RESILIENCE OF AQUATIC METACOMMUNITIES: IMPLICATIONS FOR DISTURBANCE RECOVERY

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Doctoral thesis:

Resilience of aquatic metacommunities

Implications for disturbance recovery

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Resilience of aquatic metacommunities

Implications for disturbance impacts and recovery

David Cunillera Montcusí

2020

Doctoral Programme in Water Science and Technology

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Thesis submitted in fulfilment of the requirements to obtain the Degree of Doctor at Universitat de Girona

The following thesis contains five annexes containing the supplementary materials corresponding to several chapter of the work

Hereby, Dr. Stephanie Gascón and Dr. Dani Boix from Universitat de Girona,

CERTIFY:

That the doctoral thesis entitled *Resilience of aquatic metacommunities: Implications for disturbance impacts and recovery* submitted by David Cunillera Montcusí to obtain the degree of Doctor at Universitat de Girona has been completed under our supervision and meets the requirements to opt for the *International Doctor* mention

In witness whereof and for such purposes as may arise, the following certification is signed:

Dr. Stephanie Gascón
Universitat de Girona

Dr. Dani Boix
Universitat de Girona

Girona, 31/10/2019

ALS PETITS DE CASA.

Que el present incendiari

on us ha tocat viure,

sigui l’adob d’un

futur ple de

LLIBERTATS.

Ho intentaré, ho intentarem.

UNA HISTÒRIA DE MIL MERDERS,

UNA HISTÒRIA DE MIL GRÀCIES!

Dia de Nadal, ben d'hora al matí. Mai m'hauria pensat que seria tan difícil agrair-vos tot el què heu fet perquè avui jo estigui aquí, al costat d'una estufa de butà, rumiant com dir gràcies a tothom qui d'una manera o altra, ha format part d'aquesta història. Però clar... on comença tot això? O encara més important, quin nivell de contribució mereix una menció en aquests folis? I aquestes *contribucions mereixedores de citació* (a partir d'ara CMC) han d'incloure les passives (CMC1)? O només les dèbilment actives (CMC2), les moderadament actives (CMC3) i les fortament actives (CMC4)? I clar... ara que hi penso, aquestes contribucions dependran de cada persona específicament i la seva percepció de la meva tesi. Per tant, potser s'hauria de fer algun model que representés un patró general per diferents CMC i d'allà saber qui mereix ser citat, qui no (ja em sap greu... no és res personal, però ho diu un algoritme i estem al segle XXI, o sigui que no si pot fer res) i en quin ordre segons rellevància (Ep! Al lloro, considerant totes les CMC juntes eh! No sigui cas que ens ho rebutgin...). Com ja deveu estar sospitant, hi ha una pregunta que encara no he fet i que crec que amb el que fins ara heu llegit, que em temo que només entendran alguns escollits més del tipus CMC4, ja s'hauria de començar a intuir: fins on arribarà aquest noi/sonat per dir unes simples gràcies? Quantes pàgines ocuparà? M'haurà dit gràcies a mi al final?

Doncs bé, més enllà de quin CMC sou, si esteu llegint això i heu arribat fins aquí, vol dir que alguna cosa teniu a veure amb mi i amb aquesta història que com gairebé tot va més enllà de la tesi. Un savi em va dir que a la natura mai pots dir un "tot" perquè sempre hi haurà quelcom que en sigui l'excepció. En conseqüència, ja us dono les meves més sinceres **GRÀCIES** per començar i així, si teniu pressa, podeu deixar-ho aquí que tampoc anirem gaire més lluny... però som-hi:

Tot comença en un poblet de l'Alt Camp (Ull! Primera comarca en ordre alfabètic! No totes ho poden dir això!) on neixo jo (bé, de fet neixo a l'hospital però ja s'entén, a més, un hospital no queda tan poètic). La qüestió és que neixo i allà comença tot, no ens posarem a remenar reencarnacions passades que llavors la cosa es complica, perdem el fil i buf... massa feina. La cosa és que tot comença, no me'n recordo massa però en general diuen que bé, que va anar bé. Aquí mereixen una menció especial la mama, Rita, i el papa, Pep, que sobretot em gestionen els temes i tota la paperassa d'aquells temps. No crec que als pares se'ls pugui agrair mai prou el què han fet passivament, activament o com sigui, però en aquest cas, ells són l'espurna que engega la flama d'aquest foc, que avui crema amb força i calidesa. A ells, **GRÀCIES**. Tanmateix, quan vaig arribar a casa, no estava sol... ja n'hi havia dues... però per sort, enlloc d'enfrontar-nos en una lluita eterna i titànica de violència fraternal, ens vam ajudar i sobretot, elles dues, van oxigenar la flama que al seu costat va créixer amb més fortalesa i empenta. A elles, **GRÀCIES**. Més enllà d'aquesta casa, però, hi ha la família (so de mandolines), la sang, els portadors dels cognoms. Els padrins, els tiets, els cosins, (sonen encara més mandolines) també una nova família més recent però més intensa (tenen més nivell i em donen embotits) i, per últim i per doble importància els padrins altre cop han fet que la flama tingués bon fons, que no fos un simple foc follet. Per tot això i en el fons molt i molt més, **GRÀCIES**.

Tot va fent al llarg dels anys, molts d'aquests ja us he dit que els tinc poc clars, però ja m'ho gestionaven des de fora i la cosa doncs anava fent prou bé. Al llarg d'aquests temps vaig anar coneixen gent, molt bona gent que encara avui puc considerar amics. Amics passats, presents, futurs i atemporals o com a sinònim mandanguers (el word no m'accepta "atemporal" ni "mandanguer" i a l'optimot no li agraden massa tampoc... però aquest és el meu llibre i la definirem com: *quelcom que no inclou de forma explícita la dimensió temporal*). Tots ells formen part d'una amalgama molt curiosa de personatges amb els quals he compartit coses molt esperpèntiques (coses en el sentit ampli de la paraula però que poden ser tant negatives com positives com cap de les dues, com fastigoses, humiliants, denigrants, delirants, hilarants i mil adjectius més, ara tampoc em posaré a buscar... no cal fer-me el Josep Pla... que al final tot ho busco a l'optimot). El més fort del cas, és que després de tot el viscut encara esperen un tracte respectable quan ara es fan els seriosos tenint feines importants, fills encantadors amb noms que empoderen (Ull! Rei i profeta! Tampoc tothom ho pot dir això!) i coses per l'estil. Com ens hem de veure. Sigui com sigui, a aquests personatges esperpèntics se'ls deu més del que sembla, perquè són família i perquè al final han estat, són i seran llenya que alimenta la flama i la fa esvelta i duradora. A tots ells, passats, presents, futurs i atemporals mil **GRÀCIES**.

Tot acaba on tot comença i un bon inici és Girona. La història segueix a Girona on el canvi d'aires aporta nous personatges (sí... també d'aquests esperpèntics però sense saber fer vocals obertes... perdoneu però algú ho havia de dir). La cosa és que aquí la paperassa ja la portava jo i clar, tot es va complicant i al final no saps dir que no i t'emmerdes. I per merder aquest que us estic presentant ara mateix. De fet, podem seguir ben bé els inicis del merder remuntant-nos al febrer de 2013, ironies de la vida aquest merder el defenso un febrer de 2020, on un David del passat deia així (m'he pres el luxe de corregir les faltes ,subratllades, que aquest energumen del passat feia en el seu correu a un professor que li havia de dirigir el TFG... molt bé David (aplaudiments irònics) gran presentació... mira que n'ets de sabatot...):

Bon dia Dani,
sóc (tens sort que encara hi havia diacrítics) *en David Cunillera, alumne de 3er de CCAA i tot i ser "d'hora" estic pensant en el treball de fi*nal *de grau i les pràctiques en empreses* (empresa) *i intentar combinar-ho.*
*No tinc massa clar qu*è *vull fer amb el treball de fi*nal *de grau, però m'interessaria fer-l'ho amb tu o bueno* (soc molt fan de escriure un pronom feble maco perquè sí i seguidament un "bueno" de nivell catedral) *que si m*é*s no em donguessis* (ai la gu...) *idees o possibles coses a treballar.*

PD: Aquest (any) *vaig demanar una beca de col·laboració per a fer-la amb l'Anna Romaní però al ser un alumne de tercer no me la van donar. L'any que ve la tornaré a demanar, m'agradaria preguntar-te si tu tens algún* (tu posa-hi un accent a veure si cola que saps escriure...) *projecte o cosa per tal de pude* (no cal dir res: poder) *realitzar-la amb tu. Com pots veure no s*é *massa bé cap on enfocar-me però no ho tinc molt clar.* No saps on enfocar-te ni escriure massa bé... però bé... no faré més comentaris.

El Dani de 2013 em va respondre entre altres coses: *Penso que per correu és difícil donar-te massa consells.* Ja ho pots ben dir... em sagnen els ulls de veure com l'escric! A mi! Que en soc l'autor! Sigui com sigui, aquí es pot dir que comença el merder que m'ha dut on soc. Cal destacar que allà també em vaig trobar una bona corrua de personatges esperpèntics que no em posaré a descriure (sí, per por a represàlies, que *hi ha molta maldat en este món*) amb els quals vaig haver d'interaccionar en laboratoris enformolats, taules dinàmiques infinites i scripts traïdors. Allà, enmig del merder, la flama no va fer res més que créixer, un punt desbocada cal dir-ho, però créixer. I entre entrades i sortides de despatxos aquest grup d'esperpèntics personatges no va fer res més que atiar el foc i ajudar-lo a créixer de forma dirigida més que controlada. Res hagués pogut ser sense ells i la flama no seria flama sense ells. Per això i tot el demés, **GRÀCIES**. Ara bé, òbviament, tot merder no seria merder sense el corresponent joc d'esperpèntics personatges igual o més emmerdats que un mateix amb els quals es comparteix espai, lluites.doc, reivindicacions.doc i finalment experiències seminarials (tampoc acceptat però prou explícit). A tota aquesta metacomunitat de personatges (localment aïllats en una tesi però connectats al compartir-ne les inquietuds i experiències), també **GRÀCIES**. Sé que a lo mejor no estaréis entendiendo nada de lo que he dicho hasta ahora, pero resumiéndolo todo un poco, muchas **GRACIAS** por acogerme (sí, lo he dicho) y hacerme uno más de esta patria chica que es la banda oriental, banda, que quedó muy pegada dentro de mí. ¡Un beso enorme, fuerza i vamos arriba! Que estamos de enhorabuena (no queda muy bien decirlo aquí, pero tenía que escribir *enhorabuena* en algún lugar).

Tot continua i continua millor al costat de gent important. He parlat de merders, personatges esperpèntics, *contribucions mereixedores de menció*, correus antics, faltes d'ortografia però hi ha coses, energies o forces que sobrepassen la lògica i coherència que tants anys m'ha costat construir. I aquesta força mística que em sobrepassa ets tu. No hi ha paràgrafs al món per intentar entendre el com, el perquè i així amb totes les preguntes que hauria de fer-me per comprendre-ho, però el fet és que és així. Simplement, és. **GRÀCIES** infinites per haver-hi estat, ser-hi i per seguir-hi sent, perquè hi ha coses atemporals, que per molt incongruents o impensables que siguin, són, i això per mi és màgic. Si la flama és càlida i dolça és perquè crema al costat de la teva. **GRÀCIES** cargolí per acompanyar-me en aquesta història ben emmerdada, plena de personatges esperpèntics i de *contribucions mereixedores de menció*. És graciós, perquè enlloc de cansar-me'n no penso res més que seguir-la vivint amb tu.

Vaig començar la tesi perquè volia saber més i endinsar-me en els dubtes del moment. Crec que avui, puc assegurar que l'acabo amb encara més dubtes i ben bé al fons d'un mar d'experiències. Que sigui el temps qui decideixi com les segueixo navegant i cap a on em condueix aquest gran merder. I si naufrago, tranquils, que la flama és forta i té fons per tirar, la brasa aguantarà amb força tots els embats. Però això, sapigueu-ho, és **GRÀCIES** a valtros, a tots valtros (no massa acceptat, però marca de la casa!).

Aquesta tesi s'ha dut a terme gràcies a un ajut predoctoral FPU del Ministerio de Cultura, Educación y Deporte (FPU014/06783) així com d'un ajut a la mobilitat lligat a aquest primer (EST16/00854).

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Les imatges dels invertebrats d'algunes figures han estat cedides per Jesús Ortiz així com altres imatges i símbols d'aquestes figures han estat descarregades de l' IAN Image Library (https://ian.umces.edu/imagelibrary/login.php?referer=displayimage.php%3Fpid%3D6320).

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All thesis drawings (covers and footers) are originals property and made by Rita Montcusí Rovira. I want to acknowledge her, for being able to create such beauty. Before using or reproducing them please ask us.
Invertebrate images used in some figures have been ceded by Jesús Ortiz. Small images and symbols from figures have been downloaded from IAN Image Library (https://ian.umces.edu/imagelibrary/login.php?referer=displayimage.php%3Fpid%3D6320)

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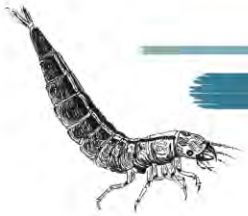

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LIST OF ABBREVIATIONS

- **%AC.FUL:** Fulvic acids percentage.
- **BACI:** Before-after control-impact design.
- **BS:** Body length.
- **CATS:** Community assembly by traits selection.
- **DAG1:** Passive aerial dispersers with aquatic adults.
- **DAG2:** Weak active dispersers with flying adults.
- **DAG3:** Moderate active dispersers with flying adults.
- **DAG4:** Strong aerial dispersers with flying adults.
- **DAG5:** Organisms with aquatic larvae and terrestrial dispersal adults.
- **DIC:** Dissolved inorganic carbon.
- **DIN/PT:** Ratio nutrient limitation indicator.
- **DIN:** Inorganic nitrogen.
- **DOC:** Dissolved organic carbon.
- **EC25:** Conductivity.
- **Fil Bi:** Filamentous algae biomass.
- **Fil:** Filamentous algae biomass.
- **Ful.Ac.%:** Fulvic acid percentage.
- **GAMM:** Generalized additive mixed models
- **GLMM:** Generalized linear mixed effect model.
- **HB:** High intensity burned ponds.
- **LB:** Low intensity burned ponds.
- **LHG1:** Dry-phase residents.
- **LHG2:** Dry-phase residents & spring recruits.
- **LHG3:** Dry-phase residents & summer recruits.
- **LHG4:** Non-dry-phase residents & spring migrants.
- **Mac Bi:** Macrophytes biomass.
- **Mac:** Macrophyte biomass.
- **Max Z:** Maximum pond depth.
- **MaxDepth:** Maximum pond depth.
- **MBACI:** Multiple before-after control-impact design.

- **ME:** Mass effects
- **nMDS:** Non-multidimensional scaling.
- **NP Ratio:** Ratio between DIN and SRP.
- **NT:** Neutral theory models
- **O_2:** Dissolved oxygen concentration.
- **PD:** Patch dynamics
- **PERMANOVA:** Permutational analysis of variance.
- **Phy:** Phytoplankton chlorophyll-a.
- **RDA:** Redundancy analysis
- **RMA:** Repeated measures analysis of variance.
- **SRP:** Soluble reactive phosphorus.
- **SS:** Species sorting
- **T1:** First iteration of model simulation
- **T100:** 100 iterations of the community recolonization process
- **TIC:** Total inorganic carbon.
- **TN:** Total nitrogen.
- **Tº:** Water temperature
- **TOC:** Total organic carbon.
- **TP:** Total phosphorus.
- **U:** Unburned ponds.
- **WTemp:** Water temperature.

LIST OF PUBLICATIONS

- **Cunillera-Montcusí, D.** (2016*). Burning waters: wildfire effects on aquatic systems.* In: H. Guasch (Ed.), Doctoral Meeting in Water science and technology (pp. 43–45). Girona: Universitat de Girona.

CHAPTER I

- **Cunillera-Montcusí, D.,** Gascón, S., Tornero, I., Sala, J., Àvila, N., Quintana, X. D., & Boix, D. (2019). D*irect and indirect impacts of wildfire on faunal communities of Mediterranean temporary ponds.* Freshwater Biology, 64(2), 323–334. doi: 10.1111/fwb.13219.

CHAPTER II

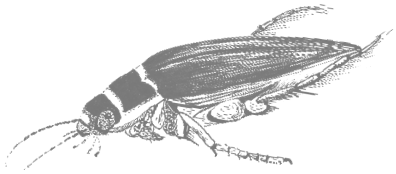
- **Cunillera-Montcusí, D.,** Arim, M., Gascón, S., Tornero, I., Sala, J., Boix, D., & Borthagaray, A.I. (2019). *Trends in traits selection in a temporary pond metacommunity in response to wildfire disturbance and seasonal succession.* Manuscript under review.

CHAPTER III

- **Cunillera-Montcusí, D.,** Boix, D., Sala, J., Compte, J., Tornero, I., Quintana, X. D., & Gascón, S. (2019). *The role of pond isolation, distance and direction in the colonization pattern of weak and strong dispersers.* Manuscript submitted.

CHAPTER IV

- **Cunillera-Montcusí, D.,** Borthagaray, A.I., Boix, D., Gascón, S., Sala, J., Tornero, I., Quintana, X. D., Arim, M. (2019). *Metacommunity network structure determines nonlinear transitions in biodiversity resilience to wildfire disturbances.* Manuscript submitted.



RESUM

Les pertorbacions són uns dels factors més determinants en l’estructuració de les comunitats, ja que poden tenir-hi un efecte al llarg de diferents nivells (p. ex. presència d’espècies, canvis ambientals o dels recursos disponibles). Així doncs, el rol que tenen les pertorbacions en condicionar-ne l’estructuració, esdevé clau a l’hora d’entendre quines dinàmiques intervenen i determinar quines espècies seran presents en uns hàbitats concrets. Tanmateix, no només les pertorbacions són rellevants per definir quines comunitats trobem als nostres ecosistemes. Altres característiques a nivell regional, com l’estructura del paisatge, hi juguen també un paper important. Per tant, si es vol comprendre millor com aquestes dues capes —els efectes d’una pertorbació i l’estructura del paisatge— influeixen a les comunitats a nivell ecològic, caldrà considerar les dues escales on aquestes tenen influència: escala local i escala regional. En conseqüència, s’haurà de tenir en compte tant les comunitats físicament delimitades (escala local) com l’intercanvi d’individus entre elles a través de la dispersió (escala regional), definint així el que es coneix com una metacomunitat. És per tot això, que si es vol aprofundir en quina és la influència sobre les comunitats de pertorbacions amb un marcat efecte tant regional com local (p. ex. incendis), esdevé necessària una perspectiva metacomunitària.

La integració d’una teoria lligada a les pertorbacions a nivell metacomunitari és molt incipient, però el seu desenvolupament permetrà la creació d’un marc teòric necessari per entendre i gestionar escenaris futurs (cada cop més catastròfics; p. ex. augment dels incendis, fragmentació dels hàbitats, pèrdua d’espècies, etc.). Conseqüentment, avui ens trobem amb la necessitat d’elaborar estudis més integradors (ja siguin observacionals, experimentals i/o basats en simulacions) que siguin capaços d’analitzar els efectes de grans pertorbacions (incendis) des d’una perspectiva metacomunitària (analitzant per exemple els seus efectes sobre les comunitats d’una xarxa d’estanys temporanis) per tal de poder comprendre millor la resposta de les comunitats a escala local i regional.

Els incendis es consideren pertorbacions importants a nivell global esdevenint grans moduladors de les comunitats i ecosistemes. Tot i això, el coneixement de les seves conseqüències s'ha focalitzat majoritàriament en sistemes terrestres, deixant de banda altres sistemes com els aquàtics. A més, dins d'aquests, els efectes dels incendis sobre els estanys temporanis han estat pràcticament ignorats. En el futur, s'espera que l'impacte d'aquestes pertorbacions augmenti en zones mediterrànies, sobretot pel que fa a la seva extensió i freqüència, on precisament els estanys temporanis són abundants i, des del punt de vista de la conservació de la biodiversitat, rellevants. En conseqüència, cada cop és més urgent entendre com els impactes dels incendis influeixen sobre les comunitats faunístiques d'aquests sistemes, ja alterats per altres efectes de la intervenció humana.

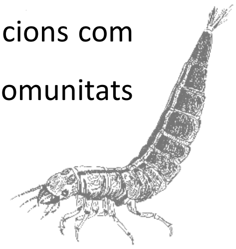
A partir de totes aquestes consideracions, l'objectiu principal d'aquesta tesi ha estat estudiar l'impacte dels incendis sobre les comunitats faunístiques dels estanys mediterranis temporanis. Al llarg d'aquest treball, s'han analitzat les implicacions que té una pertorbació com aquesta sobre les comunitats de macroinvertebrats i amfibis d'una xarxa d'estanys temporanis a través de tres aproximacions. La primera, basada en estudis observacionals de camp que han analitzat els efectes d'un incendi que va cremar parcialment un conjunt d'estanys temporanis (*Capítol I i Capítol II*); la segona, un experiment realitzat al medi on s'han utilitzat mesocosmos col·locats al voltant de basses temporànies situades en punts de la xarxa amb característiques diferents pel que fa al seu grau d'aïllament (*Capítol III*) i finalment, un model de simulació on s'ha tingut en compte l'estructura del paisatge, així com la dinàmica de recuperació de després de l'incendi (*Capítol IV*). Per tal de dur a terme aquestes tres aproximacions s'han estudiat dos sistemes situats al nord-est de la península ibèrica: les basses temporànies de l'Albera (zona baixa mediterrània) i les basses temporànies dels clots de Guils de Cerdanya (alta muntanya pirinenca). D'aquests dos sistemes se n'han mostrejat les comunitats faunístiques, incloent macroinvertebrats i amfibis i s'han identificat amb la màxima resolució taxonòmica possible.

Els resultats lligats a aquestes tres aproximacions, i especialment pel que fa als estudis observacionals, han indicat que els impactes d'un incendi sobre els estanys temporanis són tant directes com indirectes. Així, en els impactes dels incendis sobre sistemes aquàtics cal incloure l'efecte directe del foc sobre els organismes que passen la fase seca de la llacuna en el sediment i es dispersen passivament. Alhora, també es posa de manifest com la capacitat de dispersió és determinant en la recuperació postincendi (*Capítol I*). En conseqüència, els incendis acaben afectant les metacomunitats canviant-ne els seus mecanismes d'estructuració i fent-los fluctuar al llarg de l'eix neutral-nínxol. Tot i això, més enllà de l'incendi, les fluctuacions lligades a la dinàmica de la comunitat al llarg de l'hidroperíode segueixen marcant el patró general de successió en els sistemes temporanis (*Capítol II*).

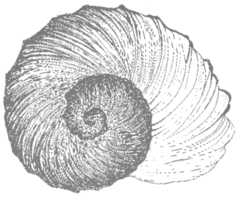
En segon lloc, els experiments realitzats al camp han posat de relleu la importància tant de la xarxa com de la localització dins d'aquesta en vehicular el flux d'individus, així com, altre cop, les diferències entre organismes amb capacitats y modes de dispersió diferents. Tanmateix, altres característiques del paisatge, de nivell més local, com la distància entre basses o el veïnatge d'altres basses, també han influït en la dispersió, i fins i tot han contrarestat els efectes del patró més general de centralitat i aïllament (*Capítol III*). En veure la rellevància de considerar les diferents capacitats de dispersió, tant pel que fa a la resposta postincendi com per la pròpia dinàmica metacomunitària, en els models de simulació es van considerar diferents percepcions del paisatge (diferents xarxes). En considerar-les, es va fer palès, tant per la recuperació postincendi com per la resiliència general de la metacomunitat, que les diferents capacitats de dispersió responien tal com s'havia observat empíricament, mostrant respostes diferents en considerar nous escenaris de pertorbació (*Capítol IV*).

Aquests models, en línia amb el que s'havia observat al *Capítol I* i al *Capítol II*, indicaven que un incendi amb característiques similars al que va succeir a la zona a l'estiu del 2012 no va comprometre en termes generals la metacomunitat dels estanys temporanis ja que aquests es recuperen ràpidament. En general, els resultats observats d'una ràpida recolonització i una alta resiliència dels sistemes estudiats estan lligats a dos factors.

Primerament, a la resiliència intrínseca de les comunitats faunístiques d'aquestes basses, molt adaptades a la sequera. En segon lloc, a l'estructura de la xarxa de basses que va afavorir una ràpida recolonització dels estanys afectats (efecte rescat de la metacomunitat). Tot i això, la futura tendència lligada a l'increment dels incendis així com també a la pèrdua d'hàbitat podrien comprometre realment la capacitat de recuperació de les metacomunitats estudiades, comportant un declivi abrupte de la resiliència a nivell regional, que implicaria un impacte molt més extens i intens que el que s'ha observat en aquest treball. En conseqüència, en un futur, pertorbacions com l'incendi estudiat poden realment comprometre la recuperació de les comunitats d'estanys mediterranis temporanis.



Es requereixen més estudis encaminats a entendre millor la interacció entre metacomunitats i pertorbacions, alhora que cal concebre les metacomunitats en un marc més obert, sense estudiar específicament quins són els mecanismes afavorits o afectats. Nous estudis amb voluntat d'integrar diferents aproximacions, així com analitzar els canvis en la metacomunitat com un gradient fluctuant, ajudaran a caminar cap a una teoria de pertorbacions a nivell metacomunitari. Emprendre aquesta direcció és actualment una necessitat imperiosa per tal d'afrontar amb condicions el present futur incendiari.



RESUMEN

Las perturbaciones son unos de los factores más determinantes en el ensamblaje de las comunidades pudiendo tener un efecto en diferentes niveles (ej., presencia de especies, cambios ambientales o recursos disponibles). Así pues, el rol que las perturbaciones tienen como condicionantes de su estructuración es clave para entender qué dinámicas intervienen en la presencia de según qué especies en según qué sitios. Sin embargo, no sólo las perturbaciones son determinantes en definir qué comunidades encontramos en nuestros ecosistemas. Otras características a nivel regional, como la estructura del paisaje, juegan también un papel muy importante. Por lo tanto, a fin de comprender mejor cómo estas dos capas -los efectos de una perturbación y la estructura del paisaje- influyen en las comunidades a nivel ecológico, se tendrán que considerar las dos escalas donde éstas tienen influencia: escala local y escala regional. En consecuencia, se deberá tener en cuenta tanto las comunidades físicamente delimitadas (escala local) como el intercambio de individuos entre ellas a través de la dispersión (escala regional), definiendo así lo que se conoce como una metacomunidad. Es por todo esto, que si se quiere profundizar en cuál es la influencia sobre las comunidades de perturbaciones con un marcado efecto tanto regional como local (ej., Incendios), es necesaria una perspectiva metacomunitaria.

A nivel metacomunitario, la integración de una teoría vinculada a las perturbaciones és muy incipiente, pero su desarrollo permitirá la creación de un marco teórico necesario para entender y gestionar escenarios futuros (ej., aumento de los incendios, fragmentación hábitats, pérdida de especies). Consecuentemente, hoy nos encontramos con la necesidad de llevar a cabo estudios más integradores (ya sean observacionales, experimentales y/o basados en simulaciones) que sean capaces de analizar los efectos de grandes perturbaciones (incendios) desde una perspectiva metacomunitaria (como la de las comunidades de una red de charcas temporales) para poder comprender mejor la imagen global de esta interacción.

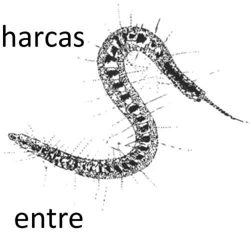
Los incendios son considerados una de las perturbaciones más importantes a nivel global convirtiéndose en grandes estructuradores de las comunidades y ecosistemas. Sin embargo, el conocimiento de sus consecuencias se ha focalizado mayoritariamente en sistemas terrestres, dejando de lado otros sistemas, como los acuáticos, que también las sufren. Además, dentro de estos, los efectos de los incendios sobre las charcas temporales han sido prácticamente ignorados. En el futuro, se espera que el impacto de estas perturbaciones aumente en zonas mediterráneas, sobre todo en cuanto a su extensión y frecuencia, donde precisamente las charcas temporales son muy abundantes y, desde el punto de vista de la conservación de la biodiversidad, relevantes. En consecuencia, cada vez es más urgente entender como los impactos de los incendios influyen sobre las comunidades faunísticas de estos sistemas, ya de por si amenazados por otros efectos de la intervención humana.

Partiendo de todas estas consideraciones, el objetivo principal de esta tesis ha sido estudiar el impacto de los incendios sobre las comunidades faunísticas de las charcas mediterráneas temporales. A lo largo de este trabajo se han analizado las implicaciones que una perturbación como ésta tiene sobre una red de charcas temporales a través de tres aproximaciones. La primera, basada en estudios observacionales de campo que han analizado los efectos de un incendio que quemó parcialmente un conjunto de charcas (*Capítulo I* y *Capítulo II*), la segunda, un experimento realizado en el campo donde se han utilizado mesocosmos colocados alrededor de charcas temporales situadas en diferentes puntos de una red con características diferenciadas por lo que respecta al grado de aislamiento (*Capítulo III*) y finalmente, un modelo de simulación donde se ha tenido en cuenta la estructura del paisaje así como la dinámica de recuperación de las comunidades después del incendio (*Capítulo IV*). Para llevar a cabo estas tres aproximaciones se han estudiado dos sistemas situados en el noreste de la península ibérica: las charcas temporales de la Albera (zona baja mediterránea) y las charcas temporales de los Clots de Guils de Cerdanya (alta montaña pirenaica). En estos dos sistemas se han muestreado las comunidades faunísticas, incluyendo macroinvertebrados y anfibios, que se han identificado con la máxima resolución taxonómica posible.

Los resultados relativos a estas tres aproximaciones, y especialmente en cuanto a los estudios observacionales, han indicado que los impactos de un incendio sobre las charcas temporales son tanto directos como indirectos. Este hecho, abre una nueva concepción sobre los impactos de los incendios sobre sistemas acuáticos: el efecto directo del fuego sobre los organismos que pasan la fase seca en el sedimento de la laguna y se dispersan pasivamente. Asimismo, también destaca como la capacidad de dispersión es determinante en la recuperación post-incendio (*Capítulo I*). En consecuencia, los incendios acaban afectando a las metacomunidades cambiando sus mecanismos y haciéndoles fluctuar a lo largo del eje neutral-nicho. Sin embargo, más allá del incendio, las fluctuaciones relacionadas con la dinámica de la comunidad a lo largo del hidroperíodo siguen marcando el patrón general de sucesión en los ecosistemas temporales (*Capítulo II*).

En segundo lugar, los experimentos realizados en el campo han puesto de relieve la importancia de la red y la localización dentro de ésta en vehicular el flujo de individuos, así como, otra vez, las diferencias entre organismos con capacidades y modos de dispersión diferentes. Sin embargo, otras características del paisaje, de nivel más local, como la distancia entre charcas o la vecindad de otras charcas, también han influido en la dispersión e incluso han contrarrestado los efectos del patrón más general de centralidad y aislamiento (*Capítulo III*). Finalmente, y dada la importancia de las diferentes capacidades de dispersión en la respuesta post-incendio y en la propia dinámica metacomunitaria, en los modelos de simulación se consideraron diferentes percepciones del paisaje (diferentes redes). Al considerarlas, se puso de manifiesto como tanto para la recuperación post-incendio como para la resiliencia general de la metacomunidad, las diferentes capacidades de dispersión respondían como se había observado empíricamente, mostrando diferentes patrones al considerar nuevos escenarios de perturbación (*Capítulo IV*). Así pues, el incendio no comprometió en términos generales a la metacomunidad de las charcas temporales y éstas se recuperaron de forma rápida.

En general, los resultados observados de una rápida recolonización y una alta resiliencia de los sistemas estudiados están ligados primeramente a la resiliencia intrínseca de las comunidades faunísticas de estas charcas, muy adaptadas a la sequía, y, en segundo lugar, a la estructura de la red de charcas que favoreció una rápida recolonización de las charcas afectadas (efecto rescate de la metacomunidad).Sin embargo, la futura tendencia ligada al incremento de los incendios, así como también a la pérdida de hábitat, podrían comprometer realmente la capacidad de recuperación de las metacomunidades estudiadas, comportando un declive abrupto de la resiliencia a nivel regional que implicaría un impacto mucho más extenso e intenso de lo observado en este trabajo. En consecuencia, en un futuro, perturbaciones como el incendio estudiado pueden realmente comprometer la recuperación de las comunidades de charcas mediterráneas temporales.



Se requieren más estudios encaminados a entender mejor la interacción entre metacomunidades y perturbaciones, a la vez que hay que concebir las metacomunidades en un marco más abierto, sin estudiar específicamente qué mecanismos están siendo propiciados o afectados. Nuevos estudios con voluntad de integrar diferentes aproximaciones como analizar los cambios en la metacomunidad como un gradiente fluctuante ayudarán a caminar hacia una teoría de perturbaciones a nivel metacomunitario. Emprender esta dirección es actualmente una necesidad imperiosa para afrontar con condiciones el presente futuro incendiario.

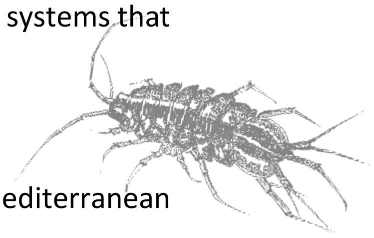
SUMMARY

Disturbances are main determinants of community assembly and have an effect throughout all system levels. Their role in shaping community outcome is key to better understand mechanisms and dynamics affecting species presence. Nevertheless, not only specific disturbance effects play a role in determining the post-disturbance community outcome. Landscape structure and the regional context of each local community also modulates its resilience. These two layers —disturbance effects and landscape structure— acts at both local and regional scales. The interaction among several delimited local communities through regional dispersal of individuals constitutes what is called a metacommunity. Thus, a metacommunity perspective is as a consequence, essential to disentangle how disturbances that have a regional affectation such as wildfires, can determine species assembly processes.

Joining metacommunity theory within disturbance theory is a current need in the sense of advance toward a theory of metacommunity disturbance. However, how disturbances interact with metacommunities is still unclear, blurring the creation of a general framework that would help to better comprehend and consequently, cope, with future disturbance scenarios (e.g., wildfire increase, habitat fragmentation, species loss). Therefore, we currently are in the need of more integrative studies (e.g., including simulation models, experiments and observational studies) that will account with disturbances (e.g., wildfires) from a metacommunity perspective (e.g., network of temporary ponds) in order to better disentangle and comprehend the full picture of disturbances interaction with metacommunities.

Wildfires are among the most worldwide distributed disturbances and are considered main natural community shaping agents. However, wildfire impacts on aquatic systems have been historically poorly addressed in comparison to their impacts on terrestrial systems. Furthermore, within aquatic systems, ponds, including temporary ones, have

received still less attention regarding wildfire impacts. The future scenario pictured regarding wildfires in dry regions such as the Mediterranean, where temporary ponds are abundantly found, draws an increase in wildfire intensity and recurrence. Consequently, there is an urgent need in comprehending wildfire impacts if some considerations must be taken to maintain and conserve these endangered systems that hold unique communities.



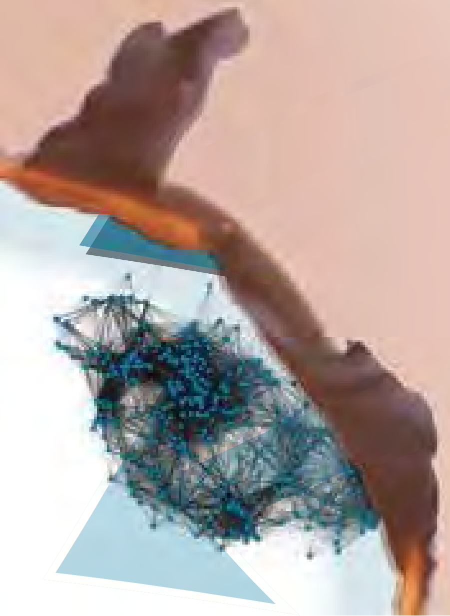
The main objective of the current thesis was to study wildfire impacts on Mediterranean temporary ponds faunal communities. In this thesis we analysed the implications of such disturbance and pond network relevance through three different approaches: observational field study through the analysis of samples from a natural wildfire event, which partially affected a temporary pond network (*Chapter I* & *Chapter II*), an experimental field study, using mesocosms around temporary ponds at different locations within the network (*Chapter III*) and a simulation model, which accounted with landscape structure and post-wildfire recovery patterns (*Chapter IV*). To carry all these analyses the Albera temporary ponds (lowland Mediterranean temporary ponds) and Guils de Cerdanya temporary ponds (high altitude Pyrenean temporary ponds), located in north-eastern Iberian Peninsula, were sampled. Their macrofaunal community, including macroinvertebrates and amphibians, was identified to the maximum taxonomic possible level in order to develop all these analyses.

Our results indicated that in the observational study, temporary ponds were affected both directly and indirectly by the wildfire, providing a new perspective on wildfire impacts on aquatic systems (i.e., direct burning of aquatic organisms aestivating in the pond sediment) and highlighting species dispersal ability relevance in post-disturbance recovery (*Chapter I*). These factors affected pond metacommunity assembly mechanisms, which fluctuated along the neutral-niche gradient, but were mainly influenced by hydroperiod successional changes which still determined successional changes overcoming wildfire consequences (*Chapter II*). Moreover, the results also pointed out the different behaviour according to the dispersal ability of the organisms and highlighted the relevance of location within the pond network. Although distance and direction from the source also modulated dispersal dynamics along colonization trends (*Chapter III*). Thus, to consider several landscape perspectives (i.e., dispersal abilities) and the network structure appeared key in order to better approach metacommunity dynamics. Finally, the simulation models showed how dispersal ability —species landscape perception— was a key resilience and recovery driver determining post-disturbance dynamics with a strong link with pond network structure (*Chapter IV*). The studied wildfire did not compromise community recovery at a landscape scale, but an increase on this disturbance intensity might represent a rapid and pronounced decline in ecosystem resilience at the regional level (i.e., marked non-linear fall of community resilience at determined wildfire intensities).

In general, a high resilience to wildfire was observed in the studied metacommunity, mainly due to their intrinsic drought-adapted community and the dense network structure that fostered recolonization of affected ponds and that was helped by the important role of dispersal in these systems (i.e., metacommunity rescue effect). Nevertheless, future scenarios, which encompass an increase in wildfire regimes and greater habitat fragmentation, do not ensure that such intrinsic resilience of temporary ponds will be maintained. Consequently, future wildfires may really compromise Mediterranean temporary ponds faunal communities and the current metacommunity rescue effect.

More studies aimed to disentangle disturbance effects on metacommunities are needed and they must open the current metacommunity perspective and not focus on which mechanisms is being affected or fostered. New integrative approaches and the current perspective of several driving forces acting together along a changing gradient may help to improve the picture of a metacommunity disturbance theory, which is necessary to face the present incendiary future.

1. GENERAL INTRODUCTION



1.1 Disturbances, patchy landscapes and their interaction on community outcome

Disturbances constitute an intrinsic mechanism of natural systems and have an effect throughout all the levels of ecological organisation (White and Pickett 1985). Disturbance consequences represent a shaping agent of communities and their assembly processes, which at the end, influence —together with other constraints— which species will be present in the studied system (Weiher and Keddy 1999). The study of disturbance impacts has fostered ecological research in order to better comprehend such phenomena and their implications in both biodiversity but also in management (Pulsford et al. 2016). Consequently, White & Picket (1985) in their intention to move toward a theory of disturbance proposed the following definition: "*A disturbance is any relatively discrete event in the time that disrupts ecosystem, community or population structure and changes resources, substrate availability or the physical environment*". In this broad definition, they attempted to include a wide range of phenomena going from less frequent catastrophic events that completely disrupt systems functioning to smaller more frequent disasters. Although this is a definition and as such, can have its own arbitrariness, here it will be used as the conceptual reference of disturbance. Therefore, wildfire will be considered as a disturbance (e.g., Gresswell 1999, Minshall et al. 2001, Whitney et al. 2015, Han et al. 2018), but also temporary pond hydroregime —understood as pond recurrent drought and refilling process— because both processes disrupt community structure changing their resources or physical environment (e.g., Boix et al. 2004, Florencio et al. 2009, O'Neill 2016). Of course, they act at different temporal frames and with differential disruptive powers, but they together constitute community assembly modulators interacting on the same communities over a common landscape.

Landscape structure and properties are tightly linked with disturbance consequences due to the heterogeneity that they can produce in a landscape (Taylor et al. 1993, Poff 1997, Turner et al. 2015). Thus, as disturbances change system equilibrium

differently along the landscape, they also generate different gradients of alteration — i.e., different environmental pressures. Therefore, disturbances create a mosaic of patches, which at the end constitute an ecological system where interactions among patch —i.e., patch dynamics— will determine species presence and system functioning (White and Pickett 1985, Wu and Levin 1994, Turner et al. 2015). Additionally, the nature of the landscape will greatly determine disturbance consequences on resource availability and/or species presence. For example, the isolation of a disturbed community in the landscape, in terms of its biotic or abiotic properties, will be crucial for its post-disturbance response and thus, for the successional process that will define future community (White and Pickett 1985, Economo and Keitt 2010, Borthagaray et al. 2015a).

Layers defining post-disturbance community outcome

In this sense, two different *layers* appear as post-disturbance community drivers (Figure 1.1): disturbance specific consequences (i.e., the alteration of selection pressures or the environmental changes) and landscape structure and properties (i.e., affected patches distribution, patch number, and patch network characteristics). Indeed, these two *layers* are driving post-disturbance community outcome mainly through two of the four high-level processes in community ecology: *selection* and *dispersal*, though *drift* and *speciation* also play an important role (Vellend 2016). *Selection* might change community species fitness and consequently influence species abundance or presence due to post-disturbance new conditions –i.e., environmental pressures (Earl and Blinn 2003, Laliberté et al. 2013, Zhang et al. 2019). On the other hand, *dispersal* is tightly related with patch network structure determining species success in immigration throughout all network patches and thus, determining post-disturbance recovery (Woods et al. 2016, González-Trujillo et al. 2019, Mausbach and Dzialowski 2019). Finally, *drift* and *speciation* will for sure play a role in driving post-disturbance community outcome (Vellend 2016), firstly because stochasticity will obviously drive such process (Hubbell 2001) and secondly, although in a larger temporal scale,

speciation might also influence community outcome (i.e., evolutive or genetic changes related to a disturbance; Evanno et al. 2009).

Interaction among these four high-level processes proposed by Vellend (2016) – mainly *selection* and *dispersal*– can be addressed from a regional perspective (i.e., landscape) and consequently, considering several patches that are interacting through the exchange of species individuals among them. Therefore, community assembly is being strongly determined by dispersal and individual fluxes among patches, also known as *local communities,* whose limits can be stablished (e.g., ponds, lakes, forests surrounded by fields, islands) and that are distributed along the landscape. Due to this strong exchange of individuals among *local communities*, they cannot be considered as completely disconnected from their neighbours, but, due to their marked limits they neither can be considered as the same community. In this context, when considering the regional and local perspectives of several communities linked by dispersal, the *metacommunity* concept arouses following Leibold et al. (2004) words, "*as a set of local communities that are linked by dispersal of multiple potentially interacting species*". This theory has determined the last decades of community ecology and is still surrounding the frameworks of most community ecology research (Holyoak et al. 2005, Logue et al. 2011, Vellend 2016).

The "2.0. version" of metacommunity theory

Metacommunity theory is currently evolving and moving toward a "version 2.0" of its theoretical framework (Leibold and Chase 2018). At its beginning, four main paradigms where considered to be metacommunities main assembly determinants: *species sorting*, *patch dynamics*, *mass effects* and the *neutral paradigm*. Each of these paradigms were defined and proposed as four broad lines or approaches that until that moment theoretical and empirical work on metacommunities had fall along (Leibold et al. 2004, Holyoak et al. 2005, Borthagaray et al. 2015a). However, these four paradigms were understood as strictly defined alternative hypotheses and led to a closed view of metacommunity ecology where the aim was to seek at which paradigm each community

belonged or by which paradigm it was being assembled (Cottenie 2005). Nowadays, this more closed view has opened to a more nuanced view where all the processes of each approach act together, but with different relative importance in the metacommunity assembly (Logue et al. 2011, Brown et al. 2017). In this sense, the four paradigms have been renamed as *metacommunity archetypes* to better conceptualize this perspective change (Leibold and Chase 2018). Hence, the current metacommunity theory is understood as the balance among several processes or driving forces (e.g., environmental conditions, dispersal, stochasticity or colonization-extinction patterns) that will modulate the assembly of local communities conforming the metacommunity in a continuous gradient that goes from a strongly deterministic to a totally neutral assembly. Leibold and Chase (2018) redefine the four archetypes as follows. *Species sorting (SS)* archetype focuses on the environmental differences among local communities that determine the number of species that can inhabit them. Thus, is strictly deterministic and assumes a strong linkage among environment and species presence in each local site (i.e., niche assembly). *Patch dynamics (PD)* archetype focuses on the balance between colonization and extinction that mediate species coexistence in the metacommunity. Here, the niche strength is smaller and the stochasticity in extinction due to demographic or disturbance-related processes is the driving force that for instance allows coexistence of competitively divergent species in the same metacommunity in a competition-colonization balance. *Mass effects (ME)* archetype focuses on source-sink relations that can counter-balance competitively inferior species or harsh environments via dispersal. Consequently, the exchange of individuals among local communities is mostly driving metacommunity assembly. Finally, *neutral theory (NT)* archetype focuses on stochasticity in dispersal and demography as main determinants of the assembly of local communities. Here, niche forces are unimportant in determining species presence thus, being only stochastically driven processes, the ones playing a role in metacommunity assembly.

Disturbances and metacommunities

The linkage between the regional and local contexts considered in metacommunity ecology has still not been fully fitted within disturbance theory (Pulsford et al. 2016). The spatial relevance in shaping disturbance impacts as well as post-disturbance succession is not specifically accounted within the metacommunity ecology (Leibold and Chase 2018). However, disturbances have been and are being studied from a metacommunity point of view, considering its consequences on the assembly process. Therefore, although not specifically addressed (Pulsford et al. 2016), disturbances are accounted as another of the shaping agents of metacommunity assembly process, which change environmental conditions —i.e., selective pressures—, dispersal dynamics, stochasticity or colonization-extinction patterns (Leibold and Chase 2018). Several works in fact, already contain disturbance consequences from a metacommunity perspective ranging from *observational studies*, to *laboratory or field experiments* and *simulation models*. Studies that will integrate all three or some of these perspectives together will for sure have a better perspective of disturbance consequences on metacommunity assembly. Consequently, they will improve the comprehension of both the specific disturbance impact and the assembly mechanism driving the metacommunity.

Observational studies, account with the most realistic picture of disturbance effects on metacommunities because disturbance consequences are difficult to simulate in an experimental context. Although *observational studies* might suffer from accounting with too much variability, they constitute the best way to catch all disturbance-produced consequences. Studied disturbances can be natural and intrinsic from the system, such as floods and drought, or human-related, such as pesticide impact, invasive species, deforestation, fragmentation, fire or mining. Although different in some of their properties, they all disrupt metacommunity dynamics and determine post-disturbance recovery. Post-disturbance consequences can lead to a change in habitat conditions and environmental characteristics, which determine metacommunity and a strong niche assembly, being species sorting archetype the main metacommunity determinant (Urban 2004, Laliberté et al. 2013, Han et al. 2018, Zhang

et al. 2019). However, greater dispersal can drive post-disturbance succession, decreasing environmental filters importance —mass effect archetype— (Vanschoenwinkel et al. 2013, Woods et al. 2016, Cai et al. 2017, González-Trujillo et al. 2019, Hernández-Ordóñez et al. 2019). In this sense, network structure and connectivity play a major role in shaping post-disturbance metacommunity dynamics, recovery or biodiversity loss (Arthaud et al. 2013, Campbell et al. 2015, Kuglerová et al. 2015, Rosati et al. 2017, Horváth et al. 2019, Mausbach and Dzialowski 2019). Nevertheless, disturbances can reset metacommunities and foster dispersal and demographic stochasticity in the recolonization process thus, promoting a neutral assembly in post-disturbance communities (Catano et al. 2017, Dong et al. 2017, Rosati et al. 2017), which can be traced into the genetic structure (Evanno et al. 2009). Therefore, disturbance modulates the balance between environmental and stochastic processes and thus the post-disturbance assembly process (Boudell and Stromberg 2008, Vanschoenwinkel et al. 2010) acting as a shaping agent that changes niche and neutral forces in community assembly.

Laboratory or field experiments on the other hand, represent an advance towards specifically address metacommunity processes. Laboratory-based experiments normally represent metacommunities using microorganisms (e.g., protists, bacteria) and creating some connection or flux of individuals among local communities (Cadotte 2007). Such experiments help to better comprehend the importance of individual fluxes after a disturbance, which is difficult to determine and quantify (Altermatt et al. 2011, Limberger and Wickham 2012, Ojima and Jiang 2017). Furthermore, such experiments can help also in understanding the landscape structure relevance in determining post-disturbance diversity (Altermatt and Holyoak 2012). Other studies account with species interactions and the relevance of priority effects in post-disturbance recovery (Ojima and Jiang 2017). In this line, laboratory-based experiments can better account with stochastic driven processes —neutral dynamics—, which can be fostered by disturbances concomitantly with a decrease of environmental control (Fukumori et al. 2015). Similarly, field-based experiments can specifically focus on dispersal and disturbance interaction, where connectivity plays a key role in post-disturbance

recovery (Starzomski and Srivastava 2007, Trekels et al. 2011, Thrush et al. 2013). Moreover, field-based experiments account with species real interactions after a disturbance that can counterbalance dispersal-driven processes (e.g., predation) or produce dominance shifts in affected communities (Howeth and Leibold 2010, Eggers et al. 2012, Sinclair et al. 2015). Therefore, both experimental approaches can better disentangle dispersal-mediated dynamics in a more realistic context than for example simulation models. However, to simulate or experimentally reproduce a disturbance (e.g., habitat loss, pesticide) has always some drawbacks and difficulties in being realistic in terms of its characteristics (e.g., extension, intensity or patchiness).

Simulation models based on theoretical approaches have been largely used along metacommunity ecology considering the specific impact of disturbances. Some works have addressed the post-disturbance relevance of the spatial structure in heterogeneous or homogeneous landscapes —i.e., fragmentation— or the threshold by which recover would become compromised (Alados et al. 2009, Brodie et al. 2016, Catano et al. 2017). The inclusion of species interactions together with dispersal dynamics has also been accounted by some simulations, where interactions among specialists or generalists with divergent dispersal capacities or the trophic chain link with regional processes is emphasised (Guichard and Steenweg 2008, Eklöf et al. 2012, Takimoto et al. 2012, Büchi and Vuilleumier 2014, 2016). In this line, niche and neutral processes in different scales have been studied and how both can appear in the same metacommunity but at divergent scales (Kadmon and Benjamini 2006, Leibold et al. 2019). Finally, the consideration of disturbances as diversity generators —intermediate disturbance hypothesis (Pulsford et al. 2016)— where the consideration of regional structure, dispersal as well as drift has proved that regionally, disturbance can enhance diversity metrics (Büchi et al. 2009, Lichstein and Pacala 2011, Tuytens et al. 2014, Harvey et al. 2018). Nevertheless, the usage of purely simulation models has its drawbacks as it is generally difficult to account with all possible processes in natural systems. On the other hand, such approaches have demonstrated the relevance of dispersal driven processes or purely stochastic ones in realistically shape metacommunity assembly, which is difficult to assess in laboratory or field studies.

Joining metacommunity theory within disturbance theory and vice versa is a current need (Pulsford et al. 2016). However, not in the sense of studying the impact of disturbances on metacommunities, which is something that is already being done, but in the sense of advance toward a *theory of metacommunity disturbance*. Nevertheless, in order to move in this way, more studies accounting with disturbances consequences in metacommunities are needed (Pulsford et al. 2016, Leibold and Chase 2018). Do disturbance impacts promote neutrality? Or they promote highly niche assembled metacommunities? The answer to these questions is still not fully pictured, because both answers have been detected (e.g., Woods et al. 2016, Dong et al. 2017, Han et al. 2018). Consequently, we currently are in the need of more studies of any kind (e.g., theoretical models, experiments or observational studies) accounting with example disturbances (e.g., wildfires) and example metacommunities (e.g., network of temporary ponds) in order to better disentangle and comprehend the full picture of *disturbances in metacommunities*.

1.2 Wildfire as the disturbance

Wildfires are among the most studied disturbances from several perspectives and science fields (Wright and Bailey 1982, Díaz-Delgado et al. 2002). The relevance of wildfires in shaping or affecting earth ecosystems constitutes a natural disturbance, intrinsic of some systems, that has been and is being fostered by human activities (Bowman et al. 2009, Pausas and Keeley 2009, Pausas and Ribeiro 2013). Wildfire regimes vary among regions due to climatic characteristics and fuel structure; these variations rise fire frequency and intensity in areas such as Mediterranean climates (Pausas 2004, Lavorel et al. 2007, Le Page et al. 2008, Schultz et al. 2008, Pausas and Ribeiro 2013). However, these climatic characteristics are predicted to change globally with an increase in temperatures and rainfall decreases, especially during summer months (Giorgi and Lionello 2008, Kovats et al. 2014, Calbó Angrill et al. 2016). Such conditions will promote wildfires (Moritz et al. 2012, Pausas and Fernández-Muñoz 2012) increasing its predicted frequency, intensity and extension (Turco et al. 2018).

Although such changes in wildfire regimes are globally expected (Le Page et al. 2008, Flannigan et al. 2009, Moritz et al. 2012), some areas with Mediterranean climate will experience a particular increase in wildfire activity and regimes (Pausas and Fernández-Muñoz 2012, Botija Llasat et al. 2016, Turco et al. 2018).

Within the current climatic emergency context derived from both climatic alteration as well as changes in human-environment interactions (e.g., field abandonment, reforestation, fossil fuel usage, stockbreeding) and considering both climatic predictions and wildfire regime predictions in the Mediterranean regions (Kovats et al. 2014, Botija Llasat et al. 2016, Calbó Angrill et al. 2016), the study and deepening of wildfire consequences on all ecosystems is currently an imperious need to better cope and comprehend future scenarios, that are already present. While writing these chapters clear examples of greater and more catastrophic wildfires are being reported such as Terra Alta wildfire as the greatest since 2012 in Catalunya: 6500 burned hectares (3/24 news 2019), Siberian wildfires: 2.6 million hectares (Jenner 2019), Alaska wildfires: 2.5 million hectares (Law 2019) and Amazonian wildfires still burning and with the peak day 700% higher than the average for the same date over the past 15 years (Watts 2019). Accordingly, in Mediterranean regions as well as globally, terrestrial ecosystems have been deeply studied and current proposals and interests are mainly focused on management and wildfire prevention (Kovats et al. 2014, Botija Llasat et al. 2016). Contrastingly, aquatic ecosystems have been less studied from a wildfire impact perspective and consequently, little is known and further needs to be disentangled regarding wildfire consequences in aquatic systems (Bixby et al. 2015, McCullough et al. 2019).

Burning waters. What do we know?

Although historically wildfire impacts on aquatic ecosystems have been poorly studied, with firsts studies being published in 1989 (Christensen et al. 1989, Minshall et al. 1989), some aquatic systems such as permanent rivers and great lakes have received some attention from researchers since those dates (Gresswell 1999, Minshall 2003). Aquatic

systems can suffer direct and indirect impacts from a wildfire, being direct impacts related to the specific wildfire event and indirect impacts related to the post-fire days, months or years (Minshall et al. 1989, Gresswell 1999). Most of these consequences are linked with wildfire impacts on the watershed because aquatic systems will receive water from the burned area, concentring wildfire consequences in the aquatic system (Prepas et al. 2009).

All these impacts can be grouped in five main consequences, mainly derived from vegetation loss and ash generation/input in the watershed (Minshall et al. 1989, McCullough et al. 2019). First, an increase in runoff after a wildfire is generally expected due to watershed vegetation loss and soil impermeabilization by ash, which produces a system wash up (Vieira et al. 2004, 2011, Whitney et al. 2015, Pereira et al. 2016), but also the creation or modification of habitats (Kleindl et al. 2015). Second, vegetation loss (e.g., specially canopy) favour light incidence and consequently water temperature rise after the wildfire and during the following years (Minshall et al. 1997, Hossack and Corn 2008, Isaak et al. 2010, Mahlum et al. 2011, Rhoades et al. 2011, Beakes et al. 2014, Cooper et al. 2015, Rodríguez-Lozano 2015, Rosenberger et al. 2015). Third, ash and debris inputs into the aquatic system, both during the wildfire and during the post-fire years will change water properties with an increase of nutrient loads (Spencer and Hauer 1991, Spencer et al. 2003, Bladon et al. 2008, Mast and Clow 2008, Smith et al. 2011, Mast et al. 2016) as well as debris, turbidity and sediment transport (Bêche et al. 2005, Reale et al. 2015, Vaz et al. 2015). Fourth, because of light, temperature and nutrient increase algal blooms are normally enhanced after a wildfire, thus changing aquatic habitat resources from allochthonous to autochthonous (Cowell et al. 2006, Verkaik et al. 2013, Silins et al. 2014, Cooper et al. 2015, Klose et al. 2015). Finally, aquatic faunal organisms can also be affected by wildfire consequences, affecting all the trophic chain, from bacteria (Ferrenberg et al. 2013) to macroinvertebrates, amphibians (Hossack and Corn 2007, Muñoz et al. 2019) and fish (Beakes et al. 2014). The response can strongly vary in accordance with post-fire condition (e.g., floods, droughts, other fires) and can led to a strong decrease of general abundance (Vieira et al. 2011, Whitney et al. 2015). However, together with the rise in temperature and trophic resources some groups are

benefited and consequently increase their abundance (Mihuc and Minshall 1995, Scrimgeour et al. 2001, Minshall et al. 2001, Malison and Baxter 2010a, b, Beganyi and Batzer 2011, Oliver et al. 2012, Brown et al. 2013, Lewis et al. 2014, Verkaik et al. 2015, Rodríguez-Lozano et al. 2015, Venne et al. 2016, Robson et al. 2018). Such biomagnification process can return to the terrestrial system with increasing flying macroinvertebrate emergence and the consequent increase of terrestrial predators in what has been named the *fire pulse* by Malison and Baxter (2010).

Although all these reported wildfire consequences, the impacts of wildfire strongly depend on system characteristics and therefore on ecological context (McCullough et al. 2019). Therefore, wildfire impacts will not be the same in rivers, where post-fire floods can be strongly determinant (Vieira et al. 2011), or in permanent boreal lakes, where nutrient accumulation can led to greater eutrophication and increase of generalist species (Scrimgeour et al. 2001, Araneda et al. 2013, Lewis et al. 2014, Mendoza et al. 2015). In this same line, drought can also compromise wildfire impacts on temporary aquatic systems changing these beforementioned consequences in comparison to permanent aquatic systems (Verkaik et al. 2013, 2015). However, if studies on aquatic permanent systems and wildfires are few, studies regarding aquatic temporary systems are scarce and specially regarding wetlands (Bixby et al. 2015). Therefore, to comprehend and directly know which are the impacts of wildfires on temporary wetlands and their response to such disturbance appears within this context as an important need. Because, in front of the future pictured scenario, the idea of not knowing how these systems are going to be affected is more threatening that the scenario itself.

1.3 Temporary ponds as the metacommunity

Temporary ponds and small aquatic systems have a great potential as model systems to study and respond ecological questions and specially to studies having a metacommunity perspective (Blaustein and Schwartz 2001, De Meester et al. 2005, Beklioglu et al. 2014). Such systems are ubiquitous around the globe, although in some climatic areas (i.e., Mediterranean regions) they can be more commonly found (Williams 2006). They are normally grouped on the landscape conforming a network of aquatic habitats that have more or less a similar hydro-regime based on their recurrent drought during the dry season. Consequently, they host species with capacity to tolerate such harsh conditions (i.e., drought) as well as benefit from them as reproductive spots due to the lack of fish (Griffiths 1997, Williams 2006, Gómez-Rodríguez et al. 2009, Boulton et al. 2014, Boix et al. 2016). All these particular characteristics have made temporary ponds hotspots of biodiversity and ecological interest for conservation purposes as well as from a metacommunity perspective understanding of assembly processes.

Wet islands in the dry land

Temporary ponds constitute a network of different water bodies that are connected among them through individual dispersal in a land matrix —i.e., a metacommunity (Leibold et al. 2004). Therefore, each one of the temporary ponds in the network can be considered as a node and the individual fluxes among ponds as functional links between them thus, assuming the temporary pond network as a graph (Urban and Keitt 2001, Fortuna et al. 2006, Minor and Urban 2008, Dale 2017). This conceptualisation, allows to better approach temporary ponds functioning at the landscape scale integrating the regional and the local perspectives (Leibold et al. 2004, Borthagaray et al. 2015a).

Network properties such as the position of network nodes (i.e., water bodies) and how are they spatially distributed throughout the landscape has a strong relevance in species distribution, species fluxes and system biodiversity (Economo and Keitt 2008,

2010, Horváth et al. 2019). Indeed, considering only network spatial structure, using neutral models, has provided relevant predictions of diversity patterns (Hubbell 2001, Muneepeerakul et al. 2008, Rosindell et al. 2011). Therefore, accounting with network characteristics in order to understand temporary ponds metacommunity dynamics will enhance our understanding on the pure landscape influence, improving our understanding on their functioning and providing an innovative framework for management against disturbances —e.g., wildfires among others (Estrada and Bodin 2008, Chang et al. 2013, Borthagaray et al. 2014, Sokol et al. 2015, Munoz et al. 2018)

In fact, the influence of network characteristics (i.e., landscape structure) plays a key role in driving temporary ponds metacommunity dynamics due to its effect on species dispersal ability (Cañedo-Argüelles et al. 2015, Jones et al. 2015, Grainger and Gilbert 2016, Vannette and Fukami 2017, Shanafelt et al. 2018). Indeed, dispersal ability of organisms strongly determine their capacity to move through the network and consequently their landscape perception (Heino 2013, Borthagaray et al. 2015b, Hill et al. 2017b). Therefore, when considering temporary ponds metacommunities, we must account with the several dispersal strategies that organisms inhabiting such habitats have developed (Williams 2006, Oertli et al. 2008, Boix et al. 2016).

