylon gillnets of 30 x 1.5 m in length and height respectively, were set in the lagoons during two different sampling times (March and September 2008) (Fig. 1). Each gillnet consisted on a series of twelve panels (2.5 m width) composed by different randomly distributed meshes ranging from 5.0 to 55.0 mm bar length (see Table 1 for mesh sizes) to avoid confusion of the mesh size with environmental gradients, and following a geometric progression to optimize efficiency (Kurkilahti et al., 2002). The nets were made of monofilament twine (ranging from 0.10 to 0.20 mm depending on the panel) and the hanging ratio oscillated between 0.493 and 0.5 depending on the panel mesh size. Nets were set on late afternoon and hauled the next morning, hence being an average soak time of 12 h. Twenty-four fyke nets of 2 m long with a hoop diameter of 80 cm and 5.0 mm mesh size, were set and hauled at the same time and sampling points of the gillnets. All fish specimens were identified to species level sorted by mesh size, and fork length (hereafter FL) was measured to the nearest millimetre.

A total of 164 females of *Atherina boyeri* from the Ebro Delta coastal lagoons were collected in 1982 during the reproduction months (Sostoa, 1983). In the same year, the fish specimens were measured for total body length (to the nearest mm) and dissected. The gonads were kept in Gilson's fluid (Bagenal and Braum, 1978) so oocytes, once separated, could be counted and classified according to their development stage. The presence of ripe oocytes in the ovaries of each female, which indicates maturity, was then recorded. This dataset was used to establish the relationship between body length and maturity stage in order to determine the MLS of the species.

2.3. Data analysis

Gillnet selectivity for each species (Table 1) was estimated by the SELECT method (Share Each Length's Catch Total) through R (2.8.1 version) code developed by Russell Millar (available at: <http://www.stat.auckland.ac.nz/~millar/selectware/R>). The SELECT method is a generalized linear model that assumes a Poisson distribution of the gillnet catch data to fit them to a specified model using maximum-likelihood (Millar, 2000). Selectivity curves are defined as the relative probability of a fish of a given length to be captured when

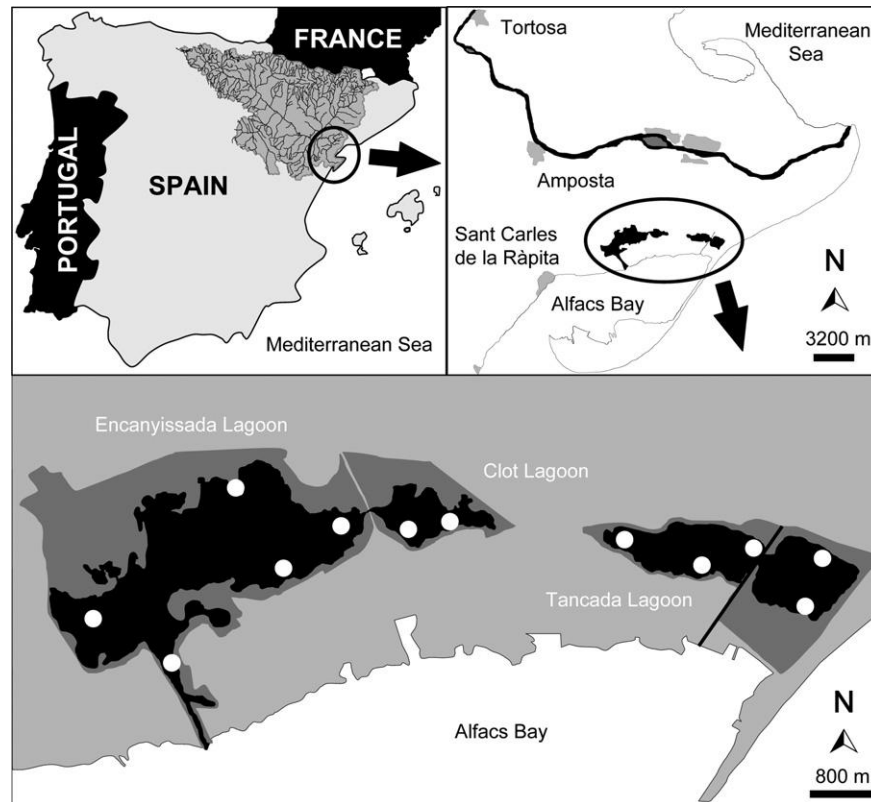


Fig. 1. Location of the study area and map of the three lagoons with the sampling points for gillnets and fyke nets (white circles).

contacting to a mesh of a determinate size (Millar, 2000). Five models were tested, the four previously available in the R scripts of SELECT (Millar, 2000) (normal location, normal scale, gamma and lognormal) and the inverse Gaussian model, developed and added by us (Table 2). All five models are unimodal and consist of two parameters describing the location and dispersion of the curves. The normal location and normal scale models are based on the normal distribution, whereas the other three are skewed curves with positive asymmetry. In addition to its statistical accuracy and

availability of several models, SELECT method analyses the data of all meshes within a single model, increasing statistical precision and power (Carol and Garcia-Berthou, 2007). All models were fitted under the assumption of equal effort of mesh size and assuming fishing power proportional to mesh size. The goodness of fit was performed by referring model deviances to a chi-squared distribution with df degrees of freedom (Madsen et al., 1999), with the lowest deviance values corresponding to the best fitting model ($p < 0.05$ denotes lack of fit). The Baranov's principle of geometric

Table 1

Total gillnet captures (N) by species and mesh size. Species mean size (fork length or total length (*) depending on the species) and range (mm) is also shown.

Fish species		Mesh size (mm)													Fish length	
Species name	Common name	N	5.0	6.25	8.0	10.0	12.5	15.5	19.5	24.0	29.0	35.0	43.0	55.0	Mean	Range
<i>Anguilla anguilla</i>	Eel	1									1				320*	320*
<i>Atherina boyeri</i>	Sand smelt	1172	361	500	303	8									65	40–90
<i>Engraulis encrasicolus</i>	European anchovy	3		3											68	67–69
<i>Alburnus alburnus</i>	Bleak	29	2		14	2	6	5							91	45–136
<i>Barbus graellsii</i>	Barbel	3				1	1		1						139	90–185
<i>Carassius auratus</i>	Goldfish	12						1	1	2	2	3		3	177	82–324
<i>Cyprinus carpio</i>	Carp	36						3	5		5	13	5	5	187	81–331
<i>Pseudorasbora parva</i>	Topmouth gudgeon	175	1	45	75	48	5			1					70	41–96
<i>Chelon labrosus</i>	Thicklip grey mullet	16			1	7	1	1		2	2	1	1		190	72–360
<i>Liza aurata</i>	Golden mullet	29		4		1	6	6	4	5		1	1	1	206	55–387
<i>Liza ramada</i>	Thinlip mullet	389		3	14	86	130	44	43	48	10	6	3	2	184	64–395
<i>Liza saliens</i>	Leaping mullet	341	15	65	53	56	78	51	17	6					140	32–310
<i>Mugil cephalus</i>	Flathead mullet	26				1		1	1	12	3	2	5	1	247	83–495
<i>Dicentrarchus labrax</i>	Sea bass	12					1	2	5		2		2		198	110–337
<i>Pomatoschistus microps</i>	Common goby	25	25												40*	35–49*
<i>Sparus aurata</i>	Gilthead seabream	6								4	2				144	127–169
<i>Solea senegalensis</i>	Senegalese sole	1							1						244*	244*
<i>Sander lucioperca</i>	Pikeperch	18					1	5	3	5	4				251	215–281
<i>Silurus glanis</i>	Wels catfish	2								1	1				205*	196–213*
<i>Syngnathus abaster</i>	Black-striped pipefish	1		1											80*	80*

Table 2

Fitting parameters of the five models tested with the SELECT method by species (in **bold** best model fit) in the Ebro Delta lagoons. Parameters 1 and 2 are k and σ for normal location model; k_1 and k_2 for normal scale model (spread proportional to mesh size); α and k for gamma model; μ_1 and σ for lognormal model and k_1 and k_2 for inverse Gaussian model. Deviance statistic measure goodness of fit. Significant results $p < 0.05$ indicate lack of fit.

