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TESIS DOCTORAL

Estudio de los bancos explotables de *Donax trunculus* y *Chamelea gallina* en el sector sur del golfo de Valencia y factores ambientales que influyen en su abundancia

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ABSTRACT

Sandy beaches are very dynamic and unstable environments that present a wide variety of life forms and in some areas high productivity. In the Gulf of Valencia, we can find two species of bivalves with high commercial interest, such as *Donax trunculus* and *Chamelea gallina*, which are part of the same *métier*. Gandia is a key area for the artisanal fishery of both species in the Gulf of Valencia. However, in the last decade the catches of both species have suffered a sharp decrease, which led to the closure of the fishery in June 2015. Therefore, the main objective of this thesis was to analyse the *méterier* composed by *D. trunculus* and *C. gallina* in the Gandia Fishermen's Guild work area, which covers from Tavernes de la Valldigna to Denia, until its closure, the evolution of the populations of *D. trunculus* after it, as well as the environmental factors which could affect the beds of both species and the rest of the sublittoral organisms of the sandy beach.

In Gandia, the catches, the fishing fleet and CPUE of the studied *métier*, in which *D. trunculus* dominates, suffered a sharp decrease during the period 2004-2014, becoming more abrupt from 2009 and 2010. The catches of *D. trunculus* decreased from 100 000 kg in 2004 to 1600 kg in 2014, and for *C. gallina* from 21 000 in 2007 to 800 in 2014. In addition, this fishery showed a marked seasonality with catches and fishing effort peaks in spring-summer and greater fishing pressure in the areas located south of Gandia Port. The growth parameters of *D. trunculus* obtained showed some differences to other studies from different geographical areas, highlighting that the values obtained for asymptotic length and growth constant in 2015 were lower than those obtained 25 years earlier in the same region. In addition, the population of *D. trunculus* showed size differential distribution between depths and a period of recruitment in summer.

The benthic fauna showed a clear bathymetric zonation explained primarily by the mean grain size of the sediment and the amount of organic matter. A community was found dominated by a larger mean grain size and lower organic matter content, in the shallowest and turbulent zone, in which the bivalve *D. trunculus* dominated in biomass. And another community related to the higher content of organic matter

in the sediment and a finer grain size present in greater depth, in which the bottom is more stable, dominated by deposit feeder polychaetes and in which *C. gallina* develops. The macrobenthic fauna showed a temporary pattern with highs abundances in summer that was not so evident in *D. trunculus* and *C. gallina* due to its exploitation.

The density and biomass values obtained two years after the closure, not only showed that there had not been a population recovery, but also that the situation had worsened. Therefore, the closure of the fishery had not had the desired effect in the short term. The overexploitation suffered in *D. trunculus* and *C. gallina* beds could have aggravated the adverse effects caused by other factors, thus hindering the recovery of populations.

Finally, it is important to note that the deficiencies in the fishery's management plans led to overexploitation and that the closure of the already collapsed fishery did not allow its recovery in the short term.

RESUMEN

Las playas arenosas son ambientes muy dinámicos e inestables que presentan una gran variedad de formas de vida y en algunas zonas una gran productividad. En el golfo de Valencia, encontramos dos especies de bivalvos con elevado interés comercial, como son *Donax trunculus* y *Chamelea gallina*, los cuales forman parte del mismo *métier*. Gandia es un área clave para la pesquería artesanal de ambas especies en el golfo de Valencia. No obstante, en la última década las capturas de ambas especies han sufrido una fuerte disminución, lo que condujo al cierre de la pesquería en junio de 2015. Por ello, el objetivo principal de la presente tesis fue analizar la pesquería del *métier* compuesto por *D. trunculus* y *C. gallina* en el área de trabajo de la Cofradía de Pescadores de Gandia, que abarca desde Tavernes de la Valldigna a Denia, hasta su cierre, la evolución de las poblaciones de *D. trunculus* tras el mismo, así como los factores ambientales que pudieron afectar a los bancos de ambas especies y del resto de organismos de la biocenosis del sublitoral de playas arenosas.

En Gandia, las capturas, la flota pesquera y CPUE del *métier* estudiado, en el que domina *D. trunculus*, sufrieron una fuerte disminución durante el período 2004-2014, volviéndose más abrupta a partir de los años 2009 y 2010. Las capturas de *D. trunculus* disminuyeron de 100 000 kg en 2004 a 1600 kg en 2014, y de 21 000 en el 2007 a 800 en el 2014 para *C. gallina*. Además, esta pesquería presentó una marcada estacionalidad con incrementos de captura en primavera-verano y una mayor presión pesquera en las zonas ubicadas al sur del puerto de Gandia. Los parámetros de crecimiento de *D. trunculus* obtenidos mostraron algunas diferencias con los estudios de otras áreas geográficas, destacando que los valores obtenidos para la longitud asintótica y la constante de crecimiento en el año 2015 fueron más bajos que los obtenidos 25 años antes en la misma región. Además, la población de *D. trunculus* mostró una distribución diferencial de tamaños entre las profundidades y un período de reclutamiento en verano.

La fauna bentónica mostró una clara zonación batimétrica definida fundamentalmente por el tamaño medio del sedimento y la cantidad de materia orgánica. Se encontró una comunidad dominada por un mayor tamaño de grano

medio y menor contenido en materia orgánica, en la zona más somera y turbulenta, en la que dominó en biomasa el bivalvo *D. trunculus*. Y otra comunidad sujeta al mayor contenido de materia orgánica en el sedimento y un tamaño de grano más fino, presente a mayor profundidad, en la que los fondos son más estables, dominada por poliquetos depositívoros y en la que se desarrolla *C. gallina*. La macrofauna bentónica siguió una pauta temporal con máximos de abundancia en verano que no fue tan patente en *D. trunculus* y *C. gallina* debido a su explotación.

Los valores de densidad y biomasa obtenidos dos años después del cierre, no solo mostraron que no había habido una recuperación de la población, sino también que la situación había empeorado. Por lo tanto, el cierre de la pesquería no tuvo el efecto deseado a corto plazo. La sobreexplotación que sufrieron los bancos de *D. trunculus* y *C. gallina* podría haber agravado los efectos adversos provocados por otros factores dificultando así la recuperación de las poblaciones.

Finalmente, es importante destacar que las deficiencias en los planes de manejo de esta pesquería propiciaron la sobreexplotación y que el cierre de la pesquería ya colapsada no permitió su recuperación a corto plazo.

RESUM

Les platges arenoses són ambients molt dinàmics i inestables que presenten una gran varietat de formes de vida i en algunes zones una gran productivitat. En el golf de València, trobem dues espècies de bivalves amb elevat interès comercial, com són *Donax trunculus* i *Chamelea gallina*, els quals formen part del mateix *métier*. Gandia és una àrea clau per a la pesqueria artesanal de les dues espècies al golf de València. No obstant això, en l'última dècada les captures de les dues espècies han patit una forta disminució, el que va conduir al tancament de la pesqueria al juny de 2015. Per això, l'objectiu principal de la present tesi va ser analitzar la pesqueria del *métier* compost per *D. trunculus* i *C. gallina* a l'àrea de treball de la Confraria de Pescadors de Gandia, que va des de Tavernes de la Valldigna a Dénia, fins al seu tancament, l'evolució de les poblacions de *D. trunculus* després del mateix, així com els factors ambientals que van poder afectar els bancs d'ambdues espècies i de la resta d'organismes de la biocenosi del sublitoral de platges arenoses.

A Gandia, les captures, la flota pesquera i CPUE del *métier* estudiat, en què domina *D. trunculus*, van patir una forta disminució durant el període 2004-2014, tornant-se més abrupta a partir dels anys 2009 i 2010. Les captures de *D. trunculus* van disminuir de 100 000 kg en 2004 a 1600 kg al 2014, i de 21 000 en 2007 a 800 al 2014 per a *C. gallina*. A més, aquesta pesqueria va presentar una marcada estacionalitat amb increments de captura en primavera-estiu i una major pressió pesquera a les zones situades al sud del port de Gandia. Els paràmetres de creixement de *D. trunculus* obtinguts van mostrar algunes diferències amb els estudis d'altres àrees geogràfiques, destacant que els valors obtinguts per a la longitud asimptòtica i la constant de creixement en l'any 2015 van ser més baixos que els obtinguts 25 anys abans a la mateixa regió. A més, la població de *D. trunculus* va mostrar una distribució diferencial de grandàries entre les profunditats i un període de reclutament a l'estiu.

La fauna bentònica va mostrar una clara zonació batimètrica definida fonamentalment per la grandària mitja del sediment i la quantitat de matèria orgànica. Es va trobar una comunitat dominada per una major grandària de gra

mitjà i menor contingut en matèria orgànica, a la zona més somera i turbulenta, en la que va dominar en biomassa el bivalve *D. trunculus*. I una altra comunitat subjecta al major contingut de matèria orgànica en el sediment i una mida de gra més fi, present a més profunditat, en la qual els fons són més estables, dominada per poliquets depositívoros i en la que es desenvolupa *C. gallina*. La macrofauna bentònica va seguir una pauta temporal amb màxims d'abundància a l'estiu que no va ser tan patent en *D. trunculus* i *C. gallina* a causa de la seva explotació.

Els valors de densitat i biomassa obtinguts dos anys després del tancament, no només van mostrar que no hi havia hagut una recuperació de la població, sinó també que la situació havia empitjorat. Per tant, el tancament de la pesqueria no va tenir l'efecte desitjat a curt termini. La sobreexplotació que van patir els bancs de *D. trunculus* i *C. gallina* podria haver empitjorat els efectes adversos provocats per altres factors dificultant així la recuperació de les poblacions.

Finalment, és important destacar que les deficiències en els plans de gestió d'aquesta pesqueria van propiciar la sobreexplotació i que el tancament de la pesquera ja col·lapsada no va permetre la seva recuperació a curt termini.

ÍNDICE

GLOSARIO

CCA	Análisis de correspondencia canónica ("*Canonical Correspondence analysis*")
CHJ	Confederación Hidrográfica del Júcar ("*Júcar Hydrographic Confederation*")
Chl_*a*	Clorofila *a* ("*Chlorophyll a*")
Chl_*b*	Clorofila *b* ("*Chlorophyll b*")
CPUE	Capturas por unidad de esfuerzo ("*Catch-per-unit-effort*")
DIN	Nitrógeno inorgánico disuelto ("*Dissolved Inorganic Nitrogen*")
DFV	Velocidad de caída adimensional
GT	Arqueo bruto ("*Gross tonnage*")
H	Altura de la onda
HPLC	Cromatografía líquida de alta resolución ("*High Performance Liquid Chromatography*")
K	Constante de crecimiento ("*Growth constant*")
K-S	Test de bondad de ajuste Kolmorov-Smirnov ("*Kolmogorov-Smirnov goodness of fit test*")
L	Longitud de concha ("*Shell length*")
L_{∞}	Longitud asintótica (*"Asymptotic length"*)
LOA	Eslora total ("*Overall length*")
M	Masa total ("*Total mass*")
MGS	Tamaño medio de grano ("*Mean grain size*")
MK	Tests de Man-Kendall
OGP	Índice de eficiencia del crecimiento ("*Overall growth performance index*")
OM	Materia orgánica ("*Organic matter*")
RTR	Rango de marea relativo
SAIH	Sistema automático de información hidrológica de la Confederación Hidrológica del Júcar ("*Automatic Hydrological Information System*")

SCOR	Comité científico de investigación oceánica (“*Scientific Committee on Oceanic Research*”)
SS	Sólidos suspendidos (“*Suspended solids*”)
TP	Fósforo total (“*Total phosphorus*”)
VBGF	Modelo de crecimiento de von Bertalanffy (“*The von Bertalanffy growth model*”)
WWTP	Planta de tratamiento de aguas residuales, EDAR, (“*Waste water treatment plant*”)
λ	Longitud de onda
ϕ'	Índice estándar de crecimiento (*“Growth performance index”)*

Capítulo 1

Introducción

1. INTRODUCCIÓN

1.1 PLAYAS ARENOSAS

Las playas arenosas expuestas dominan gran parte de la costa en todos los continentes y se caracterizan por ser un ambiente dinámico, donde el agua y el sedimento están en continuo movimiento. Se define como playa a la acumulación de sedimentos no consolidados que se sitúa entre la línea de máximo alcance del oleaje, en temporal y pleamar viva, y una profundidad que corresponde a la zona donde deja de producirse un movimiento activo de los sedimentos debido a la acción del oleaje (Ley et al., 2007). Comprende por lo tanto una zona sumergida, entre la línea de marea baja hasta donde dejan de percibirse los efectos de las olas sobre la estabilidad del fondo marino (generalmente entre 5 y 20 m de profundidad), otra intermareal, y una última de playa seca (Ley et al., 2007). Aunque las playas constituyen un único sistema geomorfológico, pueden diferenciarse 2 sistemas ecológicos en ellas: uno marino habitado por organismos marinos y fuertemente influenciado por la energía de las olas y otro dunar terrestre poblado por organismos terrestres y fuertemente influenciados por la energía del viento (McLachlan y Defeo, 2018). El oleaje, el viento y las corrientes son los responsables de transportar la arena, tanto en períodos de tormenta como de calma, en la playa sumergida y en la zona seca. Las playas arenosas son ambientes que se ven definidos principalmente por las características de la arena, las olas y las mareas, las cuales determinan su morfodinámica (McLachlan et al. 1995; McLachlan et al. 1996; Short, 1996; Defeo y McLachlan, 2005). El dinamismo que presentan estas playas hace que las especies marinas que las habitan sean altamente móviles y capaces de enterrarse, por lo que a simple vista pueden parecer desiertos marinos. Sin embargo, albergan una gran cantidad de vida, ya sea microscópica o macroscópica, siendo a menudo ecosistemas muy productivos (McLachlan, 1983; Smith y Smith, 2001; Defeo y McLachlan, 2005).

1.2 CARACTERIZACIÓN FÍSICA

1.2.1 Componentes estructurales de las playas

La arena es el elemento principal que define estos sistemas. Su fuente fundamental son los sistemas fluviales que transportan sedimentos originados por erosión de sistemas terrestres, aunque también, en menor medida, procede de fuentes marinas tanto biogénicas como de la erosión de acantilados costeros (Cantera et al., 1994). Estas arenas pueden ser de cuarzo o carbonatadas, diferenciándose además en su densidad y en su forma, siendo las partículas carbonatadas más irregulares. La característica más importante de las arenas es el tamaño de sus partículas, clasificándose generalmente en función de la escala Wenworth y pueden abarcar diámetros desde 63 µm a 2 mm. Otras características del sedimento son la porosidad, volumen de espacio libre en el volumen total de sedimento, y permeabilidad, tasa de flujo de agua a través de la arena. Generalmente cuanto más fina es la arena mayor será su porosidad, aunque la permeabilidad se incrementa de arenas finas a gruesas. La penetrabilidad de la arena, relacionada con el tamaño de partícula, porosidad y otros factores, juega un papel importante para la macrofauna, ya que determina la facilidad de excavación por parte de estos organismos (McLachlan y Defeo, 2018).

La deposición de sedimento se ve condicionada principalmente por el oleaje, pero también intervienen las mareas y la topografía (Short, 1996). Las olas, fundamentalmente las ondas de gravedad de superficie, y las corrientes que inducen constituyen las fuerzas impulsoras que gobiernan la mayoría de los procesos que ocurren en las playas arenosas. Las olas son generadas por la tensión del viento en contraposición a la tensión superficial. La energía del viento se transfiere a la masa de agua y finalmente a la playa, dónde pueden disiparse cantidades ingentes de energía. La acción de las olas penetra en la columna de agua hasta una profundidad de aproximadamente la mitad de la longitud de onda (λ). A medida que el mar profundo disminuye hacia tierra, hay un punto donde la ola toca el fondo del mar (Short, 1999; Masselink y Hughes 2003). Según la definición de Short (1999), este punto es el límite inferior de la playa -la zona cercana a la costa-. Las olas comienzan a compactarse, y la relación entre la altura de la onda (H) y la longitud de la onda (H: λ) aumenta, es el proceso conocido como asomeramiento. En esta zona comenzará el transporte de material además del de energía. Cuando la relación H: λ alcanza 1/7, la onda colapsa. Este es el punto de ruptura donde comienza la zona de surf. La última fase de la acción de las olas es donde la playa es cubierta y expuesta alternativamente por el barrido de las olas, la zona de swash. El rango mareal en las costas abiertas puede variar desde los pocos centímetros, como es el caso del Mediterráneo, hasta más de 10 m. Se considera micromareal si el rango de marea es menor de 2 m, mesomareal entre 2-4 m y macromareal cuando es mayor a 4 m (Davies y Moses, 1964). En las playas arenosas es más importante para la ecología el rango de mareas que el tipo de marea.

1.2.2 Morfodinámica de playas arenosas

En las playas arenosas podemos encontrar un rango de estados morfodinámicos como resultado de la interacción entre la arena, las olas y las mareas (Short, 1996; Defeo et al., 2009). El modelo conceptual propuesto por McLachlan y Defeo (2018) basado en la velocidad de caída adimensional (DFV) y el rango de marea relativo (RTR), clasifica las playas en tres grandes grupos: reflectivas, intermedias y disipativas.

En el caso de las playas micromareales dominadas por el oleaje existen 6 tipos diferentes, siendo los más extremos las playas reflectivas y las disipativas, mientras que los 4 restantes se considerarían estados intermedios (Wright y Short, 1984; Short, 1996; Defeo et al., 2009).

Las playas reflectivas se caracterizan por presentar condiciones tranquilas, un sedimento grueso con almacenamiento en la costa y en la zona intermareal, la presencia de zonas profundas en la orilla y por la inexistencia de la zona de surf, rompiendo las olas directamente sobre la orilla. Además, en este tipo de playa es típico encontrar cúspides u ondulaciones causadas por las olas de borde. En estas playas por lo general el rango mareal es pequeño y se encuentran en zonas dónde las olas son inferiores a 0.5 m y las arenas más gruesas que 400 µm (Short, 1996; Defeo y McLachlan, 2005; Defeo et al., 2009; McLachlan y Defeo, 2018).

Por el contrario, las playas disipativas presentan una pendiente suave con amplia zona de swash y una zona de surf extensa donde se produce un almacenamiento de arena en múltiples bancos. En este tipo de playas el oleaje no rompe directamente sobre la orilla, si no que la energía de las olas se disipa en la zona de surf. Aunque a lo largo de la costa el swash sea suave, usualmente se presentan ondas infragravitatorias. Este tipo de playas se suelen presentar en zonas donde no se superan los 2 m de altura de ola y las arenas presentan tamaños más finos de 200 µm (Short, 1996; Defeo y McLachlan, 2005; McLachlan y Defeo, 2018).

Entre las playas reflectivas y disipativas podemos encontrar cuatro tipos de playas intermedias. Este tipo de playas se caracterizan por presentar una elevada variabilidad temporal. El almacenamiento de arenas se produce entre la zona de surf y la playa, con bancos de arena variables y corrientes de resaca que dominan en la zona de surf, presentando arenas de tamaños de finos a medios (Short, 1996; Defeo y McLachlan, 2005).

En el caso de las zonas con marea mayor a 2 m, la situación se vuelve más compleja. A mayor rango de mareas, el impacto de los procesos de la zona de swash y de surf se reducen y pasan a ser más importantes las olas de asomeramiento, lo que hace que se suavice el perfil de la playa en la zona intermareal. Cuando el rango de marea se encuentra entre 3 y 12 m las playas pueden presentar tres estados. El primero de ellos reflectivo con terraza de bajamar en el que prevalecen las olas de menor altura y la pendiente es empinada. En el intermedio aumenta la altura de ola y en la terraza de bajamar se forman barras y surcos y canales de retorno. Finalmente, el estado ultradisipativo se conforma cuando el rango de marea es mayor a 7 m, y las olas mayores a 2 m, volviéndose las playas planas y con arena fina (McLachlan y Defeo, 2018).

1.2.3 Las playas del sector sur del golfo de Valencia

Las playas arenosas son el elemento morfológico más abundante del golfo de Valencia. Si bien existen diferencias sustanciales entre algunas de ellas, todas están sujetas a características comunes que están determinadas por la existencia de condiciones generales similares: i) la existencia de un rango de mareas muy pequeño; ii) baja energía de las olas la mayor parte del tiempo; iii) el origen primario de los sedimentos de la mayoría de las playas procede de contribuciones fluviales; y iv) la acción humana es muy importante y ha determinado su evolución reciente (especialmente durante el último siglo) a causa del cambio del ritmo y volumen de sedimentos suministrados al mar (Pardo-Pascual y Sanjaume, 2019).

Una de las características que mejor define a las playas es su gran dinamismo temporal y espacial. Aunque todos los cambios, en cualquier escala, tienen alto interés, aquellos que ocurren en rangos temporales de corto o mediano plazo (días, años, décadas) son especialmente importantes desde un punto de vista social ya que afectan al uso humano de la playa como recurso. En la Comunidad Valenciana, las playas son el principal recurso de un territorio en el que el turismo concentra el 13.2 % del PIB regional y 14.4 % del empleo (IMPACTUR, 2015). Por esta razón, las alteraciones experimentadas por las playas pueden tener, en algunos casos, un impacto social y económico de gran importancia. Esto explica, a su vez, la gran cantidad de intervenciones directas que se han realizado y se siguen llevando a cabo en muchas de las playas del golfo de Valencia (Pardo-Pascual y Sanjaume, 2019).

Las playas del sector sur del golfo de Valencia se caracterizan por lo tanto por su marcado carácter micromareal (< 30 cm) y una morfodinámica generalmente intermedia-disipativa (Escrivá, 2013, Pardo-Pascual y Sanjaume, 2019). En las últimas décadas algunas playas se han vuelto más intermedias debido a causas antropogénicas. La escasez de sedimentos determina un aumento de la pendiente en el perfil cercano a la costa y la zona de surf se vuelve más estrecha (Pardo-Pascual y Sanjaume, 2019). En esta zona, las corrientes dominantes circulan de norte a sur (CEDEX, 1997; Millot, 1999), y la altura significativa de ola no suele alcanzar el metro de altura (Puertos del Estado, 2019). En la playa sumergida, las fracciones de sedimento que dominan son las arenas finas, con tamaño de partículas que varía de 125 a 250 µm, incrementando las fracciones más finas en profundidad (Ramón, 1993; Escrivá, 2013).

1.3 ECOLOGÍA DE LAS PLAYAS ARENOSAS

Las playas arenosas albergan una gran variedad de organismos marinos en las que se pueden encontrar la mayoría de *phyla* de los invertebrados, tanto intersticiales (micro y meiofauna) que viven entre los granos de sedimento como organismos de mayor tamaño (> 500 µm) a los que denominamos macrofauna. No presentan organismos sésiles fijados al sustrato y por lo tanto no se encuentran productores primarios macroscópicos. Dada la inestabilidad del sustrato y el fuerte oleaje, los organismos presentan una elevada movilidad y una gran capacidad para excavar rápidamente (McLachlan y Defeo, 2018). Estos organismos pueden ocupar tanto la superficie del sedimento (epifauna) como vivir entre el sedimento (infauna) (Smith y Smith, 2001; Augier, 2007). Aunque en las playas arenosas habitan organismos de todos los taxones, generalemente los moluscos, crustáceos y poliquetos son los más abundantes en este tipo de playas (Martí et al., 2007; McLachlan y Defeo, 2018).

1.3.1 Adaptaciones de los organismos al ambiente de las playas

El sustrato arenoso no proporciona una superficie sólida para la fijación de los organismos, además, debido a la inestabilidad que presenta y a la acción de las olas, los organismos de estos tipos de playas requieren ciertas adaptaciones (Smith y Smith, 2001; McLachlan y Defeo, 2018). Entre estas adaptaciones destacamos las relacionadas con la locomoción, elección del hábitat y nutrición, pero también existen otras como la adaptación a los ritmos de mareas, las respuestas sensoriales y orientación, la respiración, la reproducción, la agregación y el gregarismo, la evitación de los depredadores y la plasticidad fenotípica (Brown, 1996; McLachlan et al., 1996; Scapini, 1997; McLachlan y Defeo, 2018).

En las playas de arena la locomoción juega un papel mucho más importante en la vida de los animales que en otros tipos de costa. Encontramos varias formas diferentes de locomoción: aunque el único modo común a toda la macrofauna de una playa de arena es la excavación, podemos encontrar organismos nadadores, los que surfean sobre la arena, los que se arrastran y los que saltan o corren. En el proceso de excavación distinguimos dos fases, la primera es la penetración en el sustrato y la segunda son los movimientos de excavación. La penetración se ve facilitada por las propiedades tixotrópicas de los sustratos particulados. La velocidad de la penetración varía en función del tamaño de grano del sedimento, siendo más lenta en playas reflectivas y aumentando la velocidad hacia las playas disipativas. De hecho, es en las playas barridas por el surf donde el peligro de ser desplazado siempre está presente, por lo que es obligatoria la penetración en solo unos segundos, aplicando mayor fuerza. Los movimientos de excavación se basan en la aplicación alternativa de dos tipos de anclaje, cada uno de los cuales permite moverse a otra parte del cuerpo en la arena. En el primer anclaje se produce la dilatación de la región posterior del cuerpo mientras la parte anterior penetra, en el segundo se produce la situación contraria. Para permitir tales cambios en la forma, los animales de cuerpo blando han desarrollado característicamente grandes cavidades llenas de líquido que afectan tanto los cambios en la forma como la transferencia de las fuerzas musculares. Las diversas etapas del ciclo de excavación están, por lo tanto, acompañadas por cambios de presión en el celoma o en el hemocele (Trueman y Brown, 1985). El surfeo, también es la forma preferencial en los organismos sometidos a la acción del swash, el surf e incluso por la acción del oleaje sobre el fondo antes de la rompiente, siendo muy común entre los organismos de playas disipativas y perdiendo importancia en las reflectivas o en las costas relativamente más protegidas (McLachlan y Defeo, 2018).

La selección del hábitat entre la fauna psammófila suele realizarse al final del desarrollo larvario o en la metamorfosis, pero también puede hacerse en los adultos repetidamente a medida que su comportamiento migratorio o condiciones impredecibles (como las tormentas) transportan a los animales a sitios menos favorables. Los factores que pueden desencadenar las respuestas suelen ser las corrientes de agua, los cambios en los patrones de luz polarizada y en la presión hidrostática. Sin embargo, los dos factores más importantes que definen el hábitat de la playa para la macrofauna bentónica son la textura de la arena (Lastra y McLachlan, 1996) y los flujos hidrodinámicos generados en la playa. En áreas afectadas por la descarga de agua dulce, como cerca de las desembocaduras de los ríos, la salinidad también puede ser un factor importante (Fanini et al., 2012). El tamaño de partícula de arena es quizás el factor más crítico, puesto que ejerce una influencia considerable en la excavación (McLachlan y Defeo, 2018).

La macrofauna de playas arenosas ha adaptado su alimentación tanto a las condiciones hidrodinámicas como a la inestabilidad. Así la ausencia de macrófitos elimina la presencia de herbívoros, dominando los filtradores y los carroñeros, aunque también podemos encontrar depredadores carnívoros. Los filtradores, más abundantes en los ambientes más turbulentos del sublitoral cuentan con un suministro más constante pero más pobre (p.ej. *Donax* ingiere una amplia gama de tamaños de partículas y puede hacer uso de microalgas, de partículas de detritus, de la espuma resultante de la descomposición del fitoplancton y las bacterias, aunque no todos estos se eliminan e ingieren en la misma medida) (Brown et al., 1989). Los carroñeros cuentan con una disponibilidad altamente errática de su suministro de alimentos y sobre todo de la calidad alimenticia de esa carroña (p.ej. algunos misidáceos son a la vez suspensívoros y depositívoros). Habitualmente, aceptarán una amplia variedad de alimentos, y, también, se convertirán en depredadores si es necesario (McLachlan y Defeo, 2018).

En las playas de arena son habituales que los organismos presenten unos sistemas sensoriales muy bien desarrollados que permiten respuestas como la reotaxis (Sigl y Laforsch, 2016), la quimiotaxis (Harris et al., 2019), la escototaxis y la fototaxis (Scapini et al., 2019), que permitirán adaptarse a los ritmos de mareas, buscar alimento u orientarse en un ambiente altamente inestable.

En la zona intermareal de la playa pueden ser cruciales adaptaciones para hacer frente a la desecación, y a las variaciones de salinidad (McLachlan y Defeo, 2018).

En playas de arenas protegidas (y particularmente en arenas lodosas), las adaptaciones respiratorias son necesarias para tolerar tensiones de oxígeno pobres o decrecientes, especialmente durante la marea alta, mientras que, en playas de arena expuestas al oleaje, las tensiones de oxígeno son habitualmente altas a una profundidad de un metro o más abajo de la superficie de la arena, de modo que los organismos no están sometidos a la hipoxia (McLachlan y Defeo, 2018).

Uno de los problemas que enfrentan los animales de las playas de arena es cuándo reproducirse; puede ser crítico que coincida la producción de gametos, la cópula o la liberación de crías con una fase particular del ciclo de las mareas. Por lo tanto, no es sorprendente que los ritmos quincenales o mensuales de actividad reproductiva sean comunes (Marques et al., 2003). La mayoría de los organismos son estrategas de la r, como en el caso *Donax*, que producen un gran número de gametos, los ovocitos son pequeños y contienen poca yema; la fertilización es externa y se producen larvas planctotróficas de natación libre. A menudo hay una temporada de reproducción prácticamente ilimitada, aunque presentan picos de actividad reproductiva marcados (Negreiros-Fransozo et al., 2002; Delgado y Defeo, 2008).

Los organismos psamófilos móviles tienden a agregarse, en contraste con la distribución típicamente no agregada de los invertebrados que viven en tubos en las costas más protegidas. Tales agrupaciones ciertamente pueden surgir debido a la acción de las corrientes de agua, particularmente en aquellos animales que surfean, y esto puede intensificarse en el caso de carroñeros carnívoros entorno a las acumulaciones de los cadáveres. Además, pueden aparecer picos de agregación coincidentes con los periodos de abundancia máxima de hembras reproductivas. Por lo tanto, aunque las fuerzas físicas pueden desempeñar un papel importante (al menos en algunas especies), otros fenómenos (como señales visuales o estímulos químicos, incluidas las feromonas) también pueden ser importantes (McLachlan y Defeo, 2018).

Para hacer frente a la depredación, encontramos mecanismos como la excavación profunda, la inactividad diurna por parte de los crustáceos semiterrestres, o respuestas de escape, mediante la natación o los saltos como en el caso de *Donax* (Ansell y Trevallion, 1969).

El comportamiento de los organismos en entornos variables debe ser flexible para hacer frente a las condiciones cambiantes. Por eso, los patrones de comportamiento en la macrofauna de playas arenosas no son rígidos, sino que se modifican de acuerdo con las circunstancias físicas y biológicas, lo que resulta en diferencias de comportamiento entre poblaciones de la misma especie en diferentes ambientes (Laudien et al., 2003). Esto sugiere una plasticidad en los organismos (Brown, 1996), es decir, el mismo genotipo puede mostrar diferentes fenotipos en diferentes condiciones ambientales.

1.3.2 Estructura y funcionamiento de las playas arenosas

La distribución, la abundancia y la riqueza de las comunidades macrobentónicas de playas arenosas se ven condicionadas por variables físico-químicas como la temperatura, pH, salinidad, oxígeno disuelto, los nutrientes y los relacionados con la morfodinámica de la playa tales como la pendiente, tamaño de grano del sedimento, hidrodinamismo, etc., así como por variables nutricionales y factores biológicos (ciclos de vida, mortalidad, depredación y competitividad por recursos, entre otros) (Hewitt et al., 1997; Brazeiro, 2001; Defeo y McLachlan, 2005; Lastra et al., 2006; Rodil et al., 2007; Ortega Cisneros et al., 2011). Sin embargo, la teoría ecológica de las playas arenosas establece que las características físicas controlan la estructura de la comunidad macrobentónica, excepto en las playas más disipativas (Ortega Cisneros et al., 2011).

Muchas playas, especialmente las disipativas, soportan una rica biota donde los bivalvos son un componente importante, especialmente en biomasa (McLachlan et al., 1996). Se ha demostrado que la riqueza, abundancia y biomasa de especies aumenta de playas reflectivas a disipativas (Defeo y McLachlan, 2005; McLachlan, 2001). La menor riqueza de especies en playas reflectivas se puede deber al incremento de tamaño de grano del sedimento, descenso de la energía de las olas, o una combinación de ambas (McLachlan et al., 1995). Las variaciones en las propiedades morfodinámicas han sido utilizadas para desarrollar teorías ecológicas que expliquen los patrones de la composición de la macrofauna en playas arenosas (Defeo y Gómez, 2005). McLachlan (2001) definió tres paradigmas para comprender los factores que controlan los patrones de riqueza de macrofauna bentónica a gran escala en las playas arenosas:

- La hipótesis autoecológica: en ambientes en los que los factores físicos proporcionan el control ambiental del sistema, son las características autoecológicas de las especies las que controlarían su presencia y abundancia, no siendo relevantes los aspectos sinecológicos.

- Los modelos morfodinámicos de las playas: dado que la morfodinámica de la playa está vinculada al oleaje, marea y tamaño de grano del sedimento, son estas características las que estarían explicando la distribución y abundancia de los organismos.
- Riqueza y abundancia acopladas con el estado de la playa: en la zona intermareal, la riqueza y abundancia se incrementan desde las playas reflectivas micromareales a las disipativas macromareales.

Para explicar estos patrones, McLachlan (2001) además formuló otras hipótesis:

- Latitud de la playa: las playas tropicales son las que presentan mayor número de especies.
- Extensión de la playa: las playas con mayor longitud soportan mayores abundancias por las condiciones más favorables del oleaje y su baja probabilidad de erosión durante las tormentas.
- Control de la arena: el tamaño del grano de arena puede controlar la riqueza de especies. En arenas gruesas, asociadas a playas reflectivas, la mayoría de especies presentan una menor capacidad de excavación. Además, los organismos pequeños no se encuentran en dicho tipo de arenas debido al triturado y abrasión que sufren en ella. Las arenas de finas a medias parecen ser óptimas para la mayoría de la fauna.
- Swash: La hipótesis de exclusión de swash (McLachlan et al., 1993) explica que el ambiente de swash en playas disipativas es lo suficientemente adecuado para permitir mantener poblaciones de macrofauna viables, puesto que el periodo de inundación es más prolongado y se producen flujos laminares de agua. A medida que el tipo de playa cambia hacia reflectiva, el swash se vuelve más inhóspito, con periodos de inundación mucho más cortos y flujos de agua

turbulentos que terminarán excluyendo cada vez más especies de fauna hasta quedar solo las formas supralitorales.

1.3.3 Zonación de las playas de arena

Debido a los fuertes gradientes que presentan las playas de arenas, es normal encontrar zonaciones en las comunidades macrobentónicas. La zonación puede verse influenciada en algunos casos por la competencia interespecífica, pero principalmente es el resultado de las respuestas al entorno físico (McLachlan y Defeo, 2018).

Los modelos de zonación más destacados son los que fueron propuestos por Dahl (1952) y Salvat (1964). Dahl (1952) estableció una zonación en función de los crustáceos presentes, clasificándose en tres zonas, la franja subterrestre, la zona mediolitoral y la franja sublitoral. Por otra parte, la clasificación llevada a cabo por Salvat (1964) distinguió cuatro zonas en función del nivel de humedad de la arena: la zona seca, la zona de retención, la de resurgencia y de saturación. No obstante, las especies presentes en las playas arenosas no muestran unos límites definidos, presentando gran variabilidad y superposiciones con otras especies. En las playas arenosas cada especie responde de manera diferente a los gradientes físicos, presentando las distribuciones de especies variaciones supradiarias, por lo que no se puede establecer una zonación muy precisa. Las playas reflectivas suelen presentar menos especies y zonas que las disipativas. Son numerosos los estudios que respaldan la separación en tres zonas, aunque no se pueden considerar fijas ya que pueden variar en función del tipo de playa, reconociéndose hasta 4 zonas en playas disipativas anchas (Defeo y McLachlan, 2005). Generalmente las costas se dividen en zona supralitoral, litoral e infralitoral (o sublitoral) (Figura 1.1), distinguiéndose en función de la fauna característica (Smith y Smith, 2001).

La zona supralitoral se caracteriza por presentar arena seca que se mueve por la acción del viento. En ella podemos encontrar crustáceos que respiran aire fuera de la zona de swash, pero que regresan a ella para alimentarse y reproducirse. Además de insectos, que son un componente importante de la fauna, también se encuentran anfípodos Talitridae, isópodos Oniscidae (*Tylos*), cangrejos Ocypodidae e isópodos

Cirolanidae del género *Excirolana* (Brazeiro y Defeo, 1996; McLachlan y Defeo, 2018).

En la zona litoral la arena se presenta húmeda y es lavada en cada marea, además muestra una energía de swash moderada. En esta zona encontramos macrofauna intermareal, isópodos entre los que se incluyen los Cirolanidae (incluyendo algunas especies del género *Excirolana*), Haustroriidae y otros anfípodos, poliquetos Spionidae como *Scolelepis*, Opheliidae como *Euzonus* y moluscos (Brazeiro y Defeo, 1996; McLachlan y Defeo, 2018).

En la zona sublitoral la arena está saturada, presenta olas de alta energía y corrientes de fondo, además de un sustrato móvil. Podemos encontrar cangrejos Hippidae como por ejemplo *Emerita*, misidáceos, Idoteidae, Oedicerotidae, anfípodos Haustoriidae, moluscos como los bivalvos donácidos, *Tellina*, poliquetos Nephtyidae y Glyceridae (Brazeiro y Defeo, 1996; McLachlan y Defeo, 2018).

Además, en la zona sublitoral se pueden distinguir la zona de surf (también conocida como zona turbulenta interior), la zona de transición y la zona turbulenta exterior (Figura 1.1). La zona de surf tiene un límite hacia tierra, en el afloramiento de la capa freática, y otro hacia el mar en el punto de ruptura de las olas. En la zona de surf, las corrientes impulsadas por las olas hacen que el fondo sea muy inestable. La fauna, predominantemente crustáceos y moluscos altamente móviles, es más rica alrededor del límite de marea baja y desaparece en la región más turbulenta alrededor del punto de ruptura de las olas. Por el contrario, la zona turbulenta exterior, es el área donde el fondo se vuelve más estable y se desarrolla una rica macrofauna, aumentando la riqueza de especies y la biomasa e incluyendo formas que construyen madrigueras permanentes a medida que el fondo se estabiliza. Entre estas dos zonas (la zona turbulenta exterior relativamente estable y la zona de surf dinámica) se encuentra una zona de transición que corresponde aproximadamente al área entre el punto de ruptura y el límite exterior de las celdas de circulación de surf. Esta zona de transición es turbulenta e incluye elementos de la fauna tanto de la zona de surf, como especies de la zona turbulenta exterior a medida que las condiciones se vuelven más tranquilas (McLachlan y Defeo, 2018).