Resting in drought, bound by wind and flight

Drought timing and hydro-regime characteristics represents main determinants of temporary ponds (Williams 2006, Boulton et al. 2014, Boix et al. 2016). Organisms inhabiting such systems must face water loss at some point in order to survive in these ecosystems. Therefore, several strategies have been developed in order to cope with drought as well as its unpredictability (Wiggins et al. 1980, Williams 1985, Batzer and Boix 2016). In the broad sense, two main strategies coexist: drought resistance and drought avoidance. Drought resistance consists in remaining in the pond sediment after drying in some life-form (e.g., resistant eggs, buried adults, diapause) and hatch when ponds refill again. On the other hand, drought avoidance consists in leaving the pond before it is completely dried and disperse to more permanent systems (e.g., permanent

ponds, streams, artificial ponds). These two broad strategies, at the end, constitute two ways of surviving drought, but also of interacting with neighbouring ponds and landscape defining passive and active dispersal abilities (Wiggins et al. 1980, Bilton et al. 2001, Heino 2013).

Passive dispersal is mostly understood as the transport of organisms' propagules, which can tolerate harsh conditions (i.e., drought) from one place to another by different physical or biological vectors (Incagnone et al. 2015 and references therein). Some examples of these vectors can be strong winds that can transport small propagules (Cáceres and Soluk 2002, Vanschoenwinkel et al. 2008a, Horváth et al. 2015) or bigger organisms that can transport these propagules with them (e.g., intestinal tract, attached, …) and that can transport them throughout large distances (Bohonak and Whiteman 1999, Frisch et al. 2007, Vanschoenwinkel et al. 2008b, Valls et al. 2016, Lovas-Kiss et al. 2019). Contrastingly, active dispersal is generally and broadly understood as the ability of an organism to move, more or less consciously, from one place to another (Dixon et al. 1993, Bilton et al. 2001). However, within active dispersal several capacities have been clearly identified, mostly in relation to flight capacity, body size or covered distance (Wiggins et al. 1980, De Bie et al. 2012, Heino 2013). These several capacities define divergent landscape perceptions and thus, different networks according to species dispersal ability (Borthagaray et al. 2012, 2015b).

Furthermore, active dispersal is also strongly determined by environmental, source habitat and target habitat conditions (e.g., weather, drought, predator presence), which strongly determine the risk-benefit balance of flying from one habitat to another (Boix et al. 2011, Boda and Csabai 2012, Trekels and Vanschoenwinkel 2017). While all these determinants have been greatly studied (Bilton et al. 2001), the focus on how network structure and the surrounding matrix can affect dispersal fluxes has remained less studied, though some studies in streams highlight their importance (Bogan and Boersma 2012, Cañedo-Argüelles et al. 2015). Consequently, to consider and include network structure and properties when analysing dispersal-driven processes (i.e., temporary ponds metacommunity dynamics) and its effects on community

assembly can provide a more realistic picture of network relevance as well as their different determinants (i.e., different dispersal perspectives, broad versus fine scales).

Fragile jewels: biodiversity and conservation

Although temporary ponds can be considered model systems to study ecological processes (Blaustein and Schwartz 2001, De Meester et al. 2005, Williams 2006), these systems and particularly those in the Mediterranean region have drawn researchers attention, but unfortunately for their fast disappearance (Skinner and Zalewski 1995, Rhazi et al. 2012, Bagella et al. 2016, Boix et al. 2016). Human activity has decreased temporary ponds numbers throughout years mostly by draining these systems (Richter et al. 1997, Brown 1998, Euliss and Mushet 1999, Rhazi et al. 2001, Wood et al. 2003). Thus, the ecosystem service that they carry as well as their valuable and unique flora and fauna are endangered specifically by human impact and interference (Skinner and Zalewski 1995). Indeed, climate change is not impacting temporary ponds per se and contrarily to what could be expected, seasonal waters are increasing, though in detriment of permanent ones (Pekel et al. 2016). Consequently, human-mediated impacts constitute the main threat for temporary ponds degradation and loss.

Disappearance of these habitats represent a relevant biodiversity loss, as they harbour or interact with lots of species, which rely on these systems for reproduction (Griffiths 1997, Gómez-Rodríguez et al. 2009) or have evolved to cope with seasonal drought and depend on it (Boix et al. 2016). In fact, though they have been generally underestimated, temporary ponds can present as much or surpass species numbers of more “important” aquatic systems (Boix et al. 2001), specially for invertebrate taxa. Seasonality (i.e., flooding and drying recurrence) is one of the main determinants of these habitats because defines habitat suitability and constitutes a refugium from fish predation. Although in other regions some fish species have strategies to survive drought (e.g., Pazin et al. 2006, Laufer et al. 2009, Lanés et al. 2014), in the Mediterranean basin, fish cannot tolerate complete and unpredictable drought and are consequently excluded from temporary ponds, which allows the existence of a totally

different assemblage adapted to drought such as large branchiopods or other crustacean groups as well as amphibians (Williams 2006, Pretus 2009, Boix et al. 2016). Moreover, not only temporary ponds faunal taxa have this marked uniqueness, these systems are of huge interest floristically (Casas et al. 1998, Font and Vilar 1998, Font 2001, Bagella et al. 2010). The relevance and uniqueness of these habitats —especially in the Mediterranean area— has led to their inclusion in the European Habitats Directive as Priority habitats (Habitats 1992).

Mediterranean temporary ponds constitute key habitats for ecological studies as well as biodiversity conservation but because of their small size they are very vulnerable and they have been historically underestimated although their global importance (Downing 2010, Bagella et al. 2016). Therefore, the comprehension of these systems functioning as well as their response against disturbances such as wildfires, which has not been addressed for these type of wetland habitats (Bixby et al. 2015) is a step forward in Mediterranean temporary ponds conservation. Furthermore, to consider their landscape structure from a network perspective, accounting with species specific dispersal abilities, has seldom been implemented in these systems although landscape importance has been repeatedly reported (De Meester et al. 2005, Boix et al. 2016, Horváth et al. 2019). Consequently, the development and improvement of landscape scale approaches focused on comprehending temporary ponds regional responses to disturbances (e.g., habitat loss, habitat degradation, wildfire, etc.) is a current imperious need in order to predict or at least draw some future scenarios. Finally, regardless of simulations or any other approach, the basic knowledge of these habitats list of species represents by itself, an improvement in terms of highlighting the biodiversity that temporary ponds hold, fragile jewels on which unique species depend to survive in the current dramatically changing world.

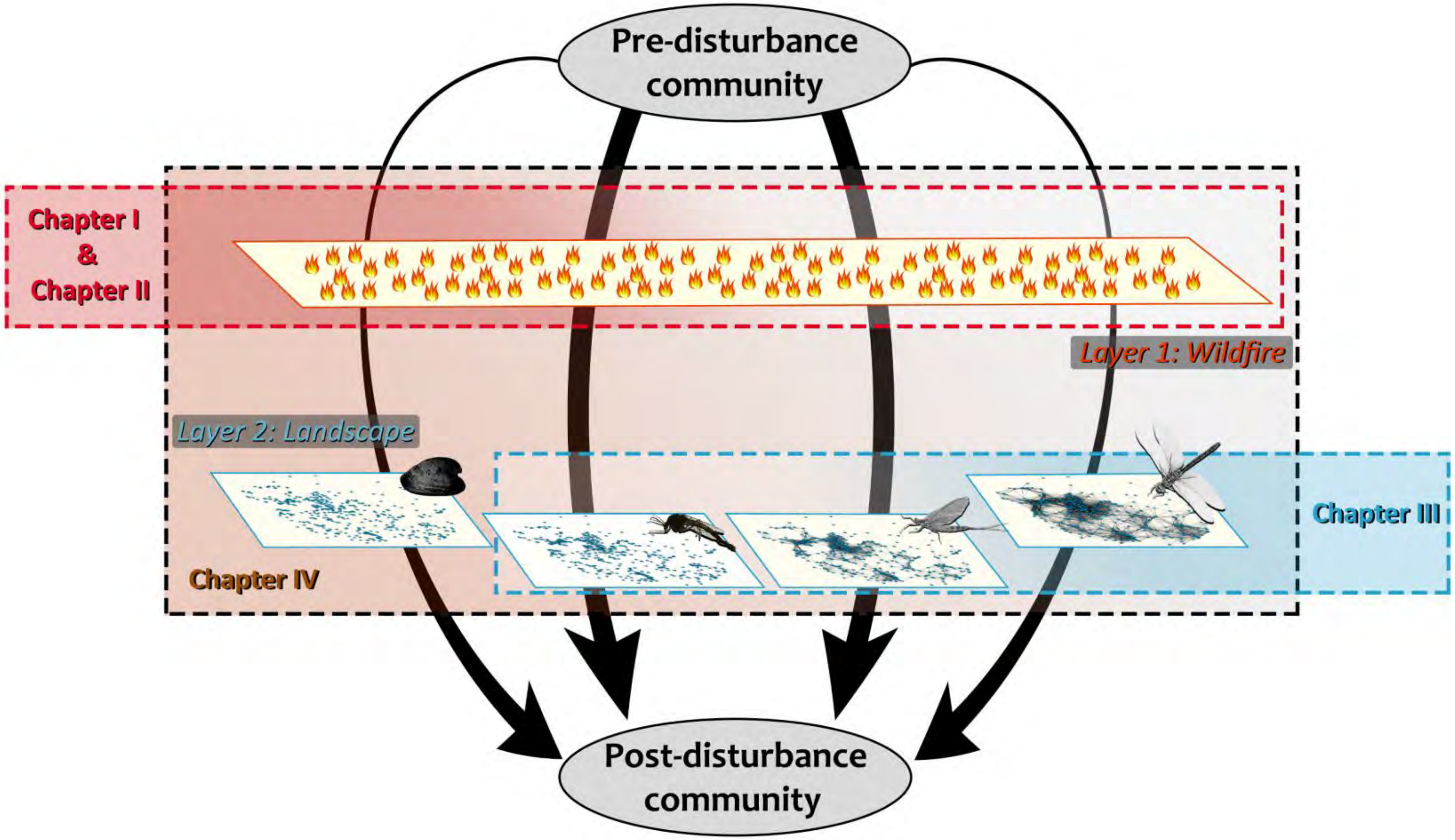
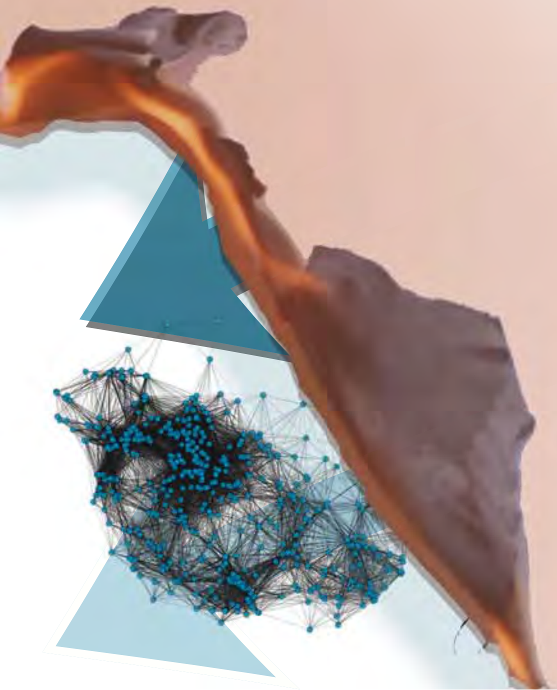


Figure 1.1: *The two layers defining post-disturbance community outcome: 1) the effects linked to the wildfire impact and 2) the regional perspective of each species that will determine its post-disturbance response. Black arrows correspond to each species evolution throughout these two layers that will modulate its final abundance or presence in the post-disturbance community. Chapter I and Chapter II explicitly focus on wildfire impacts on temporary ponds. Chapter III on network location and species regional perspective relevance. Finally, Chapter IV simulates several wildfire scenarios considering the different dispersal abilities.*

2. GENERAL OBJECTIVES



Considering the future climatic perspectives, the current knowledge on the response of temporary ponds to wildfire disturbance, the relevance of metacommunity perspective to better comprehend disturbance consequences, and the interaction among these processes in metacommunity recovery and colonization dynamics; **the main objective of this thesis is to study the wildfire impacts on Mediterranean temporary ponds fauna and their implications on metacommunity assembly processes combining observational, experimental and simulated perspectives**. To attaint this main objective, this thesis has been structured in specific aims, which constitute the four chapters (Figure 1.1), and covers different aspects that are crucial to understand metacommunity response to disturbances.

Firstly, an observational study dealing with the impact of wildfire from a taxonomic and functional perspective was carried out with the following specific objectives:

- To study wildfire effects on Mediterranean temporary ponds assessing its impacts on environmental characteristics and macrofauna composition (*Chapter I*).

- To unravel trait selection and main metacommunity assembly forces in Mediterranean temporary ponds after a wildfire disturbance (*Chapter II*).

- Although similar studies dealing with temporary lentic systems and wildfire impacts are scarce, in a *Mediterranean context we could expect an intrinsic high resilience of the studied systems, since these habitats are often affected by wildfires and its drought recurrence implies a selection of resilient species and traits*.

Secondly, because dispersal dynamics might be crucial to better understand the post-disturbance *metacommunity rescue effects*, a field experiment was conducted with the following specific objective:

- To determine network influence on dispersal dynamics considering different dispersal abilities (i.e., strong dispersers and weak dispersers) and considering broad and small-scale perspective of the isolation gradient (*Chapter III*).

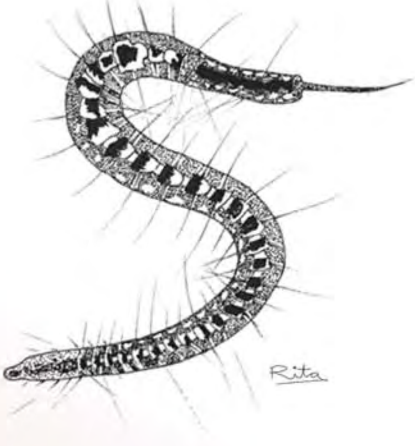
We expect that, *recolonization processes will be tightly linked with network structure and species dispersal ability (e.g., moderate active, strong active dispersers) and will modulate post-disturbance dynamics, being location within the network a key feature in favouring greater individual arrivals*.

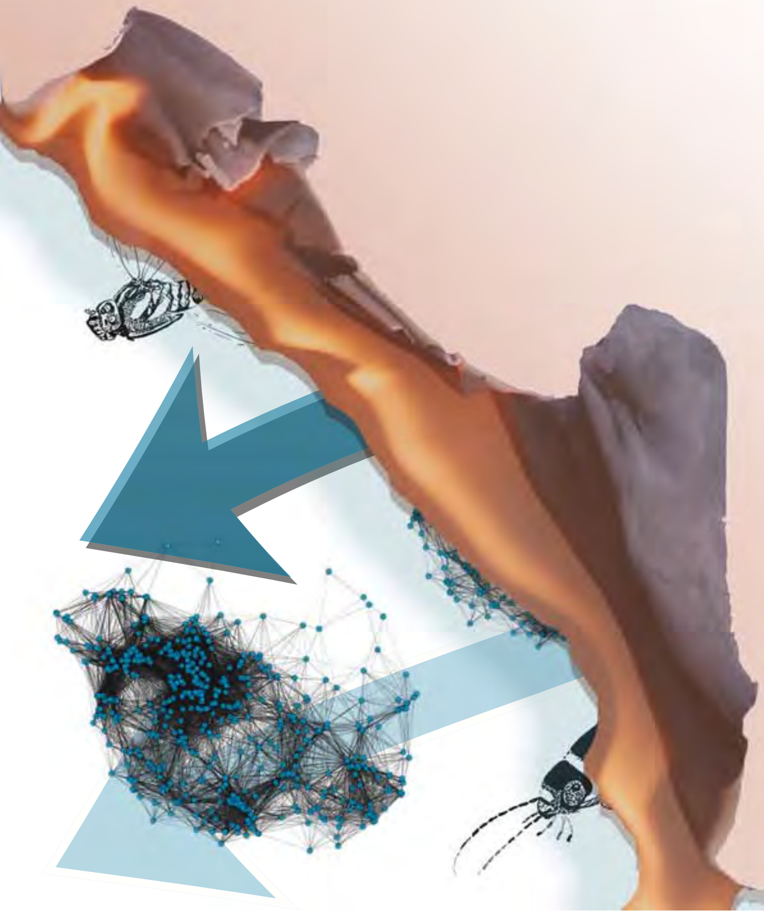
Thirdly, wildfire impacts might differ depending on its regime (intensity and area). Therefore, in this thesis a simulation study has been carried to better asses temporary ponds resilience to different wildfire regimes, with the following specific objective:

To analyse the response at metacommunity level of temporary pond communities subject to different wildfire regimes, explicitly accounting with the different landscape perception of the organisms affected (*Chapter IV*).

Our expectation is that *wildfire regime will highly determine the metacommunity rescue effect and that metacommunity response will differ according to specific species regional perspective.*

3. GENERAL METHODS





3.1 Study sites and climatic overview

The current thesis is mainly focused in a Mediterranean temporary ponds system located in the Albera region in north-eastern Iberian Peninsula (Figure 3.1), with three of four chapters using data or being based on data in relation with this region (*Chapter I, Chapter II* and *Chapter IV*). Furthermore, the *Chapter III* field experiment was carried out in another temporary pond system situated in the Pyrenees mountain range (2000 m.a.s.l.): Guils de Cerdanya (Figure 3.1). This change of location in *Chapter III* was made due to Guils de Cerdanya network configuration as it have ponds located throughout a marked central-isolation gradient, its low human pressure and the short hydroperiod length of these systems, in which dispersal plays a main role for their biotas due to the narrow time-window of suitable conditions for colonization (Wissinger et al. 2016). These factors made surveying and set up less complex and endurable along the whole hydroperiod. Each pond system is part of a greater water bodies network (Figure 3.1), which has a different structure (e.g., number of water bodies, distribution throughout the landscape, etc.), but they have in common that the ponds are mainly temporary (i.e., they are completely dry during summer). For the Albera region, ponds normally fill during autumn rains and keep water until summer beginning, although this pattern is variable according to every year rainfall (Ribera and Aguilera 1996, Font 2001, Ruhí et al. 2013b). On the other hand, Guils de Cerdanya ponds has a shorter hydroperiod that begins with snow melting, end of April, and keeps water until mid-July.

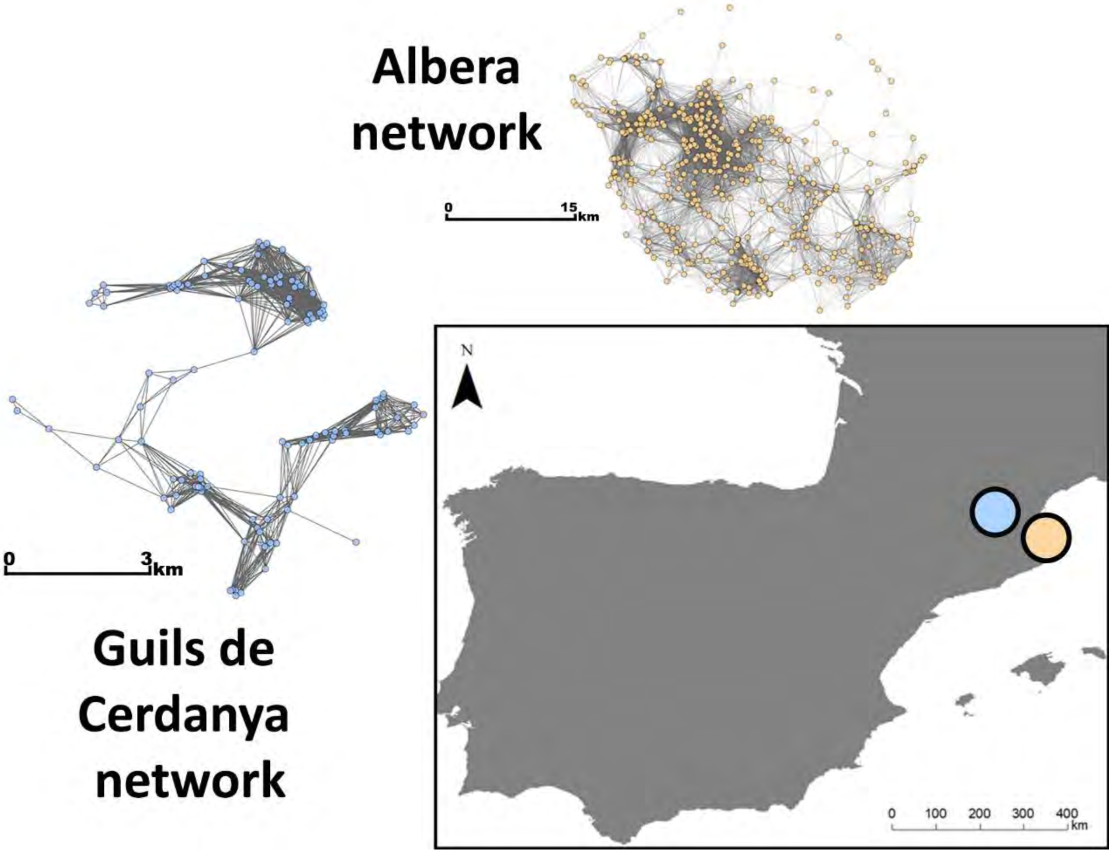


Figure 3.1: Studied pond networks location in the Iberian Peninsula. The Albera region (orange circle and network) and Guils de Cerdanya (blue circle and network). Both networks represent the region water bodies and links are based on networks percolation distances (Albera: 3849 meters and Guils de Cerdanya: 1033 meters).

The two sampled regions are considered within the Mediterranean climate type by the Catalan Meteorological Service (SMC-Meteocat; Figure 3.2). The Albera region belongs to the Northern prelitoral Mediterranean region with a mean annual precipitation ranging between 750-1000 mm with their maximums in spring and autumn (Figure 3.3A). Guils de Cerdanya belongs to the occidental Pyrenees Mediterranean climatic region with its mean annual precipitation ranging between 1000-1300 mm and their rainfall maximums in spring/summer (Figure 3.3B). Both systems are mainly rainfall feed and present a similar morphology: shallow land depressions with a sediment/rock bottom that when flooded present vegetation inside the water body (hydrophytes and helophytes). To better picture studied ponds and its general aspect see images in Supplementary S3.1 and S3.2.

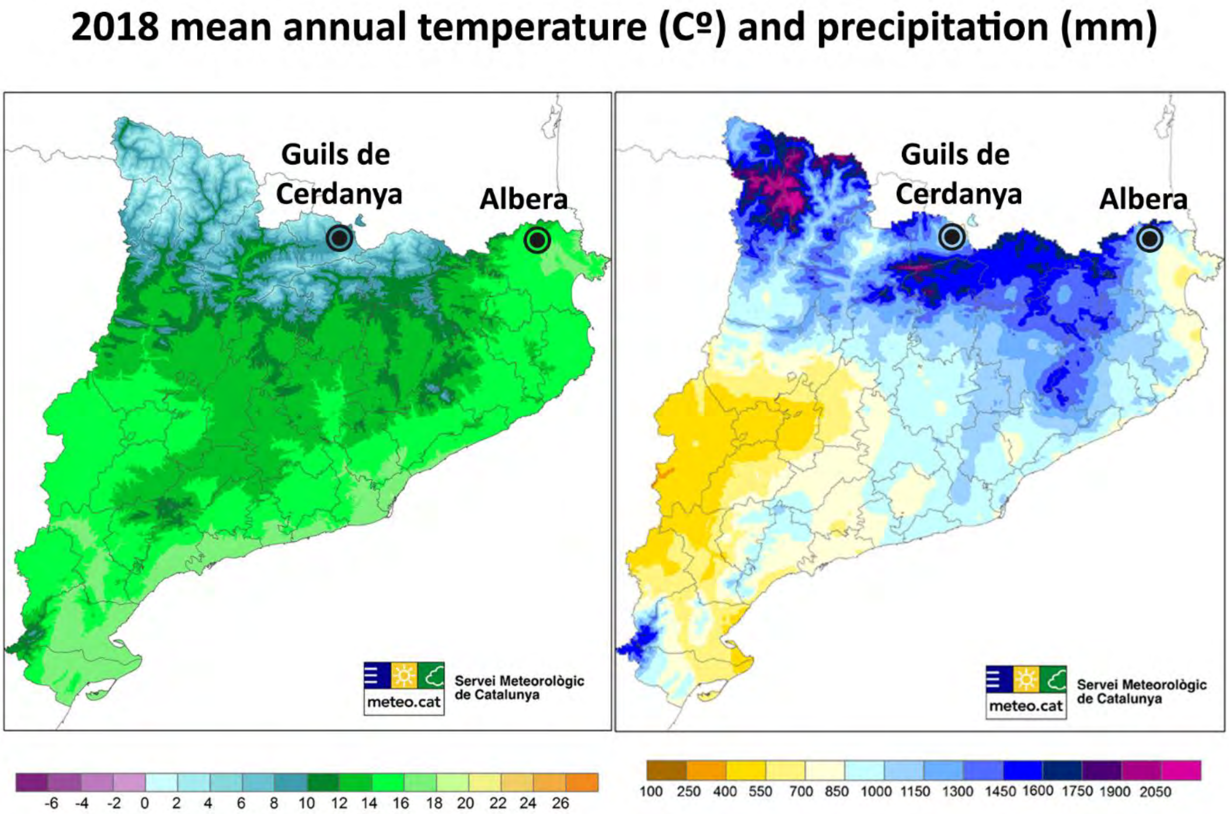
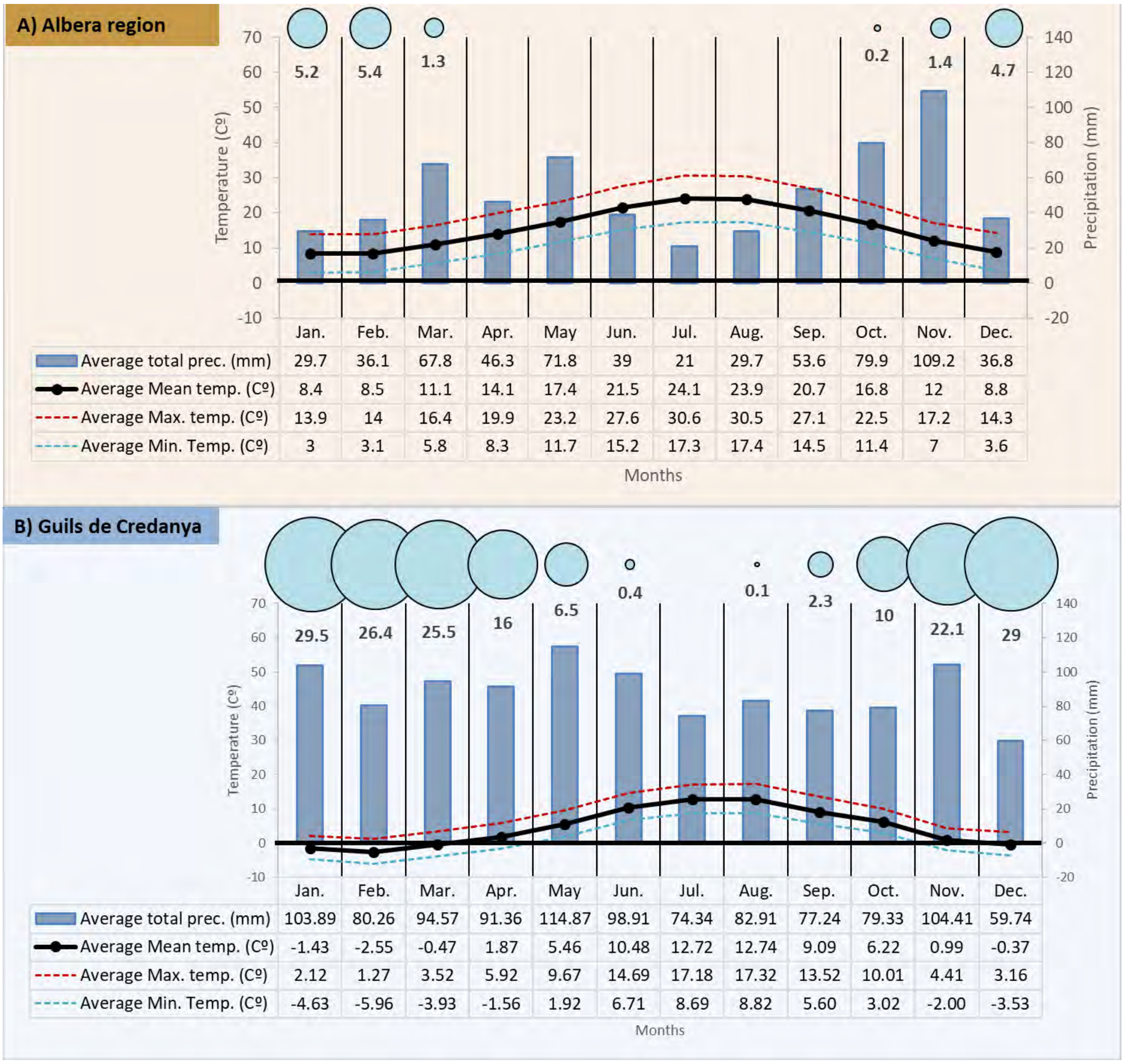


Figure 3.2: *Catalonia maps of 2018 registered mean annual temperature (left panel; ºC) and mean annual precipitation (right panel; mm) from the Catalan meteorological service indicating the two studied regions (Albera and Guils de Cerdanya) and their mean 2018 values: 14-16 ºC and 8-10 ºC, respectively; and around 850 mm and 1000 mm of precipitation, respectively.*



	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.
▬ Average total prec. (mm)	29.7	36.1	67.8	46.3	71.8	39	21	29.7	53.6	79.9	109.2	36.8
●— Average Mean temp. (Cº)	8.4	8.5	11.1	14.1	17.4	21.5	24.1	23.9	20.7	16.8	12	8.8
---- Average Max. temp. (Cº)	13.9	14	16.4	19.9	23.2	27.6	30.6	30.5	27.1	22.5	17.2	14.3
---- Average Min. Temp. (Cº)	3	3.1	5.8	8.3	11.7	15.2	17.3	17.4	14.5	11.4	7	3.6

	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.
▬ Average total prec. (mm)	103.89	80.26	94.57	91.36	114.87	98.91	74.34	82.91	77.24	79.33	104.41	59.74
●— Average Mean temp. (Cº)	-1.43	-2.55	-0.47	1.87	5.46	10.48	12.72	12.74	9.09	6.22	0.99	-0.37
---- Average Max. temp. (Cº)	2.12	1.27	3.52	5.92	9.67	14.69	17.18	17.32	13.52	10.01	4.41	3.16
---- Average Min. Temp. (Cº)	-4.63	-5.96	-3.93	-1.56	1.92	6.71	8.69	8.82	5.60	3.02	-2.00	-3.53

Figure 3.3: *Climatic diagrams of the two studied regions (panel A: Albera, and panel B: Guils de Cerdanya) based on all available data from the two nearest Catalan meteorological service stations from Espolla town for the Albera region (reference period from 2007 to 2016), and from Malniu for Guils de Cerdanya (reference period from 1999 to 2018). Blue bubbles represent the number of days per month below zero.*

3.2 Studied systems network construction

The studied networks (Figure 3.1), are formed by both temporary ponds and other water bodies (e.g., permanent ponds, cattle drinking trough, etc.). These networks were determined following the following steps (Figure 3.4): 1) Locating the initial ponds, considering the initial ponds as those from which we already had information from previous studies (Ballón et al. 2016); 2) Drawing an initial polygon encompassing all sampled ponds; 3) Calculating the maximum distance between the furthest ponds of the initial polygon; 4) Enlarging the polygon considering this maximum distance as buffer distance; and 5) identifying all water bodies within this larger polygon, using digital cartography (Google LLC 2019, Institut Cartogràfic i Geològic de Catalunya 2019).

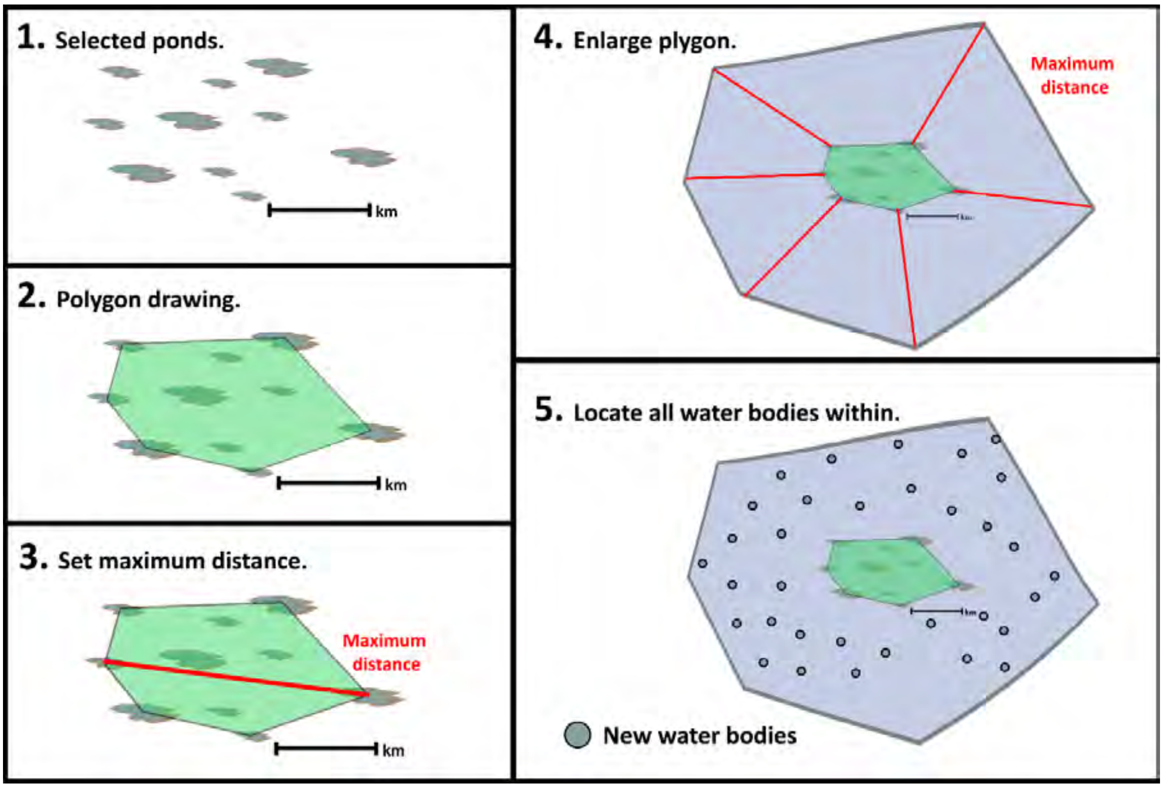


Figure 3.4: *Network determination procedure used to create the two studied networks. This procedure follows the indicated steps: 1) Locating the ten main selected ponds; 2) drawing polygon representing the shape that they conform; 3) calculating the maximum distance between two of the selected ponds; 4) enlarging the polygon according to this distance to keep ponds shape; and 5) identifying the global network including all water bodies within the new larger polygon .*

Following these steps, we could locate all water bodies (other ponds, smaller puddles, cattle drinking troughs, etc.) surrounding the initial temporary ponds and preserving the shape of the initial polygon. In the two networks, the initial ponds represented a subset of the global network centrality-isolation gradient —calculated with the closeness metric (Figure 3.5). The Albera ponds were enclosed within a dense network, were main ponds where located in a central position (Figure 3.5) whereas the

Guils de Cerdanya ponds configured a less dense network, were ponds where located along a more pronounced centrality-isolation gradient (Figure 3.5). Two extreme ponds in this isolation-centrality gradient were selected in order to compare colonization patterns (see *Chapter III*).

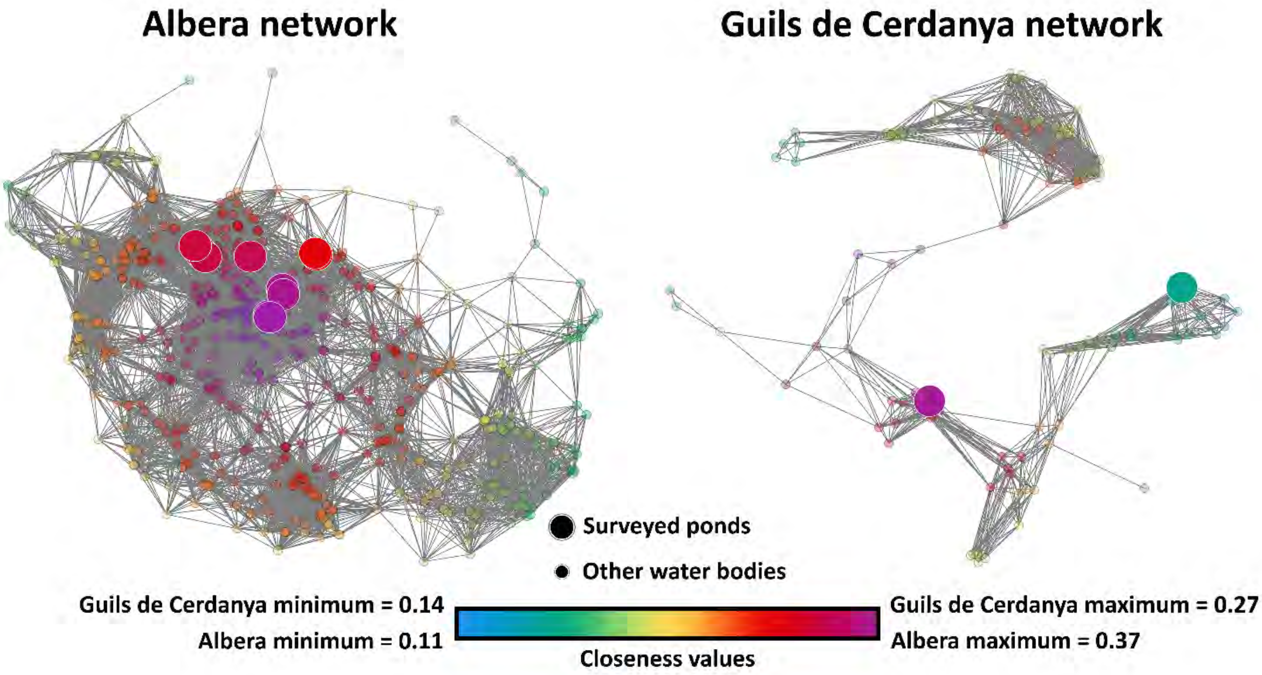


Figure 3.5: *The two determined networks coloured according to their closeness values, which indicate how central or isolated all ponds are in relation with all their neighbours. The ponds that have been surveyed along this thesis in both networks are highlighted being bigger. In the Albera network ten ponds were surveyed, whereas in the Guils de Cerdanya network two ponds were surveyed.*

3.3 Pond macroinvertebrate sampling methodology

The sampling procedure used for each pond, from both sampled systems, and during each sampling survey is a standardized methodology for wetlands ecological evaluation from the Catalan water agency (Agència Catalana de l'Aigua 2006). This procedure requires a dip-net sweep with specific measurements (Figure 3.6A) and a mesh pore size of 250 µm, basic field equipment (plastic jars, plastic trays, metal mesh of 200 µm pore size, …) and some preserving agent, in the current thesis formalin 4%. Sample collection consists in carry 20 deep sweeps evenly distributed throughout all pond mesohabitats (vegetated shore, sediment shore, submerged vegetation, etc.; Figure 3.6B). Moreover, each dip-net seep must be fast to catch most invertebrates, have an approximate half-meter distance and must not touch the ground (Figure 3.6B). Once the 20 sweeps have been done, all the collected material is stored, preserved and brought to the laboratory.

In the laboratory, the plastic jar with all sampled material is washed, removing the preserving agent, and cleaned, removing big debris (e.g., vegetal parts, coarse sediment, filamentous algae, etc.). Then, individuals are manually collected from the entire sample, in order to detect less abundant individuals that normally are also bigger. Secondly to obtain an adequate abundance estimation of all macroinvertebrates the sample is divided in several fractions using a specific dividing container in order to decrease dense samples and improve quantification (Figure 3.6C). Then, individuals from the needed fractions are collected until we have a robust number —around 300 approximately. The fraction where that number is reached is then used to extrapolate individuals sample abundance. All collected individuals are preserved in ethanol 70%. Finally, macroinvertebrates are identified and counted using a stereomicroscope and specific identification taxonomic material to obtain the total number of individuals per litre for each taxon accounting with the number of needed fractions to extrapolate the number of individuals per litre. All collected individuals were identified to species level when possible. Poorly developed individuals were identified to genus level. In the case of Turbellaria and Oligochaeta, the specimens were identified to family or subfamily level.

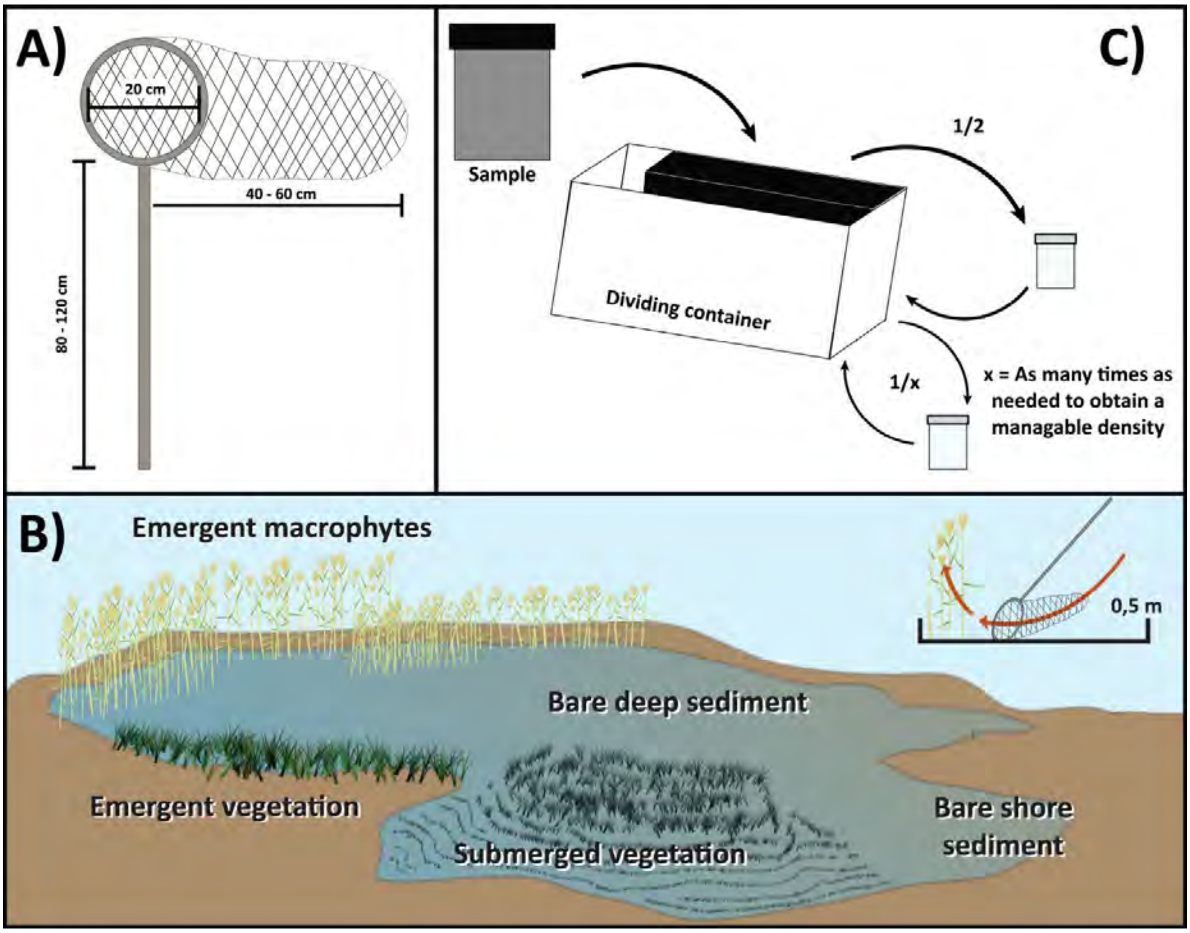


Figure 3.6: *General macrofauna sampling and samples processing materials and procedures. A: Sampling dip-net measures. B: Pond different mesohabitats and sweep sampling process. C: Sample processing and fractioning to sort, count and collect macrofauna individuals.*

3.4 The studied 2012 Empordà wildfire

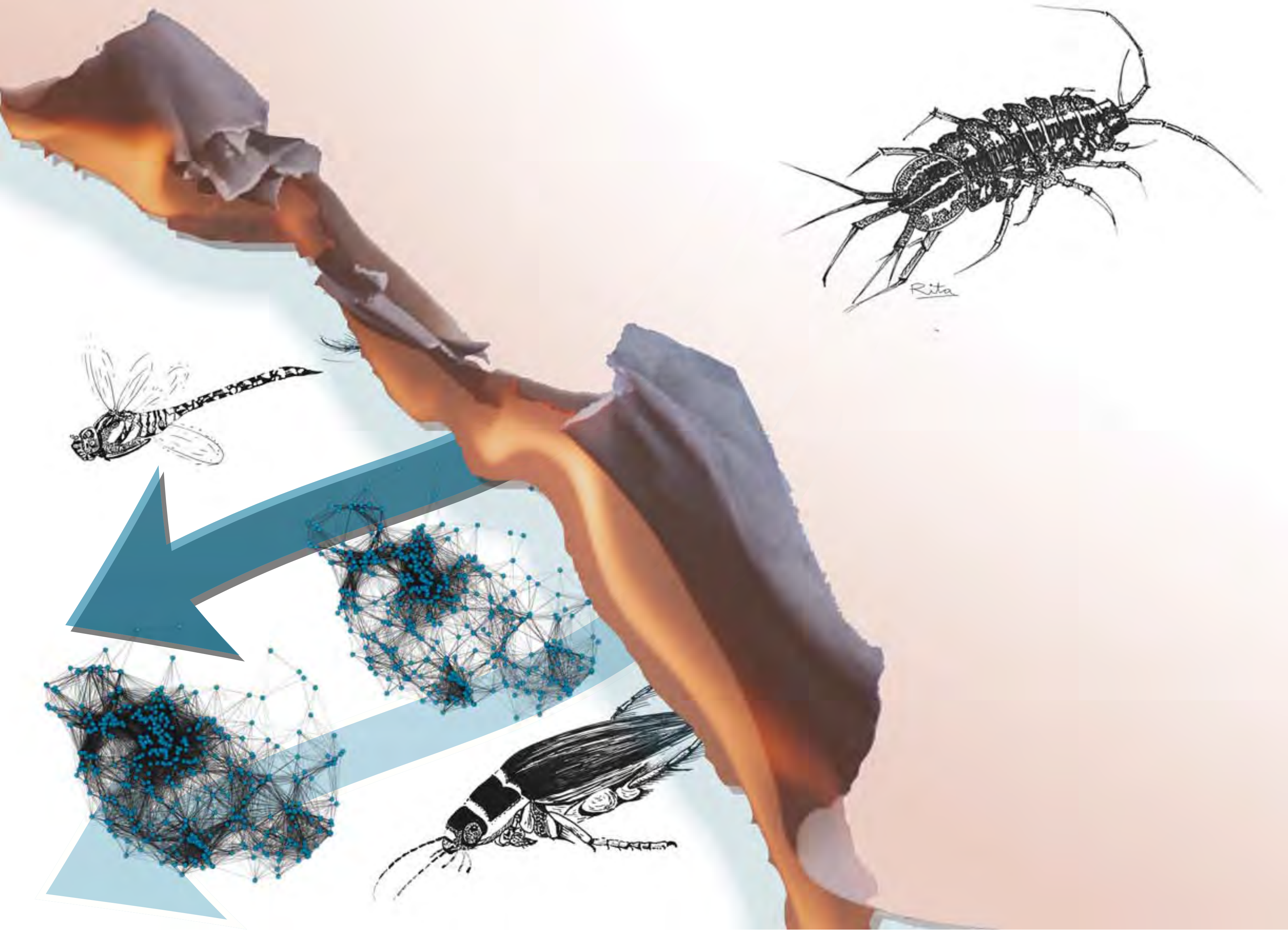
In July 22nd of 2012 near a parking in La Jonquera, in the north-east of Iberian Peninsula, a huge wildfire begun. Fostered by weather conditions it advanced at maximum velocities of 8 km/hour becoming one of the top 10 worst wildfires of the century in Spain with 10,469 hectares burned (Serveis territorials de Girona 2012, Greenpeace 2019). Within the affected area, the Albera region, in which several Mediterranean temporary ponds were found. These ponds are of great conservation importance as priority habitats by European habitats directive (Habitats 1992) and are therefore considered as a national interest landscape by the Government of Catalunya

(Departament de la Presidència 1986). In summer 2012 ponds were mostly dry following their natural hydrologic cycle and as a consequence, the wildfire passed over some pond of the protected area, completely burning the pond basin sediment and vegetation of some of them (see Supplementary S3.1). Interestingly, during spring 2012 —before the wildfire— and in the context of a different research project (RETROMED; CGL2011-23907) a sampling survey had been carried out in these Mediterranean temporary ponds, thus generating a pre-disturbance set of samples. Therefore, during the following hydroperiod —after the wildfire— from October 2012 until Summer 2013 we returned to those previously sampled ponds, some of which affected by the wildfire and some not, to survey them along the post-disturbance hydroperiod.

4. CHAPTER I

Direct and indirect impacts of wildfire on faunal communities of Mediterranean temporary ponds.

This is the peer reviewed version of the following article: ***Cunillera-Montcusí, D.,*** *Gascón, S., Tornero, I., Sala, J., Àvila, N., Quintana, X. D., & Boix, D. (2019). Direct and indirect impacts of wildfire on faunal communities of Mediterranean temporary ponds. Freshwater Biology, 64(2), 323–334,* which has been published in final form at doi: 10.1111/fwb.13219**.** This article may be used for non-commercial purposes in accordance with Wiley Terms and Conditions for Use of Self-Archived Versions.



Chapter brief summary

The Mediterranean region has historically been affected by wildfires. However, studies addressing wildfire impacts have traditionally focused first on terrestrial systems and second on lotic systems. This bias has left a gap in knowledge related to wildfire effects on some of the emblematic habitats of the Mediterranean region such as temporary ponds. We hypothesize that temporary ponds will experience indirect consequences of wildfires like those of other aquatic systems (e.g., nutrient increases or trophic alterations). However, the fact that temporary ponds are dry during the summer, when most wildfires occur, could add a new path of disturbance such as an impact on organisms aestivating in the pond sediment due to their direct burning. Based on these hypotheses, the present study analyses wildfire impacts on faunal communities adapted to temporality, focusing on species traits related to expected indirect and direct impacts (i.e., dispersal ability, life-history or feeding habits). We took advantage of a wildfire (summer 2012) that partially affected a Mediterranean temporary pond-network, comparing the environmental and the faunal community responses from before and after the wildfire, and between the burned and unburned ponds during the subsequent hydroperiod. As hypothesized, our results indicated different wildfire effects. First, changes in abundances of some trophic groups from before and after the wildfire but also strong fluctuations at the beginning of the hydroperiod between burned and unburned ponds. Second, a decrease in abundance of organisms that remain in the pond sediment during drought in burned ponds, being probably affected directly by the wildfire. Only one hydroperiod appears to be sufficient for burned ponds to recover their similarity to unburned ponds, which highlights the high resilience of these communities. Despite their resiliency these communities could become compromised in the future since global change scenarios predict increase wildfire frequency and intensity in the Mediterranean region.

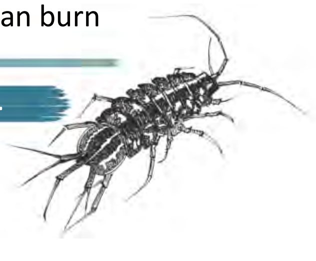
4.1 Overview

Due to global change, wildfires are projected to increase in frequency and intensity (Bowman et al. 2009, Pausas and Ribeiro 2013). Wildfire effects on terrestrial ecosystems have been widely studied from different perspectives for a long time (Wright and Bailey 1982, Díaz-Delgado et al. 2002). However, wildfire effects extend past terrestrial ecosystems and have important impacts on freshwater systems, bringing wildfire impacts into aquatic ecosystems as they drain the burned landscape (Prepas et al. 2009). Traditionally, the main consequences of wildfires in aquatic systems have been considered to be produced indirectly by wildfire impacts on terrestrial systems, but these assumptions were mostly based on perennial aquatic ecosystems (Gresswell 1999), and may not apply to temporary aquatic systems. There are three main indirect consequences: ash input, canopy loss, and basin vegetation loss (Minshall et al. 1989). Firstly, ash input increases the amount of nutrient (Spencer and Hauer 1991, Horwitz and Sommer 2005, Mast et al. 2016), sediment (Reale et al. 2015), and dissolved organic matter entering the aquatic system (even affecting water colour; Holden et al. 2012). Consequently, wildfires modify water quality, and aquatic systems become more nutrient enriched. Secondly, canopy loss increases light incidence, increasing water temperatures (Britton 1991, Cooper et al. 2015) and reducing leaf inputs into the system (Rodríguez-Lozano et al. 2015). Finally, the loss of basin vegetation cover increases runoff and coarse-sediment transport (Vaz et al. 2015), especially after post-fire storms that produce morphological alterations to habitat (i.e., Mediterranean autumn storms; (Verkaik et al. 2013, Klose et al. 2015). These wildfire effects can promote bottom-up processes in aquatic communities (Lewis et al. 2014, Rodríguez-Lozano et al. 2015). Thus, nutrient enrichment plus the increase in water temperature and light favour algal blooms (Scrimgeour et al. 2001, Spencer et al. 2003, Cooper et al. 2015). These blooms coupled with the loss of aquatic plants (de Szalay and Resh 1997, Coffman et al. 2010) change the resources available for the community.

Environmental conditions and resources altered by wildfire can benefit some faunal trophic groups, which increase their abundance, shifting community composition toward a more generalist community (Mihuc and Minshall 1995, Scrimgeour et al. 2001, Oliver et al. 2012, Araneda et al. 2013, Rugenski and Minshall 2014, Robson et al. 2018). These increases in abundance for specific groups create a bottom-up process or fire pulse (according to Malison and Baxter 2010). These pulsed effects occur in a short time lapse, while other impacts, more linked to vegetation loss and runoff increase can last for years (Minshall et al. 2001). However, a short-term pulse does not always occur (Britton 1991, Batzer 2013); if post-fire rains are particularly strong (100-year flood), they can compromise community recovery (Vieira et al. 2011). Furthermore, low post-fire rainfall and high fire recurrence can also decrease primary production and invertebrate biomass in lotic systems (Whitney et al. 2015). Therefore, depending on wildfire intensity and on immediate post-fire conditions (e.g., storms), the faunal community response may change drastically (Jackson et al. 2015, Robson et al. 2018).

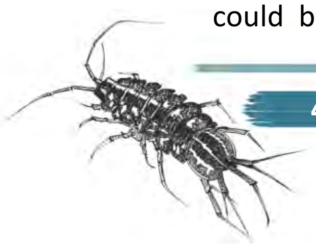
Most research on aquatic systems and wildfires is focused on lotic habitats while lentic habitats, especially temporary ponds and wetlands, have received less attention (Bixby et al. 2015). Temporary pond systems are commonly found in more arid regions such as the Mediterranean, where wildfires are also frequent due to climatic characteristics (Pausas and Vallejo 1999, Pausas 2004, Le Page et al. 2008). Although found in a fire-prone region and despite their high biodiversity value and protected status (Habitats 1992), no research has been conducted on the effects of wildfire on Mediterranean temporary ponds. As a result, how communities in these ecosystems cope with wildfires remains poorly understood.

By coupling information on the effects of wildfire on aquatic systems with the characteristics and functions specific to Mediterranean temporary ponds, we expect different consequences of wildfire for these temporary systems compared to perennial or lotic systems. Firstly, we expected direct effects on the biota since wildfires mainly occur in the summer in Mediterranean areas, when temporary ponds are usually dry. Therefore, unlike permanent ponds or lakes, Mediterranean temporary ponds can burn



completely, including pond sediment and vegetation. This could compromise the egg bank and hatching success during inundation after wildfire (Chittapun 2011). Secondly, we expected indirect effects (like changes in habitat structure arising from vegetation loss or high algal biomass) to limit the arrival of active dispersers, compromising community composition after pond inundation. Finally, as in permanent lentic systems, we expect other indirect wildfire impacts related to increased nutrient input. As ponds are endorheic, ash will accumulate in their basins after the first rains, increasing nutrient concentration in the water. Consequently, we also expect increased algal growth resulting in bottom-up processes that could also change community composition. Thus, community composition after wildfire in temporary ponds may be the result of direct processes, caused by direct burning of organisms or their resting structures, and indirect processes, such as habitat changes and bottom-up processes.

The present study analyses these predicted effects of wildfire on a set of temporary ponds that are part of a larger Mediterranean pond-network that was partially affected by a natural wildfire in the summer of 2012. At the time of the wildfire, most of the temporary ponds were already dry, and the wildfire burned the entire pond basins including the pond sediment. However, due to strong winds, wildfire severity varied, leaving two types of affected ponds. Ponds with the whole pond sediment completely burned, hereafter high intensity burned ponds (HB) and ponds with just some areas of the pond sediment burned, hereafter low intensity burned ponds (LB). Fortunately, pre-disturbance information on environmental conditions and aquatic community composition was available for all ponds. Thus, as we had pre-disturbance values and ponds unburned after the wildfire (a multiple before-after, control-impact design, MBACI), we could discriminate which ecosystem changes were caused by wildfire and which were the result of natural variability. In terms of environmental conditions and based on the likely scenario of wildfire consequences on Mediterranean temporary ponds, we expected (1) an increased nutrient content in the burnt ponds and (2) an increase in primary producer chlorophyll-*a* content. For faunal community composition, we expected burnt and unburnt communities to differ. This difference could be produced by both the effects of direct burning, causing (3) a decreased



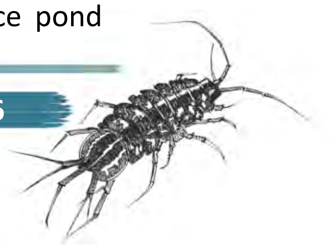
abundance of organisms that remain in pond sediment in the dry-phase; and by indirect effects of wildfire, such as habitat and trophic changes that may (4) decrease the abundance of active dispersers arriving in fire-affected ponds, and (5) shift faunal community composition toward species with more generalist strategies following a bottom-up process. Finally, we expected that all these consequences will be greater in high intensity burned ponds (HB) than in low intensity burned ponds (LB).

4.2 Methods

Study site

The studied pond-network is located in the northeastern Iberian Peninsula, in the lowlands of the Albera Mountain Range (Alt Empordà region; Figure 4.1). This pond-network has more than thirty Mediterranean temporary ponds that usually flood with rainwater during the wet seasons (especially in the autumn and/or spring). In July 2012, an intense wildfire burned 13,000 ha of this region, and the pond-network was partly affected (Figure 4.1). Some ponds of the network remained unburned (U), allowing for comparison among these unburned ponds and those exposed to low (LB) and high (HB) wildfire intensities. To see pond typologies pictures, and visual differences between severely burned ponds and less severely burned ponds, see Supplementary S4.1.

Pre-fire data were available from the year before the wildfire. Thus, this study focused on 10 ponds of the Albera network that were sampled before (between March and April 2012) and after (from December 2012 to July 2013) the wildfire. Of the 10 sampled ponds, 3 experienced a high intensity wildfire, 3 experienced a low intensity wildfire and 4 remained unburned (Figure 4.1). Therefore, we had a multiple BACI design (MBACI), having sampling survey from several control and impact ponds from before (1 sampling survey) and after (5 sampling surveys) the disturbance (Downes et al. 2002). After the wildfire, sampling surveys were conducted every two months since pond



flooding (starting in December 2012 and ending in July 2013), covering the first hydroperiod after the wildfire thus allowing us to analyse short-term wildfire effects. All ponds were sampled in each survey, except in February and July 2013, when a few ponds were already dry, due to naturally variable hydroperiods.