Species	Model	Equal fishing power					Fishing power relative to mesh size				
		Par. 1	Par. 2	Deviance	df	P	Par.1	Par. 2	Deviance	df	P
<i>Atherina boyeri</i>	Normal fixed	9.57	4.21	82.69	142	1.000	9.62	4.22	85.04	142	1.000
	Normal scale	9.74	0.41	42.02	142	1.000	9.78	0.41	42.02	142	1.000
	Gamma	225.68	0.04	41.74	142	1.000	226.68	0.04	41.74	142	1.000
	Lognormal	3.88	0.07	42.56	142	1.000	3.89	0.07	42.56	142	1.000
	Inverse Gaussian	9.72	2158.61	42.62	142	1.000	9.76	2168.19	42.62	142	1.000
<i>Chelon labrosus</i>	Normal fixed	10.19	63.21	45.78	96	1.000	10.80	63.26	46.86	96	1.000
	Normal scale	11.71	11.11	46.83	96	1.000	12.58	9.67	47.47	96	1.000
	Gamma	13.92	0.81	42.11	96	1.000	14.92	0.81	42.11	96	1.000
	Lognormal	4.45	0.25	40.10	96	1.000	4.51	0.25	40.10	96	1.000
	Inverse Gaussian	11.16	163.45	40.04	96	1.000	11.94	173.13	39.92	96	1.000
<i>Liza saliens</i>	Normal fixed	11.77	33.68	712.02	1020	1.000	12.41	34.53	686.82	1020	1.000
	Normal scale	12.62	10.50	766.22	1020	1.000	13.43	9.55	775.26	1020	1.000
	Gamma	16.66	0.75	672.19	1020	1.000	17.66	0.75	672.19	1020	1.000
	Lognormal	4.09	0.24	640.07	1020	1.000	4.15	0.24	640.07	1020	1.000
	Inverse Gaussian	12.41	200.11	640.27	1020	1.000	13.19	210.99	638.68	1020	1.000
<i>Liza aurata</i>	Normal fixed	10.60	102.27	106.46	230	1.000	12.57	117.33	110.18	230	1.000
	Normal scale	14.84	25.16	103.93	230	1.000	16.32	20.11	106.59	230	1.000
	Gamma	8.31	1.76	93.64	230	1.000	9.31	1.76	93.64	230	1.000
	Lognormal	4.45	0.37	90.47	230	1.000	4.59	0.37	90.47	230	1.000
	Inverse Gaussian	14.89	100.03	90.01	230	1.000	17.30	110.83	89.69	230	1.000
<i>Liza ramada</i>	Normal fixed	9.84	55.98	993.76	1418	1.000	10.83	59.40	962.22	1418	1.000
	Normal scale	10.69	5.22	782.44	1418	1.000	11.17	4.91	784.13	1418	1.000
	Gamma	19.21	0.56	771.94	1418	1.000	20.21	0.56	771.94	1418	1.000
	Lognormal	4.18	0.24	787.13	1418	1.000	4.24	0.24	787.13	1418	1.000
	Inverse Gaussian	10.81	173.98	795.73	1418	1.000	11.49	183.47	797.20	1418	1.000
<i>Mugil cephalus</i>	Normal fixed	8.23	25.00	37.67	173	1.000	8.32	25.23	38.22	173	1.000
	Normal scale	8.57	0.60	30.09	173	1.000	8.64	0.59	30.08	173	1.000
	Gamma	116.11	0.07	30.44	173	1.000	117.11	0.07	30.44	173	1.000
	Lognormal	4.45	0.09	30.68	173	1.000	4.45	0.09	30.68	173	1.000
	Inverse Gaussian	8.56	959.04	30.68	173	1.000	8.63	967.36	30.68	173	1.000
<i>Alburnus alburnus</i>	Normal fixed	8.93	10.26	30.76	102	1.000	9.03	10.32	31.69	102	1.000
	Normal scale	9.26	0.82	24.34	102	1.000	9.35	0.82	24.34	102	1.000
	Gamma	99.81	0.09	24.40	102	1.000	100.81	0.09	24.40	102	1.000
	Lognormal	3.83	0.10	24.48	102	1.000	3.84	0.10	24.48	102	1.000
	Inverse Gaussian	9.28	898.95	24.46	102	1.000	9.38	907.97	24.46	102	1.000
<i>Cyprinus carpio</i>	Normal fixed	5.41	22.24	49.27	173	1.000	5.48	22.41	48.76	173	1.000
	Normal scale	5.48	0.30	37.99	173	1.000	5.54	0.30	37.98	173	1.000
	Gamma	92.97	0.06	39.14	173	1.000	93.97	0.06	39.14	173	1.000
	Lognormal	4.44	0.11	39.81	173	1.000	4.45	0.11	39.81	173	1.000
	Inverse Gaussian	5.49	486.32	39.83	173	1.000	5.55	491.67	39.84	173	1.000
<i>Carassius auratus</i>	Normal fixed	4.87	15.84	14.47	58	1.000	4.94	15.99	14.70	58	1.000
	Normal scale	5.08	0.36	14.27	58	1.000	5.15	0.35	14.27	58	1.000
	Gamma	72.13	0.07	14.19	58	1.000	73.13	0.07	14.19	58	1.000
	Lognormal	4.36	0.12	14.18	58	1.000	4.37	0.12	14.18	58	1.000
	Inverse Gaussian	5.09	363.48	14.17	58	1.000	5.16	368.49	14.17	58	1.000
<i>Pseudorasbora parva</i>	Normal fixed	8.23	10.73	212.43	233	0.829	8.43	11.04	210.31	233	0.855
	Normal scale	8.59	0.82	107.77	233	1.000	8.68	0.81	107.70	233	1.000
	Gamma	83.15	0.10	114.58	233	1.000	84.15	0.10	114.58	233	1.000
	Lognormal	3.76	0.11	121.40	233	1.000	3.77	0.11	121.40	233	1.000
	Inverse Gaussian	8.60	661.77	123.50	233	1.000	8.72	670.13	123.61	233	1.000
<i>Sander lucioperca</i>	Normal fixed	11.13	71.00	47.81	62	0.908	12.20	79.83	48.02	62	0.904
	Normal scale	12.20	25.52	48.54	62	0.894	14.10	22.41	48.61	62	0.893
	Gamma	9.40	1.43	48.08	62	0.903	10.40	1.43	48.08	62	0.903
	Lognormal	5.10	0.32	47.83	62	0.907	5.20	0.32	47.83	62	0.907
	Inverse Gaussian	13.78	131.87	47.81	62	0.907	15.28	144.43	47.79	62	0.908
<i>Dicentrarchus labrax</i>	Normal fixed	8.45	12.49	6.48	46	1.000	8.52	12.48	6.48	46	1.000
	Normal scale	8.58	0.54	6.11	46	1.000	8.64	0.54	6.11	46	1.000
	Gamma	137.23	0.06	6.02	46	1.000	138.23	0.06	6.02	46	1.000
	Lognormal	4.67	0.09	5.99	46	1.000	4.68	0.09	5.99	46	1.000
	Inverse Gaussian	8.58	1179.51	5.99	46	1.000	8.64	1188.05	5.99	46	1.000

similarity (Baranov, 1948) is the main used assumption of gillnet selectivity. This principle interprets gillnet captures as a mechanical process that depends only on the relative geometry of the mesh and the fish, stating that “Since all meshes are geometrically similar and all fish of the same species (within a reasonable size range) are also geometrically similar, the selectivity curves for different mesh sizes must be similar” (Hamley, 1975). Four of the models fitted

here showed geometric similarity with both length and spread of the curves increasing proportionally to mesh size and only the normal location curve (with the modal length proportional to mesh size but the spread fixed over mesh sizes) (Millar and Fryer, 1999), did not present the geometric similarity.

Atherina boyeri minimum landing size (MLS) was estimated by a logistic regression model relating the total body length as the

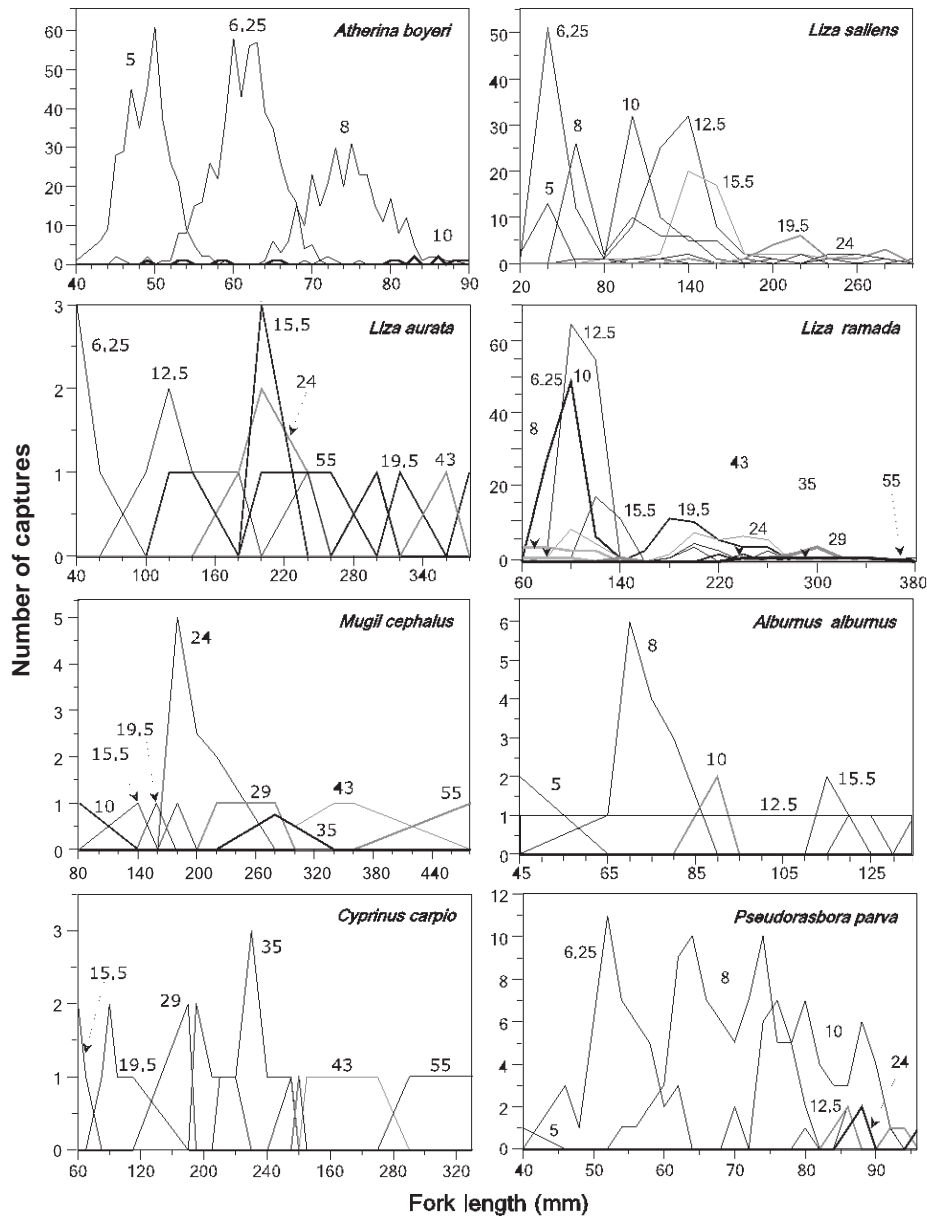


Fig. 2. Catch frequency per fork length class of the eight most captured fish species in the Ebro Delta lagoons (Figures shown above lines = bar length in mm).

quantitative predictor with the maturation stage as a qualitative response variable (“mature female” vs. “immature female”). This model allowed the estimation of the female probability to be mature according to its total length. The χ^2 value (and its associated p value) that compares the full model with the constant-only model and Nagelkerke’s R^2 , which is similar to the ordinary determination coefficient (Tabachnick and Fidell, 2001), are reported for the logistic regression. Size-at-maturity (L_x) of a population, is the total length (TL) at which X% of the individuals are mature and was estimated from the fit of logistic regression (Stergiou et al., 1996; Somarakis et al., 2005). The percentage of *A. boyeri* mature individuals were compared among gears and mesh sizes with a G-test of independence (Sokal and Rohlf, 1995). Analysis of variance (ANOVA) was used to compare model deviances between both approaches and to compare deviances among models and fished species, followed by Games-Howell post-hoc tests (hereafter, GH tests). The GH tests are among the most powerful and most robust to unequal variances of post-hoc multiple comparison methods

(Day and Quinn, 1989). The association of model deviances with catch number and number of meshes in which one species is captured was analysed with Spearman’s correlation coefficient (r_s). All statistical analyses were performed using SPSS v.17.0 software.