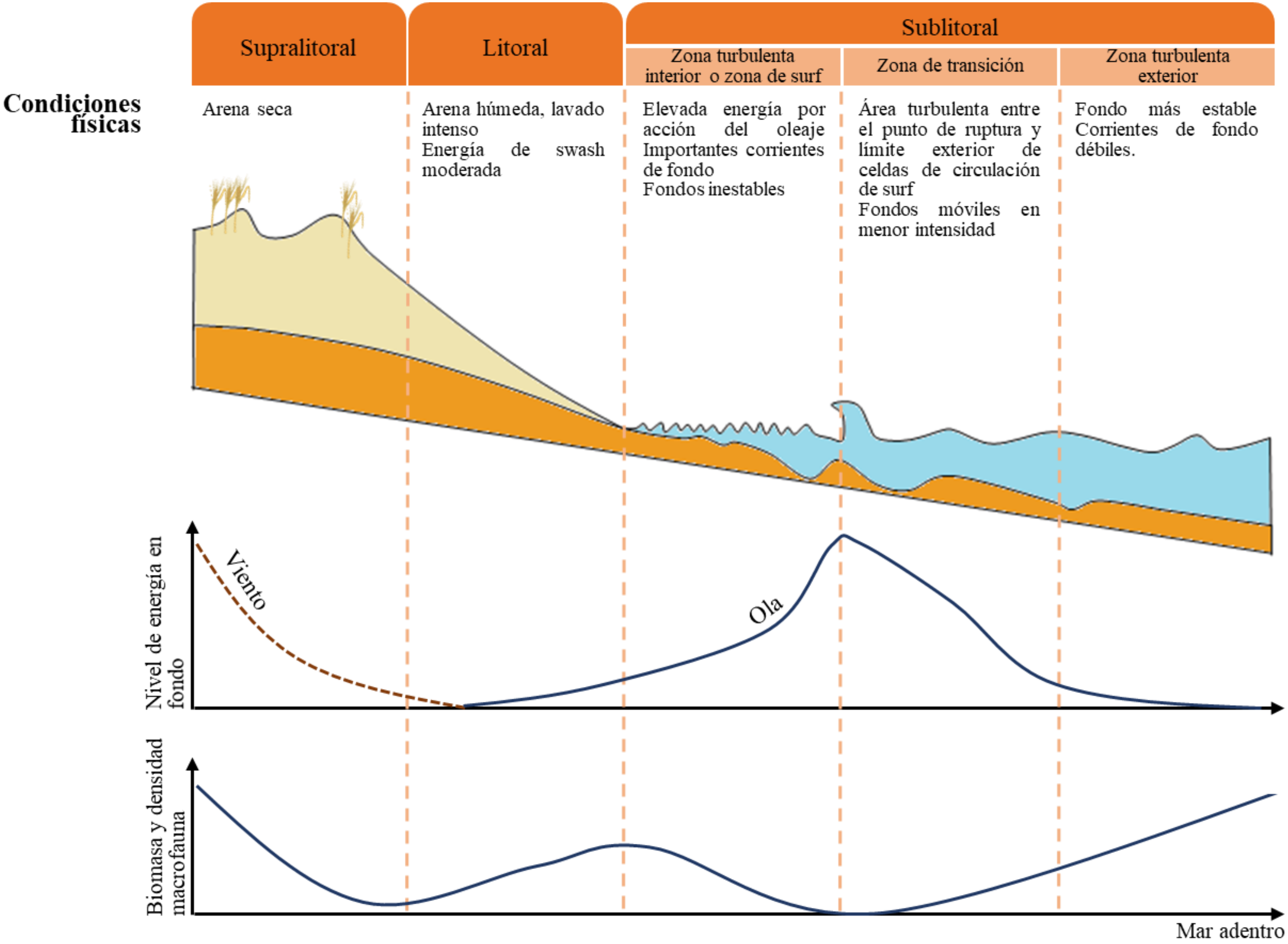

Figura 1.1. Esquema general de zonación. Adaptado de McLachlan y Defeo (2018).

1.3.4 Ecología de las playas del sector sur del golfo de Valencia

En el golfo de Valencia, Ramón y Abelló (2004) identificaron dos comunidades de macrofauna bien diferenciadas, una entre los 0.5-2.5 m de profundidad caracterizada por el bivalvo *Donax trunculus* y el cangrejo *Portumnus latipes* y otra más diversa a mayor profundidad con especies de bivalvos como *Chamelea gallina*, *Donax semistriatus*, *Spisula subtruncata*, *Acanthocardia tuberculata*, *Mactra stultorum*, gasterópodos como *Nassarius mutabilis* y cangrejos como *Liocarcinus vernalis* y *Diogenes pugilator*. En la misma zona de estudio Martí et al. (2007) confirmaron esas biocenosis y determinaron las fuertes variaciones temporales de densidad que se producen entre el invierno y la primavera-verano. Por otra parte, identificaron que los factores que determinan la distribución espacial son las condiciones meteorológicas y climáticas marinas, la hidrodinámica local, el período del año y la presión antrópica. La presión del turismo y los efluentes fluviales son los factores fundamentales que combinados con la circulación y la hidrodinámica alcanzan gran relevancia. Sin embargo, concluyeron que la distribución espacial de los organismos está más relacionada con las características de los sedimentos derivadas de la batimetría y de las condiciones hidrodinámicas. El trabajo de Escrivá (2013) refuerza esa idea, ya que observó incrementos de riqueza y diversidad en profundidad, siendo mínima la riqueza en la zona de surf e incrementándose cuando las condiciones eran menos turbulentas. Concluyendo que la distribución y abundancia de la macrofauna bentónica estuvo condicionada por la profundidad, la granulometría, así como el contenido de materia orgánica.

1.4 PESQUERÍAS EN PLAYAS ARENOSAS

Las costas arenosas albergan especies de interés pesquero, siendo las pesquerías artesanales de invertebrados la forma más común de explotación de estas costas. Estas pesquerías son de gran relevancia socioeconómica en las zonas donde se llevan a cabo, presentando en algunos casos altos ingresos por exportación (Defeo, 2003). Las pesquerías de playas arenosas son los sistemas que más se desconocen sus relaciones entre la estructura y dinámica de las poblaciones, la pesca (recreativa y comercial) y un sistema de gestión influenciado por aspectos culturales, sociales y políticos (Defeo, 2011).

A pesar de que la pesca es una actividad estrictamente regulada, está sufriendo una importante crisis, ya que los caladeros están sujetos a una fuerte explotación, lo que sugiere una mala gestión (Santos-Martín et al., 2015).

1.4.1 Tipo de pesquerías, recursos y prácticas de extracción

La pesca en playas arenosas comprende tres tipos de pesquerías: i) la de subsistencia basada en la recolección de organismos para consumo familiar, ii) la recreativa o deportiva basada en la pesca para cebo o alimentación, sin venta ni dependencia del recurso, y representa un gran porcentaje de capturas en algunas de las pesquerías de playas arenosas y iii) la comercial que implica la recolección para la venta, generando una fuente de empleo considerable. En muchos casos, estos tres tipos de pesquerías pueden llegar a coexistir (McLachlan et al., 1996; McLachlan et al., 2013). Las playas arenosas son ideales para el desarrollo de la pesca recreativa, principalmente de especies intermareales como son los donácidos, lo que ha generado un incremento de los pescadores recreativos, generando un conflicto con el sector comercial (Schoeman, 1996; Defeo, 2003).

McLachlan y Defeo (2018) clasifican dos categorías de recursos explotados en las playas arenosas, la fauna de la zona sublitoral y la fauna de invertebrados bentónicos de la zona supralitoral e intermareal. En la zona sublitoral se capturan especies de peces pelágicas y demersales importantes tanto para la pesca comercial como recreativa, además de crustáceos. Estos organismos pueden ser capturados mediante redes de cerco (p.ej. jábegas y chinchorros), redes de arrastre (p.ej. gánguil), redes de enmalle y trasmallo, redes de empuje, dragas, palangre, anzuelos y trampas. En la zona más cercana a la costa los pescadores artesanales suelen utilizar embarcaciones pequeñas. La zona sublitoral se ha visto afectada por el agotamiento de las poblaciones de especies objetivo en muchas áreas, tanto por la pesca recreativa como por la comercial. Por otra parte, la pesca de invertebrados bentónicos supralitorales, intermareales o sublitorales poco profundos generalmente se llevan a cabo por pescadores individuales bien con palas, recolectados a mano, buceo, arpones, trampas y dragas mediante embarcaciones de pesca artesanal. Cabe destacar también la existencia de la pesca para recolección de poliquetos y crustáceos intermareales, que se utilizan como cebos, la cual también requiere de planes de gestión similares a los de la pesca tradicional. También, es importante señalar, tal y como describió Defeo (1996), que la actividad pesquera, además de capturar especies objetivo, causan mortalidad accidental sobre las especies no explotadas, ya sea por daños directos, indirectos o alteración del hábitat.

1.4.2 Pesquerías de bivalvos

Las pesquerías más significativas tienen como especie objetivo los bivalvos, pescándose al menos alguna especie de bivalvo en las playas expuestas de todos los continentes (McLachlan et al., 1996). Los bivalvos son un componente importante en dichas costas en términos de biomasa, siendo recolectados tanto recreativamente como en pesquerías artesanales o comerciales (McLachlan et al., 1996). En el caso de la pesca comercial, puede tener lugar directamente desde la orilla o desde pequeñas embarcaciones (McLachlan y Defeo, 2018).

Al menos 20 especies de bivalvos son explotados a lo largo de las playas arenosas de todo el mundo (McLachlan y Defeo, 2018), algunos ejemplos son presentados en la Tabla 1.1.

Tabla 1.1: Aspectos generales sobre algunas de las pesquerías de playas arenosas. Modificado de McLachlan y Defeo (2018).

Principal área de distribución	Especies	Hábitat	Tipo de pesquería	Nombre común
África	*Donax serra*	I y S	A y R	White sand mussel
Europa	*Donax trunculus*	I y S	A y R	Coquina, tellina, cadelucha, wedge clam
	Chamelea gallina	S	A	Chirla, rossellona, striped venus clam
	Callista chione	S	A	petxina lluent, smooth clam
	Spisula subtruncata	I y S	A	---
	Mactra stultorum	I y S	A	Pechina lisa
	Cerastoderma edule	S	A	Berberecho, common edible cockle
	Ensis siliqua	S	A	Navaja, pod razor Shell
Caribe	*Donax denticulatus*	I	A	Beach clam
	Donax striatus	I	A	Chipi-chipi

Principal área de distribución	**Especies**	**Hábitat**	**Tipo de pesquería**	**Nombre común**
Asia	*Donax cuneatus*	I	A	---
	Donax faba	I	A	---
Australia	*Donax deltoides*	I y S	A y R	Pipi
América del Sur	*Mesodesma mactroides*	I	A y R	Yellow clam
	Mesodesma donacium	I y S	A	Macha
Nueva Zelanda	*Paphies ventricosa*	I	A y R	Toheroa
	Paphies subtriangulata	I y S	A y R	Tuatua
	Paphies donacina	I y S	A y R	Tuatua, southern tuatúa
América del Norte	*Siliqua patula*	I y S	A y R	Pacific razor clam
	Tivela stultorum	I y S	A y R	Pismo clam
Caribe y América del Sur	*Tivela mactroides*	I y S	A y R	Guacuco, trigonal clam

En hábitat: I: intermareal; S: submareal. En Tipo de pesquería: A: artesanal; R: recreativa.

Han sido varias las pesquerías de bivalvos que han presentado declives en sus poblaciones, llegando en algunos casos al colapso. En una playa de Nueva Zelanda, el bivalvo *Paphies ventricosa* presentó un declive continuado de capturas pese al establecimiento de medidas para contrarrestarlos, tales como, temporadas de veda, clausuras en determinadas zonas por más de 20 años, restricciones de tamaño, de cupo de pesca, las cuales fueron ineficientes para recuperar las poblaciones a un nivel sostenible para la pesca (McLachlan et al., 1996). Este declive de las poblaciones de *P. ventricosa* fue justificado por la erosión y pérdida de arena que presentaba la zona (Beentjes et al., 2006). En California, también se observó un caso similar con *Tivela stultorum*, en el que, pese a la adopción de medidas de gestión, las poblaciones alcanzaron niveles insostenibles para su explotación (McLachlan et al., 1996). En Uruguay (Atlántico), en 1987, la pesquería de *Mesodesma mactroides* mostró un declive que llevó al cierre de la misma durante casi tres años, siendo tras el primer año de cierre cuando se pudo observar una recuperación de las poblaciones (Defeo, 1996; Brazeiro y Defeo 1999; Ortega et al., 2012). Sin embargo, casi a los tres años de cierre el aumento de la población estuvo acompañado por una disminución notable en el reclutamiento, destacando que un período de cierre prolongado puede causar consecuencias negativas (Defeo, 1996;

Brazeiro y Defeo, 1999). Otro ejemplo de colapso lo observamos en un estudio llevado a cabo por Baeta et al. (2018) en Catalunya sobre la pesquería del bivalvo *Callista chione*. La actividad pesquera fue suspendida durante un año para recuperar la población de este bivalvo después de detectar su declive. Una vez abierta la pesquería de nuevo, no se observó una recuperación, por lo tanto, muchos pescadores decidieron abandonar la pesquería de este bivalvo, presentando una actividad residual hasta el cierre permanente 6 años después. Baeta et al. (2014) destacan que la ausencia de un manejo integrado de la zona costera en la que era explotada *C. chione*, provocó una situación crítica en sus poblaciones llevando a la pesquería al colapso, debido principalmente al dragado periódico de arena en la zona y la sobrepesca. Por otra parte, en Galicia la pesquería de *Cerastoderma edule* también colapsó debido a la alta mortalidad de individuos generada por el parásito *Marteilia cochillia* (Villalba et al., 2014). Garcia et al. (2018) observaron una elevada mortalidad de *D. trunculus* en Francia durante los años 2008-2011, vinculado con la aparición de los parásitos *Mikrocytos veneroïdes* y *Mikrocytos donaxi*.

En las pesquerías comentadas, a pesar de que se implementaron diferentes herramientas de gestión, las poblaciones no se recuperaron a niveles adecuados para su explotación. La sostenibilidad de las pesquerías depende del establecimiento de unas medidas de gestión adecuadas en las que deben ser considerados tanto aspectos biológicos como ecológicos de las especies objetivo (Gaspar et al., 2002). Esto es importante para evitar problemas pesqueros y favorecer la regeneración de recursos, adaptando el esfuerzo pesquero a los recursos disponibles y cerrando las pesquerías cuando sea necesario (Bombace, 1991). Además, es imprescindible llevar a cabo una gestión a largo plazo sólida y eficaz basadas en una visión de las playas como sistemas socioecológicos, con subsistemas biofísicos (ecológicos y geomorfológicos) y humanos (aspectos culturales, económicos, políticos y éticos) que interactúan. Tener en cuenta solo uno de los subsistemas llevará a una explotación insostenible de los recursos (McLachlan et al., 2013). Defeo et al. (2016) observaron que los planes de cogestión, donde el gobierno y los usuarios locales comparten las tareas y responsabilidades de gestión, ayudaron a mejorar algunas de las pesquerías latinoamericanas. Autores como Baeta et al. (2018) señalaron la necesidad de llevar a cabo planes de gestión que favorezcan la cogestión basada en los ocho principios establecidos por Ostrom (1990). Estos principios son: (1) límites de los usuarios y recursos claramente definidos; (2) congruencia entre las reglas de apropiación y las de provisión, ajustándose ambas a las condiciones locales; (3) decisiones colectivas; (4) seguimiento de los usuarios y recursos; (5) sanciones graduales en función de la seriedad y el contexto de la violación de las reglas; (6) mecanismos de resolución de conflictos; (7) reconocimiento mínimo de los derechos de organización y (8) organización en niveles anidados. Estos ocho principios son robustos para pesquerías a pequeña escala (Cox et al., 2010).

1.4.3 Pesquerías en las playas del sector sur del golfo de Valencia

En el golfo de Valencia, Gandia y Cullera son las dos zonas más importantes donde se pueden encontrar bancos de los bivalvos infralitorales *D. trunculus* (coquina o tellina) y *C. gallina* (chirla), presentando ambas zonas la mayor flota marisquera y el mayor volumen de capturas del golfo (Ramón, 1993). Según la GVA (2016), de estas dos zonas, Gandia representó el 60 % de las capturas de *D. trunculus* y *C. gallina* en el golfo de Valencia entre los años 2005 y 2012. Las áreas poco profundas con sustratos arenosos son hábitats específicos para estas especies (Lleonart et al., 1990). *D. trunculus* se puede encontrar en un rango batimétrico comprendido entre 0 y 2 m de profundidad en la costa mediterránea (Ramón et al., 1995; Gaspar et al., 2002), en un área donde los organismos están continuamente sujetos a turbulencias. La coquina es un organismo filtrador que puede alimentarse de fitoplancton, detritos de plantas y materiales resultantes de la muerte de fitoplancton y bacterias, si bien tiene una mayor eficacia de asimilación con una dieta basada en microalgas (McLachlan y Defeo, 2018). En el caso de *C. gallina*, se encuentra en una zona más profunda, distribuyéndose entre los 3 y 12 m de profundidad (Moschino y Marin, 2006) y encontrándose en la capa superficial, la cual le ofrece refugio (Pampanin et al., 2002). La chirla es un organismo filtrador que filtra tanto fitoplancton como otras partículas de seston (Romanelli et al., 2009).

Ambas especies forman parte del mismo *métier*, esto se debe a que combinan similares artes de pesca, área de explotación, período de pesca y un patrón de explotación similar (European Commission, 2010). La técnica de pesca se basa en la utilización de dragas tiradas por mariscadores a pie en las zonas menos profundas (exclusivo para la coquina, al distribuirse en una zona más somera) y cuatro dragas remolcadas por pequeñas embarcaciones de pesca artesanal en aguas más profundas (Ramón, 1993). En esta última, dependiendo de la especie objetivo, variará el tamaño de la luz de malla que recubre la draga y la profundidad en la que se trabaja. Las dragas utilizadas en las embarcaciones (Figura 1.2A) consisten en una caja rectangular de acero recubierta con una malla, con una abertura de 70 cm de ancho y una barra dentada para facilitar su introducción al sedimento. En esta caja rectangular se van recolectando los organismos a medida que avanza la embarcación, siendo descartados el sedimento y organismos pequeños a través de la malla. El tamaño de la luz de malla metálica que recubre las dragas es de 11.7 x 11.7 mm para *D. trunculus* y 24 x 11 mm para *C. gallina*. Los buques dedicados a esta pesquería en Gandia son normalmente pequeños, presentando un tonelaje bruto de 1 a 9, longitud total de 7 a 12 m y potencia de 16 a 74 kW (European Commission DG MARE, 2018). La draga manual (Figura 1.2B) es una caja rectangular de acero de 55-70 cm de ancho, dentada y un tamaño de luz de malla de 11.7 mm. Esta draga se ata a la cintura de la persona y se va arrastrando a lo largo de la costa con la ayuda de un mango largo. En la Comunidad Valenciana, la captura de estos bivalvos está regulado por unas capturas máximas anuales y diarias, así como el tamaño mínimo (14 mm para *D. trunculus* y 25 mm para *C. gallina*), y también por el establecimiento de zonas de pesca, horarios y características de los artes de pesca.

Durante las últimas décadas, en algunas partes de España y el Mediterráneo, se ha observado una reducción drástica de las reservas naturales en ambas especies (Ruiz-Azcona et al., 1996; Lart et al., 2003; Padella y Finco, 2009; Romanelli et al., 2009; Martínez-Pita et al., 2012; Baeta et al., 2018). La disminución de las capturas en la Comunidad Valenciana llevó al cierre de esta pesquería a principios de junio de 2015 (Resolució de la Comunitat Valenciana del 3 de juny, 2015).

Figura 1.2: Dragas utilizadas para la pesca de *C. gallina* y *D. trunculus*, respectivamente, desde embarcación (A) y draga utilizada por los mariscadores a pie para la extracción de *D. trunculus* (B).

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Capítulo 2

Objetivos y estructura de la tesis

2. OBJETIVOS Y ESTRUCTURA DE LA TESIS

El objetivo principal de esta tesis fue analizar la pesquería del *métier* compuesto por *Donax trunculus* y *Chamelea gallina*, en el área de trabajo de la Cofradía de Pescadores de Gandia, hasta su cierre en junio de 2015, la evolución de las poblaciones de *D. trunculus* tras el mismo, así como los factores ambientales que pudieron afectar a los bancos de ambas especies.

Además, para alcanzar este objetivo se han planteado los siguientes objetivos específicos:

1. Estudiar la evolución de la pesquería de *D. trunculus* y *C. gallina* en el área de trabajo de la Cofradía de Pescadores de Gandia, entre 2004 y 2015, así como la estacionalidad de la pesca, la distribución de las capturas en las diferentes zonas de pesca y la idoneidad del sistema de gestión de esta pesquería.
2. Estudiar aspectos ecológicos y biológicos de *D. trunculus* en las poblaciones del área de trabajo de la Cofradía de Pescadores de Gandia, tales como las variables de crecimiento, reclutamiento y la distribución batimétrica.
3. Analizar la densidad y la biomasa de *D. trunculus* antes del cierre de su pesquería y su evolución durante los cinco meses posteriores, así como el estado de la población 2 años después del cierre.
4. Determinar si la comunidad macrobentónica del sublitoral de playas arenosas responde a diferentes factores físicos, químicos y nutricionales.

Para alcanzar estos objetivos, la presente tesis se organiza en capítulos que corresponden a los artículos científicos redactados, en los cuales se lleva a cabo una introducción en la que se enmarcan los objetivos, un apartado de material y métodos que incluye una descripción de la zona de estudio, y se presentan resultados y discusión.

Capítulo 3. "The state of artisanal bivalve fishing exploitation in the Spanish Mediterranean coastal: Gandia Coast case". En este capítulo se analiza la evolución del *métier* compuesto *D. trunculus* y *C. gallina*, en el área de trabajo de la Cofradía de Pescadores de Gandia, que abarca desde Tavernes de la Valldigna a Denia, entre los años 2004 y 2015. Esto se lleva a cabo a partir de los datos diarios de capturas, de la evolución mensual entre los años 2011 y 2015 así como la anual entre 2004-2014. Además de ello se analizan las variaciones de la presión pesquera a lo largo la zonificación establecida por la legislación y la estacionalidad del *métier* basada en la biología de las especies y el rendimiento económico. Además de ello se revisa la idoneidad de las medidas de gestión del *métier* y se proponen alternativas de gestión. Este capítulo fue enviado a la revista *Fisheries Research*, y se encuentra en proceso de revisión.

Capítulo 4. "Changes of *Donax trunculus* populations after the fishery's closure in Gandia (Western Mediterranean Sea, Spain)". En el capítulo 4 se analizan las densidades y la biomasa de *D. trunculus*, en el área de trabajo de la Cofradía de Pescadores de Gandia, justo en el momento previo al cierre de su pesquería y la evolución de ambos valores durante los cinco meses inmediatamente posteriores al cierre, además del estado de las poblaciones dos años después del cierre. En este capítulo también son analizados varios aspectos biológicos y ecológicos tales como las variables de crecimiento, el reclutamiento y la diferenciación por tallas en función de la batimetría, así como algunos aspectos pesqueros. Este artículo ha sido enviado a la revista *Journal of Shellfish Research*, y se encuentra bajo proceso de revisión.

Capítulo 5. "Driving forces that structure macrobenthic communities of microtidal intermediate sandy beaches". En este capítulo se estudia la respuesta de las comunidades bentónicas a las variables ambientales en el sublitoral de las playas arenosas intermedias del sector sur del golfo de Valencia, haciendo hincapié en los bivalvos de interés pesquero en la zona de estudio *D. trunculus* y *C. gallina*. Para ello, se lleva a cabo una caracterización de la dinámica temporal y espacial de la estructura de la comunidad macrobentónica, se analizan los patrones temporales y espaciales para variables físicas, químicas y nutricionales, y se determina la influencia de éstas en la estructura de la comunidad macrobentónica. Este artículo fue enviado a la revista *Estuarine, Coastal and Shelf Science*, en la que se encuentra en estado aceptado pendiente de la realización de revisiones.

A continuación, en el **Capítulo 6**, se lleva a cabo una discusión general en la que se engloban los resultados de los tres capítulos anteriores y se da una visión más amplia sobre la pesquería de *D. trunculus* y *C. gallina* y sus poblaciones, así como los factores que pueden haber afectado a la recuperación de la pesquería. Finalmente, en el **Capítulo 7** se presentan las conclusiones generales obtenidas.

Capítulo 3

The state of artisanal bivalve exploitation in the Spanish Mediterranean Coastal: Gandia Coast case

3. THE STATE OF ARTISANAL BIVALVE FISHING EXPLOITATION IN THE SPANISH MEDITERRANEAN COASTAL: GANDIA COAST CASE

3.1 ABSTRACT

This study analyses the métier composed by *Donax trunculus* and *Chamelea gallina* through the data provided by the Gandia Fishermen's Guild (Gulf of Valencia, Western Mediterranean Sea) both daily (between June 2011 and May 2015) and annually (2004-2014). Gandia is a key area for exploitation of both species in the Gulf of Valencia. In the study area, these clams are exploited in an artisanal way, with dredges towed by a small vessel and also, in the case of *D. trunculus*, captured by on-foot shellfish gatherers using hand dredges. In Gandia, catches and fishing fleet of this métier suffered a sharp decrease during the study period, becoming more abrupt from 2009-2010. In addition, high seasonality could be observed with catches and fishing effort peaks in spring-summer, and clear differences in the fishing pressure along coastal zones, resulting in some areas being more exploited than others. On the other hand, it was observed that the high daily quotas established as a management measure for the fishery did not limit the catches of both species. Overall, it is suggested that the management plans for this fishery were not effective. Changes in management measures for better management of the fishery have been proposed.

Keywords: métier; *D. trunculus*; *C. gallina*; management; clam

3.2 INTRODUCTION

The exploitation of coastal marine resources has been carried out for more than 150 000 years by humans (Marean et al., 2007). Particularly, on the Spanish coast, the human use of bivalve molluscs, as described by Jordá et al. (2016), started during the Upper Pleistocene and the Early Holocene. Therefore, fishing of these organisms has being of great importance since the beginning of human coastal settlements. Currently, artisanal fisheries are of great relevance both economically and socially in the Spanish Mediterranean coastal region, involving a large number of people (Alarcón-Urbistondo, 2002), representing 54.2 % of the Spanish fishing fleet in the Mediterranean (Forcada et al., 2010).

Exposed sandy beaches dominate a great part of the coast in all continents, offering high value commercial resources, as they have a rich biota. Bivalves are an important component in these coasts in terms of biomass, being collected both recreationally, and by artisanal or commercial fisheries (McLachlan et al., 1996). Spain has a great shellfish tradition, bivalve fishing being practiced to a greater or lesser extent along the Mediterranean coast (Ramón, 1993). Although fishing is a strictly regulated activity, it is suffering a crisis as the fishing grounds are subjected to heavy exploitation, suggesting poor management (Santos-Martín et al., 2015). FAO (2016) estimated that, in 2013, 31.4 % of the world's fish stocks were overexploited. In the case of the Mediterranean and Black Sea 59 % of the populations were unsustainably exploited. Also, it is estimated that 60 % of the Spanish fishing grounds are or have collapsed at some point (Santos-Martín et al., 2015). In the Spanish Mediterranean, bivalve beds have gradually declined due to high demand and lack of control of their exploitation (Ramón, 1993).

In the Gulf of Valencia, Gandia and Cullera are the two most important zones where beds of *Donax trunculus* and *Chamelea gallina* can be found. These areas have the largest shellfish fleet and catches (Ramón, 1993). According to GVA (2016), Gandia represented 60 % of the catches of *D. trunculus* and *C. gallina* in the Gulf of Valencia between 2005 and 2012. The species, which are part of the same métier, are fished by dredges towed by small vessels (Ramón, 1993), in which mesh size and depth are modified, depending on the target species. In addition, in the case of *D. trunculus*, it can also be captured by manual collectors using hand dredges (Ramón, 1993) because it is distributed in a shallower area than *C. gallina*.

Shallow areas with sandy substrates are specific fishing beds for these clams (Lleonart et al., 1990). *D. trunculus* inhabits sandy beaches, at a bathymetric range between 0 and 2 m depth on the Mediterranean coast (Ramón et al., 1995; Gaspar et al., 2002), in an area where organisms are continuously subject to turbulence. The biocoenoses of fine surface sands -biocoenoses described by Peres and Picard (1964) which applies for the whole Mediterranean and Europe- is found between a depth of 0 and 3 m, where *D. trunculus* is the most characteristic and abundant species (McLachlan and Brown, 2006; Augier, 2008). *D. trunculus* is a filter feeder which can feed on phytoplankton, plant detritus and foams resulting from phytoplankton and bacteria death, but it has a greater assimilation efficiency with a diet based on phytoplankton (McLachlan and Brown, 2006). The reproductive period is influenced by water temperature and annual phytoplankton production (McLachlan et al., 1996). For example, in the Spanish Mediterranean coast, the reproduction of this clam occurs during a long period of time between June and October (Ramón, 1993), while in the Spanish Atlantic coast, it occurs between March and October, with peaks from May to August (JACUMAR, 2009). In one study carried out by Ramón (1993) off the coast of Cullera -directly to the north of our study area- two periods of recruitment of *D. trunculus* were determined, the first between July and August and the second in December, as well as the size at first maturation 12.7 mm. The recruitment of this species can be compromised in areas with elevated anthropic pressure (Delgado and Silva, 2018). In other species of the same genus such as *D. hanleyanus,* the salinity and phytoplankton were determined to be important factors which can affect recruitment (Defeo and de Alava, 1995). In the Gulf of Valencia, *D. trunculus* can reach a maximum size of between 41.8 and 46.0 mm, with a rate of growth from 0.6-0.7 and a lifespan of 2 to 3 years (Ramón et al., 1995). The fine well-sorted sands community -according to Peres and Picard (1964)- is found at a greater depth, which extends from 2 to approximately 20 m.

Species of high commercial interest, such as *C. gallina*, are found in shallower areas of this biocoenoses (Costa et al., 1984; Giner and Soler, 1989; Augier, 2008), which is distributed between a depth of 3 to 12 m (Moschino and Marin, 2006). *C. gallina* is a suspension feeder with a reproductive period that extends from May to August in the Gulf of Valencia and one period of recruitment between August and October (Ramón, 1993). The size at first maturation of this species is smaller than the previous one, at 12.3 mm. The maximum size reached by *C. gallina*, with a rate of growth from 0.3-0.4, is between 36-40 mm, with a lifespan of 3 or 4 years (Ramón, 1993).

The survival of these organisms, and therefore their density, can be in certain cases due to the temperature of the water. The decrease of water temperature (Manca Zeichen et al., 2002) together with the salinity due to torrential rain (Neuberger-Cywiak et al., 1990) or high temperatures (Monari et al., 2007) can affect the abundance of both species of clams. An influence was also observed due to the decrease in the flow of freshwater from the rivers (Romanelli et al., 2009). Mazé and Laborda (1988) and Lart et al. (2003) determined a clear influence of the granulometry in the presence of those organisms. As well as these environmental variables that can determine the density or mortality of these species, certain parasites can provoke a high death rate. Moreover, starfish, fish, crabs and gastropods are the main predators of the infralittoral bivalves (Ramón, 1993).

During the last few decades, in some parts of Spain and the Mediterranean, a drastic reduction of natural reserves in both species has been observed (Ruiz-Azcona et al., 1996; Lart et al., 2003; Padella and Finco, 2009; Romanelli et al., 2009; Martínez-Pita et al., 2012; Baeta et al., 2018). In the Valencian Community, capture of these clams is carried out according to regulations that establish maximum annual and daily catches, as well as individual minimum size (14 mm for *D. trunculus* and 25 mm for *C. gallina*). In the Gulf of Valencia, artisanal fishing vessels have almost been restricted to the ports of Cullera and Gandia in the last two decades. The decrease in catches in the Valencian Community led to the closure of this fishery at the beginning of June 2015 (Valencian Community Resolution of June 3, 2015).

The vessels dedicated to this fishery in Gandia are normally the smaller ones of the artisanal fleet. Each small vessel usually has two crew members aboard, although on occasion only one crew member is necessary. However, when there is an exceptional amount of clams, up to four crew members can be aboard. These small vessels can work on other artisanal fisheries, which fluctuates from year to year, depending on the quantity and price of the catches (GVA, 2016). In the year 1993, Ramón (1993) established that approximately 70 small vessels would work in Gandia. However, an inventory carried out in 2000-2001 by Alarcón-Urbistondo (2002) estimated that for each of the species of clam, there were 25 small vessels fishing it, which meant 50 fishermen to each clam. On the other hand, the hand-dredges are less common than the small vessels and have only got a maximum of four fishing licenses in our area of study (personal communication secretary manager of Gandia Fishermen's Guild).

The aim of this work was to study the evolution of *D. trunculus* and *C. gallina* catches in the Gandia Fishermen's Guild work area, between 2004 and 2015, as well as the fishing seasonality based on the species biology and economic performance. In addition, the distribution of catches in the different fishing zones was analysed and the suitability of the management system carried out in this artisanal fishery.

3.3 MATERIAL AND METHODS

3.3.1 Study zone

The study area is located on sandy beaches of the southern sector of the Gulf of Valencia in micro tidal Western Mediterranean Sea, off the eastern coast of Spain (Fig. 3.1). It covers the area from Tavernes de la Valldigna to Dénia, where the Gandia Fishermen's Guild's fleet works. This area was divided according to the classification established by the Regional Ministry of Agriculture, Fisheries, Food and Water as production areas for bivalve molluscs in the Valencian Community waters (Valencian Community Resolution of February 19, 2013). Of the total fishing zones established by this regulation, 12 coincide with the areas in which the Gandia Fishermen's Guild's fleet fishes. These 12 zones, in this study, were reorganized into 10 areas, bringing together zones 7, 8 and 9 into a single area (Area VII), due to the fishermen's confusion about catches made in these zones. The seabed in these areas has a greater amount of fine sands with respect to the rest of the sediment fractions, increasing the percentage of fine sands with depth (Ramón, 1993). It should be noted that the region and has heavy tourist pressure, mainly north of Gandia Port (Area III). The main freshwater input is the River Serpis. In addition, there are small streams and irrigation channels (Fig. 3.1). The River Serpis has a marked seasonality, with a dry period during the summer and torrential rains during the autumn (Sebastiá et al., 2013). In addition, this river is artificially regulated by dams and irrigation channels that entails a decrease in the flow along the river and the absence of a permanent flow at the mouth (Sospedra et al., 2018).

3.3.2 Legislation

During the studied period, different regulations were established in order to manage the fishery of both species. These have been summarised in Table 3.1. In July 1993, the Valencian Community Order of June 7, 1993 was published, which regulated the fishery of shellfish in the Valencian Community for *C. gallina* and *D. trunculus*. The law stated that all the individuals caught for both species had to be sieved on board through a metal plated sieve perforated with circular holes with a minimum diameter of 19 mm for the *C. gallina* and 14 mm for *D. trunculus*. Then all the specimens that had been passed through the sieve had to be returned to the sea. This law was developed or complemented by the Valencian Community Resolution of February 12, 1996 and Valencian Community Resolution of April 13, 2007, which established the hours of shellfish fishery on the coast each year. Furthermore, it was modified by the Valencian Community Decree 67/1996, which regulated shellfish licenses. In the end it was repealed and replaced by the Valencian Community Decree 94/2013 which regulated shellfish fishing of *C. gallina* and *D. trunculus* on the coast of the Valencian Community.

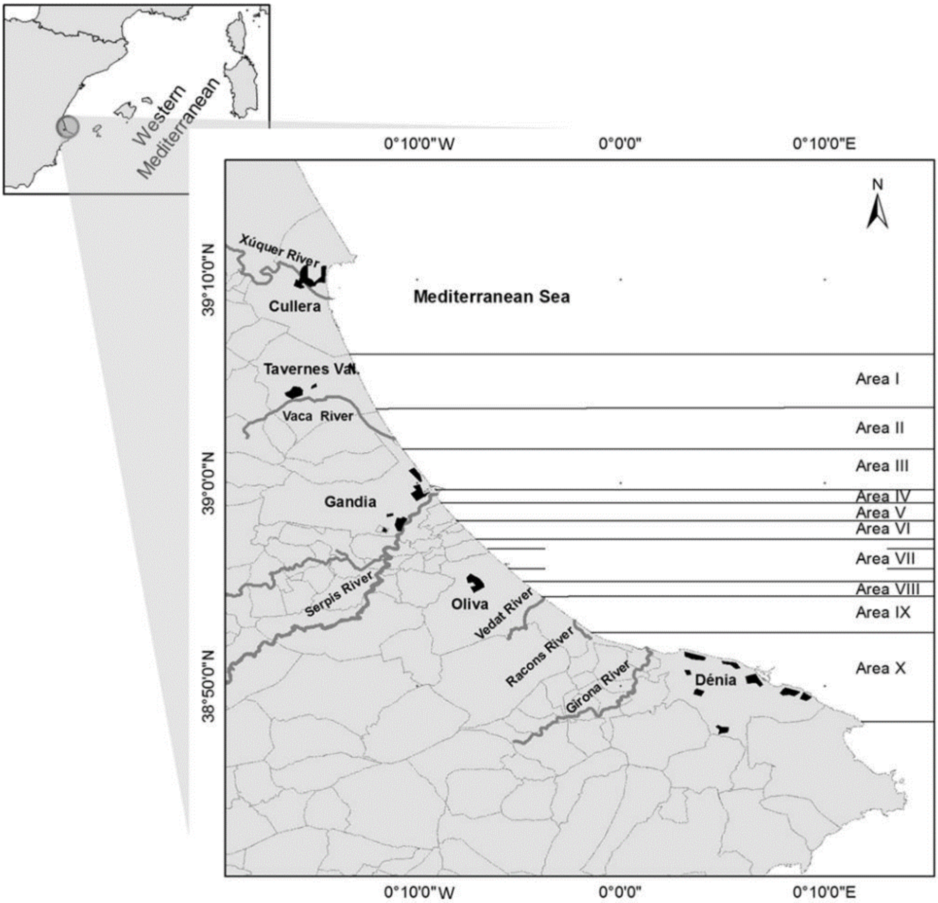

Figure 3.1. Study zone with the 10 fishing areas.

Additionally, the Valencian Resolution of February 19, 2013 was published which approved a listing of the areas of production of bivalve molluscs, gastropods and equinoderms in the water of the coast of the Valencian Community, and their classification.