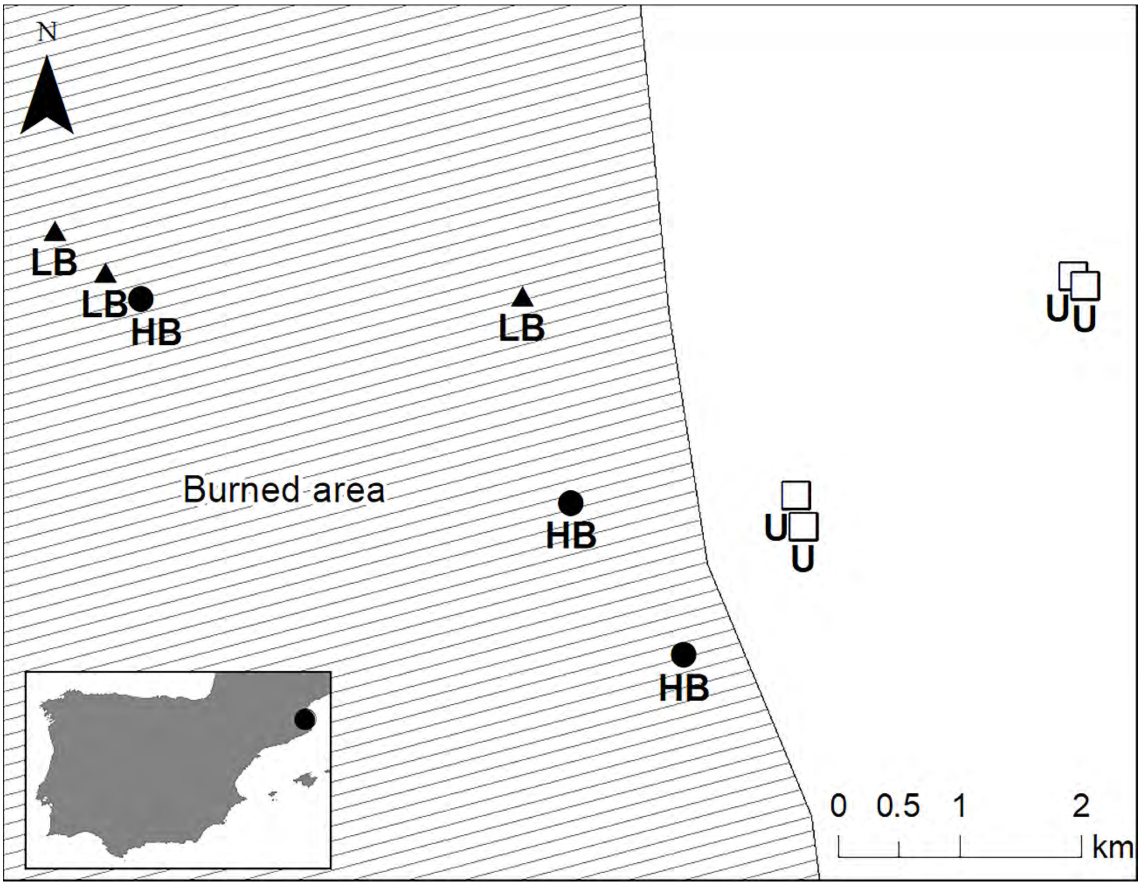


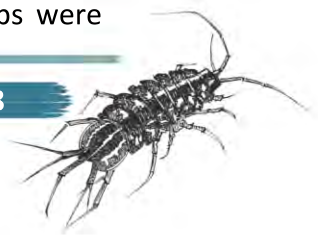
Figure 4.1: *Study area in northeastern Iberian Peninsula. The burned area represented by the dashed surface. The high intensity burned ponds (HB) are in circles, the low intensity burned ponds (LB) are in triangles, and the unburned ponds (U) are in squares.*

Environmental variables

We measured physical and chemical variables to detect changes caused by wildfire. We measured variables such as pond maximum depth (Max Z), dissolved oxygen concentration (O_2), conductivity (EC25) and pH in situ (model HACH HQ30d). Also, we measured dissolved nutrients such as soluble reactive phosphorus (SRP), inorganic nitrogen (ammonia, nitrite and nitrate, DIN), total nitrogen (TN), total phosphorus (TP) and NP Ratio (a nutrient limitation indicator). In relation to carbon composition, we measured total inorganic carbon (TIC), total organic carbon (TOC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC) and fulvic acid percentage (Ful.Ac.%). To measure these environmental variables, we followed the methodology described in Ballón et al. (2016). Additionally, primary producer biomass and chlorophyll-*a* content was measured. Macrophyte biomass (Mac; g/m^2) was estimated from the dominant vegetation as the mean dry weight of three circular replicates of 50.26 cm^2 that were collected at random from each pond. Filamentous algae biomass (Fil; g/cm^2) was also estimated as the mean dry weight of three circular replicates of 171.56 cm^2 that were collected at random from each pond. Phytoplankton (Phy) chlorophyll-*a* content (µg/l) was extracted by filtering water samples (Whatman GF/F filters) and using 90% acetone following Talling and Driver (1963). Chlorophyll-*a* analyses were carried out following the adaptation of Àvila et al. (2016) of the method described by Zapata et al. (2000).

Faunal sampling and functional traits

Integrated samples of aquatic macrofauna (both amphibians and macroinvertebrates) were taken. As they are protected, amphibian adults were identified *in situ* and returned to the pond. Amphibian larvae and macroinvertebrates were rapidly preserved *in situ* in 4% formaldehyde for subsequent identification in the laboratory. Both macroinvertebrates and amphibians were sorted, counted and identified to species level when possible. Poorly developed individuals and immature stages were identified to genus level. In the case of Turbellaria, Chironomidae and Oligochaeta the specimens were identified to family or subfamily level. For the functional traits, we classified all taxa considering their dispersal ability, life-history and feeding traits. Dispersal ability groups (DAG) were based on Heino (2013). Heino's study proposed 4 groups based on species overland dispersal abilities and related to their landscape perceptions, from passive dispersers with aquatic adults (DAG1) to strong aerial dispersers with flying adults (DAG4) and between them weak (DAG2) and moderate (DAG3) active dispersers with flying adults. In this study, we added a new group for organisms with aquatic larvae and terrestrial dispersal adults (i.e., amphibians) as DAG5. Life history groups were



based on the groups proposed by Wiggins et al. (1980) that include dry-phase residents (LHG1), dry-phase residents & spring recruits (LHG2), dry-phase residents & summer recruits (LHG3) and non-dry-phase residents & spring migrants (LHG4). Finally, feeding groups were based on Merritt and Cummins (1996) and Tachet (2000) with 7 feeding categories: collectors, filterers, scrapers, shredders, piercers, parasites and predators.

Data analysis

In this study we accounted for 3 different sources of variability: the burn status and two different temporal patterns. Burn status corresponds to pond typologies according to wildfire impact: HB, LB and U. The first temporal pattern was the strict before-after control-impact comparison, which separates the effect of the wildfire from the effects of other temporal changes in the pre to post-fire period, hereafter referred as pre/post-fire comparisons. The second temporal pattern was obtained only considering the sampling surveys done after the wildfire. Its inclusion allowed us to analyze possible differences in temporal patterns among ponds with different burn status. Thereby measuring recovery, in the sense that unburned ponds were seen as the reference condition.

We conducted a non-metric multidimensional scaling (nMDS) ordination, using the Bray-Curtis similarity index, and used PERMANOVA to test the wildfire effect on faunal assemblages. We used a nested PERMANOVA design with two main factors: burn status and before-after the wildfire, nesting sampling survey within before-after to account for the post-fire trend. When necessary, pairwise comparisons were carried out between control (U) and affected ponds (HB and LB).

In order to detect if there were initial differences among pond types prior to disturbance, we used analysis of variance for both environmental variables and faunal traits data. Afterward, for the same data, we compared burn status and pre/post-fire conditions using a multiple before-after control-impact design or MBACI. To have a

balanced MBACI comparison (Downes et al. 2002), we tested for wildfire impacts before and after the impact but only including sampling campaigns from the same month (i.e. March-April 2012 vs. April 2013). We used a mixed model, including pond identity as random in the analysis. Then, as we had a post-impact monitoring of the same ponds, we tested for significant differences in burn status and the post-fire trend with a repeated measures ANOVA (RMA) using a mixed model, again including pond identity as random in the analysis. Prior to these analyses, environmental variables were log-transformed, except pH, the number of individuals of DAGs and LHGs were fourth root transformed and functional feeding groups log-transformed. In both analyses, MBACI and RMA, the interaction between burn status and pre/post-fire comparisons (MBACI) or burn status and post-fire trend (RMA) were considered to identify temporal differences among pond typologies. To assess possible differences among pond typologies according to their burn status (comparing control vs. affected ponds) Dunnet's test was used to correct the expected inflated error type I due to multiple testing.

All analyses were conducted using *nlme, lsmeans* and *vegan* packages with R-3.4.1 and Primer 6 and PERMANOVA+ program. The use of an *LME* model is recommended in BACI analyses, because the underlying assumptions of the model are likely to be satisfied, and the interpretation of the estimated parameters is straightforward (McDonald et al. 2000).

4.3. Results

Environmental variables

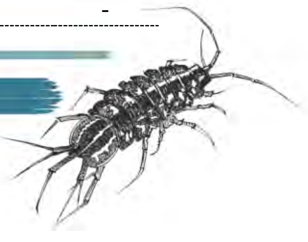
Burned (both high and low intensity) and unburned ponds had similar environmental conditions before the wildfire (Supplementary S4.2 and S4.3). Only maximum depth

differed among ponds ($F_{2,7}$=5.77, p-value=0.03), because unburned and low intensity ponds were deeper than high intensity ponds.

When comparing before and after (MBACI), the only differences detected in relation to burn status was maximum depth (Table 4.1), as expected because high intensity ponds were shallower than the others. Maximum depth differences were maintained during the following hydroperiod (Supplementary S4.4). The MBACI detected pre/post-fire differences for most environmental variables (Table 4.1). Strong temporal differences in most environmental variables were also detected for the post-fire trend with the RMA (Table 4.1). The interaction between burn status and post-fire trend (Table 4.1) showed that phytoplankton peaked earlier in the high intensity ponds (December 2012) than in the unburned ponds, where it continued increasing throughout the hydroperiod (Figure 4.2a). Conversely, macrophytes increased earlier in the unburned and low intensity ponds than in the high intensity ponds (Figure 4.2b). O_2 concentration constantly decreased throughout the hydroperiod leaving unburned ponds slightly higher concentrations at the end of the hydroperiod (Figure 4.2c). Nutrients presented different patterns across the hydroperiod: SRP strongly increased at the end of the hydroperiod mainly in low and high intensity ponds (Figure 4.2d), whereas total nitrogen differed initially (highly burned ponds had slightly higher values), followed by a strong decrease in mid-hydroperiod for all ponds. Finally leaving unburned ponds with slightly higher total nitrogen concentrations toward the end of the hydroperiod (Figure 4.2e). For detailed information on the environmental variable values (averages and standard deviations) before and after wildfire samples, see Supplementary S4.5.

Table 4.1: *Environmental variables results for multiple before-after-control-impact (MBACI) and repeated measures ANOVA (RMA). The included sources of variation were burned status (high intensity burned ponds, low intensity burned ponds and unburned ponds), pre/post-fire comparisons (before and after), post-fire trends (only surveys after the fire) and the interaction between burned status and both temporal patterns. Acronyms are Max Z (maximum pond depth), EC25 (conductivity), O2 (dissolved oxygen), SRP (soluble reactive phosphorus concentration), DIN (dissolved inorganic nitrogen), TN (total nitrogen), TP (total phosphorus), NP Ratio (nitrogen-phosphate ratio), DIC (dissolved inorganic carbon), DOC (dissolved organic carbon), TIC (total inorganic carbon), TOC (total organic carbon), Ful.Ac.% (percentage of fulvic acids), Phy (chlorophyll-a content of phytoplankton),Fil (biomass of filamentous algae) and Mac (biomass of macrophytes). Significant results are indicated (p<0.05) in gray shading and bold.*

	MBACI	**Burn status**		**Pre/post-fire**		**Interaction**	
		χ^2_2	**p-value**	χ^2_1	**p-value**	χ^2_2	**p-value**
Environmental variables	Max Z	**11.06**	**<0.001**	3.60	0.06	**7.94**	**0.02**
	EC25	2.48	0.29	0.76	0.38	-	-
	pH	0.01	1.00	3.79	0.05	-	-
	O2	1.71	0.42	**9.99**	**<0.001**	-	-
	SRP	2.98	0.23	0.00	0.99	-	-
	DIN	0.88	0.65	**4.30**	**0.03**	-	-
	TN	0.86	0.65	**12.98**	**<0.001**	-	-
	TP	0.19	0.91	0.52	0.47	-	-
	NP Ratio	0.70	0.70	2.94	0.08	-	-
	DIC	2.26	0.32	**6.88**	**<0.001**	-	-
	DOC	1.06	0.59	**53.11**	**<0.001**	-	-
	TIC	2.34	0.31	1.33	0.25	-	-
	TOC	1.10	0.58	**35.89**	**<0.001**	-	-
	Ful.Ac.%	1.10	0.58	**5.66**	**0.02**	-	-
	Phy	2.88	0.24	2.84	0.09	-	-
	Fil	1.88	0.38	**4.60**	**0.03**	-	-
	Mac	0.99	0.60	0.97	0.32	-	-
	RMA	**Burn status**		**Post-fire trends**		**Interaction**	
		χ^2_2	**p-value**	χ^2_4	**p-value**	χ^2_8	**p-value**
	Max Z	**7.00**	**0.03**	**36.47**	**<0.001**	-	-
	EC25	1.45	0.48	**26.10**	**<0.001**	-	-
	pH	0.75	0.69	**36.48**	**<0.001**	-	-
	O2	1.12	0.57	**39.56**	**<0.001**	**16.47**	**0.03**
	SRP	3.61	0.16	**22.08**	**<0.001**	**16.18**	**0.04**
	DIN	2.10	0.35	2.84	0.59	-	-



RMA	Burn status		Post-fire trends		Interaction	
	χ^2_2	p-value	χ^2_4	p-value	χ^2_8	p-value
NT	0.30	0.86	**24.05**	**<0.001**	**16.79**	**0.03**
PT	0.50	0.78	**16.16**	**<0.001**	-	-
NP Ratio	1.33	0.51	4.18	0.38	-	-
DIC	2.63	0.27	**39.91**	**<0.001**	-	-
DOC	1.58	0.45	**21.96**	**<0.001**	-	-
TIC	2.37	0.31	**38.80**	**<0.001**	-	-
TOC	1.79	0.41	**18.62**	**<0.001**	-	-
Ful.Ac.%	5.99	0.05	**17.67**	**<0.001**	-	-
Phy	5.34	0.07	**12.45**	**<0.001**	**44.93**	**<0.001**
Fil	0.14	0.92	7.23	0.12	-	-
Mac	4.44	0.10	**67.73**	**<0.001**	**20.11**	**<0.001**

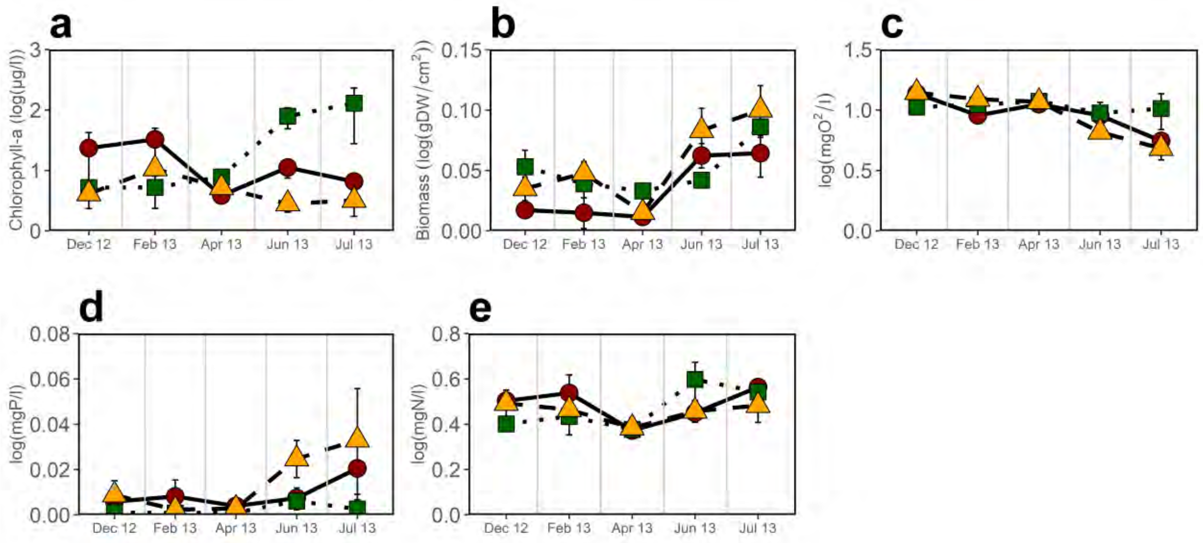


Figure 4.2: *Changes in (a) phytoplankton chlorophyll-a, (b) macrophyte biomass, (c) O2 concentration, (d) soluble reactive phosphorus, and (e) total nitrogen along the post-fire hydroperiod for the high intensity ponds (red circles with solid line), low intensity ponds (orange triangles with dashed line) and unburned ponds (green squares with dotted line). Error bars indicate the standard error of the mean.*

Similarly, PERMANOVA analysis showed strong differentiation among sampling surveys (Pseudo-F$_{4,36}$= 4.43, p-value<0.001, Figure 4.3b) and differences among ponds of different burnt status (Pseudo-F$_{2,36}$= 2.06, p-value= 0.01, Figure 4.3a); pairwise comparisons showed no differences between unburned and low intensity ponds (HB-U: t$_{2,36}$=1.17, p-value=0.23 and LB-U t$_{2,36}$=1.45, p-value=0.07). PERMANOVA did not detect significant differences for pre/post-fire comparisons (i.e. differences between before and after wildfire: Pseudo-F$_{1,36}$= 0.86, p-value= 0.33), nor for the interaction between burn status and pre/post-fire comparison (Pseudo-F$_{2,36}$= 1.15, p-value= 0.28), nor for the interaction between burn status and the post-fire trend (Pseudo-F$_{8,36}$= 0.63, p-value=0.99).

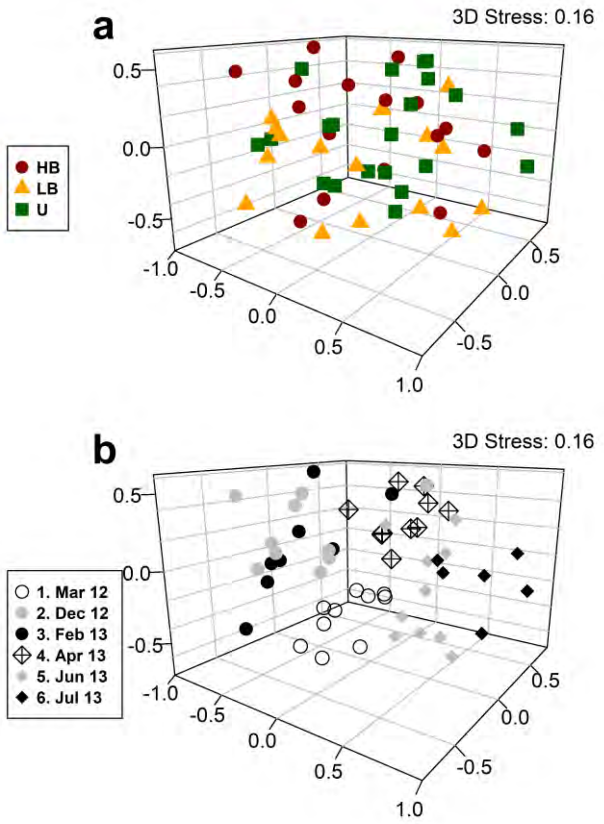
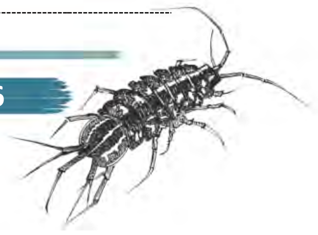


Figure 4.3: *Ordination 3D plots representing change in macrofaunal assemblage composition according to pond burn status (a) and different sampling surveys (b). Panel a high intensity ponds (red circles), low intensity ponds (orange triangles), and unburned ponds (green squares). Panel b sampling surveys from before the wildfire: March 2012 (white circles), and from after the wildfire: December 2012 (grey circles), February 2013 (black circles), April 2013 (cross diamonds), June 2013 (grey diamonds), and July 2013 (black diamonds).*

When analysing the functional groups, we did not find any significant differences among pond typologies before the wildfire (Supplementary S4.2, S4.3). However, when compared with the after samples we again detected strong temporal patterns, including pre/post-fire differences (using MBACI) and a post-fire trend (by means of RMA) for most functional groups (Table 4.2). Moreover, there were also differences among pond burn status. The MBACI analysis (Table 4.2) showed that there were fewer collectors in LB than in unburned ponds (LB-U z-value$_7$=-2.80, p-value<0.001). Furthermore, the interaction between burn status and pre/post-fire (Table 4.2) also indicated that weak active dispersers, terrestrial active dispersers, dry-phase residents & spring recruits all increased their numbers after the wildfire (Figure 4.4). When we only considered the post-disturbance year (RMA; Table 4.2) we detected the same differences in burn status for collectors, but also in functional groups related to passive dispersal ability and aestivation during drought in the pond sediment (Figure 4.5). Pairwise comparisons indicated lower abundances of these organisms (HB-U z-value$_7$=-2.31, p-value=0.03) and of passive dispersers (HB-U z-value$_7$=-2.20, p-value=0.05). Finally, we detected significant differences in the interaction between burn status and post-fire trends for some feeding groups such as shredders, scrapers and piercers (Table 4.2): each showed different trends in relation to wildfire disturbance (Figure 4.6). Piercer abundance increased in high intensity burned ponds toward the end of the hydroperiod (Figure 4.6a). Scrapers also increased at the end of the hydroperiod, but similarly in all pond typologies (Figure 4.6b). In contrast, shredders had higher abundances in high intensity ponds at the beginning of the hydroperiod and then decreased across the hydroperiod (Figure 4.6c). For detailed information on the functional trait values (averages and standard deviations) before and after the wildfire, see Supplementary S4.6.

Table 4.2: *Functional traits data results for multiple before-after-control-impact (MBACI) and repeated measures ANOVA (RMA). The included sources of variation were burned status (high intensity burned ponds, low intensity burned ponds and unburned ponds), pre/post-fire comparisons (before and after), post-fire trends (only surveys after the fire) and the interaction between burned status and both temporal patterns. Acronyms are passive dispersers (DAG1), weak aerial active dispersers (DAG2), aerial active dispersers (DAG3), strong aerial active dispersers (DAG4), terrestrial dispersers (DAG5), dry-phase residents (LHG1), dry-phase residents & spring recruits (LHG2), dry-phase residents & summer recruits (LHG3) and non-dry-phase spring migrants (LHG4). Significant factors (p<0.05) in gray shading and bold.*

	MBACI	**Burn status**		**Pre/post-fire**		**Interaction**	
		χ^2_2	**p-value**	**χ^2_1**	**p-value**	**χ^2_2**	**p-value**
Faunal traits	DAG1	2.37	0.31	0.41	0.52	-	-
	DAG2	0.08	0.96	**21.46**	**<0.001**	**6.02**	**0.04**
	DAG3	2.26	0.32	**9.81**	**<0.001**	-	-
	DAG4	0.51	0.77	1.70	0.19	-	-
	DAG5	2.46	0.29	**228.68**	**<0.001**	**7.08**	**0.03**
	LHG1	2.09	0.35	0.36	0.55	-	-
	LHG2	0.87	0.65	**28.57**	**<0.001**	**6.51**	**0.04**
	LHG3	0.12	0.94	**4.09**	**0.04**	-	-
	LHG4	0.07	0.96	2.63	0.10	-	-
	Collectors	**8.42**	**0.01**	**5.39**	**0.02**	-	-
	Shredders	1.72	0.42	**4.77**	**0.02**	-	-
	Scrapers	4.77	0.09	0.39	0.53	-	-
	Filterers	0.51	0.77	3.06	0.08	-	-
	Piercers	1.39	0.50	0.09	0.77	-	-
	Predators	1.22	0.54	2.57	0.11	-	-
	Parasites	1.24	0.54	**19.50**	**<0.001**	-	-
	RMA	**Burn status**		**Post-fire trends**		**Interaction**	
		χ^2_2	**p-value**	**χ^2_4**	**p-value**	**χ^2_8**	**p-value**
	DAG1	**8.24**	**0.01**	5.91	0.21	-	-
	DAG2	0.69	0.71	**20.07**	**<0.001**	-	-
	DAG3	0.27	0.87	**76.68**	**<0.001**	-	-
	DAG4	2.88	0.24	**67.79**	**<0.001**	-	-
	DAG5	0.01	1.00	**184.54**	**<0.001**	-	-
	LHG1	**9.10**	**0.01**	5.65	0.23	-	-



RMA	***Burn status***		***Post-fire trends***		***Interaction***	
	χ^2_2	p-value	χ^2_4	p-value	χ^2_8	p-value
LHG2	0.76	0.68	**29.55**	**<0.001**	-	-
LHG3	0.48	0.79	**40.26**	**<0.001**	-	-
LHG4	2.15	0.34	**120.15**	**<0.001**	-	-
Collectors	**9.19**	**0.01**	3.69	0.45	-	-
Shredders	3.89	0.14	**32.65**	**<0.001**	**23.52**	**<0.001**
Scrapers	2.72	0.26	**35.32**	**<0.001**	**15.87**	**0.04**
Filterers	0.00	1.00	**31.78**	**<0.001**	-	-
Piercers	2.18	0.34	**37.58**	**<0.001**	**27.96**	**<0.001**
Predators	0.25	0.88	**23.67**	**<0.001**	-	-
Parasites	2.55	0.28	**17.45**	**<0.001**	-	-

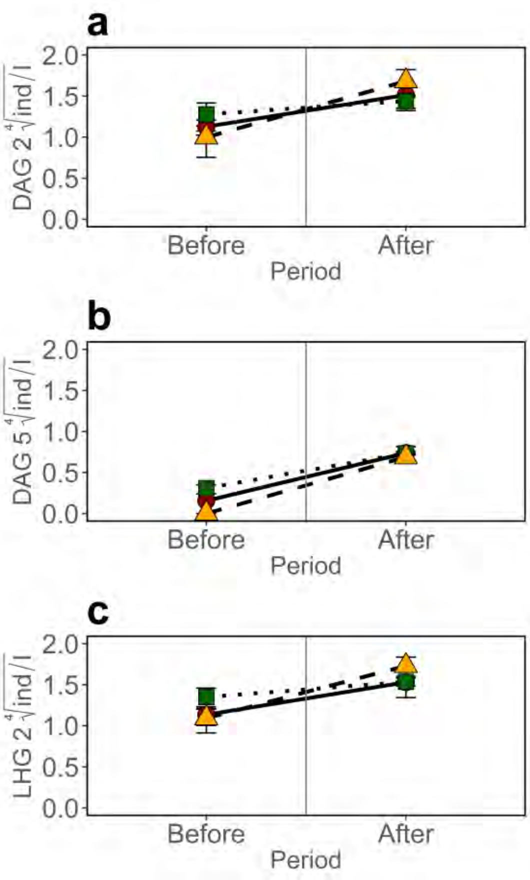


Figure 4.4: *Changes in abundance of weak active dispersers (DAG2; a), terrestrial active dispersers (DAG5; b), and dry-phase residents & spring recruits (LHG2; c) between before and after for the high intensity ponds (red circles with solid line), low intensity ponds (orange triangles with dashed line) and unburned ponds (green squares with dotted line). Error bars indicate the standard error of the mean.*

4.4. Discussion

As expected, wildfire affected Mediterranean temporary ponds, although it did not compromise post-disturbance recovery. Comparisons indicated a strong temporal variability between before and after the wildfire, but also throughout the post-fire hydroperiod. Strong patterns of temporal change are well known in temporary lentic waterbodies where faunal communities vary across the hydroperiod following seasonal changes or depending on time of inundation (Williams 2006, Boulton et al. 2014, Boix et al. 2016). Direct and indirect impacts of wildfire were also detected, demonstrating its effects in temporary systems. However, the indirect impacts (e.g. increase in nutrients, phytoplankton and some feeding groups) disappeared rapidly, showing the strong resilience of these temporary systems to wildfire. Only animals that aestivate in pond sediments were negatively affected throughout the whole post-disturbance hydroperiod (having lower abundance values in highly burned ponds). As ponds dry during summer, life-stages remaining in the sediment are vulnerable to direct burning by wildfire. Therefore, direct impacts appear to be the strongest wildfire effects on Mediterranean temporary pond communities.

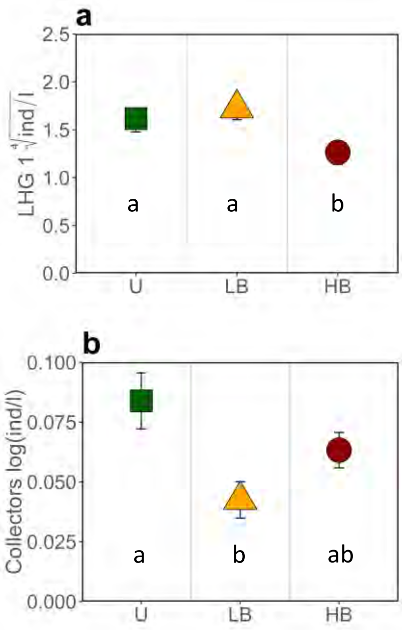


Figure 4.5: *Wildfire effect in abundance of dry-phase residents & spring recruits (LHG1; a) and collectors (b) in the post-fire year for high intensity ponds (red circles), low intensity ponds (orange triangles) and unburned ponds (green squares). Error bars indicate the standard error of the mean. Lowercase letters indicate statistical differences among treatments.*

Environmental variables

Wildfire did influence environmental characteristics of ponds, although not as strongly as first hypothesized. Like other studies in peatland areas (Holden et al. 2012), we did not observe any alteration in inorganic or organic carbon in affected ponds. Changes in the forms of carbon present were mainly detected across the hydroperiod and occurred regardless of pond burn status. However, as expected, we observed different patterns in nutrient concentrations (total nitrogen and SRP) across the post-disturbance hydroperiod. Burnt ponds slightly increased their nutrient concentration but, over time, ponds became more similar in their nutrient content except for an increase in SRP in low intensity ponds at the end of the hydroperiod. Nitrogen (ammonia and nitrate) fluctuations have been linked to smoke diffusion into water due to nitrogen volatilization, while SRP concentrations have been linked to ash accumulation and leaching (Spencer and Hauer 1991, Minshall et al. 1997, Smith et al. 2001, Earl and Blinn 2003). After the wildfire, ponds remained dry for some months, which could have decreased the amount of ash accumulated due to wind dispersal and therefore reduced nutrient inputs to the ponds once flooded.

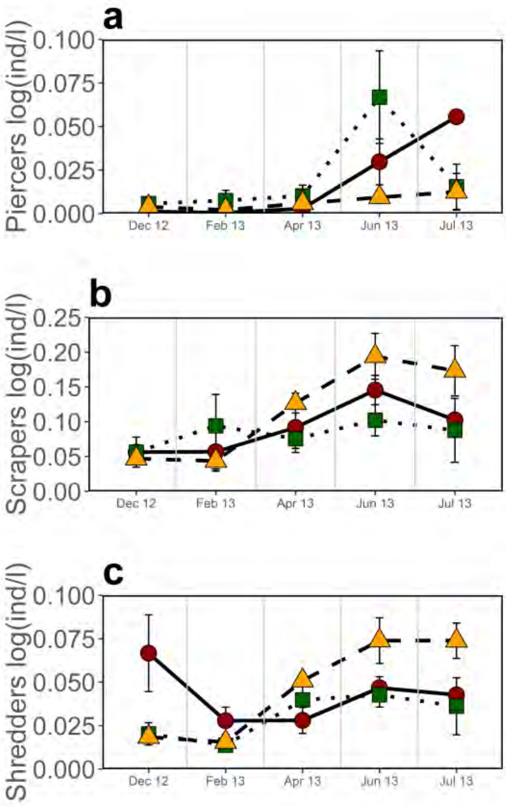
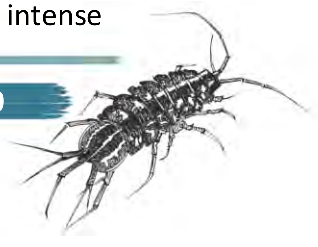


Figure 4.6*: Changes in abundance of shredders (a), scrapers (b), and piercers (c) along the hydroperiod for the high intensity ponds (red circles with solid line), low intensity ponds (orange triangles with dashed line) and unburned ponds (green squares with dotted line). Error bars indicate the standard error of the mean.*

Primary producer dynamics were also modified after the wildfire, probably linked to nutrient differences. Phytoplankton growth is closely related to nutrient availability in freshwater ecosystems, which are normally limited by phosphorus (Vallentyne 1974). Therefore, in the affected ponds, the increase in both total nitrogen and dissolved phosphorus concentrations probably favoured fast responses (blooms) by phytoplankton (Scrimgeour et al. 2001, Mendoza et al. 2015). In addition, these phytoplankton blooms might also be favored by increased light incidence, due to the low macrophyte biomass detected in burnt ponds at the beginning of the hydroperiod. We did not observe an increase in filamentous algae, as is usually observed in rivers and wetlands after wildfires (Spencer et al. 2003, Verkaik et al. 2013, Bixby et al. 2015). However, in other lentic waterbodies (i.e., small boreal lakes), increased primary production after a wildfire is normally caused by phytoplankton (Moser et al. 2002, Lewis et al. 2014, Mendoza et al. 2015).

Community composition and functional traits

Species with common life strategies and functional characteristics were similarly impacted by wildfire, highlighting that wildfire impacts in Mediterranean temporary ponds go beyond species identity. While the taxonomic approach did not detect wildfire impacts, the functional approach detected stronger consequences of wildfire on all functional groups. Both direct and indirect effects were detected: a decrease in organisms remaining in pond sediment and alterations linked to trophic changes. Direct impacts of fire such as mortality (Gresswell 1999) are commonly found in stream fishes and are associated with an increase in water temperatures or ammonia intoxication due to smoke diffusion (Isaak et al. 2010, Beakes et al. 2014). However, in this study, the decreased abundance during the subsequent hydroperiod of organisms that aestivate in sediment may indicate direct burning or the effects of radiant heat on these animals while resting in the sediment during the dry periods (Cowell et al. 2006). Decreased abundances of specific groups after wildfire due to direct burning have been recorded for zooplankton in rice paddies and crayfish in streams and temporary wetlands (Chittapun 2011, Johnston et al. 2014). Although other studies with less intense



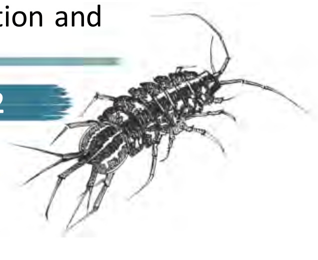
controlled burns detected no direct effect of fire on anostracan cysts (Wells et al. 1997). Therefore, wildfire intensity could be the key to understanding these impacts of fire (Malison and Baxter 2010b). Indeed, in our study, high intensity burned ponds were the ones that showed a substantial decrease in the abundance of these taxa. However, we must emphasize that this impact only decreased their abundance, some of these taxa survived, so their populations were not compromised in the short-term post-fire scenario.

As expected, trophic groups were also affected by wildfire, indicating that temporary pond fauna are also affected by indirect consequences of wildfire. The variation and decline of groups such as collectors, shredders, scrapers or piercers could be a consequence of lower macrophyte densities at the beginning of the hydroperiod, higher nutrient content and phytoplankton blooms. These habitat alterations and trophic changes concur with most studies dealing with wildfires and aquatic systems (Minshall et al. 2001, Lewis et al. 2014). In lotic systems, these changes are normally linked to post-fire floods or morphological changes that delay recolonization (Vieira et al. 2011, Klose et al. 2015), but in temporary ponds, these changes may result in less vegetation cover or a delay in vegetation growth, as seen in the firsts months of our study. However, these changes did not affect colonization by active dispersers, and instead seemed mainly to alter the available resources. Alterations in trophic resources were expected to shift the faunal community (Mihuc and Minshall 1995). However, we found that as the hydroperiod commenced, within the first 2-4 months after fire in burnt ponds, shredders had higher abundances. This fast increase in shredder abundance has also been observed in other lentic habitats (Scrimgeour et al. 2001, Munro et al. 2009, Lewis et al. 2014). This rapid rise and subsequent decrease could be described as a *fire pulse effect* (Malison and Baxter 2010a). Post-disturbance conditions favour groups that rapidly exploit newly available resources (Jackson et al. 2015, Venne et al. 2016), so, other groups may concomitantly decrease in abundance (Oliver et al. 2012). Piercers and scrapers increased their abundance at the end of the hydroperiod in all ponds, but their increase was higher in burned ponds. Most likely, tolerance to burned conditions

and freshly grown vegetation facilitated the increase of piercers and scrapers in the affected ponds (Beganyi and Batzer 2011).

Seasonal trends and community recovery

As predicted, in Mediterranean temporary ponds the direct effects of wildfire played a major role compared to perennial aquatic ecosystems. Moreover, indirect impacts linked to habitat and trophic alterations of the community were also detected, similar to other lentic systems affected by wildfire (de Szalay and Resh 1997, Beganyi and Batzer 2011, Lewis et al. 2014). Therefore, intermittency adds a new path of disturbance for the aquatic community: a decrease in abundance of organisms resting in sediments during the dry phase, probably directly burnt by the wildfire. Additionally, intermittency also altered the outcome of wildfire compared to perennial waterbodies for variables such as nutrient concentration. Nevertheless, our results also show that although there were some immediate wildfire-related impacts, there was a strong temporal pattern operating regardless of wildfire. Recovery in one hydroperiod (10 months) is faster or as fast as that of other studied systems, such as temperate streams (Gresswell 1999, Minshall et al. 2001) or temperate lakes (1-2 years; Scrimgeour et al. 2001, Lewis et al. 2014) and temporary Mediterranean streams (1-4 years; Kotze 2013, Verkaik et al. 2013, Rodríguez-Lozano et al. 2015). In comparison to lotic or permanent lentic ecosystems, communities from temporary ponds and streams have evolved to cope with drought and the intra and inter annual variation of this phenomenon (Verkaik et al. 2015, Boix et al. 2016). Adaptation to a disturbance such as drying or the unpredictability of flooding appears to indirectly increase resilience to post-fire changes in temporary ponds and riparian systems (Pettit and Naiman 2007). Therefore, communities of temporary Mediterranean aquatic systems appear resilient to wildfire, recovering faster than perennial habitats (Verkaik et al. 2013, Robson et al. 2018). Despite high resilience, these aquatic systems are still impacted over shorter timeframes. Thus, if as predicted, there is an increase in the intensity and recurrence of wildfire impacts, communities from dry areas may be substantially affected (Whitney et al. 2015) both directly and indirectly. Within this context, the importance of pond but also pond-network conservation and



connectivity among ponds may be key to conservation, including buffering wildfire impacts at the landscape scale (metacommunity rescue effect; Leibold et al. 2004). Future approaches should be directed toward better understanding of how network connectivity can enhance system resilience against these impacts.

5. CHAPTER II

Trends in traits selection in a temporary pond metacommunity in response to wildfire disturbance and seasonal succession.

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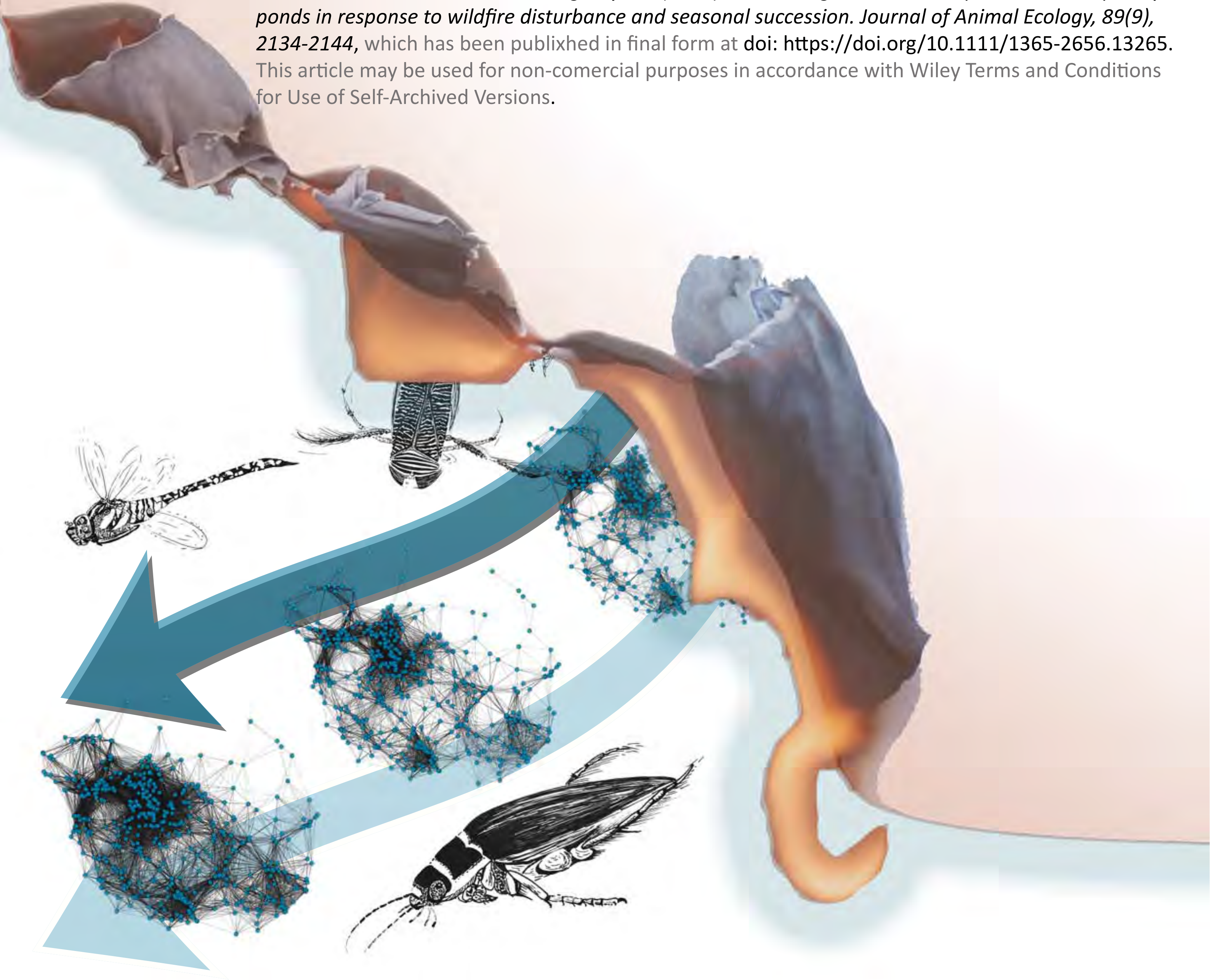


Chapter brief summary

Understanding community assembly mechanisms has been a prevalent aim in ecology. The theory of community assembly by traits selection (CATS) introduced a conceptual and methodological framework for identifying trait selection and its effect on species abundances. Mediterranean temporary ponds experience environmental unpredictability within years and can also be impacted by wildfire disturbances. Wildfires can change the selective pressures determining the trait-dependent responses of communities but may also foster stochasticity because of random variation in recruitment and priority effects. A natural wildfire partially burned a metacommunity of temporary ponds surveyed before and after the wildfire. Capitalizing on this natural before-after-control-impact design, we used the CATS approach to identify assembly mechanisms, their dependence on the wildfire disturbance and the trend across the postfire hydroperiod. The environmental gradients determined trends in the selected traits among ponds, particularly at the beginning of the hydroperiod, with effects that diminished with the advance of the hydroperiod. Before the wildfire, the burned and unburned ponds showed no difference in the selected traits. Wildfire changed the selection of traits such as body size, dispersal ability and feeding group (e.g., scrapers and collectors). Nonetheless, some environmental characteristics were directly more relevant than the wildfire effect, indicating the strong influence of environmental conditions on the biodiversity of temporary ponds. CATS analysis allowed the detection of a wildfire effect on the selection forces assembling communities, which was added to the strong species sorting operating among other traits and environmental gradients. Both niche and neutral processes played an important role in pond seasonal succession. Selection was strong at the beginning of the hydroperiod but decreased in strength at the end of the hydroperiod. Disentangling the main assembly mechanisms of communities has not been straightforward. Here, we report the occurrence of a temporally dependent balance among assembly drivers in which disturbance, local conditions, and species traits interact in determining biodiversity patterns. The explicit identification of selected traits and their role on community assembly represent an advance in the understanding of the mechanisms beyond biodiversity response to disturbances and successional changes in metacommunities.

5.1 Overview

Understanding the drivers of community assembly and the role of species traits in the involved mechanisms has been a cornerstone of ecological theory (Darwin and Wallace 1858, Hutchinson 1959, Schoener 1974, Weiher and Keddy 1999, Vellend 2016). Community structure was originally considered to be the outcome of species interactions and environmental filters affecting species because of their unique combinations of traits (Weiher and Keddy 1999, Chesson 2000). Neutral theory broke this conceptualization, showing that emergent properties of biodiversity could originate in spite or because of the lack of a functional effect of differences in species traits (Hubbell 2001). In addition, the original focus on local determinants of community assembly, such as species interactions, productivity, heterogeneity, or area (Rosenzweig 1995), was expanded, with a main role of organism dispersal being identified (Leibold et al. 2004). Metacommunity theory provides a synthesis of these local and regional mechanisms, recognizing species sorting, patch dynamics, mass effects, and neutral processes as main determinants of local and regional biodiversity (Mouquet and Loreau 2003). While the role of these four mechanisms is generally accepted, the empirical evaluation of their relative importance in community assembly is not straightforward (Logue et al. 2011, Grainger and Gilbert 2016). Furthermore, these mechanisms are currently more understood as part of a gradient, which is product of the balance among several assembly forces (e.g., stochasticity, dispersal and the environment), a conceptualization that surpasses the relatively more closed four-types view of metacommunity ecology (Brown et al. 2017, Leibold and Chase 2018).

The theory of *community assembly through trait selection*, hereafter CATS, introduced a conceptual synthesis and novel methodological approach for the explicit evaluation of assembly mechanisms (Shipley et al. 2006, Shipley 2010, Warton et al. 2015). CATS, focuses on a trait-based habitat filtering model that predicts relative species abundances in a local community on the basis of the unique trait combinations

(Shipley et al. 2006). Furthermore, selection coefficients representing the selective advantage provided by each trait in a local community are determined by CATS. The magnitude and sign of the selection coefficients provide an estimate of the selective pressures operating during community assembly. Therefore, CATS may be used to infer the roles of the different metacommunity mechanisms (Borthagaray et al. 2015a). First, species sorting involves differences in selection pressures among local communities (Leibold et al. 2004). This should be reflected in differences in the selection coefficients of traits in relation to environmental factors (Borthagaray et al. 2015a). Second, the concept of patch dynamics implies changes in selection coefficients in the balance between colonization and extinction rates among species. Third, the mass effect determines the persistence of species in local communities based on the inflow of dispersers (Leibold et al. 2004), which promotes local independence among species traits and abundances in sink communities (Borthagaray et al. 2015a). The latter mechanism may be reflected as a reduction in the strength of selection coefficients in sink communities. Finally, neutral mechanisms result in a lack of association between species traits and abundances, decreasing selection coefficient strength in all communities (Shipley 2010). Therefore, greater coefficient values, either positive or negative, indicate niche assembly mechanisms (e.g., species sorting), whereas lower coefficient values indicate a dominance of more neutral mechanisms, thus, placing the studied system along the niche-neutral gradient (Figure 5.1).

Wildfires are disturbances that can change selective pressures in different communities (Wright and Bailey 1982, Bowman et al. 2009, Prepas et al. 2009). Specifically, in aquatic systems, wildfires can affect nutrient input (Spencer and Hauer 1991, Mast et al. 2016), habitat structure (Britton 1991, Vieira et al. 2011, Cooper et al. 2015), and resource availability (Christensen et al. 1989, Mihuc and Minshall 1995, Gresswell 1999, Lewis et al. 2014). Species responses to these changes depend on their traits, such as those related to life history, diet, dispersal, and body size (Oliver et al. 2012, Rodríguez-Lozano et al. 2015, Robson et al. 2018). In this context, CATS may provide an explicit evaluation of the role of these traits in system response to wildfire

(e.g.,Bargmann et al. 2016) as well as the trends of general metacommunity mechanisms and the balance between niche (i.e., species sorting) and neutral assembly mechanisms.

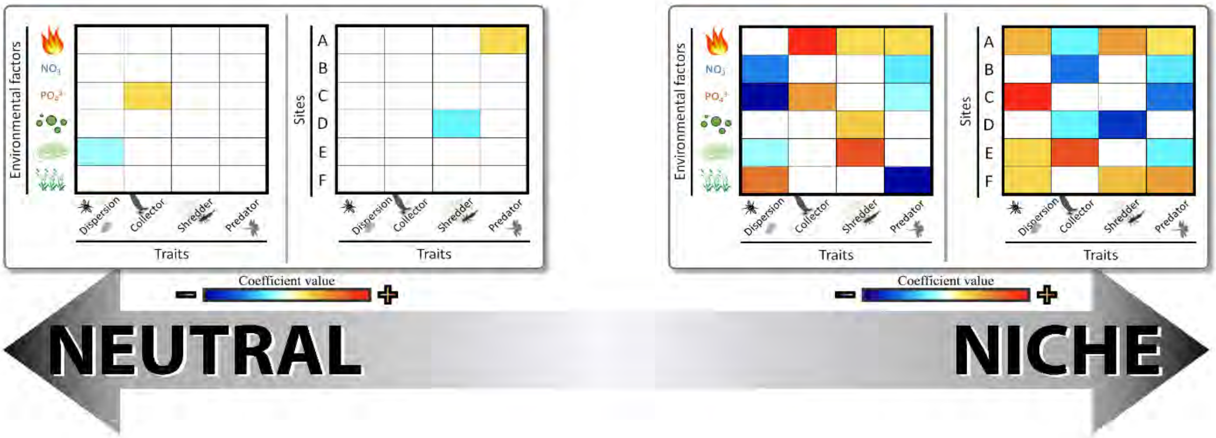


Figure 5.1: *Graphical representation of the expected results of CATS coefficients along the niche-neutral gradient. Coefficient values close to zero, either positive or negative, indicate a smaller relation among traits and environmental factors and thus, a dominance of neutral assembly mechanisms. On the other hand, greater coefficient values, either positive or negative, indicate a stronger relation among traits and environmental factors and thus, a dominance of niche assembly mechanisms.*

Lentic water bodies, such as temporary ponds, typically form metacommunities and have been identified as model systems for the evaluation of metacommunity dynamics (Wood et al. 2003, Chase et al. 2010, Logue et al. 2011). In Mediterranean regions, these systems have variable hydroperiod lengths that change intra- and interannually. Therefore, Mediterranean temporary pond communities are adapted to cope with drought and hydroperiod unpredictability (Boix et al. 2001, 2016). In addition, Mediterranean areas are fire-prone regions where an increase in such disturbances is predicted (Pausas 2004, Kovats et al. 2014, Turco et al. 2018). However, few studies have addressed wildfire impacts on these temporary pond communities and the connection with the mechanisms determining metacommunity dynamics and resilience after such fires (Bixby et al. 2015, Cunillera-Montcusí et al. 2019). In this study, we took advantage of a natural wildfire that partially burned a temporary pond network that was being surveyed and continued this survey during the hydroperiod following the wildfire. We used this naturally produced experiment and the CATS approach to identify the metacommunity- and wildfire-related mechanisms determining the assembly of pond invertebrate communities after this disturbance. Specifically, we explored the spatial

and temporal trends in selection pressures on species traits related to body size, dispersal ability, life cycle, and functional feeding preferences. We expect that trends in local conditions, including wildfire, will determine the trends in the selected traits and assembly processes, which will be evidenced by the selection coefficients obtained with CATS.

5.2 Methods

Study site

The studied pond network is located on the north-eastern Iberian Peninsula and is formed by nearly thirty temporary ponds (Figure 5.2). These temporary ponds often flood with autumn and/or spring rainfalls and dry during the summer. It is also in the summer when most wildfires occur in Mediterranean regions; therefore, ponds can burn completely, including the pond sediment, if they are affected by a wildfire. Seven ponds in the studied system were sampled over time following a before-after control-impact (BACI) design. The first survey was performed before the wildfire, hereafter May12. In July 2012, the wildfire burned some of the ponds. Once flooded, we continued the surveys throughout the following hydroperiod from its beginning until the ponds completely dried, carrying out 4 sampling surveys in December 2012, February 2013, April 2013 and June 2013, hereafter Dec12, Feb13, Apr13 and Jun13, respectively. The wildfire partly affected the pond network, leaving some of the ponds unaffected. Thus, after the wildfire, we had three completely burned ponds and four unburned ponds, which were sampled in one pre-fire campaign and four postfire campaigns. For more detailed information on the wildfire and how the ponds were affected, see Cunillera-Montcusí et al. (2019) and general methods section.

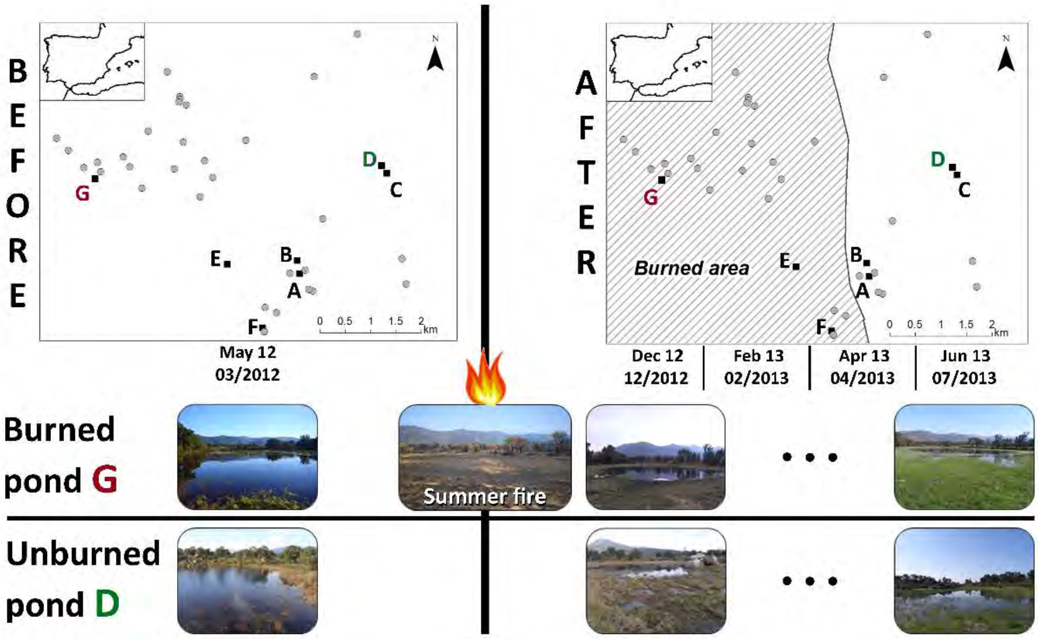


Figure 5.2: *Pond location on the north-eastern Iberian Peninsula. Surveyed ponds are represented by black squares with a corresponding letter (i.e., A, B, C, D, E, F). Dashed area represents the area affected by the summer 2012 wildfire. Images correspond to changes over time in the two sampled ponds represented by different colours (i.e., burned pond G in red and unburned pond D in green).*

We measured environmental variables related to water physical and chemical properties: maximum pond depth (MaxDepth), water temperature (WTemp), dissolved oxygen (O2), conductivity (EC25), pH, soluble reactive phosphate (SRP), inorganic nitrogen (ammonia, nitrite and nitrate, DIN), total nitrogen (TN), total phosphorus (TP), a nutrient limitation indicator (DIN/PT ratio), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC) total inorganic carbon (TIC), total organic carbon (TOC) and fulvic acids percentage (%AC.FUL.). Additionally, primary producer biomass and chlorophyll-a content were measured for macrophytes biomass (Mac Bi; g/m^2), filamentous algae biomass (Fil Bi; g/cm^2) and phytoplankton chlorophyll-a content (Phy; µg/l). Macroinvertebrates were preserved *in-situ* in 4% formaldehyde for posterior identification to the species level whenever possible. See Àvila et al. (2016), Ballón et al. (2016), Cunillera-Montcusí et al. (2019) and general methods section for more detailed information on the sampling procedure, sample processing and water analyses.

Functional traits

A trait database was constructed considering discrete and continuous traits related to the possible consequences of wildfire disturbance on aquatic faunal communities. Four traits were included: i) average body size of the species; ii) dispersal ability group (DAG), which is based on the species' overland dispersal ability and their landscape perceptions (following Heino 2013); iii) life history group (LHG), which is based on the groups proposed by Wiggins et al. 1980), integrating dispersal ability, drought-resistance strategy and other aspects of the life cycle of each species; and iv) feeding category: collector, filterer, scraper, shredder, piercer or predator (from Merritt and Cummins 1996, Tachet 2000). All the considered traits and information related to their categorization are summarized in Table 5.1.

CATS regression

The *community assembly through trait selection,* or CATS (Shipley et al. 2006, Shipley 2010), procedure formerly required the use of maximal entropy formalism (Jaynes 2003) to calculate the selection coefficients for each trait. Fortunately, it was shown that the maximum entropy estimates of selection coefficients can also be obtained from a generalized linear mixed effect model (GLMM). CATS regression estimates species abundances across sites as a function of the species traits, environment, and trait-environment interactions (Warton et al. 2015). This function links species performance with species traits along environmental gradients. By calculating the coefficients relating species abundances with species traits, CATS regression quantifies the expected change in species abundance with the change in trait state (e.g., body size or dispersal mode). These parameters represent coefficients of selection for each trait (Shipley et al. 2006, Shipley 2010). The trait-environment interactions specifically estimate how the trait-abundance relationship is magnified or attenuated with the change in local environmental conditions (Warton et al. 2015, Loranger et al. 2018). In this way, interaction parameters represent the change in selection coefficient for each trait along an environmental gradient. For example, these coefficients may represent the change

in selection pressure for body size across a gradient of pond volumes or the change in selection pressure for dispersal mode associated with burned versus non-burned ponds. In this sense, positive coefficients indicate those traits that are most favoured, while negative coefficients indicate traits that are more disfavoured by selection pressures. The matrix of trait-environment coefficients has been called the fourth-corner matrix (Legendre and Legendre 2012) because it is estimated by combining three other matrices: those including data on species abundances per site (abundance matrix), environmental conditions across sites (environmental matrix), and trait states among species (species traits matrix; Brown et al. 2014).

Trait-environment relations, including the wildfire effect, were analysed with CATS regression using a negative binomial distribution (Brown et al. 2014, Warton et al. 2015). Only environmental variables with a Pearson correlation of less than 0.8 were included in the analysis. In addition, we added the disturbance effect as a factor to test for the effect of wildfire on trait importance (i.e., burned=1 and unburned=0). We ran a fourth-corner model for each of the five sampling surveys. We related the matrix of species abundance per site (Y) with the environmental matrix (X1) and the species trait matrix (X2) to obtain their interaction matrix (X1*X2) following Brown et al. (2014). We used a LASSO penalty to shrink non-significant parameters to zero (Warton et al. 2015). Because all variables were standardized, the value of the coefficient of each trait–environment interaction indicates its relative importance. Furthermore, a matrix of selection coefficients for each trait in each pond was constructed. We used the square root of species abundance and then standardized the CATS regression coefficients following Bring (1994). Both procedures using CATS regression were implemented in the R package ‘mvabund’, which allows the analysis of multidimensional data (Wang et al. 2012). All data analyses and visualizations were conducted in R (R-Core Team 2013).

Table 5.1: *Functional traits used in the CATS regression. Species body size (BS), dispersal ability groups (DAG), life history groups (LHG). The functional feeding groups consisted of 6 groups. Species affiliation to each category was weighted according to the literature, ranging from 0 to 3.*

Species traits		
Body size	BS	Body length (mm)
Dispersal ability groups	DAG	1- Passive dispersers with aquatic adults 2- Weak active dispersers with flying adults 3- Moderate active dispersers with flying adults 4- Strong active dispersers with flying adults
Dry-phase residents	LHG1	Binary code 1-0
Dry-phase residents & spring recruits	LHG2	Binary code 1-0
Dry-phase residents & summer recruits	LHG3	Binary code 1-0
Non-dry-phase residents & spring migrants	LHG4	Binary code 1-0
Functional feeding groups	Collector	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity
	Filterer	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity
	Scraper	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity
	Shredder	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity
	Piercer	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity
	Predator	0- No affinity 1- Small affinity 2- Medium affinity 3- Strong affinity

5.3 Results

The variables related to inorganic carbon, dissolved organic carbon and total inorganic carbon were highly correlated with conductivity. Similarly, the total organic carbon, total nitrogen and total phosphorus were highly correlated with the dissolved inorganic carbon (Supplementary S5.1). Thus, to avoid collinearity problems we selected MaxDepth, pH, WTemp, O2, EC25, SRP, DIN, DOC, % AC.FUL., Mac Bi, Fil Bi and Phy to be included in the models. The CATS models detected strong variation in traits and trait-environment coefficients across ponds and surveys (Figure 5.3 and Supplementary Table S5.2, S5.3). The magnitude of positive and negative selection coefficients was relatively larger at early stages of the hydroperiod (Figure 5.4). Thus, species traits had a larger role in determining species abundances at the initial stages of community succession.

Before the wildfire (May12), we detected no differences between ponds that would burn during summer and unburned ponds (burned factor; Figure 5.3a). This result must be highlighted because it supports a lack of difference in the role of traits in community assembly between ponds that were posteriorly burned and those that were not. Consequently, the subsequent differences in the interaction coefficients for traits and the burned factor could be interpreted as a consequence of changes in assembly mechanisms after wildfire and not a result of pre-existing differences among ponds. On the other hand, almost all the other environmental gradients showed a significant effect on trait selection, indicating that they influence the differentiation of local communities, and sometimes presented larger coefficients than the wildfire effect. For example, trends in dissolved inorganic nitrogen, dissolved inorganic carbon or water temperature were more important determinants of variation, which is evidenced by these variables having larger coefficients than the ones linked to the burned effect (Figure 5.3).