3. Results

3.1. Fitting selectivity curves

A total of 2297 fish belonging to 20 different fish species were gillnet caught (24 nets) and valid for SELECT analysis. Nevertheless, for gillnet selectivity estimation only fish species captured in at least four different panels and in enough number of individuals were considered (Tables 1 and 2). The mean length of captured fish increased with mesh size (Fig. 2). Although wedged and entangled fish were excluded from gillnet selectivity analysis, there were a few fish in mesh sizes larger than expected (e.g. one 199 mm *Mugil cephalus* in the 15.5 mm mesh) or in smaller meshes than

expected (e.g. one 316 mm *Liza ramada* in the 8 mm mesh or several *Liza saliens*) (Fig. 2). Also quite apparent was the increased size variability of catches with increasing meshes (i.e. geometric similarity) particularly in fish species with a higher number of captures (e.g. *Atherina boyeri*, *L. saliens*, *L. ramada*, *Pseudorasbora parva*) (Fig. 2). A contrasting pattern was found related to fish length range, so, the smaller *A. boyeri*, *L. ramada*, *P. parva*, and *Alburnus alburnus* were mainly captured in smaller mesh sizes, while larger *Cyprinus carpio* and *Carassius auratus* were principally caught in larger mesh sizes. The rest of fish species, with larger length ranges, were captured in a wider number of meshes (Table 1).

Fitting parameters estimated by the SELECT method for all models and fish species are given in Table 2 (also see Fig. 3 for best model fitting the eight most captured species). Overall, it was not observed a common pattern, since best fitting model varied among fish species. Assuming equal fishing power for all meshes, the normal scale (proportional spread) model had the lowest deviance value (i.e. better fit) for *Mugil cephalus*, *Alburnus alburnus*, *Cyprinus carpio* and *Pseudorasbora parva* (Fig. 3). For the *Chelon labrosus*,

Liza aurata, *Carassius auratus* and *Dicentrarchus labrax* (but only on the third decimal place), the best fit model was the inverse Gaussian model (Fig. 3). The gamma distribution model had the lowest deviance for the *Atherina boyeri* and *Liza ramada*; thus inverse Gaussian model was the best fit when having low number of captures, while gamma was the best fit with higher captures. For *Liza saliens* the best fit model was the lognormal model; and for *Sander lucioperca* was the normal location model (fixed spread) but only on the third decimal place (Table 2; Fig. 3). For all fish species, goodness of fit tests (Table 2) indicated no deviation of the observed catch for the best model predictions ($p > 0.90$). None of the models showed a lack of fit ($p < 0.05$) indicating the accuracy of results obtained. But, the normal location (for 9 species) and the normal spread model (for the rest of the species) were a worse fit than the rest of the models (Table 2). Similar results were obtained when fishing power relative to mesh size was assumed; only for *L. saliens* and *S. lucioperca* the best fit model changed to inverse Gaussian model due to smaller deviances values changes (Fig. 2). Thus, both approaches were suitable to estimate gillnet selectivity

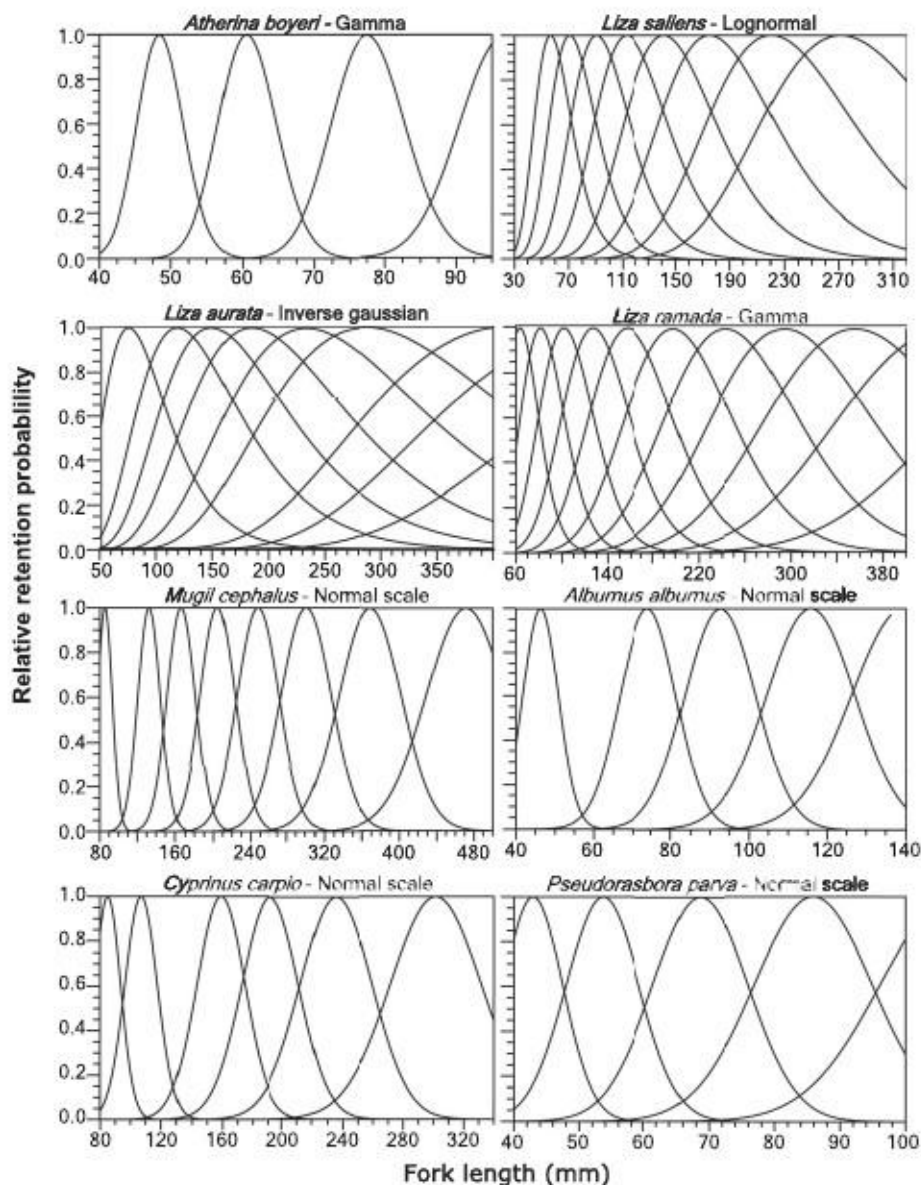


Fig. 3. Fitted selective curves for the eight most captured fish ($N = 20$) assuming equal fishing power to mesh size: Only model with best fit for each species is shown (Table 2). Meshes are shown in size ascending order consecutively from 5.0 mm mesh size (see Table 1).

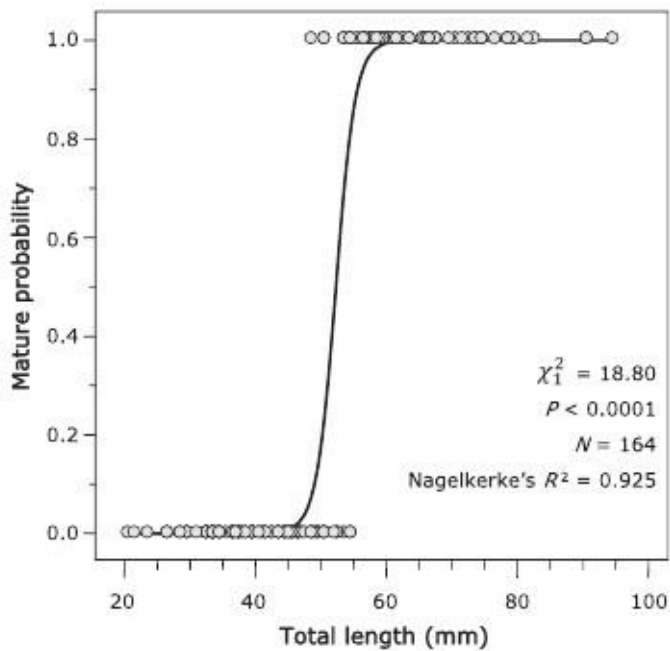


Fig. 4. Relationship between maturity stages of *Atherina boyeri* with total length. Circles shown the maturity of the individuals (0; immature and 1; mature) and black line illustrates the predicted probability to be mature at a determined length.

Model deviance (Table 2) also did not show significant differences between both gillnet estimation approaches (ANOVA, $F_{1, 44} = 0.75$, $p = 0.39$) but significantly depended on fish species ($F_{11, 44} = 51,776.4$, $p < 0.0001$) mostly because an effect of sample size (correlation between deviance and catches; Spearman's $r_s = 0.71$, $p < 0.001$) and mainly due to the number of meshes in which one species was captured (Spearman's $r_s = 0.52$, $p < 0.0001$), since species with larger samples sizes and captured by a wider range of meshes had larger deviance than less captured species or present in only a few number of panels (Tables 1 and 2). There was also a significant model \times species interaction ($F_{44, 44} = 121.92$, $p < 0.001$) because the fit of different models showed opposite patterns in different species; for instance when the normal scale model was the best fit (lowest deviance), normal location was the worst (e.g. *P. parva* and *A. alburnus*) when the normal location or lognormal were the best, normal scale was the worst (e.g. *S. lucioperca* and *L. saliens*). There was no significant evidence for both approach \times model or approach \times species interactions ($p > 0.16$) effects.