With the objective of fixing an annual limit on the fishing of *C. gallina* and *D. trunculus*, the Valencian Community Resolution of August 9, 2013 was established. The limit of annual catches of both species was fixed for the whole Valencian

Community. This was repealed and replaced by the Valencian Community Resolution of January 17th, 2014.

Finally, in June 2015, the Valencian Community Resolution resolved the closure of the fishery of both species due to the reduction of each species.

In May 2016, the Management Plan was published in the Valencian Community Decree 62/2016. This Decree established the objectives of the Management Plan and reference points, as well as the monitoring of the objectives, the minimum landing size (25 mm for *C. gallina* and 14 mm for *D. trunculus*), the suitable areas for shellfish, reserve zones, the conditions for the authorization of the activity, a control and monitoring of shellfish fishing and evolution of the populations, in all areas where it is authorized, plus the need to create a monitoring commission.

3.3.3 Fishing gear

The fishing techniques for the bivalves *D. trunculus* and *C. gallina* were widely described by Ramón (1993), and they consist of hand dredges pulled by on-foot shellfish gatherers in shallower waters and four dredges towed simultaneously by small vessels in deeper waters. The dredges are a steel rectangular collecting box with an opening of 70 cm in width with a toothed bar to facilitate its introduction to the sediment. The metallic mesh size that covers the dredges is of 11.7 x 11.7 mm for *D. trunculus* and 24 x 11 mm for *C. gallina*. The Gandia Port fishing fleet has a gross tonnage (GT) range of 1 to 9, overall length (LOA) from 7 to 12 m and power from 16 to 74 kW (European Commission DG MARE, 2018). The hand dredge is a steel rectangular collecting box 55-70cm wide, with diving teeth, and a mesh size of 11.7 mm. The dredge is pulled from the waist and held with a towing long handled pole.

Table 3.1. Legislation in force during the studied period.

	O. 1993	R. 1996	D. 1996	R. 2007	D. 2013	R. Feb. 2013	R. Aug. 2013	R. 2014	R. 2015
Authorised fishing gears									
Small vessel dredge									Closure
Max. width (vessel with 3 dredges)					85 cm				Closure
Max. width (vessel with 4 dredges)	75 cm				75 cm				Closure
Hand dredge max. width	85 cm				85 cm				Closure
Timetable of fishing in study zone									
	Mon. to Fri., max. 9 h	Mon. to Fri. 5 am -2 pm 1st Jul. to 15th Sep. until 11 am		15th Jun. to 15th Sep. 4-11 am	Mon. to Fri. 5 am-2 pm 15th Jun. to 15th Sep. 4-11 am				Closure
Min. landing size									
D. trunculus	14 mm				14 mm				Closure
C. gallina	19 mm				25 mm				Closure
Daily joint allowable catches									
Hand dredge	35 kg				20 kg				Closure
Small vessel with 1 crew member					40 kg				Closure
Small vessel with 2 crew members	65 kg				65 kg				Closure
Small vessel with 3 crew members	90 kg								Closure
Small vessel with 3 or more crew members					90 kg				Closure
Small vessel with 4 crew members	125 kg								Closure
Maximum annual catches in whole Gulf of Valencia									
D. trunculus							41896 kg	22210 kg	Closure
C. gallina							22665 kg	6302 kg	Closure
Max. hand dredge licences									
	25				15				Closure
Max. period of licences									
			5 years						Closure
Zones in Gulf of Valencia									
Study Zone areas						CVA1.3.6-CVA1.3.17			Closure

O. 1993: Order of June 7, 1993; R. 1996: Resolution of February 12, 1996; D. 1996: Decree 67/1996; R. 2007: Resolution of April 13, 2007; D. 2013: Decree 94/2013; R. Feb. 2013: Resolution of February 19, 2013; R. Aug. 2013: Resolution of August 9, 2013; R. 2014: Resolution of January 17, 2014; R. 2015: Resolution of June 3, 2015

3.3.4 Data

Historical data concerning the métier that covers the Gandia fleet and on-foot shellfish gatherers' fishing zones were provided by the Gandia Fishermen's Guild. Also, total annual catch data for *D. trunculus* and *C. gallina* in the whole Spanish Mediterranean were provided by the Ministry of Agriculture and Fisheries, Food and Environment of Spain for the period 2004-2014 (MAPAMA, 2017). The annual evolution of *D. trunculus* was analysed for the period 2004-2014 in Gandia, based on data from annual catches in kilograms plus the fishing effort (number of trips) for each small vessel and on-foot shellfish gatherers. For *C. gallina*, this data was provided for the 2007-2014 interval, only for small vessels because its distribution occurs in a higher bathymetric range. The catch-per-unit-effort (CPUE) was calculated for small vessels (kg small vessel^{-1} d^{-1}) and for manual collectors (kg on-foot shellfisher^{-1} d^{-1}). From June 2011 to May 2015, the data provided by Gandia fishermen's Guild for both species were daily catches per small vessel and, in the case of *D. trunculus*, also per on-foot shellfish gatherer, in which the area where they were collected was specified. On the basis of this data, monthly catches, fishing effort and CPUE were calculated for small vessels and for on-foot shellfish gatherers for these periods, as well as daily catches per small vessel. In addition, annual catches collected per km of coastline for each area during the 2012-2014 period were estimated. Finally, the monthly sale price of the target species was provided during the entire study period (2004-2015).

In order to detect the significant trend of catches and CPUE in the time series of both species, non-parametric Man-Kendall (MK) and Sen's slope tests were used. In order to assess if there were statistically significant differences in the small vessels' characteristics, simple ANOVA or Kruskal-Wallis (in case of non-normality in the data) tests were performed.

3.4 RESULTS

3.4.1 Annual evolution of *D. trunculus* and *C. gallina* during 2004-2014 period

Table 3.2 presents annual catches, fishing effort, CPUE, as well as fishing fleet and number of on-foot shellfish gatherers for *D. trunculus* in Gandia during the period 2004-2014. The annual catches suffered an abrupt decrease during the fore mentioned period, falling from 100 000 kg yr^{-1} in 2004 to 1600 kg yr^{-1} in 2014. The downward trend was continuous over the years, however, in 2007, a specific and significant reduction was observed with respect to the general descent pattern. The results of the application of the MK test showed a statistically significant decreasing trend in *D. trunculus* catches ($p = 0.0001$). The results for the Sen's slope test indicated a decreasing trend of 11 086 kg per year. In the period in which there was on-foot shellfish gathering, between 2004 to 2011, their catches represented a very low percentage with respect to the total annual figure, with a minimum value of 0.3 % in 2004 and a maximum of 1.6 % in 2006, most years usually being lower than 1 %. As a result, information provided by on-foot shellfishing was considered negligible. The fishing effort also showed a marked decrease in the case of small vessels, from around 2100-2600 fishing trips in 2004, 2005 and 2006 to less than 300 trips in 2014. The year 2007 was also atypical with a specific decrease in the number of trips. The small vessels CPUE, however, remained relatively constant in the range between 37-42 during the period 2004-2008, including 2007, as both the catches and the fishing effort showed a notable decrease in that year. However, from 2010, an abrupt decline in the CPUE was observed, falling from 18.8 to 5.5 in 2014. According to MK test the CPUE revealed a statistically significant decreasing trend ($p = 0.0002$), with an annual decrease of 3.8. As regards the small vessels that worked in Gandia, the values oscillated throughout the years of this study from a maximum of 32 in 2005 to a minimum of 10 in 2014, showing a gradual decrease in this period.

Table 3.2. Annual catches, fishing effort, CPUE, fishing fleet and number of on-foot shellfishers for *D. trunculus* in Gandia during the period 2004-2014. Total annual catches of *D. trunculus* in the whole Spanish Mediterranean is also shown.

Year	Catches in Spanish Mediterranean Sea (kg)	Gandia							
		Small vessel catches (kg)	Hand dredge catches (kg)	Small vessel Fishing effort (trips)	Hand dredge Fishing effort (trips)	CPUE for small vessel	CPUE for on-foot shellfisher	Fishing fleet (small vessels)	On-foot shellfishers
2004	228 350	101 627	263	2611	36	38.9	7.3	27	1
2005	256 458	98 249	867	2381	155	41.3	5.6	32	4
2006	200 454	90 664	1433	2137	234	42.4	6.1	30	4
2007	82 880	23 426	187	624	44	37.5	4.2	21	4
2008	152 358	62 912	856	1694	83	37.1	10.3	22	2
2009	183 233	57 026	857	1846	77	30.9	11.1	23	2
2010	145 623	30 976	126	1645	16	18.8	7.9	25	2
2011	103 529	18 571	121	1224	51	15.2	2.4	18	1
2012	98 549	12 273	0	807	0	15.2	0	14	0
2013	102 973	7272	0	501	0	14.5	0	11	0
2014	104 195	1597	0	291	0	5.5	0	10	0

In addition, the total annual catches of *D. trunculus* in the whole Spanish Mediterranean are shown in Table 3.2. These values were between 200 000-250 000 kg yr^{-1} during 2004 and 2006. In 2007 there was a clear decline that increased again in the period 2008-2010 and remained stable around 100 000 kg from 2011 to 2014.

On the other hand, Table 3.3 shows the same variables for *C. gallina* during the 2007-2014 period. The annual catches of this clam were at their highest in 2007 and 2008 with 21 000 and 37 000 kg yr^{-1} respectively and a minimum in 2012 and 2013 being almost nonexistent. The fishing effort presented a similar evolution to the catches, going from maximum values of around 900 trips in 2007 and 2008 to only 1 trip in 2012 and 2013. As regards the CPUE, the trend was similar to the previous variables, with maximum values of 39.1 in 2008 and a minimum value of 1 in 2013. The MK test revealed no statistically significant trend in *C. gallina* catches and CPUE (p = 0.083 in both cases) during the 2007-2014, probably as a consequence of the catch and CPUE increases in 2014. The fishing fleet showed a marked decline, with values around 20 small vessels in 2007, 2008 and 2010, to just one fishing small vessel in 2012 and 2013. Regarding the total catches of this species in the whole Spanish Mediterranean, the maximum was reached in 2008, exceeding 110 000 kg yr^{-1}, while the minimum occurred in 2012 with values close to 21 000 kg yr^{-1}.

Table 3.3. Annual catches, fishing effort, CPUE and fishing fleet for *C. gallina* in Gandia during the period 2007-2014. Total annual catches of *C. gallina* in the whole Spanish Mediterranean is also shown.

Year	Catches in Spanish Mediterranean Sea (kg)	Gandia			
		Catches (kg)	Fishing effort (trips)	CPUE	Fishing fleet (small vessels)
2007	78 233	21 162	839	25.2	23
2008	111 483	36 999	947	39.1	23
2009	55 477	252	27	9.3	9
2010	53 103	1652	133	12.4	21
2011	24 310	419	73	5.7	14
2012	21 417	5	1	5.0	1
2013	49 428	1	1	1.0	1
2014	31 818	810	57	14.2	8

In general, this type of artisanal fishery usually has a small fishing fleet and viable fishing days are few, due to shallow fishing areas and, to a large extent, sea conditions (Ramón, 1993). As the fishing fleet for this métier varied considerably between 2004 and 2014, it would be interesting to establish a classification, differentiating between occasional and habitual fishing fleets based on the number of fishing days. It is proposed to define as habitual fleet, small vessels which fish at least 50 % of the trips when compared with the small vessel that has made the greatest fishing effort for that particular year. In Figure 3.2, based on this classification, the number of occasional and habitual small vessels can be observed for each year. From 2004 to 2010, 13 small vessels remained relatively constant as habitual fleet. From the decrease in catches and CPUE that occurred in 2010, there was a progressive fall in the number of habitual small vessels, leaving a total of only 3 small vessels. This represented a decline of 77 % in the habitual fleet. Of the 10 small vessels which ceased in this fishery, 50 % of the cases stopped their activity (European Commission DG MARE, 2018) while the rest decided to dedicate fishing effort to other fisheries such as octopus. As regards to on-foot shellfish gatherers for this fishery, their contribution was considered irrelevant as only one shellfish gatherer could have been considered as habitual during the study period.

When different characteristics between occasional and habitual fleets were compared, the LOA showed significant differences ($p = 0.0001$), being lower in the habitual type. The power showed a trend close to significance ($p = 0.06$), in which the habitual fleet was lower than the occasional one. On the other hand, the GT did not present differences between the two types of small vessel. The smaller size and power of the habitual small vessels of this métier caused difficulties when adapting to other fisheries such as those which operate as occasional fleet. This is mainly due to the lack of available work and storage space on deck, as the gear is more voluminous than the one used in this artisanal fishery.

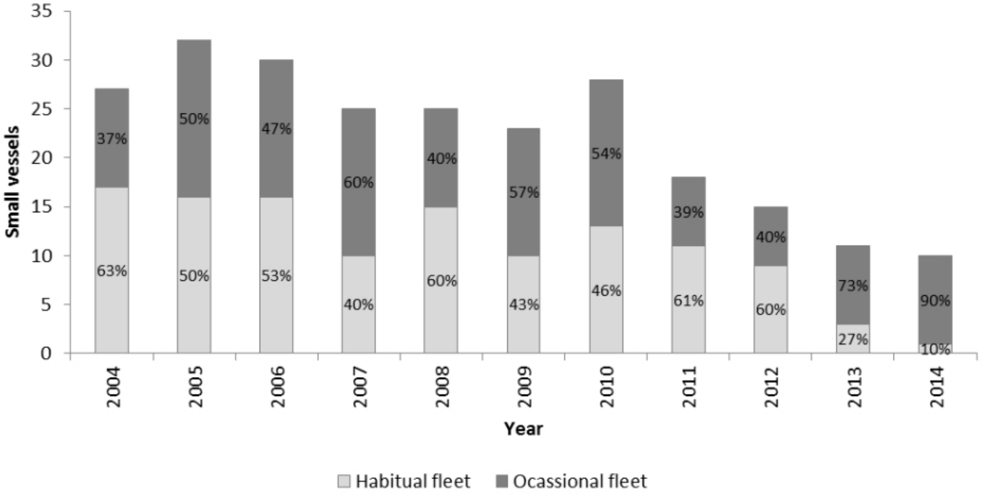

Figure 3.2. Number of occasional and habitual small vessels for the métier, between 2004 and 2014.

3.4.2 Monthly evolution of *D. trunculus* and *C gallina* during 2011-2015 period

In Figure 3.3A, evolution of monthly catches and fishing effort of small vessels for *D. trunculus* from June 2011 to May 2015 are shown. In 2011, 2012 and 2013 the highest monthly catches were obtained in summer (June and July) but these peaks disappeared in 2014 and 2015. It can also be seen that the evolution of fishing effort presented a similar trend to the catches.

The CPUE showed peaks between the months of April and July in the period 2011-2013, with maximums of 28 in June 2011, 21 in May 2012 and 18.5 in June 2013 (Fig. 3.3B). From 2014, the CPUE decreased and remained low until the two months prior to the fishery closure. The season when there was the greatest number of fishing small vessels *D. trunculus* was in summer (July-August) with maximum values that reduced from 19 small vessels in July 2011 to 9 in July 2014. In 2015, when the fishery closed at the beginning of June, the maximum occurred in March with 2 small vessels.

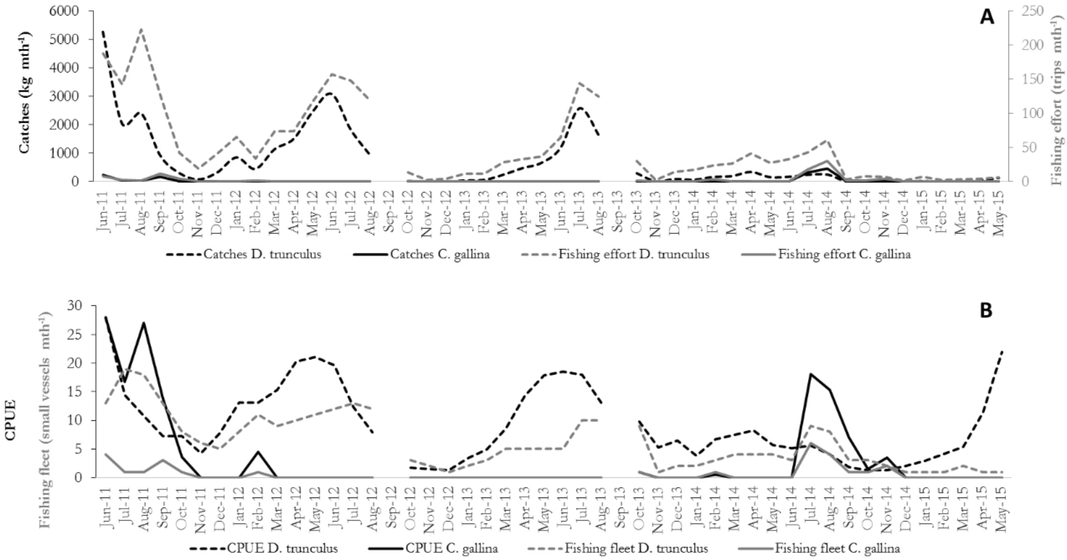

Figure 3.3. Monthly catches and fishing effort of small vessels (A) and CPUE and fishing fleet (B) for *D. trunculus* and *C. gallina* in Gandia from June 2011 to May 2015.

Regarding the evolution of monthly catches of *C. gallina* from June 2011 to May 2015 (Fig. 3.3A), it can be highlighted that these were almost non-existent over the months and years showing only some amounts in June 2011, September 2011, July 2014 and August 2014 with maximum values that varied between 170 and 460 kg mth^{-1}. The same evolution was observed for the fishing effort with maximum values that increased from 8 trips in June 2011 to 30 in August 2014 (Fig. 3.3A). Regarding the CPUE (Fig. 3.3B), the highest value of 27.9 was observed for June 2011 decreasing to a maximum of 18.11 in July 2014. Finally, the number of small vessels fishing during the months in those that presented maximum values varied between 3 and 6, with 1 or 2 small vessels working during the rest of the months.

3.4.3 Daily catches per small vessel

Table 3.4 shows, for *D. trunculus*, the minimum and maximum daily catches per small vessel per month during the years of the study. In 2011, 2012 and 2013 the minimum daily catches were between 0 and around 3.5 kg, and the maximum reached values of 100 kg in 2011, while in 2012 and 2013 barely passed the 50 kg.

In 2014 the minimum ranged between 0-1.5 kg and the maximum decreased notably (between 2 and 19.5 kg). In 2015, these values were between a minimum of 1.4-18 kg a maximum of 4.3 and 32 kg.

Table 3.4. Minimum and maximum daily catches (kg) of *D. tunculus* per small vessel per month between 2011 and 2015. ND: no data and (----) month without fishing for this species.

Year	2011	2012	2013	2014	2015
	Min-Max	Min-Max	Min-Max	Min-Max	Min-Max
January	ND	2-33.5	1-7.5	1-8	1.4-4.7
February	ND	0-33	1-9.5	1.5-15.5	3.9-4.3
March	ND	2-32	0-21	0.5-19.5	1.9-7.4
April	ND	2-52	0-26	0-16.5	3-26
May	ND	1-49	0-34	0-12.5	18-32
June	3-54	2-47	1.5-35	0-14.5	-----
July	0-38.5	1-28	0-50.5	0-15.5	-----
August	2.5-100	0-17.5	0-29	0-10.5	-----
September	0-21.5	-----	-----	1-2.5	-----
October	0-17.5	0.5-3.5	2-24.5	0-3	-----
November	0.5-9	1-2	3.5-7	0.7-2	-----
December	0-17.5	0.5-2.5	2-10.5	2	-----

Daily catches per small vessel for *C. gallina* were nonexistent during most of the months and years in the period from June 2011 to May 2015, highlighting only June 2011, September 2011, July 2014 and August 2014 with ranges of 19-37.5, 2.5-23.5, 4.5-30.5 and 4-26.5 kg respectively.

Also, in Figure 3.4 we can observe the catch-frequency distribution of the daily catches per small vessel, from June 2011 to May 2015, for *D. trunculus*. The limits of daily catches established by the legislation are also shown. Daily catches per small vessel were analysed based on the daily maximums established by the existing regulations between 2011 and 2015. Based on these standards, catches of virtually 100 % of the trips were lower than the lowest quota established for both species per small vessel, that is to say less than 65 kg from June 2011 to July 2013, and less than 40 kg between 2013 and 2015 (Fig. 3.4). The only exception occurred in August 2011 during a trip for *D. trunculus* (representing 0.4 % of total trips in August 2011) in which 100 kg were caught. In the case of on-foot shellfishers, their catches were always less than 5 kg d^{-1}, clearly below the allowable limits. In the case of C. *gallina*, the highest daily averages were presented in 2011 and 2014, reaching an accumulated frequency of catches of 80 % between 20-25 kg. In other years the catches were less than 5 kg.

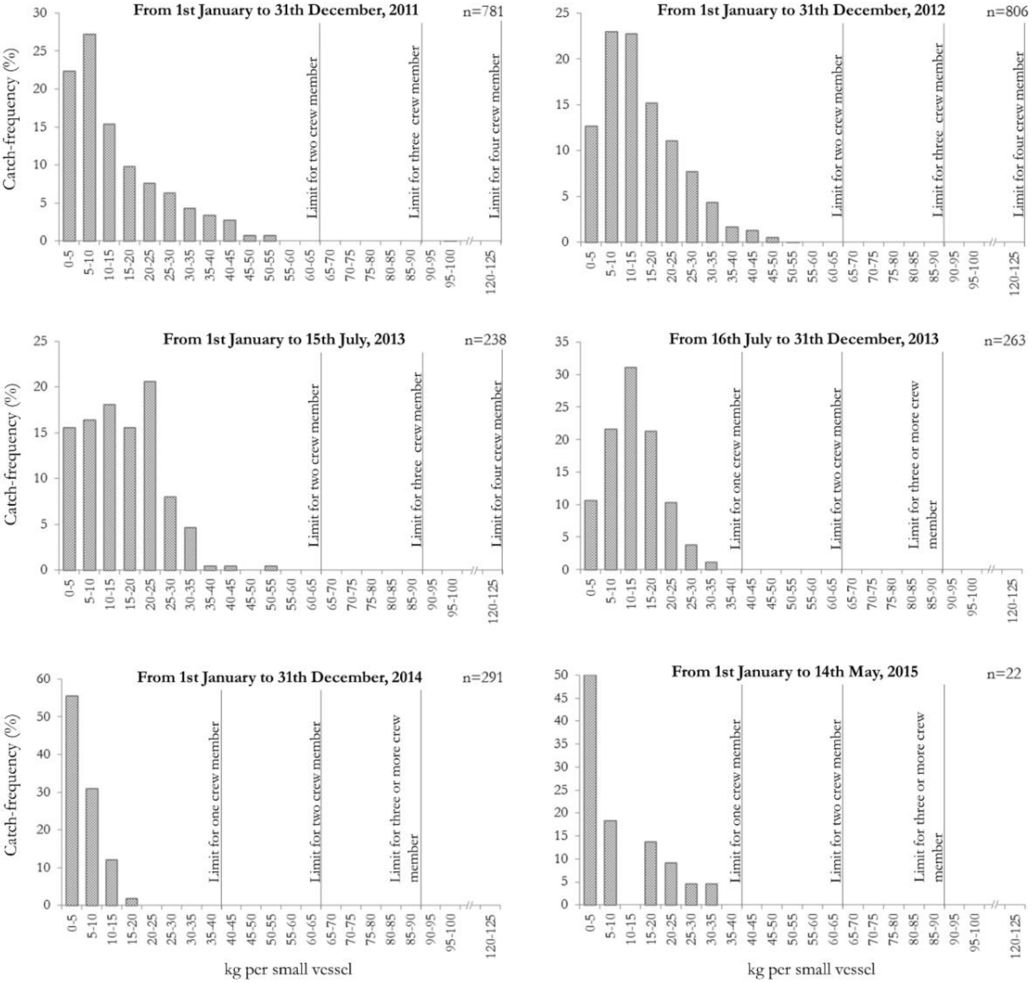

Figure 3.4. Catch-frequency distribution of the daily catches per small vessel of *D. trunculus*, from June 2011 to May 2015. The year 2013 presents two graphs due a change in the regulation.

3.4.4 Coastal zoning

Table 3.5 presents, for *D. trunculus*, the annual catches, fishing effort, CPUE and catches per km of coast, in each one of the areas of the study zone during the period 2012-2014. In 2012, areas V, VI and VII were the ones with the highest catches, going beyond 2000 kg per year in each of them, while in 2013, area VII was the only one that exceeded this value. Finally, in 2014 there was a marked decrease in all the areas, registering the highest catches in areas V and VII, with 340 and 504 kg yr^{-1} respectively. A similar distribution to that of the catches in each area was observed for fishing effort.

Regarding CPUE, it was between 6.8 (area I) and 16.0 (area IX) in 2012. In 2013, this range was similar, but in different areas (6.8 in area III and 16.3 in area II). However, in 2014 the maximum value was close to the minimum of the previous two years with only 7.9 in area IX and the minimum of 1.2 in area III.

The maximum catches per km of coastline in 2012 were found in areas V and VI with values around 1000 kg km^{-1}. Areas IV, V, VI and VII had the highest number of catches per km in 2013, although they were between 350 and 500 kg km^{-1}. For 2014, the maximum fell and again areas IV, V and VII showed the greatest catches per km, with approximately 170 kg km^{-1} in the first two areas and about 100 kg km^{-1} in area VII.

In the case of *C. gallina*, zoning for the years 2012 and 2013 does not make sense due to the negligible total catches made (Table 3.3). In 2014, area IV produced 680 kg followed by areas V (98 kg) and VII (21 kg). Regarding the fishing effort and CPUE, areas IV, V and VII were also the ones with the highest values with 43, 8 and 2 trips and 15.8, 12.2 and 10.4 kg $vessel^{-1}$ d^{-1} respectively. With regard to catches per km of coastline, in 2014 area IV showed the greatest value with 746.5 kg km^{-1} followed by areas V and VII, although with much lower catches of only 48.9 kg km^{-1} and 4.1 kg km^{-1} respectively.

Table 3.5. Annual catches, fishing effort, CPUE and catches per km of coast in each one of the ten fishing areas for *D. trunculus* during the period 2012-2014. NC refers to catches without specified area.

		Catches (kg)			Fishing effort (trips)			CPUE			Catches per km (kg km^{-1})		
	Year	**2012**	**2013**	**2014**	**2012**	**2013**	**2014**	**2012**	**2013**	**2014**	**2012**	**2013**	**2014**
Area	**I**	270	418	10	40	28	4	6.8	14.9	2.4	51	79	2
	II	659	781	43	71	48	14	9.3	16.3	3.1	161	190	10
	III	1187	196	4	123	29	3	9.7	6.8	1.2	297	49	1
	IV	488	316	161	43	37	40	11.4	8.5	4.0	537	347	177
	V	2129	815	340	178	77	69	12.0	10.6	4.9	1064	408	170
	VI	2566	855	160	184	70	47	13.9	12.2	3.4	1199	399	75
	VII	3888	2617	504	293	187	90	13.3	14.0	5.6	767	516	99
	VIII	219	407	122	23	34	19	9.5	12.0	6.4	114	212	64
	IX	496	552	222	31	43	28	16.0	12.8	7.9	103	115	46
	X	210	177	2	14	18	1	15.0	9.8	1.5	10	8	0
	NC	160	138	31	13	14	8	12.3	9.8	3.9	---	---	---

3.4.5 Monthly price evolution

Regarding price evolution of *D. trunculus* (Table 3.6), in 2004 and 2005 the minimum values of about 4.5 € kg^{-1} were reported in March and April while the highest prices were found in August. In 2006, February had the lowest price (3.4 € kg^{-1}), and the highest value was 11.4 € kg^{-1} in December, followed by August. Between 2007 and 2014, July and mostly August were the months with the highest price, varying between 10 and 22 € kg^{-1}. Finally, in 2015, the year in which the fishery was closed at the beginning of June, the minimum price was presented in February with a value of 7.5 € kg^{-1}, while the maximum was 14.1 € kg^{-1} in March.

The highest prices for *C. gallina* varied from 7.0 to 11.1 € kg^{-1} and they were generally observed in summer (July and August) and winter (December and February) (Table 3.6). The minimums were around 3.5 € kg^{-1} in the period 2004-2008, however, from 2009, they increased to 7.8 € kg^{-1} in 2011. For the years 2012, 2013 and 2015 only the monthly prices in which *C. gallina* was collected are shown in Table 3.6.

Table 3.6. Minimum and maximum average monthly selling price (€ kg^{-1}) of *D. trunculus* and *C. gallina.*

Year	***D. trunculus*** **Min-Max**	***C. gallina*** **Min-Max**
2004	4.3(Mar)-13.9(Aug)	3.4(Nov)-10.4(May)
2005	4.5(Apr)-9.3(Aug)	3.5(Jan)-8.6(Aug)
2006	3.4(Feb)-11.4(Dec)	4.1(Sep)-8.6(May)
2007	4.3(Apr)-13.6(Aug)	3.6(Feb)-9.0(Aug)
2008	3.2(Feb)-10.0(Aug)	3.5(Apr)-7.0(Dec)
2009	4.0(Apr)-12.6(Aug)	6.3(Mar)-9.0(Aug)
2010	5.6(Apr)-21.6(Aug)	5.5(Apr)-11.1(Dec)
2011	7.7(Feb)-19.3(Jul)	7.8(May)-11.1(Jul)
2012	7.5(Jan)-18.8(Aug)	8.0(Feb)
2013	6.0(Apr)-19.0(Aug)	7.0(Oct)
2014	8.4(Mar)-21.6(Aug)	5.9(Nov)-8.0(Feb)
2015	7.5(Feb)-14.1(Mar)	8.0(Jun)

3.5 DISCUSSION

Regarding annual catches during the period 2004-2014, it is worth noting how important the Gandia fishery is when compared to the catches made along the whole Spanish Mediterranean coast (Tables 3.2 and 3.3). Between 2004 and 2008, *D. trunculus* catches from Gandia represented around 40 % of the total catches in the whole Spanish Mediterranean, except 2007 when they dropped to 30 %. From 2009, a clear downward trend begins with 30 % for that year, reaching 2 % in 2014. In the case of *C. gallina*, the percentage caught by Gandia in the whole Spanish Mediterranean was lower, presenting the highest values in 2007 and 2008 with 27 and 33 % respectively while the rest of the years were between 0 and 3 %. In both cases, the trends in catches in Gandia and the rest of the Spanish Mediterranean were different, showing a very sharp decrease in our study area. This decline could have been due to deficiencies in the métier's management plans, which is reflected in the CPUE (Tables 3.2 and 3.3). The CPUE is considered as an index of relative abundance (Gatica and Hernández, 2003), and gives us information about the state of species exploitation. The decrease in CPUE values in recent years could indicate that the fishery had suffered overexploitation. With regard to 2007, the atypical year in which there was a specific decline in catches of *D. trunculus*, it should be noted that it was the result of less fishing effort, with a relatively stable CPUE (Table 3.2). In that year, 2007, 35 % of the small vessels that were dedicated to this fishery on a regular basis, both in previous and subsequent years, dedicated a greater fishing effort to another fishery such as the octopus (unpublished data). It is interesting to note that the decline in catches during 2007 was not only observed in Gandia, but also occurred in a generalized manner in the Spanish Mediterranean, which could also have been the result of poor general conditions of the sea for fishing in shallow waters.

Previous studies in the Mediterranean Sea, carried out by Romanelli et al. (2009) showed a decline of *C. gallina* in the Adriatic Sea from the mid-1980s, despite the management measures implemented. In the same study, they concluded that the reduction of freshwater inputs to the sea was a relevant factor in the decrease of the population of this species. In the Catalan coast (Northwestern Mediterranean) Baeta et al. (2018) observed a decline in *C. gallina* catches from 2000 to 2015 due to high fishing pressure and inadequate management. In addition, Ramón et al. (2005) remarked that beds of this clam in Cullera and Gandia had partially disappeared. *C. gallina*, as mentioned above, inhabits sandy bottoms up to 12 m deep, however, fishermen only collect them up to 5 m, as they are limited by their habitual fishing gear (personal communication secretary manager of Gandia Fishermen's Guild). This indicates that there is a very large bathymetric range that is not used, which should be studied in order to determine the reason why these no-take zones did not help to recover the fishery that was carried out in shallower waters.

Declines in catches for *D. trunculus* were also observed in the Mediterranean coasts of Lazio, where changes in sediment granulometry were considered the most likely cause (Lart et al., 2003). Baeta et al. (2018), in the Catalan coast, observed a marked drop in *D. trunculus* landings between 2007 and 2015 as a consequence of high fishing pressure, inadequate management plus undetermined factors. The reduction of *D. trunculus* catches in Gandia, along with a decrease in the fleet and fishing effort, led to relatively stable values of CPUE from 2004 to 2008. According to fishermen, a minimum of 20 kg per working day should be obtained in order to be profitable per small vessel. However, from 2010, it was observed that the CPUE dropped to values below 19, and despite this, the fishery still continued for another four years. This would indicate that one way or another it was profitable to continue with the exploitation of this species, leading to the suspicion that part of the catches may not have been declared.

On the other hand, from the monthly evolution of this fishery, a marked seasonality with an increase in fishing effort, catches, CPUE, fishing fleet and price in spring and summer has been observed (Fig. 3.3A and B and Table 3.6). Due to the rising temperature in spring and summer, there is an increase in physiological processes, such as growth and reproduction (Ansell and Lagardère, 1980; Manca Zeichen et al., 2002), so these organisms require a greater source of food owing to a high metabolic demand (Dionicio and Flores, 2015). Therefore, when the highest temperatures are present, these clams could be located closer to the sediment surface in order to have easy access to food, thus more accessible for fishing. In addition, the summer months showed the greatest price increases of *D. trunculus* and *C. gallina* caused by gastronomic interest fuelled by the rise of tourists in the area thus further increasing the seasonality and profit margin of this métier. Ramón (1993) also observed the price effect on fishing activity along the Cullera coast located immediately north of the study area. Also, Sardá et al. (2000) observed seasonality in these types of bivalve fishing during spring and summer in the Bay of Blanes (Spanish Mediterranean coast), which was linked to the most favourable sea conditions. In shallower areas, it is important to take into account the sea conditions, as bad weather could cause serious accidents for fishermen. The increase in fishing pressure in spring and summer coincides with the period of reproduction of these organisms which could affect the survival of the species. The measures of management of this fishery in the study zone do not take into account seasonal closure. It would be advisable to maintain periods of closure during the reproductive and recruitment periods, which could be the months of July and

August, to facilitate reproductive potential of natural beds. These periods of closure of shellfish fishing during reproduction are already carried out for both species in other parts of Spain such as Catalonia and Andalusia as well as in Portugal (Andalusia Community Order of March 25, 2003; Lart et al., 2003; Silva et al., 2014a; GENCAT, 2015; Andalusia Community Order of February 22, 2018). In Italy, the seasonal closure is to ensure the settling of larvae of *C. gallina* (Padella and Finco, 2009). However, as the July and August are months with a high demand for bivalves and therefore have a high price in the market, a period of closure could result in socioeconomic repercussions, although this repercussion could be compensated by a recovery of natural population. As an alternative to a closure period, no-take zones could be established during the period with the highest market value in order to avoid a serious socioeconomic repercussion.

Together with periods of cessation, a measure to take into account for not compromise the reproduction potential of *D. trunculus* would be to establish a more appropriate minimum landing size, taking into account the biology of the species. This size is fixed at 14 mm, which is close to the size at first maturation, established by Ramón (1993) near the zone of study (12.7mm). In other areas of Spain such as the Gulf of Cadiz and Catalonia and countries like Portugal, there is a minimum landing size established at 25 mm (Lart et al., 2003; Silva et al., 2014a; GENCAT, 2015; Andalusia Community Order of February 22, 2018). Since the increase in size is accompanied by an increase in the production of oocytes (Silva and Juárez, 2009. In: Delgado et al., 2013), the extraction of organisms that do not reach the sufficient size to maximize its reproductive potential could affect the survival of the species. Moreover, the overfishing of large individuals would considerably reduce the production of oocytes which would affect the structure of the population and recruitment (Silva and Juárez, 2009. In: Delgado et al., 2013). Recruitment is also subject to environmental conditions as well as by predation. Recruitment is an important component due to the population of bivalves strongly depends on recruitment for its conservation and sustainability (Rufino et al., 2018). In the case of *C. gallina*, the minimum landing size marked in the zone of study is 25 mm since

July 2013, and is equal to the rest of the mentioned zones of production (Lart et al., 2003; Padella and Finco, 2009; Silva et al., 2014b; GENCAT, 2015).

Regarding daily catches, is interesting to note that the catch frequency was rarely close to the established set maximums (Fig. 3.4). The annual maximum catches were only established for 2013 and 2014, but these limits were set for the whole Gulf of Valencia fishery. Taking into account that Gandia represented 60 % of the catches in the Gulf of Valencia (GVA, 2016), we can assert that the annual catches were not close to the annual maximums established (Tables 3.2 and 3.3). Therefore, it could be said that the prior and current legislations regarding quotas are not adequate to control catches of both species and/or a relevant part of catches could have been illegally traded, as Baeta et al. (2018) observed in the Catalan coast, where 1-20 % of the catches were sold outside official channels. A deficiency in the allocated quota as a means of management of this métier had also been observed by Ramón (1993) in Cullera, where a maximum catch of 65 kg per day per two crew member vessel had been established via internal agreements by the Cullera Fishermen's Guild. Quotas for fisheries management have been used in many cases, although not always successfully (Morgan, 1997). In some cases, these quotas were established at very high values against scientific recommendations (Santos-Martín et al., 2015) or were not even set up under scientific advice (Baeta et al., 2018). In Portugal, the change from daily to weekly quotas favourably increased productivity due to reduction of fishing days and fuel consumption (Oliveira et al., 2013). Sometimes, the maximum daily catches are fixed to maintain economic sustainability and not to sustain exploited populations, as Morales-Nin et al. (2010) observed in the Balearic Islands (Western Mediterranean).

It is worth pointing out that the quotas in the zone of study were fixed by the legislation on the basis of studies and the monitoring of fishing activity at its beginning (1988-1995) (GVA, 2016). Consequently, these quotas do not reflect an adaption to the evolution of the state of the natural population and remain obsolete for the correct management of fishery. The daily quotas should be established based on monitoring current surveys of bivalve beds. For example, in the fishing of *D.*

trunculus in Andalusia, a monitoring of the fishing activity has been carried out since 2013 in order to improve and adjust its management and to reach sustainability (Silva et al., 2014a) and since 2007 for *C. gallina* (Silva et al., 2014b). In the legislation of our zone of study, the management plan was established in 2016 by the Valencian Community Decree 62/2016. Coastal zone management should be carried out from an ecological and biological perspective to avoid fishing problems and favour the regeneration of resources, therefore controlling fishing effort based on the available resources and closing fisheries when necessary (Bombace, 1991). Defeo et al. (2016) observed that co-management plans, where the government and local users share the management tasks and responsibilities, helped to improve fisheries. Baeta et al. (2018) pointed out the need to carry out new management plans favouring co-management based on the eight principles established by Ostrom (1990).