After the wildfire and at the beginning of the hydroperiod (Dec12 and Feb13 samplings), differences in trait selection were detected between the burned and

unburned ponds (Figure 5.3b and c). Species with a large body size and scrapers were favoured in burned ponds, while those with strong active dispersal were underrepresented. For all the other considered traits and all the environmental gradients, several significant interactions were detected. This implies a large change in the trait-abundance relationship among ponds, indicating a species sorting assembly among communities (i.e., different traits selected in different ponds). During the middle and towards the end of the hydroperiod (Apr13 and Jun13), the burned factor still appeared to be an important determinant of the selected traits, but a reduction in the magnitude of coefficients was observed (Figure 5.3d and e, Supplementary Table S5.2). Scrapers were still favoured in burned ponds, with the addition of a slightly higher representation of predators. Collectors and dry-phase residents (i.e., LHG1) were underrepresented among the burned ponds. A general reduction in coefficient strength was observed for all interactions and traits. Regarding the coefficient values for each pond across the surveys (Figure 5.3 f-j), we observed strong variation among sites regardless of wildfire impact that changed throughout the surveys, highlighting the idiosyncratic selection patterns among sites, which were suggested by the coefficients observed for the trait-environment interaction (Figure 5.3 a-e).

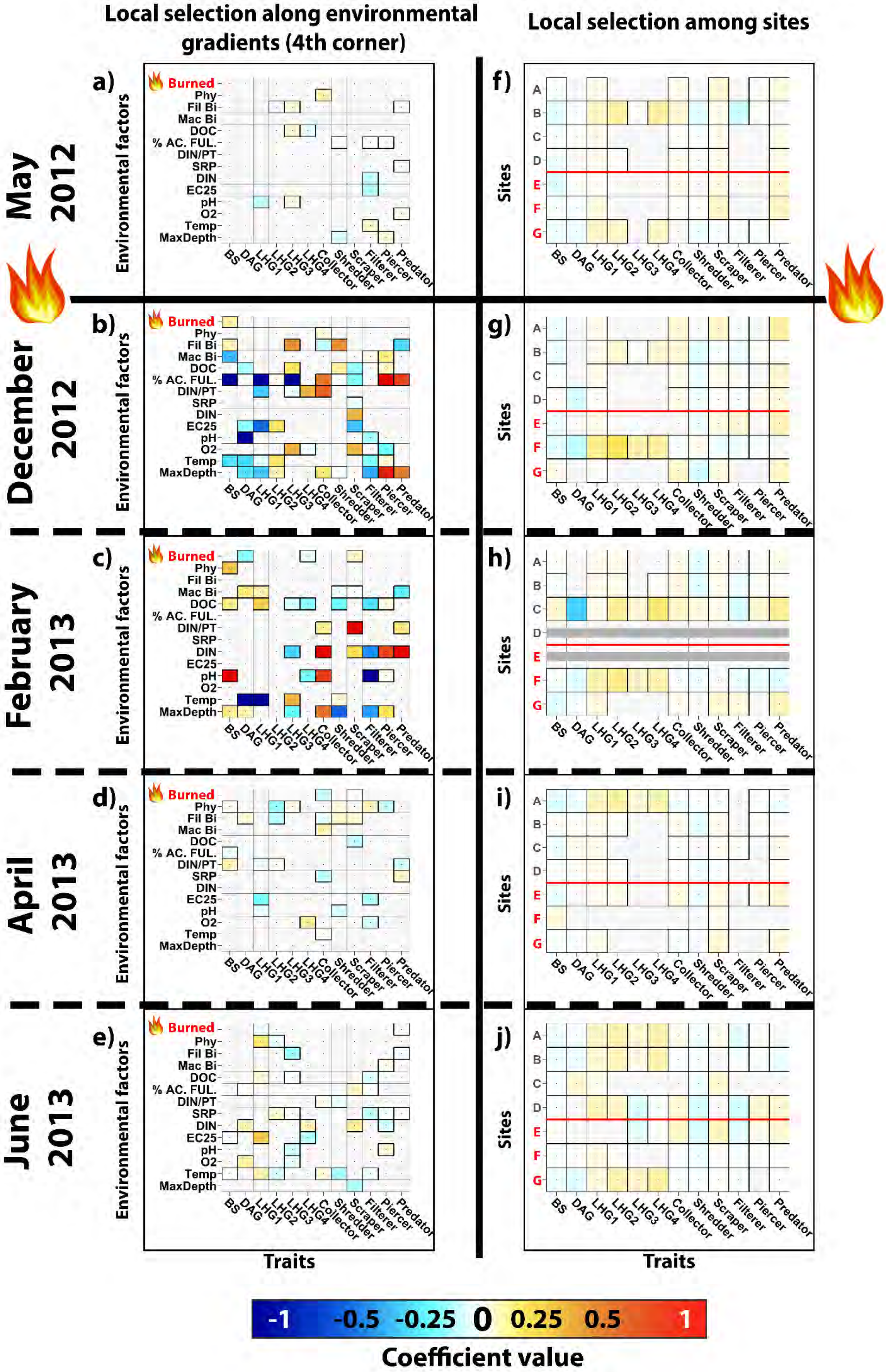

Local selection along environmental gradients (4th corner)
Local selection among sites
May 2012
December 2012
February 2013
April 2013
June 2013
a)
b)
c)
d)
e)
f)
g)
h)
i)
j)
Environmental factors
Burned
Phy
Fil Bi
Mac Bi
DOC
% AC. FUL.
DIN/PT
SRP
DIN
EC25
pH
O2
Temp
MaxDepth
Sites
A
B
C
D
E
F
G
BS
DAG
LHG1
LHG2
LHG3
LHG4
Collector
Shredder
Scraper
Filterer
Piercer
Predator
Traits
-1
-0.5
-0.25
0
0.25
0.5
1
Coefficient value

Figure 5.3: *Graphs in the left column (a, b, c, d, e) represent the selection coefficients for each trait-environment interaction (i.e., fourth-corner models), including the burned effect. Graphs in the right column (f, g, h, i, j) represent the selection coefficients for each surveyed pond, unburned (A, B, C, D) and burned (E, F, G). Colour intensity indicates coefficient magnitude, while red colours indicate positive values and blue colours indicate negative values. Each panel corresponds to a sampling survey: a and f correspond to May12, b and g correspond to Dec12, c and h correspond to Feb13, d and i correspond to Apr13, and e and j correspond to Jun13. Coefficients were estimated using community assembly via trait selection (CATS regression). Fourth-corner models were calculated with a LASSO penalty that shrinks non-significant effects to zero.*

5.4 Discussion

The theory of community assembly by trait selection, or CATS, is emerging as an interesting framework to improve our understanding of ecological mechanisms that shape biodiversity patterns (Harabiš and Dolný 2018, Krishnadas et al. 2018, Li et al. 2018, Loranger et al. 2018). The evaluation of the relative roles of traits in explaining species abundance allowed us to better picture the trend in assembly processes in response to wildfire as well as the actions and relative importance of general mechanisms, such as species sorting and neutral dynamics. Disturbances, such as wildfires, may promote relatively more neutral mechanisms due to stochasticity in species recolonization (Didham et al. 2005, Didham and Norton 2006, Fukami 2015), but they can also promote deterministic assembly or the rapid transition towards it when the consequences of wildfire constitute a strong environmental filter (Chase 2007, 2010, Ferrenberg et al. 2013). Our study provides two contributions to this framework that should be highlighted. First, we found that wildfire promoted the action of novel selection forces for specific traits. These results imply that there were disturbance-generated trait responses, which were added to the strong species sorting processes operating among other traits and environmental gradients also detected in the CATS analyses. Second, we found support for a main role of niche processes, with species sorting as the main determinant of metacommunity assembly. This is supported by the changes in selected traits among ponds and environmental conditions across the hydroperiod (Ferrenberg et al. 2013, Borthagaray et al. 2015a). Nonetheless, neutral

processes also played an important role, which was supported by the attenuation of the selection coefficient at the end of the hydroperiod. Therefore, we identified a trend in the relative importance of traits for species performance across pond succession, supporting a transition from niche to neutral assembly (Figure 5.1).

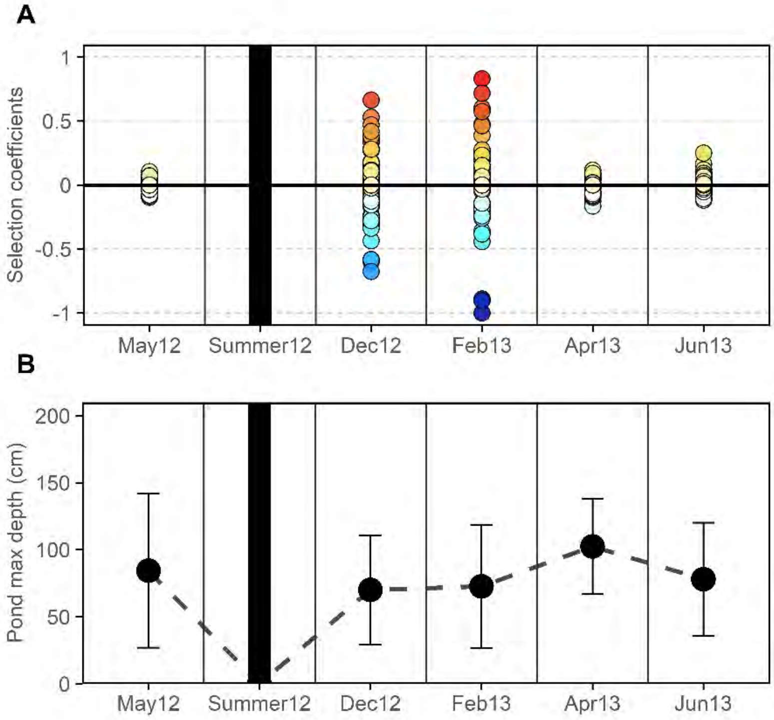


Figure 5.4: *Panel A shows all the selection coefficient values for each of the five fourth-corner models: May12, Dec12, Feb13, Apr13 and Jun13. Colour intensity indicates coefficient magnitude, while red colour indicates positive values and blue colour indicates negative values, as shown in Fig. 3. Note that higher magnitudes were detected in Dec12 and Feb13. Panel B: Mean pond depth along all sampling surveys. The black line indicates summer 2012 wildfire, when ponds were completely dry.*

The wildfire effect had a larger impact on the identity of the selected traits at the beginning of the hydroperiod and progressively decreased over time. Larger sizes seem to be favoured after a wildfire (Bargmann et al. 2016), likely due to a recolonization dynamic, in which relatively large organisms are capable of reaching disturbed sites tolerating new conditions. Additionally, in the studied systems, these organisms likely benefit from a lack of competence among directly burned organisms (Cunillera-Montcusí et al. 2019). Across the subsequent stages, the CATS results indicated that strong active dispersal, migrant life-histories and collector strategies were negatively

affected in burned ponds, while scraper and predator feeding strategies were favoured. The alteration of feeding strategies after the wildfire and during almost the entire subsequent hydroperiod may have been the result of a change in resource characteristics among the burned ponds. Such alterations have been identified as the main impacts of wildfire in aquatic systems (Scrimgeour et al. 2001, Oliver et al. 2012, Lewis et al. 2014, Robson et al. 2018). However, although wildfire effects were detected, it should be highlighted that the wildfire was not the only, and not even the dominant, environmental variable affecting the local community assembly. Therefore, in addition to the impact of wildfire, local environmental conditions still greatly determine temporary pond community assembly.

Niche assembly mechanisms, particularly species sorting, play a key role in the biodiversity of the studied Mediterranean temporary ponds. This assertion is supported by the observed variation in the strength of the selection coefficients among the environmental gradients and, consequently, among ponds (Borthagaray et al. 2015a). Temporary ponds are systems subject to strong inter- and intra-annual variation in Mediterranean climate areas (Williams 2006, Boulton et al. 2014, Boix et al. 2016) and their specific local conditions (Arim et al. 2011). The amount, regularity, and distribution of rainfall during the hydroperiod determine pond community assembly (Boix et al. 2001, De Meester et al. 2005, Vanschoenwinkel et al. 2008c, Schriever and Lytle 2016). This favours great spatial and temporal variation in selected traits, thus promoting strong local filters (i.e., species sorting), which vary with the dominant conditions (Borthagaray et al. 2015a, Leibold and Chase 2018). Indeed, this mechanism has been detected in other pond systems (Chase 2007, Florencio et al. 2014, Hill et al. 2017a, b). However, the attenuation of selection strength during more stable conditions (i.e., Apr13, Jun13) also indicates a relaxation of species sorting mechanisms favouring community assemblies that are closer to neutral dynamics. This attenuation reflects a loss in the of influence of species traits on species abundance. This trend may coincide with the conceptual successional model of phases alteration proposed for temporary ponds (Kenk 1949, Williams 2006, Boix et al. 2016). This model proposes a change over time in the main determinants of community structure (Lake et al. 1989, Bazzanti et al.

1996, Boix et al. 2004). At the beginning of the hydroperiod, abiotic factors represent a strong filter for temporary pond community structure, thus increasing the role of species traits in surpassing such filters (i.e., species sorting). Then, in the middle of the hydroperiod, biotic factors become the main determinants of temporary pond community structure. Biotic interactions, such as predation and competition, could be trait-independent forces (Hubbell 2001, Arim et al. 2011) if all prey species have similar predation pressures, for example, because no species experiences a body size refuge (Morin 1983). Nonetheless, it is also possible that our matrix of species traits was appropriate in terms of capturing those traits related to the abiotic filtering process but not in regard to capturing those traits determining species fate in competitive and predatory interactions (i.e., biotic interactions). On the one hand, this would demand novel studies covering a larger set of traits able to better represent such mechanisms (Shipley 2010). On the other hand, the used trait matrix considered species body size and diet. These attributes are directly related to the predatory and competitive ability of the species, but the role of these traits did not increase with the attenuation of the roles of other traits. Consequently, not discarding alternative explanations, the observed reduction in the strength of selection coefficients may represent a transition from niche to neutral assembly (Figure 5.1). Therefore, community assembly in temporary ponds may fluctuate over time along the niche-neutral gradient, reinforcing the existence of a change in the balance between niche and neutral mechanisms over time at the metacommunity level (Chase 2007, Ng et al. 2009, Ferrenberg et al. 2013, Mouillot et al. 2013, Leibold and Chase 2018).

The use of the CATS approach for the evaluation of invertebrate communities may provide a novel and thoughtful understanding of the assembly mechanisms in animal communities (Shipley et al. 2016). CATS regression analysis has been recently applied in faunal communities (Bargmann et al. 2016, Harabiš and Dolný 2018), expanding its previous domain of empirical evaluation of plant communities (Sonnier et al. 2010, Laughlin et al. 2011, Baastrup-Spohr et al. 2015, Janssen et al. 2017, Krishnadas et al. 2018). In addition, its application to improve the evaluation of disturbance impacts on community assembly has provided new insights regarding community response to

and resilience against such impacts (Mouillot et al. 2013, Bargmann et al. 2016, Niu et al. 2016) as well as concerning associated management strategies (Harabiš and Dolný 2018). As a result of this study, through the use of the CATS approach, we have a better understanding of how wildfire impacted Mediterranean temporary pond communities, which is needed to better understand the impact of this increasing disturbance on these systems (Bixby et al. 2015). In temporary ponds, the impact of wildfire has been mainly linked to the direct burning of organisms resting in the sediment (Cunillera-Montcusí et al. 2019). However, the strong environmental control of temporary ponds and the consequent adaptation of their faunal communities seem to increase resilience against postfire consequences, diminishing the other expected negative consequences. Furthermore, the recolonization of affected ponds through dispersal may also increase recovery and consequently system resilience through a metacommunity rescue effect (Leibold et al. 2004). Within this context, the CATS trait-based approach appears to be an appropriate tool to help disentangle the main assembly mechanisms of communities (Loranger et al. 2018). Such an approach could facilitate extrapolation and comparisons among regions or among ecosystems (Shipley 2010), becoming a powerful tool to understand the impacts on and patterns of biodiversity at larger regional scales and against greater regional threats.

6.CHAPTER III

The role of pond isolation, distance and direction in the colonization pattern of weak and strong dispersers

This is a pre-print of an article published in Aquatic Ecology: ***Cunillera-Montcusí, D.,*** *Boix, D., Sala, J. et al. Large- and small-regional-scale variables interact in the dispersal patterns of aquatic macroinvertebrates from temporary ponds. Aquat Ecol (2020).* The final authenticated version is available online at: https://doi.org/10.1007/s10452-020-09792-8.

Chapter brief summary

Dispersal is one the main determinants of metacommunity assembly, yet its dynamics and relation with network properties and structure have not been fully pictured. Wide regional perspectives such as the network centrality-isolation gradient will influence communities abundance and richness, but smaller regional perspectives such as direction or distance among water bodies might influence them too. Furthermore, taxa regional perspectives might also determine their interaction with these regional perspectives. All these dispersal drivers are generally difficult to analyse, but mesocosms approaches, combined with network metrics constitute an interesting framework to disentangle relevance of this drivers. In this study, our aim was to bring together both approaches to analyse how dispersal dynamics are influenced by wider and smaller (distance and direction from the source pond) regional perspectives and how such dynamics behave between different dispersal abilities. Therefore, we set up several mesocosms (20l plastic containers) around two source ponds having divergent locations within network centrality-isolation gradient. Mesocosms were installed at different distances from the source pond (10 m and 100 m) and at different directions (towards the closest neighbour pond or not). We monitored mesocosms along the whole hydroperiod (9 weeks) and one week after ponds drying (10th week) surveying their macroinvertebrates fauna and environmental variables. We also sampled source ponds in three occasions (hydroperiod beginning, middle and end). We analysed temporal trends among mesocosms considering 1) location and distance and; 2) location and direction, differentiating between taxa dispersal ability (weak and strong dispersers) using generalized additive models and model selection. We also analysed mesocosms community composition and pond community composition. The wider regional perspective, being central or isolated, determined dispersal groups abundance and richness along the hydroperiod, with greater values in central locations. However, small regional perspectives, specially direction (i.e., being between two ponds) modulated this general trend making isolated mesocosms present similar patterns as central ones. Mesocosms community composition was also determined by location but not by distance nor direction. These values suggested a main role of mass effect archetype at

a global scale, though this big general pattern might be modulated by both smaller scales and taxa intrinsic landscape perceptions. Throughout this study, the interaction of these different regional perspectives drove dispersal dynamics influencing communities biodiversity and consequently, whole metacommunity assembly.

6.1 Overview

Dispersal understood as the ability of organisms to move from a source habitat to another habitat has been largely studied (Fernando 1958, 1959, Landin 1980, Safriel and Ritte 1980, Ferriere et al. 2000, Bilton et al. 2001, Bohonak and Jenkins 2003, Incagnone et al. 2015, Morán-Ordóñez et al. 2015). Several works have focused on disentangling the patterns of arrival, species colonization or which cues promote dispersal dynamics (Landin and Stark 1973, Williams et al. 2007, Boix et al. 2011, Bogan and Boersma 2012, Boda and Csabai 2012). In this context, temporary ponds constitute ideal ecosystems to study dispersal patterns, because pond drying compromise aquatic organisms life histories forcing them to cope with drought via resistance structures or by dispersing to permanent aquatic refugia (Landin 1976, Wiggins et al. 1980, Williams 2006, Incagnone et al. 2015). Furthermore, fauna of temporary ponds includes taxa with different dispersal abilities, which constitute distinct regional perceptions and consequently, imply different interactions with the landscape and its structure (Rundle et al. 2002, De Bie et al. 2012, Heino 2013, Padial et al. 2014, Borthagaray et al. 2015b, Horváth et al. 2015). Some taxa have weaker capacity to actively move from pond to pond (e.g., Chironomidae; Delettre and Morvan 2000, Dettinger-Klemm 2003) while other taxa can cover larger distances and rapidly colonize recently flooded places (e.g., Coleoptera, Heteroptera or Odonata; Fernando 1959, Nilson 1996, Angelibert and Giani 2003, Miller and Bergsten 2016). However, dispersion of these taxa not only depends on their intrinsic ability to move from one site to another, weather or habitat conditions are also main drivers of this process. Weather strongly determines the benefit-risk balance of moving from one pond to another, because humidity, wind velocity or temperature can compromise flying capacities and at the end, dispersal success (Murillo 1985, Gatehouse 1994, Velasco and Millan 1998, Csabai and Boda 2005, Boix et al. 2011, Bogan and Boersma 2012). In a similar way, time of the day and sun inclination mediate daily changes in flight activity and thus on dispersal patterns (Landin 1968, Csabai et al. 2006, Williams et al. 2007, Boda and Csabai 2009, Boda et al. 2014). Finally, the target habitat conditions can alter arrival patterns too, as predator presence or habitat environmental

characteristics can diminish dispersal success (Waser 1985, Resetarits and Binckley 2009, Trekels and Vanschoenwinkel 2017). Nonetheless, all these local drivers of dispersal dynamics must be considered together with a more regional perspective, which also modulates these dynamics (Audet et al. 2013, Henriques-Silva et al. 2015, Leibold and Chase 2018).

Despite taxa intrinsic abilities and habitat conditions, the regional context also determines dispersal patterns (Jenkins and Buikema 1998, Bohonak and Jenkins 2003, Louette and De Meester 2005, Borthagaray et al. 2015b). Geographical channels such as natural wet corridors (e.g., rivers) or abundance of water bodies and their characteristics (i.e., water permanence, water body size, habitat heterogeneity, etc.) mediate organisms fluxes (Oertli et al. 2002, 2008, Bogan and Boersma 2012, Phillipsen and Lytle 2013, Cañedo-Argüelles et al. 2015, Geismar et al. 2015, Gall et al. 2017, Sarremejane et al. 2017). The distribution, location and organisms fluxes among such habitats at the end constitute a network of connected habitats (i.e., Metacommunity; Leibold et al. 2004). These connections depend on the distance among habitats (i.e., water bodies and ponds) and consequently, from a regional perspective, on the distribution and location of such habitats within the network (Urban and Keitt 2001, Rozenfeld et al. 2008, Borthagaray et al. 2014). Ponds located in a more central position of the pond network will probably be subject to different arrival patterns (e.g., higher arrival frequency or greater new taxa incorporation) than those more isolated ponds (Economo and Keitt 2008, 2010, Chase and Shulman 2009). Consequently, we could expect a divergence in the relative importance of metacommunity archetypes along this centrality-isolation gradient (Borthagaray et al. 2015a, Leibold and Chase 2018). In more central locations, higher density of ponds and less difficulties to reach them will probably promote dominance of source-sink dynamics as well as dominance of stronger competitors. On the other hand, in more isolated locations, specific habitat characteristics will become more important and weaker competitors, if able to reach those locations, may be benefited in stronger competitor absence (Vanschoenwinkel et al. 2008a, Chase and Shulman 2009, Borthagaray et al. 2015a, Morán-Ordóñez et al. 2015, Castillo-Escrivà et al. 2017, Hill et al. 2017b).

The role of different dispersal determinants (i.e., pond location) are difficult to analyse or quantify experimentally (Bilton et al. 2001) and their study has required sometimes the usage of dispersal traps (i.e., mesocosms) in order to disentangle main dispersal drivers and dynamics (Maguire 1963, de Szalay et al. 1996, Shurin 2000, Boix et al. 2011, Bogan and Boersma 2012, Trekels and Vanschoenwinkel 2017). Indeed, mesocosms constitute a useful approach to comprehend such dynamics, although there is always a mesocosms effect and some species might be less attracted to them (de Szalay et al. 1996, Williams et al. 2007). Nonetheless, mesocosms utilization allows to specifically focus on key dispersal determinants such as the distance or the direction of source and sink ponds. The geographical distance between ponds determines the possibility of organisms to reach suitable habitats, being closer distances more easily reachable for a wider range of organisms (Chaput-Bardy et al. 2010, Astorga et al. 2012, Geismar et al. 2015, Sarremejane et al. 2017). Furthermore, the direction of closer ponds, understood as the direction in which dispersal can be more successful, may also drive dispersal fluxes. Indeed, not all the possible directions from the source pond must have the same number of suitable habitats and thus, it is possible that dispersal fluxes will have preferential directions (Resetarits and Binckley 2013, Gall et al. 2017). Consequently, these dispersal determinants are, at the end, compromising community assembly at a smaller regional scale, as they play a role in driving species composition and metacommunity assembly patterns together with wider regional scales (i.e., entire network location; Heino 2013, Phillipsen and Lytle 2013, Henriques-Silva et al. 2015, Hill et al. 2017b).

In order to better understand the importance of dispersal determinants such as direction or distance in a regional context, in this study, we set up a field experiment using mesocosms placed at different directions and distances from ponds that have divergent locations within the isolation-centrality gradient of the studied pond-network. We selected a high-altitude temporary pond-network to build this field experiment due to its network configuration as it have ponds located throughout a marked central-isolation gradient, its low human pressure and the short hydroperiod length of these systems, in which dispersal plays a main role for their biotas due to the narrow time-

window of suitable conditions for colonization (Wissinger et al. 2016). Furthermore, high altitude temporary ponds are of great interest in the Pyrenees, as their faunal communities –specially macroinvertebrates– have not received the same attention than in central and north Europe or north America (Schneider 1999, Wissinger et al. 1999, Hinden et al. 2005, Oertli et al. 2008, Wissinger et al. 2016) nor than other aquatic habitats (e.g., streams and lakes; Verneaux et al. (2004), de Mendoza and Catalan (2010), Khamis et al. (2014)). Although some studies, focused on fish negative impacts on Amphibians, have reported temporary alpine ponds biotas (Miró et al. 2018). The consideration of smaller (i.e., direction and distance) and wider (i.e., location within the centrality-isolation gradient) regional perspectives together as drivers of dispersal dynamics is still not clearly understood and there is a need of more empirical approaches (Ferriere et al. 2000, Henriques-Silva et al. 2015). In addition, the consideration of the whole network and their centrality-isolation values in a field-experiment using mesocosms represent a step forward the empirical comprehension of network structure and its influence on dispersal dynamics and consequently at metacommunity level (Borthagaray et al. 2015a). Therefore, in this study, we aim to bring together both classic (i.e., mesocosms) and novel (i.e., network structure) concepts and approaches to analyse how dispersal dynamics are influenced by smaller and wider regional perspectives and how such dynamics behave between different dispersal abilities – i.e., weak and strong active disperser.

6.2 Methods

Study site

The studied pond network is located in north eastern Iberian Peninsula, in the Pyrenees mountain range (2000 m.a.s.l.). It is composed of nearly 120 water bodies being some of them temporary ponds (Figure 6.1). The hydroperiod in this area is short and starts in April-March with snow melting and ends in June-July with pond complete drying due to summer temperatures (Supplementary S6.1). In order to select two suitable temporary

ponds having contrasting locations in the isolation-centrality gradient of the network (i.e., one central and one isolated) we followed several steps. Firstly, we worked with the graph network built with the minimum spanning tree, which connects all local ponds with a minimum linkage distance between them. Then, we estimated the isolation-centrality gradient using the closeness metric, which considers the whole network to determine for each water body a value of centrality accounting with all network water bodies (Urban and Keitt 2001, Borthagaray et al. 2015a). Secondly, we considered ponds physical, chemical and biological characteristics, mainly substrate typology and vegetation cover to ensure similitude between the two source ponds. Finally, we took into account the feasibility to reach the selected ponds, considering accessibility and ponds similarities regarding human and cattle presence to select the most suitable ponds. Once all these properties were considered (Supplementary S6.2) we selected the two more similar and suitable ponds, as the source pounds to setting up the mesocosms experiment (Figure 6.1).

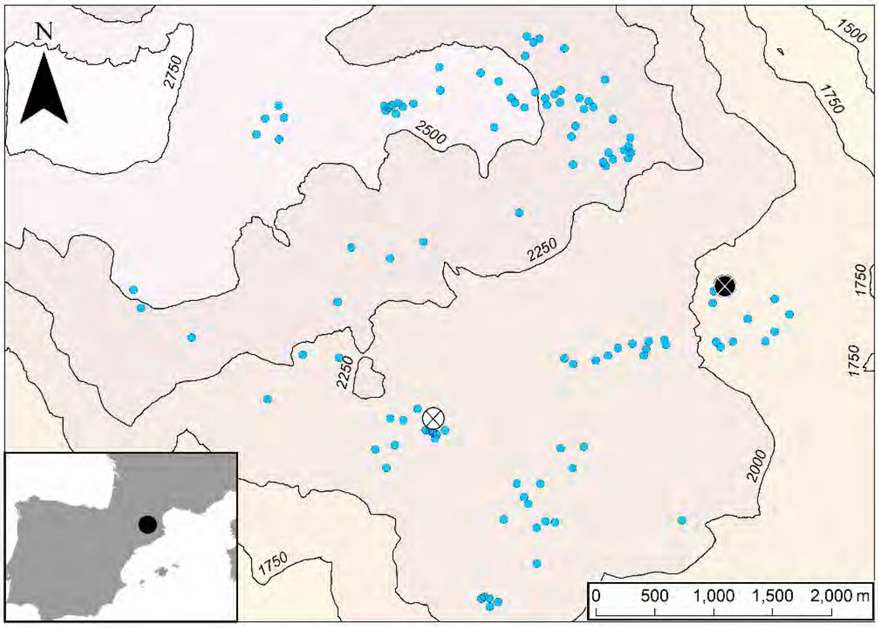


Figure 6.1: *Guils de Cerdanya area with all its water bodies indicated (blue dots). Most water bodies are found around 2000 and 2500 m.a.s.l. The two selected ponds, one central (white dot with black cross) and one isolated (black dot with white cross) are around the 2000 m.a.s.l. and separated by approximately 2,5 km distance.*

Mesocosms experimental set up

The mesocosms consisted of plastic containers filled with 20 litres of filtered water, using a sieve with a mesh size of 50 µm, and with overflow holes to maintain a constant water level. They were grouped in sets of four mesocosms with two meters between them and surrounded by a protection fence to avoid cattle entrance (Figure 6.2A). We built three sets, that where placed at different distances and directions. The first set, hereafter the Control set, was placed at ten meters from the pond. The second set, hereafter the Between set, was placed at ten meters from the pond but directed towards the closest neighbour pond. The third set, hereafter the Distant set, was placed at one hundred meters from the pond and in the same direction as the control set. This setting was repeated twice, one for the central pond and the other for the isolated pond. Overall, we placed 24 mesocosms covering two possible effects on dispersal: distance (all mesocosms in the same direction but at a distance of 10 m and 100 m from the source pond; control sets versus distant sets) and direction (all mesocosms at 10 m from the source pond but at opposite directions; control sets versus between sets). This setting was repeated in both types of source ponds; one located in a central position within the network and the other in an isolated position (Figure 6.2B).

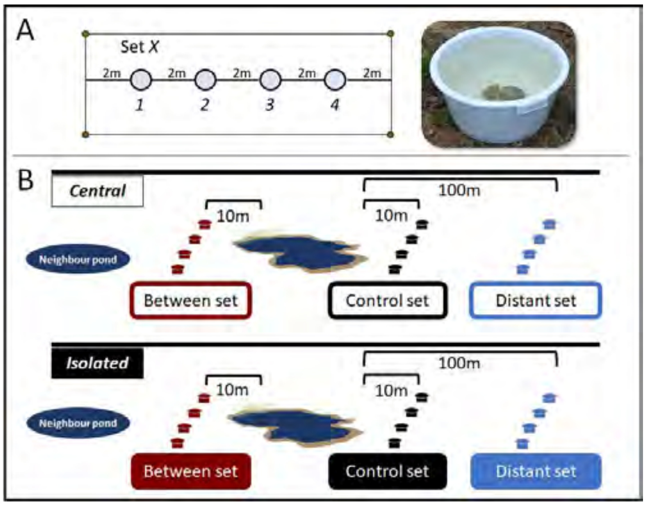


***FIgure 6.2**: A: Mesocosms sets configuration and a mesocosm picture. B: All three set typologies (control set (black), distant set (blue) and between set (red)) configuration in each studied pond location: central pond (panel B Top; white dot with black cross) and isolated pond (panel B bottom; black dot with white cross).*

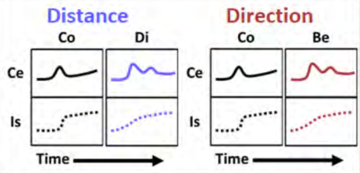
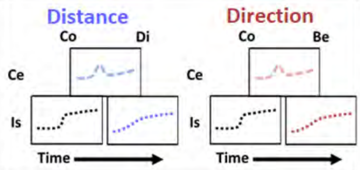
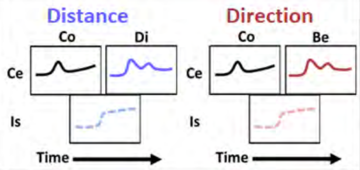
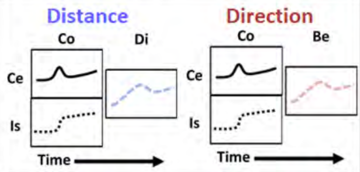
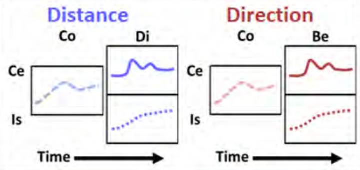
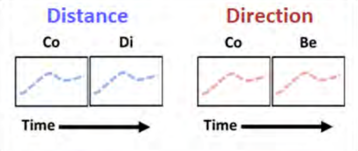
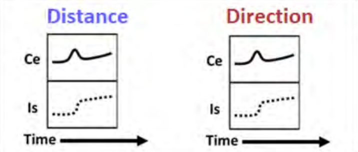
Mesocosms colonization trends

We measured colonization trends in mesocosms encompassing the whole hydroperiod that lasted 9 weeks (May 18th – July 19th 2016). We filled all mesocosms on 18th May with filtered water from a river using a sieve of 50µm mesh size. Additionally, we placed two medium/large rocks to provide shelter for colonizers and weight mesocosms down. Since then, once per week until ponds complete drying, we monitored all mesocosms maintaining their water levels. We also measured water temperature (Tº), dissolved oxygen (O2) and conductivity (EC25) in situ (field datalogger model HACH HQ30d) of all mesocosms and of the two selected ponds. Finally, in each monitoring campaign we sampled mesocosms organisms. All samples were taken following two steps: first, we conducted visual surveys and sampled large organisms (e.g., Coleopterans, Heteropterans). Second, we took one litre of mesocosm water and filtered it through a 250-µm-mesh net to quantify more abundant and smaller organisms (i.e., Diptera larvae). All sampled organisms were preserved in situ with 70% ethanol. We then sorted, counted and identified all organisms to species level whenever possible in the laboratory. Poorly developed individuals and immature stages were mainly identified to genus level and only in few cases they were identified to family level. Once identified, all species were classified according to their dispersal ability in weak dispersers (i.e., Chironomidae, Ceratopogonidae, Ephydridae and Culicidae) and strong dispersers (i.e., Coleoptera) following Tachet (2000) and Heino (2013) proposals.

We analysed differences among environmental mesocosms characteristics with an analysis of variance for each sampling survey to detect variations in water characteristics throughout the experiment. Faunal colonization trends were evaluated considering 3 different response variables: (1) weak dispersers abundance, (2) strong dispersers abundance and (3) total accumulated richness. Each one of these variables were treated independently. We used generalized additive mixed models (GAMM with a negative binomial distribution and using mesocosm identity as random intercepts) to fit the response of the above-mentioned variables along time (smooth term). As we wanted to test the importance of the centrality-isolation gradient (explanatory variable

with two levels; central site versus isolated site) in relation to distance and direction separately, we conducted for each response variable two different analyses according to the tested effects (Table 6.1): “Distance based”, and “Direction based”. The main difference between these two analyses is that the “Distance based” includes, as explanatory variable, the distance of the mesocosms to the source pond (two levels; control set versus distant set), whereas the “Direction based”, includes the direction in which the mesocosms were settled (two levels; control set versus between set). To sum up, we considered three response variables and analysed them accounting for 1) centrality-isolation and distance (Distance based analysis) and; 2) for centrality-isolation and direction (Direction based analysis).

Table 6.1: *Generalized additive mixed models (GAMMs) used in the model selection process for mesocosms weak active dispersers abundance, strong active dispersers abundance and accumulated richness. Each row corresponds to each one of the proposed models that describe different colonization patterns. Model description shows the GAMMs accounted terms. Graphical representation indicates the considered patterns for each model (variable evolution throughout time). Biological meaning describes for each model what is understood if it is selected as the one best describing the colonization pattern.*

	Model description	Graphical representation	Ecological interpretation
Model A (Full model)	***Main terms:*** Centrality (Ce; Is) Distance (Co; Di) or Direction (Co; Be) ***Smooth term:*** Time (Week) ***Interaction term:*** Centrality x Distance (CeCo; CeDi; IsCo; IsDi) or Centrality x Direction (CeCo; CeBe; IsCo; IsBe)		All mesocosms typologies present **different colonisation patterns** along time.
Model B1 (merging central sets)	***Main terms:*** Centrality (Ce; Is) Distance (Co; Di) or Direction (Co; Be) ***Smooth term:*** Time (Week) ***Interaction term:*** Centrality x Distance (Ce; IsCo; IsDi) or Centrality x Direction (Is; IsCo; IsBe)		All **central** mesocosms present the same pattern regardless of distance/direction. ↓ **Central** locations colonization patterns are not influenced by distance/direction.
Model B2 (merging isolated sets)	***Main terms:*** Centrality (Ce; Is) Distance (Co; Di) or Direction (Co; Be) ***Smooth term:*** Time (Week) ***Interaction term:*** Centrality x Distance (CeCo; CeDi; Is) or Centrality x Direction (CeCo; CeBe; Is)		All **isolated** mesocosms present the same pattern regardless of distance/direction. ↓ **Isolated** locations colonization patterns are not influenced by distance/direction.
Model B3 (merging distant/between sets)	***Main terms:*** Centrality (Ce; Is) Distance (Co; Di) or Direction (Co; Be) ***Smooth term:*** Time (Week) ***Interaction term:*** Centrality x Distance (CeCo; IsCo; Di) or Centrality x Direction (CeCo; IsCo; Be)		All **distant**/**between** mesocosms present the same pattern regardless location. ↓ Distance/Direction homogenize the **control pattern**.
Model B4 (merging control sets)	***Main terms:*** Centrality (Ce; Is) Distance (Co; Di) or Direction (Co; Be) ***Smooth term:*** Time (Week) ***Interaction term:*** Centrality x Distance (Co; CeDi; IsDi) or Centrality x Direction (Co; CeBe; IsBe)		All **control** mesocosms present the same pattern regardless location. ↓ Distance/Direction influence colonization patterns differently in both locations.
Model C (merging control & distant/between sets)	***Main terms:*** Distance (Control; Distant) or Direction (Control; Between) ***Smooth term:*** Time		Mesocosms colonization patterns are not influenced by **location**. ↓ Only distance/direction drive colonization.
Model D (merging central & isolated)	***Main terms:*** Centrality (Central; Isolated) ***Smooth term:*** Time		Mesocosms colonization patterns are not influenced by distance/direction . ↓ Only **location** drive colonization.

In order to determine which combination of explanatory variables was shaping the colonization process, all possible combinations of explanatory variables were tested (Table 6.1) in order to identify the more explicative one (Zuur et al. 2009). To do so, we begun with the full model that include all possible combinations of explanatory variables (Model A in Table 6.1). From there, we tested if it could be simplified merging explanatory variable levels (subsequent models from B to D in Table 6.1) and tested if this merging supposed a significant loss of information. The rationale behind this "merging-testing" procedure was as follow: if the explanatory variables were not important for describing the colonization pattern, then the merging of its levels would not suppose a significant loss of information. For example, if the observed colonization pattern was not influenced by the centrality-isolation gradient, we would expect similar trends in both locations. To test it, a comparison of a model that describes the trend separately for central and isolated sites with one with a joined trend (obtained merging both levels) has to be performed. Then, if when comparing both models a non-significant result is obtained, this would indicate that the pattern observed was the same regardless of the isolation-centrality gradient because no loss of information occurs when merging isolated and central sites. To select the best model among the ones tested we calculated the following values: 1) Akaike's information criterion of the second order (AICc); 2) the AICc differences among the tested models; and 3) the AICc weights (Burnham and Anderson 2002, Zuur et al. 2009). The best model corresponded to the one with the lowest AICc, with the AICc differences below 2, and with the highest AICc weight. This best model contains the combination of explanatory variables that better described the colonization trend observed and thus, which variables determined such trend. All GAMM models were done using the "gamm4" function available in the "gamm4" package (Wood and Scheipl 2013), and the AICc, the AICc differences and the AICc weights were obtained using the function "model.sel" available in the "MuMIn" package (Barton 2012). Both packages were written in R language (R-Core Team 2013).

Mesocosms post-drought community

After ponds complete drying, we did not remove the mesocosms and we carried a last survey one week after (July 27th 2016). In this case, we filtered the entire mesocosms water through a hand net with a mesh size of 250 µm. All organisms were preserved in 70% ethanol and posteriorly sorted, counted, identified to the maximum taxonomic level possible and classified according to their dispersal abilities (Tachet 2000, Heino 2013). We conducted a non-metric multidimensional scaling (nMDS) ordination, using the Bray- Curtis similarity index, and used PERMANOVA to test differences on mesocosms assemblages according to centrality and set typology (i.e., control, distant and between).

Pond and mesocosms dissimilarities

During the whole study, the two selected ponds were also surveyed in three occasions, at the beginning (May 18th 2016), the middle (June 16th 2016) and the end (July 12th 2016) of the hydroperiod following other studies (Boix et al. 2017). For each pond, integrated samples of aquatic macrofauna (amphibians and macroinvertebrates) were taken using a sampling effort proportional to the representativeness of each microhabitat. All sampled organisms were preserved in 70% ethanol and then sorted, counted and identified to the maximum level possible. We then conducted a non-metric multidimensional scaling ordination (nMDS) and PERMANOVA test, using the Bray-Curtis similarity index, to determine the level of dissimilarity between ponds and mesocosms communities. To make this comparison, we considered the mesocosms community composition of the same survey and then calculated the total average abundance of each species in each location (central and isolated). All nMDS ordination plots were done using "vegan" package available in R language (Oksanen et al. 2010). Similarity analyses and PERMANOVAs tests were conducted using Primer 6 and PERMANOVA+ program (Anderson et al. 2008).

6.3 Results

All mesocosms remained intact during the whole experiment although cattle presence. Water levels were mostly maintained due to rainfall and only additional filtered water was added in the last surveys to maintain mesocosms water levels, from June 22d until July 19th. Along the experiment mesocosms water characteristics differed between isolated and central locations. However, these differences were mainly due to the sampling daytime, normally having 2 hours of difference among central and isolated mesocosms measures (Supplementary S6.3A). We also found some differences among mesocosms sets, but only in some sampling surveys (Supplementary S6.3A). Although these detected differences, mesocosms water characteristics were largely like ponds water characteristics or varied within ponds variation range (Supplementary S6.3B, C and D).

In total, at the end of the experiment, twenty taxa arrived at the mesocosms, fourteen in the central ones and seventeen in the isolated ones. These taxa corresponded to three Coleoptera families (Dytiscidae, Helophoridae and Hydrophilidae) and four Diptera families (Chironomidae, Culicidae, Ephydridae and Ceratopogonidae). We observed some differences between isolated and central locations, with some families being more diverse in isolated mesocosms (Dytiscidae and Chironomidae; Supplementary S6.4). Furthermore, some Anura adults were also found in two isolated mesocosms. Finally, in some mesocosms of the isolated location we unexpectedly found Turbellaria, Cladocera, Copepoda and Ostracoda individuals, these arrivals happened only in three mesocosms and therefore were considered as incidental findings and not further studied. For more detailed information on found species, see supplementary S6.5A.

Mesocosms colonization.

Organisms abundance in all mesocosms presented a similar evolution along the hydroperiod in all set types, constantly increasing. However, in isolated pond mesocosms, the between set presented larger values than the control and the distant sets whereas in central pond mesocosms all three sets had more or less similar values (Supplementary S6.6). A similar pattern was observed with accumulated richness between central and isolated locations and among the different mesocosm sets of each location (Supplementary S6.6).

Centrality, distance and direction played different roles in determining the observed colonization trends also being differently important in relation to dispersal ability and accumulated richness (Table 6.2 and supplementary S6.7). Thus, the location of the source pond (central or isolated) is important in the sense that, in general, the isolated site showed less marked trends. Thus, abundances and accumulated richness usually showed higher values in the central location (Table 6.2 and Figure 6.3). The only two cases in which the location of the ponds seems of no importance is in the case of weak dispersers abundance and accumulated richness, when taking into account direction (so merging control sets and between sets). In these cases, no differences were found between central and isolated sites (Table 6.2 and Figure 6.4).

Focusing on the importance of the distance based analyses, our results indicated that distance was only important for weak dispersers (Table 6.2 and Figure 6.3). Thus, distant mesocosms showed similar colonization trends for weak dispersers, regardless of its location on the network (i.e., close to a central or an isolated source pond). In fact, the most marked trend, with rapid rise in abundance, was obtained in mesocosms located close to the central pond (i.e. central control set), whereas the weaker trend (smaller increase and lower abundances) was obtained in mesocosms located close to the isolated pond (i.e. isolated control set). In between these two trends, we found the trend observed for the weak dispersers but in distant mesocosms (Figure 6.3A). On the other hand, direction based analyses results suggested that the direction was relevant

only for active dispersers (Figure 6.4). Here, the between sets, regardless of its location in a central or isolated part of the network, had intermediate abundance values and presented the same abundance peak of central control sets at week six, but less marked. Isolated control sets presented a flatter pattern (Figure 6.4B).

Finally, it is important to stress that we detected different colonization trends according to species dispersal ability, indicating different patterns between them. Weak active dispersers abundance constantly increased along the hydroperiod, while strong active dispersers presented a marked peak, more evident in control sets (situated closer to the source ponds), in their abundance in week number six of the experiment (Figure 6.3B and Figure 6.4B) and probably linked with pond depth loss (Supplementary S6.8).

Table 6.2: *Best selected GAMMs for each considered explanatory variable: weak dispersers abundance - distance (control sets versus distant sets), weak dispersers abundance direction (control sets versus between sets), strong dispersers abundance - distance (control sets versus distant sets), strong dispersers abundance - direction (control sets versus between sets), accumulated richness - distance (control sets versus distant sets), accumulated richness - direction (control sets versus between sets.*

Distance based analyses				
Weak dispersers — **Model B3**				
Terms:	*Estimate*	*Std. Error*	*z value*	*Pr(>|z|)*
Central & Control	1.36	0.34	4.05	0.00
Isolated & Control	**-1.03**	**0.48**	**-2.17**	**0.03**
Distant	0.51	0.39	1.31	0.19
Model R-sq.(adj.)	0.58			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central & Control	**3.46**	**3.46**	**30.88**	**<0.01**
Week : Isolated & Control	**1.86**	**1.86**	**25.73**	**<0.01**
Week : Distant sets	**3.51**	**3.51**	**74.25**	**<0.01**
Strong dispersers — **Model D**				
Terms:	Estimate	Std. Error	z value	*Pr(>|z|)*
Central	-0.04	0.18	-0.24	0.81
Isolated	**-1.38**	**0.33**	**-4.13**	**0.00**
Model R-sq.(adj.)	0.61			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central sets	**5.73**	**5.73**	**47.16**	**<0.01**
Week : Isolated sets	1.00	1.00	0.50	0.48
Accumulated richness — **Model D**				
Terms:	*Estimate*	*Std. Error*	*z value*	*Pr(>|z|)*
Central	0.96	0.09	10.37	<0.01
Isolated	**-0.77**	**0.17**	**-4.62**	**<0.01**
Model R-sq.(adj.)	0.68			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central sets	**1.68**	**1.68**	**25.64**	**<0.01**
Week : Isolated sets	**1.86**	**1.86**	**33.22**	**<0.01**
Direction based analyses				
Weak dispersers — **Model D**				
Terms:	*Estimate*	*Std. Error*	*z value*	*Pr(>|z|)*
Central	1.39	0.20	7.02	0.00
Isolated	-0.35	0.27	-1.31	0.19
Model R-sq.(adj.)	0.44			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central sets	**2.69**	**2.69**	**50.01**	**<0.01**
Week : Isolated sets	**1.00**	**1.00**	**41.46**	**<0.01**

Strong dispersers — Model B3				
Terms:	Estimate	Std. Error	z value	*Pr(>|z|)*
Between	-0.51	0.21	-2.37	0.02
Central & Control	**0.73**	**0.33**	**2.23**	**0.03**
Isolated & Control	-0.42	0.40	-1.07	0.29
Model R-sq.(adj.)	0.49			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central & Control	**4.11**	**4.11**	**14.68**	**<0.01**
Week : Isolated & Control	1.00	1.00	0.37	0.54
Week : Between sets	**5.11**	**5.11**	**32.10**	**<0.01**
Accumulated richness — Model D				
Terms:	*Estimate*	*Std. Error*	*z value*	*Pr(>|z|)*
Central	0.66	0.13	5.24	0.00
Isolated	-0.04	0.18	-0.21	0.83
Model R-sq.(adj.)	0.56			
Smoothing terms:	*edf*	*Ref.df*	*Chi.sq*	*p-value*
Week : Central sets	**1.00**	**1.00**	**30.07**	**<0.01**
Week : Isolated sets	**1.63**	**1.63**	**39.31**	**<0.01**

Post-drought mesocosms community

Similar to the observed colonization trends, after ponds complete drought, all central mesocosms had similar values of abundance, whereas the between set of isolated mesocosms had greater abundances of organisms than the other two isolated sets (Supplementary S6.9). These differences were also observed for accumulated richness (Supplementary S6.9). Again, centrality was determinant for both weak dispersers (Figure 6.5A) and strong dispersers community composition (Figure 6.5B). The PERMANOVA results corroborated these differences between central and isolated locations (Weak dispersers Pseudo-$F_{1,18}$= 4.23, p<0.01 and strong dispersers Pseudo-$F_{1,14}$= 6.28, p<0.001). In the other hand, no significant differences were found regarding the type of mesocosm set (Weak dispersers Pseudo-$F_{2,18}$= 1.38, p=0.22 and strong dispersers Pseudo-$F_{2,14}$= 6.0.93, p=0.51) nor for the interaction between centrality and the type of mesocosm set (Weak dispersers Pseudo-$F_{2,18}$= 1.92, p=0.07 and strong dispersers Pseudo-$F_{2,14}$= 1.45, p=0.21).

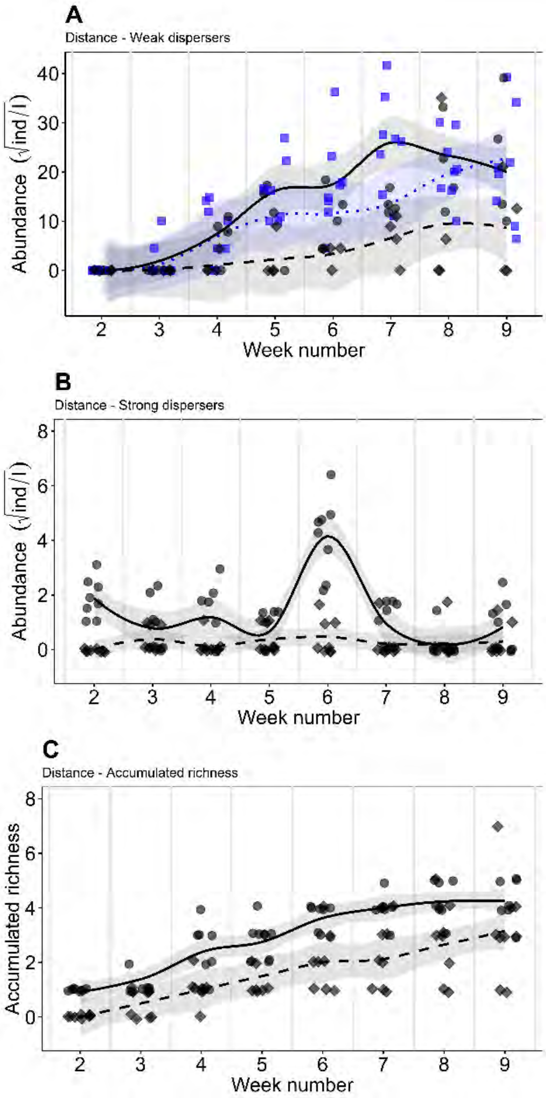


Figure 6.3: *GAMMs results for distance comparisons (control sets versus distant sets) along time (surveyed weeks) for A: Weak active dispersers abundance (square rooted individuals per litre), B: Strong active dispersers abundance (square rooted individuals per litre) and C: Accumulated richness of both groups. Each panel shows the best selected model that explains the observed colonization pattern (based on Table 6.2). Weak active dispersers (Panel A) responded to the B3 model, having 3 distinguishable patterns (Central Control sets (solid black line and black circles), Isolated control sets (dashed black line and black diamonds) and central and isolated distant sets considered together (dotted blue line and blue squares)). Strong active dispersers (Panel B) and accumulated richness (Panel C) responses better fitted with C model, having only 2 distinguishable patterns linked to centrality: all central sets (solid black line and black circles) and all isolated sets (dashed black line and black diamonds). Confidence bands indicating 95% confidence interval.*

Pond and mesocosms dissimilarities

Differences between ponds and mesocosms were clearly marked by several taxa that were only present in the ponds. Some groups such as Trichoptera did not arrive in the mesocosms although they were highly abundant in the ponds as well as some Culicidae species, that did not colonise mesocosms (Supplementary S6.5B). Consequently, PERMANOVA results indicated differences between ponds and mesocosms (Pseudo-$F_{1,11}$= 16.70, p=0.02), but not in their interaction with time and centrality (Pseudo-$F_{2,11}$= 2.84, p=0.11 and Pseudo-$F_{1,11}$= 2.02, p=0.23, respectively) indicating that despite differences in composition the trend was similar along time. The nMDS ordination plot

suggested some temporal organization of the samples and differences between ponds and mesocosms communities (Figure 6.6). However, differences related to time were non-significant (Pseudo-$F_{2,11}$= 3.22, p=0.09) neither for centrality (Pseudo-$F_{1,11}$=2.18, p=0.2) nor their interaction (Pseudo-$F_{2,11}$=0.9, p=0.5).

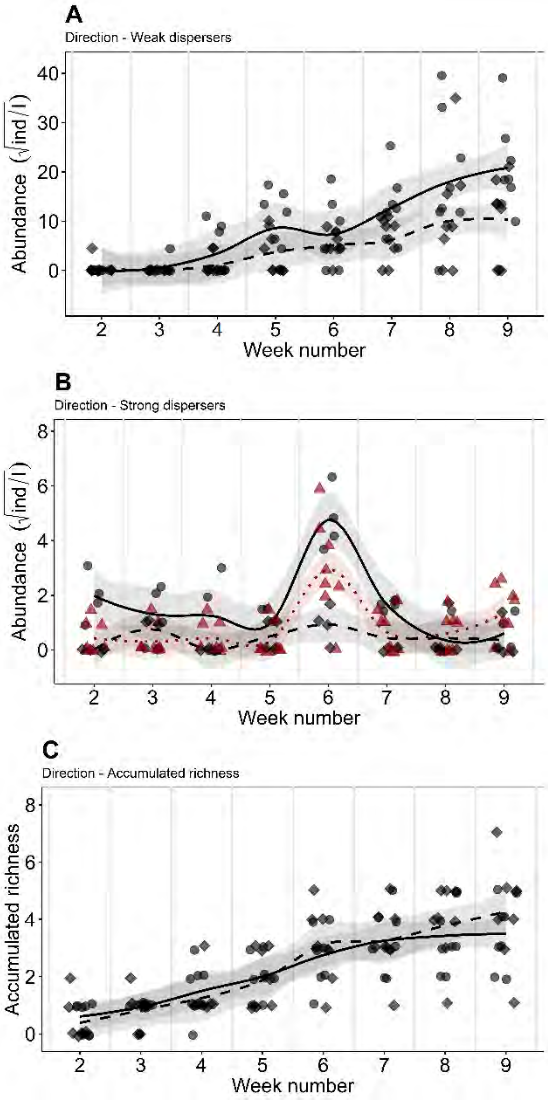


Figure 6.4: GAMMs results for direction comparisons (control sets versus between sets) along time (surveyed weeks) for A: Weak active dispersers abundance (square rooted individuals per litre), B: Strong active dispersers abundance (square rooted individuals per litre) and C: Accumulated richness of both groups. Each panel shows the best selected model that explains the observed colonization pattern (based on Table 6.2). Weak active dispersers (Panel A) and Accumulated richness (Panel C) responded to the C model, having only 2 distinguishable patterns linked to centrality: all central sets (solid black line and black circles) and all isolated sets (dashed black line and black diamonds). Strong active dispersers (Panel B) responded to the B3 model, having 3 distinguishable patterns: Central Control sets (solid black line and black circles), Isolated control sets (dashed black line and black diamonds) and central and isolated between sets considered together (dotted red line and red triangles). Confidence bands indicating 95% confidence interval.

6.4 Discussion

The relevance in metacommunity dynamics of habitats location along the centrality-isolation gradient has been proposed as one of the main drivers of metacommunity assembly (Chase et al. 2010, Borthagaray et al. 2015a). The studied ponds location –in two opposite extremes of the centrality-isolation gradient– influenced colonization trends being mostly the best explanatory variable of the temporal pattern for weak and strong dispersers abundance and for accumulated richness. Generally, and accordingly with our results, greater values of abundance and richness were detected in central locations than in isolated ones (Henriques-Silva et al. 2015). Furthermore, both distance and direction affected the observed patterns appearing as additional modulators for colonization patterns. Divergent metacommunity dynamics between organisms having different dispersal abilities have been documented in several studies (Heino 2013, Borthagaray et al. 2015b, Cañedo-Argüelles et al. 2015, Jones et al. 2015, Castillo-Escrivà et al. 2017). However, the importance of smaller regional scale variables such as direction to the closest pond or distance between studied habitats can also play a role as additional metacommunity assembly drivers (Resetarits and Binckley 2013, Morán-Ordóñez et al. 2015, Gall et al. 2017). Therefore, the smaller regional context might act as a modulator of the wider regional pattern and consequently both are playing a role in the metacommunity assembly process (Kneitel and Chase 2004, Heino and Tolonen 2017). The centrality-isolation gradient also drove mesocosms composition differences, but in this case without having differences due to distance nor direction comparisons. Thus, communities differed in the bigger regional scale (i.e., centrality-isolation gradient) but not in the smaller regional one (i.e., distance or direction).

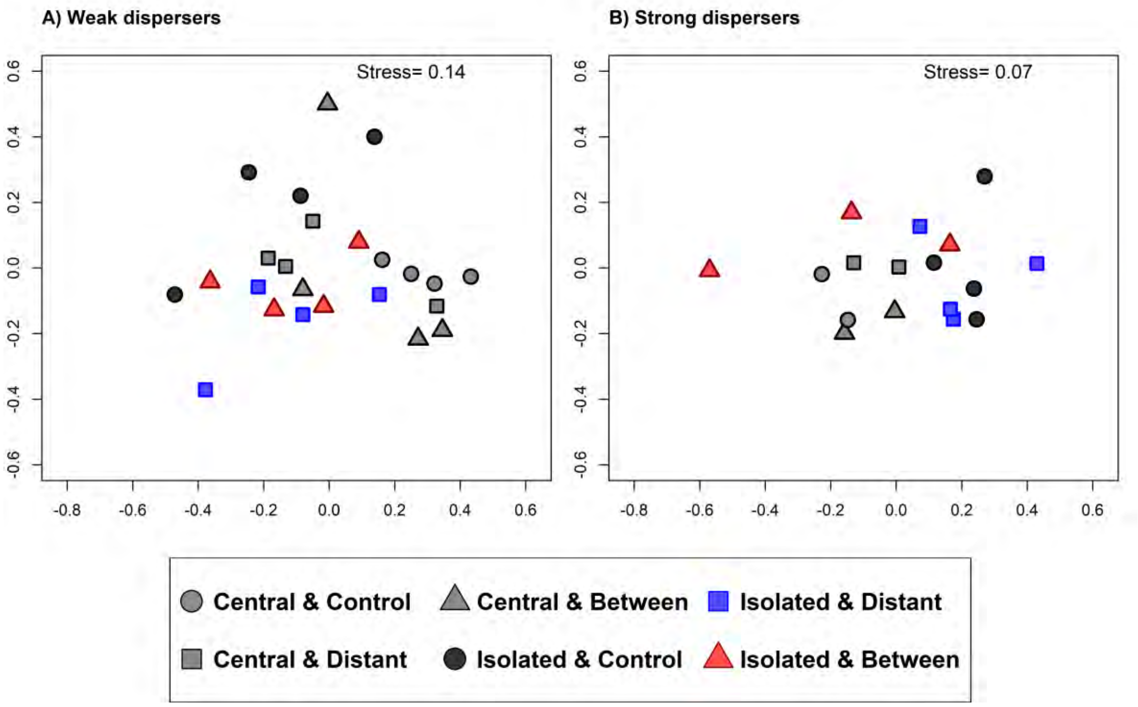


Figure 6.5: *nMDS plot for weak active dispersers (Panel A) and strong active dispersers (Panel B) mesocosms communities at the post-drought survey (one week after ponds complete drying).*

Nevertheless, organisms dispersal ability interacted with all these observed trends along the centrality-isolation gradient. Weak dispersers presented a great variation between mesocosms and followed almost the same pattern during the experiment, while strong dispersers presented clearly differentiated patterns between central and isolated locations. Moreover, colonization trends were markedly different across weak and strong dispersers responding to their different landscape perception but probably to their different weather perception too (Jonsson et al. 1980, Gardarsson et al. 2004, Csabai and Boda 2005, Bouchard 2007, Csabai et al. 2012, Borthagaray et al. 2015b). Weak dispersers increased more or less constantly along the hydroperiod and similarly across locations, this pattern might suggest a constant increase of individuals flux that would respond to an improvement of weather conditions along the hydroperiod (e.g., higher temperature, lower wind; Boda and Csabai 2009, Boix et al. 2011). On the other hand, strong dispersers presented a marked peak during the sixth week of the experiment, which coincided with a decrease in ponds level, more pronounced in the central pond, due to lack of rainfall and temperatures rise. Thus, strong dispersers responded rapidly to pond fluctuation. Nevertheless, this rapid response to pond water level decrease by strong dispersers was different between

control sets in central and isolated locations being higher in central ones. Interestingly, mesocosms sets located between ponds, both central and isolated, behaved similarly to the central control sets also presenting a peak in abundance at week six. Hence, when situated between two ponds, the effect of location in organism abundances was attenuated due to an increase in individuals flux, which was triggered by pond water level decrease (Yee et al. 2009, Boix et al. 2011, Boda and Csabai 2012). Likewise, when accounting for differences in accumulated richness, direction (control sets versus between sets) did not generate differences for the isolation-centrality gradient, probably as a consequence of the greater flux between ponds that increased species arrival and thus, compensated the isolation effect (Cadotte 2006, Chase and Shulman 2009, Economo and Keitt 2010). Habitat suitability (e.g., wet corridors) can drive individual movement influencing community assembly processes (Winterbourn and Crowe 2001, Oertli et al. 2008, Bogan and Boersma 2012, Cañedo-Argüelles et al. 2015). Similarly, in this study, the direction towards more suitable habitats (i.e., another pond) changed the observed trend for strong dispersers in isolated between sets, making them similar to centrally located ones. Thus, mitigating the general observed pattern of smaller values in isolated locations.