3.2. *A. boyeri* selectivity and minimum landing size

A total of 1625 *Atherina boyeri* individuals were captured in the three lagoons through fyke net and gillnet gears. By number, the *A. boyeri* only represented the 3.8% of the total catches from the fyke nets, being the fourth most important species ($N = 453$), after *Pomatoschistus microps* (6161), *Gambusia holbrooki* (3890) and *Aphanius iberus* (694); but it was the main catch species by gillnets (1172) (Table 1) corresponding to the 51% of the gillnet total captures. *A. boyeri* was only captured by the four smaller meshes (from 5.0 to 10.0 mm mesh; see Table 1). *A. boyeri* mean fork length was significantly different between gears (ANOVA, $F_{1, 1413} = 444.1$, $p < 0.001$), and fish length from fyke nets was smaller ($\mu \pm$ standard deviation = 46.66 ± 12.52) than those from gillnets (61.14 ± 10.50). Mean fork length was also significantly different among gillnet meshes ($F_{4, 1410} = 870.89$, $p < 0.001$); with fish length related to

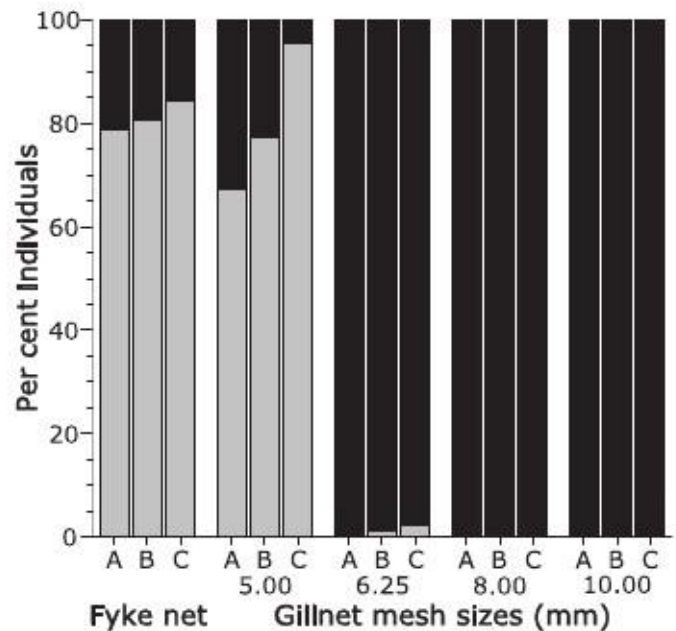


Fig. 5. Percentage of mature (■) and immature (□) individuals of *A. boyeri* by fyke nets and gillnets (by mesh size) per different lengths at maturity (A = L_{50} , B = L_{75} , C = L_{95}) estimated from the fit of logistic regression (see Material and methods).

mesh size 5.0 mm (49.08 ± 3.21); 6.25 mm (61.56 ± 3.92); 8.0 mm (74.21 ± 5.48) and 10.0 mm (80.80 ± 8.62). Thus fyke net and the 5.0 mm mesh size panel captured the smallest individuals, while the 8.0 and 10.0 mm panels caught the largest fish (GH post-hoc tests; fyke nets < 5.0 mm $<< 6.25$ mm $<< 8.0$ mm = 10.0 mm). Summarizing, fyke nets captured less and smaller individuals than gillnets, and the 6.25 mm mesh size panel and higher, were the best for the fishery of *A. boyeri*; since an increase in both catches and fish length were observed.

From the total *Atherina boyeri* individuals examined for maturity ($N = 164$), all fish with a total length below 45 mm were sexually immature; and fish with a length over 55 mm were all mature, so maturity was significantly related to fish length ($p < 0.0001$) (Fig. 4). The L_{50} of the *A. boyeri* (estimated from the logistic model) was 52.27 mm TL (Fig. 4), and less than 2 mm more were necessary to reach the L_{75} (53.92 mm TL). The L_{95} and L_{99} were estimated from the logistic model at sizes of 56.67 and 59.12 mm total length respectively. Interestingly, the percentage of mature individuals captured significantly varied among meshes ($G_4 > 1495.393$; $p < 0.0001$ for L_{50} , L_{75} and L_{95}); and the percentage of mature fish caught were positively related to mesh size ($r_s > 0.542$; $p < 0.0001$ for L_{50} , L_{75} and L_{95}) (Fig. 5). Therefore, fyke nets and 5.0 mm mesh gillnets captured less and smaller individuals and with a higher percentage of immatures, than larger meshes. Increasing mesh size only 1.25 mm (i.e. up to 6.25 mm), the number of captures increased with a percentage of mature individuals close to 100% (Fig. 5).

4. Discussion

4.1. Gillnet-fit and selectivity curves

The normal scale model had the best fit in four of the twelve fish species. The lognormal and the normal location models had the best fit for one fish species; only on equal-effort approach combined with inverse Gaussian model on unequal-effort approach, with the best fit for four different fish species. The

gamma model, showed the best fit for two fish species; any model, also, showed a lack of fit indicating the accuracy of the results. In general, the worst approach was achieved by the normal fixed model mostly due to its fixed spread, because assumes no geometrical symmetry (McAuley et al., 2007). Overall, gillnet selectivity curves approaches normal curves when most fish are wedged or gilled, but when many fishes are entangled or snagged, catch data are skewed to the right and are better fitted by gamma, lognormal, inverse Gaussian models or multimodal mesh models (Hamley, 1975; Hovgård, 1996; Dos Santos et al., 2003; Erzini et al., 2003). According to this, some studies have reported that the binomial model may better fit that unimodal models, when fishes are caught by a combination of different processes (i.e. gilled, entangled, wedged or snagged) and therefore smaller or larger fish than expected are found in a determinate mesh (Holt, 1963; Hovgård, 1996; Erzini et al., 2003). Nevertheless, although the binomial model was not available in the R-scripts of SELECT, in the present study the presence of entangled or wedged fish were lower than the 2%, so bimodality was not evident in our catch data.

Model fit behaved opposite pattern in different species; when the normal location or lognormal models were the best (lowest deviance), normal scale was the worst. Residual symmetry analysis provides an explanation for this opposite behaviour (Carol and Garcia-Berthou, 2007). All models, except normal location model, incorporate positive asymmetry, thus when normal location model z length residuals had a positive asymmetry, disappearing in the rest of the models that incorporated it. Nevertheless, for species best fitted with the normal location model, uniform and symmetric residuals are expected and a length variability reduction with increasing mesh size; meanwhile models with positive asymmetry would produce larger residuals with negative asymmetry (see Millar and Holst, 1997) since the variability increase with increasing mesh size (i.e. geometric similarity). This variability may have been influenced by different factors, for instance if fishes were entangled or wedged; also, differences in body shape, allometric growth, girth at the point of capture, different behaviour towards gillnets and the presence of appendages are important in determining how fish are caught by gillnets (Campos and Fonseca, 2003), and if selection curves are skewed or bell-shaped (Hamley, 1975).

Species with typical fish body shape such as the cyprinids *Cyprinus carpio*, *Pseudorasbora parva* or *Alburnus alburnus* have lesser allometry, existing a clear linkage between fish length and shape and both increasing with gillnet mesh size. Thus they were best fitted with normal scale model. However, species that have greater allometry, for instance the *Carassius auratus* (higher body depth e fish length relationship) and mugilids (higher fish length e body girth relationship), or species with gill bony appendages such as *Sander lucioperca* and *Dicentrarchus labrax*, were best fitted with skewed models (gamma, lognormal and inverse Gaussian models).

4.2. Gear selectivity implication on the *A. boyeri* fisheries management

According to the Code of Conduct for Responsible Fisheries (FAO, 1995) fisheries management strategy should be developed from a multispecific point of view. However, multispecies management is arduous to implement due to the difficulty of having enough information on species biology, ecology and interactions among species (Sainsbury et al., 2000; Campos and Fonseca, 2003). *Atherina boyeri* is a widespread and locally abundant species along the Mediterranean Sea, and it is commercially exploited in different Mediterranean areas such as Croatia, Greece or Italy (Maci and Basset, 2010) being one of the most important fisheries in the Spanish Mediterranean coast (Andreu-Soler et al., 2003). For instance its captures triplicate in a period of three years

in Mar Menor (1998–2000), being one of the commercial species of major interest in this area (Andreu-Soler et al., 2006). This commercial interest contrast with the lack of studies on its biology and ecology mainly when referred to the Spanish Mediterranean coast, where only Sostoa (1983) and Andreu-Soler et al. (2003) have described some aspects of its biology in the Ebro River Delta and Mar Menor lagoon respectively. This lack of knowledge is translated into an absence of the fisheries regulation. Although the establishment of an adequate MLS is the most basic measure in fisheries management (Stewart, 2008; Stergiou et al., 2009), *A. boyeri* was the only commercial species present in the lagoon without an MLS regulated by the European Union (CE 1967/2006) or the Spanish government (RD 1615/2005). In order to enhance fish recruitment and juvenile survival, it is necessary to establish a valid MLS and to improve the knowledge of the fisheries gears selectivity (Wileman et al., 1996). In the Ebro Delta coastal lagoons, *A. boyeri* represented only 3.8% of total fyke nets catches but accounted for 51% of gillnets catches; although it was present in only four of the 12 gillnet panels. For the three smaller gillnet panels (5.0 mm, 6.25 mm and 8.0 mm) separately, *A. boyeri* accounted for 89%, 81% and 66% respectively. Fyke nets fished all the *A. boyeri* size range and significantly captured smaller individuals (i.e. higher proportion of immature fish) than gillnets, even when compared with gillnet 5.0 mm mesh size. Different gillnet meshes captured a narrow size range with fish mean length increasing on larger panels and reducing the proportion of immature fish captured. Thus, in the case of *A. boyeri*, gillnets demonstrated its higher specific and size selectivity usefulness, suggesting that they could be a better tool for the assessment of *A. boyeri* fisheries allowing, thus, obtaining more suitable data for management recommendations when compared to fyke nets. Since the selectivity of the gear is the major factor determining the exploitation of a fish stock (Suuronen et al., 2007) should be related to a valid MLS. Furthermore, an ecological MLS should be close to L_{50} , but if the management aim is to protect juveniles, then the L_{75} criteria would be applied (Stewart, 2008). In our study we found that mesh sizes over 6.25 mm had a similar effect in *A. boyeri* catch, independently of both L_{50} and L_{75} approaches, even if an L_{95} was assumed. Thus, from a fisheries management point of view mesh sizes lower than 6.25 mm should be avoided, together with fyke nets of the same mesh sizes. The use of 6.25 mm bar length mesh appeared to be the most appropriate tool for the exploitation and management of *A. boyeri* fishery, since it captured higher number of individuals allowing the establishment of a L_{75} MLS (near 54 mm TL) in order to enhance juvenile survival (Stewart, 2008).

Gear selectivity and valid MLS are commonly used to improve fisheries management, restricting the size of fish captured (Machelis et al., 1994; Dos Santos et al., 2003). As far as we are concern, this is the first time that those are applied to the fisheries of a small fish species, and particularly to *Atherina boyeri* (in our study TL = 100 mm). This framework must be the first step to stop the low surveillance fishing situation. Thus, future research should focus on exploring how management actions may affect *A. boyeri*, the rest of fish community and the relationship among them in order to establish a global management plan.