In general, the areas that presented the greatest fishing effort and catches for *D. trunculus* were areas V, VI and VII located south of Gandia Port (Table 3.5). However, it should be noted that these were not always the ones that presented the highest CPUE. For *C. gallina*, the highest values of CPUE, catches and fishing effort were found in the two areas immediately south of Gandia Port (areas IV and V). A greater fishing effort dedicated in the areas closest to the port is a consequence of shorter commuting time so optimizing the time and fuel costs dedicated to fishing. Moreover, the areas toward the south of Gandia Port are directly affected by the River Serpis -as dominant currents come from the North (Sebastiá and Rodilla, 2013)- which could favour higher productivity. As a result, these areas had the highest catches per km of coastline and suffered the greatest fishing pressure. In order to improve fishery management, densities of each bivalve bed should be taken into account in each regulated shellfish area. Managing each zone separately would help to distribute the fishing pressure on them, based on the state of populations and the extension of each zone. This management could be carried out by distributing the working days in each of the zones or by rotating the opening of zones or, as was done in Catalonia, by establishing independent quotas in each one (GENCAT, 2015), in order to distribute the fishing pressure. It is important to bear in mind that, in summer periods, the working day finishes earlier due to the presence of bathers, which makes it difficult to exploit the areas farthest from the port due to length of commuting time.

3.6 CONCLUSION

In the study area, the métier is mainly dominated by *D. trunculus* catches, as the fishing effort dedicated to *C. gallina* was highly variable over the years due to the low catches.

Lack of suitable management measures in response to the decline in CPUE, triggered a sharp drop from 2010 that led to the fishery closure in June 2015, with extremely low CPUEs in 2014.

Fishery management by quotas was not adequate, because established maximums were so high that the catches were not limited by them, thus showing a deficit in the management measures. These quotes must be established according to the state of the natural population; therefore, they must be monitored.

Deficiencies in the métier management, in addition to the effect on natural populations of both species, led to socioeconomic problems. The small vessels, with 1 or 2 crew members, that habitually worked this artisanal fishery had difficulty adapting to other fisheries, as the occasional fleet did, which is why some of them ceased their activity.

In addition, the exploitation of these clams had a markedly seasonal character, concentrating mainly in spring-summer. In order to guarantee the sustainability of the exploitable beds a closure period is proposed of at least two months during spring and summer, which coincides with the reproduction and start of the recruitment period. An alternative would be the establishment of no-take zones in order to avoid the consequence of serious socioeconomic repercussions that could be caused by the establishment of cessations during the months with the highest economic value and demand of these bivalves.

In future, good management measures would be required in order to regulate spatial exploitation of the entire fishing zones according to natural populations densities in each area, guaranteeing no-take zones that would facilitate species reproduction, as well as avoiding intensive exploitation of the areas closest to the port.

Other aspects such as the minimum landing size of *D. trunculus* must be revised, taking into account the biology of the species and that guarantees an efficient reproduction of the species.

3.7 REFERENCES

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Capítulo 4

Changes of *Donax trunculus* populations after the fishery's closure in Gandia (Western Mediterranean Sea, Spain)

4. CHANGES OF *Donax trunculus* POPULATIONS AFTER THE FISHERY'S CLOSURE IN GANDIA (WESTERN MEDITERRANEAN SEA, SPAIN)

4.1 ABSTRACT

In the sandy shores of the Spanish Mediterranean, *Donax trunculus* Linnaeus, 1758 has a high commercial interest. In the Gulf of Valencia, the catches of this clam showed a marked decline in the last decade leading to the closure of the fishery at the beginning of June 2015. The objective of this study was to analyse densities and biomass of *D. trunculus* in the main fishing area of the Gulf of Valencia, which belongs to the Gandia Fishermen's Guild's fleet, just before the closure of this fishery and its subsequent five-month evolution, as well as the state of the population 2 years after the closure. Moreover, some ecological and fishing characteristics were analysed. The growth of *D. trunculus* in this study showed some differences to other studies from different geographical areas, highlighting that the values obtained for asymptotic length and growth constant in 2015 were lower than those obtained 25 years ago in the same region. Furthermore, the population showed size differential distribution between depths and a period of recruitment in summer. The biomass and density evolution showed a seasonal pattern in both variables with maximum values in summer. In 2015, before the fishery closure, biomass ranged from < 0.05 to 120.56 kg ha^{-1} and during the five subsequent months after the closure from 7.79 to 102.43 kg ha^{-1}. However, two years later, in 2017, a considerable decrease in both biomass and density were observed showing minimum values of 0.20 kg ha^{-1} and maximum values of 17.33 kg ha^{-1} for biomass. This indicate that the closure did not improve the state of the population, therefore, other variables which could be affecting the abundance or survival of this bivalve must be analysed and taken into account for further management strategies.

Keywords: clam; wedge clam; bivalve fishery; population structure; fishing gear selectivity

4.2 INTRODUCTION

The wedge clam *Donax trunculus* Linnaeus, 1758 is a filter feeder organism that live on sandy beaches in highly energetic environments, due to their ability to bury themselves in sediment (Ramón, 1993; Gaspar et al., 1999b; McLachlan and Brown, 2006). On the Mediterranean coast these organisms can be found up to 4 m deep, although they are preferentially distributed between 0 and 2 m deep (Ramón, 1993; Ramón et al., 1995; Gaspar et al., 2002; Manca Zeichen et al., 2002) and between 0 and 6 m on the Atlantic coast (Gaspar et al., 2002). In general, juveniles are found in shallower areas while adults are more abundant in deeper ones (Manca Zeichen et al. (2002), Deval (2009), Baeta et al. (2018) in Mediterranean coasts and Ansell and Lagardère (1980), Gaspar et al. (2002) in Atlantic coasts). However, in some areas, such as the Atlantic Moroccan coast, the inverse distribution has been observed by Bayed and Guillou (1985). This species shows a rapid increase in size and a short lifespan, lasting in the Spanish Mediterranean between 2 to 3 years (Ramón et al., 1995). *D. trunculus* has a high reproduction potential, due to its small size at first maturity and spawning between tens of thousands to millions of oocytes (Ramón, 1993; Tirado and Salas, 1998; Louzán, 2015; Delgado and Silva, 2018). Different factors can affect the survival and growth of this species such as temperature (Neuberger-Cywiak et al., 1990; Manca Zeichen et al., 2002), food availability (Ansell and Bodoy, 1979), sediment grain size (Mazé and Laborda, 1988; Lart et al., 2003; La Valle et al., 2011), parasites (Ramón et al., 1999; Ramadan and Ahmad, 2010; de Montaudouin et al., 2014; Delgado and Silva, 2018) and/or lethal or sublethal predation (Salas et al., 2001).

This species has a high commercial interest in the sandy shores of the Spanish Mediterranean. In the last two decades, fishing in the Valencia region has been almost limited to the fleet of the harbours of Cullera and Gandia. *D. trunculus* is harvested using two types of fishing gears: hand-operated dredges and small vessel-operated dredges. This fishery has never had appropriate management plans based on real data about the abundance and recruitment of *D. trunculus*, resulting in a decline in catches over the last decade that led to the closure of the fishery in June 2015.

The establishment of fishery management measures related to the improvement of the selectivity and efficiency of fishing gears and the protection of areas inhabited by juveniles are essential in order to avoid the overfishing of the target species (Gaspar et al., 2002). For some bivalves like *Spisula solida* Linnaeus, 1758 (Gaspar et al., 1999a; Gaspar et al., 2003), *Chamelea gallina* Linnaeus, 1758 (Sala et al., 2014), *Venus striatula* da Costa, 1778 and *Ensis siliqua* Linnaeus, 1758 (Gaspar et al., 1999a), the selectivity of fishing gears has been analysed. However, there is little research in the case of *D. trunculus*. One example for *D. trunculus* in the Spanish Mediterranean is the study made by Ramón (1993), in which the selectivity of fishing gear was analysed based on the mesh size of the dredges used. In other parts of Europe such as Portugal, the selectivity has been investigated based on mesh size (Lart et al., 2003), but with fishing gears different to those used in the Gulf of Valencia.

Furthermore, in order to establish adequate management measures, the biology and ecology of the species has to be considered (Gaspar et al., 2002). Several studies have been carried out about population structure and growth of *D. trunculus* in the Mediterranean Sea, in the Atlantic region and Sea of Marmara (Ansell and Lagardère, 1980; Bodoy, 1982; Mazé and Laborda, 1988; Neuberger-Cywiak et al., 1990; Ramón, 1993; Ramón et al., 1995; Gaspar et al., 1999b; Manca Zeichen et al., 2002; Deval, 2009; Çolakoğlu, 2014; Hafsaoui et al., 2016; Delgado et al., 2017). The von Bertalanffy growth parameters obtained by these authors were estimated generally from length-frequency distributions, although in some of them the parameters were estimated from acetate peel method, thin section or external growth rings.

If the ecological data of the species is not taken into account when establishing the fishing management measures, it can cause a lack of sustainability of the natural beds leading to the decline and consequently the fishery closure. Moreover, the study of the fishing gear efficiency is another point to take into consideration. The objective of this research was to analyse *D. trunculus* densities and biomass before the closure of this fishery and its subsequent five-month evolution, and the state of the population 2 years after the closure.

4.3 MATERIAL AND METHODS

The area of study is located in the sandy beaches in the southern sector of the Gulf of Valencia in the western Mediterranean Sea, off the eastern coast of Spain. It encompasses all the beaches from Tavernes de la Valldigna until Denia, the area which belongs to the Gandia Fishermen's Guild's fleet, dedicated to the fishery of *D. trunculus* (Fig. 4.1). These coasts are composed of sandy sea beds with a high proportion of fine sands which increases with depth (Ramón, 1993). Two sampling campaigns were carried out: the first one in 2015 that included the month just before the fishery's closure and the five subsequent months, and the second one two years later in 2017 with 4 months of duration. Eleven sampling stations (St1, …, St11) were located in order to cover heterogeneity of population, according to the areas previously determined by the Regional Ministry of Agriculture, Fisheries, Food and Water as production areas for bivalve molluscs in the Valencian Community waters (Valencian Community Resolution of February 19, 2013). Between May and November 2015, samples were obtained at two depths at each station, the first one from 0.5 to 1.5 m of depth (bathymetric range 1 or Stx_1) and the second one between 1.5 m and 3 m of depth (bathymetric range 2 or Stx_2). The first sampling was taken at each station between late May and early June, 2015 just before the closure of the fishery on June 10th, 2015 (Valencian Community Resolution of June 3, 2015). Furthermore, immediately after the closure, monthly samplings were taken at stations St1, St4, St6, St8, St10 and St11, and bimonthly at stations St3, St5, St7 and St9 between July and November, 2015. In 2017, between June and September, samples were taken monthly at five stations (St1, St3, St4, St7 and St8), but only in the bathymetric range 1.

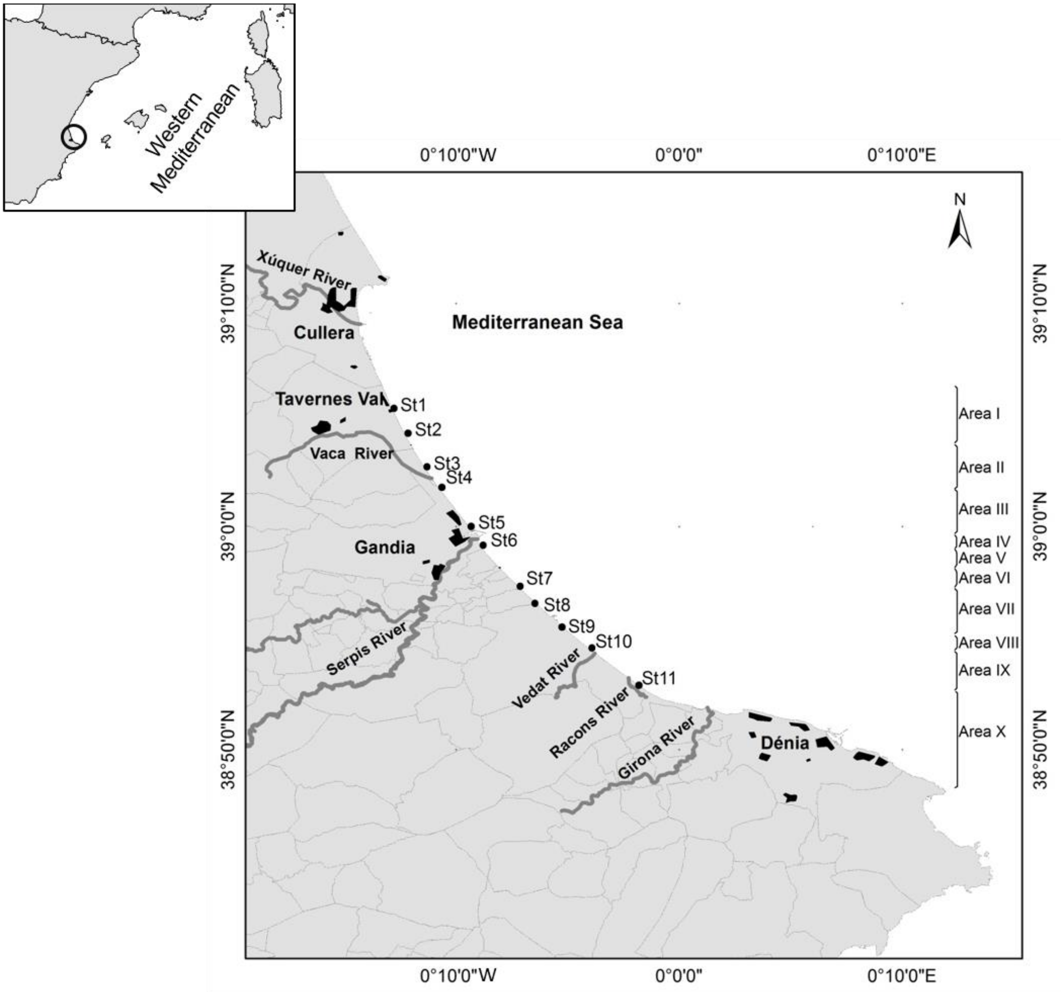

Figure 4.1. Study area with sampling station

The samples were collected using a shellfishing vessel specially equipped with four dredges (70 cm wide each one) to catch *D. trunculus.* Dredges were towed, parallel to the coastline, over a standard length of 100 m, every dredge covering an area of 70 m^2. Three of the four dredges had a metallic grid with a mesh size of 11.7 x 11.7 mm, which is standardly used at this fishery, and the fourth had a mesh size of 5 x 5 mm in order to catch individuals of different sizes of adults and juveniles. The commercial size of this species was established in 2013 as 14 mm (Valencian Community Decree 94/2013), considering this value, in this study, as the minimum size of adults. A similar size (15 mm) was used by Manca Zeichen et al. (2002) to divide between juveniles and adults.

The anterior-posterior length of each clam caught in the 5 mm mesh size dredge was measured to the nearest 0.1 mm using a calliper. To obtain the relationship between biomass and length of the wedge clam, the anterior-posterior length of individuals of all size classes were measured and weighed total mass (shell + tissue) (precision ± 0.01 g). The parameters of this relationship were estimated by regression analysis

$$M=a\,L^{b} \qquad (1)$$

where M is the total mass (g), L the shell length (mm), a is the ordinate at origin and b is the slope. The coefficient of determination (r^2) was used as an indicator of the quality of linear regression.

Total biomass and density of *D. trunculus* were determined for each sampling station in the 5 mm mesh size dredge for all samples. Temporal differences of the population density and biomass were analysed statistically by one-way ANOVA test, and the Tukey *post hoc* test to identify significant differences between groups, only for those stations sampled monthly in 2015 and 2017. These tests were carried out using the software package Statgraphics Centurion XVII.

The length-frequency distribution was calculated for each month and sampling site, separating individuals at intervals of 1 mm. The length-frequency distribution obtained was divided into cohorts applying the Bhattacharya's method using FISAT II tool (Gayanilo et al., 2005). Differences between size frequency distributions among depths, in the case of samples taken in 2015, were analyzed on the basis of data pooled over time for the whole period, for each sampling station. The Kolmogorov-Smirnov goodness of fit test (K-S) for two samples was applied, with a significance level of 95 % ($\alpha = 0.05$).

The von Bertalanffy growth model (VBGF) parameters were estimated for 2015 and 2017 from an analysis of the length-frequency distribution of all the samples, using 'ELEFAN I' routine of the FISAT II software package (Gayanilo et al., 2005), which estimates the growth parameters: asymptotic length (L_∞) and growth constant (K). Estimated L_∞ (mm) and K (yr^{-1}) were used to calculate the growth performance index (ϕ'), used to compare growth in terms of length (Pauly and Munro, 1984), and the overall growth performance index (OGP) (Pauly, 1979), defined as

$$\phi' = 2 \log_{10} (L_\infty) + \log_{10} K \qquad (2)$$

$$\mathrm{OGP} = \mathrm{Log}\,(K\,[L_\infty]^3) \qquad (3)$$

Furthermore, a t-test was carried out to determine if significant differences existed between the ϕ' of the current study and the data obtained in the literature.

In addition, organisms caught with the other three dredges of 11.7 mm in size were photographed *in situ* and returned to the sea. The individual sizes were obtained by means of digital analysis using the programme ImageJ. The fishing gear selectivity was analysed for all samples taken in 2015, through length-frequency distribution comparison between 5 mm and 11.7 mm mesh sizes dredges. Also, data on mesh size selectivity was analysed and the parameters obtained using FISAT II (Gayanilo et al., 2005).

Monthly seawater temperatures for the area of study were obtained from NASA Goddard Space Flight Center (Ocean Biology Processing Group, 2018) for the periods May to November 2015 and June to September 2017.

4.4 RESULTS

4.4.1 Study of populations in 2015

4.4.1.1 Biomass and density evolution

The first sampling campaign was carried out in 2015, between May and November. The mean sea surface temperature increased from late May-June (21.7 ºC) until August (26.8 ºC) and from the end of summer, a temperature decrease was noted with values of 24.9 ºC in September, 22.3 ºC in October and 19.3 ºC in November (NASA Goddard Space Flight Center, Ocean Biology Processing Group, 2018).

The observed relationship between mass and length of *D. trunculus* (n = 193) was $M=9x10^{-5}L^{3.1433}$, with an $r^2=0.9782$. The regression analysis demonstrated significant correlation between the mass and length ($p < 0.0001$). The equation obtained was employed to determine the weight of the rest of the individuals.

Biomass in 5 mm mesh size dredge is shown just before and five months after the fishery closure in both bathymetric ranges in Table 4.1. In late May-early June 2015, total biomass ranged between 9 and 121 kg ha^{-1} for the bathymetric range 1, while the minimum and maximum were lower for the bathymetric range 2 (< 0.05 and 93 kg ha^{-1} respectively). The commercial size individuals (≥14 mm) represented between 83.1 and 98.5 % of the total biomass at bathymetry 1, and varied between 87.4 and 100% at the deepest range. After the closure, biomass fluctuated between 11 and 86 kg ha^{-1} at bathymetric range 1 while the bathymetry 2 covered a wider range of values between 8 and 102 kg ha^{-1}. In this period, commercial size individuals represented between 66.7 and 99.1 % of the total biomass in bathymetric range 1 and from 78.6 to 99.9 % in bathymetry 2. The sampling station which showed the highest biomass before the fishery closure was St3 in both bathymetric ranges, followed by St7_1 and St4_1. After the closure, the highest value was observed at St3_2 in August (102.43 kg ha^{-1}), while the lowest was located at St11_2 in October (7.79 kg ha^{-1}).

Table 4.1. Total biomass (in kg ha^{-1}) of *D. trunculus* in 5 mm mesh size dredge before and after the closure of the fishery in 2015. The values of *D. trunculus* commercial size ($\geq$ 14 mm) are shown in brackets.

Sampling site	**Just before the closure**	**After the closure**				
		Jul-15	**Aug-15**	**Sept-15**	**Oct-15**	**Nov-15**
St1_1	8.73 (7.76)	58.63 (52.64)	32.12 (26.28)	55.04 (40.97)	42.50 (39.29)	22.05 (21.47)
St1_2	5.19 (5.19)	63.07 (62.56)	54.04 (51.70)	59.28 (54.38)	55.41 (51.56)	34.72 (34.36)
St2_1	22.89 (20.53)					
St2_2	< 0.05					
St3_1	120.56 (108.05)		68.94 (66.23)		25.11 (22.21)	
St3_2	92.88 (86.23)		102.43 (100.66)		49.62 (44.18)	
St4_1	66.51 (60.89)	77.32 (73.27)	43.08 (38.64)	18.14 (13.05)	30.78 (26.98)	24.62 (23.41)
St4_2	10.43 (9.67)	76.06 (74.44)	72.52 (69.38)	29.53 (25.03)	22.37 (20.38)	47.26 (45.13)
St5_1	16.55 (14.96)	18.17 (17.71)		12.35 (9.17)		11.26 (10.46)
St5_2	16.23 (15.25)	14.64 (12.59)		7.97 (7.20)		15.59 (14.78)
St6_1	17.33 (16.68)	59.17 (58.57)	13.29 (12.77)	31.32 (27.71)	25.10 (24.51)	13.94 (13.66)
St6_2	16.11 (15.41)	25.70 (25.01)	35.87 (34.36)	25.72 (22.96)	45.01 (44.36)	24.61 (24.15)
St7_1	78.35 (77.14)	77.57 (66.96)		85.66 (75.26)		25.29 (24.66)
St7_2	41.72 (40.32)	85.46 (79.58)		49.47 (43.15)		40.35 (39.76)
St8_1	34.54 (28.70)	28.47 (26.18)	19.54 (13.03)	20.63 (14.58)	34.44 (32.81)	34.64 (33.89)
St8_2	59.68 (58.04)	40.95 (39.00)	67.00 (59.81)	35.95 (31.45)	29.61 (28.43)	32.21 (31.75)
St9_1	26.35 (25.80)		45.17 (43.08)		19.76 (18.36)	
St9_2	16.59 (16.36)		27.32 (21.47)		22.39 (21.47)	
St10_1	20.47 (19.20)		22.94 (20.39)	11.60 (10.57)	26.40 (25.62)	27.40 (27.14)
St10_2	9.40 (8.22)		12.45 (11.75)	8.43 (7.99)	14.03 (13.80)	15.18 (15.02)
St11_1	21.67 (20.16)	41.60 (40.99)	28.47 (26.34)	53.93 (46.58)	12.37 (11.41)	16.58 (16.26)
St11_2	11.44 (11.14)	43.60 (43.56)	15.98 (14.95)	16.16 (14.67)	7.79 (7.21)	12.90 (12.50)

Total density of the sampling sites sampled monthly is shown in Table 4.2. In late May-early June a mean density of 2.68 ind m^{-2} was observed which increased to values higher than 5 ind m^{-2} in July, August and September, after the fishery closure. From October, the mean density decreased and in November reached a value similar to that obtained before the closure, albeit slightly higher. The densities varied between 0.39 and 7.29 ind m^{-2} before the closure, showing the minimum at St1_2 and the maximum at St4_1. Following the closure, the minimum densities varied between 1.10 and 3.00 ind m^{-2}, showing the lowest density at St6_1 in November, while the maximum values ranged between 5.91 and 12.36 ind m^{-2}, the highest density having been obtained in September at ST1_1. The mean densities and minimum values during the subsequent months after the closure were always higher than before the closure, while the maximum density in November was lower than the maximum in late May-early June.

Table 4.2. Total density (ind m^{-2}) of *D. trunculus* in 5 mm mesh size dredge of the sampling sites sampled monthly in 2015, before and after the closure of the fishery. Also the mean, median, maximum and minimum data are presented. The values of *D. trunculus* commercial size (≥ 14 mm) are shown in brackets.

Sampling site	**Just before the closure**	**After the closure**				
		Jul-15	**Aug-15**	**Sept-15**	**Oct-15**	**Nov -15**
St1_1	0.97 (0.61)	9.64 (5.69)	5.99 (2.66)	12.36 (4.64)	5.64 (4.33)	2.36 (2.04)
St1_2	0.39 (0.39)	6.03 (5.67)	6.47 (5.04)	7.40 (5.03)	7.74 (6.14)	3.29 (3.11)
St4_1	7.29 (4.77)	9.09 (7.13)	6.49 (4.23)	4.47 (1.93)	5.14 (3.57)	3.11 (2.63)
St4_2	1.14 (0.80)	7.53 (6.84)	7.76 (6.27)	5.04 (2.83)	3.59 (2.63)	5.91 (4.97)
St6_1	1.33 (1.04)	5.04 (4.64)	1.43 (1.19)	4.59 (2.47)	2.41 (2.11)	1.10 (0.97)
St6_2	1.60 (1.33)	3.20 (2.87)	4.73 (3.76)	3.89 (2.57)	3.96 (3.70)	2.10 (1.89)
St8_1	6.50 (2.14)	3.49 (2.19)	5.11 (1.57)	4.94 (1.94)	3.99 (3.26)	3.80 (3.31)
St8_2	4.74 (3.66)	4.79 (3.91)	9.50 (6.04)	5.80 (3.61)	3.71 (3.21)	3.20 (3.01)
St11_1	1.97 (1.39)	3.73 (3.43)	3.49 (2.36)	8.26 (5.23)	2.09 (1.69)	1.97 (1.84)
St11_2	0.91 (0.79)	3.00 (2.99)	1.86 (1.40)	2.00 (1.41)	1.49 (1.26)	1.44 (1.29)
Mean	2.68 (1.69)	5.55 (4.54)	5.28 (3.45)	5.87 (3.17)	3.98 (3.19)	2.83 (2.51)
Max.	7.29 (4.77)	9.64 (7.13)	9.50 (6.27)	12.36 (5.23)	7.74 (6.14)	5.91 (4.97)
Min.	0.39 (0.39)	3.00 (2.19)	1.43 (1.19)	2.00 (4.41)	1.49 (1.26)	1.10 (0.97)

Regarding biomass and density temporal evolution during the sampled months in 2015, it was observed that the density had significant statistical differences between months ($p = 0.0062$) while the biomass did not ($p = 0.0715$). Using the Tukey test, the density showed significant differences between late May-early June and September, the highest values noted in September.

4.4.1.2 Length–frequency distribution and estimation of growth parameters

In Fig. 4.2 we can observe the length-frequency distribution of *D. trunculus* for the 5 mm mesh size, from St4_1, as an example. The use of Bhattacharya's method allowed the separation of two or three cohorts depending on the month when the sample was collected. Just before the closure, two cohorts were found. The mean length for the first cohort was 11.57 ± 1.78 mm and 20.34 ± 2.37 mm for the second. July and August presented three cohorts. In both months the first cohort presented a mean length slightly lower than before the closure (11.00 ± 1.49 and 10.74 ± 1.82 respectively). Furthermore, we can observe that the mean size of the first cohort was increasing from September. In November the mean size of both cohorts were the highest, with 16.52 ± 2.73 mm in the first and 23.24 ± 0.90 mm in the second. Also, in Fig. 4.2 can be observed one recruitment period during August and September.

In the set of samples, the shell length of individuals in the 5 mm mesh ranged from 5.3 mm to 30.4 mm in bathymetry 1, and between 5.0 and 28.8 mm in bathymetry 2 before the closure. In July the minimum length in both bathymetric ranges were higher than before the closure (6.7 and 7.2 mm respectively), and the maximum values were 29.4 mm in bathymetric range 1 and 29.3 mm in bathymetric range 2. The range in August was between 6.6-31.0 mm (bathymetry 1) and 6.7-30 mm (bathymetry 2). The minimum and maximum lengths in September were slightly lower than in August, with minimum values varying between 5.9 (St11_1) and 9.5 mm (St11_2) and maximum lengths between 25.0 (St4_1) and 30.8 mm (St11_2). In October, the smallest wedge clam was caught at St4_1 and had a length of 4.4 mm and the biggest was 29.3 mm caught at St1_1. In November, a range from 6.0 to 29.0 mm and 6.1 to 28.5 mm was observed in the bathymetric range 1 and 2 respectively. Out of all the samples, the length frequency distribution showed the highest proportion of individuals to be between 14 and 18 mm, the most frequent range of sizes being between 11 - 17 mm in bathymetry 1, and 14 - 18 mm in bathymetry 2.

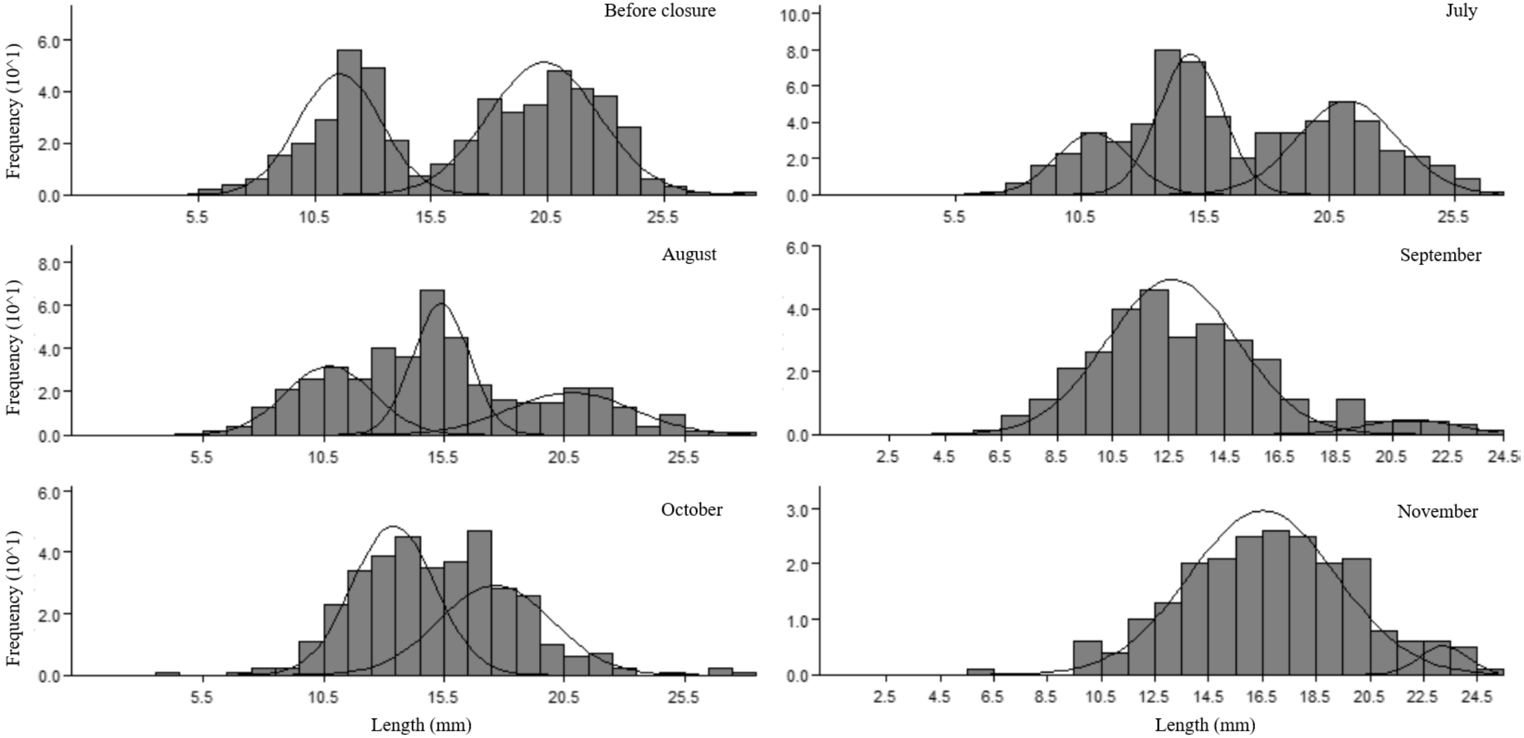

Figure 4.2. Monthly length-frequency distribution of *D. trunculus* for the 5 mm mesh size dredge of the 2015 sampling campaign, sampling site St4_1. Curves represent the cohorts.

The K–S test revealed the existence of significant statistical differences in the length frequency distributions between depths ($p < 0.05$) in virtually 100 % of the sampling stations where the appearance of juvenile individuals was the most frequent in the shallowest areas. The two exceptions were at St9, which showed a trend close to significance ($p = 0.06$), and at St10 which did not display statistically significant differences ($p = 0.36$).

With respect to the growth parameters, the von Bertalanffy growth constant presented a value of 0.44 yr^{-1}, and the L_∞ was 39 mm in 2015. The calculated standard growth index ϕ' was 2.83 and the OGP 4.42. The computed growth curve using these parameters is shown over the length distribution in Figure 4.3.

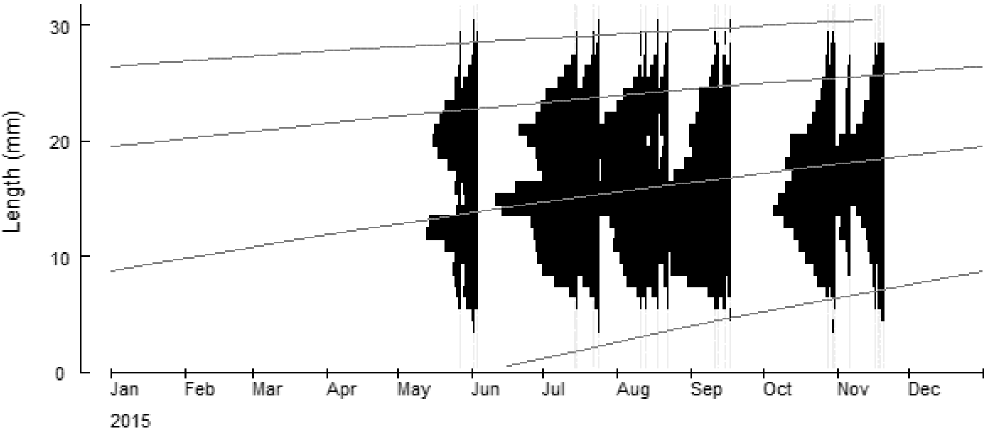

Figure 4.3. Length frequency distribution with growth curves superimposed using ELEFAN I for the sampling campaign 2015.

4.4.1.3 Selectivity

In Fig. 4.4 and 4.5, the size frequency for 5 and 11.7 mm mesh size dredges in the sampling sites St8_1 and St8_2 is illustrated, as an example. Out of all the samples, the size of virtually 100 % of individuals caught in the 11.7 mm mesh size dredge was bigger than the minimum value established by legislation (14 mm). When individuals were noted to be smaller than the minimum size, these organisms did not exceed 1 % of the total sample of *D. trunculus* in bathymetry 1 and 1.3 % in bathymetry 2. In the case of the 11.7 mm mesh size dredge, the individuals which appeared more frequently were those with lengths between 21 and 25 mm. The 5 mm mesh size dredges showed the existence of juveniles in the population from 5 mm in length; however, one case was noted when an individual was smaller (4.4 mm at St4_1 in October). The juveniles (< 14 mm) that were not caught by the 11.7 mm mesh size dredge, represented a percentage between 5 and 70 % of the organisms caught in the 5 mm mesh in bathymetry 1, meanwhile a lower percentage, between 0 and 57 %, was observed in the deeper area. The most frequent size caught by the 5 mm mesh size dredge varied over the sampled months.

The analysis of the fishing gear selectivity for *D. trunculus* was carried out using FISAT II software package, and showed that the 25 % retention length was 24.69 mm, while the L50 % was 25.25 mm and the L75 % showed a value close (25.81 mm).

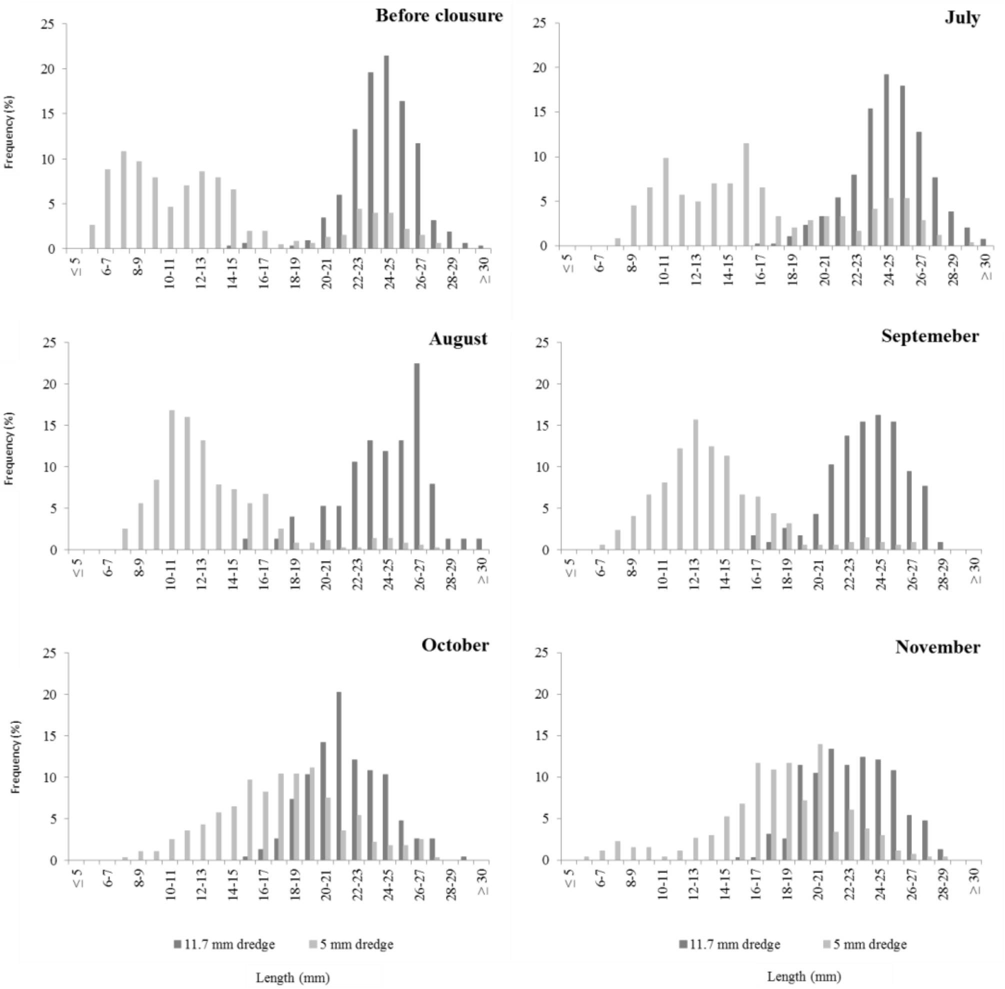

Figure 4.4. Length-frequency distribution of *D. trunculus* population for 5 mm and 11.7 mm mesh size dredges of the 2015 sampling campaign, sampling site St8_1.