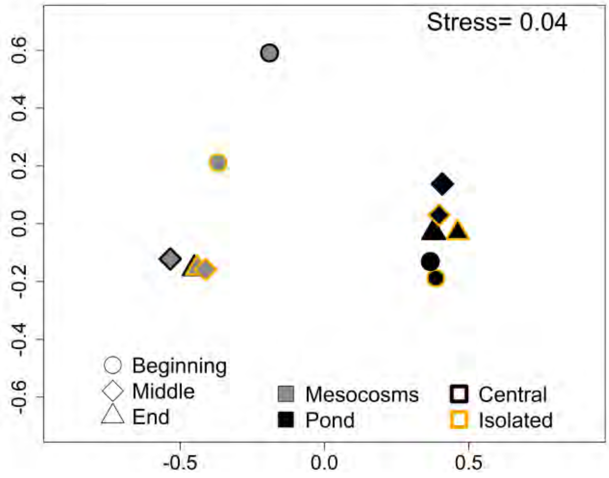


Figure 6.6: *nMDS plot for the two source ponds (central and isolated) and the avergage mesocosms communities composition of each location (central mesocosms and isolated mesocosms) corresponding to the three surveys carried on the begining of the hydroperiod (small triangles), the middle of the hydroperiod (medium triangles) and the end of the hydroperiod (big triangles). Ponds in black triangles and mesocosms in grey triangles. Central locations triangles have black contour and isolated locations triangles have orange contour.*

Ponds and mesocosms shared several species, mainly of Coleoptera and Diptera, but there were also some groups that did not colonize mesocosms such as Trichoptera or Odonata species. These differences are mainly linked to the effect of the mesocosm (de Szalay et al. 1996, Williams et al. 2007), which may represent non-suitable habitats for some taxonomic groups for example due to the lack of emergent vegetation, some specific resources or impossibility for eggs laying (Wiggins et al. 1980). Nevertheless, ponds and mesocosms were influenced equally by successional changes. Temporal changes were strong along the hydroperiod going from snow covered shores to completely vegetated pond bottoms. Such changes determined communities composition along time being hydroperiod beginning the more differentiated in terms of composition in both ponds and mesocosms and regardless of their location. Such fluctuation was not affected by the fact of being a pond or a mesocosm nor by location nor by distance or direction. Hence, indicating that a greater temporal pattern, mainly linked to environmental and habitat changes –of both ponds and mesocosms–, was strongly determining communities successional changes (Williams 2006, Boulton et al. 2014, Boix et al. 2016, Wissinger et al. 2016).

Along this study, mesocosms usage was proved to work well in partially capturing ponds biodiversity and also organisms movement among studied sites. Nevertheless, we must assume that we did not capture the whole picture of the metacommunity assembly process. We could not account with complete pond biodiversity (e.g., egg-bank), which can strongly determine pond community assembly (Wisnoski et al. 2019) and the already commented mesocosm effect that increased differences between ponds and mesocosms communities. However, although what we obtained was a partial snapshot of the hydroperiod succession process and temporary ponds assembly process, mesocosms did help in specifically capture weak and strong dispersers movement among ponds and the regional perspectives (i.e., wide and small scale) influencing such movement.

Divergent patterns between central and isolated locations might appear due to the differences in the number of available habitats or the presence or absence of determined organisms (Chase and Shulman 2009, Chase 2010, Borthagaray et al. 2015a, Trekels and Vanschoenwinkel 2017). In fact, central locations presented higher number of individuals, similar communities but on the other hand, a smaller number of species. As expected, these values would suggest a greater role of mass effects processes and stronger competitors in central locations while the opposite pattern is observed in more isolated ones (Borthagaray et al. 2015a). Of course, environmental changes (e.g., weather or pond depth) also determined these dynamics as could be expected for temporary ponds (Florencio et al. 2014, Boix et al. 2017, Gianuca et al. 2017, Hill et al. 2017a). Nevertheless, although the general picture suggested a general mass effect along the central-isolation gradient, when considering direction, we saw an attenuation of this general pattern in isolated between mesocosms. Thus, although general patterns can be pictured along wide regional scales (i.e., network centrality-isolation gradient), smaller regional scales (i.e., direction towards suitable habitats or distance from source habitat) can modulate this more general pattern as well as the different landscape perceptions -dispersal abilities- of metacommunity organisms. Here, we observed that there are different landscape layers shaping dispersal dynamics (wide, small and species-specific perceptions) in a real field experiment that integrated different approaches (i.e., mesocosms and network structure analysis). To consider all these different layers appears as a powerful approach to specifically test dispersal driven processes (Bogan and Boersma 2012, Tonkin et al. 2018) and must become a key step to unravel landscape –at all scales– influence on metacommunity assembly.

7. CHAPTER VI

Metacommunity network structure determines nonlinear transitions in biodiversity resilience to wildfire.

Chapter brief summary

Disturbances are the main drivers of biodiversity dynamics and ecosystem functioning at regional scales. Wildfires represent natural disturbances, but their frequency and intensity are expected to increase because of global change. At the landscape level, wildfires generate a mosaic of disturbed and undisturbed patches, changing the current metacommunity network structure, which has been recognized as a key determinant of community resilience. Here, we combine empirical information with metacommunity simulations to advance the understanding of the role of metacommunity network structure in biodiversity resilience to wildfire. The response of a pond's metacommunity to a wildfire that burned 13,000 hectares in the NE Iberian Peninsula in 2012 involved minor changes among strong dispersers but the overrepresentation and underrepresentation of weak active and passive dispersers, respectively. The dispersal distances of organisms were used to determine linkage distances between local communities, defining metacommunity network structures for the 542 water bodies recorded in the area. For each taxon-dependent metacommunity network, the local communities' resilience to wildfire was analyzed by running a spatial explicit lottery model for gradients of wildfire size and intensity. Biodiversity resilience was nonlinearly related to metacommunity network structure and wildfire disturbance, indicating a sharp transition from high to low resilience with increases in wildfire size and intensity. This transition is expected at the percolation distance—the minimum distance that allows organisms to reach all landscape patches. However, it was observed at half the percolation distance, indicating a more complex connection between biodiversity and landscape structure than expected. The strong resilience of the metacommunity can become compromised in the coming future, since resilience is subject to specific conditions that can be surpassed with the predicted increase in fire impacts. Thus, theoretical approaches incorporating a regional perspective can provide valuable insights into understanding metacommunity dynamics and coping with future fire scenarios.

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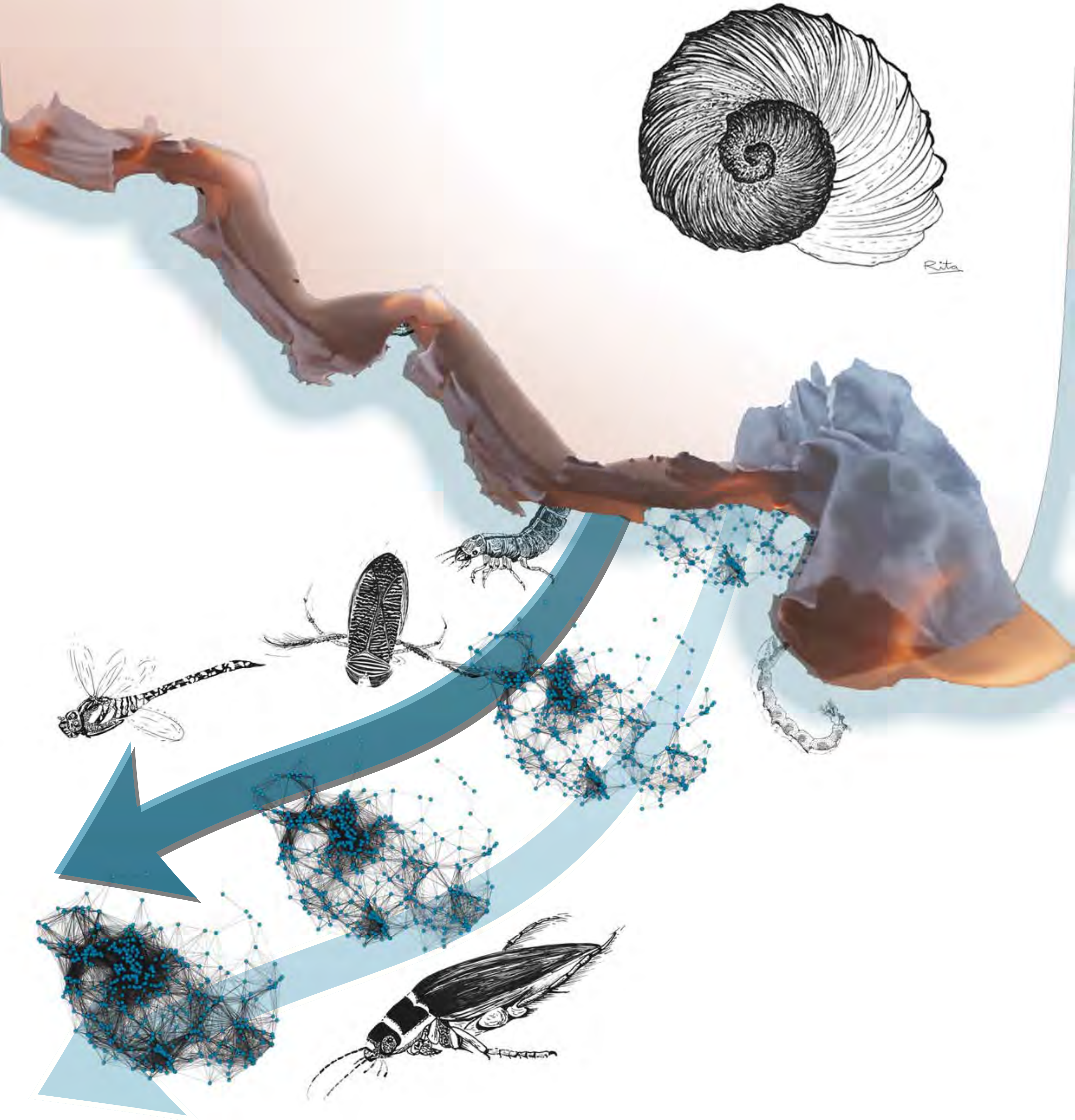
8. GENERAL DISCUSSION



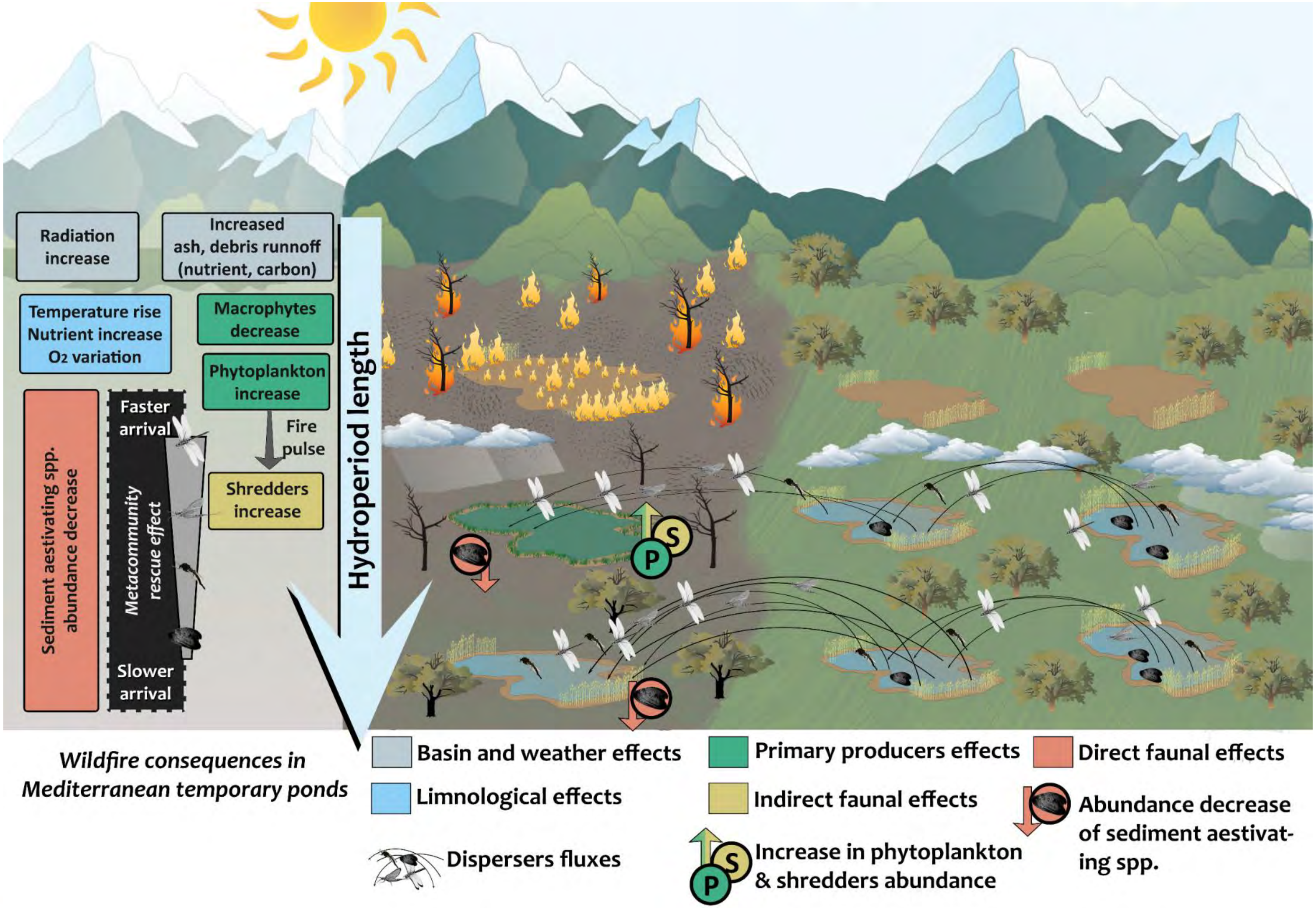
8.1 Wildfire impacts on Mediterranean temporary ponds

Several works have dealt with wildfire impacts on aquatic systems (Gresswell 1999, Verkaik 2010), though mostly on permanent rivers and lakes (Bixby et al. 2015). Thus, some main wildfire consequences on aquatic habitats can be obtained from that, but the relevance of the specific ecosystem context (McCullough et al. 2019) can still provide new insights on wildfire consequences on aquatic habitats, in our case, Mediterranean temporary ponds.

Drought is one of the main determinants of Mediterranean temporary ponds faunal communities (Williams 2006, Boulton et al. 2014, Boix et al. 2016) and in the context of wildfire disturbance, drought is also determining its impacts on macroinvertebrate communities (*Chapter I*). The fact that Mediterranean temporary ponds are mainly dry during summer —wildfire season— opens a new path of disturbance on the aquatic habitat which is the direct burning of sediment-aestivating organisms (such as Oligochaeta and Gasteropoda). Gresswell (1999) stated two main different types of wildfire effects: direct and indirect. Direct effects where mainly related to impacts happening during the wildfire event (e.g., rise in water temperature, mortality, water toxicity, etc.), while indirect effects occurred after the wildfire and along time (e.g., vegetation changes, sediment increase, algal blooms, etc.). But, interestingly, direct impacts where mainly linked to fish and macroinvertebrates mortality due temperature rise in small tributaries (Christensen et al. 1989) and toxicity due to smoke ammonia diffusion (Spencer and Hauer 1991). Of course, the proposed framework by Gressweel (1991) and Minshall et al. (1989) was in the context of permanent temperate streams and rivers and therefore, it did not account with the fact of being dry during the wildfire. In this thesis, the decrease in abundance of species remaining in the pond sediment during drought in highly burned ponds suggested that a new direct effect happened in temporary lentic systems such as Mediterranean temporary ponds: the direct burning (*Chapter I*). Consequently, one of the main wildfire direct effects on Mediterranean temporary ponds, also expected in other temporary waters (i.e., temporary streams), is the direct burning of organisms aestivating in the dry sediment (Figure 8.1).

While direct wildfire effects on Mediterranean temporary ponds seemed to provide new perspectives in the theoretical framework of wildfire disturbances and aquatic systems, indirect effects appeared more similar to permanent lentic systems consequences. An increase of nutrients (i.e., nitrogen and phosphorus), phytoplankton concentration and some trophic macroinvertebrate groups abundance following a fire pulsed effect are the main expected consequences for lentic systems (Scrimgeour et al. 2001, Malison and Baxter 2010a, Beganyi and Batzer 2011, Lewis et al. 2014, Venne et al. 2016, McCullough et al. 2019). Therefore, indirect effects in temporary lentic systems faunal communities would be similar to those observed in permanent lentic systems (Figure 8.1). Other consequences of wildfire similar to permanent systems would be the ones related to environmental characteristics such ash accumulation (i.e., increase in nutrient content) or macrophyte decrease (for streams canopy loss), which would contribute to the fire pulsed effect (Malison and Baxter 2010a). Having observed all these consequences we could try to build a conceptual framework of what would be expected of wildfire disturbance on Mediterranean temporary ponds faunal communities, which is represented in Figure 8.1. Nevertheless, in this conceptual framework, we must consider time, understood as hydroperiod successional changes —Figure 8.1 top (hydroperiod beginning) to Figure 8.1 bottom (hydroperiod end)— that as it has been seen along this thesis (*Chapter I* and *Chapter II*) is key to pond recovery. In this case, direct impacts lasted along the whole hydroperiod with directly burned species having always lower abundance values (Figure 8.1 red square), whereas indirect wildfire impacts where mainly concentrated at the beginning of the hydroperiod (i.e., fire pulsed effect; Figure 8.1 green and yellow squares).

Being temporary adds a new direct effect of wildfire on Mediterranean temporary ponds, which will already suffer wildfire indirect effects. However, this same factor of being temporary and consequently, have a drought-adapted and unpredictability-adapted community, might increase resilience to wildfire effects and thus, promote system recovery (i.e., less than one hydroperiod). Verkaik (2010) and Verkaik et al. (2013) highlighted the importance of Mediterranean lotic systems and their strong and frequent inter- and intra- annual variation in promoting aquatic communities adapted to such conditions. These adaptations concomitantly favoured resilience against less frequent, though recurrent, wildfire-induced effects. The fast basin vegetation recovery might accelerate Mediterranean streams recovery, returning to pre-fire conditions some years after the wildfire (Verkaik et al. 2015). Similarly, Mediterranean temporary ponds recovered rapidly and within one hydroperiod —less than one year was needed by the system to have recovered its pre-disturbance dynamics (*Chapter I*). Indeed, Mediterranean temporary ponds are characterised by having strong inter- and intra-annual unpredictability in both hydroperiod length and hydroperiod beginning (Florencio et al. 2009, Rouissi et al. 2014). Such unpredictable conditions promote an adapted-to-drought community in two aspects: some species can cope with drought by resisting in the dry pond sediment (e.g., buried or by resistance structures) while others can actively and rapidly move from permanent habitats to recently flooded temporary ones (Boix et al. 2016). Therefore, similarly to Mediterranean streams, the intrinsic characteristics of Mediterranean temporary ponds systems, related to frequent intra- and inter- annual variations, concomitantly increased their resilience to wildfire effects and their fast recovery. Indeed, only macroinvertebrate species having passive dispersal ability (i.e., those directly burned by wildfire) presented lower abundance levels after the disturbance. This suggested that recovery was strongly linked with dispersal capacity and thus, with species ability to reach affected habitats after the disturbance, being passive dispersers (such as Oligochaeta or Gasteropoda) the ones having greater difficulties to recover (*Chapter IV*; Figure 8.1 black square).



Wildfire consequences in Mediterranean temporary ponds

Figure 8.1: *Conceptual model of the reported impacts of wildfire on temporary ponds that can be drawn from this thesis chapters.* ***Left section:*** *First impacts are related to basin and post-wildfire meteorological conditions (grey boxes), linked with runoff and the increase in radiation. These initial effects are transmitted to the water body with the consequent increase in temperatures and nutrient, which also generate* O_2 *variation (blue box). Primary producers respond differently to such changes. Macrophytes present smaller abundances at the beginning of the hydroperiod while phytoplankton increase its abundance (green boxes) due to the increase in temperature and nutrients. These indirect consequences generate a pulsed effect (grey small arrow) which increase shredders abundance in burned ponds (light-green box). This fire pulse was concentrated at the beginning of the hydroperiod (initial and mid part of the hydroperiod blue arrow). On the other hand, species that aestivate in the pond sediment and have a passive dispersal ability decreased their abundances along the whole hydroperiod (until the end of the hydroperiod; blue arrow), thus being directly burned by wildfire (red box). The fact that some ponds remained unaffected, contributed to recolonization along the hydroperiod, but at different velocities according to dispersal ability (black box).* ***Right section:*** *Temporal evolution of studied ponds starting with wildfire (August 2012; top image) and finishing at the end of the hydroperiod (July 2013; bottom image). Dispersers fluxes increase among unburned and burned ponds along time after pond filling due to an improvement of weather conditions and burned ponds condition (first rains).*

8.2 Do not let wildfire burn what the network bounds: The metacommunity rescue effect

Wildfire disturbance usually generates a patchy landscape with great heterogeneity (Wright and Bailey 1982, White and Pickett 1985, Turner et al. 2015), but the overall results obtained in this thesis stress the importance of the pond network structure —hereafter pondscape— to better understand the subsequent recovery. In this sense, the simulations conducted in *Chapter* assumed system neutrality and thus, provided an explicit evaluation of the landscape influence on metacommunity dynamics (Urban and Keitt 2001, Economo and Keitt 2008, 2010, Muneepeerakul et al. 2008). Therefore, the reported rapid recolonization by the simulations was mostly linked to system specific landscape structure thus, proving that the pondscape was playing a key role in post-disturbance recovery. In fact, our results showed that the studied pondscape, promotes metacommunity resilience against wildfire disturbance for most dispersal abilities. Only individuals with poor dispersal were expected to have their recovery compromised after the wildfire (*Chapter IV*). Moreover, the high density of water bodies in the studied network, with burned and unburned ponds in very central locations (see Figure 8.2), favoured a rapid recolonization process which did not required great times to reach and recolonize burned habitats. Indeed, distances among burned and unburned ponds where not great, being less than 2 km in some cases. Therefore, most species inhabiting those systems where capable to reach burned sites based on their dispersal distances, going from weak or moderate active dispersers with flying adults (Kovats et al. 1996, Nilson 1996, Delettre and Morvan 2000, Dettinger-Klemm 2003, Vallenduuk and Moller Pillot 2007, Astorga et al. 2012) to strong active dispersers with flying adults (Nilson 1996, Conrad et al. 1999, Angelibert and Giani 2003, Chaput-Bardy et al. 2010, Miller and Bergsten 2016). In the case of macroinvertebrates that disperse passively and have aquatic adults, these distances might have been within their maximum dispersal capacity ranges but still within them (Van De Meutter et al. 2007, Kappes and Haase 2012 and refferences therein), and so might be the ones having more difficulties to

recolonize (i.e. needing more time; *Chapter IV*). However, as it was seen in *Chapter III*, and in agreement with previous studies (Economo and Keitt 2010, Altermatt 2013, Borthagaray et al. 2015a), the central location of studied ponds (both burned and unburned) within the network, may have fostered dispersal with the consequent enhancement of recolonization processes (Figure 8.2). Centrality was calculated using the closeness metric which considers all network points to calculate how central or isolated are each one from the rest. Once these values were obtained (Figure 8.2), they clearly showed the marked central location of surveyed ponds of the Albera region, reinforcing the idea of the relevance of location within the network. Consequently, location determined community assembly (*Chapter III*), but also community resilience, greatly increasing macroinvertebrate arrival and contributing to the observed post-fire fast recovery (*Chapter I*).

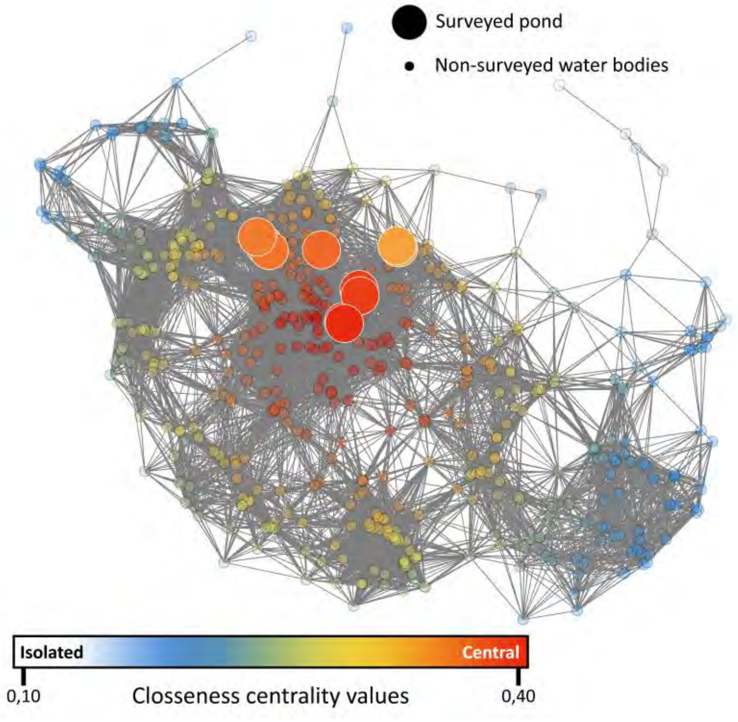


Figure 8.2: *The studied network graph of the Albera region. Surveyed ponds are highlighted by their greater size. Colour gradient indicates closeness centrality values ranging from 0,10 to 0,40. Closeness centrality quantifies how central or isolated is a network node (i.e., water body) from the rest of the nodes in the whole network. Note that surveyed ponds present orange to red colours indicating their high centrality within the studied network and the high density of links (grey lines) around sampled ponds, which imply greater fluxes of individuals among nodes in that area.*

Moving toward a metacommunity perspective, the central location within the network of burned ponds coupled with the abundance of water bodies around the area probably helped recolonization through dispersal and the consequent greater influence of mass effects (*Chapter III;* Chase and Shulman 2009, Economo and Keitt 2010, Borthagaray et al. 2015a, Cadotte and Tucker 2017) during the post-fire hydroperiod. However, this rapid colonization was coupled with the post-wildfire patchy landscape, which leaved unburned ponds within the burned area or some partially affected ponds (*Chapter I*) that acted as colonization sources of the surrounding burned ponds (*Chapter III*). Wildfire heterogeneity appeared as key to metacommunity fast recovery and determined metacommunity resilience when simulating different and more catastrophic wildfires scenarios (*Chapter IV*). Consequently, wildfire severity was strongly modulated by its intensity (i.e., number of water bodies affected within the burned area) and thus, on post-fire network fragmentation which is key for ponds biodiversity but also for general systems biodiversity maintenance (Scheffer et al. 2012, Calhoun et al. 2017, Horváth et al. 2019, Vidal et al. 2019).

To sum up, the observed rapid recovery during the post-disturbance hydroperiod mainly responded to the intrinsic resilience of Mediterranean temporary ponds faunal communities, enhanced for being a drought-resistant Mediterranean community (Verkaik et al. 2013, Boix et al. 2016), but also by the structure of the ponds network or pondscape (*Chapter IV*) and the particular wildfire behaviour, which partially affected the area. This rapid recovery was also strongly mediated by dispersal and recolonization of individuals from unburned ponds in what could be understood as a *metacommunity rescue effect* (Leibold et al. 2004).

Although its relevancy, the role of the egg-bank (i.e., species, mainly microcrustaceans, which remained in the sediment and survived wildfire) remained as not specifically considered, neither in simulated models (C*hapter IV*) nor in the field-experiment (*Chapter III*). Its role in determining post-disturbance succession in Mediterranean temporary ponds should not be disregarded. Indeed, the relevance of dormancy in driving metacommunity dynamics has been not greatly addressed (Leibold and Chase

2018, Wisnoski et al. 2019), although it can be a key determinant for some communities such as temporary ponds (e.g., priority effects; Fukami 2015). The egg-bank and propagules that remain in the sediment after ponds drying will be the first pioneers when ponds refill again and thus, will determine the initial post-fire community (O'Neill 2016, Wisnoski et al. 2019). The post-fire communities that we simulated (*Chapter IV*) did not account with a part of the community remaining in the pond sediment during the dry phase (we simulated the worst-case-scenario of community complete destruction). Therefore, we might assume that in fact, the real scenario will still be more positive than what was pictured, due to the coupled influence of an aerial mass effect (*Chapter III* and *Chapter IV*), but also of a ground mass effect that surely promoted the observed fast recovery.

Future perspectives of wildfire disturbance are basically an increase in area and intensity, both globally and in the Mediterranean regions (Pausas and Fernández-Muñoz 2012, Kovats et al. 2014, Botija Llasat et al. 2016, Turco et al. 2018), something that pictures a threatening scenario for Mediterranean temporary ponds. Currently we can assume that Mediterranean temporary ponds seem resilient to wildfire disturbance due to their intrinsic resilience (*Chapter I and Chapter II*) but our results indicate that their recovery depend on their location within the network, the magnitude of network affectation, and also on its own structure (*Chapters I, Chapter III and Chapter IV*). Moreover, future scenarios might compromise community recovery also for great active dispersers if for example, wildfires affect greater areas or more importantly completely burn all water bodies within the affected area —high intensity (*Chapter IV*). Wildfire recurrence can also exacerbate metacommunity recovery also affecting egg-bank hatching capacity (Wells et al. 1997, Chittapun 2011, Johnston et al. 2014, Whitney et al. 2015). Therefore, the current assumption of high resilience of Mediterranean temporary ponds metacommunities must not be understood as a non-worrying fact. There is a delicate balance and the shift from resilient network to catastrophic consequences can happen abruptly and having a strong non-linear behaviour (Urban and Keitt 2001, Scheffer 2009, Gilarranz and Bascompte 2012, Borthagaray et al. 2014). This non-linear shift was linked to disturbance intensity (i.e., heterogeneity; *Chapter IV*) and thus, to the existence of post-fire refugia that promote recolonization to unburned ponds or in other words habitat fragmentation.

Within temporary ponds threats, habitat disappearance is the main one (Skinner and Zalewski 1995, Wood et al. 2003), consequently, greater network fragmentation coupled with intense disturbance regimes (i.e., area, intensity and recurrence) may overcome the current Mediterranean temporary ponds resilience. Knowledge of pond responses to wildfires and other disturbances is key, but also the consideration of the whole pondscape as well as comprehend its implications on system dynamics —metacommunity assembly mechanisms— and recovery are fundamental elements for temporary ponds conservation in order to cope with future scenarios (Calhoun et al. 2017, Horváth et al. 2019).

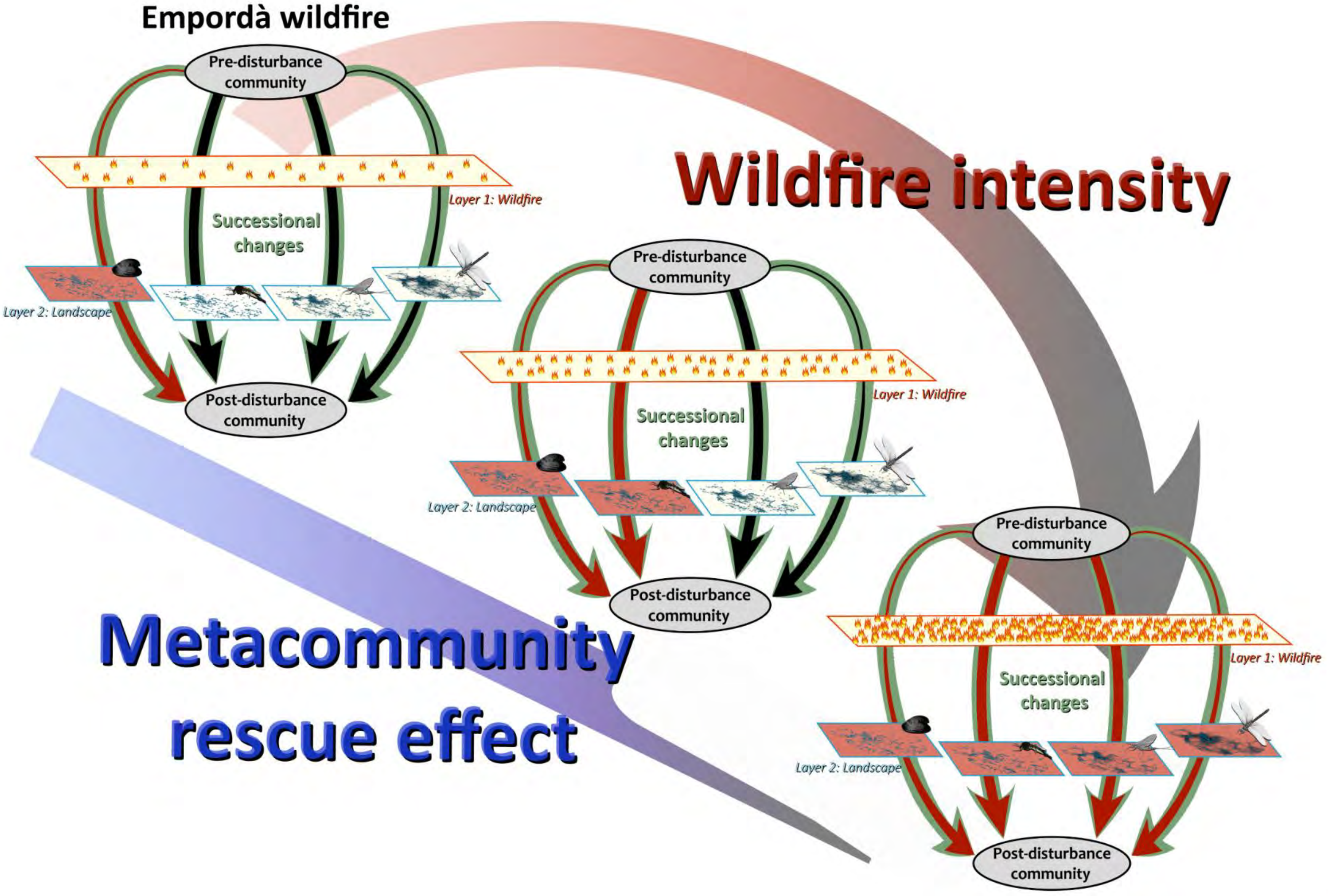
8.3 Neither one nor the other, disturbance consequences on metacommunity assembly: Surfing along the balance.

Disturbance consequences and their influence on metacommunity dynamics constitute one of the great questions in current community disturbance theory (Pulsford et al. 2016). Several divergent responses have been reported after a disturbance in metacommunities from an increase of species sorting dynamics (niche forces; Urban 2004, Laliberté et al. 2013, Han et al. 2018, Zhang et al. 2019) to an increase in stochasticity (neutral forces; Catano et al. 2017, Dong et al. 2017, Rosati et al. 2017) and also a strong influence of mass effects after post-disturbance (Vanschoenwinkel et al. 2013, González-Trujillo et al. 2019, Hernández-Ordóñez et al. 2019). Consequently, it is difficult to have one unique response to the question of how disturbances are going to affect metacommunity dynamics. However, at the same time, by assuming that any archetype can be fostered after a disturbance we are already picturing an answer, as with wildfire and aquatic systems (McCullough et al. 2019): it will strongly depend on system (i.e., metacommunity) and disturbance contexts (e.g., intensity of the disturbance, type of disturbance, etc.). Consequently, to try to specifically determine

which is the archetype by which post-disturbance metacommunity is being assembled, when all types can be detected, is still maintaining the more closed view on metacommunity ecology. In order to move towards a *2.0. version of metacommunity ecology,* more integrative approaches are needed (Logue et al. 2011, Leibold and Chase 2018), where the understanding of metacommunity assembly forces is on the balance between neutral and niche forces (Brown et al. 2011, Logue et al. 2011), which determine all possible archetypes through interaction between main community high-level processes (i.e., selection, dispersal, drift, speciation; Vellend 2016).

Communities in temporary ponds have a strong niche influence and are as a consequence, mostly related to a species sorting archetype (*Chapter II*, Chase 2007, Florencio et al. 2014, Gascón et al. 2016, Hill et al. 2017a, b). However, there is also a strong dispersal-driven component that determines community succession and thus temporary pond assembly (Boix et al. 2004, 2016, Ruhí et al. 2009, 2013a), which is determined by network structure and species dispersal abilities (*Chapter IV, Chapter III*, Grönroos et al. 2013, Borthagaray et al. 2015a, b). These two main driving forces, which determine more niche assembled or more neutrally assembled metacommunities respectively, are continuously acting throughout time (i.e., hydroperiod) and thus gaining or losing importance along succession concomitantly with habitat changes, more related to hydroregime metrics (i.e., pond water-level; *Chapter II*). Within this context, the *two layers* defined in the introduction, which we assumed that will *define post-disturbance community outcome* and that were linked to wildfire consequences and to landscape structure (i.e., pondscape and dispersal-driven dynamics) did influence studied ponds community outcome. However, temporary ponds successional changes (i.e., hydroperiod environmental changes; *Chapter II*) strictly determined these fluctuations and thus, the whole metacommunity assembly process. Therefore, considering the wildfire impacts (*Chapter I*), the relevance of successional changes and their fluctuations on metacommunity assembly (*Chapter II*) and post-disturbance metacommunity rescue effect and colonization trends (*Chapter III and Chapter IV*) we can update the current knowledge of expected wildfire disturbance effects in Mediterranean temporary ponds. In this sense, Figure 8.3 summarises this framework where wildfire intensity —layer 1— is the main determinant of metacommunity resilience and recovery (Figure 8.3 red arrows) and where the metacommunity rescue effect is counterbalancing its consequences —layer 2. However, there is a tipping point as wildfire intensity increases where the resilience is completely compromised (Figure 8.3 blue square shrinking), affecting the whole metacommunity recovery (Figure 8.3 bottom right part). Thus, the increase in wildfire intensity is related to the decrease in the metacommunity rescue effect, with a marked fall. Furthermore, all species metacommunity perceptions (Figure 8.3 individual black or red arrows) are constantly determined by successional changes linked to hydroregime fluctuation (Figure 8.3 background green arrows) regardless of both disturbance and landscape layers, that are strongly determining metacommunity assembly too.

Figure 8.3: *Wildfire effects on the two layers defining post-disturbance community outcome (wildfire and landscape) along a wildfire intensity gradient (red arrow). The whole outcome for each species landscape perception (the four black/red arrows of each scenario) is globally affected by hydroregime fluctuation and successional changes (green background arrows). The Empordà wildfire (top left graph) was of not great intensity (understood as number of burned water bodies within the area) and only compromised species having a smaller regional perspective (red arrow) not heavily compromising the whole metacommunity. Simulations of greater wildfire intensities (mid central and right bottom graphs) might represent a greater compromise on metacommunity outcome for species with higher landscape perceptions (red arrows). The metacommunity rescue effect is maintained until a tipping point where it loses its strength and therefore, at some intensity level, post-disturbance community would become truly compromised (blue bar). Images represent the four main dispersal groups: passive dispersers with aquatic adults (Gasteropoda), weak active dispersers with flying adults (Diptera), moderate active dispersers with flying adults (Ephemeroptera) and strong active dispersers with flying adults (Odonata).*


Empordà wildfire
Pre-disturbance community
Layer 1: Wildfire
Successional changes
Layer 2: Landscape
Post-disturbance community
Wildfire intensity
Pre-disturbance community
Layer 1: Wildfire
Successional changes
Layer 2: Landscape
Post-disturbance community
Metacommunity rescue effect
Pre-disturbance community
Layer 1: Wildfire
Successional changes
Layer 2: Landscape
Post-disturbance community

8.4 Final considerations

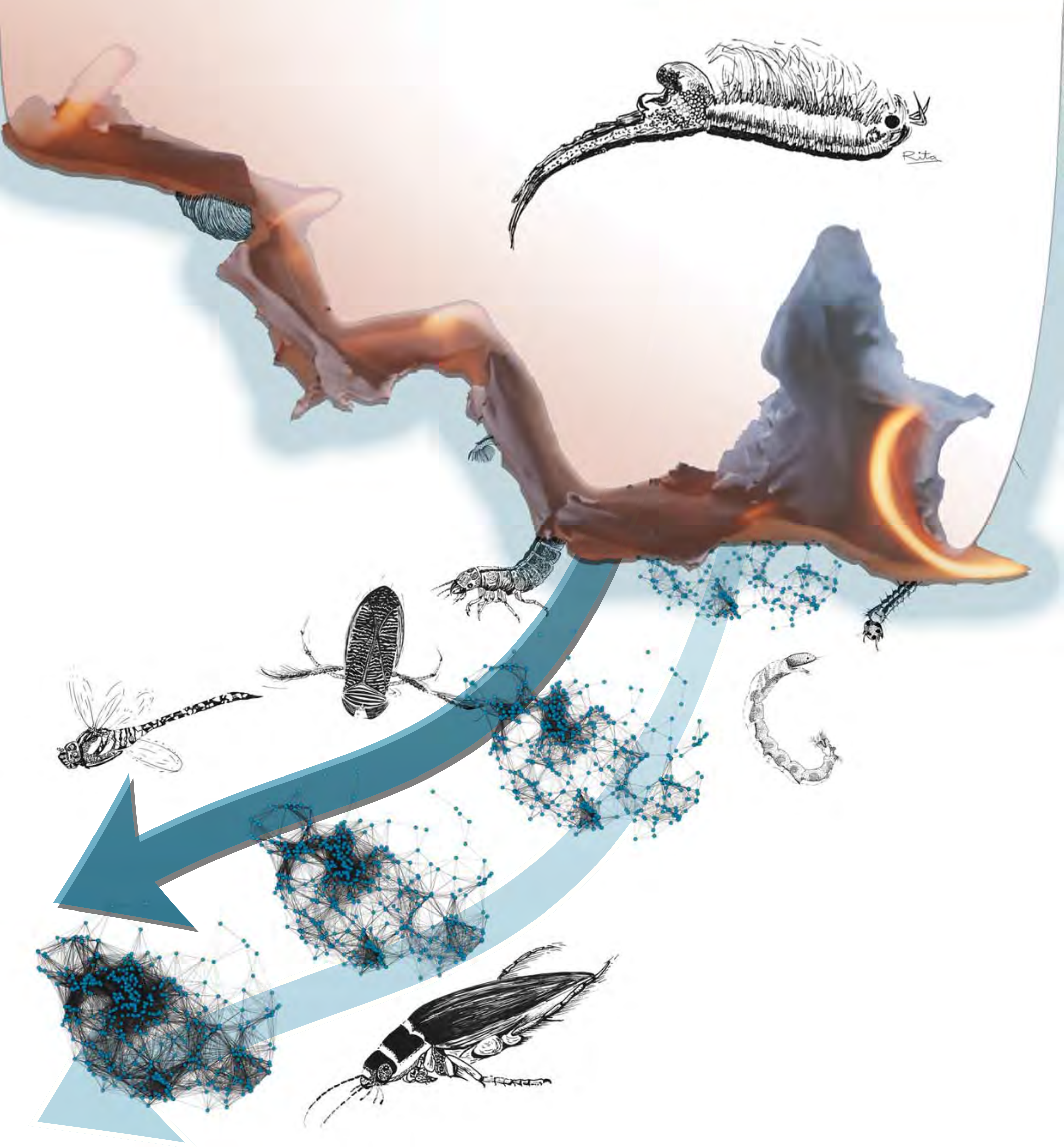
There is still a need for further studies integrating disturbances in a metacommunity perspective as well as an integrative framework of both theories (Pulsford et al. 2016, Leibold and Chase 2018). To achieve this, the usage of all available approaches as well as methodologies is required in order to advance in metacommunity ecology comprehension. Theoretical studies, based for example on simulation models (*Chapter IV*), field-based experiments, using for example mesocosms (*Chapter III*) and more common observational studies (*Chapter I* and *Chapter II*) are all needed in order to advance and better know how all layers define post-disturbance community outcome. Therefore, to jointly considerate all these different perspectives will surely provide a truly deep understanding of ecological processes and the interaction between disturbance and metacommunity dynamics, which far from define a specific common answer, seem to draw a more nuanced and fluctuating pattern along niche and neutral assembly forces balance (*Chapter II;* Leibold and Chase 2018).

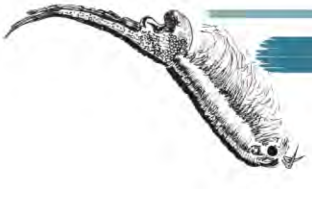
Along this thesis, we have conducted several surveys in particularly unique, endangered and protected habitats (i.e., Mediterranean temporary ponds and high-altitude temporary ponds). Thus, besides the main topic of this thesis, the fact of collecting, identifying and listing the macroinvertebrate species inhabiting such ecosystems is, by itself, a contribution to knowledge and of course to empowerment of these threatened biodiversity reservoirs (Supplementary S8). Ponds have been historically considered a neglected habitat and have been consequently erased from landscapes (Wood et al. 2003, Downing et al. 2006, Céréghino et al. 2008, Downing 2010) including temporary (Brown 1998, Calhoun et al. 2017). This historical tendency has left few pristine areas where ponds and more importantly, the whole pondscape remain unspoiled, because for these systems, as seen along this thesis, network interaction through dispersal is crucial for whole biodiversity maintenance and metacommunity resilience (Gómez-

Rodríguez et al. 2009, Horváth et al. 2019). Recently, awareness was raised against farming activities near some Albera Mediterranean temporary ponds (Vilà 2018), which emphasized the need to scientifically highlight the huge biodiversity that these habitats harbour (Vilà 2018, IAEDEN association 2019). In this thesis, in only 10 ponds surveyed in different occasions, nearly 207 different taxa including macroinvertebrates and amphibians were identified. Individually, ponds presented around 100 different taxa being Estany Gran dels Torlits (TORG) pond with 111 taxa the one having the maximum number of taxa and Estany de la Cardonera de la Gutina (GUTC) and Estany de la Rajoleria de la Gutina (GUTR) with 83 and 84 taxa, respectively, the ones having less number of taxa. Similarly, in Guils de Cerdanya up to 80 taxa where found in the only two ponds surveyed in three occasions along the hydroperiod and their list might constitute, to our knowledge, one of the few studies of Pyrenean temporary ponds that include all the macroinvertebrate fauna assemblage (Specific taxonomic list in supplementary S8).

The presence and survival of Mediterranean temporary ponds taxa from the Albera region (*Chapter I and Chapter II*) as well as high altitude temporary ponds from the Pyrenees in Guils de Cerdanya (*Chapter III*) stands on their historical contingency but also on their regional structure, which have withheld their uniqueness throughout all kind of disturbances, going from catastrophic wildfires to human presence (*Chapter IV*). Its maintenance and conservation reside in keeping their intrinsic characteristics and their surrounding landscape, which must be considered in order to move towards truly and coherent biodiversity conservation policies to face the pictured incendiary climatic future.

9. GENERAL CONCLUSIONS





- Our results indicated that temporary ponds were affected both directly and indirectly by the wildfire, providing a new perspective on wildfire impacts on aquatic systems (i.e., direct burning of sediment organisms) and highlighting species dispersal ability relevance in post-disturbance recovery (*Chapter I*).

    - Indirect impacts were concentrated at the beginning of the post-wildfire hydroperiod (first sampling surveys). They followed a bottom-up process (i.e., fire pulse) that implied an alteration in nutrient content in affected ponds, an increase in phytoplankton chlorophyll-a concentration and an increase in shredders abundance.

    - Direct impacts were sustained over the whole post-wildfire hydroperiod and implied the decrease in abundance of organisms aestivating in ponds sediment and having a passive dispersal ability, but they were not eliminated from burned ponds.

- During the post-wildfire hydroperiod, burned ponds recovered their pre disturbance conditions. (*Chapter I*).

- Throughout the post-wildfire hydroperiod, ponds faunal community composition of all typologies (highly burned, lowly burned and unburned) became more similar.

    - At the end of the hydroperiod environmental characteristics such as macrophytes biomass and nutrients were not different among pond typologies.

- All wildfire indirect and direct impacts, as well as successional changes linked to hydroregime fluctuation (i.e., pond level), affected pond metacommunity assembly forces, which fluctuated along a neutral niche gradient. However, this fluctuation was greatly influenced by intrinsic ponds successional changes and regardless of wildfire (*Chapter II*).

- Although wildfire changed selection forces of some dispersal abilities as well as on body size and some functional feeding groups, they were not as important as other environmental changes, more related to hydroregime fluctuations that greatly influenced trait selection (*Chapter II*).

- Network location, understood as being in a central or in an isolated location within the pond network, influences dispersal dynamics following a mass effects archetype. Thus, central locations present greater abundance and accumulated richness values. This influence is generalized among dispersal groups (i.e., weak and strong active dispersers) independently of their different colonization patterns (*Chapter III*).

    - Mesocosms community composition behaves similarly having central and isolated locations different community compositions after pond complete drying but not being different regarding distance or direction.

- Smaller regional perspectives (distance and direction) and the specific network structure represent additional modulators of dispersal dynamics and thus of main metacommunity dynamic (*Chapter III*).

    - Direction towards other suitable habitats (i.e., ponds) enhance organism fluxes and consequently it increases organism abundances regardless of location and generating a "central-like" pattern in isolated locations.

- Dispersal ability —species landscape perception— appeared as a key driver of colonization patterns (*Chapter III*) but also of resilience and recovery, determining post-disturbance dynamics with a strong link with pond network structure (*Chapter IV*).

- The simulation based on the original wildfire of la Jonquera corroborated the intrinsic resilience of temporary ponds (i.e., the Albera network; *Chapter IV*). Moreover, it also corroborated the relevance of landscape structure (i.e., dispersal abilities or regional perspectives; *Chapter IV*) in the reported post-disturbance recovery (*Chapter I & Chapter II*).

- Metacommunity simulated resilience responds to the intensity and area of the wildfire, being intensity —the number of ponds burned within an area— the more relevant (*Chapter IV*).

    - Simulated responses of resilience to different wildfire scenarios showed a strong non-linear pattern linked mostly to intensity. There is a tipping point where metacommunity resilience becomes compromised and rapidly falls from an almost all species recovery situation to practically any species recovery after disturbance.

    - Intensity (i.e., the number of ponds burned within an area) is key as it represents the maintenance of colonization sources (*Chapter III*) within the burned area. Something that determines post-disturbance recolonization.

- The predicted future increase in wildfire intensity and area in the Mediterranean region and globally, coupled with habitat fragmentation due to temporary ponds loss can deeply affect the current reported metacommunity resilience, compromising the whole metacommunity recovery after a disturbance such as wildfires.

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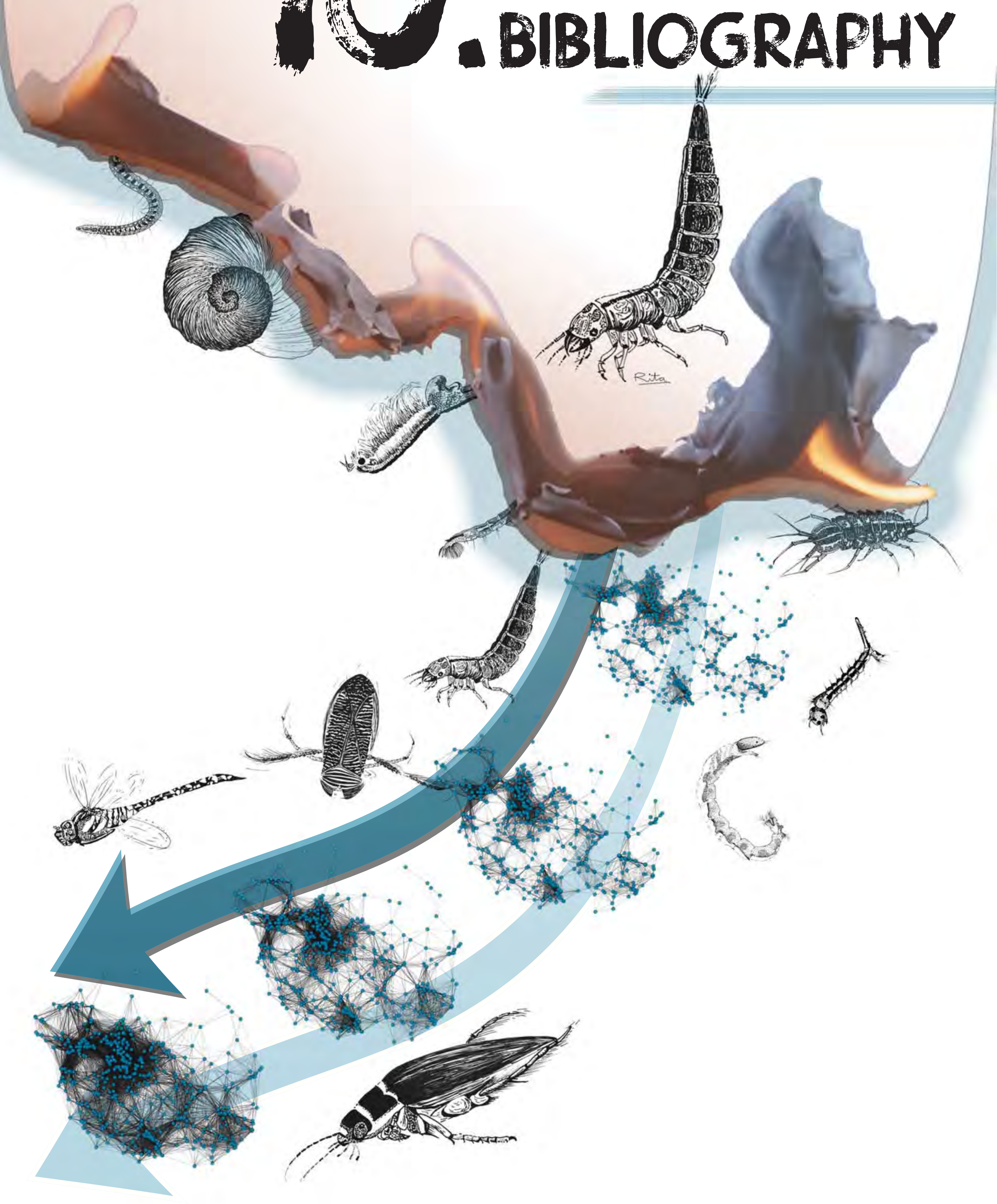

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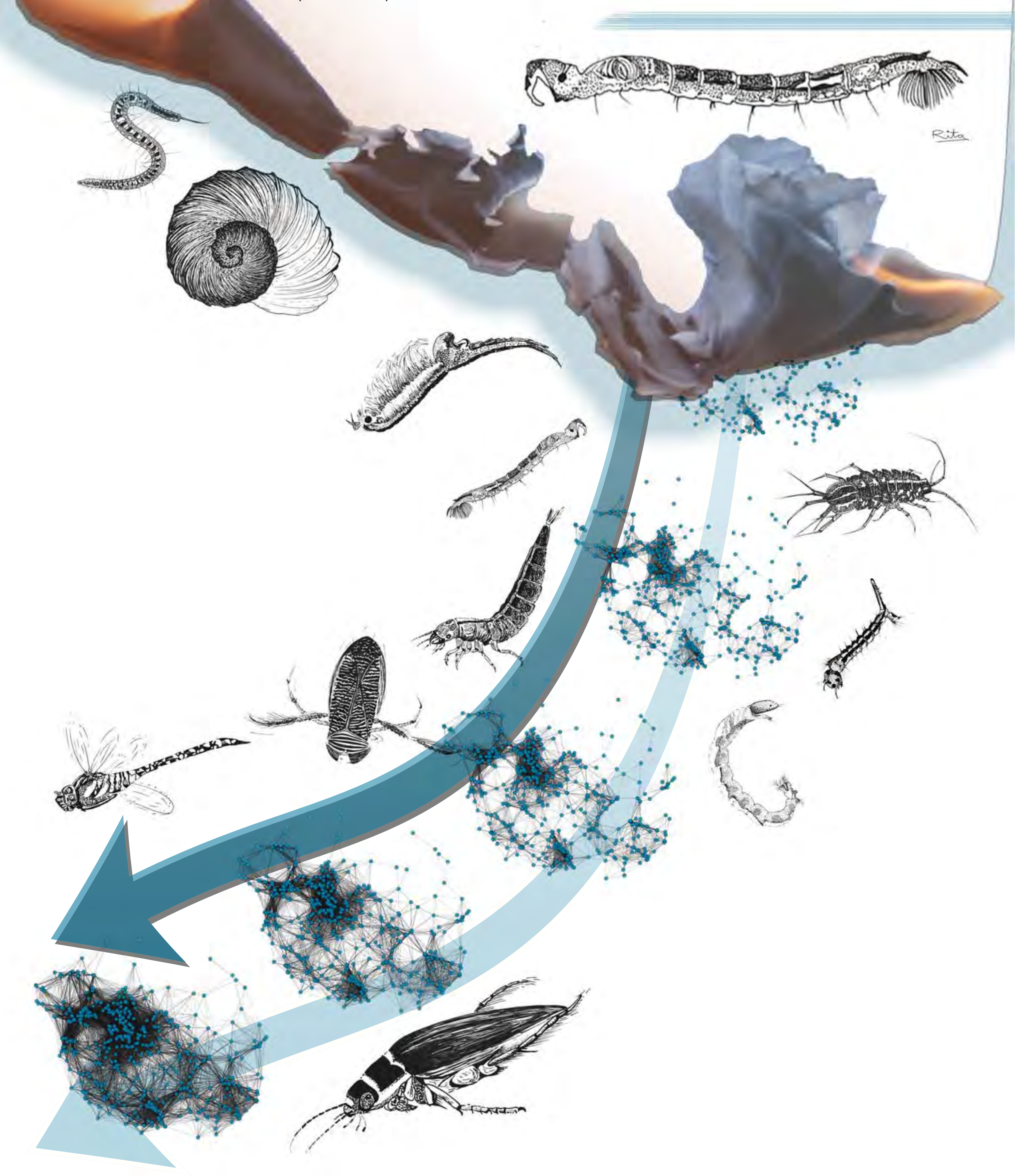
11. SUPPLEMENTARY MATERIALS



SUPPLEMENTARY MATERIAL

S3

PHOTOGRAPHIC REGISTER

Supplementary S3.1: Photographic images from some of the Albera region sampled ponds (CANB, TORC, GUTR TORP ponds) from several years before the wildfire, right after the Empordà wildfire (when they were still dry) and from the surveys after the wildfire of December 2012, February 2013, April 2013 and June 2013.

Some images of ponds several years before the wildfire

CANB 2011 —Estany de Baix de Canadal—

GUTR 2009 —Estany de la Rajoleria de la Gutina—

TORC 2009 —Estany de la Cardonera dels Torlits—

Summer 2012 – right after the wildfire

Highly burned ponds

CANB summer 2012 —Estany de Baix de Canadal—

TORC summer 2012 —Estany de la Cardonera dels Torlits—

SERR summer 2012 —Estany de Serrallobera—

Lowly burned ponds

CANP summer 2012 —Estany Petit de Canadal—

MOLES summer 2012 —Estany de les Moles—

December 2012 survey campaign — first survey afer the wildfire —

CANB December 2012 —Estany de Baix de Canadal—

TORC December 2012 —Estany de la Cardonera dels Torlits—

GUTR December 2012 —Estany de la Rajoleria de la Gutina—

TORP December 2012 —Estany Petit dels Torlits—

February 2013 survey campaign — second survey afer the wildfire —

CANB february 2013 —Estany de Baix de Canadal—

TORC February 2013 —Estany de la Cardonera dels Torlits—

GUTR February 2013 —Estany de la Rajoleria de la Gutina—

TORP February 2013 —Estany Petit dels Torlits—

April 2013 survey campaign — third survey afer the wildfire —

CANB April 2013 —Estany de Baix de Canadal—

TORC April 2013 —Estany de la Cardonera dels Torlits—

GUTR April 2013 —Estany de la Rajoleria de la Gutina—

TORP April 2013 —Estany Petit dels Torlits—

June 2013 survey campaign — fourth survey afer the wildfire —

CANB June 2013 —*Estany de Baix de Canadal*—

TORC June 2013 —*Estany de la Cardonera dels Torlits*—

GUTR June 2013 —*Estany de la Rajoleria de la Gutina*—

TORP June 2013 —*Estany Petit dels Torlits*—

Supplementary S3.2: Photographic images from the two surveyed Guils de Cerdanya ponds (central pond and isolated pond) from the three pond surveys realized there in the corresponding chapter. Surveys correspond to hydroperiod beginning (May 18th2016), hydroperiod middle (April 19th2016) and hydroperiod end (July 19th2016).