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Salinity as the main factor structuring small-bodied fish assemblages in hydrologically altered Mediterranean coastal lagoons

SÍLVIA RODRÍGUEZ-CLIMENT, NUNO CAIOLA and CARLES IBÁÑEZ

IRTA Aquatic Ecosystems, E-43540 Sant Carles de la Ràpita, Catalonia, Spain. E-mail: nuno.caiola@irta.cat

SUMMARY: In the Ebro Delta coastal lagoons, one of the main anthropogenic pressures is the artificial freshwater input. Each coastal lagoon has different water management schemes causing profound changes in its physicochemical characteristics. The main objective of this water management is to favour some bird species with interest either for conservation or hunting activities. The present study assesses the influence of hydrological alteration on the fish assemblages of three coastal lagoons in the Ebro Delta. The small-bodied fish fauna was mainly composed of five families: Gobiidae, Poeciliidae, Cyprinodontidae, Atherinidae and Mugilidae. Salinity was found to be the main factor structuring fish community in the lagoons. The dominant species was the common goby (*Pomatoschistus microps*) when the lagoons reached higher salinity values, whereas the invasive eastern mosquitofish (*Gambusia holbrooki*) dominated during the period of higher freshwater inputs. The juveniles of the family Mugilidae showed low catch per unit effort, especially during the period of lower salinity. This same pattern was found for the endangered Spanish toothcarp (*Aphanius iberus*). Overall, introduced species were favoured by low salinity, which highlights the importance of changing the present water management by reducing the freshwater inputs in order to maintain suitable levels of salinity to favour native species that are important for both commercial and conservation purposes.

Keywords: fish community, salinity, water management, Ebro Delta, coastal lagoons.

RESUMEN: La salinidad como el principal factor en la estructuración de comunidades de peces de pequeño tamaño en lagunas costeras mediterráneas hidrológicamente alteradas. – Una de las mayores presiones antropogénicas sobre las lagunas costeras del Delta del Ebro es la entrada artificial de agua dulce. Cada laguna costera presenta medidas de gestión distintas que alteran profundamente sus propiedades fisicoquímicas. Esta gestión del agua tiene como objetivo favorecer algunas especies de aves con interés para la caza y/o conservación. El presente estudio evalúa la influencia de la alteración hidrológica en la comunidad de peces de tres lagunas costeras del Delta del Ebro. La fracción de la ictiofauna integrada por individuos de pequeño tamaño estaba mayormente compuesta por cinco familias: Gobiidae, Poeciliidae, Cyprinodontidae, Atherinidae y Mugilidae. La salinidad fue el factor que más contribuyó a la estructuración de la comunidad de peces de las lagunas. Con valores máximos de salinidad, la especie dominante en las lagunas fue el Gobito (*Pomatoschistus microps*), mientras que la *Gambusia holbrooki*, una especie invasora, fue dominante durante el período con mayores aportes de agua dulce. Los juveniles de la familia Mugilidae presentaron valores bajos de Capturas por Unidad de Esfuerzo (CPUE), especialmente durante el período de menor salinidad. El mismo patrón fue encontrado para el Fartet (*Aphanius iberus*), especie en peligro de extinción. En general, las especies introducidas fueron favorecidas por una baja salinidad, hecho que subraya la importancia de modificar la presente gestión del agua reduciendo los aportes de agua dulce con el fin de mantener niveles adecuados de salinidad que favorezcan a las especies nativas, importantes tanto desde el punto de vista comercial como conservacionista.

Palabras clave: comunidad de peces, salinidad, gestión del agua, Delta del Ebro, lagunas costeras.

INTRODUCTION

Coastal lagoons are located at the land-sea interface and are generally characterized by high biological

productivity and shallowness (Barnes 1980, Brehmer *et al.* 2011). They are used by many fish species either as feeding, nursery or spawning grounds (Pérez-Ruzafa *et al.* 2004, Oertli *et al.* 2005, Ribeiro *et al.*

2006, Verdiell-Cubedo 2009). Usually dominated by marine and estuarine species, coastal lagoon fish assemblages may have different compositions and structures, depending mainly on the balance between marine and freshwater inputs (Gordo and Cabral 2001). The Mediterranean coastal lagoons are highly modified ecosystems due to the impacts of human activities such as intensive agriculture, urban and industrial land uses, canal construction and impoundments, dredging and tourism. These activities cause serious changes in the natural ecological cycles, particularly water regime alterations caused by artificial freshwater inputs (Pérez-Ruzafa *et al.* 1991, Day *et al.* 2000, Caiola *et al.* 2001a, b, Cañedo-Argüelles *et al.* 2012, Verdiell-Cubedo 2009). The most common consequences of these hydrological alterations are the decline in salinity and the increase in both eutrophication and environmental pollution as a result of increased nutrient and chemical contaminant inputs (Cloern 2001, Lucena *et al.* 2002, Cañedo-Argüelles *et al.* 2012). Despite the severe impacts that these hydrological alterations can have on the fish communities, studies focusing on this issue in the Mediterranean area are scarce (Poizat *et al.* 2004, Badosa *et al.* 2007).

Situated in Catalonia (north-east Spain), the Ebro Delta, covering an area of 320 km², is one of the most important estuarine systems in Europe. Due to the high ecological interest, 25% of its area, including the coastal lagoons, bays and adjacent coastline, has been protected as a natural park since 1983. The natural park conservation status means that the protection of the natural areas must be compatible with the use of their resources and the activities of their inhabitants, such as fisheries, hunting and agriculture. The rest of the area is used for agriculture, mainly rice cultivation (Day *et al.* 2006). To sustain this agricultural activity two main canals, regulated at their source (40 km upstream of the river mouth), are split into numerous secondary canals and ditches to bring freshwater to all rice fields in the delta. Freshwater supply is only interrupted between January and April to prepare the fields for the next crop. In the past, waste water from the rice fields was drained to the lagoons, and from these systems to the sea through the outlets. This water management scheme caused eutrophication in the coastal lagoons, a problem that was overcome in the 1990s with the construction of circumvallation canals that prevent drainage water from passing through the lagoons (Forés *et al.* 2002). At present the Ebro Delta coastal lagoons still receive freshwater artificially, mainly for habitat management to favour game birds. The freshwater comes both from the river, through the irrigation canals, and from the rice field drainage. The quality of the river water has increased significantly in the last 15 years (Nebra *et al.* 2011, Ibáñez *et al.* 2012a,b) and the water input from rice field drainage is only opened in periods of low pesticide and fertilizer input (Forés *et al.* 2002). As a result, water quality in the lagoons has improved, but

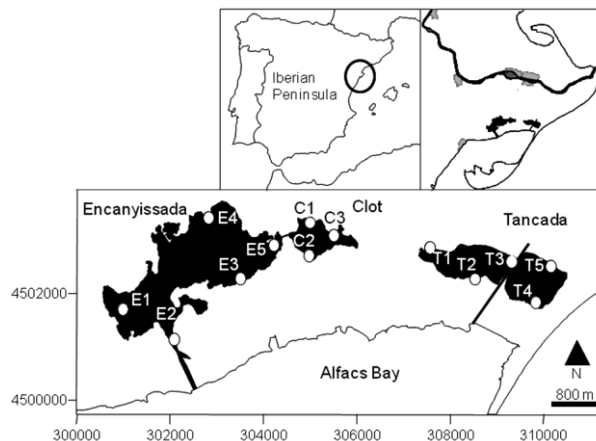


FIG. 1. – Map showing the location of the Ebro Delta and the sam-pled coastal lagoons. White spots represent sampling sites.

their hydrology is still severely altered. The effects of water management on the coastal lagoons' fish assemblages remain unknown. The approach of sampling small-sized resident fish species and juveniles of migratory species was chosen because the capacity of movement of larger species and the adults of migratory species make them unreliable as descriptors of spatial distribution (Mariani, 2001). Moreover, if only small-bodied fishes are analysed, target species are not taken into account, so the effect of fisheries on the fish community is excluded.

The present study aims to describe the composition of small-bodied fish assemblages from the Ebro Delta coastal lagoons and to investigate the relationship between assemblage structure and distribution patterns and the main environmental features that depend on artificial water management. The degree of association of each species with the studied physicochemical parameters is also assessed.

MATERIALS AND METHODS

Study area

Three coastal lagoons located in the Ebro Delta were selected for this study: Encanyissada, Clot and Tancada (Fig. 1). The Encanyissada, with an area of 4.18 km², is the largest lagoon in the delta; it has a mean depth of 77 cm (121 cm maximum depth) and a salinity ranging between 2 and 36. It is connected naturally to the Alfacs Bay (Mediterranean Sea) by the Sant Pere outlet (see Fig. 1) and to the Clot lagoon by a floodgate. The Clot is a small (0.56 km²) and shallow lagoon, with a mean depth of about 53 cm (maximum depth around 90 cm) and a salinity ranging between 1 and 29. Closest to the sea, the Tancada lagoon has a surface area of 1.85 km², a mean depth of 58 cm (80 cm maximum depth) and a salinity ranging between 8 and 37. It is connected to the Alfacs Bay by several artificial canals (Fig. 1).

Sampling procedures

Two sampling periods were selected, March and September 2008, coinciding with the periods of minimum and maximum artificial freshwater inputs, respectively. Fish were collected with fyke nets, a passive fishing gear that is increasingly used for scientific surveys (Poole *et al.* 2007), particularly for surveying small-bodied fish (Brazner *et al.* 1998, Pampoulie *et al.* 2001, Uzarski *et al.* 2005, Fredberg *et al.* 2009, Wedderburn *et al.* 2012). The fyke nets used were 2 m long with a hoop diameter of 80 cm and mesh size of 5.0 mm. A total of 26 fishing sets were settled in 13 different sampling locations so as to cover the maximum spatial gradient in each lagoon (5 in the Encanyissada and Tancada lagoons, 3 in the Clot lagoon) (Fig. 1). As coastal lagoons are shallow water bodies with a quite constant depth, and therefore aquatic habitat conditions, the fyke nets were set in the littoral zone where there is a higher probability of capturing fish due to habitat characteristics (higher macrophyte coverage and presence of helophytic and littoral vegetation). In fact, this same procedure is performed by local fishermen. Moreover, setting the fyke nets in the littoral zone allowed a constant depth of approximately 80 cm to be maintained between all the sampling points, corresponding to the fyke nets' hoop diameter (the whole water column was sampled for small-bodied fish). All nets were set in late afternoon and hauled the next morning, so average soak time was 12 hours. The fish were sorted, identified at species level and counted. This procedure complied with the fish specimen capture requirements as stated in the scientific capture permit issued by the Catalan Government: eliminating exotic species according to the current law (in this case with excess of anaesthetic MS-222) and releasing the native species after handling them. Each sampling site was also sampled for environmental descriptors that are directly affected by the water management. These descriptors are related to water physicochemical parameters, water nutrient loading variables, and water level expressed as depth (cm). With regard to the physicochemical parameters, dissolved oxygen (mgL^{-1}), temperature ($^{\circ}\text{C}$), salinity and pH were measured with an YSI 556 multi-parameter probe. To determine the water nutrient loading water samples were collected, preserved on ice in the absence of light, transported to the laboratory and stored at -20°C until a dissolved nutrient concentration analysis was done. The nutrients analyzed were ammonium (N-NH_4), nitrite (N-NO_2), nitrate (N-NO_3) and phosphate (P-PO_4), following Koroleff (1977).