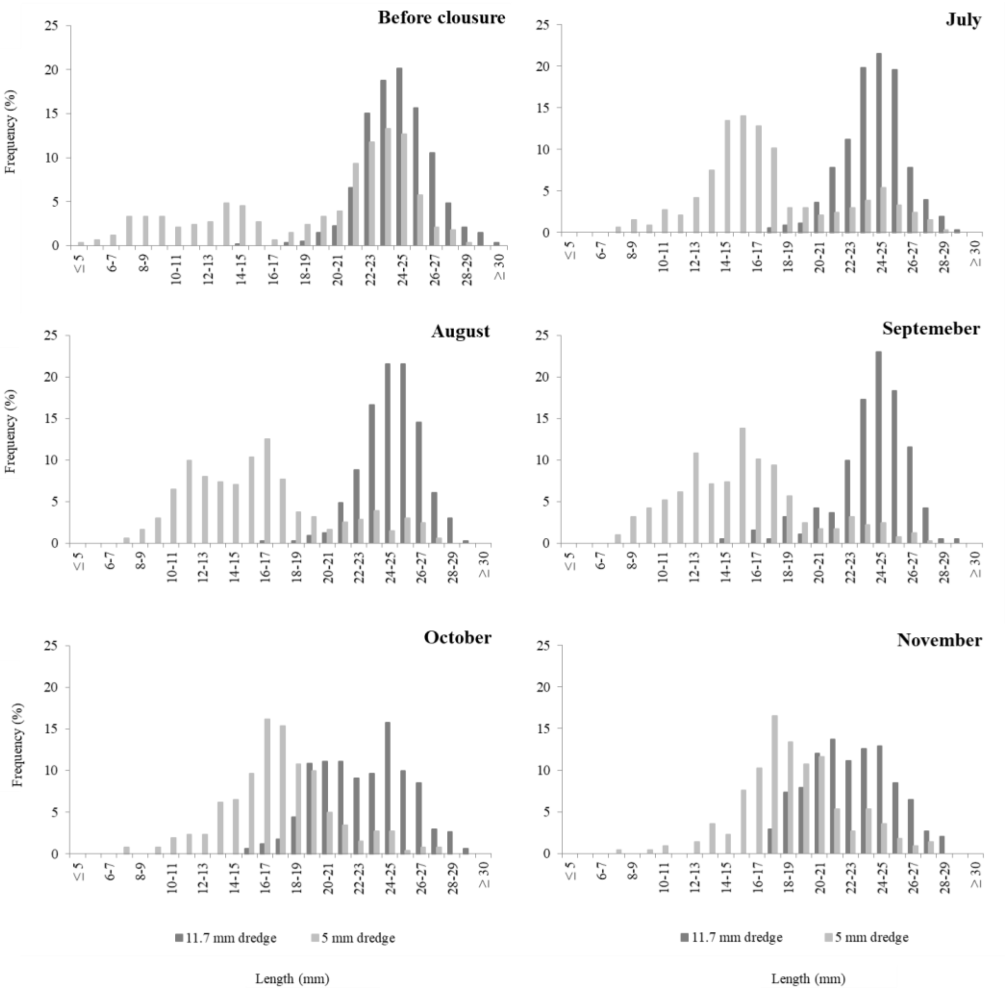

Figure 4.5. Length-frequency distribution of *D. trunculus* population for 5 mm and 11.7 mm mesh size dredges of the 2015 sampling campaign, sampling site St8_2.

4.4.2 Study of populations in 2017

4.4.2.1 Biomass and density evolution

In the 2017 sampling campaign, that included four months (June to September), the mean sea surface temperature was 24.5 ºC in June, 25.5 in July, 26.8 in August and 25.2 in September (NASA Goddard Space Flight Center, Ocean Biology Processing Group, 2018).

In Table 4.3, the biomass and density values for the period June to September 2017 are shown. In general, the maximum biomass was observed in July (between 10-17 kg ha^{-1}), with the exception of St3_1 and St8_1 sampling sites that showed their highest values in June (11 and 13 kg ha^{-1} respectively). A similar pattern was observed for the commercial size biomass with maximum values higher than 10 kg ha^{-1}. Regarding the density, although St3_1 and St8_1 had their maximum densities in September and June respectively, the maximum mean density was observed in July (1.21 ind m^{-2}). St4_1 was generally the station with the highest values for both biomass and density. The commercial size individuals represented the most frequent sizes in the majority of samples with a percentage higher than 50 % of the total density, with the exception of St1_1 in August and September.

Table 4.3. Total biomass (in kg ha^{-1}) and density (ind m^{-2}) of *D. trunculus* in 5 mm mesh size dredge for the period June to September 2017. Also the mean values are presented. The values of *D. trunculus* commercial size (≥ 14 mm) are shown in brackets.

	Sampling site	Jun-17	Jul-17	Aug-17	Sept-17
Biomass	St1_1	7.25 (7.05)	14.26 (13.80)	11.13 (9.07)	1.92 (0.99)
	St3_1	10.66 (10.28)	9.87 (9.72)	6.78 (6.60)	7.43 (6.48)
	St4_1	15.77 (14.91)	17.33 (16.18)	9.80 (9.05)	9.86 (7.74)
	St7_1	9.00 (8.66)	13.13 (13.03)	5.21 (5.16)	2.40 (2.20)
	St8_1	12.85 (12.23)	9.66 (9.58)	3.03 (2.94)	0.20 (0.17)
	Mean	11.11 (10.63)	12.85 (12.46)	7.19 (6.56)	4.36 (3.52)
Density	St1_1	0.64 (0.55)	1.26 (0.93)	1.86 (0.64)	0.58 (0.13)
	St3_1	0.92 (0.75)	0.81 (0.72)	0.64 (0.53)	1.09 (0.60)
	St4_1	1.85 (1.51)	2.20 (1.70)	1.38 (0.96)	2.19 (0.83)
	St7_1	0.93 (0.79)	1.03 (0.99)	0.55 (0.49)	0.29 (0.20)
	St8_1	1.38 (1.10)	0.76 (0.71)	0.33 (0.27)	0.02 (0.01)
	Mean	1.14 (0.94)	1.21 (1.01)	0.95 (0.58)	0.84 (0.35)

Although significant statistical differences were not observed for the density between months ($p = 0.7885$), they were detected for the biomass ($p = 0.0062$). The Tukey test showed that the differences were between the months of June and

September, and July and September, September being the month with the lowest biomass.

4.4.2.2. Length–frequency distribution and estimation of growth parameters

The length frequency distribution of sampling site St1_1 showed the highest frequency for the sizes to be between 9 to 13 mm. At St3_1 the lengths between 18-19 mm and 21-23 mm were the most frequent. St4_1 showed the highest frequency for sizes between 13 and 17 mm. Finally, St7_1 and St8_1 sampling sites had the most frequent lengths, between 15 and 24 mm and between 18-22 mm respectively. Out of all the samples, the length frequency distribution showed the highest proportion of individuals to be between 14 and 17 mm. Moreover, a recruitment period was observed between August and September, mainly during the latter.

The shell length of individuals in the 2017 sampling campaign (Table 4.4) varied between 6.9 and 29.2 mm in June. A wider range was observed in July (5.5-29.3 mm), but in August the range was slightly lower, from 6.2 to 28.5 mm. Finally, the smallest and biggest individuals of the whole sampling campaign were obtained in September, with 5 mm (at St4_1) and 31.1 mm (at St3_1).

Table 4.4. Minimum and maximum length in each month and sampling site in 2017.

Sampling site	Jun-17	Jul-17	Aug-17	Sept-17
St1_1	7.0-29.2	7.0-29.1	6.2-27.2	6.6-25.4
St3_1	11.7-27.1	6.3-25.7	6.5-28.2	7.3-31.1
St4_1	7.0-26.5	5.5-28.6	6.4-27.7	5.0-26.1
St7_1	9.0-29.2	12.9-28.5	6.3-28.5	7.5-25.0
St8_1	6.9-26.3	7.7-29.3	8.1-26.2	10.6-25.8

Regarding growth parameters estimated for 2017, the asymptotic length was 35.9 mm, while *K* presented a value of 0.65 yr^{-1}. From these data, ϕ' (2.92) and OGP (4.48) were obtained.

4.5 DISCUSSION

4.5.1 Ecological aspects of *D. trunculus* populations

From the length frequency distribution of the 2015 sampling campaign (Fig. 4.4 and 4.5), it was observed that smaller organisms were found in the shallower areas (bathymetric range 1). However, the most frequent sizes obtained in 2017 in bathymetry 1 were somewhat bigger than in 2015 and were even similar to the sizes from bathymetric range 2 of 2015. The K-S test confirmed the different distribution between sizes based on depth, with the highest frequency of smaller sizes in shallower areas. The spatial distribution pattern with depth segregation found in virtually 100 % of sampling stations, was also found in several other studies carried out in different zones, on the Catalan coast of Spain (Baeta et al., 2018), Portugal (Gaspar et al., 2002), the French Atlantic coast (Ansell and Lagardere, 1980), Italy (Manca Zeichen et al., 2002) and Turkey (Deval, 2009). These studies detected the prevalence of smaller individuals in shallower waters, while the larger ones were mainly distributed in the deeper bathymetric ranges. However, the contrary was also observed on the coasts of Morocco by Bayed and Guillou (1985). Gaspar et al. (2002) explained the distinction in depth as a survival strategy, in which the juveniles once grown, migrate towards deeper areas giving them more space for the settlement of larvae. Moreover, Scheltema (1971) stated that the establishment of juveniles in shallower areas is due to a higher availability of food and better environmental conditions for their development. Furthermore, the shallower areas can function as a source of *D. trunculus* due to currents and hydrodynamism passively pushing the larvae into this zone favouring colonization (Manca Zeichen et al., 2002).

Through the analysis of the evolution of cohorts, an increase in mean size was observed during 2015 which confirmed that the populations thrived with time and the mean size of *D. trunculus* increased each month. The recruitment period observed during summer (August and September) in 2015 (Fig. 4.2) and 2017, consequently showed juvenile individuals to have increased in biomass and density. A period of summer recruitment was also observed by Ramón (1993) and Ramón et al. (1995) in Cullera (just north of the study zone) between July and September, and another in winter (December to February). On the southwest Spanish coast (Atlantic), Delgado et al. (2017) determined a period of recruitment similar to our study zone during July as well as another a prior period between February and March. Meanwhile on the northwest of Spain (Atlantic) a first recruitment period was observed between February and April and a second in October (Mazé and Laborda, 1988). On the French Atlantic Coast and on the Mediterranean coast of Israel, a summer recruitment was also detected, from July to September (Ansell and Lagardère, 1980; Neuberger-Cywiak et al., 1990). Additionally, differences between other areas of Europe were observed, such as in Italy where recruitment happened in a unimodal way between December and January (Manca Zeichen et al., 2002), or Portugal where recruitment occurred between March and August (Gaspar et al., 1999b). Differences in recruitment periods with respect to other areas may be due to different environmental conditions such as salinity and food availability (Delgado et al., 2017; Defeo and de Alava, 1995).

The growth parameters obtained for *D. trunculus* in the two sampling campaigns showed some differences to other studies from different geographical areas (Table 4.5). In another study carried out in the Gulf of Valencia (Ramón et al., 1995), these parameters, obtained by using the same method (length-frequency distribution method), were generally higher, with a L_∞ of 46 mm and K of 0.58 year^{-1}. This growth constant was higher than that obtained in 2015 but lower than that of 2017 of the present study. Nevertheless in other parts of the western Mediterranean such as Italy and Algeria, the asymptotic lengths showed higher values than those estimated in the current study, though the growth constants were lower (Manca Zeichen et al., 2002; Hafsaoui et al., 2016). In contrast, a lower L_∞ and a higher K were observed in France by Bodoy (1982). In the case of the Sea of Marmara, these values were generally higher than those obtained in this study (Deval, 2009; Çolakoğlu, 2014), with the exception of the growth constant found by Deval (2009) through external growth rings method. Moreover, in the Atlantic, in the northwest and southwest of Spain and Portugal, higher values of the VBGF parameters were obtained, however the growth constants were lower than the 2017 K from this research (Mazé and Laborda, 1988; Gaspar et al., 1999b; Delgado et al., 2017). In another study carried out in the Atlantic, specifically in France, these parameters varied in a wide range (Ansell and Lagardere, 1980). The growth performance indexes obtained in this study were a bit lower compared to the different locations in other studies, an exception was the value calculated for Algeria and Italy (Table 4.5). However, ϕ' values obtained in the current study did not showed significant statistical differences with respect to the other research (p = 0.0582 for 2015; p = 0.7544 for 2017). Vakily (1992) stated that due to the higher temperatures found in lower-latitude regions, bivalves grow faster because of an increase in their metabolic rates. Nevertheless, this was not the case in the data found in different areas (see Table 4.5), likely due to little latitudinal variation of the location analysed. Gaspar et al. (1999b) observed that growth parameters were not affected by latitude either. Therefore, growth parameter variations can be due to different environmental conditions in each area (Vakily, 1992; Gaspar et al., 1999b), such as food availability, population density, type of sediment, temperature, among others (Vakily, 1992). Changes in growth patterns found in different areas of the world suggest a notable growth plasticity of *D. trunculus* under different environmental conditions (Manca Zeichen et al., 2002).

Regarding the overall growth performance index (OGP), the area of study presented slightly lower values than the other studies with the exception of Algeria (Table 4.5). According to Laudien et al. (2003) and Herrmann et al. (2009) the OGP is habitat-specific, categorizing Donacidae in species that could inhabit tropical/subtropical regions, temperate or upwelling. In our case, the population of *D. trunculus* studied would belong, as expected, in the temperate species group, just as Delgado et al. (2017) observed on the southwest Spanish coast.

Table 4.5. Parameters of VBGF of *D. trunculus* obtained by different authors and different methods. L_∞: asymptotic length, K: growth constant, ϕ': growth performance index, OGP: overall growth performance index.

Location	**L_∞ (mm)**	**K (yr^{-1})**	**ϕ'**	***OGP***	**Source**
Atlantic (France)	32.25-43.48 [1]	0.150-0.78	2.21-3.09	3.73-4.68	Ansell and Lagardere (1980)
Mediterranean Sea (France)	35.99 [1]	0.956	3.09	4.65	Bodoy (1982)
Atlantic (Spain)	52.84 [1]	0.55	3.19	4.91	Mazé and Laborda (1988)
Mediterranean Sea (Spain)	46.0 [1]	0.58	3.09	4.71	Ramón et al. (1995)
Mediterranean Sea (Spain)	41.8 [2]	0.71	3.09	4.75	Ramón et al. (1995)
Atlantic (Portugal)	47.3 [2]	0.58	3.11	4.79	Gaspar et al. (1999b)
Mediterranean Sea (Italy)	47.56 [1]	0.30	2.83	4.51	Manca Zeichen et al. (2002)
Marmara Sea (Turkey)	44.15 [3]	0.62	3.08	4.73	Deval (2009)
Marmara Sea (Turkey)	42.44 [4]	0.75	3.13	4.76	Deval (2009)
Marmara Sea (Turkey)	44.10 [1]	0.76	3.17	4.81	Çolakoğlu (2014)
Mediterranean Sea (Sidi Salem, Algeria)	54.70 [1]	0.15	2.65	4.39	Hafsaoui et al. (2016)
Mediterranean Sea (Echatt, Algeria)	55.22 [1]	0.14	2.63	4.37	Hafsaoui et al. (2016)
Atlantic (Spain)	46.0 [1]	0.48	3.01	4.67	Delgado et al. (2017)
Atlantic (Spain)	46.7 [4]	0.47	3.01	4.68	Delgado et al. (2017)
Mediterranean Sea (Spain)	39.0 [1]	0.44	2.83	4.42	Present study, data 2015
Mediterranean Sea (Spain)	35.9 [1]	0.65	2.92	4.48	Present study, data 2017

[1] Length-frequency distribution method
[2] Acetate peel method
[3] External growth rings method
[4] Thin section method

4.5.2 Biomass and density evolution

In the 2015 sampling campaign, which included May to November, a rapid increase in densities and biomass, both for the total and commercial size individuals, was observed in the area of study after the fishery closure (July-August-September, 2015) (Table 4.1 and 4.2). However, only significant statistical differences in density were noted between late May-early June and September. This post-closure increase was likely due not only to the lack of fishing activity but also due to higher temperatures which cause an increase in physiological processes such as feeding (Dionicio and Flores, 2015) and reproduction (Ansell and Lagardère, 1980; Manca Zeichen et al., 2002), which could make individuals stay closer to the sediment surface. Later, when the temperature decreases (October and November), larger organisms tend to bury themselves deeper (McLachlan and Brown, 2006), causing a decrease in density and biomass of both total and commercial sizes. Manca Zeichen et al. (2002), in a study carried out on the South Adriatic coast (Italy), also detected a decline in the densities of *D. trunculus* during autumn-winter (between November and January), in which a decrease in water temperature was a determining factor in the population density variations. A decline of densities was also reported by Neuberger-Cywiack et al. (1990) in January (winter) when minimum temperatures reached 16 ºC, as well as a decrease in salinity.

Regarding the samples taken in 2017, from June to September, a slightly different trend was observed for the biomass and densities. In both 2015 and 2017, the maximum values were found in July, but August and September in 2017 showed a decrease compared to those obtained in June. However, significant statistical differences were only observed for biomass between June and September and between July and September. The fact that June 2017 showed higher values than August and September 2017, the contrary to what happened in 2015, could be because there was still fishing activity of *D trunculus* in the area in 2015. A general and considerable decline for biomass and density in 2017 could be seen compared to the 2015 values.

4.5.3 Fishing aspects of *D. trunculus*

As previously mentioned in "Selectivity" section, the individuals smaller than 5 mm were not caught by the 5 mm mesh size dredge, just as expected (with the exception of one case). In some cases, the larger organisms that were caught by the 11.7 mm mesh size dredges were not caught by the 5 mm mesh size dredge (Fig. 4.4 and 4.5). This could be due to the 5 mm mesh size dredge not being able to penetrate well because of the sediment clogging in the mesh (Ramón, 1993). As a consequence, smaller individuals would be caught because larger sizes are able to bury themselves deeper which avoids intraspecific competition (McLachlan and Brown, 2006). A factor that can be ruled out is that some organisms can actively escape, because of their slow mobility (Ramón, 1993).

With respect to the selectivity parameters obtained using FISAT II, the L25, L50 and L75 values (24.69, 25.25 and 25.81 mm respectively) were higher than the minimum size landing (14 mm) established by legislation and higher than the first maturation size of 12.7 mm (Ramón, 1993), therefore the fishing gear could be considered to have an appropriate selectivity. In a previous selectivity study carried out in the Gulf of Valencia, lower 25, 50 and 75 % retention lengths (19.7, 22.1 and 24.5 mm) for *D. trunculus* were noted. From then on, it was agreed that no other modification of mesh size was necessary (Ramón, 1993). Nevertheless, although the fishing gear analysed showed an adequate selectivity, it would be advisable to use a larger commercial size of 25 mm, to benefit the sustainability of exploited beds. In another study carried out in Portugal, the fishing gear selectivity was also analysed, based on the mesh size (Lart et al., 2003). In that case, the mesh size for *D. trunculus* fishing (25 mm), was considerably bigger than the one used in our zone, and the 25, 50 and 75 % retention lengths were 19.41 mm, 23.59 mm, and 27.77 mm respectively, generally lower values than our own. The fact that the retention lengths were lower in spite of having a mesh size higher could be due to the use of dredges with net bag. When a dredge with a net bag is being towed the holes of the mesh stretches and organisms can't pass through the mesh, unlike what happens with the metallic grid.

Before the fishery closure, the biomass of commercial size individuals ranged broadly from <0.05 to 108 kg ha^{-1} (Table 4.1). Only in 45 % of the cases, the minimum profitable level of 20 kg ha^{-1} (E. Ferrer, personal communication, Gandia Fishermen's Guild, 2015) was reached o exceeded, thus the fishing had a low profitability. From this 45 % of cases, 70 % was found in the bathymetric range 1. In the months subsequent to the closure in 2015, the percentage of cases exceeding 20 kg ha^{-1} for commercial size individuals increased ranging from 56 % to 86 % (in September and July respectively), showing the highest values in July, August and October. However, two years after the closure in 2017, this improvement in the biomass of commercial size individuals could no longer be seen. In this year, the biomass of the sampling sites analysed never reached the minimum profitable level of 20 kg ha^{-1} (Table 4.3), therefore it was not deemed profitable to reopen the fishery.

According to the biomass data obtained in 2015, stations St3, St4 and St7 were considered the most profitable to be fished. In 2017, although the minimum profitable level was not reached in any of the cases, St4 also showed the highest value of biomass.

Due to the high reproduction potential of this species, it was expected to have had a fast population recovery. However, two years after the fishery closure, this recovery was not observed. Other bivalve fisheries which were closed in order to encourage population recovery have been studied in the Mediterranean and Atlantic. Baeta et al. (2018) carried out an analysis of the bivalve fishery of *Callista chione* Linnaeus, 1758 in Catalonia (Spain, NW Mediterrannean) where the fishing activity was suspended for a year in order to recover the population. At the end of the suspension, a recovery was not observed therefore many fishermen decided to abandon the fishery of this clam, leaving residual activity until the permanent closure 6 years later. In contrast, in Uruguay (Atlantic), the fishery of *Mesodesma mactroides* Reeve, 1854 showed an immediate recovery after a 1 year suspension (Defeo, 1996; Brazeiro and Defeo, 1999; Ortega et al., 2012). Nevertheless, when the suspension lasted more than 3 years, the increase of population was accompanied by a notable decrease in recruitment, thus such a long period of closure may cause negative consequences (Defeo, 1996; Brazeiro and Defeo, 1999). Therefore, it is worth noting that fishery closures are not always effective.

In our current study, a lack of recovery of the population of the wedge clam *D. trunculus* two years after the closure, means that there are other factors affecting the density and the survival of this specie in the Gulf of Valencia. Previously the influence of temperature on the species abundance as a determining factor was discussed. However, other studies related to the survival or abundance of the species determined that sediment grain size, parasitism and predation can also be factors to be taken into account. Sediment granulometry is a variable which has been studied many times to determine the influence it has on populations of *D. trunculus* in the Mediterranean Sea and in the Atlantic coast of Spain (SW). The distribution and abundance of these organisms depend on sediment grain size and are susceptible to changes in the sediment both produced by humans (nourishment, coastal modifications…) or naturally (Mazé and Laborda, 1998; Guillou and Bayed, 1991; Lart et al., 2003; La Valle et al., 2011). Moreover, this variable can have a great influence on the ecological efficiency of individuals such as a decrease in their burrowing rate, growth and metabolism (De la Huz et al., 2002).

In addition, authors such as Ramadan and Ahmad (2010) in the Mediterranean coast of Egypt, de Montaudouin et al. (2014) in the Atlantic coast of Morocco and France, and Delgado and Silva (2018) in the Atlantic coast of Spain, determined that *Bacciger bacciger* Rudolphi, 1819 parasitation could cause a severe castration of *D. trunculus*, generating survival problems of population. This trematode was also found in *D. trunculus* in the Gulf of Valencia with a low prevalence, which decreased reproduction capacity of this bivalve although without significantly affecting the population (Ramón et al., 1999).

Another aspect to take into account is predation both in benthic and planktonic phases. The Donax clams are subject to predation by gastropods such as *Neverita josephinia* Risso, 1826, crabs, birds and fishes, among others (Schneider, 1982; Neuberger-Cywiak et al., 1990; McLachlan and Brown, 2006). Salas et al. (2001), in a study developed in the South of Spain (Malaga), determined that the foot nipping on Donacidae produced principally by crabs can be sub-lethal, because the individuals affected were more susceptible to predation. Besides this type of sublethal foot-predation and lethal predation, predation of larvae in their planktonic phase needs to be investigated in greater depth, due to the lack of studies focused to it.

4.6 CONCLUSIONS

The growth of *Donax trunculus* in this study showed differences compared to other locations, highlighting that the values obtained for L_∞ and K in 2015 were lower than those obtained 25 years ago in the same region. However, the growth performance indexes confirmed that they were within the values expected, with possible variations due to environmental factors. Furthermore, the summer recruitment and juvenile and adult size differential distribution between depths coincided with the literature.

The analysis of selectivity of fishing gear showed adequate values based on the minimum landing size and the first maturation size. Nevertheless, it would be advisable to increase the minimum catch size, and therefore to adjust the mesh size of dredges to the new size.

The variability in density and biomass showed a seasonal pattern with increases in summer mainly due to the variation of temperature and associated biological processes. The values of density and biomass in 2017, not only showed that there had not been a population recovery, but also that the situation had worsened. Therefore, the closure of the fishery had not had the desired effect in the short term. This poses the need for further studies regarding factors or variables which could be affecting the abundance or survival of the population of *D. trunculus* in the study area.

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Capítulo 5

Driving forces that structure macrobenthic communities of microtidal intermediate sandy beaches

5. DRIVING FORCES THAT STRUCTURE MACROBENTHIC COMMUNITIES OF MICROTIDAL INTERMEDIATE SANDY BEACHES

5.1 ABSTRACT

Sandy beaches are very dynamic and changeable environments that present a wide variety of life forms, and, in some areas, high productivity. In the Gulf of Valencia (Western Mediterranean Sea, Spain) these areas have endured an economically important shellfishery of clams (*Donax trunculus* and *Chamelea gallina).* In the Mediterranean, a sea with a notable oligotrophic character, rivers are particularly important because of the contributions of fresh water that carry nutrients and organic matter to coastal ecosystems. Consequently, in the vicinity of the mouths of the rivers, it is possible to investigate the functions of physical, chemical and biological gradients on the macrofauna structuring of sandy beach communities. The objective of this work was to determine the response of benthic communities to environmental variables in microtidal intermediate sandy beaches of the south of the Gulf of Valencia. Samples of benthic macrofauna, water and sediment were collected at 5 stations located at different distances to freshwater sources, each one sampled at five depths, and in three sampling campaigns. In general, the physicochemical and nutritional variables (phytoplanktonic groups) of the water showed a spatial variation alongshore, and only a few variables (dissolved inorganic nitrogen and salinity) also had an across-shore variation. This variability was due to the different contributions of fresh water. The sediment variables (mean grain size and organic matter) presented clear patterns related to depth. With respect to the macrofauna, positive relationships with depth were observed for the total density, density of bivalves, polychaetes, crustaceans and *C. gallina*, while *D. trunculus* showed the opposite pattern. The proximity to sources of fresh water, favoured a greater abundance of polychaetes in the deepest zone only. Regarding the temporal variation, two very different phytoplanktonic compositions between winter and summer could be distinguished, while, for the macrofauna, differences in abundance were observed with maximum values in summer. By means of Canonical correspondence analysis, it was found that the communities of benthic macrofauna were strongly influenced by physical factors such as mean grain size, as well as the nutritional variables (organic matter in sediment and microalgae in the water column) and the distance to the freshwater discharges. The identification of these variables is crucial in order to develop an adequate coastal management, paying special attention to those anthropogenic activities that may be modifying the environment and that therefore will affect the biocoenoses.

Keywords: sandy-beach invertebrates; environmental factors; macrofaunal assemblages; microtidal sandy beaches

5.2 INTRODUCTION

Exposed sandy beaches are very dynamic systems in space and time, inhabited by biological communities that are fundamentally structured by physical drivers (Defeo and McLachlan, 2005). In most ecological systems, including those in the oceans, community structure arises from a complex interaction between biotic interactions and abiotic environmental factors (Cury et al., 2008; Blanchette et al., 2009). However, the ecological theory of sandy beaches establishes that physical characteristics control macrobenthic community structure on all beaches, except on the most dissipative ones (Ortega Cisneros et al., 2011).

Sandy beaches, despite their dynamism, harbour a wide variety of life forms, becoming very productive in some areas (McLachlan, 1983; Defeo and McLachlan, 2005; McLachlan and Defeo, 2018). Due to instability and complex environmental changes that sandy beaches undergo, fauna that inhabit them develop adaptations related to locomotion, sensory responses, nutrition, respiration, reproduction, among others (McLachlan and Defeo, 2018). In general, in these ecosystems, molluscs, crustaceans and polychaetes are found as dominant groups of macrofauna, often constituting more than 90 % of the species and biomass (Gray and Elliott, 2009; McLachlan and Defeo, 2018). At the meso-scale, variations of macrofauna communities can be observed both alongshore and across-shore

(Defeo and McLachlan, 2005; McLachlan and Defeo, 2018). Distribution, abundance and richness of macrobenthic communities are conditioned by physical-chemical variables such as temperature, pH, salinity, dissolved oxygen, nutrients and those related to beach morphodynamics (slope, sediment particle size, hydrodynamism ...), as well as nutritional and biological factors (life cycles, mortality, depredation and competitiveness by resources, among others) (Hewitt et al., 1997; Brazeiro, 2001; Defeo and McLachlan, 2005; Lastra et al., 2006; Rodil et al., 2007; Ortega Cisneros et al., 2011). Authors such as Ortega Cisneros et al. (2011) highlighted the importance of combining physical, chemical and nutritional variables for studies on community structure. In most of the investigations carried out to explain patterns of community structure, only abiotic characteristics tend to be taken into account, but it would be more appropriate to also consider the potential biotic interactions (Defeo et al., 1997; Hewitt et al., 1997).

Beach morphodynamic state can be defined by physical factors such as wave energy, tidal range and sand particle size (McLachlan, 2001), while the physical factors that influence macrofauna community structure would primarily be sediment grain size, as well as coast slope and length, among other factors (McLachlan, 1996; Rodil and Lastra, 2004; McLachlan and Dorvlo, 2005; Lastra et al., 2006; Lercari and Defeo, 2006). There have also been several authors who have observed an influence of salinity on community abundance and richness, both decreasing in the areas closest to the freshwater contributions and during periods of greater precipitation (Lercari et al., 2002; Lercari and Defeo, 2006; Lozoya et al., 2010; Ortega Cisneros et al., 2011). In addition, other investigations highlighted the importance of the feeding of populations -organic matter in sediment or availability of food in water column (Lastra et al., 2006; Rodil et al., 2007; McLachlan and Defeo, 2018).

River mouths are key locations to investigate the influence of physical, chemical and nutritional variables on benthic fauna. These environments are particularly important in oligotrophic seas such as the Mediterranean, which depends strongly on the contributions of fresh water that carry nutrients, necessary for primary production, and organic matter which are sources of food for many organisms (Ludwig et al., 2009; Schlacher and Connolly, 2009; Levinton, 2011). In the Spanish Mediterranean coast, Pinedo et al. (1997) analysed the effect of precipitation, hydrographic factors, concentration of chlorophyll *a*, granulometry and organic matter in sediment, on the trophic groups in macrobenthic community in the Bay of Blanes (Northwestern Mediterranean Sea). In this study, the spatial distribution of the benthic macrofauna was explained by the depth, the granulometry and the organic matter, while the temporal distribution by the rainfall, the chlorophyll *a* and the organic matter in sediment, without any significant effect from temperature or salinity. In the Gulf of Valencia (Western Mediterranean Sea, Spain) few studies have been carried out in relation to determining the influence of environmental variables on benthic community structure, hence it is still a subject of investigation. Martí et al. (2007) analysed the temporal changes of bivalves and polychaetes from Cullera Bay (Western Mediterranean Sea), observing the influence of tourist pressure and water contributions in the shallowest zone, while, in deeper areas, granulometry and climatic and meteorological conditions were more relevant. In these studies, it is important to take into account the repercussions that changes in the ecosystems may have on benthic communities, where species of socio-economic relevance can be found. In the case of the Gulf of Valencia, two species of bivalves, *Donax trunculus* and *Chamelea gallina*, have a high fishing interest but they have suffered notable declines in their populations in the last decade, leading to the closure of the fishery in June 2015 (Valencian Community Resolution of June 3, 2015).

To determine whether the macrobenthic community responds to different physical, chemical and nutritional factors, three sampling campaigns were carried out at five sampling along-shore stations, each one sampled at five depths, and located at different distances to freshwater sources in Gandia (Western Mediterranean Sea, Spain). To address this aim, the objectives were: (1) to characterize the temporal and spatial dynamics of the macrobenthic community structure; (2) to determine the temporal and spatial patterns for physical, chemical and nutritional variables;

and (3) to determine which physical, chemical and nutritional variables were most influential in describing changes in the macrobenthic community structure.

5.3 MATERIAL AND METHODS

5.3.1 Study area

The study area is located on the microtidal sandy beaches of the Gulf of Valencia (Western Mediterranean Sea, Spain) (Fig. 5.1) close to: i) the Serpis River mouth, which is the main river that provides fresh water to this area and it is artificially regulated by a complex system of weirs and irrigation channels that provide fresh water to irrigated crops of the Safor County; ii) the Gandia Port mouth, that receives waters from the San Nicolas ravine, a 14 km long watercourse of which the flow relies strongly on torrential rainfalls and only in its last 1.5 km the flow is continuous, due to the input of fresh water draining the Safor Wetland (Sebastiá et al., 2012) and; iii) the submarine outfall of the wastewater treatment plant (WWTP) of Gandia that discharges wastewater at 17 m depth and at 1900 m from the coast, in front of the harbour. Climatically, this area is characterized by its water scarcity in the dry season (summer) and abundant and brief rainfall in the wet season (autumn) (Garófano-Gómez et al., 2009). The annual precipitation of the zone shows values over 800 mm (Miró et al., 2018). Morphodynamically, the beaches of the zone were classified as intermediate exposed sandy beaches by Escrivá (2013).

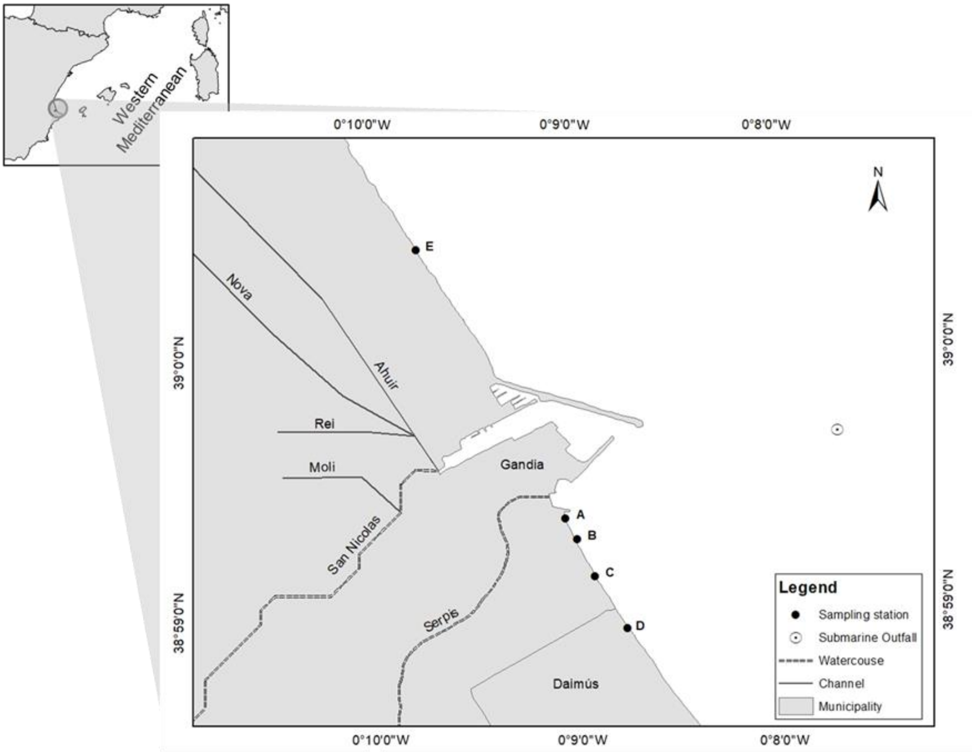

Figure 5.1. Study area in the Gulf of Valencia (Western Mediterranean Sea, Spain) indicating the position of the five sampling stations (from A to E).

5.3.2 Sampling design

Three sampling campaigns were carried out: the first one (July 2013) from 30th July to 5th August 2013; the second (February 2014) between 3rd and 14th February 2014 and the third (December 2014) between 12th and 17th December 2014.

In each campaign, samples of water, sediment and benthic macrofauna were obtained at five sampling stations: four of them located to the south of the mouth of the Serpis River at 50 (A), 200 (B), 500 (C) and 1000 (D) m away - presumably influenced by freshwater contributions from both the river and the Port of Gandia, since the dominant currents move from north to south (CEDEX, 1997; Millot, 1999) - and a fifth sampling station at 1000 (E) m distance north of the mouth, away from the influence of fluvial inputs (Fig. 5.1).

At each station, transects were established perpendicular to the coastline, taking samples in the sublittoral zone at five different depths (0.5, 1, 2, 3 and 4 m): at 0.5 m in the zone between the low tide line and the area where the waves break, at the break point (1 m), and at 2, 3 and 4 m in the transition and outer turbulent zone.

5.3.3 Environmental and nutritional variables analysis

5.3.3.1 River flow and precipitation

The Serpis flow and rainfall in the area of influence were obtained from the Automatic Hydrological Information System (SAIH) of the Júcar Hydrographic Confederation (CHJ). The values of the flow and precipitation were gathered from the Assut d'En Carròs meteorological and gauging station, located about 10 km from the estuary, as it is the closest station to the mouth of the river where the information is available.

5.3.3.2 Water variables

In order to analyse the physicochemical characteristics of the water, samples were taken at each sampling station and depth. To do this, a Niskin oceanographic bottle was used to collect water from 50 cm above the seabed.

In situ, temperature and salinity were measured using the multiparametric probe Multi 340i WTW. In the laboratory, the following variables were analysed: photosynthetic pigments, suspended solids (SS), total phosphorus and dissolved nutrients (ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), phosphate (PO_4^{3-}) and silicate ($Si(OH)_4$). The dissolved nutrients were analysed using the methods proposed by Aminot and Chaussepied (1983) and adapted by Baumgarten et al. (2010). Dissolved inorganic nitrogen (DIN) was calculated as the sum of: ammonium, nitrite and nitrate. To determine total phosphorus (TP), digestions of unfiltered samples were made according to Valderrama (1981) before its determination using the same method for PO_4^{3-}. The photosynthetic pigments (alloxanthin, chlorophyll *a* (Chl_*a*), chlorophyll *b* (Chl_*b*), fucoxanthin, lutein, neoxanthin, peridinin, prasinoxanthin, violoxanthin, zeaxanthin, 19'butanoyloxyfucoxanthin, 19'hexanoyloxyfucoxanthin) were analysed using the high-performance liquid chromatography (HPLC) technique. The HPLC method employed was that proposed by Wright et al. (1991), recommended by the Scientific Committee on Oceanic Research (SCOR), and slightly modified as per Hooker et al. (2000). Suspended solids were analysed according to APHA (2012).