May 2016 — hydroperiod beginning —

Isolated pond

Central pond

June 2016 — hydroperiod middle —

Isolated pond

Central pond

July 2016 — final hydroperiod —

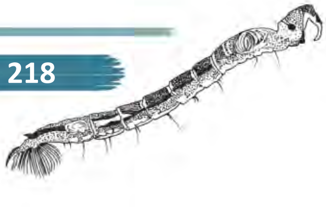
Isolated pond

Central pond

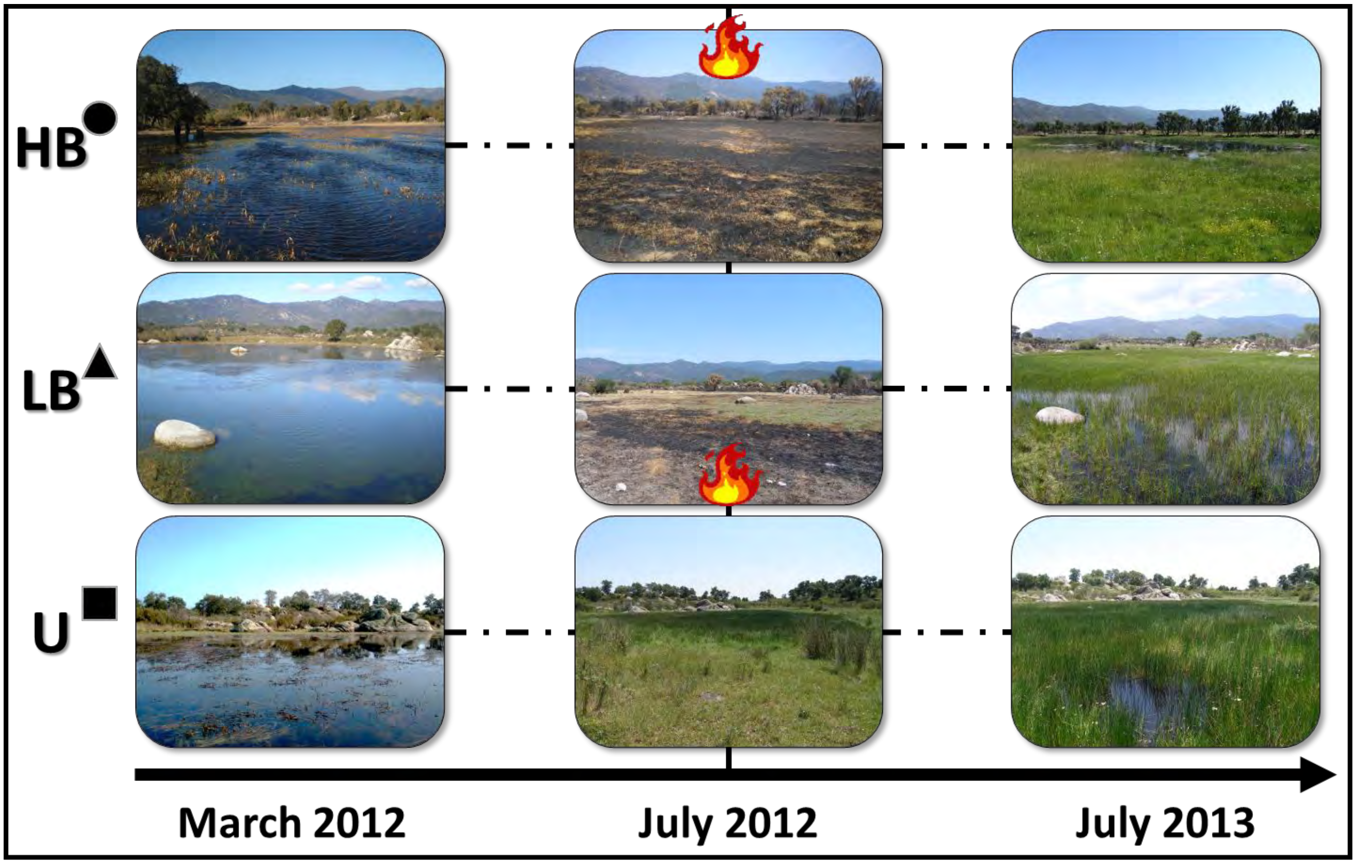
SUPPLEMENTARY MATERIAL

S4

CHAPTER 1 SUPPLEMENTARY MATERIAL



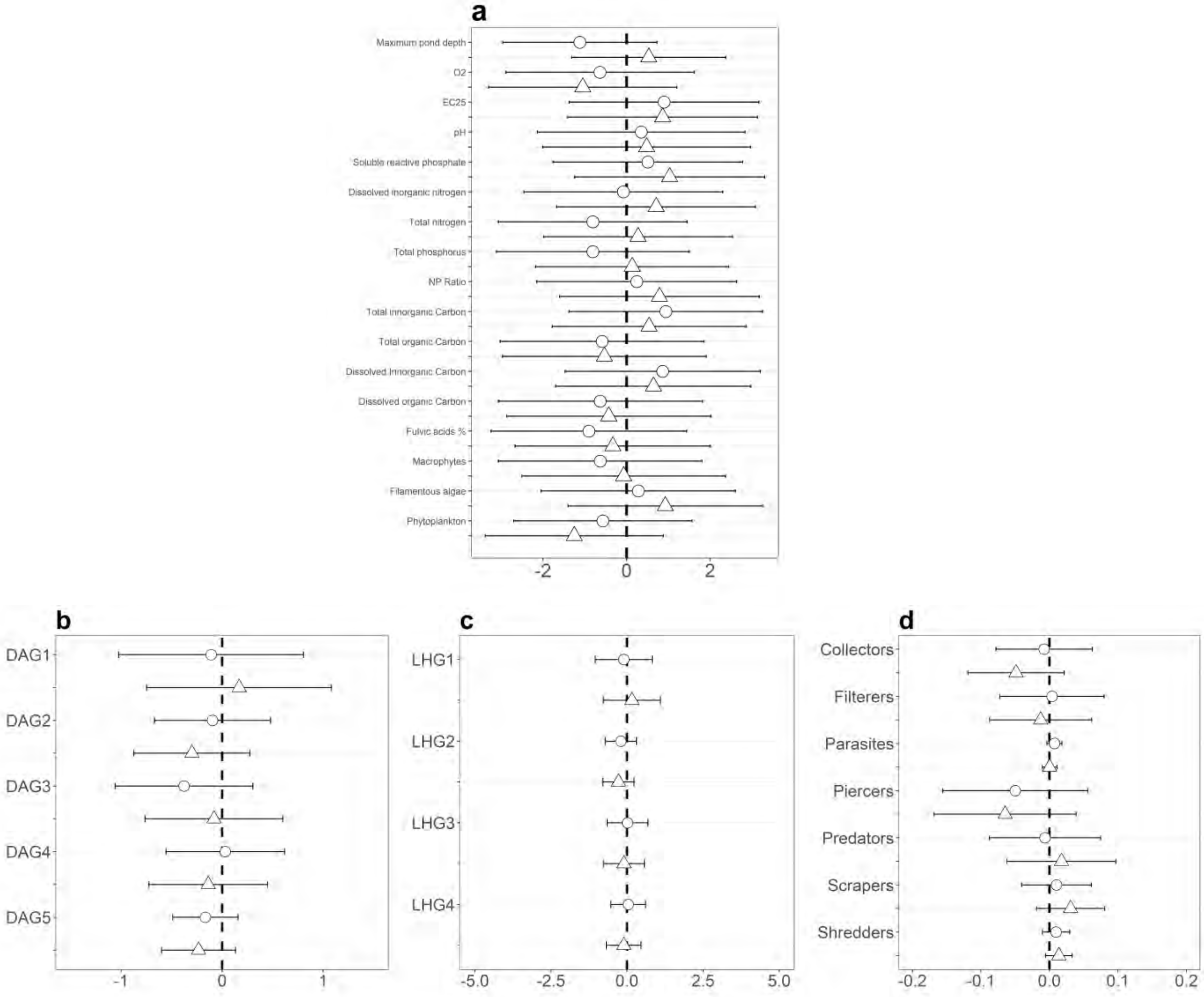
Supplementary S4.1: *Picture panel showing pond typologies according to their burn status (unburned: U, high: HB, and low LB) along time. March 2012 corresponds to ponds before the wildfire, July 2012 corresponds to dried ponds appearance just after the wildfire, and July 2013 to the last sampling surveys of post-fire year.*



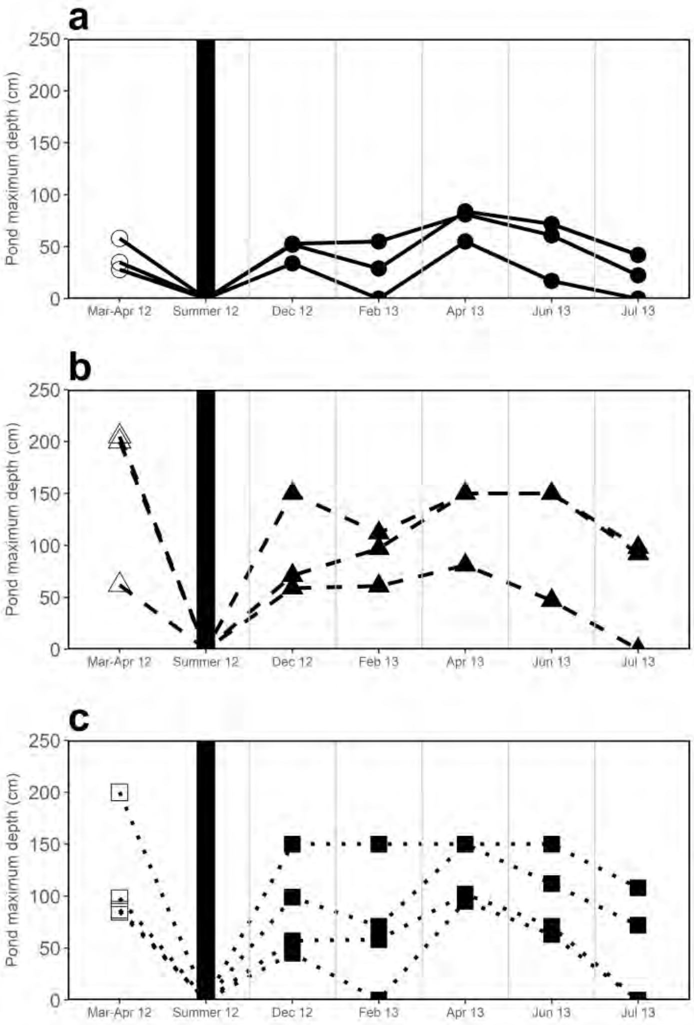
Supplementary S4.2: *ANOVA results for environmental variables and functional traits (dispersal ability groups (DAG), life-history groups (LHG) and functional feeding groups (FFG)) before the wildfire. For variable names and definitions see material and methods section. Significance levels are p<0.05 in gray shading and bold.*

Environmental variables	$F_{2,7}$	p-value	Functional traits	$F_{2,7}$	p-value
			DAG		
Ful.Ac.%	0.65	0.55	DAG1	0.35	0.71
EC25	0.91	0.44	DAG2	1.19	0.36
DIC	0.66	0.55	DAG3	1.40	0.31
DIN	0.31	0.74	DAG4	0.36	0.71
O2	0.98	0.42	DAG5	1.03	0.41
DOC	0.27	0.77	**LHG**		
Fil	0.70	0.53	LHG1	0.31	0.74
Mac	0.05	0.95	LHG2	1.44	0.30
Max Z	**5.77**	**0.03**	LHG3	0.14	0.87
NP Ratio	0.24	0.79	LHG4	0.28	0.77
pH	0.18	0.84	**FFG**		
Phy	2.18	0.18	Collectors	2.27	0.17
SRP	0.94	0.44	Filterers	0.15	0.86
TIC	0.74	0.51	Scrapers	1.16	0.32
TN	1.00	0.41	Shredders	1.58	0.22
TP	0.93	0.44	Piercers	1.29	0.28
			Predators	0.26	0.77

Supplementary S4.3: *Mean values and the confidence intervals (95%) of the differences between the high intensity ponds and unburned ponds (circles) and between the low intensity ponds and unburned ponds (triangles) before the wildfire. Environmental variables are in (a), dispersal ability groups (DAG) are in (b), life history groups (LHG) are in (c), and functional feeding groups are in (d). For variable names and definitions see material and methods section. The dashed central line represents the reference conditions (unburned ponds). No significant differences found.*



Supplementary S4.4: *Pond maximum depth among high intensity ponds (a), low intensity ponds (b) and unburned ponds (c). White symbols correspond to before the wildfire surveys and black symbols to the after the wildfire surveys. Black vertical bar represents the summer drought where the wildfire affected the systems and the separation between before and after periods.*



Supplementary S4.5: *Summary table of the environmental variables values obtained in the study. The average and standard deviation are shown for each environmental variable calculated in this study. For variable names and definitions see material and methods section. Values are separated by burned status (unburned: U, high: HB, and low LB) and by sample survey including the survey before the wildfire.*

	U	Before		December 12		February 13		April 13		June 13		July 13	
Environmental variables	Max Z	117.5	±47.89	87.75	±41.15	93.00	±40.65	124.2	±25.87	99.00	±34.82	90.00	±18.00
	O2	8.94	±0.85	9.58	±0.49	10.04	±1.35	10.78	±1.91	8.44	±3.68	9.27	±3.38
	EC25	159.3	±42.26	224.43	±100.72	331.67	±45.90	197.8	±74.45	226.1	±83.46	412.00	±86.00
	pH	7.30	±0.10	6.98	±0.37	8.43	±0.47	8.00	±1.13	6.97	±0.30	7.50	±0.11
	SRP	0.00	±0.00	0.00	±0.00	0.00	±0.00	0.00	±0.00	0.01	±0.02	0.01	±0.00
	DIN	0.04	±0.02	0.02	±0.02	0.02	±0.00	0.02	±0.01	0.01	±0.00	0.03	±0.01
	TN	1.85	±0.06	1.52	±0.12	1.72	±0.30	1.39	±0.20	2.96	±1.31	2.49	±0.03
	TP	0.06	±0.02	0.06	±0.04	0.05	±0.02	0.05	±0.02	0.32	±0.30	0.15	±0.07
	NP Ratio	1.26	±0.34	1.38	±1.86	0.90	±0.36	0.82	±0.15	0.11	±0.10	0.61	±0.18
	TIC	7.58	±2.86	9.99	±8.04	12.29	±7.59	8.66	±4.28	14.48	±5.50	21.35	±5.70
	TOC	27.75	±3.89	24.32	±3.23	27.38	±5.70	22.45	±4.01	35.54	±12.99	36.20	±4.26
	DIC	7.07	±2.39	10.20	±8.27	10.89	±7.10	9.00	±4.08	13.53	±4.97	24.07	±8.04
	DOC	27.27	±4.42	24.12	±2.40	27.69	±6.44	21.53	±3.64	32.31	±10.36	34.31	±2.88
	Ac.Ful.%	40.36	±5.88	24.93	±13.79	16.73	±4.73	21.61	±7.44	34.66	±6.67	37.15	±8.12
	Phy	16.55	±12.87	4.20	±2.65	4.25	±4.08	6.79	±3.81	77.84	±51.82	128.39	±101.63
	Fil	0.001	±0.001	0.0001	±0.0001	0.0008	±0.0007	0.0009	±0.0009	0.002	±0.003	0.00	±0.00
	Mac	0.04	±0.03	0.13	±0.07	0.09	±0.08	0.08	±0.03	0.10	±0.01	0.22	±0.03

	HB	Before		December 12		February 13		April 13		June 13		July 13	
Environmental variables	Max Z	40.33	±12.81	46.33	±8.73	42.00	±13.00	73.33	±13.02	50.00	±23.76	32.25	±9.75
	O2	7.71	±1.58	10.86	±3.16	8.47	±0.80	10.17	±1.28	8.02	±1.41	4.55	±0.18
	EC25	329.9	±280.9	320.20	±185.83	347.85	±160.15	266.9	±168.3	280.3	±182.5	753.50	±529.50
	pH	7.42	±0.43	7.84	±0.91	9.04	±1.13	7.83	±0.10	7.46	±0.48	7.08	±0.09
	SRP	0.01	±0.01	0.01	±0.01	0.03	±0.03	0.01	±0.01	0.02	±0.02	0.05	±0.03
	DIN	0.04	±0.02	0.01	±0.00	0.05	±0.03	0.02	±0.01	0.02	±0.03	0.07	±0.04
	TN	1.49	±0.12	1.88	±0.43	3.04	±0.65	1.36	±0.25	1.82	±0.11	2.66	±0.22
	TP	0.05	±0.01	0.09	±0.05	0.14	±0.07	0.06	±0.03	0.08	±0.03	0.22	±0.15
	NP Ratio	1.53	±0.38	0.46	±0.35	0.64	±0.13	1.18	±0.56	0.70	±0.71	0.93	±0.25
	TIC	22.08	±23.11	17.29	±9.27	15.22	±0.36	16.96	±11.57	18.62	±10.85	57.68	±40.17
	TOC	24.84	±2.96	21.84	±4.09	33.26	±4.44	17.01	±2.77	20.19	±2.59	32.50	±3.40
	DIC	17.10	±16.60	19.01	±9.39	14.67	±0.27	16.74	±10.95	21.28	±14.07	59.52	±42.52
	DOC	24.06	±2.45	21.47	±4.50	32.59	±2.65	16.43	±2.34	20.32	±2.30	31.40	±2.99
	Ac.Ful.%	35.47	±2.56	45.23	±23.26	45.84	±20.9	40.33	±21.81	39.76	±3.95	52.50	±11.31
	Phy	10.54	±3.63	22.41	±26.87	31.64	±17.40	2.84	±1.28	10.19	±5.22	5.53	±1.31
	Fil	0.002	±0.0004	0.0005	±0.0002	0.005	±0.004	0.0003	±0.0003	0.00	±0.00	0.00	±0.00
	Mac	0.07	±0.05	0.04	±0.02	0.03	±0.04	0.02	±0.00	0.15	±0.05	0.16	±0.07

	LB	Before		December 12		February 13		April 13		June 13		July 13	
Environmental variables	Max Z	154	±65.05	93.33	±40.37	90.00	±21.40	127.0	±32.53	115.6	±48.55	95.00	±3.00
	O2	6.92	±2.29	12.94	±2.40	11.28	±0.12	10.69	±0.65	5.55	±0.74	3.81	±0.94
	EC25	323.6	±69.06	478.67	±299.98	498.67	±271.2	364.6	±155	364.1	±158.3	516.00	±191.00
	pH	7.46	±0.34	7.42	±0.35	8.06	±0.11	7.87	±0.63	7.15	±0.09	7.00	±0.15
	SRP	0.01	±0.00	0.02	±0.02	0.01	±0.00	0.01	±0.00	0.06	±0.03	0.08	±0.06
	DIN	0.07	±0.06	0.50	±0.68	0.05	±0.03	0.02	±0.01	0.07	±0.10	0.02	±0.01
	TN	1.98	±0.67	2.11	±0.64	1.91	±0.93	1.42	±0.36	1.87	±0.43	2.04	±0.48
	TP	0.07	±0.01	0.07	±0.05	0.05	±0.01	0.05	±0.02	0.13	±0.06	0.14	±0.08
	NP Ratio	2.15	±1.75	78.12	±110.07	2.19	±1.27	1.03	±0.85	0.81	±0.93	0.28	±0.04
	TIC	15.87	±6.55	16.50	6.86	20.21	±10.06	18.69	±6.57	24.51	±9.81	35.18	±13.11
	TOC	25.10	±6.25	23.30	±13.88	26.41	±13.21	21.12	±6.05	25.16	±3.61	28.09	±5.28
	DIC	14.53	±7.52	18.14	±8.49	19.87	±8.89	20.03	±8.11	24.56	±10.28	34.54	±11.81
	DOC	25.10	±6.25	23.12	±13.80	26.00	±12.93	19.94	±5.83	25.29	±3.56	28.52	±5.28
	Ac.Ful.%	38.57	±4.63	29.18	±6.29	27.09	±9.91	27.80	±3.29	48.18	±6.14	50.39	±3.47
	Phy	3.27	±1.20	3.12	±2.51	9.64	±5.89	4.19	±1.21	1.81	±1.08	2.21	±1.47
	Fil	0.004	±0.003	0.0009	±0.0008	0.002	±0.001	0.0008	±0.0008	0.0004	±0.0003	0.0009	±0.001
	Mac	0.09	±0.04	0.08	±0.04	0.11	±0.03	0.04	±0.00	0.21	±0.09	0.26	±0.08

Supplementary S4.6: *Summary table of values (Nº individuals per liter) of the functional trait groups obtained in the study. The average and standard deviation are shown for each functional trait calculated in this study, dispersal ability groups (DAG), life-history groups (LHG) and functional feeding groups. For variable names and definitions see material and methods section. Values are separated by pond burned status (unburned: U, high: HB, and low LB) and by sample survey including the survey before the wildfire.*

	U	Before		December 12		February 13		April 13		June 13		July 13	
Faunal traits	DAG1	7.6	±5.1	7.5	±6.9	5.9	±3.0	13.4	±12.5	2.4	±1.8	3.3	±1.6
	DAG2	2.7	±2.3	3.0	±2.8	2.9	±2.7	4.3	±1.7	2.4	±2.5	14.6	±9.3
	DAG3	0.6	±0.8	1.0	±1.6	1.3	±1.9	0.8	±1.1	0.8	±0.4	11.0	±5.1
	DAG4	1.1	±0.8	0.4	±0.3	0.5	±0.2	1.5	±0.2	3.4	±4.4	2.6	±1.1
	DAG5	0.0	±0.0	0.0	±0.0	0.0	±0.0	0.3	±0.3	0.1	±0.0	0.0	±0.0
	LHG1	7.5	±5.2	7.4	±7.0	5.8	±3.1	13.4	±12.6	2.3	±1.5	3.2	±1.5
	LHG2	3.3	±2.1	3.6	±3.6	4.1	±2.5	5.5	±1.2	3.0	±2.5	12.4	±5.5
	LHG3	0.1	±0.1	0.0	±0.0	0.0	±0.0	0.1	±0.0	0.1	±0.1	0.1	±0.1
	LHG4	1.0	±0.7	0.8	±1.0	0.8	±0.4	1.4	±0.2	3.7	±4.8	15.9	±10.8
	Collectors	1.7	±1.2	2.3	±1.3	2.3	±2.3	8.3	±8.7	1.2	±0.8	5.2	±2.2
	Shredders	1.2	±0.4	0.6	±0.4	0.4	±0.3	1.5	±0.5	0.9	±0.7	2.8	±1.1
	Scrappers	1.6	±0.6	1.5	±1.5	3.0	±2.4	2.8	±0.7	2.1	±1.7	6.9	±3.3
	Filterers	0.8	±0.7	1.0	±0.4	0.6	±0.4	3.4	±3.1	0.5	±0.5	0.3	±0.0
	Piercers	0.2	±0.2	0.2	±0.2	0.2	±0.3	0.3	±0.3	2.7	±4.0	1.3	±1.1
	Predators	6.1	±5.2	5.9	±5.6	3.6	±3.3	3.0	±1.5	1.4	±0.8	14.8	±9.2
	Parasites	0.3	±0.2	0.4	±0.3	0.5	±0.4	0.7	±0.2	0.4	±0.4	0.2	±0.1

Faunal traits

HB	Before	December 12	February 13	April 13	June 13	July 13
DAG1	5.9 ±4.8	2.1 ±0.7	1.4 ±0.7	3.3 ±3.1	4.2 ±4.8	1.5 ±0.5
DAG2	1.6 ±0.7	1.3 ±0.5	2.3 ±0.7	5.2 ±3.1	3.8 ±4.4	2.2 ±0.8
DAG3	0.0 ±0.0	0.1 ±0.2	0.1 ±0.0	0.2 ±0.1	1.0 ±0.4	4.8 ±1.9
DAG4	1.5 ±1.5	0.2 ±0.1	0.2 ±0.1	0.8 ±0.2	1.8 ±1.1	2.1 ±0.3
DAG5	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.3 ±0.0	0.0 ±0.0	0.0 ±0.0
LHG1	5.9 ±4.8	2.0 ±0.8	1.4 ±0.7	3.3 ±3.1	4.0 ±4.9	1.1 ±0.3
LHG2	1.7 ±0.9	1.6 ±0.6	2.4 ±0.8	5.5 ±3.1	4.3 ±3.7	5.6 ±2.1
LHG3	0.1 ±0.1	0.0 ±0.0	0.1 ±0.1	0.1 ±0.0	0.3 ±0.2	0.1 ±0.1
LHG4	1.5 ±1.4	0.2 ±0.1	0.1 ±0.0	1.0 ±0.4	2.1 ±1.2	3.6 ±1.2
Collectors	1.5 ±1.4	0.6 ±0.3	0.6 ±0.0	1.3 ±0.6	1.2 ±0.5	2.3 ±0.8
Shredders	1.1 ±1.0	0.4 ±0.3	0.3 ±0.1	0.6 ±0.1	1.4 ±0.9	1.1 ±0.4
Scrappers	1.5 ±1.0	0.4 ±0.1	0.9 ±0.5	2.5 ±1.5	4.4 ±2.9	2.8 ±1.0
Filterers	0.7 ±0.5	0.4 ±0.0	0.4 ±0.1	1.0 ±0.1	0.6 ±0.6	0.2 ±0.0
Piercers	0.0 ±0.0	0.0 ±0.0	0.0 ±0.0	0.1 ±0.0	1.0 ±0.8	1.4 ±0.5
Predators	4.1 ±3.0	1.7 ±0.7	1.4 ±0.6	3.3 ±0.3	1.5 ±1.3	2.3 ±0.8
Parasites	0.2 ±0.1	0.2 ±0.1	0.3 ±0.1	0.8 ±0.0	0.6 ±0.7	0.2 ±0.1

	LB	Before		December 12		February 13		April 13		June 13		July 13	
Faunal traits	DAG1	10.3	±6.9	5.1	±1.3	2.7	±0.5	12.4	±5.4	10.8	±8.8	17.0	±10.7
	DAG2	1.0	±1.0	0.9	±0.4	0.5	±0.2	8.0	±4.3	2.8	±3.4	8.3	±0.8
	DAG3	0.1	±0.1	0.1	±0.1	0.0	±0.0	0.3	±0.1	0.5	±0.3	3.8	±0.3
	DAG4	0.8	±0.8	0.1	±0.1	0.1	±0.1	1.6	±0.2	0.8	±0.5	1.3	±0.5
	DAG5	0.0	±0.0	0.0	±0.0	0.0	±0.0	0.2	±0.1	0.1	±0.0	0.0	±0.0
	LHG1	10.1	±7.1	5.0	±1.0	2.6	±0.4	11.6	±5.5	10.8	±8.7	17.0	±10.7
	LHG2	1.4	±1.0	1.2	±0.6	0.6	±0.2	9.0	±3.4	2.9	±3.0	6.0	±1.5
	LHG3	0.0	±0.0	0.0	±0.0	0.0	±0.0	0.1	±0.1	0.1	±0.0	0.1	±0.1
	LHG4	0.7	±0.6	0.1	±0.1	0.1	±0.0	1.8	±0.4	1.2	±0.7	7.4	±1.6
	Collectors	0.4	±0.2	0.8	±0.9	0.2	±0.0	3.2	±2.0	1.0	±1.0	2.5	±0.3
	Shredders	2.4	±1.9	0.3	±0.1	0.1	±0.0	2.7	±0.9	3.1	±2.4	5.8	±2.7
	Scrappers	6.0	±5.0	0.7	±0.1	0.3	±0.1	6.8	±2.3	8.4	±6.3	14.0	±7.3
	Filterers	0.6	±0.5	0.5	±0.4	0.5	±0.3	2.4	±0.7	0.5	±0.7	0.5	±0.2
	Piercers	0.1	±0.0	0.1	±0.1	0.0	±0.0	0.3	±0.2	0.4	±0.2	0.6	±0.4
	Predators	2.5	±1.9	3.7	±0.1	2.0	±0.5	5.3	±3.1	1.2	±1.0	6.9	±1.8
	Parasites	0.3	±0.1	0.2	±0.2	0.1	±0.0	1.6	±0.4	0.4	±0.4	0.3	±0.2

SUPPLEMENTARY MATERIAL

S5

CHAPTER II SUPPLEMENTARY MATERIAL

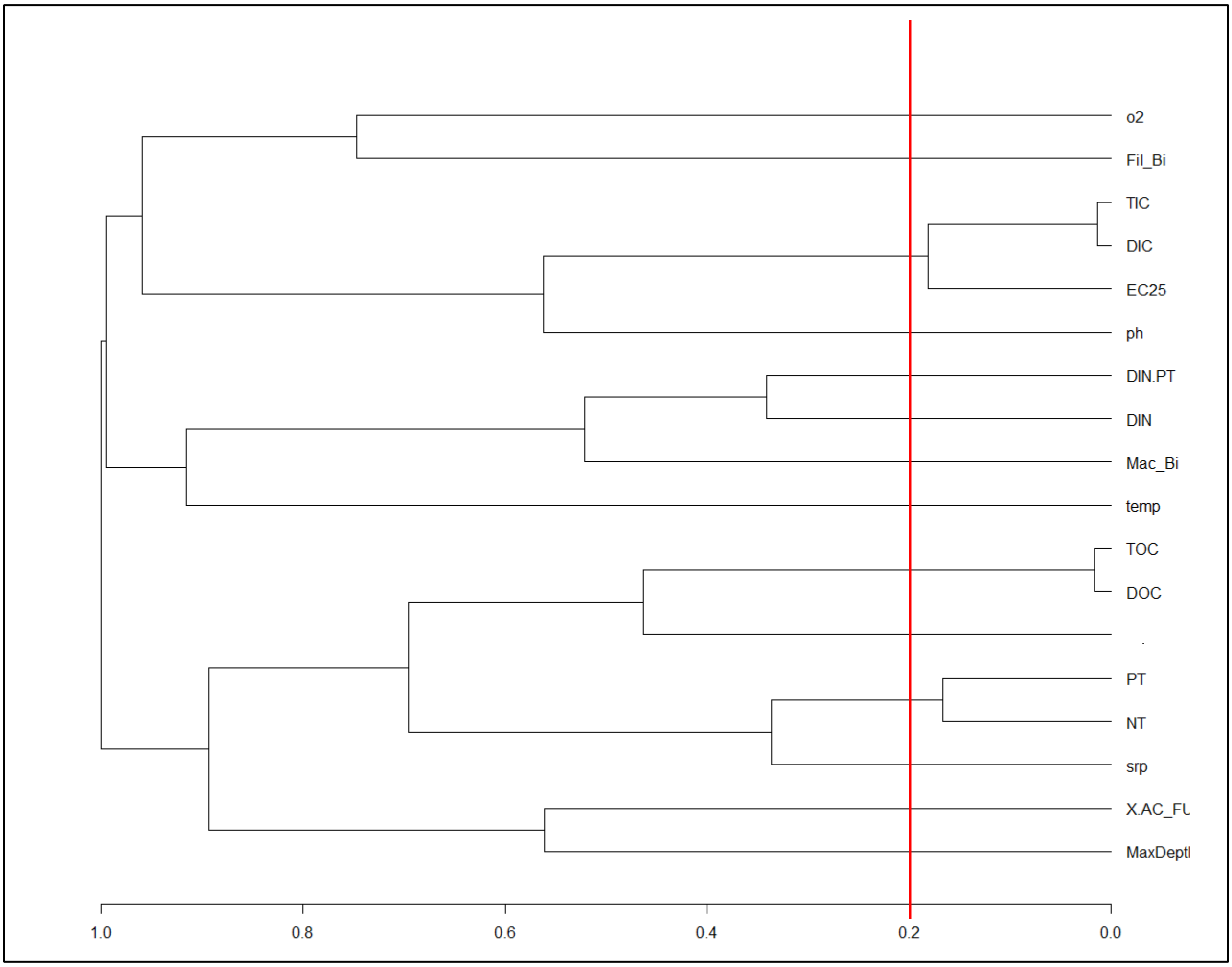
***Supplementary S5.1:** Correlation analysis to select environmental factors and avoid highly correlated variables*

***Supplementary S5.1A:** Environmental variables correlation table, indicating Pearson coefficients between pairs of environmental variables.*

	Phy	*Mac. Bi.*	*Fil. Bi.*	*O^2*	*EC25*	*ph*	*Max. Depth*	*DIN*	*SRP*
Phy	1.00	0.00	-0.01	-0.02	-0.22	-0.24	-0.21	0.15	0.37
Mac. Bi.	0.00	1.00	-0.12	-0.12	0.01	-0.23	-0.12	-0.51	-0.16
Fil. Bi.	-0.01	-0.12	1.00	-0.25	0.23	0.34	0.01	-0.03	-0.10
O^2	-0.02	-0.12	-0.25	1.00	-0.15	0.33	0.10	-0.08	-0.06
EC25	-0.22	0.01	0.23	-0.15	1.00	0.62	0.12	-0.19	-0.13
ph	-0.24	-0.23	0.34	0.33	0.62	1.00	0.16	-0.01	-0.02
Max. Depth	-0.21	-0.12	0.01	0.10	0.12	0.16	1.00	-0.19	-0.25
DIN	0.15	-0.51	-0.03	-0.08	-0.19	-0.01	-0.19	1.00	0.21
SRP	0.37	-0.16	-0.10	-0.06	-0.13	-0.02	-0.25	0.21	1.00
W. Temp.	0.19	0.08	-0.07	-0.05	-0.12	-0.06	0.22	-0.09	0.15
% Ac. Ful.	0.49	-0.06	-0.01	-0.11	-0.36	-0.25	-0.44	0.22	0.43
DOC	0.54	0.23	0.18	-0.01	0.00	-0.11	-0.13	-0.09	0.30
DIN/PT	-0.38	-0.48	-0.05	0.01	0.04	0.11	0.08	0.66	-0.31
NT	0.70	0.02	0.09	0.05	0.02	-0.04	-0.18	0.06	0.66
PT	0.65	0.07	-0.14	0.09	-0.20	-0.24	-0.14	-0.08	0.66
TOC	0.61	0.21	0.13	0.01	-0.04	-0.16	-0.11	-0.09	0.33
DIC	-0.06	-0.10	0.04	-0.15	0.82	0.44	0.05	-0.07	0.18
TIC	-0.02	-0.10	0.07	-0.18	0.84	0.46	0.04	-0.03	0.17

	W. Temp.	% Ac. Ful.	DOC	DIN/PT	NT	PT	TOC	DIC	TIC
Phy	0.19	0.49	0.54	-0.38	0.70	0.65	0.61	-0.06	-0.02
Mac. Bi.	0.08	-0.06	0.23	-0.48	0.02	0.07	0.21	-0.10	-0.10
Fil. Bi.	-0.07	-0.01	0.18	-0.05	0.09	-0.14	0.13	0.04	0.07
O^2	-0.05	-0.11	-0.01	0.01	0.05	0.09	0.01	-0.15	-0.18
EC25	-0.12	-0.36	0.00	0.04	0.02	-0.20	-0.04	0.82	0.84
ph	-0.06	-0.25	-0.11	0.11	-0.04	-0.24	-0.16	0.44	0.46
Max. Depth	0.22	-0.44	-0.13	0.08	-0.18	-0.14	-0.11	0.05	0.04
DIN	-0.09	0.22	-0.09	0.66	0.06	-0.08	-0.09	-0.07	-0.03
SRP	0.15	0.43	0.30	-0.31	0.66	0.66	0.33	0.18	0.17
W. Temp.	1.00	0.05	-0.15	-0.12	0.10	0.26	-0.11	0.13	0.10
% Ac. Ful.	0.05	1.00	0.13	-0.36	0.33	0.30	0.17	-0.07	-0.07
DOC	-0.15	0.13	1.00	-0.37	0.78	0.58	0.98	-0.09	-0.03
DIN/PT	-0.12	-0.36	-0.37	1.00	-0.42	-0.48	-0.40	0.00	0.03
NT	0.10	0.33	0.78	-0.42	1.00	0.83	0.81	0.19	0.23
PT	0.26	0.30	0.58	-0.48	0.83	1.00	0.65	0.02	0.04
TOC	-0.11	0.17	0.98	-0.40	0.81	0.65	1.00	-0.08	-0.01
DIC	0.13	-0.07	-0.09	0.00	0.19	0.02	-0.08	1.00	0.99
TIC	0.10	-0.07	-0.03	0.03	0.23	0.04	-0.01	0.99	1.00

Supplementary S5.1B: *Correlation dendrogram indicating the correlations between environmental variables. We did not consider variables with a Pearson coefficient higher than 0.8.*



The variables considered were pH, Fil Bi, DIN, SRP, EC25, O^2, % AC.FUL., MaxDepth, Mac Bi, WTemp, NT, DOC and Phy.

Afterwards, we conducted a variance inflation factor test, which eliminated total nitrogen (NT) due to its multiple correlations with several variables.

Supplementary S5.2: *Standardized fourth-corner coefficients for each trait-environmental interaction for each sampling survey. Body size (BS), dispersal ability groups (DAG), life-history group of dry-phase residents (LHG1), life-history group of dry-phase residents & spring recruits (LHG2), life-history group of dry-phase residents & summer recruits (LHG3), life-history group of non-dry-phase residents & spring migrants (LHG4) and functional feeding groups. See table 1 for more information on these traits. Phytoplankton chlorophyll-a content (Phy; µg/l), macrophyte biomass (Mac Bi; g/m^2), filamentous algae biomass (Fil Bi; g/cm^2), dissolved oxygen (O2), conductivity (EC25), pH, maximum pond depth (MaxDepth), inorganic nitrogen (DIN), soluble reactive phosphate (SRP), water temperature (Temp), fulvic acid percentage (%AC.FUL.), dissolved organic carbon (DOC), nutrient limitation indicator (DIN/PT).*

May 2012

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
BS	0	0	0	0	0	0	0	0	0	0	0	0	0	0
DAG	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LHG1	0	0	0	0	0	-0.148	0	0	0	0	0	0	0	0
LHG2	0	0	-0.018	0	0	0	0	0	0	0	0	0	0	0
LHG3	0	0	0.085	0	0	0.097	0	0	0	0	0	0.123	0	0
LHG4	0	0	0	0	0	0	0	0	0	0	0	-0.06	0	0
Collector	0.181	0	0	0	0	0	0	0	0	0	0	0	0	0
Shredder	0	0	0	0	0	0	-0.069	0	0	0	0.024	0	0	0
Scraper	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Filterers	0	0	0	0	-0.166	0	0	-0.133	0	0.126	0.009	0	0	0
Piercer	0	0	0	0	0	0	0.086	0	0	0	0.025	0	0	0
Predator	0	0	0.027	0.044	0	0	0	0	0.029	0	0	0	0	0

December 2012

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
BS	0	-0.534	0.16	0	0	0	0	0	0	-0.408	-1.004	0	0	0.192
DAG	0	0	0	0	-0.222	-1.035	-0.408	0	0	-0.429	0	-0.174	0	0
LHG1	0	0	0	0	-0.75	0	-0.483	0	0	-0.065	-1.006	0	-0.479	0
LHG2	0	0	0	0	0.314	0	0.15	0	0	0.302	0.043	0	0	0
LHG3	0	0	0.584	0.491	0	0	0	0	0	0	-1.165	0.306	-0.039	0
LHG4	0	0	0	-0.027	0	0	0	0	0	0	0	0	0.483	0
Collector	0.087	0	-0.116	0	0	0	0.285	0	0	0	0.656	0	0.724	0

December 2012

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
Shredder	0	0	0.581	0	0	0	-0.013	0	0	0	-0.016	0.203	0	0
Scraper	0	0	0	0.469	-0.503	0	0	0.475	-0.049	0	-0.265	-0.215	0	0
Filterers	0	0.046	0	0	0	-0.183	-0.581	0	0	-0.269	0	0	0	0
Piercer	0	0.247	0	-0.296	0	0	0.916	0	0	0.094	1.146	0.124	0	0
Predator	0	0	-0.435	0	0	0	0.638	0	0	0	0.811	0	0	0

February 2013

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
BS	0.445	0	0	0	0	1.036	0.24	0	0	0	0	0.249	0	0
DAG	0	0.263	0	0	0	0	0.144	0	0	-1.725	0	0	0	-0.243
LHG1	0	0.166	0	0	0	0	0	0	0	-1.56	0	0.401	0	0
LHG2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LHG3	0	0	0	0	0	0	-0.327	-0.443	0	0.476	0	-0.083	0	0
LHG4	0	0	0	0	0	-0.293	0	0	0	0	0	-0.238	0	-0.053
Collector	0	0	0	0	0	0.825	0.669	1.433	0	0	0	0	0.218	0
Shredder	0	-0.027	0	0	0	0	-0.762	0	0	0.086	0	-0.314	0	0
Scraper	0	-0.007	0	0	0	0	0	0.356	0	0	0	0	1.238	0.117
Filterers	0	0	0	0	0	-1.535	-0.622	-0.661	0	0	0	-0.417	0	0
Piercer	0	0	0	0	0	0.045	0.332	0.801	0	0	0	0.067	0	0
Predator	0	-0.361	0	0	0	0	0	0.99	0	0	0	0	0.265	0

April 2013

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
BS	0.034	0	0	0	0	0	0	0	0	0	-0.031	0	0.157	0
DAG	0	0	0.078	0	0	0	0	0	0	0	0	0	0	0
LHG1	0	0	0	0	-0.284	-0.036	0	0	0	0	0	0	-0.039	0
LHG2	-0.219	0	-0.136	0	0	0	0	0	0	0	0	0	0.033	0
LHG3	0.064	0	0	0	0	0	0	0	0	0	0	0	0	0
LHG4	0	0	0	0.2	0	0	0	0	0	0	0	0	0	0
Collector	0	0.162	-0.047	0	0	0	0	0	-0.148	0.033	0	0	0	-0.118

April 2013

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
Shredder	0.07	0	0.122	0	0	-0.08	0	0	0	0	0	0	0	0
Scraper	0	0	0.092	0	0	0	0	0	0	0	0	-0.136	0	0.026
Filterers	0.138	0	0	-0.081	-0.164	0	0	0	0	0	0	0	0	0
Piercer	-0.117	0	0	0	0	0	0	0	0	0	0	0	0	0
Predator	0	0	0	0	0	0	0	0	0.119	0	0	0	-0.097	0

June 2013

Traits	Phy	Mac Bi	Fil Bi	O2	EC25	pH	Max Depth	DIN	SRP	Temp	% AC. FUL.	DOC	DIN/PT	Burned
BS	0	0	0	0	0.006	0	0	0	0	0.016	0	0	0	0
DAG	0	0	0	0.175	0	0	0	0.159	0	0	0.035	0	0	0
LHG1	0.284	0	0	0	0.432	0	0	0	0	0.156	0	0.112	0	-0.032
LHG2	-0.041	0	0	0	0	0	0	0	0.092	-0.067	0	0	0	0
LHG3	0	0	-0.202	-0.054	0	-0.107	0	0	0.003	0	0	0.012	0	0
LHG4	0	0	0	0	-0.139	0	0	0.145	0	0	0	0	0	0
Collector	0	0	0	0	0	0	0	0	0	0.067	0	0	0.047	0
Shredder	0	0	0	0	0	0	0	0	0	-0.182	0	0	-0.053	0
Scraper	0	0	0	0	0	0	-0.166	0.207	0	0	0.132	0	0	0
Filterers	0	0	0	0	0	0	0	0	-0.128	-0.023	0.011	-0.089	0	0
Piercer	0	0.059	0	0	0	0.111	0	-0.083	0.026	0	0	0	0	0
Predator	0	0	-0.038	0	0	0	0	0	0.019	0	0	0	0	0.018

Supplementary S5.3: *Standardized coefficients for each trait in each sampled pond. Body size (BS), Dispersal ability groups (DAG), life-history group of dry-phase residents (LHG1), life-history group of dry-phase residents & spring recruits (LHG2), life-history group of dry-phase residents & summer recruits (LHG3), life-history group of non-dry-phase residents & spring migrants (LHG4) and functional feeding groups. See table 1 for more information on these traits.*

May 2012

Sampled ponds

Traits	**A**	**B**	**C**	**D**	**E**	**F**	**G**
BS	0	**-0.004**	**-0.001**	**-0.001**	**-0.003**	**-0.004**	**-0.001**
DAG	0	**0.001**	**0.001**	**0**	**0.002**	**0.002**	**0.001**
LHG1	**0.003**	**0.008**	**0.002**	**0.001**	**0.002**	**0.015**	**0.002**
LHG2	0	**0.009**	0	0	0	**0.007**	0
LHG3	0	0	0	0	0	**0.004**	0
LHG4	0	**0.009**	0	0	0	0	0
Collector	**0.003**	**0.008**	**0.001**	**0.002**	**0.002**	0	**0.002**
Shredder	0	**-0.002**	0	0	**0.001**	**0.003**	**-0.001**
Scraper	**0.006**	**0.005**	**0.002**	**0.002**	**0.005**	**0.018**	**0.002**
Filterers	0	**-0.007**	0	0	0	0	0
Piercer	**0.001**	**0.003**	0	0	0	0	**0.001**
Predator	**0.005**	**0.002**	**0.002**	**0.003**	**0.006**	**0.016**	**0.003**

December 2012

Sampled ponds

Traits	**A**	**B**	**C**	**D**	**E**	**F**	**G**
BS	**-0.002**	0	0	**0.001**	**-0.002**	**-0.004**	**0.001**
DAG	0	**0.001**	0	**-0.002**	0	**-0.011**	**0.017**
LHG1	**0.007**	0	**0.003**	**0.001**	**0.003**	**0.023**	**0.013**
LHG2	**0.001**	0	0	0	0	**0.03**	**0.009**
LHG3	0	0	0	0	0	**0.014**	0
LHG4	0	**0.001**	0	0	0	**0.017**	0
Collector	**0.01**	**0.005**	**0.005**	**0.002**	0	**-0.001**	**0.008**

December 2012

Sampled ponds

Traits	A	B	C	D	E	F	G
Shredder	0	-0.002	-0.001	-0.001	-0.001	0.005	-0.017
Scraper	0.013	0.003	0.004	0.001	0.005	0.003	0.002
Filterers	0	0	0.001	0	0.006	0	0.004
Piercer	0	0.002	0	0.001	0	0.002	-0.011
Predator	0.019	0.004	0.007	0.003	0.009	0.002	0.011

February 2013

Sampled ponds

Traits	A	B	C	F	G
BS	0.001	0	0.005	-0.001	-0.004
DAG	0	0	-0.021	-0.005	0.004
LHG1	0.002	0.002	0.002	0.014	0.007
LHG2	0.003	0.002	0.009	0.021	0
LHG3	0	0	0.004	0.01	0
LHG4	0	0.002	0.013	0.015	0
Collector	0.003	0.003	0.005	-0.002	0.002
Shredder	-0.001	-0.001	0.003	-0.001	-0.003
Scraper	0.003	0.001	0.002	0	0.007
Filterers	0	-0.001	-0.003	-0.002	0.008
Piercer	0.001	0.001	0.005	-0.002	0.001
Predator	0.003	0.001	0.01	-0.005	0.012

April 2013

Sampled ponds

Traits	A	B	C	D	E	F	G
BS	-0.003	0.002	-0.002	0	-0.002	-0.001	-0.001
DAG	-0.002	0.001	0.002	0.001	0	-0.001	0.001
LHG1	0.008	0.004	0.003	0.001	0.004	0.003	0.004

April 2013

	Sampled ponds						
Traits	**A**	**B**	**C**	**D**	**E**	**F**	**G**
LHG2	0.011	0.003	0	0.001	0.004	0.005	0
LHG3	0.005	0	0	0	0	0.002	0
LHG4	0.01	0	0	0	0	0.004	0
Collector	-0.001	0.003	0.002	0.001	0.007	0	0
Shredder	0.001	-0.002	0	-0.001	-0.002	0	0
Scraper	-0.001	0.002	0.003	0	0.007	0.001	0.005
Filterers	0	0	0	0	-0.002	-0.001	0
Piercer	-0.001	0.003	0	0	0.004	0	0
Predator	-0.002	0.002	0.003	0	0.008	0	0.004

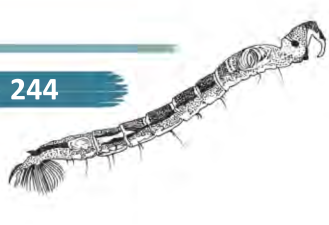
June 2013

	Sampled ponds						
Traits	**A**	**B**	**C**	**D**	**E**	**F**	**G**
BS	-0.002	-0.001	-0.002	-0.001	0.002	0	-0.002
DAG	0	-0.001	0.011	0.001	0	0	-0.001
LHG1	0.011	0.003	0.007	0.018	0	0.003	0.003
LHG2	0.013	0.005	0	0.017	0	0.002	0.007
LHG3	0.007	0.002	0	0	0	0.002	0.005
LHG4	0.012	0.006	0	0.01	0.001	0.001	0.009
Collector	0.003	0	0	0.007	0.008	0	0
Shredder	-0.003	0	-0.005	-0.006	-0.007	0	-0.001
Scraper	0.005	0	0.009	-0.003	0.016	0.001	0.001
Filterers	-0.004	0	0	-0.008	-0.002	0	0.001
Piercer	0.003	0.001	0	0.006	0.005	0.001	0
Predator	0.002	-0.001	0.001	0.005	0.011	0	0

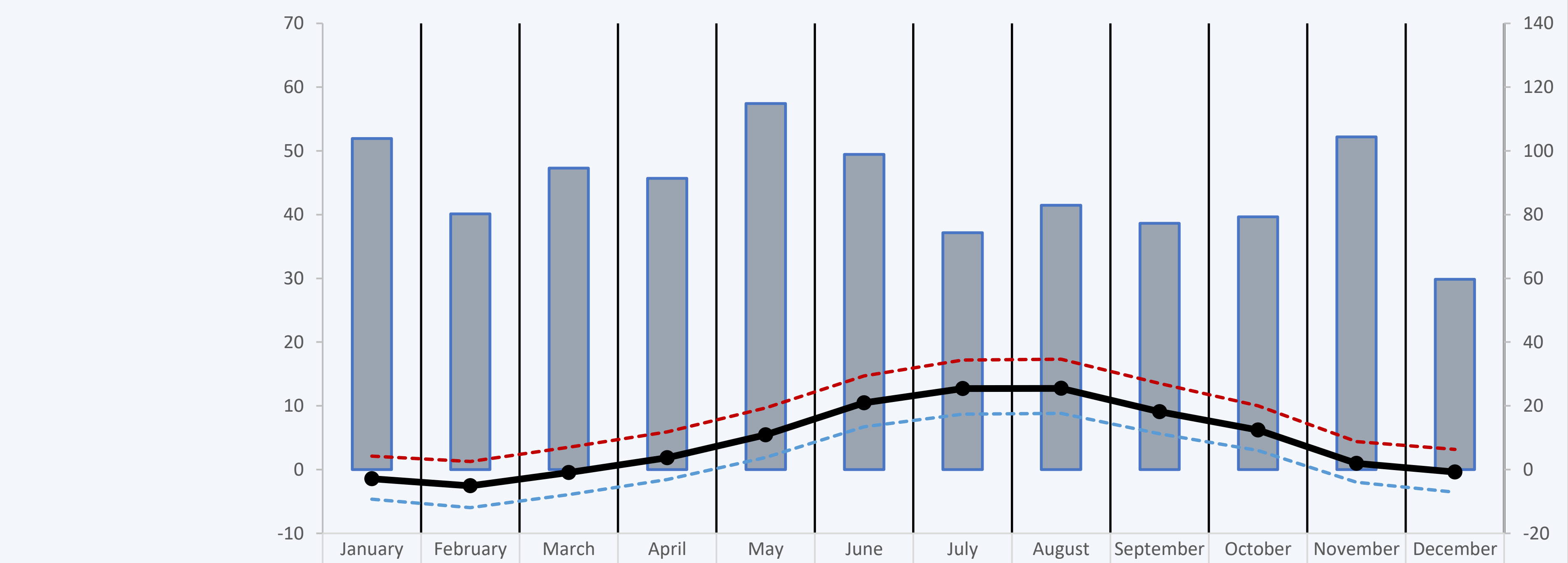
SUPPLEMENTARY MATERIAL

S6

CHAPTER III SUPPLEMENTARY MATERIAL



Supplementary S6.1: *Climatic diagram from Malniu meteorological station (Meranges municipality, Cerdanya) located next to sampled area. Data has been provided by the Catalan meteorological service (www.meteocat.cat) and goes back to 1999, when the station was installed. Average monthly total precipitation in mm (Average total prec.) in bars and average mean month temperature in Celsius degrees (Average Mean temp.) black line, average mean month maximum temperature in Celsius degrees (Average Max. temp.) dashed red line, average mean month minimum temperature in Celsius degrees (Average Min. temp.) blue dashed line.*



	January	February	March	April	May	June	July	August	September	October	November	December
▬ Average total prec. (mm)	103,89	80,26	94,57	91,36	114,87	98,91	74,34	82,91	77,24	79,33	104,41	59,74
━●━ Average Mean temp. (Cº)	-1,43	-2,55	-0,47	1,87	5,46	10,48	12,72	12,74	9,09	6,22	0,99	-0,37
- - - Average Max. temp. (Cº)	2,12	1,27	3,52	5,92	9,67	14,69	17,18	17,32	13,52	10,01	4,41	3,16
- - - Average Min. Temp. (Cº)	-4,63	-5,96	-3,93	-1,56	1,92	6,71	8,69	8,82	5,60	3,02	-2,00	-3,53

Supplementary S6.2A: *Table with closeness values for each one of the water bodies (network nodes) and their topographically assigned names. Closeness values were calculated using the "sna" package available for R studio and based on the minimum spanning tree built on the UTM location values of each one of the water bodies.*

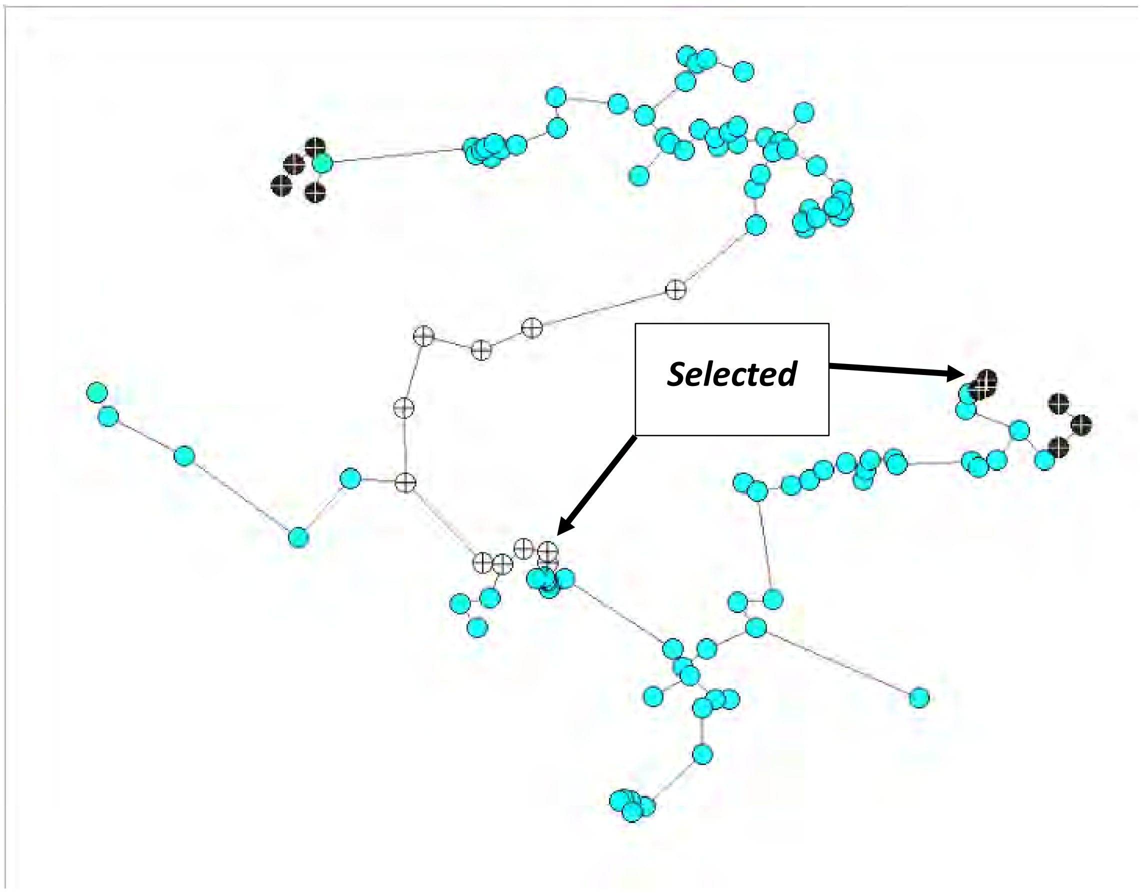
Name	Closeness values	Sampling feasibility
Els Clots de Guils 18	*0.028*	**Yes**
Pla de la Vila 1	0.029	No
Els Clots de Guils 11	0.029	No
Els Clots de Guils 17	*0.029*	**Yes**
Pla de la Vila 3	0.029	No
Els Clots de Guils 10	0.030	No
Els Clots de Guils 16	0.030	No
Pla de la Vila 2	0.030	No
Pla de la Vila 4	0.030	No
Els Clots de Guils 9	0.031	No
Els Clots de Guils 15	0.031	Yes
Pla de la Vila 5	0.031	No
Els Clots de Guils 8	0.032	No
Els Clots de Guils 14	0.032	No
Pla dels Empedrats 1	0.032	No
Els Clots de Guils 13	0.033	No
Pla dels Empedrats 6	0.033	No
Pla dels Empedrats 2	0.033	No
Els Clots de Guils 12	0.034	No
Pla dels Empedrats 3	0.034	No
Els Clots de Guils 20	0.035	No
Pla dels Empedrats 4	0.035	No
Els Clots de Guils 19	0.036	No
Pla dels Empedrats 5	0.036	No
Tartera Roja 11	0.037	No
Moscador de Dalt 4	0.037	No
Pla dels Empedrats 8	0.037	No

Pla de la Tosa 16	0.037	No
Tartera Roja 10	0.038	No
Moscador de Dalt 3	0.038	No
Pla dels Empedrats 9	0.038	No
Tartera Roja 7	0.039	No
Pla de la Tosa 18	0.039	No
Pla de la Tosa 20	0.039	No
Tartera Roja 9	0.039	No
Mulleres de Roca Colom 6	0.039	No
Pla dels Empedrats 10	0.040	No
Moscador de Dalt 2	0.040	No
Tartera Roja 8	0.040	No
Pla de la Tosa 19	0.040	No
Pla dels Empedrats 11	0.041	No
Mulleres de Roca Colom 2	0.041	No
Mulleres de Roca Colom 5	0.041	No
Tartera Roja 6	0.041	No
Moscador de Dalt 5	0.041	No
Pla de la Tosa 17	0.042	No
Pla de la Tosa 1	0.042	No
Tartera Roja 5	0.042	No
Mulleres de Roca Colom 8	0.043	No
Mulleres de Roca Colom 7	0.043	No
Moscador de Dalt 1	0.043	No
Pla de la Tosa 2	0.043	No
Tartera Roja 4	0.044	No
Pla dels Empedrats 12	0.044	No
Tartera Roja 1	0.044	No
Mulleres de Roca Colom 9	0.044	No
Mulleres de Roca Colom 11	0.045	No
Pla de la Tosa 3	0.045	No
Els Rasets 4	0.045	No

Tartera Roja 3	0.045	No
Pla de la Tosa 4	0.046	No
Tartera Roja 2	0.046	No
Mulleres de Roca Colom 10	0.046	No
Els Rasets 1	0.047	No
Coll de Mulleres 2	0.047	No
Els Rasets 3	0.047	No
Pla de la Tosa 5	0.047	No
Pla de la Tosa 6	0.048	No
La Feixa 1	0.048	No
La Creu de la Gavarrera 1	0.048	No
Mulleres de Roca Colom 4	0.048	No
La Feixa 5	0.048	No
Pla de la Tosa 8	0.049	No
Pla de la Tosa 11	0.049	No
Els Rasets 2	0.049	No
La Feixa 3	0.049	No
Coll de Mulleres 1	0.049	No
Pla de la Tosa 9	0.050	No
La Feixa 2	0.050	No
Mulleres de Roca Colom 1	0.050	No
La Feixa 4	0.051	No
Els Clots de Guils 1	0.051	No
Pla de la Tosa 10	0.051	No
Coll de Mulleres 3	0.051	No
La Feixa 1	0.052	No
Els Clots de Guils 27	0.052	No
Mulleres de l'Home Mort	0.052	No
Pla de la Tosa 7	0.052	No
Pla de la Tosa 15	0.053	No
Els Clots de Guils 21	0.053	Yes
La Feixa 3	0.053	No