Statistical analysis

Fish abundance was expressed as catch per unit effort (CPUE), an index of relative abundance that represents the success of capture. CPUE values were calculated as number of captured fish per fishing set. Fishing time was assumed to be equal, since soaking

time was roughly constant. Rare species ($\text{CPUE} < 0.833$ and/or present at just one sampling site) were excluded from the analyses. Normality and homogeneity of variances of all physicochemical variables were tested with the Shapiro-Wilk test and the Levene statistic (Clifford and Taylor 2008). Physicochemical variables were log-transformed and species data were square-root transformed when necessary in order to improve linearity and normality of variances (Quinn and Keough 2002). Analysis of variance (ANOVA) was used to compare physicochemical descriptors between lagoons and sampling periods, followed by Games-Howell post-hoc test (GH test). A Canonical Correspondence analysis (CCA) was performed in order to extract the species variation explainable by the measured physicochemical parameters (Lepš and Šmilauer 2003). To describe the individual species response to salinity, generalized additive models (GAMs) were fitted assuming a Poisson distribution and a log link function. GAMs are an extension of the general linear models that do not require the assumption of a linear relationship between environmental variables and the species distribution (Hastie and Tibshirani 1986, Lepš and Šmilauer 2003). The model complexity of GAMs was selected by the stepwise selection procedure using the Akaike Information Criterion (AIC), which considers not only goodness of fit but also parsimony, penalizing very complex models (Lepš and Šmilauer 2003). Thus, the relationship between the predictors and the dependent variable can assume any shape, from a straight line to non-parametric curves of increasing complexity (Alcaraz *et al.* 2011). Tolerance ranges of the species were analyzed by data attribute plots with salinity as the environmental variable. The statistical analyses were performed with SPSS 17.0 and CANOCO 4.5.

RESULTS

Environmental descriptors

The environmental descriptors of the three lagoons were compared between the two sampled periods (Table 1). Significant differences were found in temperature ($F_{1,92}=95.57$; $p < 0.01$), depth ($F_{1,92}=47.63$; $p < 0.01$), pH ($F_{1,92}=19.15$; $p < 0.01$) and salinity ($F_{1,94}=100.33$; $p < 0.01$), whereas no differences were found in oxygen concentration ($F_{1,91}=0.22$; $p=0.64$). The mean temperature in March was significantly lower than in September (GH, $p < 0.01$); Tancada lagoon was the coldest and Clot lagoon the warmest (GH, $p < 0.01$). All three lagoons showed significantly higher mean depth values in September, with Encanyissada lagoon being the deepest (GH, $p < 0.01$). The pH was highest in September, being statistically significant only for the Clot (GH, $p < 0.05$) and Tancada lagoons (GH, $p < 0.01$). Tancada lagoon showed the lowest pH values. In contrast, the salinity was significantly higher in March than in September (GH, $p < 0.01$). The maximum observed salinity was in the Tancada lagoon in March

TABLE 1. – Environmental descriptors for the three lagoons in the two sampled periods. Mean \pm standard deviation is shown. DO, dissolved oxygen. * statistical significance between seasons. ** statistical significance between lagoons. ns, non-significant differences.

		March 2008			September 2008		
	Clot	Encanyissada	Tancada	Clot	Encanyissada	Tancada	
T (°C)	*	19.74 \pm 0.63 **	16.97 \pm 0.99 **	11.87 \pm 0.34 **	21.52 \pm 0.47 **	20.15 \pm 1.31 **	19.78 \pm 0.96 **
DO (mg l ⁻¹)	ns	8.28 \pm 0.82	7.96 \pm 1.12	8.30 \pm 1.17	7.93 \pm 0.84	8.00 \pm 1.18	7.95 \pm 0.76
Salinity	*	9.99 \pm 1.01 **	18.40 \pm 4.91 **	30.30 \pm 0.89 **	1.20 \pm 0.16 **	9.04 \pm 2.00 **	10.29 \pm 1.78 **
pH	*	7.93 \pm 0.07 **	8.11 \pm 0.16 ns	7.72 \pm 0.09 **	8.23 \pm 0.18 **	8.10 \pm 0.11 ns	8.21 \pm 0.18 **
P-PO ₄ (mg l ⁻¹)	*	>0.001 **	0.012 \pm 0.004 **	0.006 \pm 0.003 **	0.045 \pm 0.022 **	0.039 \pm 0.011 **	0.040 \pm 0.011 **
N-NH ₄ (mg l ⁻¹)	ns	0.003 \pm 0.003	0.037 \pm 0.023	0.156 \pm 0.176	0.055 \pm 0.092	0.100 \pm 0.071	0.194 \pm 0.059
N-NO ₂ (mg l ⁻¹)	*	0.003 \pm 0.000 ns	0.006 \pm 0.003 ns	0.008 \pm 0.003 **	0.021 \pm 0.019 ns	0.007 \pm 0.002 ns	0.085 \pm 0.035 **
N-NO ₃ (mg l ⁻¹)	ns	0.010 \pm 0.006	0.157 \pm 0.165	0.075 \pm 0.077	0.191 \pm 0.320	0.023 \pm 0.044	0.374 \pm 0.265
Depth (cm)	*	39.29 \pm 11.67 **	65.07 \pm 15.23 **	56.50 \pm 10.31 **	68.43 \pm 7.57 **	91.30 \pm 17.35 **	67.50 \pm 11.63 **

TABLE 2. – List of species captured in the Ebro Delta lagoons and its abundance in CPUE. Lagoon: C, Clot; E, Encanyissada; T, Tancada. Status: N, native; I, introduced. Species with codes in **bold** were selected for the analysis.

Family	Common name	Species	Code	State	March 2008			September 2008		
					C	E	T	C	E	T
Anguillidae	European eel	<i>Anguilla anguilla</i>	AAN	N	0.167	0.083	0.250		0.083	0.083
Atherinidae	Sand smelt	<i>Atherina boyeri</i>	ABO	N	0.500	3.417	7.417	4.000	21.667	0.167
Cyprinidae	Carp	<i>Cyprinus carpio</i>	CCA	I			0.083			
Cyprinodontidae	South European toothcarp	<i>Aphanius fasciatus</i>	AFA	I		0.083				
Cyprinodontidae	Spanish toothcarp	<i>Aphanius iberus</i>	AIB	N	3.167	36.083	15.333		2.417	0.417
Gobiidae	Common goby	<i>Pomatoschistus microps</i>	PMI	N	30.917	148.917	221.750	77.083	22.083	10.833
Mugilidae	Thicklip grey mullet	<i>Chelon labrosus</i>	CLA	N		3.083	1.583		0.833	0.083
Mugilidae	Golden mullet	<i>Liza aurata</i>	LAU	N		0.167	6.167		1.250	
Mugilidae	Thinlip grey mullet	<i>Liza ramada</i>	LRA	N			4.083			
Mugilidae	Leaping mullet	<i>Liza saliens</i>	LSA	N	0.333	0.167	17.333		2.000	0.500
Mugilidae	Flathead mullet	<i>Mugil cephalus</i>	MCE	N			0.833			
Poeciliidae	Eastern mosquitofish	<i>Gambusia holbrooki</i>	GHO	I	5.333	3.250	0.333	218.333	95.917	0.417
Syngnathidae	Black-striped pipefish	<i>Syngnathus abaster</i>	SAB	N		0.250	0.083			

(30.30; *GH*, $p < 0.01$), whereas the lowest salinity was in the Clot lagoon (9.99; *GH*, $p < 0.01$). In addition, temperature, depth and pH were negatively correlated with salinity ($r = -0.87$, $p < 0.01$; $r = -0.37$, $p < 0.01$ and $r = -0.58$, $p < 0.01$, respectively).

Regarding the dissolved nutrient concentration (Table 1), significant differences in nitrite and phosphate were found between sampling periods ($F_{1,42} = 8.21$ and $F_{1,41} = 102.54$, respectively $p < 0.01$). Nitrite concentrations were significantly higher in September, Tancada lagoon being the one reaching the highest concentrations (0.085 \pm 0.035). The phosphate was significantly higher in September (*GH*, $p < 0.05$), with Clot showing the highest values. A negative correlation was found between salinity and phosphate ($r = -0.53$, $p < 0.01$). Furthermore, although no differences were found in ammonium and nitrate ($F_{1,42} = 2.92$ and $F_{1,41} = 1.09$, respectively; $p > 0.09$), their concentrations were higher in September, except for nitrate in Encanyissada.

Fish assemblages

Overall, 13 species belonging to eight different families were captured in the three lagoons during the two sampling periods (Table 2). The small-bodied fish composition in the lagoons was composed of three introduced species (23.08%) and ten native species (76.92%). Although the native species dominated in terms of richness in all lagoons and sampling seasons, in terms of abundance the introduced species *Gambu-*

sia holbrooki was dominant in September (69%). This pattern was observed in all the lagoons except in the Tancada, where the native species dominated in the two periods (Table 2). For further analysis, the rare species were excluded, so finally seven species belonging to five families were analyzed (Fig. 2).