5.3.3.3 Sediment variables

At each sampling station, sediment samples were taken at each of the depths. At each point a sample was collected using a Ponar dredge. A granulometric analysis (Shepard, 1954) was conducted on each sample in triplicate and the mean grain size was calculated (Friedman and Sanders, 1978). Furthermore, the concentration of organic matter (OM) was analysed in triplicate employing the method proposed by Pusceddu et al. (2004).

5.3.4 Macrobenthic community analysis.

At each point, sediment samples were taken in triplicate, to reduce heterogeneity in the distribution of macrofauna in the sediments, using a Ponar dredge (with an area of 0.059 m^2). The samples were sieved through a 500 μm mesh to remove excess sand and retain the benthic macrofauna as per the methodology proposed by Castelli et al. (2004). Subsequently, the organisms were submerged in a 7 % magnesium chloride solution as an anaesthetic and preserved in 10 % buffered formaldehyde. In the laboratory, the remaining sediment was removed manually, and the specimens were identified to the lowest taxonomic level possible and counted. In addition, the organism density was calculated as the number of individuals per square meter (ind m^{-2}).

5.3.5 Data analyses

Data analyses were conducted through a range of multivariate analysis. To determine the spatial influence of the different variables, ANOVA multifactorial tests were done and a Tukey post hoc test done when a significant result was found. These analyses were carried out for each variable in each sampling campaign, using as factors “Depth” and “Station”. Those variables that did not follow a normal distribution were transformed by their natural logarithm. Moreover, to identify significant differences in variables between sampling campaigns non-parametric Kruskal-Wallis tests and multiple comparisons of groups with Bonferroni adjustment of p-values were carried out. In addition, a cluster analysis was done to investigate similarity in macrobenthic composition of the different points and sampling campaigns, using the Ward method and Manhattan distance. These tests were carried out using the software package Statgraphics Centurion XVII. Finally, a canonical correspondence analysis (CCA) was conducted in order to determine the most influential physical, chemical and nutritional variables in the macrobenthic community, taking into account those species that were found in at least 20 % of the sampling points (Rodil et al., 2006). Furthermore, a second CCA was carried out on the species richness, using all the species at each sampling point. For these analyses, the CANOCO 5 software was used. The statistical significance of the relationships was evaluated using Monte Carlo permutation tests with a manual forward selection procedure.

5.4 RESULTS

5.4.1 River flow and precipitation

The annual precipitation obtained from the nearest meteorological station to the study area (Assut d'En Carròs) was 471 and 258 $l\ m^{-2}$ for the years 2013 and 2014 respectively. During the 30 days prior to the beginning of the first sampling campaign (July 2013), there was no discharge of water from the river, and the cumulative monthly rainfall was low (2 mm). In contrast, in February 2014 and December 2014, the river had a mean flow of 1 $m^3\ s^{-1}$ and 0.8 $m^3\ s^{-1}$ during each prior month respectively, and the cumulative monthly precipitation was 12.4 and 96.2 mm.

5.4.2 Environmental and nutritional variables

Table 5.1 shows the minimum and maximum values of the main environmental variables, for each station and sampling campaign, as well as the depths at which they were obtained. In July 2013, high temperatures were detected in the water, varying between 24 and 27 ºC. Regarding salinity, significant statistical differences ($p < 0.001$) were observed between stations, showing lower values at the station located 1000 m north of the mouth (E) and 200 m south (B) (Table 5.S1). With respect to nutrients, DIN ($p = 0.005$) and $Si(OH)_4$ ($p = 0.002$) and the variable TP ($p = 0.013$) also showed significant effects, with higher concentrations at station B which present an inverse pattern to salinity. The photosynthetic pigments Chl_*a* ($p = 0.023$), Chl_*b* ($p = 0.004$), fucoxanthin ($p = 0.015$) and peridinin ($p < 0.001$) followed the same pattern as the nutrients, with higher values at station B. Finally, 19'hexanoyloxyfucoxanthin ($p < 0.001$) showed the highest values at the two stations furthest away from freshwater sources (E and D). Meanwhile, the variables that presented significant statistical effects with respect to the depth were: in water, only DIN ($p = 0.005$) with higher values in the shallowest areas (0.5 and 1 m) and, in sediments, mean grain size ($p = 0.005$) and OM ($p = 0.033$) with clear patterns of variation with depth, decreasing mean grain size and increasing OM when depth increases.

In February 2014, temperature and salinity decreased to values of 11-14 ºC and 36-37 respectively (Table 5.1). In the case of SS, the highest concentrations were observed ($p = 0.001$) at the station located immediately south of the freshwater sources (A) (Table 5.S2). In this campaign, the concentrations of DIN again showed a significant effect between stations ($p = 0.020$), with higher values at B and C. With regard to the photosynthetic pigments, those that had significant effects were alloxanthin ($p < 0.001$), Chl_*a* ($p = 0.034$), 19'hexanoyloxyfucoxanthin ($p = 0.003$) with higher values at station D and Chl_*b* ($p = 0.006$) that presented higher concentrations at the station located to the north of the contributions (E). Furthermore, in sediments, the mean grain size showed a pattern of descent with the increase of depth ($p = 0.001$).

Finally, in December 2014, temperatures increased compared to February 2014, varying between 14 and 16 ºC (Table 5.1). Those variables that had significant effects between stations in this sampling campaign were SS, TP and alloxanthin ($p = 0.029$, $p = 0.003$ and $p = 0.026$ respectively) (Table 5.S3). The highest values of SS were found at station E, for TP at the two stations furthest from the freshwater contributions (D and E), while the highest alloxanthin concentrations were found at A. With respect to differences between depths, significant effects were observed for salinity, OM and mean grain size ($p = 0.003$, $p = 0.039$ and $p < 0.001$ respectively). Salinity and OM increased with depth, while mean grain size showed the opposite pattern.

Table 5.1. Minimum and maximum values of the environmental variables at each station and sampling campaign, the depths at which they were found are shown in brackets. Temp.: temperature; Allox.: alloxanthin; Perid.: peridinin; 19’hexan.: 19’hexanoyloxyfucoxanthin; MGS: mean grain size.

		July 2013					February 2014					December 2014				
		A	B	C	D	E	A	B	C	D	E	A	B	C	D	E
		Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.	Min.-Max.
Temp.	ºC	26.1-26.8	25.6-26.3	24.6-25.6	24.1-25.1	25.7-26.7	11.6-12.6	12.5-13.5	12.4-12.7	12.9-13.7	11.2-12.1	14.1-14.9	14.2-15.1	15.2-16.3	15.5-16.3	13.7-16.3
Salinity		37.6-37.8	37.3-37.5	37.6-37.8	37.3-37.6	37.1-37.5	36.7-37.1	36.8-37.0	36.8-37.1	37.0-37.2	36.7-37.0	37.5-37.6	37.3-37.6	37.1-37.7	37.5-37.6	37.2-37.8
Depth	m	(3-2;4)	(2;3-0.5;1)	(0.5-1)	(1-2;3;4)	(0.5;1-4)	(2-4)	(4-2)	(0.5;4- 1;2;3)	(0.5;2-1;4)	(0.5-2)	(0.5;2;4-1;3)	(0.5-3;4)	(0.5-3)	(0.5;1-2;3;4)	(0.5-4)
SS	mg L^{-1}	9.0-15.8	10.5-18.3	7.9-15.8	8.0-16.5	8.2-13.8	8.4-13.8	6.6-10.0	6.3-9.4	7.0-9.8	9.0-14.8	12.7-19.5	10.5-20.7	8.8-16.0	10.2-13.8	11.2-53.5
Depth	m	(2-1)	(3-2)	(0.5-2)	(3-2)	(2-1)	(2-1)	(1;3-4)	(4-0.5)	(2;4-1)	(3-0.5)	(0.5-2)	(3-0.5)	(3-0.5)	(1-0.5)	(2-0.5)
DIN	µM	0.1-3.0	3.5-6.6	0.3-5.8	0.3-6.4	0.1-3.3	3.2-4.6	3.2-5.5	3.7-4.7	1.8-4.1	3.0-4.2	3.7-6.6	4.6-7.7	2.8-10.0	3.7-6.8	2.0-4.9
Depth	m	(4-3)	(4-1)	(4-1)	(3-0.5)	(4-2)	(1-4)	(4-2)	(4-0.5)	(4-2)	(4-3)	(1-3)	(4-1)	(3-0.5)	(2-1)	(3;4-0.5)
Si(OH)$_4$	µM	3.3-3.9	5.1-6.8	2.3-4.0	2.3-6.2	2.9-5.0	0.9-3.6	0.9-6.8	1.7-5.2	1.2-2.4	0.8-1.3	4.5-5.5	4.1-8.2	3.5-10.6	4.0-5.7	3.0-4.34
Depth	m	(2-1)	(3-2)	(4-0.5)	(3-0.5)	(4-1)	(1-2)	(4-0.5)	(1-4)	(1-2)	(1-2)	(2-0.5)	(4-0.5)	(4-0.5)	(3-1)	(0.5-2)
TP	µM	0.18-0.26	0.23-0.38	0.20-0.29	0.11-0.23	0.14-0.23	0.03-0.26	0.12-0.14	0.03-0.18	0.07-0.13	0.12-0.19	0.12-0.24	0.09-0.19	0.16-0.26	0.22-0.36	0.17-0.61
Depth	m	(4-2)	(1-4)	(3;4-0.5;1)	(4-0.5)	(0.5-2)	(0.5-3)	(0.5;3-4)	(0.5-1)	(4-0.5)	(2-1)	(1-3)	(2-0.5)	(2;3-0.5;1)	(0.5-4)	(4-1)
Chl_*a*	µg L^{-1}	0.123-0.245	0.186-0.232	0.062-0.146	0.114-0.219	0.130-0.370	0.170-0.295	0.217-0.369	0.263-0.367	0.331-0.530	0.281-0.550	0.151-0.353	0.003-0.229	0.108-0.237	0.003-0.174	0.107-0.141
Depth	m	(3-2)	(1-2)	(4-1)	(3-1)	(0.5-4)	(4-1;3)	(0.5-3)	(2;4-1)	(0.5-1)	(2-4)	(2-0.5)	(0.5-3)	(2-0.5)	(0.5-1)	(1-2)
Allox.	µg L^{-1}	0.027-0.063	0.068-0.102	0.015-0.091	0.036-0.131	0.045-0.113	0.059-0.113	0.059-0.161	0.137-0.311	0.212-0.508	0.105-0.192	0.010-0.029	<0.003-0.016	0.007-0.021	<0.003-0.017	0.003-0.009
Depth	m	(3-2)	(2-0.5)	(4-1)	(3;1)	(0.5-4)	(4-3)	(2-0.5)	(4-0.5)	(3-1)	(1-3)	(2-0.5)	(0.5;4-2)	(2-0.5)	(0.5;2-1)	(1-2;4)
Perid.	µg L^{-1}	<0.03-0.16	0.11-0.13	<0.03-0.05	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03
Depth	m	(0.5;1-2)	(3-0.5)	(1-0.5)												
19’hexan.	µg L^{-1}	<0.01-0.06	0.03-0.14	<0.01-0.14	0.12-0.28	0.15-0.24	<0.01-0.07	<0.01	<0.01	0.03-0.30	<0.01-0.29	<0.01-0.01	<0.01	<0.01	<0.01	<0.01
Depth	m	(2;3;4-1)	(1-0.5)	(4-1)	(3-1)	(3-1)	(1;2;3;4-0.5)			(2-3)	(0.5;3-2)	(1;2;3;4-0.5)				
MGS	mm	0.14-0.24	0.15-0.25	0.16-0.26	0.17-0.23	0.16-0.20	0.14-0.25	0.18-0.23	0.16-0.32	0.18-0.28	0.18-0.23	0.13-0.27	0.15-0.28	0.15-0.24	0.17-0.27	0.17-0.23
Depth	m	(4-0.5)	(4-0.5)	(4-2)	(4-2)	(2;4-0.5)	(4-1)	(3;4-0.5)	(4-0.5)	(3;4-2)	(2;4-0.5)	(4-0.5;1)	(4-0.5)	(3-1)	(4-0.5)	(4-1)
OM	%	0.31-1.34	0.75-1.09	0.50-0.86	0.49-0.95	0.36-0.61	0.71-1.27	0.53-0.97	0.40-1.28	0.26-1.06	0.49-0.84	0.06-0.75	0.01-1.05	0.38-1.25	0.15-0.88	0.37-0.77
Depth	m	(1-4)	(1-3)	(2-4)	(2-3)	(1-0.5)	(0.5-4)	(1-2)	(0.5-3)	(2-3)	(4-1)	(1-0.5)	(1-4)	(0.5-4)	(0.5-3)	(0.5-3)

Regarding temporal variation of the environmental variables (Table 5.S4), the highest temperatures were observed in July 2013, coinciding with the summer period, and the lowest in February 2014, showing significant statistical differences between the three campaigns ($p < 0.001$). In the case of salinity, the differences were found between February 2014, with lower values, and the other two sampling campaigns ($p < 0.001$). The nutrients that showed significant differences between sampling campaigns were DIN and $Si(OH)_4$ ($p = 0.001$ and $p < 0.001$ respectively). The highest DIN values were observed in December 2014. However, $Si(OH)_4$ had higher concentrations in December 2014 and July 2013 and lower in February 2014, showing the same pattern as the variables TP ($p < 0.001$) and SS ($p < 0.001$). Finally, almost all the photosynthetic pigments displayed significant differences between sampling campaigns, with the exception of lutein in which the Bonferroni procedure indicated that there were no significant statistical differences. The pigments that showed significant differences between February 2014 and the remaining sampling campaigns were Chl_*a* ($p < 0.001$), Chl_*b* ($p < 0.001$), fucoxanthin ($p < 0.001$), prasinoxanthin ($p < 0.001$) and 19'butanoyloxyfucoxanthin ($p < 0.001$) presenting in all cases the highest values in February 2014. In the case of alloxanthin ($p < 0.001$) and violaxanthin ($p < 0.001$), there were differences between the three sampling campaigns, also with higher values in February 2014. Peridinin ($p < 0.001$) and 19'hexanoyloxyfucoxanthin ($p < 0.001$) showed lower concentrations in February 2014 and December 2014 than in July 2013 and neoxanthin ($p < 0.001$) higher values in February 2014 than December 2014. Finally, zeaxanthin ($p = 0.001$) had significant differences between July 2013 and December 2014, with higher concentrations in July 2013. In sediments, neither the mean grain size nor OM showed significant differences between the three sampling campaigns.

5.4.3 Macrobenthic community

In Fig. 5.2 the abundance of bivalve, crustacean and polychaete can be observed, for each station, depth and sampling campaign. In July 2013 (Table 5.S5), only the abundance of polychaete showed a significant statistical effect between sampling stations ($p = 0.004$), with higher values at those stations closer to the freshwater inputs (A, B and C). However, significant differences with respect to depth were observed for the total density ($p < 0.001$) and the density of crustaceans ($p = 0.001$), polychaetes ($p < 0.001$), *D. trunculus* ($p < 0.001$) and *C. gallina* ($p = 0.001$) showing a pattern which increases with depth, with the exception of *D. trunculus* which was the opposite. In the sampling campaign of February 2014 (Table 5.S6), the total density ($p = 0.014$) and the abundance of crustaceans ($p = 0.023$) showed a significant effect between stations, with higher values at B. As regards differences between depths, all the variables of fauna presented changes with depth. The total density ($p < 0.001$), and the density of bivalves ($p < 0.001$), crustaceans ($p < 0.001$), polychaetes ($p < 0.001$) and *C. gallina* ($p = 0.009$) showed increases with depth, while *D. trunculus* ($p < 0.001$) had lower values at the deepest points. Regarding the difference between stations in December 2014 (Table 5.S7), only the total density showed a significant effect ($p = 0.011$) with higher values at A. On the other hand, the total density ($p < 0.001$), and the abundance of bivalves ($p < 0.001$), crustaceans ($p < 0.001$), polychaetes ($p = 0.001$) and *D. trunculus* ($p < 0.001$) showed significant statistical differences with increases in depth, with the same exception for *D. trunculus*.

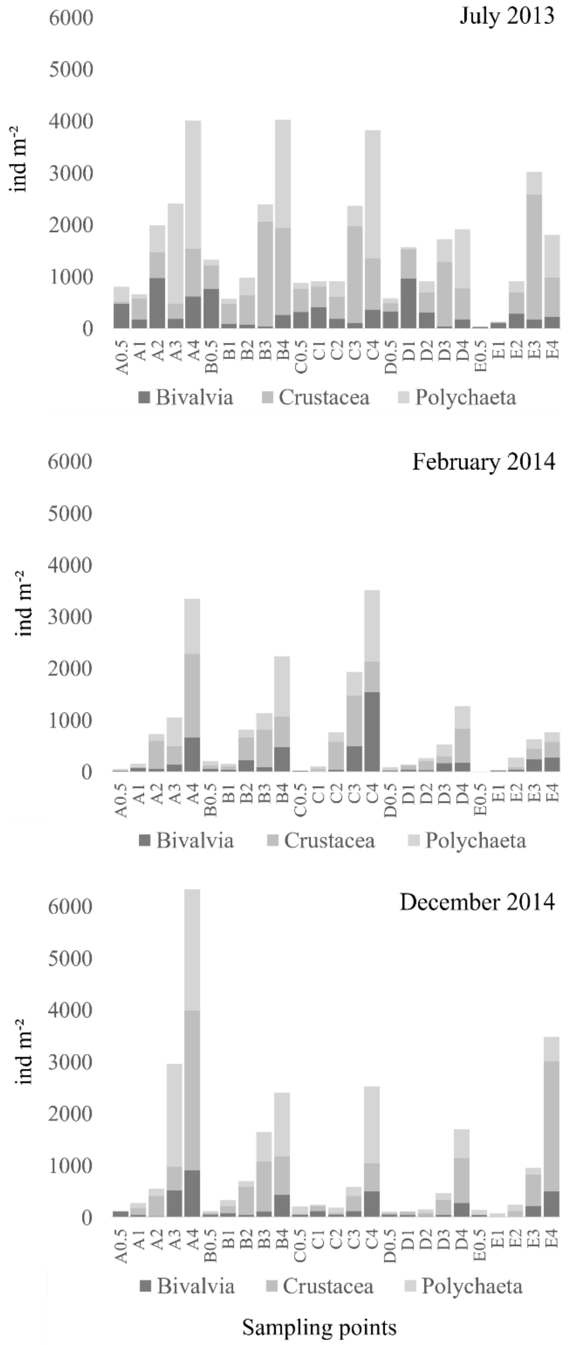

Figure 5.2. Abundance (ind m²) of main classes of macrofauna (bivalvia, crustacea and polychaeta) for each station and depth in each sampling campaign.

Regarding temporal variation of fauna (Table 5.S8), both the total density (*p* = 0.005) and the density of bivalves (*p* = 0.008) presented significant differences between July 2013 and the other two sampling campaigns, with higher values in July 2013. The density of crustaceans also showed significant differences (*p* = 0.028) but only between July 2013 and February 2014, with higher values again in July 2013. In the case of the two species of commercial interest for the study area, *D. trunculus* and *C. gallina*, their densities did not show significant differences between the three sampling campaigns, however, for both species, the densities were also higher in the July 2013 sampling campaign.

In general, the most abundant species of Bivalvia in the shallowest sampled areas was *D. trunculus.* At higher depths, 2-4 m, *D. trunculus* was replaced by *Donax semistriatus*, *Mactra stultorum* and *Loripes orbiculatus.* Regarding polychaetes, *Scolelepis squamata* predominated in the shallower area, while at 2 m depth, *Magelona johnstoni*, *S. squamata*, and *Glycera sp* were the most abundant. At deeper points (3 and 4 m) *Magelona mirabilis*, *M. johnstoni, Paradoneis armata* and *Sigalion mathildae* were the most abundant. In the case of crustaceans, *Pontocrates arenarius* was the predominant at 0.5-1 m along with *Urothoe grimaldii. Siphonoecetes sabatieri* and *Bathyporeia elegans* became the most abundant crustaceans from 2 m depth, and in some cases *Apseudopsis bacescui.*

The dendrogram obtained from the cluster analysis showed a separation of the sampling points in three clusters (Fig. 5.3). The first two were grouped according to depth, regardless of the sampling campaign. The first one comprised the points from 0.5 and 1 m depth while the second those from the intermediate and deepest zone (from 2 to 4 m). Finally, a third cluster was obtained where only the points of 4 m from the stations closest to the freshwater contributions (A, B and C) were grouped (including all three sampling campaigns). The different clusters highlighted the influence of bathymetry on the macrobenthic community structure, although the third one also incorporated the proximity to the freshwater contributions.

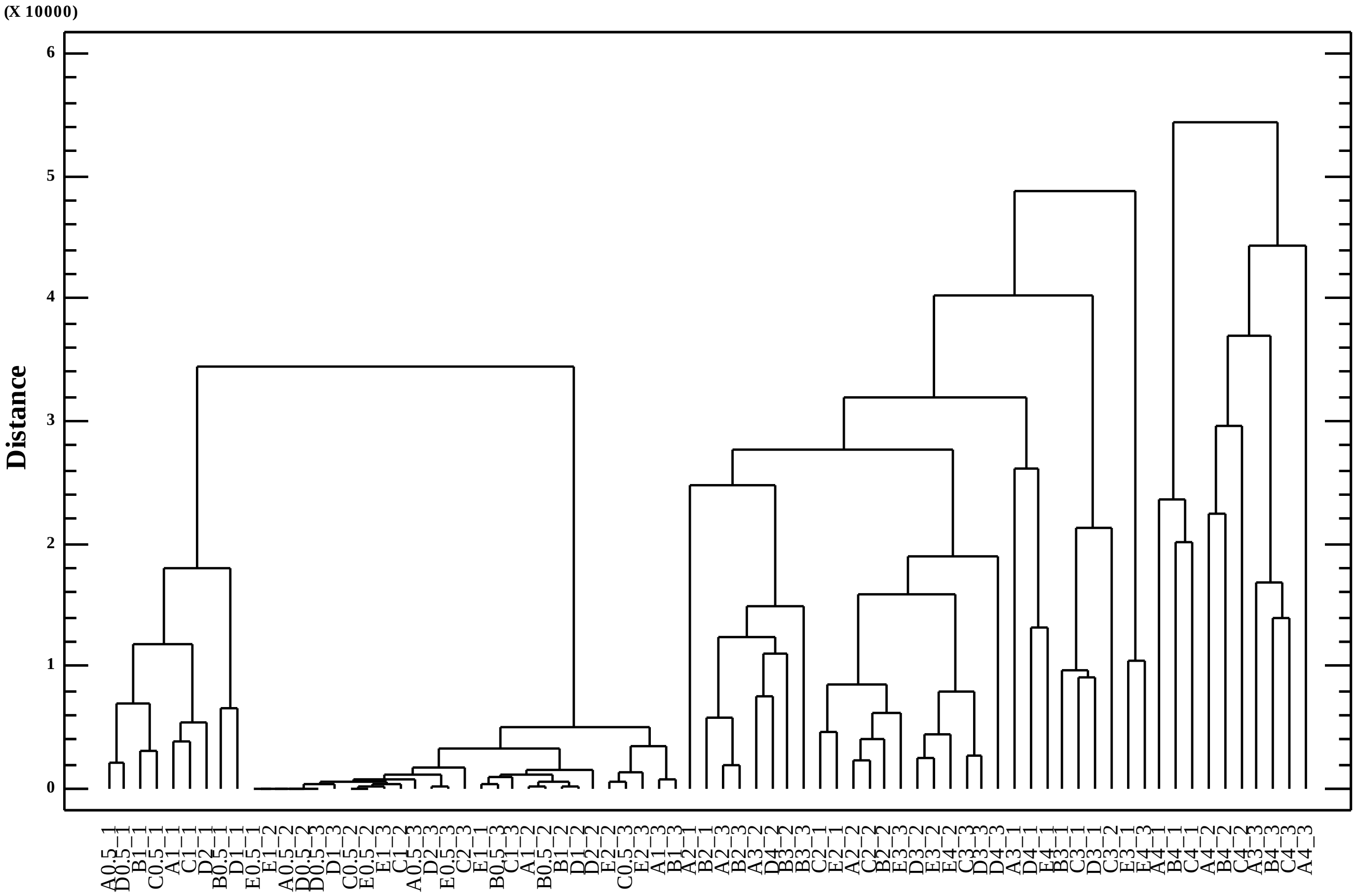

Figure 5.3. Dendrogram resulting from the cluster analysis of the macrobenthic fauna density. The labels show the sampling station (letter), depth (first number) and sampling campaign (second number; 1: July 2013; 2: February 2014; 3: December 2014).

5.4.4 Correlations between community structure and environmental variables

In order to analyse the relationship between the macrobenthic fauna species with the environmental variables, a CCA analysis was carried out (Fig. 5.4). Due to the fact that both depth (as seen in the cluster analysis, Fig. 5.3) and temperature (linked to species life cycle) are the most relevant factors which could mask the influence of any other variables, both were discarded in order to conduct the CCA. The ordination biplot for the major species of macrofauna obtained using CCA showed the species distribution, whose proximity indicates species often occurring together, along with environmental variables. The two axes explained 27.12 % of the variability. Species such as *U. grimaldii*, *S. squamata*, *Gastrosaccus sp.*, *D. trunculus*, *Schistomysis assimilis* and *P. arenarius* showed a high positive correlation with the variable mean grain size. These species are also positively influenced by the pigments 19'hexanoyloxyfucoxanthin and, to a lesser extent, by alloxanthin as well as DIN. Furthermore, the polychaetes species, specifically *M. mirabilis*, *Prionospio pygmaea*, *P. armata* and *Nephtys hombergii,* showed a positive correlation with OM, as well as peridinin and DIN, and negative when the distance to freshwater contributions increases. The numbers of *Scoloplos sp.*, *D. semistriatus* and *Glycera sp.* were the least explained by the environmental variables since their positions in the CCA model were close to the origin.

In addition, a CCA ordination diagram between the species richness and the environmental variables can be observed in Figure 5.5. In general, those points that presented lower richness were located at lower depths, relating to the mean grain size and the pigment 19'hexanoyloxyfucoxanthin. However, those points that showed higher values of richness were located at greater depths, mainly at 3 and 4 m, generally with positive correlations with OM and 19'butanoyloxyfucoxanthin, and negative correlation when the distance to the freshwater sources increases.

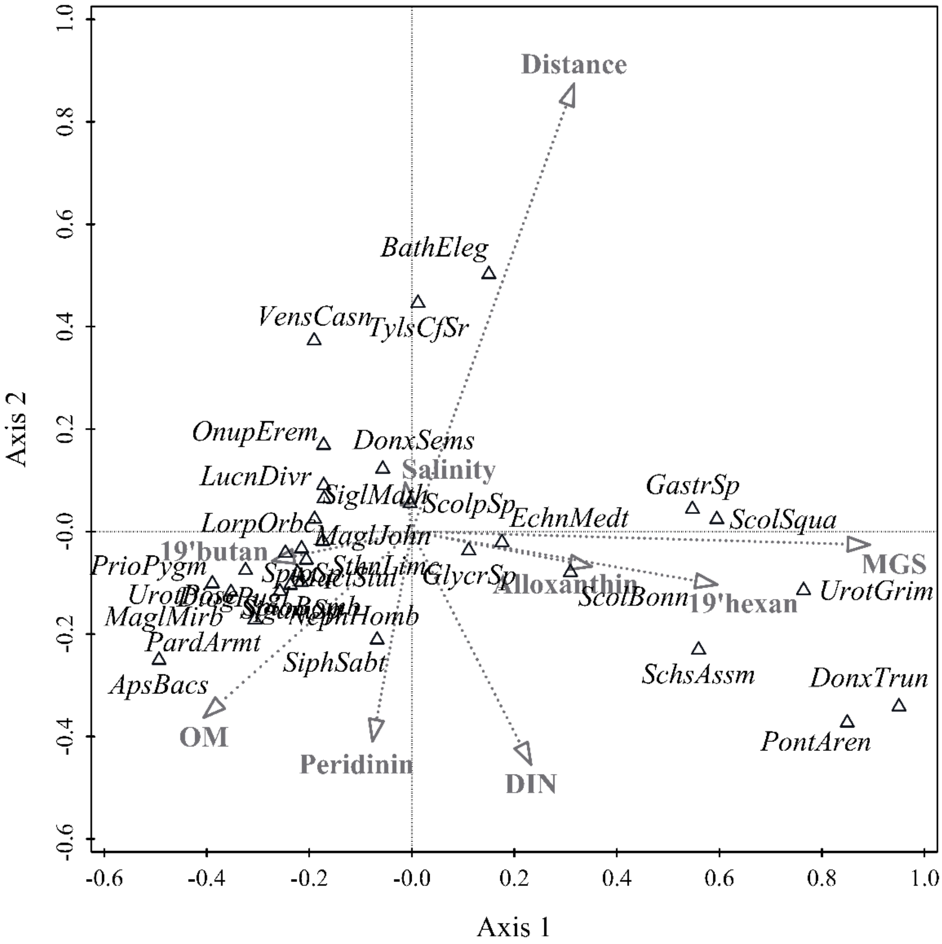

Figure 5.4. CCA ordination diagram between the densities of species (triangles) and environmental variables (arrows). The length of the arrows indicates the strength of the variable in that dimensionality of the solution, and they point in the direction of the gradient. DIN: dissolved inorganic nitrogen; MGS: mean grain size; OM: organic matter; 19'butan: 19'butanoyloxyfucoxanthin; 19'hexan: 19'hexanoyloxyfucoxanthin. *ApsBacs: Apseudopsis bacescui; BathEleg: Bathyporeia elegans; DiogPugl: Diogenes pugilator; DonxSems: Donax semistriatus; DonxTrun: Donax trunculus; EchnMedt: Echinocardium mediterraneum; GastrSp: Gastrosaccus sp.; GlycrSp: Glycera sp.; LorpOrbc: Loripes orbiculatus; LucnDivr: Lucinella divaricata; MactStul: Mactra stultorum; MaglJohn: Magelona johnstoni; MaglMirb: Magelona mirabilis; NephHomb: Nephtys hombergii; OnupErem: Onuphis eremita; PardArmt: Paradoneis armata; PontAren: Pontocrates arenarius; PrioPygm: Prionospio pygmaea; SchsAssm: Schistomysis assimilis; ScolBonn: Scolelepis bonnieri; ScolpSp: Scoloplos sp.; ScolSqua: Scolelepis squamata; SigamSp: Sigambra sp.; SiglMath: Sigalion mathildae; SiphSabt: Siphonoecetes sabatieri; SpioBomb: Spiophanes bombyx; SpioSp: Spio sp.; SthnLimc: Sthenelais limícola; TylsCfSr: Tylos cf. sardous; UrotGrim: Urothoe grimaldii; UrotPose: Urothoe poseidonis; VensCasn: Venus casina.*

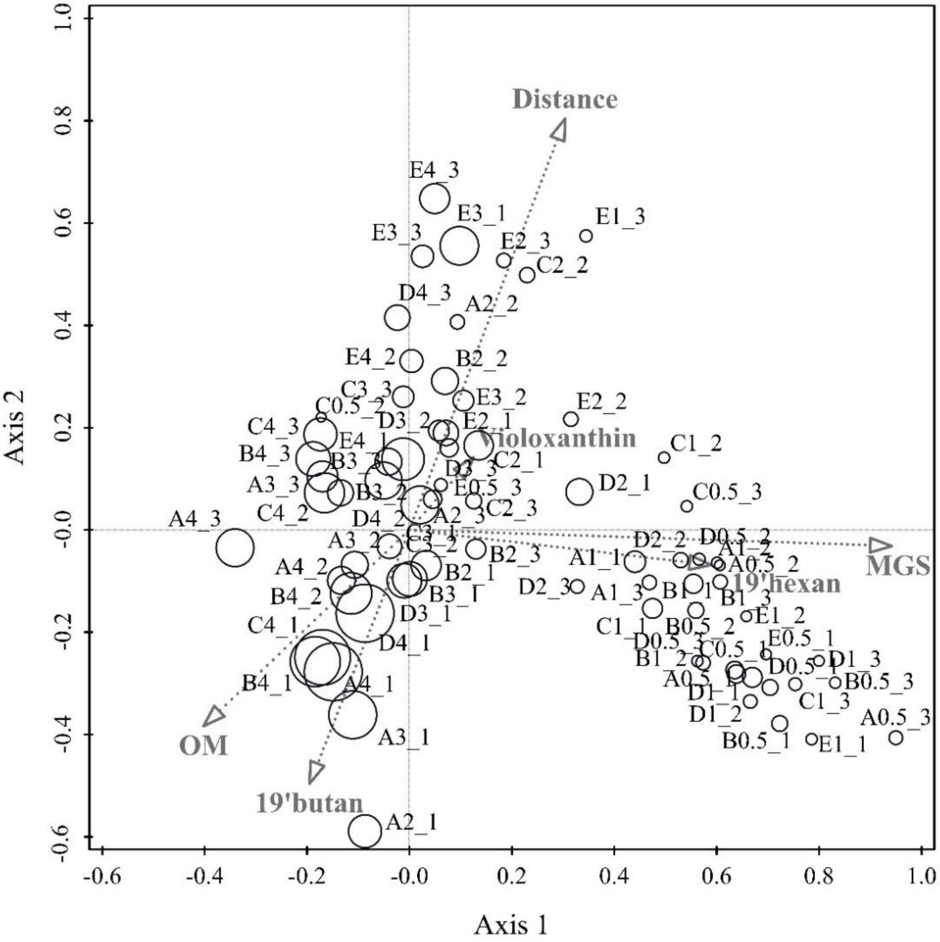

Figure 5.5. CCA ordination diagram between species richness (circles) and environmental variables (arrows). Circle size varies to reflect the value of richness: the greater the size, the larger the richness. The labels show the sampling station (letter), depth (first number) and sampling campaign (second number; 1: July 2013; 2: February 2014; 3: December 2014). MGS: mean grain size; 19'butan: 19'butanoyloxyfucoxanthin; 19'hexan: 19'hexanoyloxyfucoxanthin.

5.5 DISCUSSION

McLachlan (2001) proposes physical control (wave energy, tidal range and sand particle size) as one of the paradigms that explain macro-scale variations of macrobenthic communities in the intertidal zone of sandy beaches. However, biological factors (including food availability, competition and predation) are considered to play a role in the structuring of macrofaunal communities at the meso-scale only in physically benign habitats, such as undisturbed dissipative beaches. In extremely microtidal environments (< 30 cm), the intertidal zone is so narrow that the shallower sublittoral zone is that which harbours a community, subject to instability and continuous changes due to the action of wave breaks. It is important to determine whether the sublittoral macroinfaunal community that inhabits dynamic beaches (intermediate morphodynamic type), such as those of the study area, responds significantly to the physical environment at the meso-scale, as predicted by the theory.

Regarding the spatial distribution of the fauna variables, in general, the densities of the different groups (bivalves, polychaetes and crustaceans) along with the total density and richness were mainly influenced by depth. By means of the cluster analysis, a first grouping by similarity was observed, including the fauna from the shallowest points, 0.5 and 1 m. This area presents more dynamic and turbulent conditions, with less diversity and abundance of species (Janssen and Mulder, 2005; McLachlan and Defeo, 2018). While in deeper areas, sediments present a greater stability allowing the development of richer and denser macrofauna, showing a greater similarity between the points from 2 to 4 m (second group). In the physicochemical variables of sediment, a clear influence of depth could also be observed, in both the mean grain size and the OM. The smallest sand particle size, observed at the deepest points of the study zone, is related to higher densities of macrofauna and richness (McLachlan and Dorvlo, 2005; Rodil et al., 2006; McLachlan and Defeo, 2018). The greater concentration of OM in deeper zones is a consequence of sedimentation and accumulation on the seabed of both finer geological materials and the majority of the particulate organic matter in this area.

This pattern in the physicochemical variables of sediment was also observed by Ramón (1993) in an area located just north of our study zone. In addition, the deepest points of the stations closest to the contributions of fresh water were grouped in a third cluster, indicating an influence of freshwater sources on the fauna. The samples taken at 4 m depth from the stations A, B and C showed, in all the sampling campaigns, a higher abundance of polychaetes than the 4 m samples from stations D and E (Figure 5.2). When comparing the samples obtained at 4 m depth, the stations closest to the contributions were those that had higher levels of OM and lower mean grain size. The deepest points from A, B and C were affected by the proximity to sources of OM and sediments such as those from the Serpis River. Due to this as well as the lack of turbulence in this area, sedimentation of particulate material occurred. The habitual conditions in this zone were adequate for the development of polychaetes, mainly surface deposit feeders and some carnivores which feed on small invertebrates (Fauchald and Jumars, 1979; Jumars et al., 2015). Among these polychaetes, *M. mirabilis*, *M. johnstoni*, *Cirratulus cirratus*, *S. mathildae*, *P. pygmaea*, *Notomastus sp.*, *P. armata* and *Sigambra sp* were the most abundant. Hydrodynamism has a great influence on abundance and biomass of deposit feeders and suspension feeders in macrobenthic communities, due to their food supply depending largely on water velocity (Zhang et al., 2015). The food in the turbulent zone is in suspended form, therefore, suspension feeders dominate this area, excluding the establishment of deposit feeders (Wildish, 1977). In the deepest zone, where the sedimentation of particulate material takes place, a gradient of deposit feeders from the closest areas to the freshwater sources to the furthest ones, could be observed.