Els Clots de Guils 2	0.054	No
Pla de la Tosa 12	0.054	No
La Feixa 2	0.054	No
Estany del Refugi	0.054	No
Els Clots de Guils 25	0.055	Yes
Els Clots de Guils 5	0.055	Yes
Pla de la Tosa 13	0.055	No
Pla de la Tosa 14	0.056	No
Els Clots de Guils 4	0.056	Yes
Mulleres de Roca Colom 13	0.056	No
Els Clots de Guils 22	0.057	Yes
Els Clots de Guils 3	0.057	No
Estany Sec	0.057	No
Mulleres de Roca Colom 12	0.057	No
Els Clots de Guils 23	0.057	Yes
Mulleres de Roca Colom 3	0.058	No
Els Clots de Guils 26	0.058	No
Estany del Gespetar	0.058	No
Els Clots de Guils 6	*0.059*	**Yes**
Estany Mal	0.059	No
Els Clots de Guils 24	0.059	No
Estany del Prat Fondal	0.059	No
Els Clots de Guils 7	0.060	No
Prat Fondal	0.060	No
Estany de Malniu	0.060	No
Clots de Guils 29	0.060	No
Clots de Guils 30	0.060	No
Els Clots de Guils 28	0.060	No

Supplementary S6.2B: *Minimum spanning tree (package "ape" on R language) of the water bodies network with the ten most central (white crossed dots) and the ten most isolated water bodies (black crossed dots). Arrows point to the two finally selected ponds according to location and other criteria (see Methods section).*



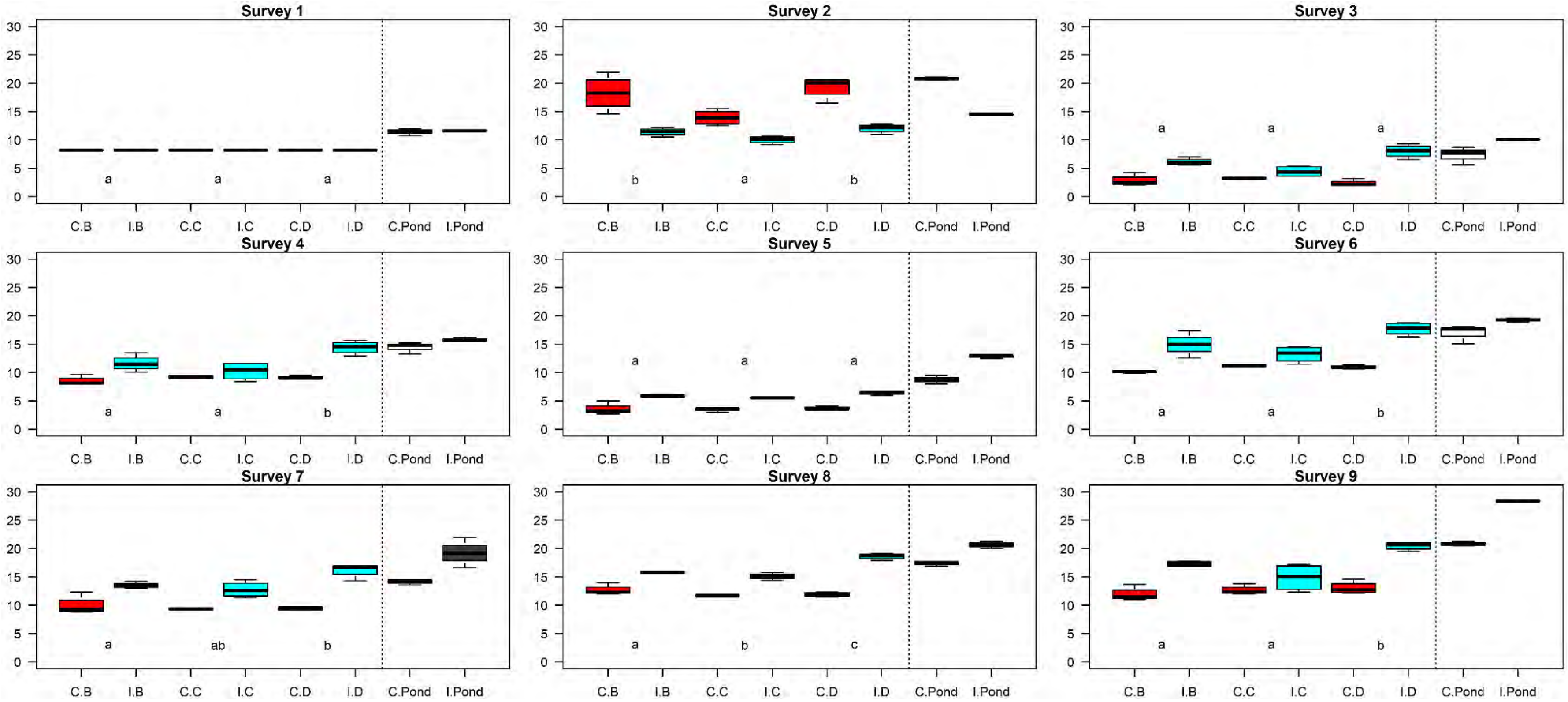
Supplementary S6.2C: *Feasible isolated and central ponds physic, chemical and biological characteristics considered for selecting the two sampled ponds. The feasible most isolated ponds (black background and white text) and the most feasible central pond (white background and black text). The two finally selected ponds are marked with a green background: Els Clots de Guils 17 as the isolated and Els Clots de Guils 6 as the central*

Name	**Pond basin surface**	**Maximum depth**	**Diss. O2**	**Cond.**	**pH**	**Water temp.**	**ECELS value**	**DIC**	**DOC**	**TIC**	**TOC**	**SRP**	**DIN**	**TN**	**TP**	**Pond vegetation cover**	**Transparency**	**Substrate**
Isolated feasible ponds																		
Els Clots de Guils 18	4151	23.000	6.09	51.9	6.8	14	91	2.70	7.26	6.17	8.87	0.00	0.03	0.69	0.06	Fully covered	Clear water	Grass
Els Clots de Guils 17	9110	91.000	8.12	55	7.79	25.1	86	5.22	9.21	5.75	9.21	0.00	0.13	0.69	0.03	Partially covered	Clear water	Rock
Central feasible pond																		
Els Clots de Guils 6	10069	45.000	8.13	19.74	6.57	28.1	93	0.34	18.06	0.87	18.08	0.00	0.09	1.43	0.11	Partially covered	Clear water	Rock

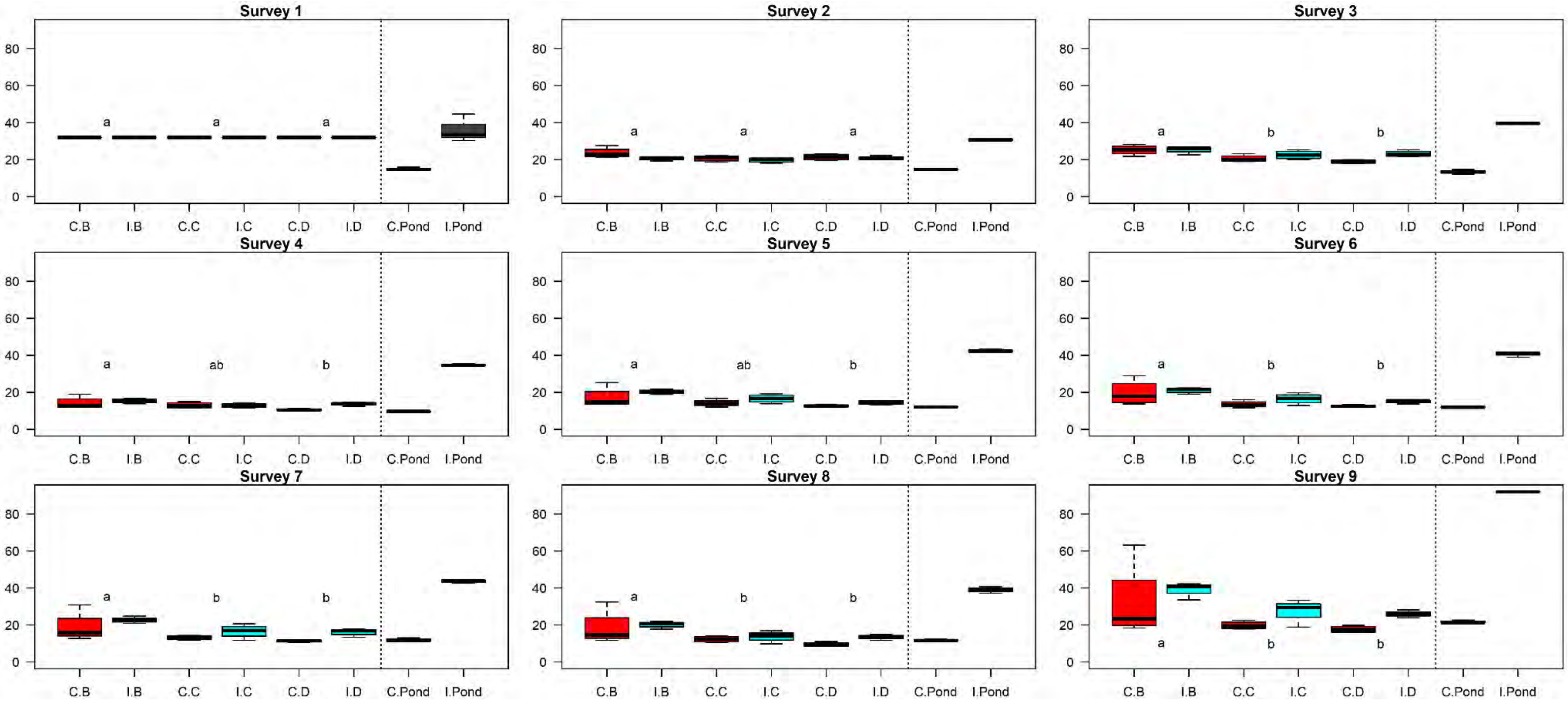
Supplementary S6.3A: *Analysis of variance results for mesocosms environmental metrics: Temperature (Cº), Conductivity (mS/cm) and dissolved oxygen (mg/l) along all sampled weeks (from first week "1" until ninth week "9"). See supplementary S3 B, C and D for post-hoc differences.*

Survey	**Centrality**		**Sets**		**Interaction**	
	F- value $_{(1,20)}$	p-value	F- value $_{(2,20)}$	p-value	F- value $_{(2,20)}$	p-value
Temperature						
1	1	0.33	1	0.39	-	-
2	**99.61**	**<0.01**	**13.48**	**<0.01**	-	-
3	**96.41**	**<0.01**	1.08	0.36	**11.05**	**<0.01**
4	**52.79**	**<0.01**	**7.2**	**<0.01**	**6.83**	**0.01**
5	**118.37**	**<0.01**	1.72	0.2	-	-
6	**117.12**	**<0.01**	**7.8**	**<0.01**	**9.21**	**<0.01**
7	**118.79**	**<0.01**	**4.01**	**0.04**	**4.62**	**0.02**
8	**409.05**	**<0.01**	**18.33**	**<0.01**	**21.39**	**<0.01**
9	**77.85**	**<0.01**	**8.29**	**<0.01**	**5.81**	**0.01**
Conductivity						
1	1	0.33	1	0.39	-	-
2	**4.73**	**0.04**	2.42	0.11	-	-
3	**7.16**	**0.01**	**9.37**	**<0.01**	-	-
4	**4.81**	**0.04**	**5.34**	**0.01**	-	-
5	**8.04**	**0.01**	**8.41**	**<0.01**	-	-
6	**6.02**	**0.02**	**11.35**	**<0.01**	-	-
7	**11.55**	**<0.01**	**9.33**	**<0.01**	-	-
8	**5.37**	**0.03**	**10.51**	**<0.01**	-	-
9	**11.78**	**<0.01**	**7.15**	**<0.01**	-	-
Dissolved oxigen						
1	1	0.33	1	0.39	-	-
2	**12.58**	**<0.01**	**10.95**	**<0.01**	**8.01**	**<0.01**
3	**5.97**	**0.03**	**5.57**	**0.01**	**6.11**	**0.01**
4	**71.29**	**<0.01**	**24.63**	**<0.01**	**33.7**	**<0.01**
5	**6.47**	**0.02**	0.22	0.81	-	-
6	**21.88**	**<0.01**	1.95	0.17	**6.67**	**0.01**
7	**14.23**	**<0.01**	0.36	0.71	-	-
8	**24.94**	**<0.01**	2.43	0.11	-	-
9	**9.41**	**0.01**	1.38	0.27	-	-

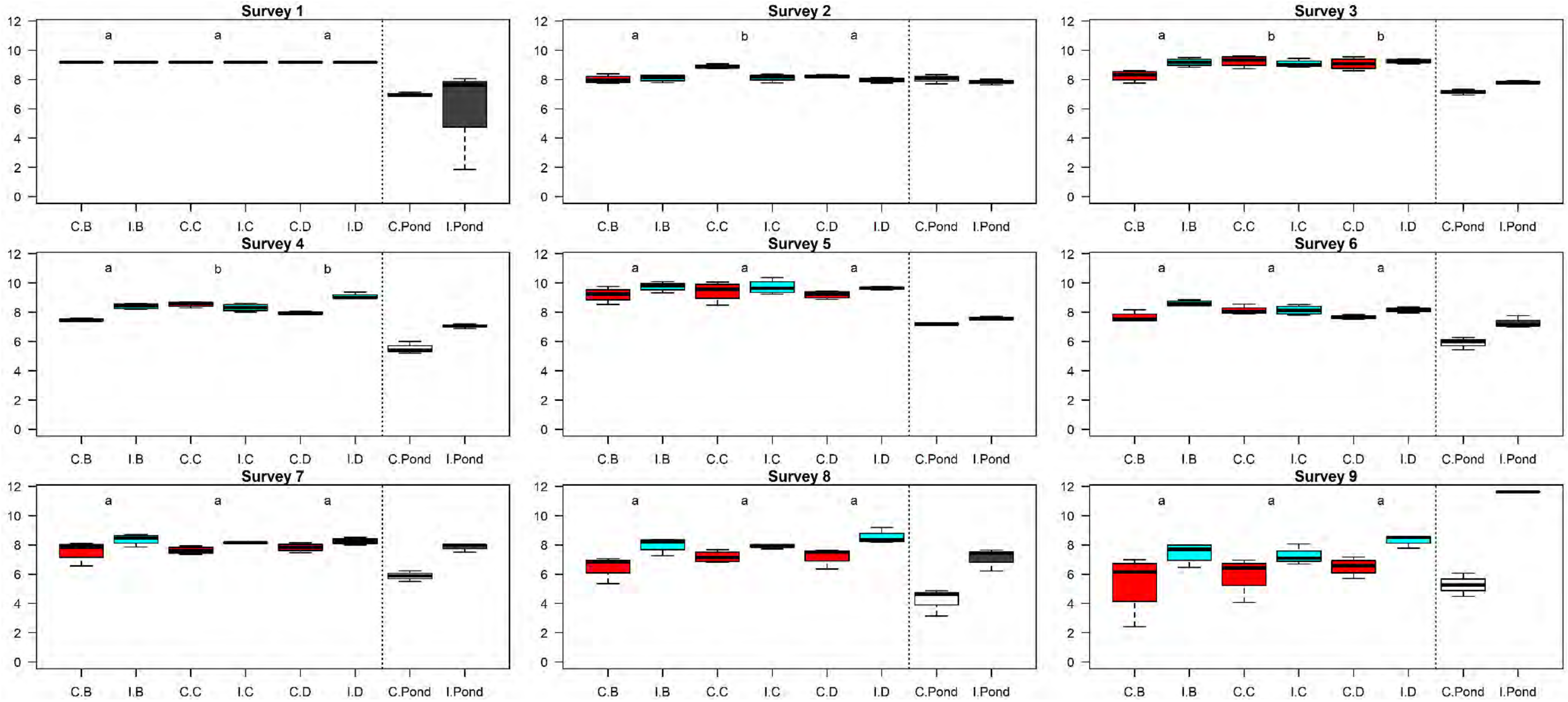
Supplementary S6.3B: *Boxplots with temperature values (ºC) for each sampled mesocosm (C.B: Central-between; I.B: Isolated-between; C.C: Central-control; I.C: Isolated-control; C.D: Central-distant and I.D: Isolated-distant) and pond (C.Pond: Central pond and I.Pond: Isolated pond) for all weekly sampling surveys (from first week "Survey 1" until ninth week "Survey 9"). Red boxplots correspond to central location mesocosms, cyan boxplots correspond to isolated location mesocosms, white boxplots correspond to central pond and grey boxplots correspond to isolated pond. Post-hoc differences within each set type (control, distant and between) are specified with lowercase letters.*



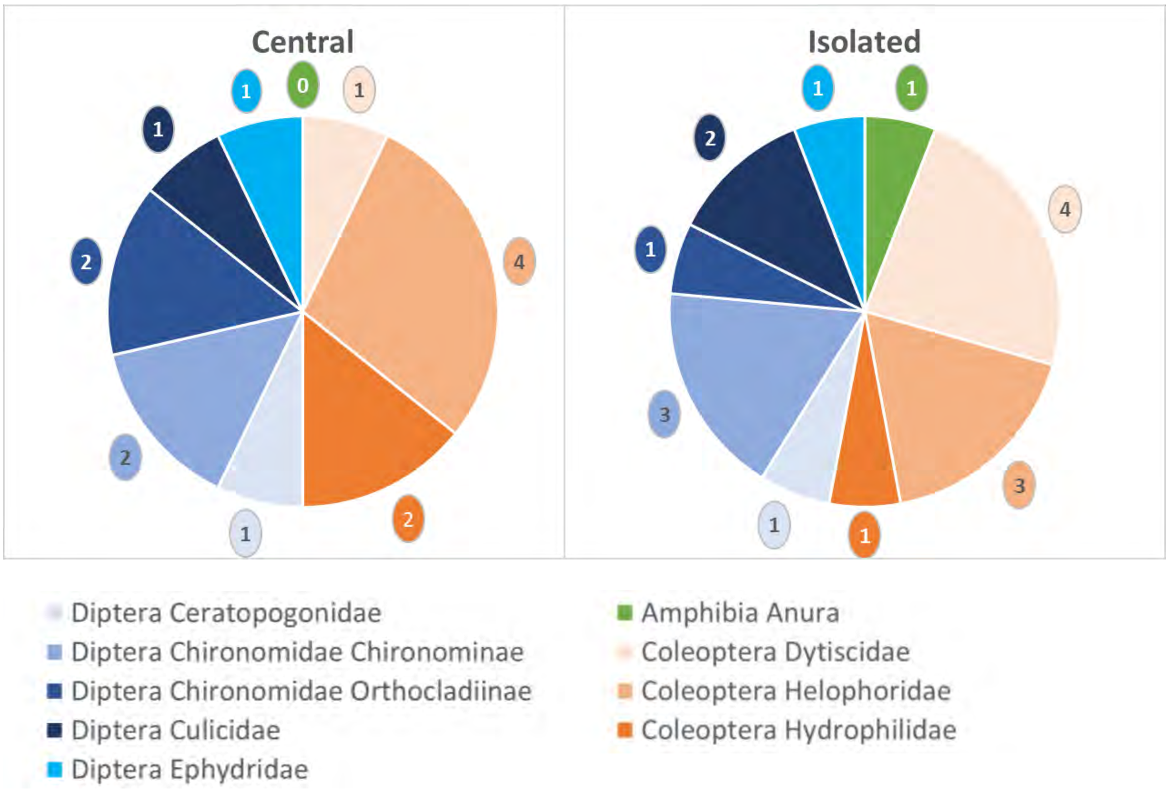
Supplementary S6.3C: *Boxplots with conductivity values (mS/cm) for each sampled mesocosm (C.B: Central-between; I.B: Isolated-between; C.C: Central-control; I.C: Isolated-control; C.D: Central-distant and I.D: Isolated-distant) and pond (C.Pond: Central pond and I.Pond: Isolated pond) for all weekly sampling surveys (from first week "Survey 1" until ninth week "Survey 9"). Red boxplots correspond to central location mesocosms, cyan boxplots correspond to isolated location mesocosms, white boxplots correspond to central pond and grey boxplots correspond to isolated pond. Post-hoc differences within each set type (control, distant and between) are specified with lowercase letters.*



Supplementary S6.3D: *Boxplots with dissolved oxygen values (mgO$_2$/l) for each sampled mesocosm (C.B: Central-between; I.B: Isolated-between; C.C: Central-control; I.C: Isolated-control; C.D: Central-distant and I.D: Isolated-distant) and pond (C.Pond: Central pond and I.Pond: Isolated pond) for all weekly sampling surveys (from first week "Survey 1" until ninth week "Survey 9"). Red boxplots correspond to central location mesocosms, cyan boxplots correspond to isolated location mesocosms, white boxplots correspond to central pond and grey boxplots correspond to isolated pond. Post-hoc differences within each set type (control, distant and between) are specified with lowercase letters.*



Supplementary S6.4: *Pie chart with mesocosms active dispersers species composition between central location and isolated location. Similar colours correspond to similar taxonomic orders: oranges to Coleoptera, blues to Diptera and greens to Amphibia. Numbers around each plot fraction, which correspond to taxonomic families, correspond to the total number of taxa found on each taxonomic family.*



***Supplementary S6.5A**: Mesocosms species checklist for each one of the mesocosms sets (control set, between set and distant set) in both studied locations (central and isolated). * Indicates small individuals that have been classified to Family level.*

	Central & Control	*Central & Distant*	*Central & Between*	*Isolated & Control*	*Isolated & Distant*	*Isolated & Between*
Turbellaria Rhabdocoela						
Gieysztoria diadema	-	-	-	-	-	X
Gieyztoria sp.1	-	-	-	X	-	X
Cladocera						
Chydorus sphaericus	-	-	-	X	-	X
Daphnia longispina	-	-	-	X	-	X
Copepoda Calanoida						
Diaptomus cyaneus	-	-	-	X	-	X
Ostracoda Cypridoidea						
Paralimnocythere psammophila	-	-	-	-	-	X
Coleoptera Dytiscidae						
Hydroporus foveolatus	-	-	-	X	-	-
Hydroporus nigrita	-	X	-	X	-	X
Hygrotus impressopunctatus	-	-	-	-	-	X
Ilybius albarracinensis	-	-	-	X	-	-
Coleoptera Helophoridae						
Helophorus discrepans	X	X	X	X	X	X
Helophorus flavipes	X	X	X	X	X	X
Helophorus gr. *maritimus*	X	X	-	-	-	-
Helophorus granularis	X	-	-	X	X	X
Coleoptera Hydrophilidae						
Anacaena limbata	-	X	-	-	-	-
Limnebius truncatellus	-	X	-	-	-	-
Limnebius sp.	X	X	-	X	X	X

Diptera Ceratopogonidae						
Dasyhelea sp.	-	X	X	X	X	X
Diptera Chironomidae Chironominae						
Chironomus alpestris	-	X	-	X	-	-
Cladotanytarsus sp.	-	-	-	-	-	X
Micropsectra lindrothi	X	X	X	X	X	X
Diptera Chironomidae Orthocladiinae						
Chaetocladius sp.	X	X	-	-	-	X
Orthocladius ashei	-	-	X	-	-	-
Diptera Culicidae						
Culex territans	-	-	X	-	X	X
Culicidae indet. *	-	X	-	-	-	-
Diptera Ephydridae						
Ephydridae indet.*	X	-	X	X	-	X
Amphibia Anura						
Rana temporania	-	-	-	-	-	X

Supplementary S6.5B: *Pond and mesocosms species checklist for the two studied locations (central and isolated). * Indicates small individuals that have been classified to Family level.*

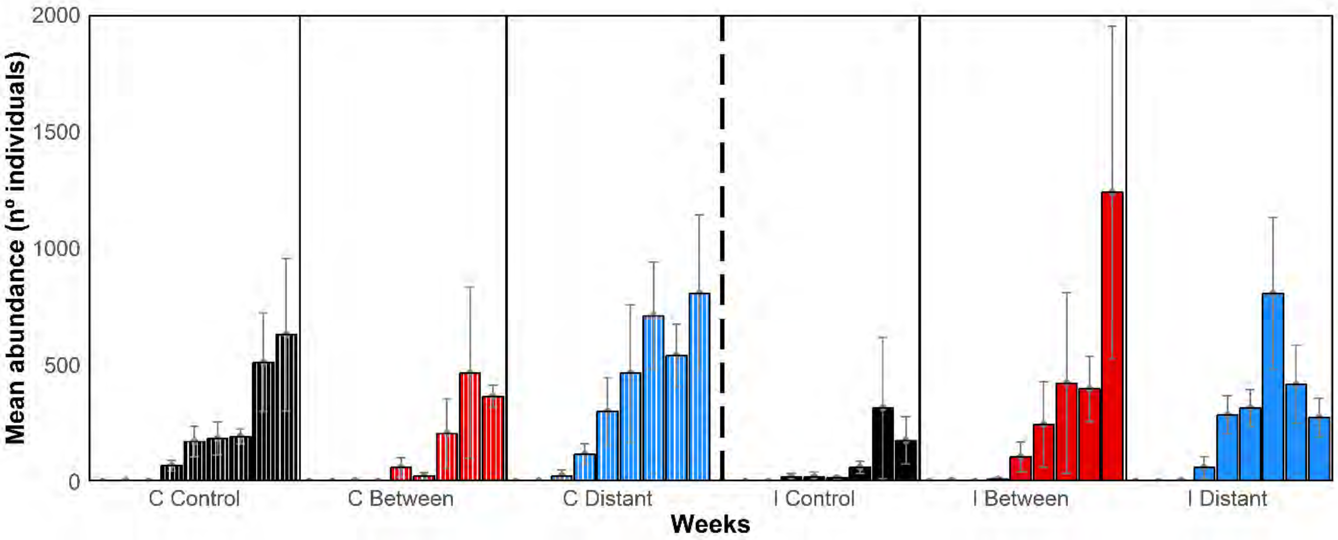
	Central mesocosm	***Central pond***	***Isolated mesocosm***	***Isolated pond***
Turbellaria Rhabdocoela				
Dalyellidae indet.	-	X	-	X
Gieysztoria diadema	-	X	X	-
Gieyztoria sp.1	-	X	X	X
Gyratrix hermaphroditus	-	-	-	X
Phaenocora sp.	-	-	-	X
Rhabdocoela indet. 1	-	X	-	-
Rhabdocoela indet. 2	-	X	-	X
Annelida Oligochaeta				
Enchytraeidae indet.	-	X	-	X
Lumbricidae indet.	-	X	-	-
Lumbriculidae indet.	-	X	-	X
Naidinae indet.	-	X	-	X
Tubificinae indet. with setae	-	X	-	X
Tubificinae indet. without setae	-	-	-	X
Bivalvia Sphaeriidae				
Pisidium casertanum	-	X	-	X
Gastropoda Lymnaeidae				
Galba truncatula	-	-	-	X
Branchiopoda Anostraca				
Chirocephalus diaphanus	-	X	-	X
Hydrachnidia				
Hydrachna skorikowi	-	X	-	X
Hydrachnidia indet.	-	X	-	-
Piona sp.1	-	X	-	X
Piona sp.2	-	X	-	X

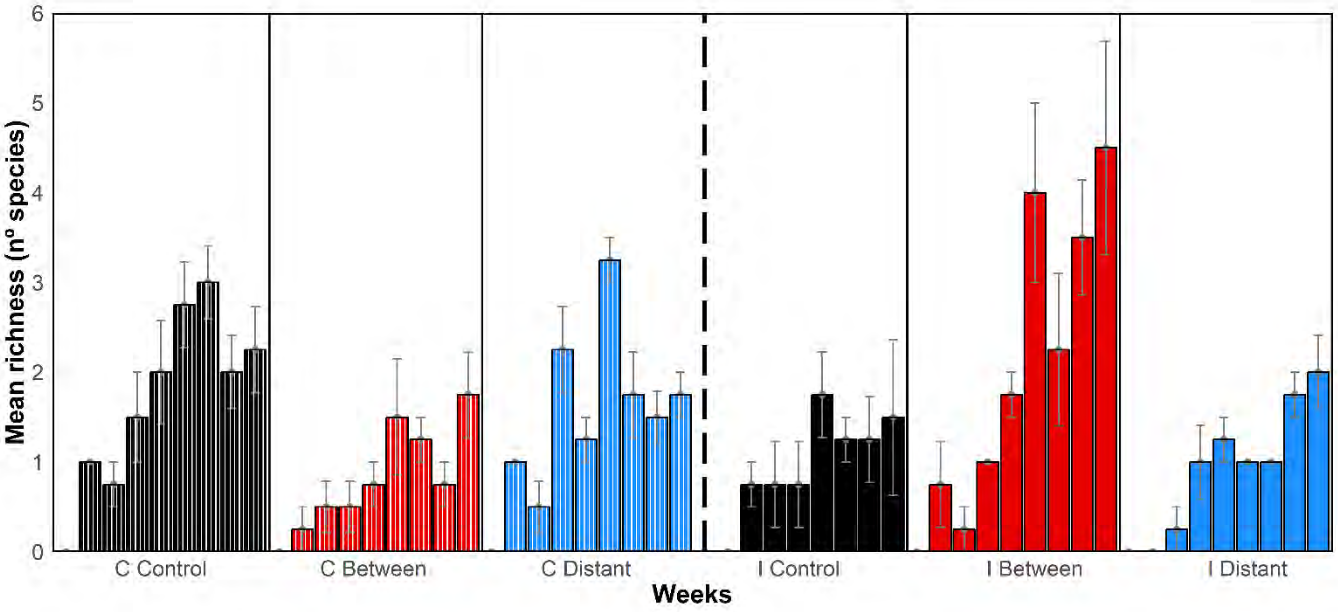
Ephemeroptera Baetidae				
Baetidae indet.	-	X	-	X
Cloeon gr. *dipterum*	-	-	-	X
Odonata Aeshnidae				
Aeshnidae indet.	-	-	-	X
Odonata Lestidae				
Lestes dryas	-	X	-	X
Odonata Libellulidae				
Sympetrum flaveolum	-	X	-	X
Hemiptera Corixidae				
Sigara nigrolineata	-	X	-	-
Hemiptera Gerridae				
Gerris costai	-	X	-	X
Hemiptera Notonectidae				
Notonectidae indet.	-	X	-	-
Hemiptera Pleidae				
Plea minutissima	-	-	-	X
Coleoptera Dryopidae				
Dryops auriculatus	-	-	-	X
Dryops sp.	-	X	-	X
Coleoptera Dytiscidae				
Agabus cf. *lapponicus*	-	X	-	-
Agabus sp. (larvae)	-	X	-	X
Deronectes aubei sanfilipoi	-	-	-	X
Deronectes sp. (larvae)	-	X	-	X
Dytiscus sp. (larvae)	-	X	-	X
Hydroporus foveolatus	-	-	X	-
Hydroporus nigrita	X	-	X	-
Hydroporus cf. *nigrita*	-	X	-	-
Hydroporus cf. *vagepictus*	-	X	-	-
Hygrotus impressopunctatus	-	-	X	-

Hygrotus marklini	-	X	-	X
Hygrotus sp. (larvae)	-	-	-	X
Ilybius albarracinensis	-	-	X	X
Laccophilus hyalinus	-	-	-	X
Coleoptera Helophoridae				
Helophorus discrepans	X	X	X	X
Helophorus flavipes	X	X	X	X
Helophorus gr. *maritimus*	X	X	-	-
Helophorus granularis	X	-	X	-
Coleoptera Hydrophilidae				
Anacaena limbata	X	X	-	-
Berosus luridus	-	X	-	X
Enochrus fuscipenis	-	-	-	X
Limnebius truncatellus	X	-	X	-
Trichoptera Limnephilidae				
Limnephilus bipunctatus	-	X	-	X
Limnephilus stigma	-	X	-	X
Limnephilus vittatus	-	X	-	X
Limnephilidae indet. *	-	X	-	-
Diptera Ceratopogonidae				
Culicoides sp.	-	X	-	X
Dasyhelea sp.	X	-	X	-
Diptera Chironomidae Chironominae				
Chironomus alpestris	X	X	X	X
Cladotanytarsus sp.	-	-	X	-
Cricotopus sp.	-	-	-	X
Micropsectra lindrothi	X	-	X	-
Polypedilum sp.	-	X	-	-
Tanytarsus sp.	-	-	-	X

Diptera Chironomidae Orthocladiinae				
Chaetocladius sp.	X	-	X	-
Cricotopus sp.	-	-	-	X
Limnophyes sp.	-	-	-	X
Orthocladinae indet. *	-	-	-	X
Orthocladius ashei	X	-	-	-
Psectrocladius (P.) sp.	-	X	-	X
Pseudosmittia sp.	-	X	-	-
Smittia sp.	-	X	-	-
Thienemannia sp.	-	X	-	X
Diptera Chironomidae Tanypodinae				
Ablabesmyia monilis	-	X	-	X
Procladius sp.	-	X	-	X
Tanypodinae indet. *	-	X	-	X
Zavrelimyia sp.	-	X	-	-
Diptera Culicidae				
Aedes (Ochlerotatus) communis	-	X	-	-
Aedes (Ochlerotatus) pullatus	-	X	-	-
Aedes (Ochlerotatus) sp.	-	X	-	X
Aedes (Ochlerotatus) surcoufi	-	X	-	X
Culex territans	X	-	X	-
Culicidae indet. *	X	X	-	X
Diptera Ephydridae				
Ephydridae indet.	X	-	X	-
Hydrellia sp.	-	X	-	-
Diptera Scathophagidae				
Scathophagidae indet.	-	-	-	X
Amphibia Anura				
Rana temporania	-	X	X	X

Supplementary S6.6: *Mean abundance (individuals/l) and mean richness values for each mesocosm set type along time (all nine surveyed weeks). Central locations are indicated in dashed contour black lines and isolated locations without any lines. Control sets in black, between sets in red and distant sets in blue.*





Supplementary S6.7: *Model selection results for each one of the six GAMM conducted for mesocosms data. For abundance of 1) weak dispersers - distance (control set vs distant sets), 2) weak dispersers – direction (control set vs between sets), 3) strong dispersers - distance (control set vs distant sets), 4) strong dispersers – direction (control set vs between sets). Finally, for both weak and strong dispersers 5) accumulated richness - distance (control sets vs distant sets) and 6) accumulated richness – direction (control sets vs between sets). AICc corresponds to the Akaike's information criterion of the second order, delta corresponds to the AICc differences among the tested models and finally the AICc weights.*

Distance based analysis

Weak dispersers — Model B3

Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-364.86	761.40	14.39	0.00
Model B1	11	-362.13	748.50	1.48	0.29
Model B2	11	-372.99	770.30	23.21	0.00
Model B3	**11**	**-361.39**	**747.00**	**0.00**	**0.60**
Model B4	11	-370.18	764.60	17.59	0.00
Model C	8	-366.68	750.60	3.53	0.10
Model D	8	-370.08	757.40	10.31	0.00

Strong dispersers — Model D

Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-134.85	301.40	22.24	0.00
Model B1	11	-127.84	279.90	0.78	0.40
Model B2	11	-138.31	300.90	21.72	0.00
Model B3	11	-145.34	314.90	35.77	0.00
Model B4	11	-142.25	308.80	29.60	0.00
Model C	8	-150.11	317.40	38.26	0.00
Model D	**8**	**-130.98**	**279.20**	**0.00**	**0.60**

Accumulated richness — Model D

Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-180.40	392.50	11.22	0.00
Model B1	11	-180.46	385.20	3.91	0.12
Model B2	11	-182.02	388.30	7.04	0.03
Model B3	11	-188.13	400.50	19.26	0.00
Model B4	11	-183.43	391.10	9.85	0.01
Model C	8	-189.86	396.90	15.64	0.00
Model D	**8**	**-182.04**	**381.30**	**0.00**	**0.85**

Direction based analysis

Weak dispersers — Model D

Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-317.56	666.80	7.42	0.01
Model B1	11	-319.46	663.20	3.78	0.06
Model B2	11	-319.19	662.70	3.24	0.08
Model B3	11	-318.05	660.40	0.97	0.24
Model B4	11	-321.34	666.90	7.54	0.01
Model C	8	-321.77	660.70	1.33	0.20
Model D	**8**	**-321.10**	**659.40**	**0.00**	**0.40**

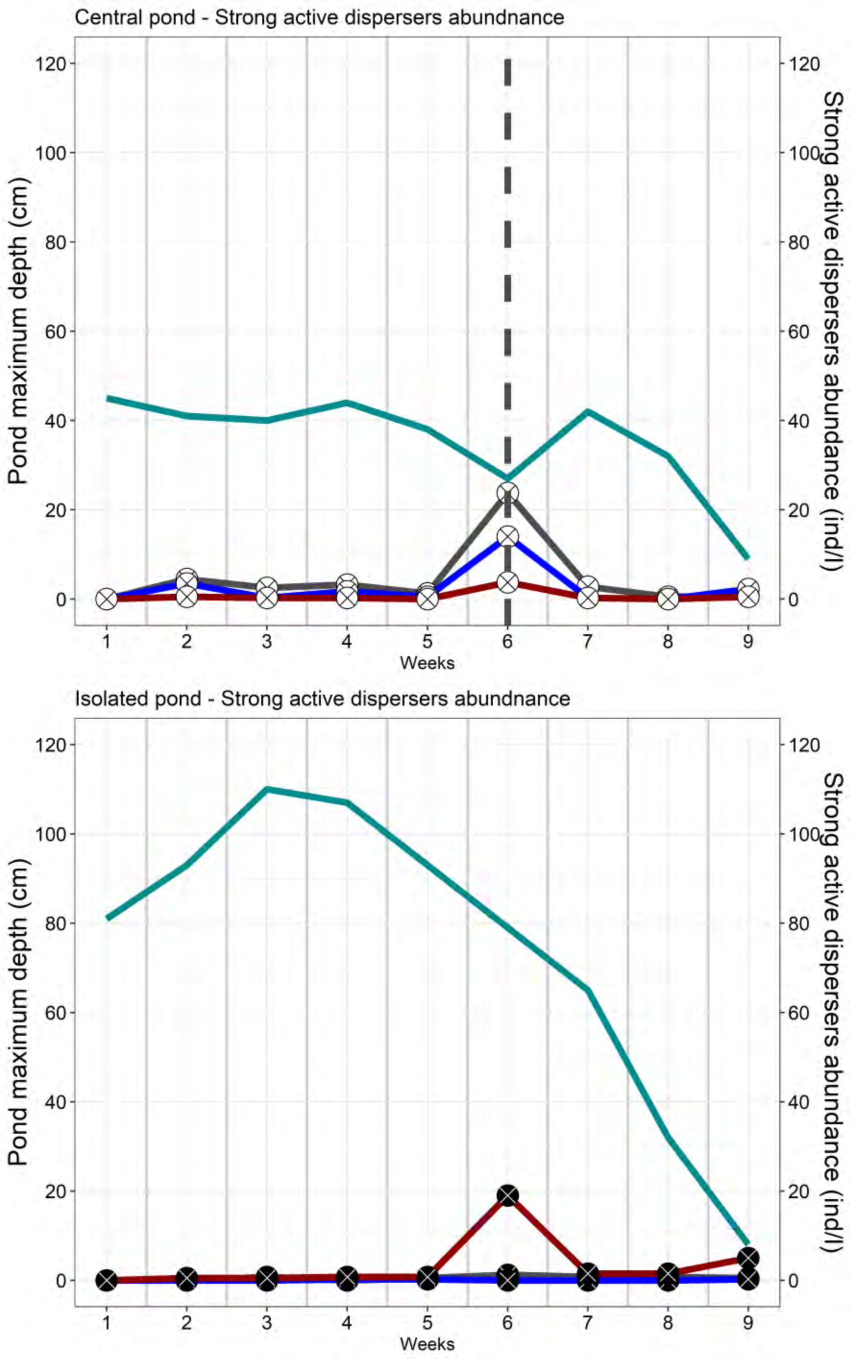
Strong dispersers — Model B3

Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-151.55	334.80	1.89	0.22
Model B1	11	-155.90	336.10	3.14	0.12
Model B2	11	-157.44	339.10	6.22	0.03
Model B3	**11**	**-154.33**	**332.90**	**0.00**	**0.57**
Model B4	11	-159.96	344.20	11.27	0.00
Model C	8	-161.37	340.00	7.02	0.02
Model D	8	-160.49	338.20	5.25	0.04

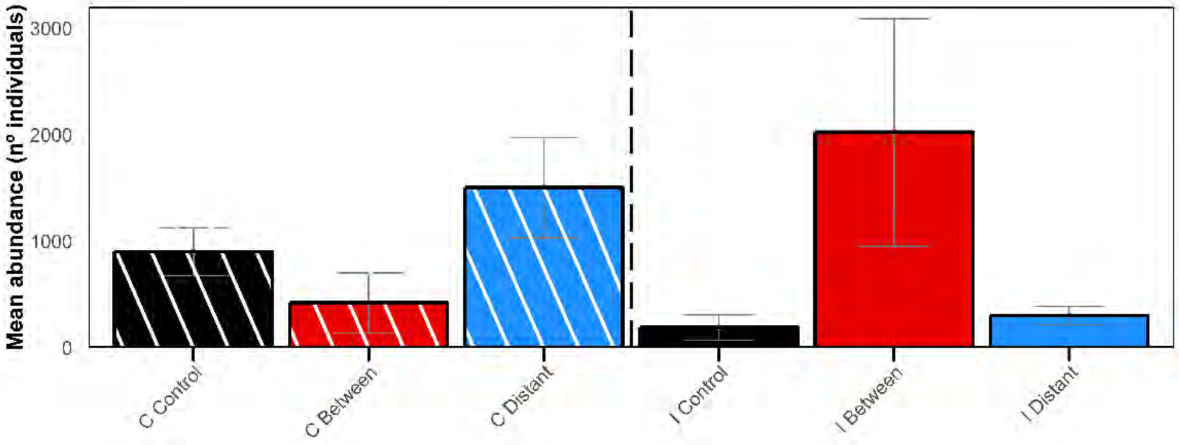
Accumulated richness — Model D

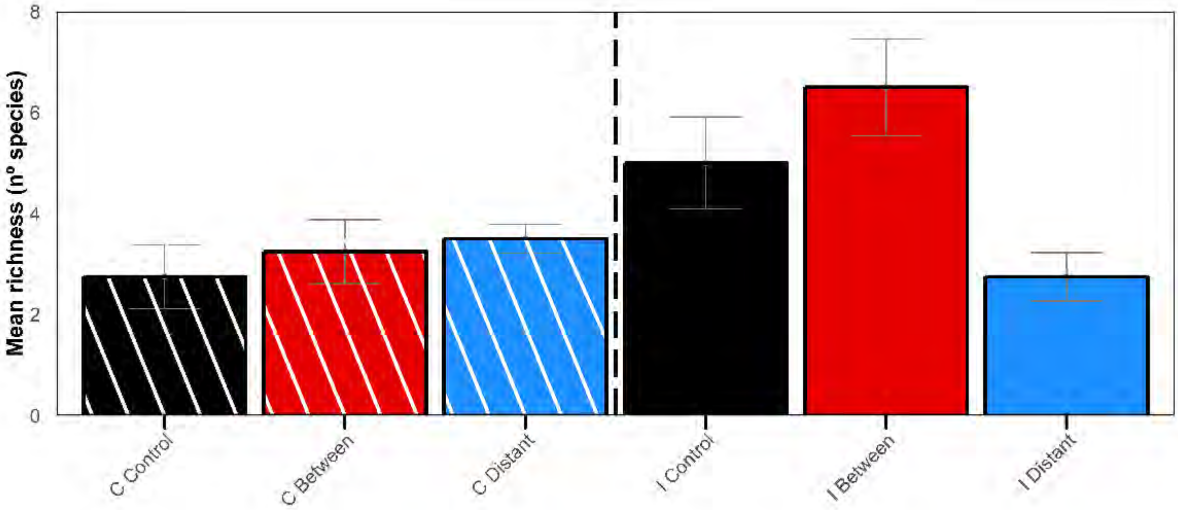
Model	*df*	*logLik*	*AICc*	*delta*	*weight*
Model A	14	-183.18	398.10	3.76	0.05
Model B1	11	-187.01	398.30	3.98	0.04
Model B2	11	-185.46	395.20	0.88	0.21
Model B3	11	-186.78	397.80	3.50	0.06
Model B4	11	-186.14	396.60	2.24	0.11
Model C	8	-188.99	395.20	0.87	0.21
Model D	**8**	**-188.56**	**394.30**	**0.00**	**0.33**

Supplementary S*6.8*: Mean abundance values of strong active dispersers for each mesocosm set type along time (all nine surveyed weeks) and pond maximum depth (cyan solid line) in the central location (top panel, white dots and black crosses) and isolated location (bottom panel, black dots and white crosses). Control sets in grey solid line, distant sets in blue solid line and between sets in red solid line. Grey dashed line indicates the central pond level fall around the sixth week that represented an increase in strong active dispersers movement.



Supplementary S6.9: *Mean abundance and mean richness values for each mesocosm set type in the post-drought mesocosms survey (one week after ponds complete drying). Central locations are indicated in dashed contour grey lines and isolated locations without any lines. Control set in black, between sets in red and distant sets in blue.*



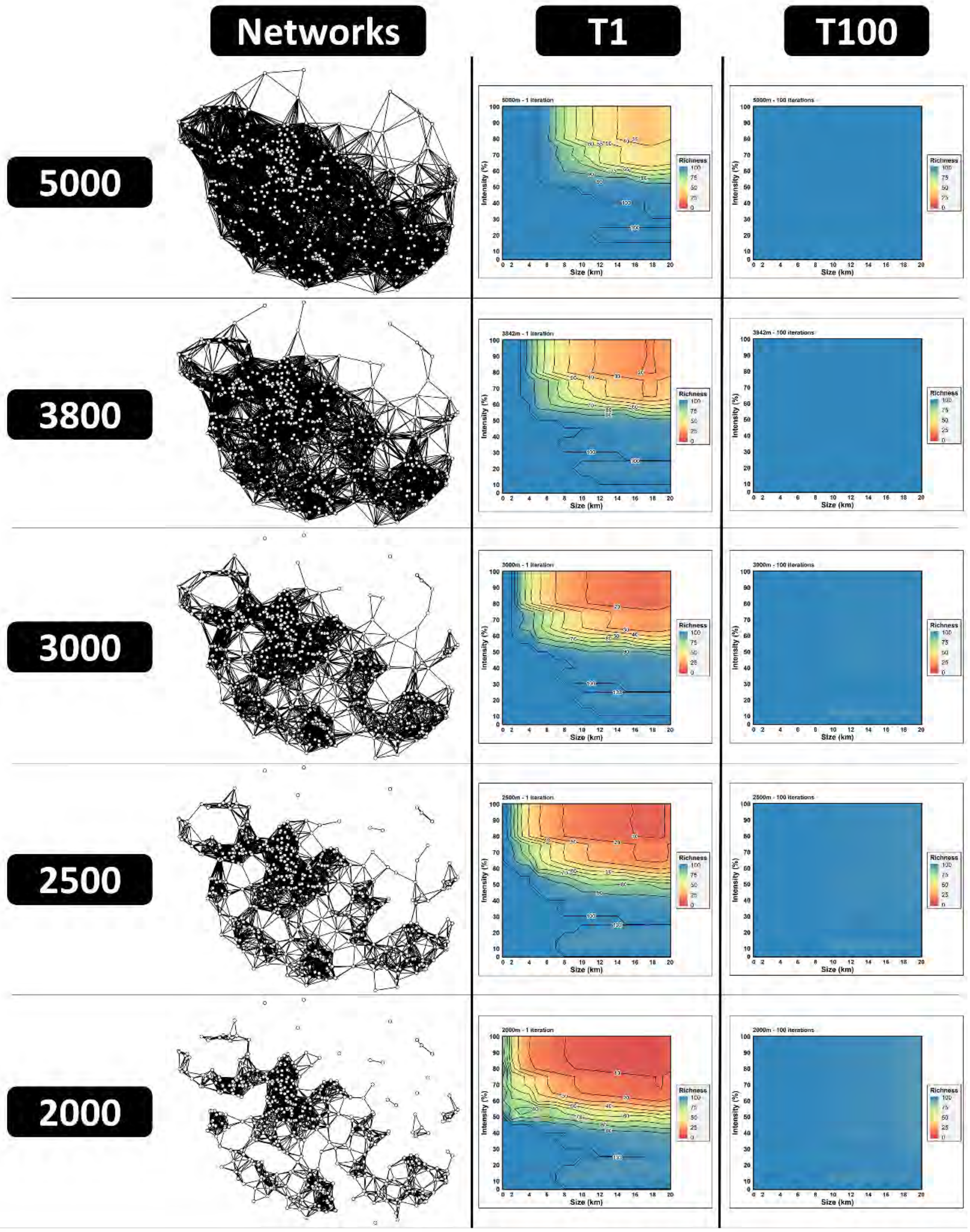


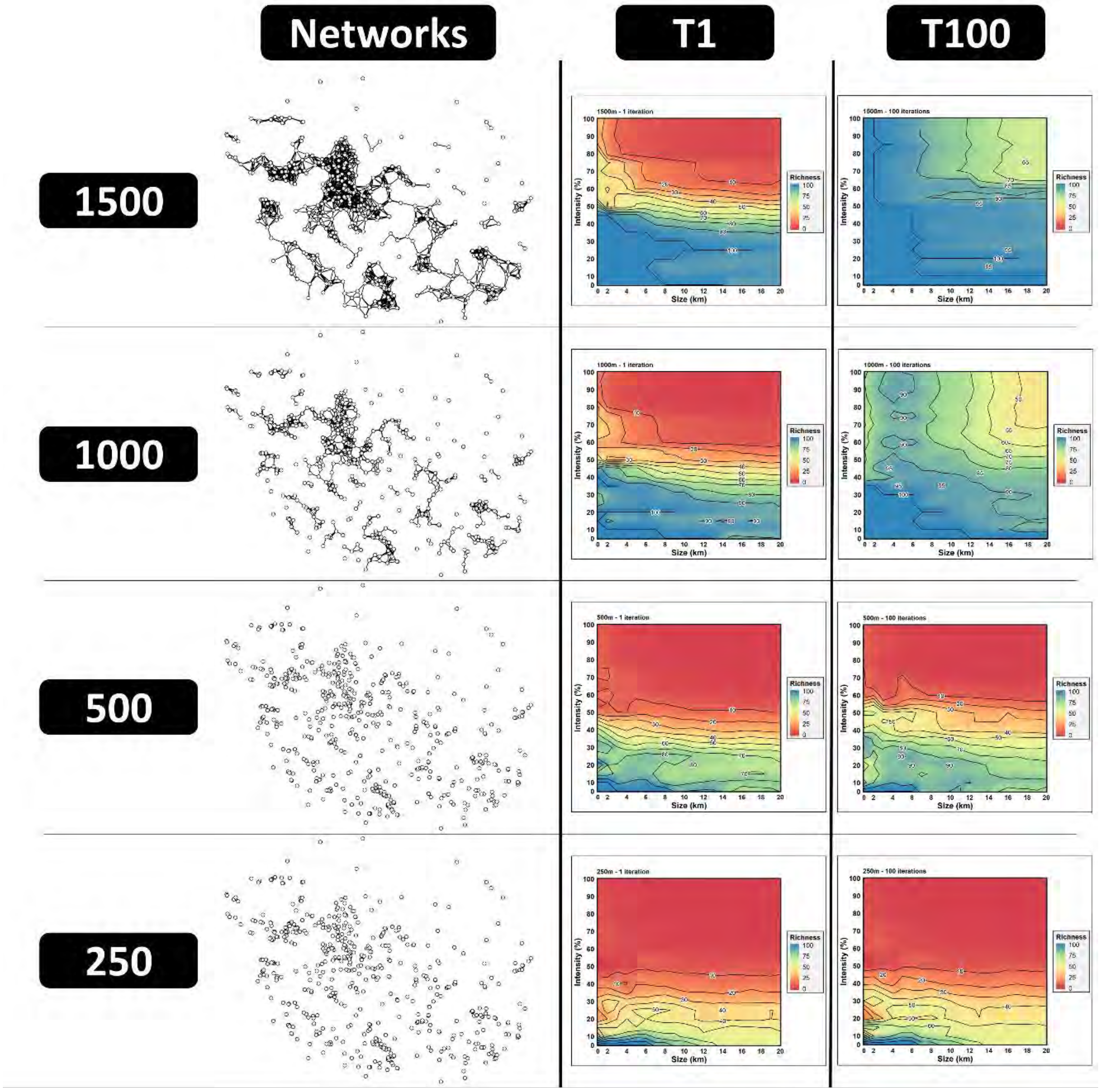
SUPPLEMENTARY MATERIAL

S7

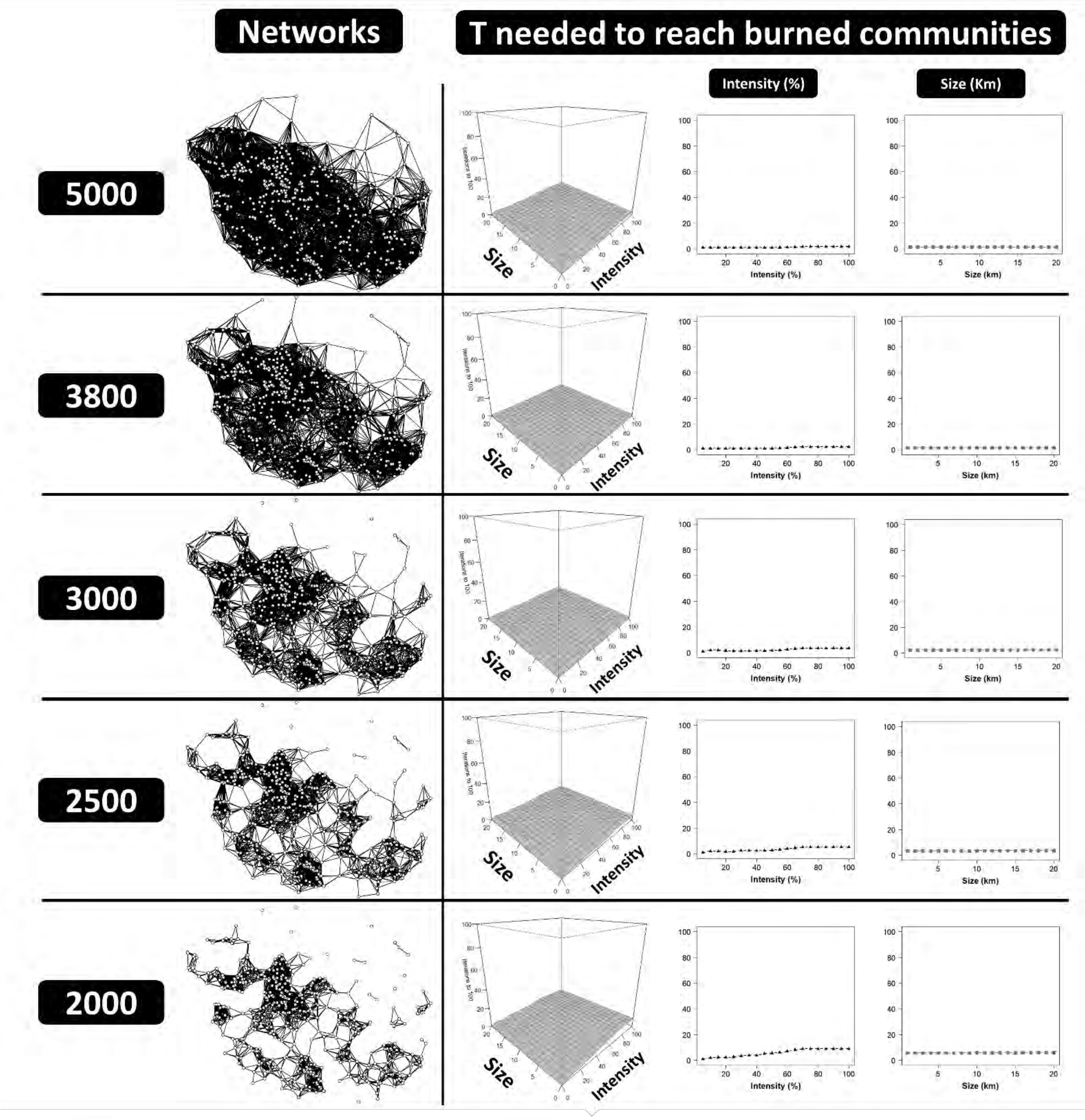
CHAPTER IV SUPPLEMENTARY MATERIAL

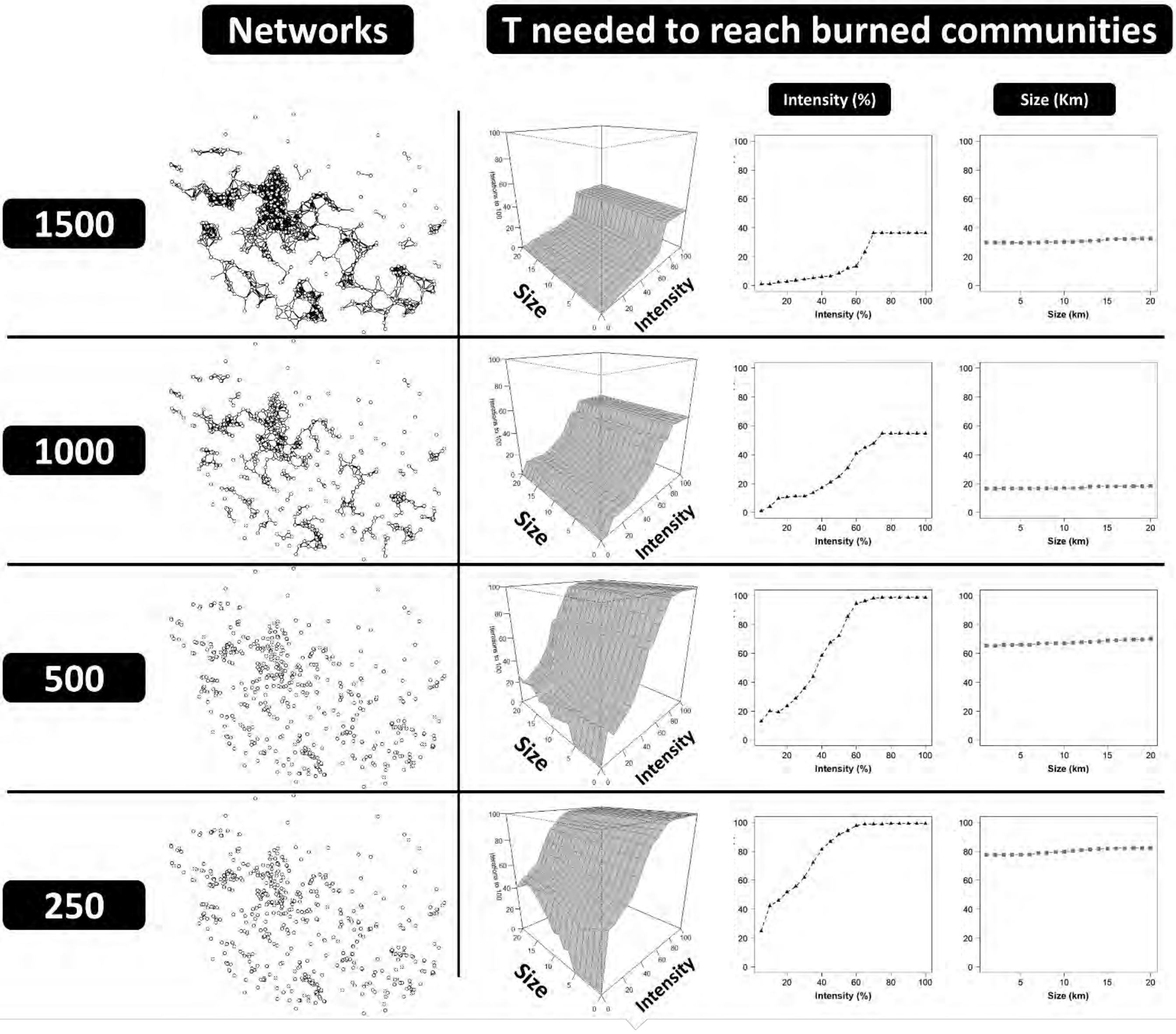
Supplementary S7.1: *Resilience of the communities to simulated wildfire at all levels of linkage distance (5000, 3842, 3000, 2500, 2000, 1500, 1000, 500 and 250 meters). Resilience is the average number of species available from the pool to community assembly after recolonization in all burned water bodies. T1 indicates communities analyzed after 1 iteration and T100 indicates communities analyzed after 100 iterations after disturbance. Size is the extension covered by the simulated wildfire, and intensity is the proportion of the burned water bodies within this extension.*




Networks
T1
T100
1500
1000
500
250
1500m - 1 iteration
1500m - 100 iterations
1000m - 1 iteration
1000m - 100 iterations
500m - 1 iteration
500m - 100 iterations
250m - 1 iteration
250m - 100 iterations
Intensity (%)
Size (km)
Richness
100
75
50
25
0

Supplementary S7.2: *Temporal trend of the communities to simulated wildfire at all levels of linkage distance (5000, 3842, 3000, 2500, 2000, 1500, 100, 500 and 250 meters). T needed to reach burned communities is the number of iterations required to connect the burned community with a community containing individuals (i.e., unburned or recolonized). Size is the extension covered by the simulated wildfire, and intensity is the proportion of the burned water bodies within this area. The strong nonlinear trend is appreciated specially along the intensity 2D plots and for lower landscape perceptions.*