The small-bodied fish community in March was dominated by *Pomatoschistus microps*, *Aphanius iberus* and *Liza saliens*. *P. microps* was present in all lagoons and sampling sites, whereas *A. iberus* was present at all sampling points, except T2 and T3. *L. saliens* was mostly present in saltier habitats, particularly in the Tancada lagoon (Fig. 2). The presence of *L. aurata* was low and it was always concentrated in the saltier environments. The invasive *Gambusia holbrooki* occurred in fresher areas (the Clot and Encanyissada lagoons). In March *Chelon labrosus* was found almost exclusively in sampling points with sea connections (the Encanyissada and Tancada lagoons).

In September the community was dominated by freshwater and brackish species such as *G. holbrooki*, *P. microps* and *Atherina boyeri*. In this period, *P. microps* was dominant in the Tancada lagoon, whereas *G. holbrooki* was dominant in both the Clot and Encanyissada lagoons. The species *A. boyeri* was abundant in the Encanyissada lagoon. The endangered *A. iberus* was present in all lagoons except Clot. The mugilids *L. saliens* and *C. labrosus* were present in the saltier habitats (Tancada lagoon), whereas the presence of *L. aurata* was restricted to the Encanyissada lagoon.

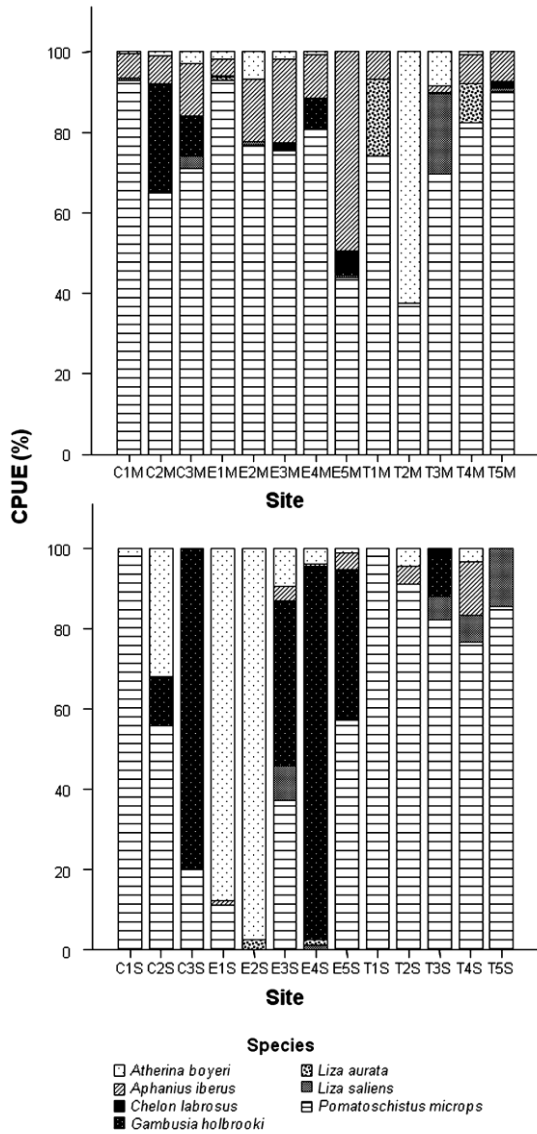


FIG. 2. – Percentage contribution of species to the overall abundance in the three lagoons sampled in March (above) and September (be- low). See Figure 1 for sampling site codes.

Effects of environmental descriptors on the fish assemblages

The first two axes of the CCA accounted for 27.4% of the total variability of the species data. The CCA summary statistics resulted in a significant first axis species-environment relationship ($r=0.836$, $p<0.01$). Salinity ($F=5.17$, $p<0.01$) and depth ($F=2.42$; $p<0.05$) were the only significant variables in the CCA analysis (Fig. 3). The first axis suggested a salinity gradient derived from the differences in the artificial freshwater inputs, which split September samples (fresher and deeper) from March samples (shallower and saltier). Regarding the species scores, brackish species such as *Aphanius iberus*, *Pomatoschistus microps*, *Liza saliens*, *Liza aurata* and *Chelon labrosus*, showed negative

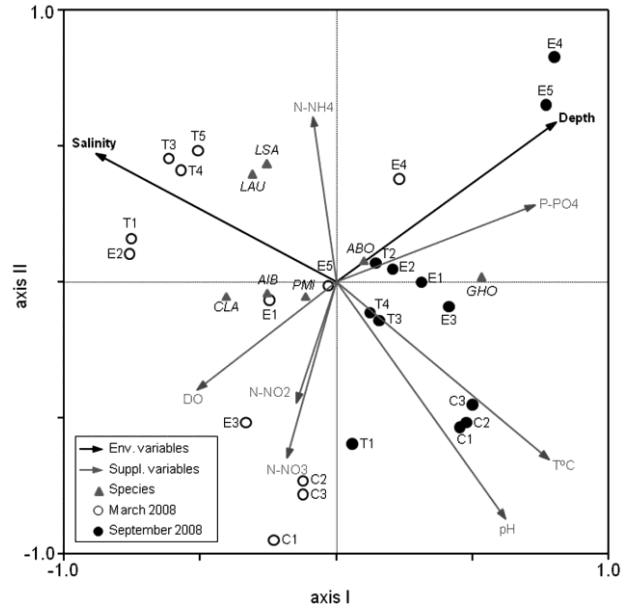


FIG. 3. – Correspondence canonical analysis triplot showing the relationships between physicochemical parameters, sampling sites and species. Significant and non-significant (supplementary) phys- icochemical parameters are plotted. Species abbreviations: ABO (*Atherina boyeri*), AIB (*Aphanius iberus*), CLA (*Chelon labrosus*), GHO (*Gambusia holbrooki*), LAU (*Liza aurata*), LSA (*Liza sali- ens*) and PMI (*Pomatoschistus microps*). See Figure 1 for sampling codes.

coordinate values, i.e. associated with higher salinity, whereas species with freshwater origin as *G. holbrooki* were situated in positive coordinates. *Atherina boyeri* was positioned near the centre of the plot. The samples ordination in the second axis (4.1% of the total variability) indicated nitrogen loading with positive coordinate values for the ammonium and negative values for nitrate and nitrite. Then, high ammonium concentration samples were situated in the upper part of the plot (Tancada in March) and low ammonium loaded samples in the lower part (Clot in March).

Species response to salinity

The response curves of species abundances (expressed as CPUE square-root transformed) to salinity are shown in Figure 4. The significance of the response curves is shown in Table 3.

TABLE 3. – Results of the generalized additive models (GAM) applied to all the selected species, showing the explained deviance of the fitted model (Deviance), the residual degrees of freedom (DF residual) the Akaike information criterion (AIC), and the model significance (F value). Asterisks denote significance levels: ** $p<0.01$; * $p<0.05$.

Species	Deviance	DF residual	AIC	F
<i>Pomatoschistus microps</i>	127.55	21	161.66	3.83 *
<i>Gambusia holbrooki</i>	230.58	22	278.27	12.18 **
<i>Aphanius iberus</i>	70.87	21	87.36	10.73 **
<i>Liza saliens</i>	63.54	21	86.00	4.83 *
<i>Liza aurata</i>	44.99	22	62.37	4.06 *
<i>Chelon labrosus</i>	28.40	21	37.75	6.02 **

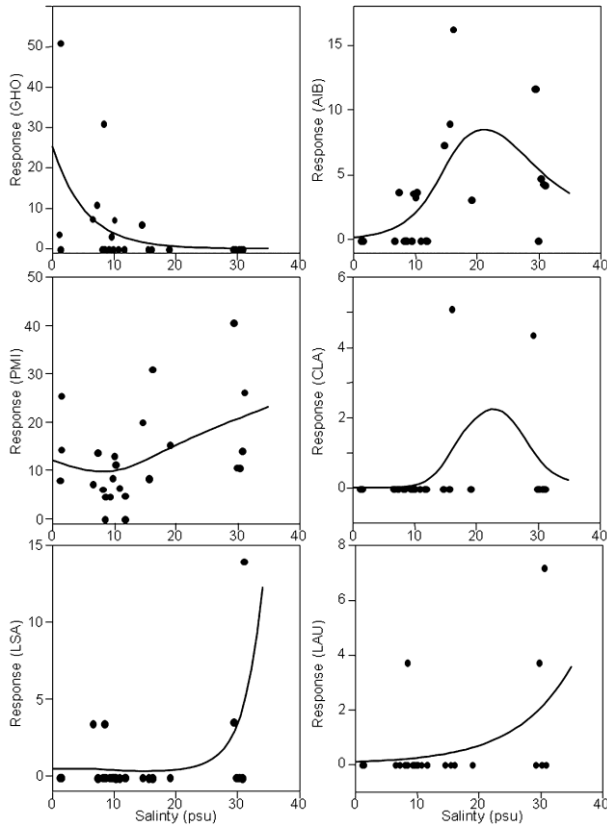


FIG. 4. – Response curves of species abundance (species selected by the model) to salinity. The curves are the generalized additive models (GAM) selected by the Akaike information criterion (AIC). Dots represent sampling sites. See Fig. 3 for species abbreviations.

Gambusia holbrooki decreased drastically with salinity, whereas *Pomatoschistus microps* increased with salinity values higher than 20. The mugilids *Liza saliens* and *Liza aurata* increased constantly with salinity, whereas the endangered *Aphanius iberus* showed salinity preferences around 20. *Chelon labrosus* and *Atherina boyeri* were not plotted as they were not selected during the GAM iterative process. The salinity range for each species is shown in Figure 5. *G. holbrooki* has its optimum in freshwater environments, whereas *A. boyeri* is more adapted to mesohaline waters (around 15). The rest of the species have their optimums in brackish to marine waters (18-30).