The most abundant macrofauna classes in the study area were molluscs, crustaceans and polychaetes, coinciding with what happens in a generalized way on sandy beaches (Martí et al., 2007; McLachlan and Defeo, 2018). In the Mediterranean, the bivalve *D. trunculus* is characterized by a higher density of individuals at depths between 0.5 and 2 m (Ramón et al., 1995; Gaspar et al., 2002), a pattern also observed in the present study. As depth increases, another species of the same genus, *D. semistriatus,* becomes more abundant (Ansell and Lagardère, 1980; Martí et al., 2007). *C. gallina*, although not one of the most abundant species, was found in higher densities at the deepest points of the study area coinciding with the bathymetric range observed by Moschino and Marin (2006). The abundance of *S. squamata* in the shallower areas may have related to the level of beach exposure (Borzone et al., 1996), and also to the high sediment permeability that is necessary for its respiratory requirements (Dauer, 1983). *S. squamata* lives in vertical tubes that form with sand and mucus that are incompatible in fine-grained sediments. Similar granulometric requirements are described by Jones (1948) for *P. arenarius*, which excavates in sands and was described by Rodil et al. (2006) as typical of this bathymetric range. Likewise, *U. grimaldii* is considered by Carvalho et al. (2001) and Carvalho et al. (2012) as an exclusive species of the shallow sublittoral areas of exposed beaches. The abundance of *S. sabatieri* at greater depth can be linked, on the one hand, to the interaction with other species (Bigot et al., 2006), but also to the availability of sedimentary detritus that serve as food (Navarro-Barranco et al., 2013) coinciding with what happens with the increase in depth. Both the polychaetes *M. mirabilis*, *M. johnstoni*, *P. armata* and *S. mathildae*, and the crustacean *A. bacescui* were found at greater depths, where the higher content of organic matter is found, most of them being surface deposit feeders (Fauchald and Jumars, 1979; Jumars et al., 2015). Deposit feeders dominate more mature ecosystems, with low hydrodynamism, where the content of organic matter and clay is high (Aneiros et al., 2018).

In the water in the study zone, the spatial variation of the different variables was linked to the sources of fresh water. It should be noted that the annual rainfall in 2013 and 2014 showed values clearly lower than what is usual in the area (Miró et al., 2018). In the July 2013 sampling, the lowest salinities were observed at stations B and E, located 200 m south and 1000 m north of the mouth of the Serpis River respectively, and the highest concentrations of nutrients at B. Although this campaign corresponded to a dry period with low rainfall and zero flow of the river, the contributions, from the drainage channels from the Safor Wetland that end up in the Gandia Port, as well as from the overflow channel of the Gandia WWTP that discharges near the Serpis mouth, had a clear influence at station B. In summer, when the population of Gandia triples due to tourist activity, the maximum capability of absorbing wastewater by the submarine outfall is exceeded, therefore some of this wastewater is discharged into the final section of the Serpis River (Sebastiá et al., 2013). In addition, in the February 2014 and December 2014 samplings, when there was a greater flow in the Serpis River, in spite of not observing significant differences in the salinity between stations, transect B also showed lower levels. At south of the mouth of the Serpis River, there is a breakwater that acts as a barrier shielding the nearest area, consequently station B was more greatly influenced by the contributions of fresh water than the nearest one (A). In addition, station B also presented the highest nutrient values. Furthermore, it should be noted that diffuse groundwater discharges from the wetland (Sebastiá et al., 2012) could occur in the area. The lower salinity at station E in July 2013, could be due to the diffuse contributions of groundwater. Regarding the spatial variation with respect to depth and consequently to the distance from the coast, salinity was higher in the deepest zone in the December 2014 sampling, while DIN showed higher concentrations in the shallowest area in July 2013. A similar pattern for DIN was also observed by Sospedra (2014) in the study area. Nitrogen fertilizers used in citrus agriculture in the area (Ramos et al., 2002), could contaminate groundwater that ultimately discharges into the coastal zone. It is also important to mention that there was no clear pattern between nutrient concentrations and photosynthetic pigments. In July 2013, the transects that showed higher concentrations of nutrients also had the highest concentration of photosynthetic pigments. However, in February 2014, DIN presented its lowest values at station D, where the highest concentrations of pigments were generally obtained. Due to the fact that primary producers need different proportions of nitrogen and phosphorus along with only some taxonomic phytoplanktonic groups that also require a nutrient such as silica, makes it possible to observe both direct and inverse patterns between nutrient concentrations and photosynthetic pigments.

The analysis of the temporal variations indicated that temperature had clear differences between the different campaigns with maximum values in July 2013 (between 24 and 27 ºC) and minimums in February 2014 (11-14 ºC). The lowest values of salinity were observed in February 2014, due to the greater flow from the Serpis River in this sampling campaign. In December 2014, with a slightly lower flow, and July 2013, without any flow, the salinity of the waters was higher and similar between them. However, the highest values of nutrients such as DIN and $Si(OH)_4$ were presented in December 2014, when the highest rainfall occurred. Higher concentrations of DIN and $Si(OH)_4$ in rainy periods were also observed by Sebastiá et al. (2013) and Gadea et al. (2013) in the study area, as a result of agricultural runoff and chemical weathering of land.

Regarding the photosynthetic pigments, in February 2014, when salinity and temperature were lower, the highest concentrations of Chl_*a*, Chl_*b*, fucoxanthin, neoxanthin, prasinoxanthin, 19'butanoyloxyfucoxanthin, alloxanthin and violaxanthin were found. Blooms of phytoplankton in late winter and early spring, when light availability increases, have also been observed in the study area by other authors (Gadea et al., 2013; Sospedra, 2014). Some of these pigments are associated with certain divisions of algae, so they reveal, at least in part, the composition of the phytoplankton population. In this way, it was possible to distinguish two very different phytoplanktonic compositions between winter (February 2014) and summer (July 2013). In February 2014, the high levels of Chl_*b*, a compound present in all green algae (chlorophyceae, prasinophyceae and euglenophyta) and cyanophytas, as well as fucoxanthin, a characteristic of diatoms, and alloxanthin, which indicates cryptophytes, would reveal a greater presence of these groups (Jeffrey and Vesk, 1997; Vidussi et al., 2001; Gallardo, 2004; Brewin et al., 2010). Also prasinoxanthin, an exclusive pigment to prasinophyceae (Jeffrey and Vesk, 1997), and 19'butanoyloxyfucoxanthin, present in both prymnesophyceae and chrysophyceae (Jeffrey and Vesk, 1997), were higher in the late winter sampling campaign. However, in summer (July 2013), the highest concentrations of 19'hexanoyloxyfucoxanthin and peridinin indicated the presence of dinoflagellates (Vidussi et al., 2001) while zeaxanthin indicated the presence of cyanophytas and prochlorophytes (Vidussi et al., 2001). In the study area, higher dinoflagellate and cyanobacteria were observed during the dry season and showed a direct correlation with temperature (Sebastiá et al., 2013). In temperate zones, dinoflagellates and cyanobacteria show an increase in summer-autumn (Levinton, 2011), while diatoms dominate during spring (Garmendia et al., 2011; Levinton, 2011; Sebastiá and Rodilla, 2013).

It is also worth analysing two variables such as SS and TP which are related, to a certain extent, to plankton. The SS includes both organic particles (organisms and non-living matter) and inorganic particles, so their concentration depends on factors such as primary production, sea state (currents, waves, ...), although anthropogenic perturbations (Bilotta and Brazier, 2008) can also affect them. The TP comprises dissolved and particulate forms, the latter including phosphorus present in organisms, dead organic matter and adsorbed on particles. These two variables showed higher levels in the December 2014 and July 2013 campaigns, unlike most of the photosynthetic pigments that had the maximum in February 2013. This would indicate that in December 2014 and July 2013 the detritus and inorganic particles were the dominant components of these variables instead of the phytoplanktonic organisms.

As regards the temporal analysis of fauna, the highest abundance of the different groups of macrofauna (bivalves and crustaceans) along with the total density, were obtained in the July 2013 sampling campaign which presented the highest temperature. This type of seasonal variation was described by McLachlan and Defeo (2018) as a habitual pattern due to the displacement of organisms that occurs in winter when the water temperature falls or the deeper burial during stormy weather. Although, in the study area, polychaetes, *D. trunculus* and *C. gallina* did not show significant differences between campaigns, the highest densities were also obtained in July 2013. The seasonality of *D. trunculus* was also associated with increases in temperature by authors such as Manca Zeichen et al. (2002) and Neuberger-Cywiak et al. (1990). The greater biomass of diatoms in late winter - early spring (Garmendia et al., 2011; Levinton, 2011; Sebastiá and Rodilla, 2013) triggers two processes: an increase in the abundance of secondary producers that couple their life cycle to a greater food availability (Zhang et al., 2015) and, furthermore, a higher level of detritus due to death and sedimentation of both primary and secondary producers. In July 2013, many species increased their abundance due to the incorporation of new recruits and, also, because of the increase of detritus that, along with the dinoflagellates, became the main sources of food in this period.

The CCA showed an influence of the sediment variables (mean grain size and OM), microalgae in the water column, and the distance to the freshwater sources on the macrobenthic community. Most species were negatively influenced by the increase in mean grain size. This influence of granulometry on organisms was also observed by different authors (McLachlan, 1996; Rodil and Lastra, 2004; McLachlan and Dorvlo, 2005; Lastra et al., 2006; Lercari and Defeo, 2006). The importance of granulometry on the benthic macrofauna is a key point to consider in areas where beach nourishment occurs, as is the case of the beaches near our study area. It is important to select an adequate sediment to preserve species that can be susceptible to habitat modifications such is the case of *D. trunculus* (De la Huz et al., 2002; La Valle et al., 2011). In addition, the OM in sediment was another important factor in the CCA. Increases of organic matter in sediments facilitate the richness and abundance of many species (Lercari and Defeo, 2006). In the water column, microalgae, that can act as a food source, is a factor that could determine the benthic macrofuana population dynamics, enrichening when food availability is greater (Lastra et al., 2006; Rodil et al., 2007; McLachlan and Defeo, 2018). Finally, the distance to sources of fresh water, encouraged the greater abundance of polychaete deposit feeders, conditioned by the contributions of finer sediments and organic matter at the deepest points of transects A, B and C. Salinity is a factor that some studies (Lercari et al., 2002; Lercari and Defeo, 2006; Lozoya et al., 2010; Ortega Cisneros et al., 2011) have demonstrated that negatively influence the abundance and richness of macrofauna, however, due to the subtle changes observed in our study, this pattern did not occur.

Our results show that in the shallower sublittoral zone, which includes the surf zone, the benthic macrofauna communities are strongly influenced by physical factors associated with the beach profile and hydrodynamics, i.e the granulometric parameters. However, in the zone beyond the breaking area, where the turbulence is lower, the importance of nutritional variables, i.e. the organic matter in the sediment and the dynamics of phytoplanktonic groups, increases. In addition, the distance to the freshwater discharges presented an influence on the fauna especially in the deepest and least turbulent zone. This highlights the importance of considering both physical and nutritional variables in the study of sublittoral benthic communities in micro-tidal zones, agreeing with the paradigms described by McLachlan (2001) and Ortega Cisneros et al. (2011). The identification of these key variables is crucial to develop adequate coastal management. Anthropogenic activities in coastal areas such as coastal structures, beach nourishment, dredging, freshwater discharges, ..., through modifying the characteristics of the environment (hydrodynamism, beach profile, granulometry and nutritional sources) will affect the biocoenoses. This could have consequences on the food web, as well as socio-economic repercussions when affecting species of high fishing interest.

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5.7 SUPPORTING INFORMATION

Table 5.S1. ANOVA multifactorial tests for the environmental variables of July 2013. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)					**Depth (m)**			
	p-value	**post hoc**				**p-value**	**post hoc**		
Salinity	0	E	X			0.4107	N.S.		
		B	X	X					
		D		X	X				
		C			X				
		A			X				
SS	0.2481	N.S.				0.052	N.S.		
DIN	0.0048	E	X			0.0047	4	X	
		A	X				3	X	X
		C	X				2	X	X
		D	X	X			0.5		X
		B		X			1		X
Si(OH)$_4$	0.0023	C	X			0.0217	4	X	
		A	X				3	X	
		E	X				2	X	
		D	X	X			1	X	
		B		X			0.5	X	
PO$_4^{-3}$	0.3124	N.S.				0.0729	N.S.		
TP	0.013	D	X			0.4488	N.S.		
		E	X						
		A	X	X					
		C	X	X					
		B		X					
Alloxanthin	0.2161	N.S.				0.1941	N.S.		

	Distance (station)					**Depth (m)**			
	p-value	**post hoc**				**p-value**	**post hoc**		
Chl_*a*	0.0231	C	X			0.6128	N.S.		
		D	X	X					
		A	X	X					
		E	X	X					
		B		X					
Chl_*b*	0.0037	C	X			0.8086	N.S.		
		A	X	X					
		D		X					
		E		X					
		B		X					
Fucoxanthin	0.0148	C	X			0.2219	N.S.		
		A	X	X					
		D	X	X					
		E	X	X					
		B		X					
Lutein	----	----				----	----		
Neoxanthin	----	----				----	----		
Peridinin	0	E	X			0.3271	N.S.		
		D	X						
		C	X	X					
		A		X					
		B			X				
Prasinoxanthin	----	----				----	----		
Violoxanthin	0.2782	N.S.				0.2188	N.S.		
Zeaxanthin	0.2856	N.S.				0.1434	N.S.		
19' butanoyloxyfucoxanthin	----	----				----	----		
19' hexanoyloxyfucoxanthin	0	A	X			0.0415	4	X	
		C	X				3	X	
		B	X				2	X	
		D		X			1	X	
		E		X			0.5	X	
Mean grain size	0.3619	N.S.				0.005	4	X	
							3	X	X
							1	X	X
							2	X	X
							0.5		X
OM	0.072	N.S.				0.0327	1	X	
							0.5	X	X
							2	X	X
							3	X	X
							4		X

Table 5.S2. ANOVA multifactorial tests for the environmental variables of February 2014. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)					**Depth (m)**				
	p-value	**post hoc**				**p-value**	**post hoc**			
Salinity	0.1347	N.S.				0.1475	N.S.			
SS	0.0014	C	X			0.123	N.S.			
		B	X							
		D	X	X						
		E		X	X					
		A			X					
DIN	0.0197	D	X			0.1059	N.S.			
		E	X	X						
		A	X	X						
		C		X						
		B		X						
$Si(OH)_4$	0.0784	N.S.				0.752	N.S.			
PO_4^{-3}	0.8074	N.S.				0.5799	N.S.			
TP	0.2928	N.S.				0.6748	N.S.			
Alloxanthin	0	A	X			0.1136	N.S.			
		B	X							
		E	X	X						
		C		X	X					
		D			X					
Chl_*a*	0.034	A	X			0.5458	N.S.			
		B	X	X						
		C	X	X						
		E	X	X						
		D		X						
Chl_*b*	0.0057	B	X			0.6085	N.S.			
		A	X							
		C	X	X						
		D	X	X						
		E		X						
Fucoxanthin	0.8304	N.S.				0.6891	N.S.			
Lutein	0.919	N.S.				0.2613	N.S.			
Neoxanthin	0.8053	N.S.				0.713	N.S.			
Peridinin	----	----				----	----			
Prasinoxanthin	0.2077	N.S.				0.5781	N.S.			
Violoxanthin	0.0691	N.S.				0.1142	N.S.			
Zeaxanthin	0.1359	N.S.				0.2771	N.S.			
19' butanoyloxyfucoxanthin	0.0856	N.S.				0.6639	N.S.			
19' hexanoyloxyfucoxanthin	0.0032	C	X			0.9707	N.S.			
		B	X							
		A	X							
		E	X	X						
		D		X						
Mean grain size	0.4492	N.S.				0.0013	4	X		
							3	X	X	
							2	X	X	X
							1		X	X
							0.5			X
OM	0.3593	N.S.				0.0854	N.S.			

Table 5.S3. ANOVA multifactorial tests for the environmental variables of December 2014. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)		**Depth (m)**	
	p-value	**post hoc**	**p-value**	**post hoc**
Salinity	0.6611	N.S.	0.003	0.5 X 1 XX 2 X 4 X 3 X
SS	0.0288	D X C X B XX A XX E X	0.2191	N.S.
DIN	0.0872	N.S.	0.1354	N.S.
$Si(OH)_4$	0.0722	N.S.	0.1514	N.S.
PO_4^{-3}	0.3435	N.S.	0.1071	N.S.
TP	0.0029	B X A XX C XX D X E X	0.679	N.S.
Alloxanthin	0.0263	E X D XX B XX C XX A X	0.7586	N.S.
Chl_*a*	0.1272	N.S.	0.9396	N.S.
Chl_*b*	0.172	N.S.	0.2687	N.S.
Fucoxanthin	0.2802	N.S.	0.8446	N.S.
Lutein	----	----	----	----
Neoxanthin	----	----	----	----
Peridinin	----	----	----	----
Prasinoxanthin	----	----	----	----
Violoxanthin	----	----	----	----
Zeaxanthin	----	----	----	----
19' butanoyloxyfucoxanthin	----	----	----	----
19' hexanoyloxyfucoxanthin	----	----	----	----
Mean grain size	0.2132	N.S.	0	4 X 3 X 2 X 0.5 X 1 X
OM	0.3682	N.S.	0.0385	1 X 0.5 X 2 X 3 X 4 X

Table 5.S4. Kruskal-Wallis and Bonferroni post hoc tests for environmental variables. The sampling campaigns are organised from the lowest to the highest value.

	p-value	post hoc			
Temperature	0	Feb-14	X		
		Dec-14		X	
		Jul-13			X
Salinity	1.90E-10	Feb-14	X		
		Dec-14		X	
		Jul-13		X	
SS	1.75866E-06	Feb-14	X		
		Jul-13		X	
		Dec-14		X	
DIN	0.00105747	Jul-13	X		
		Feb-14	XX		
		Dec-14		X	
Si $(OH)_4$	2.46E-07	Feb-14	X		
		Jul-13		X	
		Dec-14		X	
PO_4^{3-}	0.252002	N.S.			
TP	9.63E-07	Feb-14	X		
		Jul-13		X	
		Dec-14		X	
Alloxanthin	0.00E+00	Dec-14	X		
		Jul-13		X	
		Feb-14			X
Chl_*a*	2.95E-09	Dec-14	X		
		Jul-13	X		
		Feb-14		X	
Chl_*b*	1.01E-10	Dec-14	X		
		Jul-13	X		
		Feb-14		X	
Fucoxanthin	3.86E-12	Dec-14	X		
		Jul-13	X		
		Feb-14		X	
Lutein	1.35E-02	Dec-14	X		
		Jul-13	X		
		Feb-14	X		
Neoxanthin	8.81E-05	Dec-14	X		
		Jul-13	XX		
		Feb-14		X	
Peridinin	8.70E-07	Dec-14; Feb-14	X		
		Jul-13		X	
Prasinoxanthin	2.48E-11	Jul-13	X		
		Dec-14	X		
		Feb-14		X	
Violoxanthin	8.22E-10	Dec-14	X		
		Jul-13		X	
		Feb-14			X
Zeaxanthin	1.33E-03	Dec-14	X		
		Feb-14	XX		
		Jul-13		X	

	p-value	post hoc		
19' butanoyloxyfucoxanthin	3.84E-07	Dec-14	X	
		Jul-13	X	
		Feb-14		X
19' hexanoyloxyfucoxanthin	3.16E-08	Dec-14	X	
		Feb-14	X	
		Jul-13		X
Mean grain size	0.512928	N.S.		
OM	0.0616999	N.S.		

Table 5.S5. ANOVA multifactorial tests for the fauna variables of July 2013. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)				**Depth (m)**					
	p-value	**post hoc**			**p-value**	**post hoc**				
Total density	0.2495	N.S.			0	0.5	X			
						1	X			
						2	X			
						3		X		
						4		X		
Bivalve density	0.4365	N.S.			0.5021	N.S.				
Crustacea density	0.2701	N.S.			0.0006	0.5	X			
						1	X			
						2	X			
						4	X	X		
						3		X		
Polychaeta density	0.0042	E	X		0	1	X			
		D	X	X		0.5	X	X		
		B		X		2		X	X	
		C		X		3			X	X
		A		X		4				X
Donax trunculus density	0.2316	N.S.			0	4	X			
						3	X			
						2	X			
						1		X		
						0.5		X		
Chamelea gallina density	0.2928	N.S.			0.001	0.5	X			
						2	X			
						1	X			
						3	X	X		
						4		X		

Table 5.S6. ANOVA multifactorial tests for the fauna variables of February 2014. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)				Depth (m)				
	p-value	**post hoc**			**p-value**	**post hoc**			
Total density	0.0141	E	X		0	0.5	X		
		D	X	X		1	X		
		C	X	X		2		X	
		A		X		3		X	
		B		X		4		X	
Bivalve density	0.5507	N.S.			0.0002	0.5	X		
						1	X		
						2	X	X	
						3		X	X
						4			X
Crustacea density	0.023	E	X		0	0.5	X		
		D	X	X		1	X		
		A	X	X		2		X	
		C	X	X		3		X	
		B		X		4		X	
Polychaeta density	0.1276	N.S.			0	0.5	X		
						1	X		
						2		X	
						3			X
						4			X
Donax trunculus density	0.2201	N.S.			0.0098	3	X		
						4	X		
						0.5	X	X	
						2	X	X	
						1		X	
Chamelea gallina density	0.6452	N.S.			0.0093	0.5	X		
						2	X		
						1	X		
						3	X		
						4		X	

Table 5.S7. ANOVA multifactorial tests for the fauna variables of December 2014. Stations and depths are organised from the lowest to the highest in the results of the Tukey post hoc tests.

	Distance (station)				**Depth (m)**					
	p-value	**post hoc**			**p-value**	**post hoc**				
Total density	0.0105	D	X		0	0.5	X			
		E	X	X		1	X			
		C	X	X		2	X			
		B		X		3		X		
		A		X		4			X	
Bivalve density	0.0683	N.S.			0.0001	2	X			
						1	X	X		
						0.5	X	X		
						3		X	X	
						4			X	
Crustacea density	0.0316	C	X		0	0.5	X			
		E	X			1		X		
		D	X			2		X	X	
		A	X			3			X	X
		B	X			4				X
Polychaeta density	0.4467	N.S.			0.0007	1	X			
						0.5	X			
						2	X	X		
						3		X	X	
						4			X	
Donax trunculus density	0.1866	N.S.			0	4	X			
						3	X			
						2	X			
						1		X		
						0.5		X		
Chamelea gallina density	0.4362	N.S.			0.0705	N.S.				

Table 5.S8. Kruskal-Wallis and Bonferroni post hoc tests for fauna variables. The sampling campaigns are organised from the lowest to the highest value.

	p-value	**post hoc**		
Total density	0.00486453	Feb-14	X	
		Dec-14	X	
		Jul-13		X
Bivalve density	0.00826023	Feb-14	X	
		Dec-14	X	
		Jul-13		X
Crustacea density	0.027758	Feb-14	X	
		Dec-14	XX	
		Jul-13		X
Polychaeta density	0.163312	N.S.		
Donax trunculus density	0.119624	N.S.		
Chamelea gallina density	0.0793207	N.S.		

Capítulo 6

Discusión general

6. DISCUSIÓN GENERAL

6.1 EVOLUCIÓN DE LA PESQUERÍA Y POBLACIONES DE *Donax trunculus* Y *Chamelea gallina*

6.1.1 Antes del cierre de la pesquería: desde 2004 hasta junio de 2015

Como se ha descrito en el capítulo 3, la pesquería de *D. trunculus* y *C. gallina,* en las zonas de trabajo de la Cofradía de Pescadores de Gandia, para los periodos temporales 2004-2014 en el caso de *D. trunculus* y 2007-2014 para *C. gallina*, sufrió un descenso de capturas que estuvo acompañado por una caída en la flota y esfuerzo pesquero (Tablas 3.2 y 3.3). Estos descensos fueron muy acusados a partir del año 2010. Las capturas de *D. trunculus,* durante los años 2004-2008, llegaron a representar un 40 % del conjunto de capturas del Mediterráneo español (Tabla 3.2), disminuyendo a valores del 2 % en el año 2014. En el caso de *C. gallina* fueron del 33 % en el 2008 (Tabla 3.3), descendiendo a porcentajes entre 0 y 3 % durante 2012-2014. Acontecimientos similares de descensos en las capturas en el Mediterráneo fueron descritos por autores como Lart et al. (2003), Romanelli et al. (2009) y Baeta et al. (2018) como consecuencia de una mala gestión pesquera, un descenso de los aportes de agua dulce o a cambios granulométricos. No obstante, los datos de capturas en Gandia y en el resto del Mediterráneo español mostraron tendencias diferentes, presentando las de Gandia una disminución marcada mientras que las del resto fueron relativamente estables. Además, los datos evidenciaron que en el *métier* dominaron las capturas de *D. trunculus* con respecto a *C. gallina*. Ramón et al. (2005) ya afirmaron que los bancos de *C. gallina* habían desaparecido parcialmente en las costas de Cullera y Gandia.

La disminución a partir del 2010 de los valores de CPUE, considerado como índice de abundancia relativa (Gatica y Hernández, 2003), podría indicar que la pesquería sufrió sobreexplotación. En el caso de *D. trunculus*, la reducción de las capturas, junto con el menor número de embarcaciones y el esfuerzo pesquero llevó a una estabilidad de los valores de CPUE de 2004 a 2008, con valores de en torno a 40 (Tabla 3.2). A partir del 2010, pese a mostrar valores de CPUE por debajo de 19, que fue la mitad del valor promedio de CPUE de los 6 años anteriores, se continuó trabajando, aún sin alcanzar los kg mínimos rentables para su pesca que se estimó en torno a 20 kg por día de trabajo y embarcación (E. Ferrer, comunicación personal, Cofradía de Pescadores de Gandia, 2015). Esto hace sospechar una posible venta de parte de las capturas de forma ilegal, tal y como observaron Baeta et al. (2018) para la costa catalana, en las que hasta el 20 % de las capturas se comercializaron fuera de los canales oficiales.

Por otro lado, este *métier* presenta una marcada estacionalidad, con incrementos de esfuerzo pesquero, capturas, flota, CPUE y precio en primavera-verano (Figura 3.3 y Tabla 3.6). Dicha estacionalidad está vinculada, entre otros aspectos, con las condiciones más favorables del mar para su pesca (Sardá et al., 2000), así como por la mayor demanda de estos organismos durante los meses de primavera-verano. Pero, además, la estacionalidad en la pesca también está relacionada a los ciclos de vida de los organismos, ya que los procesos fisiológicos tales como el crecimiento y la reproducción, aumentan en primavera y verano debido a incrementos de temperatura (Ansell y Lagardère, 1980; Manca Zeichen et al., 2002), por lo que requieren una mayor fuente de alimento debido a la alta demanda metabólica (Dionicio y Flores, 2015), ubicándose más cerca de la superficie del sedimento y pasando a ser más accesibles para la pesca. Cabe destacar que el aumento de la presión pesquera en primavera y verano coincide con el período de reproducción de estos organismos, lo que podría afectar la supervivencia de la especie.

Existió una presión pesquera desigual en los distintos tramos del sublitoral en los que se realiza el marisqueo por parte de las embarcaciones de la Cofradía de Pescadores de Gandia, que tienen su puerto base en Gandia. Las zonas que presentaron mayor presión pesquera fueron aquellas ubicadas inmediatamente al sur del Puerto de Gandia (IV, V, VI y VII) (Figura 3.1), concentrándose en ellas el mayor esfuerzo pesquero (principalmente por la proximidad al puerto y la optimización del tiempo y gastos de combustible), además de presentar las mayores capturas por km de costa (Tabla 3.5). Dichas zonas, podrían presentar una mayor productividad al verse afectadas por los aportes del río Serpis ya que las corrientes dominantes provenientes del norte transportan la pluma de agua dulce hacia el sur (Sebastiá y Rodilla, 2013). No obstante, para el caso de *D. trunculus* estas zonas no fueron las que presentaron mayor CPUE (Tabla 3.5). Esto se debe a que, pese a que puedan presentar una mayor productividad, la explotación tan continuada que se ejerce sobre ellas podría haber reducido sus poblaciones, necesitando un mayor esfuerzo pesquero para obtener las mismas capturas, viéndose reducida así la CPUE.

El descenso de las capturas en esta década llevó finalmente al cierre de la pesquería en junio del 2015.

6.1.2 Después del cierre de la pesquería: desde junio de 2015 a 2017

En el capítulo 4 se determinó que justo antes del cierre de la pesquería, entre finales de mayo-principios de junio de 2015, la biomasa de los individuos de tamaño comercial de *D. trunculus* varió de < 0.05 kg ha^{-1} a 108 kg ha^{-1} en la zona estudiada (Tabla 4.1). Solamente en el 45 % de las 22 muestras (11 estaciones a dos profundidades), se alcanzaron o superaron los 20 kg ha^{-1} (cantidad mínima rentable para la pesca), encontrándose el 70 % de dichos casos en la zona más somera. Una vez cerrada la pesquería y analizando los cinco meses posteriores, se observó un incremento en julio, agosto y septiembre de 2015, de las densidades y biomasa de *D. trunculus* tanto de individuos comerciales como total (Tabla 4.1 y 4.2). En los meses posteriores al cierre el porcentaje de casos que alcanzaron o excedieron los 20 kg ha^{-1} para individuos de tamaño comercial varió entre el 56 al 86 %. Este incremento coincidió con la clara estacionalidad de las poblaciones en primavera-verano ya descrita, ya que estos valores volvieron a descender en los meses de octubre y noviembre. Además, en el año 2017, mediante el análisis de las poblaciones entre junio a septiembre, también se pudo observar el mismo patrón estacional, si bien nunca se alcanzó el mínimo rentable para la pesca. El patrón de disminución de las poblaciones a partir de octubre concuerda con los descensos poblacionales que observaron Manca Zeichen et al. (2002) y Neuberger-Cywiack et al. (1990) en otoño-invierno, vinculado con la bajada de las temperaturas. Cabe destacar que en 2017 se observó una clara disminución de la población con respecto a 2015, lo que nos indica que el cierre de la pesquería no resolvió el problema. Un desarrollo similar se observó en la pesquería del bivalvo *Mesodesma mactroides* en Uruguay, en la que primero, tras un año del cierre de la pesquería, se apreció una recuperación de las poblaciones, pero, cuando se alcanzaron casi los tres años de clausura, se vio repercutido el reclutamiento (Defeo, 1996; Brazeiro y Defeo 1999; Ortega et al., 2012).

Además, se pudo establecer una distribución de frecuencia de tallas a dos profundidades distintas, una de ellas entre 0.5-1.5 m (Figura 4.4) y la otra 1.5-3 m (Figura 4.5). Los organismos más pequeños fueron encontrados en la zona más somera, al igual que observaron Baeta et al. (2018) en la costa catalana, Gaspar et al. (2002a) en Portugal, Ansell y Lagardere (1980) en la costa atlántica francesa, Manca Zeichen et al. (2002) en Italia y Deval (2009) en Turquía. Algunos autores describen esta distribución diferenciada en profundidad como una estrategia de supervivencia (Gaspar et al., 2002a). Estas zonas más someras sirven como fuente de larvas de *D. trunculus* debido a que su colonización se ve favorecida por las corrientes e hidrodinámica que las arrastran hacia las zonas menos profundas (Manca Zeichen et al., 2002). Además, Scheltema (1971) explicó este suceso debido a la presencia de mejores condiciones para el desarrollo de juveniles en la zona más somera, y a una mayor disponibilidad de alimento.

Por otra parte, el aumento del tamaño medio de los organismos durante los meses siguientes al cierre en el 2015 (Figura 4.2) constató que las poblaciones se mantuvieron e incrementaron de tamaño cada mes, y que hubo un reclutamiento en verano (agosto-septiembre). Además, si bien los parámetros de crecimiento obtenidos para *D. trunculus* en 2015 y 2017 mostraron valores más bajos que otras áreas del golfo de Valencia (Ramón et al., 1995), los valores de ϕ' no presentaron diferencias estadísticas significativas con respecto a ninguno de los estudios citados en la tabla 4.5. Pese a que Vakily (1992) afirmó que los bivalvos crecen más rápido en latitudes más bajas debido a un incremento en sus tasas metabólicas, vinculado con una mayor temperatura, esto no fue así para los datos recopilados de diferentes áreas en la tabla 4.5, coincidiendo con Gaspar et al. (1999). Esto probablemente se deba a la poca variación latitudinal de las ubicaciones analizadas. La variabilidad en estos parámetros de crecimiento se puede deber a las diferentes condiciones ambientales de cada zona (Vakily, 1992; Gaspar et al., 1999), presentando *D. trunculus* una notable plasticidad bajo diferentes condiciones (Manca Zeichen et al., 2002).

Respecto a *C. gallina* después del cierre, las capturas tanto en 2015 como 2017 podemos considerarlas anecdóticas no existiendo evidencias de regeneración de los bancos de este bivalvo.

6.2 FACTORES AMBIENTALES VS EXPLOTACIÓN PESQUERA

El estudio de factores ambientales que influyen sobre la macrofauna bentónica (Capítulo 5) se realizó durante los dos años previos al cierre de la pesquería en tres campañas de muestreo: julio de 2013, febrero de 2014 y diciembre de 2014. Estos periodos de muestreo se correspondieron a dos épocas del año bien diferenciadas: la estival seca (julio de 2013) en la que no existió descarga de agua por parte del río, y hubo una precipitación mensual acumulada muy baja (< 2 mm), y por otra parte la más húmeda (febrero de 2014 y diciembre de 2014) en la que sí se apreció cierto caudal de agua en la desembocadura del río (1 $m^3 s^{-1}$ y 0.8 $m^3 s^{-1}$ respectivamente), y mayores precipitaciones mensuales acumuladas (12.4 y 96.2 mm respectivamente).

Para determinar el efecto de los distintos factores ambientales sobre la macrofauna bentónica se trabajó a mesoescala, en cinco estaciones de muestreo, y cinco profundidades, ubicadas a lo largo de la costa a diferentes distancias de las fuentes de agua dulce en el entorno de la desembocadura del río Serpis y del Puerto de Gandia (Figure 5.1).

Cuando se analizó la distribución espacial en función de la profundidad en el estudio realizado a mesoescala, las variables fisicoquímicas del sedimento mostraron una clara influencia (Tablas 5.S1, 5.S2 y 5.S3), disminuyendo el tamaño medio de grano e incrementando el contenido de materia orgánica (OM) en profundidad. Estas pautas en las variables fisicoquímicas del sedimento también fueron observadas por Ramón (1993) y Martí et al. (2007) en un área ubicada justo al norte de nuestra zona de estudio. Estos patrones son consecuencia de la sedimentación y acumulación en el fondo marino tanto de materiales geológicos más finos como de la mayor parte de la materia orgánica particulada. El tamaño de partícula de arena más pequeño está relacionado con mayores densidades de macrofauna y riqueza (McLachlan y Dorvlo, 2005; Rodil et al., 2006; McLachlan y Defeo, 2018), patrón que sigue también la zona de estudio.

En cuanto a la variación de las variables ambientales del agua con respecto a la profundidad y consecuentemente a la distancia a la costa (Tablas 5.S1, 5.S2 y 5.S3), se puede decir que la salinidad no varió con la profundidad excepto en la campaña de diciembre de 2014 donde se apreciaron niveles ligeramente más bajos en la zona más somera (con un promedio de salinidad de 37.3 en las zonas menos profundas y 37.6 en las más profundas). Para nutrientes solo en verano (julio de 2013) se encontraron mayores concentraciones de nitrógeno inorgánico disuelto (DIN) en la zona más somera. Este incremento del DIN podría ser debido a los aportes de fertilizantes nitrogenados, durante la primavera y verano, utilizados en la agricultura de cítricos presentes en la zona de estudio (Ramos et al., 2002), que finalmente entrarían en la zona costera a través de las aguas subterráneas.

Cuando se analizó la distribución espacial de la fauna en función de la profundidad en el estudio llevado a cabo a mesoescala (Capítulo 5), las densidades tanto de *D. trunculus* como de *C. gallina* mostraron claras pautas en las tres campañas de muestreo, presentando ambas especies diferencias estadísticamente significativas con la profundidad (Tablas 5.S5, 5.S6 y 5.S7). *D. trunculus* mostró mayores densidades en los puntos más someros (fundamentalmente 0.5 y 1 m de profundidad) y menores en los más profundos (3 y 4 m). Este donácido siguió el mismo patrón ya descrito por Ramón et al. (1995) y Gaspar et al. (2002a), que se caracteriza por presentar una mayor densidad de individuos entre 0.5 y 2 m de profundidad. *C. gallina* mostró una pauta de distribución contraria a *D. trunculus*, con mayores densidades en la zona más profunda, coincidiendo con el rango batimétrico observado por Moschino y Marin (2006), aunque no fue una de las especies más abundantes. En cuanto al resto de la fauna, la densidad de crustáceos y de poliquetos así como la densidad total de organismos también se incrementó con la profundidad en la campaña de verano, mientras que en febrero 2014 y diciembre 2014 además se observó esta pauta para la clase bivalvos (Tablas 5.S5, 5.S6 y 5.S7). El patrón de incremento generalizado de la diversidad y abundancia de especies en profundidad fue explicado por Janssen y Mulder (2005) y McLachlan y Defeo (2018) como consecuencia de una mayor estabilidad en los sedimentos más profundos.

La distribución espacial en función de la profundidad parece ser un factor determinante para la distribución de las especies comerciales, así como para los otros grupos de organismos (bivalvos, poliquetos, crustáceos y densidad total). Esta zonación batimétrica clara está vinculada, como ya se ha visto en el capítulo 5, al tamaño de grano medio del sedimento (McLachlan, 1996; Rodil y Lastra, 2004; McLachlan y Dorvlo, 2005; Lastra et al., 2006) y al porcentaje de OM (Lercari y Defeo, 2006) que a su vez están gobernados por la hidrodinámica de la playa. Por tanto, cambios hidrodinámicos en las playas acabarán modificando la zonación batimétrica de la comunidad biológica. La influencia de la granulometría y OM sobre la fauna bentónica además fue demostrada mediante el análisis de correspondencia canónica (Capítulo 5), coincidiendo con lo descrito por Lercari y Defeo (2006). Además, De la Huz et al. (2002) y La Valle et al. (2011) remarcaron la importancia de un sedimento con tamaño de grano adecuado para el establecimiento del bivalvo *D. trunculus*.

En el análisis de la distribución espacial en función de la distancia a los aportes de aguas dulces en el estudio a mesoescala se mostró que las variables del sedimento, tamaño de grano medio y OM, no presentaron diferencias significativas entre estaciones (Tablas 5.S1, 5.S2 y 5.S3). Sin embargo, en la zona más profunda, las estaciones más cercanas a los aportes presentaron las mayores cantidades de OM. En el caso de las variables analizadas en el agua sí que se observaron pautas de distribución espacial ligadas a los aportes de agua dulce (Tablas 5.S1, 5.S2 y 5.S3). En general la estación B, situada a 200 m al sur de la desembocadura del río Serpis, fue la más afectada, presentando menor salinidad (con un valor mínimo de 36.8 en febrero de 2014) y mayores concentraciones de nutrientes. El hecho de que al sur de la desembocadura del río Serpis exista un espigón que genera un efecto pantalla sobre la zona más próxima (Estación A), hace que la estación B se vea más influenciada por los aportes de agua dulce que la estación más cercana. Además, habría que considerar que la zona de estudio también presenta descargas procedentes de las acequias de drenaje del humedal que vierten directamente en el Puerto de Gandia, así como del aliviadero de la Estación depuradora de aguas residuales de Gandia en el tramo final del Serpis. En verano, al triplicarse la población de Gandia se supera el caudal de aguas residuales que es capaz de verter el emisario submarino, por lo que parte de estas aguas son aliviadas en la desembocadura del río Serpis (Sebastiá et al., 2013).