Networks
T needed to reach burned communities
Intensity (%)
Size (Km)
1500
1000
500
250
Size
Intensity
Iterations to 100
Intensity (%)
Size (km)
0
20
40
60
80
100
5
10
15
20

SUPPLEMENTARY MATERIAL

S8

FAUNAL CHECKLISTS

SUPPLEMENTARY MATERIAL

S8

FAUNAL CHECKLISTS

Supplementary material S8.1: The Albera ponds species checklist of the March and April 2012 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	*Dalyellia viridis*		X	X			X	X	X		X
	Gieysztoria beltrani	X		X	X	X	X		X		X
	Neorhabdocoela indet. (U)				X	X					
	Phaenocora sp.						X	X			
	Tetracelis marmorosa	X		X	X	X					
	Dalyellidae indet. (U)					X		X	X		
	Typhloplanidae indet. (U)					X		X			X
Cnidaria	*Hydra* sp.		X	X							X
Annelida Hirudinea	*Helobdella stagnalis*									X	
Annelida Oligochaeta	Enchytraeidae indet. (U)		X		X	X	X	X	X	X	
	Lumbriculidae indet. (U)	X	X	X	X			X		X	X
	Naidinae indet. (U)	X		X	X		X			X	X
	Tubificinae indet. (U)	X	X	X	X		X	X	X	X	X
Gastropoda Lymnaeidae	*Galba truncatula*							X	X		
	Stagnicola palustris								X	X	
Gastropoda Physidae	*Physa acuta*		X	X					X	X	
Gastropoda Planorbidae	*Ferrissia* sp.		X	X							
	Gyraulus crista	X	X	X		X			X	X	X
	Gyraulus sp.			X		X	X	X	X	X	X
	Hippeutis complanatus		X	X						X	

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Planorbidae indet. (U)		X	X		X	X	X	X	X	X
Isopoda Asellidae	*Proasellus coxalis*		X			X			X		
Hydrachnidia	*Piona* sp.		X	X					X	X	X
	Tiphys ornatus		X								
	Tiphys sp.		X						X	X	
	Hydrachnidia indet. (U)		X	X				X		X	X
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*	X	X	X			X			X	X
	Cloeon sp. (U)		X							X	
Odonata Aeshnidae	*Aeshna affinis*				X		X				
	Aeshna sp. (U)	X	X		X	X			X		
	Anax imperator		X								
	Aeshnidae indet. (U)	X		X							
Odonata Coenagrionidae	*Coenagrion* sp. (U)		X								
	Enallagma cyathigerum									X	
	Erythromma lindenii									X	
	Ischnura sp. (U)		X							X	
Odonata Lestidae	*Lestes barbarus*					X					
	Lestes sp. (U)						X	X	X		
Odonata Libellulidae	*Crocothemis erythraea*									X	
	Sympetrum sp. (U)								X		
	Libellulidae indet. (U)					X	X				X
Hemiptera Corixidae	*Corixa panzeri*									X	
	Corixa punctata	X					X				
	Hesperocorixa moesta								X	X	

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Sigara dorsalis							X			
	Sigara lateralis						X		X		
	Corixidae indet. (U)		X	X	X	X	X	X	X	X	X
Hemiptera Gerridae	*Gerris argentatus*		X	X		X					
	Gerris gibbifer								X		
Hemiptera Nepidae	*Ranatra linearis*							X			
Hemiptera Notonectidae	*Notonecta meridionalis*						X				
	Notonecta sp. (L)				X		X		X		
	Notonectidae indet. (U)					X					
Hemiptera Pleidae	*Plea minutissima*	X	X	X		X	X			X	
Hemiptera Saldidae	*Saldula opacula*					X					
	Saldula sp.2	X									
Coleoptera Dryopidae	*Dryops algiricus*								X		
	Dryops striatellus							X			
Coleoptera Dytiscidae	*Agabus nebulosus*						X	X			
	Agabus sp. (L)	X	X	X	X	X	X	X	X	X	X
	Bidessus goudoti				X			X		X	X
	Colymbetes sp. (L)				X						X
	Cybister lateralimarginalis	X					X				
	Dytiscus circumflexus				X						
	Dytiscus sp. (L)		X		X		X	X			X
	Graptodytes bilineatus								X		X
	Graptodytes flavipes	X			X	X	X	X	X	X	X
	Graptodytes sp. (L)				X	X	X	X	X		X
	Hydroglyphus geminus					X					
	Hydroporus sp. (L)							X			
	Hydroporus vagepictus							X			
	Hygrotus impressopunctatus				X						

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Hygrotus sp. (L)			X	X				X		
	Ilybius sp. (L)					X	X	X			X
	Laccophilus hyalinus							X			
Liopterus haemorrhoidalis					X			X			
	Porhydrus sp. (L)	X								X	X
	Rhantus sp. (L)	X	X		X		X	X	X	X	X
Coleoptera Gyrinidae	*Gyrinus dejeani*			X	X	X					
	Gyrinus sp. (L)	X									
Coleoptera Haliplidae	*Haliplus guttatus*	X					X		X	X	
	Haliplus lineaticollis						X				
	Haliplus sp. (L)		X							X	
Coleoptera Helophoridae	*Helophorus alternans*					X					
	Helophorus asturiensis							X			
Coleoptera Hydrochidae	*Hydrochus angustatus*			X				X	X		
Coleoptera Hydrophilidae	*Anacaena lutescens*							X			
	Berosus signaticollis	X		X	X	X	X	X	X		
	Enochrus nigritus	X	X	X					X		
	Enochrus quadripunctatus	X		X	X	X		X	X		
	Helochares lividus		X			X			X		
	Hydrobius fuscipes	X		X		X					
	Hydrobius sp. (L)				X	X			X		
	Hydrochara caraboides								X		
	Laccobius sp. (L)								X		
	Limnoxenus niger				X						
	Limnoxenus sp. (L)				X						
Coleoptera Hygrobiidae	*Hygrobia hermanii*					X				X	
Coleoptera Noteridae	*Noterus clavicornis*		X	X	X					X	
	Noterus laevis	X	X	X							
Coleoptera Scirtidae	*Cyphon* sp.		X								

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Trichoptera Limnephilidae	*Limnephilus* sp.1		X								
	Limnephilus sp.2						X		X		
Trichoptera Polycentropodidae	*Holocentropus stagnalis*			X					X		
Diptera Ceratopogonidae	*Alluaudomyia* sp.									X	
	Bezzia sp.									X	
	Culicoides sp.								X	X	
	Ceratopogonidae indet. (U)					X		X			
Diptera Chaoboridae	*Chaoborus flavicans*		X							X	X
Diptera Chironomidae Chironominae	*Chironomus aprilinus*						X			X	
	Chironomus prasinus									X	
	Chironomus riparius										X
	Chironomus sp. (L)	X	X	X	X	X		X	X	X	X
	Dicrotendipes sp. (L)		X								
	Endochironomus sp. (L)									X	
	Glyptotendipes sp. (L)		X								
	Kiefferulus tendipediformis		X					X		X	X
	Micropsectra lindrothi				X						
	Micropsectra sp. (L)				X	X	X	X		X	
	Microtendipes sp. (L)				X		X	X			
	Parachironomus gr. *arcuatus*						X		X	X	
	Paratanytarsus grimmii	X				X					X
	Paratanytarsus sp. (L)	X	X			X	X	X	X	X	X
	Polypedilum sp. (L)								X	X	
	Zavreliella sp. (L)									X	
	Chironominae indet. (U)	X			X	X					
Diptera Chironomidae Orthocladinae	*Corynoneura* sp. (L)	X			X	X	X		X		X
	Cricotopus sp. (L)			X	X	X	X			X	

March/April 2012 survey campaign
— before the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Cricotopus sylvestris									X	
	Limnophyes sp. (L)	X	X					X	X		
	Psectrocladius (A.) sp. (L)					X			X		X
	Psectrocladius (P.) limbatellus					X	X	X	X		X
	Psectrocladius (P.) sordidellus					X				X	
	Psectrocladius (P.) sp. (L)	X		X	X	X	X	X	X	X	X
Diptera Chironomidae Tanypodinae	*Ablabesmyia* sp. (L)						X	X		X	
	Macropelopia nebulosa						X				
	Macropelopia sp. (L)					X					
	Procladius choreus		X			X	X				
	Procladius sp. (L)		X		X		X	X		X	
	Psectrotanypus sp. (L)		X							X	
	Psectrotanypus varius		X							X	
	Xenopelopia falcigera	X	X	X					X		X
	Xenopelopia sp. (L)	X	X	X			X		X	X	X
	Zavrelimyia barbatipes						X		X		
	Zavrelimyia sp. (L)	X					X				
Diptera Culicidae	*Culiseta litorea*		X					X	X		X
Diptera Dixidae	*Dixella autumnalis*		X								
Diptera Ephydridae	*Hydrellia* sp.				X	X				X	
Amphibia Anura	*Epidalea calamita*							X			
	Pelodytes punctatus						X				
Amphibia Caudata	Salamandridae indet. (U)				X	X					X

Supplementary material S8.2: The Albera ponds species checklist of the December 2012 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

December 2012 survey campaign
— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	*Dalyellia viridis*										X
	Gieysztoria beltrani	X			X	X	X				
	Gieysztoria diadema		X	X					X		X
	Gieyztoria sp1 (U)						X				
	Phaenocora sp. (U)							X			
	Rhabdocoela indet. (U) 3						X				
	Rhabdocoela indet. (U) 1	X		X	X	X	X	X	X		X
	Rhabdocoela indet. (U) 11	X		X	X	X	X	X	X		X
Cnidaria	*Hydra sp.* (U)		X	X							
Annelida Hirudinea	*Helobdella stagnalis*									X	
Annelida Oligochaeta	*Dero* sp. (U)				X				X		
	Enchytraeidae indet. (U)	X			X	X					X
	Lumbricidae indet. (U)	X		X		X		X			
	Lumbriculidae indet. (U)	X	X	X	X	X	X				X
	Naididae indet. (U)	X	X	X	X	X		X	X	X	X
	Pristina/Pristinella sp. (U)										X
	Stylaria lacustris	X								X	X
	Tubificinae indet. (U) with setae	X	X	X		X					X
	Tubificinae indet. (U) without setae	X	X		X	X				X	X
Gastropoda	*Ferrissia* sp.		X								

December 2012 survey campaign

— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Gastropoda Lymnaeidae	*Galba truncatula*							X			
Gastropoda Physidae	*Physa acuta*			X					X	X	
Gastropoda Planorbidae	*Gyraulus crista*	X	X	X		X		X		X	X
	Gyraulus sp.			X		X	X				X
	Hippeutis complanatus		X	X		X	X				X
	Planorbidae indet. (U)					X	X		X		X
Branchiopoda Anostraca	*Chirocephalus diaphanus*					X					X
	Linderiella sp.								X		
	Tanymastix stagnalis							X			
Isopoda Asellidae	*Proasellus coxalis*		X						X		X
Hydrachnidia	*Piona* sp.1		X	X					X	X	X
	Piona sp.2									X	
	Tiphys ornatus									X	
	Pionidae indet. (U)		X							X	X
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*	X	X							X	X
	Cloeon schoenemundi									X	
	Baetidae indet. (U)	X	X	X			X	X	X	X	X
Odonata	Anisoptera indet. (U)				X						X
	Zygoptera indet. (U)					X			X		X
Odonata Aeshnidae	*Anax imperator*									X	
	Aeshnidae indet. (U)			X							
Odonata Coenagrionidae	*Coenagrion scitulum*									X	
Odonata Lestidae	Lestidae indet. (U)								X		X

December 2012 survey campaign
— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Odonata Libellulidae	Libellulidae indet. (U)			X		X				X	X
Hemiptera Corixidae	*Corixa panzeri*	X		X	X		X			X	X
	Corixa punctata	X	X	X	X		X		X		X
	Hesperocorixa moesta	X		X						X	X
	Sigara dorsalis							X		X	
	Sigara lateralis				X					X	X
	Sigara limitata					X				X	
	Sigara stagnalis				X						
	Corixidae indet. (U)									X	
Hemiptera Notonectidae	*Notonecta meridionalis*							X			
Hemiptera Pleidae	*Plea minutissima*	X	X							X	X
Coleoptera Dryopidae	*Dryops algiricus*						X				
Coleoptera Dytiscidae	*Agabus bipustulatus*								X		
	Agabus nebulosus	X									
	Agabus sp. (L)	X			X	X	X	X	X	X	X
	Colymbetes sp. (L)				X	X		X	X		X
	Graptodytes bilineatus				X						
	Graptodytes flavipes	X			X		X	X	X		X
	Graptodytes sp. (L)	X									X
	Gyraulus sp. (L)										X
	Hydroporus sp. (L)	X		X							
	Hygrotus impressopunctatus		X		X						
	Ilybius sp. (L)	X			X	X				X	X
	Laccophilus sp.								X		
	Laccohilus variegatus									X	
	Laccophilus minutus									X	
	Laccophilus variegatus		X								

December 2012 survey campaign
— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Liopterus sp. (L)								X		
	Rhantus sp. (L)	X			X		X	X		X	X
Coleoptera Gyrinidae	*Gyrinus caspius*						X				
Coleoptera Haliplidae	*Haliplus guttatus*	X			X		X				
Coleoptera Helophoridae	*Helophorus asturiensis*							X			
	Helophorus griseus							X			
Coleoptera Hydrochidae	*Hydrochus angustatus*									X	
	Hydrochus nitidicollis		X								
Coleoptera Hydrophilidae	*Berosus signaticollis*	X	X		X		X	X			X
	Enochrus nigritus		X								
	Enochrus quadripunctatus		X								
	Helochares lividus		X							X	
	Hydrobius fuscipes								X		
	Hydrobius sp. (L)							X	X		X
	Limnoxenus sp. (L)								X		
Coleoptera Hygrobiidae	*Hygrobia hermanii*				X					X	X
Coleoptera Noteridae	*Noterus clavicornis*		X								
	Noterus laevis		X								
Trichoptera Hydroptilidae	*Agraylea sexmaculata*									X	
Trichoptera Polycentropodidae	*Holocentropus stagnalis*			X					X	X	X
Diptera Ceratopogonidae	*Ceratopogon* sp.		X							X	
	Culicoides sp.								X		X
	Palpomyna sp.								X		
	Stilobezzia sp.									X	
	Phaenobezzia/Bezzia/Palpomyna sp.		X	X					X		
Diptera Chaoboridae	*Chaoborus flavicans*									X	
Diptera Chironomidae	*Chironomus* sp. (L)	X	X		X	X		X	X	X	X

December 2012 survey campaign
— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Chironominae	*Endochironomus* sp. (L)									X	
	Glyptotendipes sp. (L)									X	
	Kiefferulus tendipediformis	X							X		
	Micropsectra sp. (L)	X									
	Microtendipes sp. (L)	X							X		
	Paratanytarsus grimmii	X								X	
	Paratanytarsus sp. (L)	X	X		X		X	X	X	X	X
	Polypedilum sp. (L)	X	X						X		
	Zavreliella sp. (L)									X	
	Chironominae indet. (U)	X	X	X	X	X	X	X		X	X
Diptera Chironomidae Orthocladinae	*Corynoneura* sp. (L)	X	X	X				X	X		X
	Cricotopus sp. (L)	X	X	X	X			X	X	X	X
	Cricotopus sylvestris	X								X	
	Limnophyes sp. (L)	X	X	X	X	X	X			X	X
	Psectrocladius (A.) obvious									X	
	Psectrocladius (A.) sp. (L)						X	X		X	X
	Psectrocladius (P.) limbatellus								X	X	X
	Psectrocladius (P.) sp. (L)	X	X		X	X	X	X	X	X	X
	Pseudosmittia cf. simplex									X	
	Pseudosmittia sp. (L)		X								
	Stygocladius sp. (L)								X		
	Orthocladinae indet. (U)	X	X		X		X	X	X		
Diptera Chironomidae Tanypodinae	*Macropelopia* sp. (L)	X								X	
	Procladius choreus									X	
	Procladius sp. (L)	X								X	X
	Psectrotanypus sp. (L)		X								
	Tanypus sp. (L)									X	
	Xenopelopia falcigera										X
	Xenopelopia sp. (L)	X	X	X							X

December 2012 survey campaign
— first survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Zavrelimyia sp. (L)			X					X	X	
	Tanypodinae indet. (U)				X			X	X		X
Diptera Culicidae	*Culex theileri*										X
	Culiseta litorea		X								
	Culiseta longiareolata								X		
Diptera Dolichocephalidae	Dolichocephalidae indet. (U)		X								
Diptera Ephydridae	*Hydrellia* sp.	X								X	
Diptera Tabanidae	*Hybomitra* sp.		X	X							
	Tabanus sp.		X							X	X
Amphibia Anura	*Hyla meridionalis*					X					
	Pelodytes punctatus	X						X			
	Anura indet. (U)						X				

Supplementary material S8.3: The Albera ponds species checklist of the February 2013 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

February 2013 survey campaign
— second survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	*Dalyellia viridis*		X	X	-		X	-	X		X
	Gieysztoria beltrani	X			-	X	X	-			X
	Gieysztoria diadema			X	-		X	-			
	Rhabdocoela indet. (U) 1	X		X	-	X	X	-	X	X	X
	Rhabdocoela indet. (U) 11	X			-	X	X	-	X		X
	Bothromesostoma personatum	X			-			-			
Cnidaria	*Hydra* sp.		X	X	-			-			X
Annelida Oligochaeta	Enchytraeidae indet. (U)				-			-		X	
	Lumbricidae indet. (U)		X	X	-	X	X	-			
	Lumbriculidae indet. (U)		X	X	-			-		X	X
	Naididae indet. (U)	X	X	X	-		X	-		X	X
	Stylaria lacustris				-			-		X	
	Tubificinae indet. (U) with setae				-			-			X
	Tubificinae indet. (U) without setae	X			-	X	X	-	X	X	
Gastropoda	Gastropoda indet. (U)				-		X	-			
Gastropoda Lymnaeidae	*Stagnicola palustris*				-			-		X	
Gastropoda Physidae	*Physa acuta*			X	-			-		X	
	Physidae indet. (U)			X	-			-			
Gastropoda Planorbidae	*Ferrissia* sp.			X	-			-			
	Gyraulus crista		X	X	-	X		-		X	X

February 2013 survey campaign
— second survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Gyraulus sp.		X	X	-	X	X	-	X		X
	Hippeutis complanatus			X	-	X		-			X
	Planorbidae indet. (U)				-			-			X
Branchiopoda Anostraca	*Chirocephalus diaphanus*				-	X		-			
Isopoda Asellidae	Asellidae indet. (U)				-	X		-	X		X
	Proasellus coxalis		X		-	X		-	X		
	Proasellus sp. (U)				-			-			X
Hydrachnidia	*Piona* sp.1		X	X	-			-	X	X	X
	Pionidae indet. (U)		X		-			-			
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*	X	X		-			-	X	X	X
	Cloeon schoenemundi				-			-		X	
	Baetidae indet. *(U)*	X			-			-	X		
Odonata	Anisoptera indet. (U)				-	X		-			X
	Zygoptera indet. (U)				-			-			X
Odonata Aeshnidae	*Aeshna mixta*				-			-		X	
	Aeshnidae indet. (U)				-	X		-			X
Odonata Coenagrionidae	*Coenagrion scitulum*				-			-		X	
Odonata Lestidae	*Lestes* sp. (U)				-	X		-			
	Lestidae indet. (U)				-			-			X
Odonata Libellulidae	*Crocothemis erythraea*				-			-		X	
	Libellulidae indet. (U)				-			-	X		X
Hemiptera Corixidae	*Corixa panzeri*			X	-			-			
	Corixa punctata	X	X		-	X	X	-			

February 2013 survey campaign
— second survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Hesperocorixa moesta				-	X		-			X
	Sigara dorsalis				-			-		X	
	Sigara lateralis	X			-			-	X	X	
	Corixidae indet. (U)	X			-	X	X	-		X	
Hemiptera Notonectidae	*Notonecta meridionalis*				-	X		-			
	Notonectidae indet. (U)				-			-	X		
Hemiptera Pleidae	*Plea minutissima*		X		-		X	-		X	X
Coleoptera Dytiscidae	*Agabus nebulosus*				-	X		-			
	Agabus sp. (L)	X	X	X	-	X	X	-	X	X	X
	Colymbetes sp. (L)				-			-	X	X	X
	Dytiscus sp. (L)				-		X	-	X		
	Graptodytes bilineatus				-	X		-			
	Graptodytes flavipes				-	X	X	-	X		X
	Graptodytes sp. (L)	X			-			-			X
	Hydroporus sp. (L)		X	X	-			-			
	Hydroporus vagepictus				-	X		-			
	Hygrotus sp. (L)				-	X		-			
	Ilybius sp. (L)			X	-	X	X	-	X	X	X
	Rhantus sp. (L)		X		-		X	-	X	X	X
Coleoptera Haliplidae	*Haliplus guttatus*				-	X		-			
Coleoptera Helophoridae	*Helophorus asturiensis*				-		X	-			
Coleoptera Hydrophilidae	*Berosus signaticollis*	X			-	X	X	-			X
Coleoptera Hygrobiidae	*Hygrobia hermanii*				-			-		X	
Coleoptera Noteridae	*Noterus clavicornis*		X		-			-			
Trichoptera Hydroptilidae	*Agraylea sexmaculata*				-			-		X	
Trichoptera Polycentropodidae	*Holocentropus stagnalis*			X	-		X	-	X		X

February 2013 survey campaign
— second survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Diptera Ceratopogonidae	*Culicoides* sp.				-			-	X		
	Phaenobezzia/Bezzia/Palpomyna sp.			X	-			-	X		
Diptera Chaoboridae	*Chaoborus flavicans*				-			-		X	
Diptera Chironomidae Chironominae	*Chironomus* sp. (L)	X	X	X	-			-			
	Endochironomus sp. (L)				-			-			X
	Kiefferulus tendipediformis				-			-			X
	Micropsectra sp. (L)	X		X	-			-		X	
	Microtendipes sp. (L)				-			-	X		
	Parachironomus gr. *arcuatus*				-			-			X
	Paratanytarsus sp. (L)	X		X	-		X	-	X	X	X
	Polypedilum sp. (L)				-			-			X
	Zavreliella sp. (L)			X	-			-		X	
	Chironominae indet. (U)	X	X	X	-			-	X	X	
Diptera Chironomidae Orthocladinae	*Corynoneura* sp. (L)		X	X	-		X	-			
	Cricotopus sp. (L)	X	X		-		X	-	X	X	X
	Limnophyes sp. (L)			X	-		X	-		X	
	Psectrocladius (A.) sp. (L)				-		X	-	X		
	Psectrocladius (P.) limbatellus			X	-			-		X	
	Psectrocladius (P.) sp. (L)	X	X	X	-	X	X	-	X	X	X
	Orthocladinae indet. (U)	X	X	X	-		X	-	X		X
Diptera Chironomidae Tanypodinae	*Macropelopia nebulosa*				-			-		X	
	Procladius sp. (L)	X			-		X	-	X	X	X
	Xenopelopia sp. (L)		X	X	-			-	X		X
Diptera Tabanidae	*Tabanus* sp. (L)			X	-			-			
Amphibia Anura	*Discoglossus pictus*				-			-	X		
	Hyla meridionalis				-	X		-			
	Pelodytes punctatus				-	X	X	-	X		

Supplementary material S8.4: The Albera ponds species checklist of the April 2013 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	*Gieysztoria beltrani*	X		X		X	X				
	Gieysztoria diadema						X				
	Gieyztoria sp.1				X						
	Mesostoma ehrenbergii			X							
	Rhabdocoela indet. (U) 1	X		X			X	X	X	X	X
	Rhabdocoela indet. (U) 11	X					X				
	Rhabdocoela indet. (U) 7						X				
	Typhloplanidae indet. (U)		X	X							
	Bothromesostoma personatum	X	X	X	X	X		X			
Cnidaria	*Hydra* sp.		X	X	X						X
Annelida Hirudinea	*Helobdella stagnalis*									X	
Annelida Oligochaeta	*Chaetogaster* sp.						X				
	Enchytraeidae indet. (U)	X	X		X			X		X	X
	Lumbricidae indet. (U)		X	X		X			X		X
	Lumbriculidae indet. (U)		X	X	X	X	X				X
	Naidinae indet. (U)	X	X	X	X		X		X		X
	Stylaria lacustris									X	
	Tubificinae indet. (U) without setae	X	X	X	X	X	X		X	X	X
	Tubificinae indet. (U) with setae						X				
Gastropoda Lymnaeidae	*Galba truncatula*							X			

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Stagnicola palustris								X		
Gastropoda Physidae	*Physa acuta*		X	X					X	X	
Gastropoda Planorbidae	*Ferrissia* sp.		X								
	Gyraulus crista	X	X	X		X				X	
	Gyraulus sp.		X	X	X	X	X	X	X		X
	Hippeutis complanatus		X	X	X						X
	Planorbidae indet. (U)			X		X		X			X
Branchiopoda Anostraca	*Tanymastix stagnalis*							X			
Isopoda Asellidae	*Proasellus coxalis*		X						X		X
	Asellidae indet. (U)								X		X
Hydrachnidia	*Piona* sp.1	X	X	X					X	X	
	Piona sp.2		X							X	
	Pionidae indet. (U)		X	X						X	
	Tiphys ornatus		X								
	Hydrachnidia indet. (U)		X								
Ephemeroptera Baetidae	*Cloeon gr. dipterum*	X	X	X					X	X	X
	Cloeon schoenemundi		X				X	X		X	
	Baetidae indet. (U)					X		X	X	X	
Odonata Aeshnidae	*Aeshna affinis*										X
	Aeshna mixta		X								
	Aeshnidae indet. (U)		X	X	X						
Odonata Coenagrionidae	*Coenagrion scitulum*									X	
	Coenagrionidae indet. (U)										X
Odonata Lestidae	*Lestes barbarus*					X	X		X		X

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Lestidae indet. (U)			X	X						
Odonata Libellulidae	*Crocothemis erythraea*									X	
	Sympetrum meridionale					X					
	Sympetrum striolatum	X							X		
	Libellulidae indet. (U)	X	X	X	X	X	X	X	X	X	X
Hemiptera Corixidae	*Corixa panzeri*								X		
	Corixa punctata					X					
	Hesperocorixa moesta										X
	Paracorixa concina						X				
	Sigara lateralis	X									
	Corixidae indet. (U)	X	X	X	X	X	X	X	X	X	X
Hemiptera Gerridae	*Gerris argentatus*					X				X	
	Gerris thoracicus							X			
	Gerridae indet. (U)					X		X			
Hemiptera Naucoridae	*Ilyocoris cimicoides*			X							
Hemiptera Notonectidae	*Notonecta* sp. (U)	X	X	X		X	X	X	X	X	X
	Notonectidae indet. (U)	X	X	X		X	X			X	
Hemiptera Pleidae	*Plea minutissima*	X	X	X	X	X	X	X	X	X	X
Hemiptera Veliidae	Veliidae indet. (U)			X							
Coleoptera Dryopidae	*Dryops algiricus*						X	X			
Coleoptera Dytiscidae	*Agabus* sp. (L)	X		X	X	X	X	X		X	X
	Bidessus goudoti	X		X	X					X	
	Bidessus sp. (L)			X							X
	Colymbetes sp. (L)	X	X		X						
	Cybister lateralimarginalis										X
	Dytiscus sp.(L)	X			X	X	X	X	X		X
	Graphoderus sp. (L)			X		X					

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Graptodytes bilineatus				X	X		X			
	Graptodytes flavipes	X			X		X	X			
	Graptodytes sp. (L)	X		X		X	X	X	X		
	Hydroporinae indet. (U)		X						X		
	Hydroporus sp.(L)	X	X	X	X	X	X	X			
	Hydrovatus cuspidatus		X								
	Hydrovatus sp.(L)	X	X				X				
	Hygrobia hermanii										X
	Hygrotus sp.(L)				X						
	Hyphydrus aubei						X				
	Hyphydrus sp. (L)	X	X					X	X	X	
	Ilybius sp. (L)	X			X		X				
	Laccophilus minutus							X		X	X
	Laccophilus sp.(L)	X		X			X	X	X		
	Laccophilus variegatus		X	X						X	
	Liopterus haemorrhoidalis		X								
	Liopterus sp. (L)		X	X			X				X
	Rhantus sp. (L)							X	X		
Coleoptera Gyrinidae	*Gyrinus dejeani*					X		X		X	
	Gyrinus sp. (L)	X				X		X	X		
Coleoptera Haliplidae	*Haliplus guttatus*									X	
	Haliplus sp. (L)	X	X				X				
	Laccobius sp. (L)								X		
Coleoptera Hydrophilidae	*Berosus signaticollis*	X		X	X	X	X			X	
	Berosus sp.(L)		X	X	X	X		X	X		X
	Enochrus nigritus	X	X	X							
	Enochrus quadripunctatus				X						
	Enochrus sp. (L)			X	X	X					
	Enochrus testaceus		X							X	

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Helochares lividus		X	X							
	Hydrobius sp. (L)	X	X	X	X	X	X		X	X	X
	Hydrochara sp. (L)										X
	Hydrophilus sp. (L)						X				
	Limnoxenus sp. (L)				X			X			
Coleoptera Hygrobiidae	*Hygrobia hermanii*	X	X	X	X	X	X	X	X	X	
Coleoptera Noteridae	*Noterus clavicornis*		X	X	X					X	
	Noterus laevis		X	X						X	
Trichoptera Polycentropodidae	Polycentropodidae indet. (U)			X					X	X	
Diptera Ceratopogonidae	*Stilobezzia* sp.									X	
	Phaenobezzia/Bezzia/Palpomyna sp.			X					X	X	X
Diptera Chaoboridae	*Chaoborus flavicans*									X	
	Chaoboridae indet. (U)	X					X	X		X	X
Diptera Chironomidae Chironominae	*Chironomus aprilinus*						X				
	Chironomus sp. (L)				X	X	X	X	X	X	X
	Endochironomus sp. (L)									X	
	Kiefferulus tendipediformis									X	X
	Micropsectra sp. (L)				X	X	X	X	X	X	X
	Microtendipes sp. (L)						X		X	X	X
	Parachironomus gr. *arcuatus*									X	
	Paratanytarsus grimmii							X			
	Paratanytarsus sp. (L)	X		X	X		X	X	X	X	X
	Chironominae indet. (U)							X			
Diptera Chironomidae Orthocladinae	*Corynoneura* sp. (L)	X	X	X	X		X	X			X
	Cricotopus sp. (L)	X	X				X	X	X	X	
	Limnophyes sp. (L)							X		X	
	Psectrocladius (A.) obvious		X			X		X	X		

April 2013 survey campaign
— third survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Psectrocladius (A.) sp. (L)	X	X	X		X	X	X		X	X
	Psectrocladius (P.) limbatellus		X		X	X		X	X	X	X
	Psectrocladius (P.) sordidellus		X	X	X	X		X	X		X
	Psectrocladius (P.) sp. (L)	X	X	X	X	X	X	X	X	X	X
	Orthocladinae indet. (U)	X	X			X	X	X	X		X
Diptera Chironomidae Tanypodinae	*Ablabesmyia* sp. (L)			X			X				X
	Macropelopia sp. (L)						X				
	Procladius choreus		X								X
	Procladius sp. (L)					X	X	X	X	X	X
	Psectrotanypus sp. (L)									X	
	Tanypus sp. (L)									X	
	Xenopelopia falcigera		X	X							
	Xenopelopia sp. (L)		X	X						X	
	Zavrelimyia sp. (L)	X	X	X	X		X		X		X
	Tanypodinae indet. (U)				X	X					X
Diptera Culicidae	*Anopheles maculipennis*		X				X				
	Culex pipiens						X				
Diptera Ephydridae	*Hydrellia* sp.	X	X	X	X			X	X	X	X
Diptera Tabanidae	*Hybomitra* sp.		X								
Amphibia Anura	*Bufo calamita*		X	X	X		X	X	X		
	Discoglossus pictus		X	X				X			
	Hyla meridionalis	X	X	X	X	X	X	X	X	X	X
	Pelobates cultripes	X	X	X	X	X	X	X	X	X	X
	Pelodytes punctatus		X	X			X				
	Anura indet. (U)	X	X	X	X	X	X	X	X	X	X
Amphibia Caudata	Urodela indet. (U)	X	X		X	X		X	X	X	X

Supplementary material S8.5: The Albera ponds species checklist of the June 2013 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

June 2013 survey campaign
— fourth survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	*Gieysztoria beltrani*	X			X	X			X		
	Melostoma sp.			X							
	Rhabdocoela indet. (U) 1								X		
Cnidaria	*Hydra* sp.		X	X					X		
Annelida Hirudinea	*Helobdella stagnalis*									X	
Annelida Oligochaeta	*Chaetogaster* sp.		X	X		X			X		X
	Dero sp.				X						
	Lumbricidae indet. (U)			X		X				X	
	Lumbriculidae indet. (U)		X	X		X				X	X
	Naidinae indet. (U)		X	X	X				X	X	X
	Pristina/Pristinella sp.								X		
	Stylaria lacustris								X	X	
	Tubificinae indet. (U) without setae				X						
	Tubificinae indet. (U) with setae				X	X	X	X	X		X
Gastropoda Lymnaeidae	*Galba truncatula*							X	X	X	
	Stagnicola palustris								X		X
Gastropoda Physidae	*Physa acuta*	X	X	X	X				X	X	X
Gastropoda Planorbidae	*Ferrissia* sp.		X							X	X
	Gyraulus crista	X	X	X		X		X			
	Gyraulus sp.		X	X	X	X	X	X	X		X

June 2013 survey campaign
— fourth survey after the wildfire —

Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Hippeutis complanatus		X	X	X	X					X
	Planorbidae indet. (U)		X								
Isopoda Asellidae	*Proasellus coxalis*		X							X	X
	Asellidae indet. (U)		X							X	
Hydrachnidia	*Hydrodroma pilosa*									X	
	Piona sp.1	X	X								
	Piona sp.2	X								X	
	Tiphys ornatus									X	
	Pionidae indet. (U)	X	X	X					X	X	
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*	X	X	X	X	X	X	X		X	X
	Cloeon schoenemundi							X	X	X	
	Baetidae indet. (U)	X			X	X	X	X	X	X	X
Ephemeroptera Leptophlebiidae	*Habrophlebia* cf. *eldae*		X								
Odonata	Aeshnidae indet. (U)					X					
	Anisoptera indet. (U)	X									X
	Zygoptera indet. (U)	X			X		X	X	X		X
Odonata Aeshnidae	*Aeshna mixta*								X		
	Anax imperator									X	
	Aeshnidae indet. (U)	X			X	X			X		X
Odonata Coenagrionidae	*Ischnura* sp. (U)									X	
Odonata Lestidae	*Chalcolestes viridis*									X	
	Lestes barbarus			X	X						X
	Sympecma fusca	X	X	X		X	X	X	X	X	X
	Sympecma sp. (U)				X						
Odonata Libellulidae	*Sympetrum fonscolombii*									X	

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Sympetrum meridionale					X	X	X		X	X
	Sympetrum striolatum	X	X	X					X		
Hemiptera Corixidae	*Corixa affinis*									X	
	Corixa panzeri	X									
	Corixa punctata	X	X	X	X	X	X	X			
	Hesperocorixa linnaei									X	
	Hesperocorixa moesta			X		X		X	X		X
	Sigara dorsalis				X				X		
	Sigara falleni	X									
	Sigara limitata	X									
	Corixidae indet. (U)	X	X	X	X	X	X	X	X	X	X
Hemiptera Gerridae	*Gerris argentatus*						X	X		X	X
	Gerridae indet. (U)	X		X	X	X		X	X		X
Hemiptera Hydrometridae	*Hydrometra stagnorum*			X							X
Hemiptera Naucoridae	*Ilyocoris cimicoides*			X							
	Naucoridae indet. (U)	X	X	X						X	X
Hemiptera Nepidae	*Ranatra linearis*					X			X	X	
Hemiptera Notonectidae	*Anisops* sp. (U)							X			
	Notonecta meridionalis	X			X		X	X			
	Notonecta sp. (U)	X			X	X		X	X	X	X
	Notonectidae indet. (U)		X				X				
Hemiptera Pleidae	*Plea minutissima*	X	X	X	X	X	X	X	X	X	X
Hemiptera Saldidae	*Saldula opacula*					X	X				X
Coleoptera Curculionidae	*Bagous* sp. (L)						X				
Coleoptera Dryopidae	*Dryops algiricus*						X				
	Dryops sp. (L)			X							
Coleoptera Dytiscidae	*Agabus nebulosus*	X									

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Bidessus goudoti	X		X				X		X	X
	Bidessus sp. (L)	X		X						X	X
	Cybister sp. (L)	X	X		X	X	X	X	X	X	X
	Dytiscus sp. (L)	X							X		
	Graphoderus sp. (L)		X	X			X			X	
	Graphoderus sp. (L)	X			X	X		X			X
	Graptodytes flavipes										X
	Graptodytes sp. (L)	X			X	X		X			X
	Hydrobatus sp. (L)	X							X		X
	Hydroporus sp. (L)		X	X			X				
	Hydrovatus sp. (L)						X				
	Hygrotus sp. (L)		X								
	Hyphydrus aubei										X
	Hyphydrus sp. (L)	X	X							X	X
	Laccophilus cf. *minutus*							X			
	Laccophilus minutus					X	X			X	
	Laccophilus sp. (L)	X	X	X	X	X	X	X	X	X	X
	Laccophilus variegatus		X							X	
	Liopterus sp. (L)		X								X
	Noterus clavicornis										X
	Stictonectes lepidus							X			
Coleoptera Gyrinidae	*Gyrinus dejeani*		X	X				X			
	Gyrinus distinctus								X	X	
	Gyrinus sp. (L)									X	
Coleoptera Haliplidae	*Haliplus* sp. (L)	X									
Coleoptera Helophoridae	*Helophorus asturiensis*							X			
	Helophorus minutus						X				
Coleoptera Hydrochidae	*Hydrochus angustatus*							X			
Coleoptera Hydrophilidae	*Anacaena lutescens*	X						X			X

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Berosus signaticollis				X						
	Berosus sp. (L)	X	X	X	X	X	X	X	X	X	X
	Enochrus nigritus	X	X			X					X
	Enochrus quadripunctatus	X			X	X		X			X
	Enochrus sp. (L)	X			X	X				X	X
	Enochrus testaceus		X								
	Helochares sp. (L)									X	
	Helophorus minutus									X	
	Hydrobius fuscipes						X				
	Hydrobius sp. (L)	X	X	X	X	X			X	X	X
	Limnoxenus niger	X									
	Limnoxenus sp. (L)	X								X	X
Coleoptera Hygrobiidae	*Hygrobia hermanii*	X	X	X						X	
Coleoptera Noteridae	*Noterus clavicornis*		X		X	X				X	
	Noterus laevis		X	X						X	X
Trichoptera Polycentropodidae	*Holocentropus stagnalis*			X						X	
Diptera Ceratopogonidae	*Ceratopogon* sp.					X					
	Culicoides sp.										X
Diptera Chaoboridae	*Chaoborus crystallinus*			X							
	Chaoborus flavicans	X	X	X	X		X	X	X	X	X
	Chaoboridae indet. (U)	X	X	X	X					X	X
Diptera Chironomidae	*Chironomus* sp. (L)	X	X	X	X	X	X			X	X
Chironominae	*Endochironomus* sp. (L)	X		X					X	X	X
	Micropsectra sp. (L)							X			
	Parachironomus gr. *arcuatus*	X				X			X		
	Paratanytarsus sp. (L)	X	X	X					X	X	X
	Polypedilum sp. (L)									X	X

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Zavreliella sp. (L)			X		X				X	X
	Chironominae indet. (U)	X		X				X	X	X	X
Diptera Chironomidae Orthocladinae	*Acricotopus* sp. (L)			X							
	Corynoneura sp. (L)	X	X	X	X	X	X	X	X	X	X
	Cricotopus sp. (L)	X		X	X	X	X	X	X		
	Limnophyes sp. (L)		X		X						X
	Metriocnemus sp. (L)						X				
	Psectrocladius (A.) sp. (L)		X								
	Psectrocladius (P.) limbatellus	X		X					X	X	
	Psectrocladius (P.) sordidellus			X							
	Psectrocladius (P.) sp. (L)	X	X	X		X	X	X	X	X	X
	Orthocladinae indet. (U)			X				X	X		X
Diptera Chironomidae Tanypodinae	*Ablabesmyia* sp. (L)									X	X
	Procladius sp. (L)	X	X	X	X	X	X	X	X	X	X
	Tanypus sp. (L)				X		X				
	Xenopelopia sp. (L)	X	X	X			X		X		X
	Zavrelimyia sp. (L)					X				X	
	Tanypodinae indet. (U)			X		X		X		X	
Diptera Culicidae	*Anopheles maculipennis*				X				X		X
	Anopheles sp. (U)							X			
	Culex hortensis				X						
	Culex modestus						X				
	Culex pipiens				X						X
	Culex sp. (U)							X			
	Culex theileri				X		X				
	Culicidae inet.	X									
Diptera Ephydridae	*Ephydridae indet. (U)*						X	X			X
	Hydrellia sp.			X				X			X
Diptera Sciomyzidae	*Sciomyzidae indet. (U)*						X		X		X

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Diptera Stratiomyidae	*Oplodontha* sp.									X	
Diptera Tabanidae	*Tabanus* sp.										X
Amphibia Anura	*Hyla meridionalis*	X	X	X	X	X	X		X	X	X
	Pelobates cultripes	X		X	X	X	X	X		X	X
	Pelophylax perezi							X			X
	Anura indet. (U)			X						X	
Amphibia Caudata	*Triturus marmoratus*		X	X	X	X	X		X	X	X
	Urodela indet. (U)	X		X							

Supplementary material S8.6: The Albera ponds species checklist of the July 2013 survey campaign. CANB -Estany de Baix de Canadal-, CANG -Estany Gran de Canadal-, CANP -Estany Petit de Canadal-, GUTC -Estany de la Cardonera de la Gutina-, GUTR -Estany de la Rajoleria de la Gutina-, MOLES -Estany de les Moles-, SERR -Estany de Serrallobera-, TORC -Estany de la Cardonera dels Torlits-, TORG -Estany Gran dels Torlits-, TORP -Estany Petit dels Torlits. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Turbellaria Rhabdocoela	Rhabdocoela indet. (U) 1			X	-	-	-	-			
Cnidaria	*Hydra* sp.	X			-	-	-	-			
Annelida Hirudinea	*Erpobdella* sp.			X	-	-	-	-			
	Glossiphonia sp.			X	-	-	-	-			
Annelida Oligochaeta	*Chaetogaster*	X		X	-	-	-	-			
	Dero sp.				-	-	-	-			X
	Dina sp.		X	X	-	-	-	-			
	Lumbriculidae indet. (U)			X	-	-	-	-			
	Naidinae indet. (U)		X	X	-	-	-	-		X	
	Stylaria lacustris				-	-	-	-	X		
	Tubificinae indet. (U) without		X		-	-	-	-			
	Tubificinae indet. (U) with setae	X	X		-	-	-	-	X		
Gastropoda Lymnaeidae	*Galba truncatula*				-	-	-	-	X	X	
	Stagnicola palustris				-	-	-	-	X		
Gastropoda Physidae	*Physa acuta*	X	X	X	-	-	-	-	X	X	X
Gastropoda Planorbidae	*Ferrissia* sp.		X		-	-	-	-		X	X
	Gyraulus crista	X	X	X	-	-	-	-	X		
	Gyraulus sp.		X	X	-	-	-	-			X
	Hippeutis complanatus		X	X	-	-	-	-			X
	Physa acuta				-	-	-	-		X	

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Planorbidae indet. (U)			X	-	-	-	-			
Isopoda Asellidae	*Proasellus coxalis*		X		-	-	-	-	X	X	X
Hydrachnidia	*Eylais extendens*				-	-	-	-		X	
	Hydrachna skorikowi	X			-	-	-	-		X	
	Hydrodroma pilosa				-	-	-	-		X	
	Piona sp.1			X	-	-	-	-	X		
	Piona sp.2	X		X	-	-	-	-		X	X
	Arrenuridae indet. (U)	X	X		-	-	-	-			
	Pionidae indet. (U)	X			-	-	-	-	X		X
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*	X	X	X	-	-	-	-	X	X	X
	Cloeon schoenemundi				-	-	-	-	X		
	Baetidae indet. (U)	X	X	X	-	-	-	-	X	X	X
Odonata Aeshnidae	*Anax imperator*			X	-	-	-	-		X	
	Anax sp. (U)		X		-	-	-	-	X		X
	Aeshnidae indet. (U)	X	X	X	-	-	-	-	X	X	X
Odonata Coenagrionidae	*Ischnura elegans*				-	-	-	-			X
	Ischnura sp. (U)	X			-	-	-	-	X	X	
	Coenagrionidae indet. (U)		X	X	-	-	-	-	X	X	X
Odonata Lestidae	*Lestes* sp. (U)			X	-	-	-	-			
	Sympecma fusca	X	X		-	-	-	-	X		X
Odonata Libellulidae	*Sympetrum fonscolombii*			X	-	-	-	-	X	X	
	Sympetrum meridionale				-	-	-	-			X
	Sympetrum sp. (U)				-	-	-	-		X	
	Libellulidae indet. (U)	X		X	-	-	-	-			

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
Hemiptera Corixidae	*Corixa affinis*			X	-	-	-	-			
	Corixa panzeri	X	X		-	-	-	-	X	X	X
	Corixa punctata		X	X	-	-	-	-			
	Hesperocorixa linnaei			X	-	-	-	-	X		X
	Hesperocorixa moesta			X	-	-	-	-	X	X	X
	Micronecta scholzi	X			-	-	-	-			
	Sigara dorsalis	X			-	-	-	-	X		
	Sigara lateralis	X			-	-	-	-			
	Sigara limitata	X			-	-	-	-			
	Corixidae indet. (U)	X		X	-	-	-	-	X		
Hemiptera Gerridae	*Gerris argentatus*	X	X		-	-	-	-	X	X	X
	Gerris thoracicus				-	-	-	-		X	X
	Gerridae indet. (U)	X	X	X	-	-	-	-	X	X	X
Hemiptera Mesoveliidae	*Mesovelia vittigera*				-	-	-	-		X	
Hemiptera Naucoridae	*Ilyocoris cimicoides*		X		-	-	-	-			
	Naucoris maculatus				-	-	-	-		X	
	Naucoridae indet. (U)		X	X	-	-	-	-			
Hemiptera Nepidae	*Ranatra linearis*		X		-	-	-	-		X	X
Hemiptera Notonectidae	*Anisops sardeus*	X			-	-	-	-			
	Anisops sp. (U)	X			-	-	-	-			X
	Notonecta viridis				-	-	-	-			X
Hemiptera Pleidae	*Plea minutissima*	X	X	X	-	-	-	-	X	X	X
Hemiptera Saldidae	*Saldula opacula*			X	-	-	-	-	X		X
	Saldidae indet. (U)			X	-	-	-	-			
Coleoptera Curculionidae	*Bagous* sp. (U)		X		-	-	-	-	X		X
Coleoptera Dytiscidae	*Bidessus goudoti*	X	X	X	-	-	-	-			X
	Bidessus sp. (L)			X	-	-	-	-			
	Cybister sp. (L)	X	X	X	-	-	-	-		X	X

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Graphoderus sp. (L)	X	X	X	-	-	-	-			X
	Graptodytes bilineatus			X	-	-	-	-			
	Graptodytes flavipes	X		X	-	-	-	-	X	X	
	Hydroporinae indet. (U)				-	-	-	-			X
	Hydrovatus cuspidatus		X		-	-	-	-			
	Hydrovatus sp. (L)		X		-	-	-	-			
	Hyphydrus aubei	X			-	-	-	-	X	X	X
	Hyphydrus sp. (L)	X	X	X	-	-	-	-	X	X	X
	Laccophilus minutus			X	-	-	-	-	X	X	X
	Laccophilus sp. (L)	X	X	X	-	-	-	-	X	X	X
	Laccophilus variegatus		X		-	-	-	-		X	
Coleoptera Elmidae	*Oulimnius rivularis*		X	X	-	-	-	-			
Coleoptera Gyrinidae	*Gyrinus dejeani*				-	-	-	-			X
	Gyrinus distinctus		X		-	-	-	-			X
Coleoptera Hydrochidae	*Hydrochus angustatus*	X			-	-	-	-			
Coleoptera Hydrophilidae	*Anacaena lutescens*				-	-	-	-	X		X
	Berosus affinis				-	-	-	-	X		
	Berosus signaticollis			X	-	-	-	-			
	Berosus sp. (L)	X	X	X	-	-	-	-			
	Enochrus nigritus	X		X	-	-	-	-	X		X
	Enochrus quadripunctatus	X			-	-	-	-			
	Enochrus sp. (L)	X	X	X	-	-	-	-		X	X
	Helochares sp. (L)	X	X		-	-	-	-		X	X
	Hydrobius sp. (L)	X	X		-	-	-	-		X	X
	Hydrophilus sp. (L)			X	-	-	-	-			
	Limnoxenus niger			X	-	-	-	-			X
	Limnoxenus sp. (L)	X		X	-	-	-	-		X	X
Coleoptera Hygrobiidae	*Hygrobia hermanii*				-	-	-	-		X	
Coleoptera Noteridae	*Noterus clavicornis*			X	-	-	-	-		X	X

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Noterus laevis		X	X	-	-	-	-		X	X
	Noterus sp. (L)		X		-	-	-	-			
Trichoptera Polycentropodidae	*Holocentropus stagnalis*			X	-	-	-	-	X		X
Diptera Ceratopogonidae	*Alluaudomyia* sp.				-	-	-	-			X
	Culicoides sp.	X			-	-	-	-			
	Palpomyna sp.			X	-	-	-	-			
Diptera Chaoboridae	*Chaoborus crystallinus*		X	X	-	-	-	-	X		
	Chaoborus flavicans	X	X	X	-	-	-	-	X	X	X
	Chaoborus sp. (U)	X	X	X	-	-	-	-		X	X
	Chaoboridae indet. (U)				-	-	-	-	X		
Diptera Chironomidae Chironominae	*Chironomus* sp. (L)	X	X	X	-	-	-	-	X	X	X
	Endochironomus sp. (L)	X	X	X	-	-	-	-	X	X	X
	Kiefferulus tendipediformis				-	-	-	-		X	
	Micropsectra sp. (L)	X			-	-	-	-			
	Parachironomus gr. *arcuatus*	X	X	X	-	-	-	-			
	Paratanytarsus sp. (L)	X		X	-	-	-	-	X		
	Polypedilum sp. (L)	X			-	-	-	-	X	X	X
	Zavreliella sp. (L)			X	-	-	-	-	X	X	
	Chironominae indet. (U)	X	X	X	-	-	-	-	X	X	X
Diptera Chironomidae Orthocladinae	*Cricotopus* sp. (L)			X	-	-	-	-	X		
	Orthocladinae indet. (U)			X	-	-	-	-	X		
	Psectrocladius (A.) sp. (L)			X	-	-	-	-			
	Psectrocladius (P.) sp. (L)	X		X	-	-	-	-	X		
Diptera Chironomidae Tanypodinae	*Ablabesmyia* sp. (L)	X		X	-	-	-	-	X		
	Alotanypus sp. (L)	X			-	-	-	-			
	Procladius sp. (L)	X	X	X	-	-	-	-	X	X	X
	Tanypus sp. (L)		X		-	-	-	-	X	X	

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Taxon		CANB	CANG	CANP	GUTC	GUTR	MOLES	SERR	TORC	TORG	TORP
	Xenopelopia sp. (L)		X	X	-	-	-	-			
	Zavrelimyia sp. (L)				-	-	-	-	X		
	Tanypodinae indet. (U)	X	X	X	-	-	-	-	X	X	
Diptera Culicidae	*Anopheles maculipennis*	X		X	-	-	-	-	X		X
	Anopheles sp. (U)			X	-	-	-	-			
	Culex impudicus		X	X	-	-	-	-			
	Culex modestus			X	-	-	-	-	X		
	Culex sp. (U)				-	-	-	-	X		
	Culex theileri			X	-	-	-	-	X	X	X
Diptera Dixidae	*Dixella autumnalis*				-	-	-	-	X		
Diptera Ephydridae	Ephydridae indet. (U)				-	-	-	-	X		
	Hydrellia sp.			X	-	-	-	-			
Diptera Sciomyzidae	Sciomyzidae indet. (U)		X	X	-	-	-	-	X		
Diptera Stratiomyidae	*Odontomyia* sp.	X	X		-	-	-	-			X
Diptera Tabanidae	*Hybomitra* sp.				-	-	-	-	X		
Amphibia Anura	*Hyla meridionalis*		X		-	-	-	-		X	
	Pelobates cultripes				-	-	-	-			X
	Pelophylax perezi			X	-	-	-	-	X		X
	Anura indet. (U)			X	-	-	-	-		X	

Supplementary material S8.7: Guils de Cerdanya ponds species checklist of the May 2016 survey campaign. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

May 2016
— hydroperiod beginning —

Taxon		CENTRAL	ISOLATED
Turbellaria Rhabdocoela	*Gieysztoria diadema*	X	
	Gieyztoria sp.1	X	X
	Rhabdocoela indet. (U) 1	X	
	Rhabdocoela indet. (U) 11	X	X
	Dalyellidae indet. (U)	X	
Annelida Oligochaeta	Enchytraeidae indet. (U)	X	X
	Lumbricidae indet. (U)	X	
	Lumbriculidae indet. (U)	*X*	X
	Naidinae indet. (U)	*X*	X
	Tubificinae indet. (U) with setae	X	X
Bivalvia sphaeriidae	*Pissidium casertanum*	X	
	Pissidium indet. (U)	X	
Gastropoda Lymnaeidae	*Galba truncatula*		X
Branchiopoda Anostraca	*Chirocephalus diaphanus*	X	X
	Anostraca indet. (U)	X	X
Hydrachnidia	*Hydrachna skorikowi*		X
	Piona sp.1		X
	Piona sp.2	X	X
Odonata	Anisoptera indet. (U)		X
	Lestes dryas	X	X
	Zygoptera indet. (U)	X	X
Heteroptera Gerridae	*Gerris costai*	X	X
Coleoptera Dryopidae	*Dryops auriculatus*		X
	Dryops sp. (L)	X	X
Coleoptera Dytiscidae	*Agabus* sp. (L)	X	X
	Hygrotus marklini		X
Coleoptera Helophoridae	*Helophorus discrepans*	X	X
	Helophorus flavipes	X	
Coleoptera Hydrophilidae	*Anacaena limbata*	X	
	Berosus luridus	X	X

May 2016

— hydroperiod beginning —

Taxon		CENTRAL	ISOLATED
Trichoptera Limnephilidae	*Limnephilus bipunctatus*	X	X
	Limnephilus sp. (U)		X
	Limnephilus stigma	X	X
	Limnephilidae indet. (U)	X	
Diptera Chironomidae Chironominae	*Chironomus* sp. (L)	X	X
	Polypedilum sp. (L)	X	
Diptera Chironomidae Orthocladinae	*Psectrocladius (P.)* sp. (L)	X	
	Pseudosmittia sp. (L)	X	
	Smittia sp. (L)	X	
	Thienemannia sp. (L)		X
Diptera Culicidae	*Aedes (Ochlerotatus) communis*	X	
	Aedes (Ochlerotatus) pullatus	X	
	Aedes (Ochlerotatus) sp. (U)	X	X
	Aedes (Ochlerotatus) surcoufi	X	X
	Culicidae indet. (U)	X	X
Diptera Scathophagidae	Scathophagidae indet. (U)		X
Amphibia Anura	*Rana temporania*	X	X
	Anura indet. (U)	X	

Supplementary material S8.8: Guils de Cerdanya ponds species checklist of the June 2016 survey campaign. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

June 2016
— hydroperiod middle —

Taxon		CENTRAL	ISOLATED
Turbellaria Rhabdocoela	Gieyztoria sp.*1*	X	X
	Gyratrix hermaphroditus		X
	Dalyellidae indet. (U)	X	X
	Rhabdocoela indet. (U) 11	X	X
Annelida Oligochaeta	Enchytraeidae indet. (U)	X	X
	Lumbriculidae indet. (U)	X	
	Naidinae indet. (U)	X	X
	Tubificinae indet. (U) with setae	X	X
Bivalvia sphaeriidae	*Pissidium casertanum*	X	X
Gastropoda Lymnaeidae	*Galba truncatula*		X
Branchiopoda Anostraca	*Chirocephalus diaphanus*	X	X
Hydrachnidia	*Hydrachna skorikowi*	X	X
	Piona sp.1	X	
	Piona sp.2	X	
	Hydrachnidia indet. (U)	X	
Odonata Lestidae	*Lestes dryas*	X	X
Odonata Libellulidae	*Sympetrum* sp. (U)		X
Hemiptera Gerridae	*Gerris costai*	X	X
	Gerridae indet. (U)	X	
Coleoptera Dryopidae	*Dryops auriculatus*		X
Coleoptera Dytiscidae	*Agabus* sp. (L)	X	
	Dytiscus sp. (L)	X	X
	Hydroporus cf. *nigrita*	X	
Coleoptera Helophoridae	*Helophorus discrepans*		X
	Helophorus flavipes	X	
	Helophorus gr. *maritimus*	X	
Coleoptera Hydrophilidae	*Anacaena limbata*	X	
	Berosus luridus	X	X
Trichoptera Limnephilidae	*Limnephilus bipunctatus*	X	X
	Limnephilus stigma	X	X

June 2016

— hydroperiod middle —

Taxon		CENTRAL	ISOLATED
	Limnephilus vittatus	X	X
	Limnephilus sp. (U)	X	X
Diptera Chironomidae Chironominae	*Chironomus* sp. (L)	X	X
	Polypedilum sp. (L)	X	
	Tanytarsus sp. (L)		X
Diptera Chironomidae Orthocladinae	*Cricotopus* sp. (L)		X
	Limnophyes sp. (L)		X
	Psectrocladius (P.) sp. (L)	X	X
	Pseudosmittia sp. (L)	X	
	Smittia sp. (L)	X	
	Thienemannia sp. (L)	X	
Diptera Culicidae	*Aedes (Ochlerotatus) surcoufi*	X	

Supplementary material S8.9: Guils de Cerdanya ponds species checklist of the July 2016 survey campaign. **L** stands for individuals of Coleoptera and Diptera only found at larval stage that could not be further classified. Coleoptera without L indicate adult individuals found while Diptera indicate individuals found in pupa stage. **U** stands for undetermined individuals that could not be further classified.

July 2016
— final hydroperiod —

Taxon		CENTRAL	ISOLATED
Turbellaria Rhabdocoela	*Phaenocora* sp.		X
	Rhabdocoela indet. (U) 1	X	
	Rhabdocoela indet. (U) 11	X	
	Dalyellidae indet. (U)	X	
Annelida Oligochaeta	Enchytraeidae indet. (U)	X	X
	Lumbriculidae indet. (U)	X	
	Naidinae indet. (U)	X	X
	Tubificinae indet. (U) with setae	X	X
	Tubificinae indet. (U) without setae		X
Bivalvia sphaeriidae	*Pissidium casertanum*	X	X
Gastropoda Lymnaeidae	*Galba truncatula*		X
Ephemeroptera Baetidae	*Cloeon* gr. *dipterum*		X
	Baetidae indet. (U)	X	X
Odonata Aeshnidae	Aeshnidae indet. (U)		X
Odonata Lestidae	*Lestes dryas*	X	X
	Zygoptera indet. (U)		X
Odonata Libellulidae	*Sympetrum flaveolum*	X	X
Hemiptera Corixidae	*Sigara nigrolineata*	X	
Hemiptera Gerridae	*Gerris costai*	X	
Hemiptera Notonectidae	Notonectidae indet. (U)	X	
Hemiptera Pleidae	*Plea minutissima*		X
Coleoptera Dryopidae	*Dryops auriculatus*		X
Coleoptera Dytiscidae	*Agabus* cf. *lapponicus*	X	
	Deronectes aubei sanfilipoi		X
	Deronectes sp. (L)	X	X
	Dytiscus sp. (L)	X	X
	Hydroporus cf. *vagepictus*	X	
	Hygrotus marklini	X	X
	Hygrotus sp. (L)		X
	Ilybius albarracinensis		X
	Laccophilus hyalinus		X
Coleoptera Helophoridae	*Helophorus discrepans*		X
	Helophorus flavipes	X	X

July 2016
— final hydroperiod —

Taxon		CENTRAL	ISOLATED
	Helophorus gr. *maritimus*	X	
Coleoptera Hydrophilidae	*Anacaena limbata*	X	
	Berosus luridus	X	X
	Enochrus fuscipenis		X
Trichoptera Limnephilidae	*Limnephilus stigma*	X	
	Limnephilus vittatus	X	X
Diptera Ceratopogonidae	*Culicoides* sp.	X	X
Diptera Chironomidae Chironominae	*Chironomus* sp. (L)	X	X
	Cricotopus sp. (L)		X
	Polypedilum sp. (L)	X	
Diptera Chironomidae Orthocladinae	Orthocladinae indet. (U)		X
	Psectrocladius (P.) sp. (L)		X
Diptera Chironomidae Tanypodinae	*Ablabesmyia monilis*	X	
	Ablabesmyia sp. (L)	X	X
	Procladius sp. (L)	X	X
	Tanypodinae indet. (U)	X	X
	Zavrelimyia sp. (L)	X	
Diptera Ephydridae	*Hydrellia* sp. (L)	X	
Amphibia Anura	*Rana temporania*	X	X