DISCUSSION

The continuous freshwater inputs during nine months per year have inverted the hydrological pattern in the Ebro Delta coastal lagoons (De Sostoa and De Sostoa 1985, Comín *et al.* 1987). The inverted pattern was further confirmed in the present study by the opposite salinity cycle, since lagoons were found to have fresher water during dry seasons, when the lowest rainfall took place (Badosa *et al.* 2007). September was not only the month with the lowest salinity in the lagoons, but also the period with the highest pH values. These

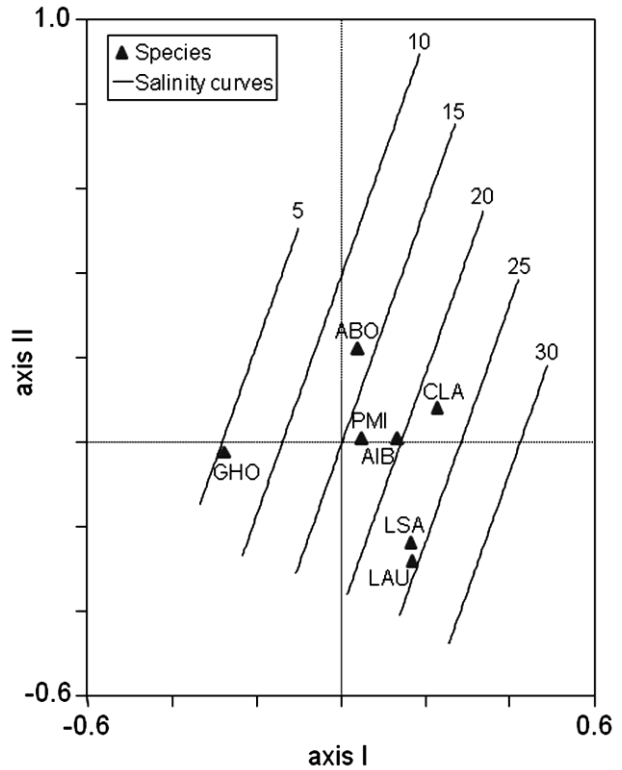


FIG. 5. – Species attribute plot showing salinity curves and the species preferences. Species abbreviations: ABO (*Atherina boyeri*), AIB (*Aphanius iberus*), CLA (*Chelon labrosus*), GHO (*Gambusia holbrooki*), LAU (*Liza aurata*), LSA (*Liza saliens*) and PMI (*Pomatoschistus microps*).

results contrasted with previous studies, which reported higher pH values in March (De Sostoa and De Sostoa 1985, Comín *et al.* 1987). These differences could be explained by the different management applied at each time, as the previous studies were carried out before the construction of the bypass canals (Forés *et al.* 2002). Furthermore, in addition to inverting the hydrological pattern, the continuous freshwater supplies tended to reduce the differences between the lagoons in September, whereas in March the differences between the lagoons were increased due to the cessation of the freshwater flow. Although the continuous freshwater inputs acted as a homogenizing agent, some differences still found between the lagoons within the same period could be attributed to a different water management scheme in each lagoon. This was the case of the Tancada lagoon, receiving the least freshwater inputs, and the Clot lagoon, receiving the greatest freshwater inputs. Therefore, these differences in freshwater inputs, in addition to the proximity to the sea, might explain the extreme differences in salinity values found between the Tancada and Clot lagoons.

The freshwater inputs in the lagoons are a mixture of drainage water from the rice fields and irrigating water from the Ebro River. During its entire course, the River Ebro collects nutrients from all the agricultural activities carried out in the basin (Mañosa *et al.* 2001). Furthermore, though rice fields are known to act

as ecological filters, the drainage water still contains a high amount of nutrients (Forés and Comín 1992). This fact might explain the negative correlation between nutrient loading and salinity that is common in coastal systems affected by agriculture activities (Chapelle *et al.* 2000, Lucena *et al.* 2002, Pérez-Ruzafa *et al.* 2005a). Moreover, this freshwater input could promote turbulence, favouring the internal recycling process in the lagoons, as has been suggested for phosphate and ammonium (Chapelle *et al.* 2000, Orfanidis *et al.* 2005, Badosa *et al.* 2007).

The amount of freshwater/marine inputs has been pointed out as the key factor in determining the fish assemblages in an ecosystem (Gordo and Cabral 2001, Mariani 2001). Therefore, any change that could modify the natural inputs of freshwater/marine water (such as the artificialization of the hydrological cycle) would consequently modify the existing fish assemblage. Fish assemblages of Mediterranean coastal lagoons with a climate-dependent hydrology (i.e. where climatic variation such as rain and floods is the only source of freshwater inputs), have been reported to be characterized by marine and brackish species with the dominance of the families Sparidae, Mugilidae, Gobiidae, Atherinidae, Soleidae, Sygnathidae and Blenniidae (Pérez-Ruzafa *et al.* 2006, Maci and Basset 2009, Verdiell-Cubedo 2009). On the other hand, fish assemblages of coastal lagoons with artificial freshwater supplies, such as the Vaccarès lagoon (France; Poizat *et al.* 2004) and the Albufera lagoon (Spain; Blanco *et al.* 2003, Blanco and Romo 2006), have families typical of freshwater environments, in addition to the aforementioned brackish and marine species. Our results suggest that the hydrological alteration of the cycle might be the responsible for the current fish community in the Ebro Delta coastal lagoons composed by brackish and marine families such as Gobiidae, Mugilidae and Atherinidae and by freshwater species belonging to the families Cyprinidae and Poeciliidae (Blanco *et al.* 2003, Poizat *et al.* 2004, Blanco and Romo 2006). Moreover, the introduction of the freshwater species (mostly the introduced *Gambusia holbrooki*) in these lagoons (Doadrio 2001) led to an alteration of the native species assemblage. The effect of the altered hydrological pattern on the proportion of native to introduced species was correlated with the management carried out in each lagoon, i.e. the higher proportion of native species found in the Tancada lagoon, even in the fresher period (September), reflects a lower perturbation of the fish community by the water management. In summary, in agreement with other studies on coastal ecosystems suffering from hydrological alterations (Flower 2001, Gordo and Cabral 2001, Poizat *et al.* 2004, Pérez-Ruzafa *et al.* 2005b, Badosa *et al.* 2007, Cañedo-Argüelles *et al.* 2012), our data confirm that the small-fish community composition of the Ebro Delta coastal lagoons was affected by the altered hydrological pattern.

The hydrological alteration was also responsible for structuring the small-bodied fish community through

salinity. The alternation of a freshwater input period with a non-input period creates a wide salinity range in the coastal lagoons that fish have to cope with. Fish have the capacity to keep their plasma osmotic concentration constant through osmoregulation (Rigal *et al.* 2008). However, this capacity differs between species, some being more or less tolerant to salinity variations (Blaber 1997, Marshall and Elliott 1998, Rigal *et al.* 2008). The abrupt decrease in abundance of the freshwater species *Gambusia holbrooki* with increasing salinities was probably due to the fact that its upper salinity tolerance is 25 (Nordlie and Mirandi 1996). This salinity tolerance could also explain its absence in March in Tancada lagoon, where the average salinity rose to 30. Moreover, a peak of abundance of the endangered *Aphanius iberus* was observed with the decrease of the invasive *G. holbrooki*. This finding could be attributed to the interaction of these two species, as some recent studies have pointed out. While some authors suggested that a sharp decrease in *A. iberus* in the presence of the *G. holbrooki* occurs not only because of the salinity preferences but also because *G. holbrooki* is more competitive due to its higher foraging ability, especially when it outnumbered the native species (Caiola and De Sostoa 2005), Rincón *et al.* (2002) hypothesized a possible predation of *G. holbrooki* on *A. iberus* juveniles. In whatever circumstances, our observed higher presence of the endangered *A. iberus* species in more saline habitats seems to be a response aimed at avoiding competition with the invasive *G. holbrooki* (Doadrio 2001, Rincón *et al.* 2002, Caiola and De Sostoa 2005).

The high osmoregulation capacity of *Pomatoschistus microps* (Rigal *et al.* 2008) was probably the reason for its presence in all three lagoons in both periods. Although this species shows a tolerance to a wide salinity range, a peak of abundance was found around 30, thus suggesting a higher preference for euhaline environments. Although *Atherina boyeri* was also found in all lagoons and periods, this species was not selected in the GAM analysis procedure due to a lack of response of its CPUE to the salinity gradient. A possible hypothesis to explain this fact could be that *A. boyeri* is the only small-bodied fish from the Ebro Delta coastal lagoons that is a target species (Rodríguez-Climent *et al.* 2012). The period of higher salinity coincides with the professional fishing season (from October to March). Therefore, the expected increase in the abundance of this species during the period of higher salinity is offset by the effect of fishery. Although members of the Mugilidae family spawn at sea (Kottelat and Freyhof 2007, Cardona *et al.* 2008), during their juvenile stages they enter coastal lagoons looking for a protected area with abundant food that will provide all the favourable conditions for their development (Verdiell-Cubedo 2009). In these habitats, salinity has been pointed out to be a key factor in determining the distribution pattern of young mugilids (Lasserre and Gallis 1975, Cardona 2006, Mićković *et al.* 2010). The low osmoregulation capacity of the juveniles of *Liza saliens* and *Liza au-*

rata, together with their optimization of growth at high salinity levels (Cardona 2006, 2008), might explain their distribution in the present study, in which they were absent in freshwater areas and showed a preference for high salinity areas. The low abundance of *Chelon labrosus* in September was in accordance with its reduced osmoregulation capacity in freshwaters, where the species is not able to survive for more than a few months (Lasserre and Gallis 1975). Furthermore, the presence of *C. labrosus* in sites closest to the sea supported the hypothesis that marine connectivity plays an important role in the abundance of this species (Cardona *et al.* 2008). Although there is no doubt about the role played by salinity in the distribution patterns of the family, some authors have hypothesized that the high trophic overlap found among young mugilids determines their survival in low resources conditions (Gisbert *et al.* 1996). Thus, the dominance of *L. saliens* over *L. aurata* (the only other species with a preference for high salinity ranges) found in our study could be explained by a possible trophic competition (Gisbert *et al.* 1995, 1996, Cardona *et al.* 2008).

CONCLUSIONS

The present study is the first research approach describing the impact of the hydrological alteration on the fish assemblages of the Ebro Delta lagoons. It can be concluded that water management practices provoking hydrological alterations have a direct impact on the structure and dynamics of the coastal lagoons' fish assemblages. The current water management scheme developed in the Ebro Delta Natural Park is favoring the introduced species in detriment of the native ones. Of special mention was the inverse abundance trends found between the endangered *Aphanius iberus* and the invasive *Gambusia holbrooki*. This finding not only evidences two opposite salinity preferences but also corroborates previous studies that demonstrate that the invasive species affects the native one negatively. In addition, the different water management applied to each lagoon allowed us to conclude that the best management scheme regarding the fish assemblages was the one undertaken in the Tancada lagoon, characterized by less freshwater inputs during the rice cultivation period (May-December). Moreover, with salinities higher than 15-20 throughout the year, a fish community with a dominance of native species important for both commercial and conservation purposes is assured. This should be set as a key criterion in future water management schemes of the Ebro Delta coastal lagoons.

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silviarodriguezcliment@gmail.com