En el estudio realizado a mesoescala, las densidades de *D. trunculus* y *C. gallina* no presentaron diferencias estadísticamente significativas en función de la distancia a los aportes de aguas dulces (Tablas 5.S5, 5.S6 y 5.S7) para ninguna época del año. No obstante, sí que presentaron diferencias significativas entre estaciones los poliquetos en la campaña de verano, la densidad total y la de crustáceos en la campaña de febrero de 2014, y la densidad total en diciembre de 2014, mostrando mayores valores las estaciones más cercanas a los aportes de agua dulce (A, B y C). Cabría destacar, que el hecho de que no se observaran diferencias significativas entre zonas para las dos especies comerciales, a diferencia del resto de grupos, se pudo deber, a que durante los períodos de muestreo todavía estaba activa la pesquería de ambas especies y los resultados podrían estar sesgados por la mortalidad que ocasiona la pesca, a diferencia del resto de organismos estudiados (Gaspar et al., 2002b; Morello et al., 2005; Martí et al., 2007). Sin embargo, en el análisis de los datos históricos de la pesquería considerados en el capítulo 3, que abarcó una área de estudio más amplia, desde Tavernes de la Valldigna a Denia (Figura 3.1), se observó, para *D. trunculus,* que las zonas situadas al sur del principal aporte de agua dulce, río Serpis y drenaje del marjal de La Safor (zonas IV, V y VI), presentaron mayores capturas totales, mayores capturas por km de costa, y a pesar de presentar mayores esfuerzos pesqueros, los valores promedio de CPUE fueron similares al del resto de las zonas indicando, por lo tanto, una mayor productividad (Tabla 3.5). Para *C. gallina*, aunque este tipo de análisis solo se pudo hacer para el año 2014, se observó que, en las zonas al sur de los aportes del río Serpis, la CPUE fue mayor. Tanto *D. trunculus* como *C. gallina* son filtradores que se alimentan de material particulado en suspensión en el agua (Romanelli et al., 2009; McLachlan y Defeo, 2018). El tamaño de las poblaciones está íntimamente relacionado con la cantidad y calidad del material particulado (Lastra et al., 2006; Lee et al., 2019). Sin embargo, en las aguas del Mediterráneo, al ser de un marcado carácter oligotrófico, las descargas de aguas dulces con sus nutrientes asociados juegan un papel clave en la productividad marina (Ludwing et al., 2009) afectando fundamentalmente a los entornos cercanos a las desembocaduras. Por tanto, a macroescala podría observarse un incremento de la abundancia de las poblaciones en relación a los aportes fluviales, que no pudo ser observado a nivel de mesoescala para las especies comerciales.

Un factor ambiental relevante a considerar es la temperatura que refleja cambios claramente estacionales en la zona de estudio. El análisis de los cambios estacionales mostró evidentes diferencias de temperatura entre campañas (Tabla 5.S4), con valores máximos en julio de 2013 (entre 24 y 27 ºC) y mínimos en febrero de 2014 (11-14 ºC) (Tabla 5.1). En el caso de la salinidad, si bien las variaciones fueron muy sutiles entre campañas y con una diferencia de 0.6 entre la media de las campañas, se observaron los valores más bajos en febrero de 2014, debido al mayor caudal aportado por el río Serpis (Tabla 5.S4). En diciembre de 2014, con un caudal ligeramente menor, y julio de 2013, sin caudal, la salinidad de las aguas fue más alta y similar entre ellas. Sin embargo, los mayores valores de nutrientes como DIN y silicio ($Si(OH)_4$) se presentaron en diciembre de 2014 (Tabla 5.S4), época en la que ocurrieron las mayores precipitaciones. Sebastiá et al. (2013) y Gadea et al. (2013) también observaron mayores concentraciones de DIN y $Si(OH)_4$ en la zona de estudio durante períodos lluviosos, como consecuencia de la escorrentía agrícola y la meteorización química del terreno.

En cuanto a las variables nutricionales del agua, incrementos en la abundancia de fitoplancton a finales de invierno y principios de primavera, cuando se incrementa la disponibilidad de luz, han sido observados en la zona de estudio por autores como Gadea et al. (2013) y Sospedra (2014), sucediendo lo mismo en el presente estudio. En febrero de 2014, cuando la salinidad y temperatura fueron menores, se encontraron las concentraciones más altas de la mayoría de los pigmentos fotosintéticos analizados (Tabla 5.S4). Algunos de estos pigmentos, al estar asociados a determinadas divisiones de algas, nos revelan, aunque sea parcialmente, la composición de la población fitoplanctónica (Jeffrey y Vesk, 1997; Vidussi et al., 2001; Gallardo, 2004; Brewin et al., 2010). De esta forma se pudo distinguir composiciones muy distintas entre finales de invierno (febrero de 2014) y verano (julio de 2013). A finales de invierno, los altos niveles de pigmentos como Chl_*a*, Chl_*b*, fucoxantina, neoxantina, prasinoxantina, 19' butanoiloxifucoxantina, aloxantina y violoxantina indicaron la presencia de algas verdes (clorofíceas, prasinofíceas y euglenofíceas), cianofíceas, criptofíceas, primnesofíceas y crisofíceas, pero fundamentalmente de diatomeas, concordando con Garmendia et al. (2011), Levinton (2011) y Sebastiá y Rodilla (2013) que indicaron que las diatomeas dominan durante el final del invierno y la primavera. Sin embargo, en la campaña de verano los pigmentos 19' hexanoiloxifucoxantina, peridinina y zeaxantina revelaron mayores abundancias de dinoflagelados y cianobacterias, al igual que observaron Sebastiá et al. (2013) en el área de estudio durante la estación seca. En las zonas templadas los dinoflagelados y cianobacterias presentan un incremento en verano-otoño (Levinton, 2011).

Dos variables relacionadas en cierta medida con el plancton como sólidos suspendidos (SS) y fósforo total (TP), que también fueron analizados, mostraron niveles más altos en las campañas de julio de 2013 y diciembre de 2014, mientras que la mayoría de los pigmentos fotosintéticos tuvieron máximas concentraciones en febrero de 2014. Los SS incluyen partículas orgánicas (organismos y materia muerta) e inorgánicas, y el TP comprende tanto al fósforo disuelto como particulado, que a su vez incluye al fósforo presente en organismos, materia orgánica muerta y adsorbido a partículas. Esto indicaría que en julio de 2013 y diciembre de 2014, los detritos y partículas inorgánicas fueron los componentes dominantes de estas variables en vez de los organismos fitoplanctónicos. Por tanto, parecería que una mayor biomasa de diatomeas hacia finales de invierno y principios de primavera desencadenó dos procesos: el primero de ellos fue un incremento de la abundancia de productores secundarios que ajustan su ciclo vital a la mayor disponibilidad de alimento (Zhang et al., 2015), mientras que el segundo fue una mayor cantidad de detritos por la muerte y sedimentación tanto de productores primarios como de algunos secundarios.

En la columna de agua, las microalgas podrían determinar la dinámica poblacional de la macrofauna bentónica, enriqueciéndola cuando la disponibilidad de alimentos es mayor (Lastra et al., 2006; Rodil et al., 2007; McLachlan y Defeo, 2018). La mayor disponibilidad de fitoplancton durante febrero de 2014, principalmente diatomeas, debería haber favorecido la presencia de filtradores, hecho que no se observó en esta campaña. El patrón de zonación del sublitoral descrito por McLachlan y Defeo (2018) propone que en las zonas más turbulentas predominan las diatomeas. En esta misma zona, Wildish (1977) y Brown et al. (1989) establecen la dominancia de filtradores con la exclusión de los depositívoros. Sin embargo, Herman et al. (1999) plantean que la comunidad bentónica integra las influencias ambientales de un lugar durante un periodo de tiempo relativamente largo, por lo que los incrementos en el sistema planctónico se verían reflejados en el bentónico con cierto desfase temporal. No obstante, los incrementos fitoplanctónicos de finales de invierno-principio de primavera sí que generan un incremento más rápido en la comunidad zooplanctónica (Rodrigues et al., 2019) que supone también la presencia de meroplancton que posteriormente se traducirá en reclutas para la comunidad bentónica (Marcus y Boero, 1998) y además contribuirá al aumento de detritos disponibles tras los picos de producción de primavera y otoño (Soetaert et al., 2000). El incremento de detritos durante la campaña de verano pudo favorecer la presencia de muchas especies de poliquetos, misidáceos… que se alimentan de ellos. La deposición de detritos se produce principalmente a mayor profundidad, siendo las condiciones de esta zona las adecuadas para el asentamiento de poliquetos depositívoros principalmente (Fauchald y Jumars, 1979; Jumars et al., 2015).

Las variaciones estacionales de la macrofauna de playas arenosas en zonas templadas están vinculadas a los cambios de temperatura del agua ya que, durante el invierno, en condiciones de baja temperatura del agua, los organismos tienden a desplazarse o enterrarse a mayor profundidad (McLachlan y Defeo, 2018). Diversos autores como Neuberger-Cywiak et al. (1990) y Manca Zeichen et al. (2002) explican la estacionalidad de la densidad de *D. trunculus* por los descensos de la temperatura del agua. Sin embargo, en el estudio a mesoescala no se pudieron apreciar cambios estadísticamente significativos en las densidades de *D. trunculus* y *C. gallina* (Tabla 5.S8), aunque sí que se mostraron valores más altos en julio de 2013, coincidiendo con las temperaturas más altas del agua. En cuanto al resto de la fauna, la densidad total, la de bivalvos y de crustáceos presentaron estadísticamente una estacionalidad marcada con mayores densidades en verano concordantes con lo descrito por Leber (1982), Degraer et al. (1999) y Martí et al. (2007). Sería nuevamente el argumento de la mortalidad por explotación pesquera el que explicaría una tendencia menos evidente en el caso de las especies sometidas a marisqueo, ya que la pesquería de *D. trunculus* y *C. gallina* también mostró una estacionalidad (Capítulo 3), con incrementos en primavera y verano del esfuerzo pesquero, capturas y CPUE. El motivo por el cual el conjunto de bivalvos mostró diferencias significativas entre épocas del año a diferencia de *D. trunculus* y *C. gallina,* se debe a que la clase bivalvos incluye, además de las dos especies comerciales, especies como *Donax semistriatus*, *Mactra stultorum*, *Loripes orbiculatus*, entre otras. Esto muestra que los bivalvos que están siendo objeto de explotación pesquera no siguen una pauta tan marcada como el conjunto de bivalvos. Por tanto, el aprovechamiento pesquero estaría modificando la variabilidad temporal natural de estas especies.

Se ha podido observar que las variaciones espaciales y temporales que presentaron las especies de interés pesquero no fueron las mismas que mostraron la fauna bentónica en general. Las poblaciones de *D. trunculus* y *C. gallina* se vieron influenciadas principalmente por la extracción pesquera, mientras que el resto de los organismos presentaron pautas de distribución y abundancia relacionadas con los factores ambientales descritos en la bibliografía como habituales. Durante los estudios llevados a cabo a meso y macroescala no se apreció la evidencia de otros factores no estudiados, tales como el parasitismo, la depredación, etc., que pudiesen afectar a las densidades de *D. trunculus* y *C. gallina* en tal magnitud como para acarrear el descenso poblacional que condujo al cierre de la pesquería. Por tanto, la presión pesquera que sufrieron estas poblaciones, debido a una mala gestión, pudo ser la que condujo a la sobreexplotación de la pesquería y su consecuente colapso. La gestión de la pesquería se llevó a cabo mediante una serie de medidas que no fueron efectivas para asegurar la sostenibilidad de los bancos explotables (Capítulos 3 y 4). Las máximas capturas diarias establecidas por la legislación, además de estar obsoletas al ser fijadas en base al monitoreo y estudio de la actividad pesquera a finales de la década de los 80 y principios de los 90, nunca limitaron las capturas de ambas especies, ya que rara vez se alcanzaron. Otra medida de gestión que se consideró poco adecuada fue la talla mínima de captura establecida para *D. trunculus*. Esta talla mínima de captura (14 mm) es muy baja y cercana a la talla de primera maduración de esta especie (12.7 mm) establecida por Ramón (1993), lo que pudo afectar a su potencial reproductor. Por otra parte, también cabe remarcar la falta de un período de veda durante el período reproductivo y de reclutamiento que asegure la supervivencia de las especies en el golfo de Valencia, además de la falta del establecimiento de un plan de gestión que lleve a cabo un seguimiento de las poblaciones naturales. Por otro lado, cabe remarcar que la selectividad del arte de pesca fue considerada adecuada tal y como se observó en el capítulo 4.

6.3 FACTORES QUE AFECTARON A LA RECUPERACIÓN DE LA PESQUERÍA

La pesquería de *D. trunculus* y *C. gallina* fue cerrada en junio del 2015 con el objetivo de permitir la recuperación de los contingentes poblacionales de ambas especies. Podemos considerar que la explotación de *C. gallina* era casi testimonial antes del cierre, debido a que sus capturas y el esfuerzo pesquero aplicado en nuestra zona de estudio fueron mínimas en comparación con las de *D. trunculus*, ya que, al tratarse del mismo *métier*, pero con rangos batimétricos de faenado diferentes, no se explotaba simultáneamente las dos especies. Por tanto, la pesquería de *C. gallina* podría asumirse que estuvo colapsada desde antes del cierre y no ha mostrado ningún signo de recuperación.

Las poblaciones de *D. trunculus,* posteriormente al cierre de la pesquería, presentaron un incremento inmediato, tal y como se comentó, debido a la estacionalidad que presentan las mismas, mientras que *C. gallina* no presentó ninguna variación de abundancia. Dos años después, en el 2017, no se observó una recuperación de las poblaciones, e incluso decayeron. Según Mackie (1984), los ciclos reproductivos se ven afectados por la interacción de factores genéticos y ambientales, lo que resulta en actividad reproductiva, y pueden clasificarse como: estrategas de la K, con períodos reproductivos cortos y limitados, y estrategas de la r, con actividad reproductiva que ocurre durante la mayor parte del año (Fretter, 1984). Los organismos del género *Donax* son organismos con ciclos de vida relativamente cortos, periodos de reproducción largos, generan grandes cantidades de larvas planctotróficas y tienen elevadas tasas de mortalidad en las fases iniciales de su ciclo vital por lo que encuadrarían en los estrategas de la r. Sin embargo, las estrategias de la r y de la K representan extremos opuestos de un espectro continuo que va desde entornos cambiantes inestables o impredecibles dominados por oportunistas a ambientes estables o relativamente constantes dominados por especies de equilibrio. La mayoría de las especies se ubicarán en algún lugar entre estos dos extremos. Ningún organismo se define por completo r o K, sino que todos deben llegar a un compromiso entre los dos extremos. En regiones templadas, Herrmann et al. (2009) revelaron que *Donax hanleyanus* en Argentina muestra dos ciclos gametogénicos anuales. En Túnez (Tlili et al., 2011) y en Portugal (Gaspar et al., 1999) *D. trunculus* tiene un ciclo reproductivo anual con una única temporada de desove. Sin embargo, en el golfo de Valencia, Ramon (1993) señaló que *D. trunculus* tiene dos eventos de desove anuales y *C. gallina* un período único que se extiende de mayo a agosto. Por todo ello, cabría esperar que las poblaciones de *D. trunculus* y *C. gallina* hubiesen presentado una rápida recuperación una vez cerrada la pesquería. La falta de recuperación de la población de ambos bivalvos dos años después del cierre, indica que pueden existir otros factores ambientales que podrían estar afectando a la densidad y la supervivencia de estas especies en el golfo de Valencia. Por ejemplo, Gadelha et al. (2019) reveló que la clorofila *a*, las precipitaciones y la materia orgánica intensificaron la frecuencia de liberación de gametos en *Donax striatus*, probablemente debido al aumento en la disponibilidad de alimentos.

Se ha podido observar que la granulometría es un factor importante que puede condicionar las poblaciones de macrofauna bentónica y en particular de las especies de interés pesquero estudiadas. En el golfo de Valencia, el transporte de sedimentos a lo largo de la costa se ve interrumpido por las diferentes obras de ingeniería costera como son los puertos. Esto genera una acumulación de arenas al norte de dichas infraestructuras y una erosión en las zonas ubicadas al sur de éstas, siendo el aporte artificial de arenas una técnica habitual para mantener las playas que han sido erosionadas (Pardo-Pascual y Sanjaume, 2019). El bivalvo *D. trunculus* es susceptible a las variaciones antrópicas y naturales de la granulometría, por lo que el tamaño de grano del sedimento va a determinar su distribución y abundancia (Guillou y Bayed, 1991; Mazé y Laborda, 1998; Lart et al., 2003; La Valle et al., 2011), pudiendo tener una gran influencia en la eficiencia ecológica de los individuos (tasa de enterramiento, crecimiento y metabolismo) tal como constataron De la Huz et al. (2002). Esto pone de manifiesto la importancia de mantener una granulometría adecuada para las poblaciones de macrofauna bentónica, siendo importante la selección del sedimento adecuado en aquellas zonas donde se producen regeneraciones de playas (La Valle et al., 2011). No obstante, el medio no muestra evidencias claras de cambios granulométricos, presentando todavía un grano adecuado para la distribución y abundancia de ambas especies.

Los factores nutricionales de la columna de agua fueron analizados en detalle durante los dos años previos al cierre de la pesquería. A partir de 2015, no ha existido ningún hecho que haga pensar que se ha modificado alguna de las pautas observadas durante el periodo 2013-2014. Ahora bien, cuando se analizan cambios que pueden ser sutiles a corto o a medio plazo y por lo tanto irrelevantes, desde una perspectiva a largo plazo una tendencia puede cobrar importancia. Como se comentó en el apartado anterior, los aportes de agua dulce a un mar oligotrófico como el Mediterráneo pueden ser determinantes para la productividad marina (Ludwig et al., 2009). La carencia de algún nutriente, fundamentalmente del que actúa como nutriente limitante, afectará al desarrollo del fitoplancton (Olivos et al., 2002). En general en el Mediterráneo se considera que el fósforo es el principal nutriente limitante (Krom et al., 2004; Siokou-Frangou et al., 2010). Ludwig et al. (2009) indicaron que el Mediterráneo ha sufrido una significativa reducción de las descargas de agua dulce (como mínimo un 20 % entre 1960 y 2000). Esto es en parte causado por sequías hidrológicas más severas, pero también por la construcción de embalses y estaciones hidroeléctricas así como por la desviación de caudales para riego (García-Ruiz et al., 2011). Además, en las últimas décadas, se ha observado una disminución generalizada de fósforo en los ríos que terminan en este mar, a causa del desarrollo del tratamiento de las aguas residuales y de la utilización de detergentes sin fosfato (Ibáñez y Peñuelas, 2019). Por lo tanto, el río Serpis, principal fuente de agua dulce al mar en la zona de estudio, podría haber sufrido además de una disminución del caudal que llega al mar, una disminución de la concentración de fósforo en las últimas décadas, presentando por consiguiente un gradual descenso de su flujo hacia el mar. Autores como Romanelli et al. (2009) observaron en la costa de Italia, en el Adriático, que el descenso que había sufrido *C. gallina* se podría deber a la disminución de las descargas de agua dulce al mar, así como al bajo contenido en fósforo que afectarían al desarrollo del fitoplancton y como consecuencia a las poblaciones de *C. gallina.*

Si bien una posible disminución de las entradas de fósforo al medio marino podría estar y haber estado afectando gradualmente a la biomasa de las poblaciones fitoplanctónicas en nuestra zona de estudio, también habría que considerar la calidad de las mismas. Las diatomeas, que predominan en las zonas más turbulentas del sublitoral (McLachlan y Defeo, 2018) donde habita *D. trunculus*, requieren de silicio además de nitrógeno y fósforo. Cuando el silicio es tenido en cuenta como un posible nutriente limitante para el crecimiento de fitoplancton, la limitación por este nutriente se vuelve un fenómeno más extendido en el Mediterráneo (Ludwig et al., 2009). Gadea et al. (2013) notaron que la comunidad fitoplanctónica del sur del golfo de Valencia estaba dominada por diatomeas en invierno y que el silicio podría actuar como nutriente limitante en el 30 % de los casos analizados en ese periodo. En los mares, a nivel global, las principales fuentes de silicio proceden de los sistemas fluviales y de las descargas de aguas subterráneas (Frings et al., 2016). Los flujos de silicio procedentes de los ríos que llegan al Mediterráneo habrían descendido a causa de la disminución de las descargas de agua dulce (Ludwig et al., 2009), pero también debido a la menor concentración de silicio que llega a la desembocadura tras quedar depositado en las presas (Humborg et al., 1997; Conley et al., 2008). Las descargas de aguas subterráneas con sus nutrientes asociados pueden representar una importante fuente de silicio para el medio marino (Niencheski et al., 2007). En la zona de estudio, Sospedra et al. (2018) establecieron que más de la mitad del silicio presente en la zona costera procede del aporte de las aguas subterráneas jugando un papel relevante los humedales de la zona debido a los cambios en los usos del suelo (Sebastiá-Frasquet et al., 2014).

Además, en zonas como la costa mediterránea de Egipto, la costa atlántica de Marruecos y Francia, y la costa atlántica de España, se observó que la parasitación de *D. trunculus* por el trematodo *Bacciger bacciger* podía causarle una castración severa y por tanto problemas en la supervivencia de las poblaciones (Ramadan y Ahmad, 2010; de Montaudouin et al., 2014; Delgado y Silva, 2018). Este mismo trematodo también fue encontrado para *D. trunculus* en el golfo de Valencia, generándole una disminución en la capacidad de reproducción, aunque sin afectar significativamente a las poblaciones (Ramón et al., 1999). No obstante, este último estudio fue llevado a cabo durante los años 1988 y 1990, por lo que en la actualidad esta situación podría haber empeorado, siendo un aspecto a tener en cuenta para su estudio. Otros parásitos, *Mikrocytos veneroïdes* y *Mikrocytos donaxi,* también fueron encontrados en *D. trunculus* en la costa atlántica de Francia (Garcia et al., 2018). En dicho estudio los autores vincularon un episodio elevado de mortalidad del bivalvo durante los años 2008-2011 con la aparición de ambos parásitos. Para *C. gallina*, durante los años 1997-1999 en Italia, mar Adriático, también fueron observados casos de mortalidades anómalas en los bancos naturales, en los que posiblemente tuvieron un papel importante las infecciones graves detectadas por protozoos del género *Nematopsis*, que pueden causar una insuficiencia respiratoria (Berrilli et al., 2000).

Otros aspectos a considerar como posible causa de la falta de recuperación de las poblaciones de *D. trunculus* y *C. gallina* podría ser la depredación tanto en fase bentónica por gasterópodos como *Neverita josephinia*, cangrejos, pájaros y peces, entre otros (Schneider, 1982; Neuberger-Cywiak et al., 1990; McLachlan y Defeo, 2018), como en la fase planctónica. Salas et al. (2001), en un estudio desarrollado en el sur de España (Málaga), determinó que la mordedura de cangrejos producida en el pie de donácidos podría ser subletal, debido a que los individuos afectados son más susceptibles a ser devorados por otros organismos. También cabe destacar la fuerte repercusión que tienen las especies invasoras sobre las comunidades bentónicas. Desde el 2014-2015 se han detectado numerosos ejemplares de *Callinectes sapidus* (cangrejo azul) en el golfo de Valencia. Aunque *C. sapidus* se alimenta de una gran variedad de organismos, los bivalvos son la principal fuente de alimentación de este crustáceo, pudiendo llegar a alcanzar en algunas zonas hasta el 50 % de su dieta (Seitz et al., 2003). Es por ello por lo que las poblaciones de *D. trunculus* y *C. gallina* podrían verse afectadas por la presencia cada vez más abundante de este decápodo.

Además de la mortalidad por depredación sobre los adultos, el consumo de las larvas en su fase planctónica afectaría al reclutamiento de *D. trunculus* y *C. gallina.* Los escifozoos pueden ejercer un fuerte control "top-down" en la comunidad planctónica (Purcell, 1992; Mills, 1995; Pérez-Ruzafa et al., 2002). En este sentido, Larson (1991) concluye que *Stomolophus meleagris* selecciona larvas veliger de bivalvos frente a otros taxones de presas en el golfo de México. En el caso de las playas estudiadas, las especies de escifozoos que han presentado blooms entre los años 2010 y 2017 han sido *Rhizostoma pulmo* y *Cotylorhiza tuberculata* que tienen un consumo relevante de larvas veliger de bivalvos en el Mar Menor (Pérez Ruzafa et al., 2002). También, otros componentes del plancton pueden generar mortalidad sobre los estadios larvarios de los bivalvos. Se encontraron tasas de mortalidad elevadas de veliger generadas por el dinoflagelado heterotrófico *Noctiluca scintillans* (Johnson y Shanks, 2003) y por larvas de poliquetos (Johnson y Brick, 1998). Sin embargo, es necesario plantear que también los filtradores bentónicos, incluso los propios bivalvos, pueden estar depredando sobre las larvas veliger (Troost et al., 2008).

Por otra parte, cabría considerar el efecto de la acidificación del mar sobre las poblaciones de bivalvos estudiadas. Flecha et al. (2015), mediante un monitoreo en el Estrecho de Gibraltar entre los años 2012-2015, detectaron una acidificación con una tendencia de -0.0044 ± 0.00006 anual. Pereira et al. (2016) determinaron mediante experimentos ex situ que reducciones de entre 0.3 y 0.6 de pH afectan a la supervivencia y desarrollo de las larvas de *D. trunculus*, produciéndose un retraso en la eclosión y un aumento de la mortalidad y deformaciones en las larvas. Range et al. (2014), determinaron la respuesta de juveniles de *C. gallina* ante la exposición a largo plazo a agua de mar acidificada con CO_2 usando un sistema experimental al aire libre, observando que descensos de pH de 0.7 causaron daños severos en la concha de *C. gallina* e incrementos de mortalidad. Por tanto, dado que son necesarias reducciones de al menos 0.3 en el pH para producir afecciones sobre estos bivalvos, y en base a la baja tasa de descenso del pH observada por Flecha et al. (2015) en el Estrecho de Gibraltar, no parece ser que la acidificación haya sido la responsable de la afección ni a *D. trunculus* ni a *C. gallina* en nuestra zona de estudio.

El incremento de la presión turística ejercida sobre las playas arenosas también puede ser un aspecto clave que afecte a las poblaciones principalmente de *D. trunculus* por encontrarse en un rango batimétrico coincidente con la zona de baño. El turismo costero se ha ido incrementando gracias a una mejora en el transporte, siendo el Mediterráneo el lugar donde se concentran el mayor número de turistas a nivel mundial (Davenport y Davenport, 2006; Almeida et al., 2008). Por ejemplo, en el año 2018 el número de turistas extranjeros que visitaron la Comunidad Valenciana duplicó a los del 2001 (de 4.4 millones a 9.2 millones) (INE, 2019). Las playas arenosas se ven fuertemente afectadas por las actividades recreativas, el turismo y el pisoteo (McLachlan y Defeo, 2018). En un estudio llevado a cabo por Reyes-Martínez et al. (2015a) en dos playas con diferente presión antrópica (área protegida y área urbanizada) en la Bahía de Cádiz, observaron que pese a presentar las mismas características físicas, climáticas y morfodinámicas, presentaban diferencias en las comunidades de macrofauna bentónica. El área protegida presentaba una mayor biomasa, densidad y riqueza, un sistema más complejo y maduro que la playa urbanizada, especialmente en verano, época que presenta mayor presión turística. Reyes-Martínez et al. (2015b) demostraron que el pisoteo es un factor que altera de forma importante la macrofauna bentónica de playas arenosas al disminuir sus densidades, resultando en una pérdida a largo plazo de la diversidad en zonas urbanizadas. En un estudio realizado por Moffet et al. (1998) en Sudáfrica, se determinó la influencia de diferentes intensidades de pisadas sobre la macrofauna de playas arenosas, observándose un daño sustancial cuando el pisoteo fue intenso, viéndose afectada *Donax serra*, aunque ligeramente, en todas las intensidades de pisoteo estudiadas, desde la más leve a la más intensa. Las especies del género *Donax*, pese a su capacidad para moverse, presentan menores densidades en zonas urbanizadas que en las zonas protegidas, relacionado con actividades recreativas y el pisoteo (Veloso et al., 2006).

Finalmente, se puede considerar que la contaminación también podría condicionar la presencia y abundancia de las especies objetivo. Neuberger-Cywiak et al. (2003) determinaron que la presencia de metales pesados como cadmio y zinc afectan al comportamiento en el enterramiento de *D. trunculus* pudiendo provocarle una muerte indirecta, debido a la importancia que tiene la capacidad de enterramiento para la supervivencia de esta especie en la zona que habita. La concentración de metales pesados en *D. trunculus* fue analizada en diferentes puntos de la Comunidad Valenciana por Martínez-Guijarro et al. (2015), presentando nuestra zona de estudio generalmente los valores más bajos. Además, Tlili et al. (2010) estudiaron el efecto de la contaminación por descargas de agua no depuradas, desechos industriales, policlorobifenilos…. en *D. trunculus*, determinando que el tamaño medio de longitud y la tasa de crecimiento de las cohortes se reducen considerablemente en sitios contaminados. Asimismo, se encontraron diferencias en la reproducción e índices biológicos en esta especie entre zonas contaminadas y no contaminadas (Tlili et al., 2011). Debido a que nuestra zona de estudio presentó los valores más bajos de concentración de metales pesados en *D. trunculus*, no se podría considerar la contaminación por metales un factor determinante, al igual que las descargas de aguas no depuradas, ya que nuestra zona no se producen este tipo de vertidos. *D. trunculus* es considerada una buena especie bioindicadora para realizar seguimientos de playas arenosas en diferentes partes del Mediterráneo, incluida la zona de estudio (Neuberger-Cywiak et al., 2003; Neuberger-Cywiak et al., 2007; Tlili et al., 2010; Tlili et al., 2011; Tlili et al., 2013; Martínez-Guijarro et al., 2015; Tlili et al., 2019).

Por tanto, los bancos explotables de *D. trunculus* y *C. gallina* no presentaron recuperación pese a la elevada capacidad reproductiva de ambas especies. Esto podría deberse a numerosos factores relacionados con los cambios en los aportes de agua dulce al mar, la depredación tanto en fase bentónica como planctónica, el parasitismo, así como el incremento de la presión turística. Por el contrario, los cambios granulométricos, la acidificación y la contaminación no parecen haber sido relevantes para considerarlos como posibles causas de afección. Es probable que la intensidad de la afección de cualquiera de los factores considerados en este punto hubiera sido mucho menor si se hubiese producido sobre poblaciones de *D. trunculus* y *C. gallina* en buen estado. Debido al estado de sobreexplotación que se alcanzó en este *métier* de pesca, posiblemente las poblaciones de estos bivalvos fueron más susceptibles al efecto adverso de estos factores.

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Capítulo 7

Conclusiones generales

7. CONCLUSIONES GENERALES

En este apartado se presentan las principales conclusiones obtenidas a partir de la discusión de los diferentes capítulos que componen la memoria de la presente tesis.

Respecto a la pesquería

- La pesquería de los bivalvos *Donax trunculus* y *Chamelea gallina* en las zonas de trabajo de la Cofradía de Pescadores de Gandia, que abarca desde Tavernes de la Valldigna a Denia, sufrió un declive que condujo a su cierre en junio del 2015. Las capturas *D. trunculus,* en el año previo al cierre, llegaron a representar tan solo un 1.6 % de los 100 000 kg capturados en el 2004 y para *C. gallina* fueron de un 3.8 % con respecto a los 21 000 kg capturados siete años antes.
- *D. trunculus* y *C. gallina* forman parte del mismo *métier*, donde se utilizan similares artes de pesca, áreas de explotación y período de pesca, en el que *D. trunculus* presentó una clara dominancia con respecto a *C. gallina.*
- Este *métier* presentó una clara estacionalidad, concentrándose principalmente durante los meses de primavera y verano, vinculada a aspectos climáticos y económicos así como a características biológicas de la especie.
- Los descensos de CPUE, que fueron muy marcados a partir del 2009 para *C. gallina* y del 2010 para *D. trunculus*, indicaron una sobreexplotación de la pesquería, alcanzando el colapso. Esto se debió a una serie de medidas de manejo que no fueron efectivas para asegurar la sostenibilidad de los bancos explotables.
- Las cuotas de capturas máximas diarias por embarcación fijadas por la legislación, que incluía a las dos especies del *métier*, fueron demasiado altas para limitar la extracción de ambas especies. Hasta la primera mitad de 2013, la cuota máxima de 65 kg fijada por la legislación en vigor solo fue superada en 1 ocasión, estando el 95 % de las capturas diarias por embarcación de toda la flota por debajo de los 35 kg. Posteriormente, hubo un cambio en la legislación que

modificó la cuota máxima diaria por embarcación a 40 kg, si bien el 97 % de las capturas de toda la flota estuvieron por debajo de los 25 kg.

- El área de pesca de la Cofradía de Pescadores de Gandia sufrió una presión pesquera desigual en las distintas zonas fijadas en la Resolución de la Comunidad Valenciana del 2013. Las zonas situadas al sur del principal aporte de agua dulce, río Serpis y drenaje del marjal de La Safor, presentaron mayores esfuerzos pesqueros, capturas totales, mayores capturas por km de costa aunque con valores promedio de CPUE similares al del resto de las zonas. La cercanía al puerto de Gandia y la mayor productividad de estas zonas fueron las causas de esta distribución.
- La selectividad del arte de pesca empleada para *D. trunculus* fue considerada adecuada en base a la talla mínima de captura de 14 mm establecida por la legislación. Sin embargo, para garantizar la eficiencia en la reproducción de *D. trunculus*, la talla mínima establecida debería ser revisada e incrementada a 25 mm, y por tanto adaptar las artes de pesca para una selectividad acorde a esta nueva talla.
- El declive de la pesquería generó el abandono total de la actividad pesquera del 40 % de las embarcaciones que se dedicaban a este *métier* ya que, por su pequeño tamaño, no se pudieron adaptar a otras pesquerías, generando por tanto claras repercusiones socioeconómicas.
- Para garantizar la sostenibilidad de los bancos explotables se propone el establecimiento de una gestión pormenorizada de cada una de las zonas en función de las densidades poblacionales, como biomasa extraíble en cada zona, periodos de pesca, cupo de capturas, número de embarcaciones máximas y asignación de zonas de pesca. Además, pueden plantearse cierres temporales durante el período de reproducción y reclutamiento (primavera-verano) o bien, el establecimiento de zonas totalmente vedadas para la pesca.

Respecto a los aspectos biológicos y ecológicos de *D. trunculus*

- Las poblaciones de *D. trunculus* en la zona de estudio mostraron una distribución diferenciada por tallas en profundidad, con los organismos más pequeños a menor profundidad y los más grandes en zonas más profundas, además de un período de reclutamiento en verano. A pesar de que el índice estándar de crecimiento (ϕ') de *D. trunculus* no mostró diferencias con otros estudios de otras zonas templadas, la longitud asintótica (L_∞) y la constante de crecimiento (K) en 2015 mostraron valores más bajos que los obtenidos 25 años antes en la misma región.

Respecto a la estructura y funcionamiento de la comunidad del sublitoral de playas arenosas

- La fauna bentónica mostró una clara zonación batimétrica definida fundamentalmente por el tamaño medio del sedimento y la cantidad de materia orgánica. Se encontró una comunidad dominada por un mayor tamaño de grano medio y menor contenido en materia orgánica, en la zona más somera y turbulenta, en la que dominó en biomasa el bivalvo *D. trunculus* y en la que podemos destacar otros organismos como *Urothoe grimaldii*, *Scolelepis squamata*, *Gastrosaccus sp.*, *Schistomysis assimilis* y *Pontocrates arenarius.* Y otra comunidad sujeta al mayor contenido de materia orgánica en el sedimento y un tamaño de grano más fino, presente a mayor profundidad, en la que los fondos son más estables, predominando principalmente los poliquetos *Magelona mirabilis*, *Prionospio pygmaea*, *Paradoneis armata* y *Nephtys hombergii.* Entre estas dos comunidades encontramos una zona intermedia, zona de transición en la que podemos encontrar una mezcla de especies de ambas comunidades. Esta zonación está por tanto regida por el hidrodinamismo que presentan las playas.
- A mesoescala, la influencia de la cercanía a los aportes de agua dulce sobre la fauna solo fue evidente en las zonas más profundas, donde el movimiento del agua es menor. En las zonas más cercanas al río Serpis a 4 m de profundidad, encontramos sedimentos más finos y mayor acumulación de materia orgánica, lo que se traduce en mayores abundancias de poliquetos depositívoros.

- La disponibilidad y tipo de alimento determinó la dinámica poblacional de la macrofauna bentónica, observándose un acople entre la parte pelágica y la bentónica. La mayor biomasa de fitoplancton hacia finales de invierno y principios de primavera se tradujo en un incremento de la abundancia de zooplancton, incluido meroplancton, que ajusta su ciclo vital a la mayor disponibilidad de alimento. Posteriormente, en verano, la muerte y sedimentación tanto de productores primarios como de algunos secundarios, generó una mayor cantidad de detritos, lo que pudo favorecer la presencia de muchas especies depositívoras, principalmente poliquetos. Pero, además, un incremento de abundancia en verano también se debió al reclutamiento del meroplancton que estaba presente a finales de invierno y principios de primavera.

- La tendencia temporal de las especies explotadas no fue tan clara como la que mostró el resto de fauna bentónica, como consecuencia de la mortalidad que ocasiona la pesca a dichas especies. Además, no presentó una pauta con respecto a la distancia al aporte de agua dulce a nivel de mesoescala.

Respecto a la recuperación de las poblaciones

- El incremento de densidades y biomasa inmediatamente después del cierre de la pesquería en 2015, donde se llegó a observar incluso un período de reclutamiento (entre agosto y septiembre), se debió principalmente a la estacionalidad que presentan las poblaciones, no pudiéndose vincular exclusivamente al cierre. Los valores de densidad y biomasa obtenidos dos años después, en 2017, no solo mostraron que no había habido una recuperación de la población, sino también que la situación había empeorado. Por lo tanto, el cierre de la pesquería no tuvo el efecto deseado a corto plazo.

- La sobreexplotación que sufrieron los bancos de *D. trunculus* y *C. gallina* podría haber agravado los efectos adversos provocados por factores tales como los cambios en los aportes de agua al mar, la depredación tanto en fase bentónica como planctónica, el parasitismo, así como el incremento de la presión turística, lo que podría haber dificultado la recuperación de las poblaciones.